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Department of Environment
Water Resources Division
St. John's, Newfoundland

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Inland Waters Directorate
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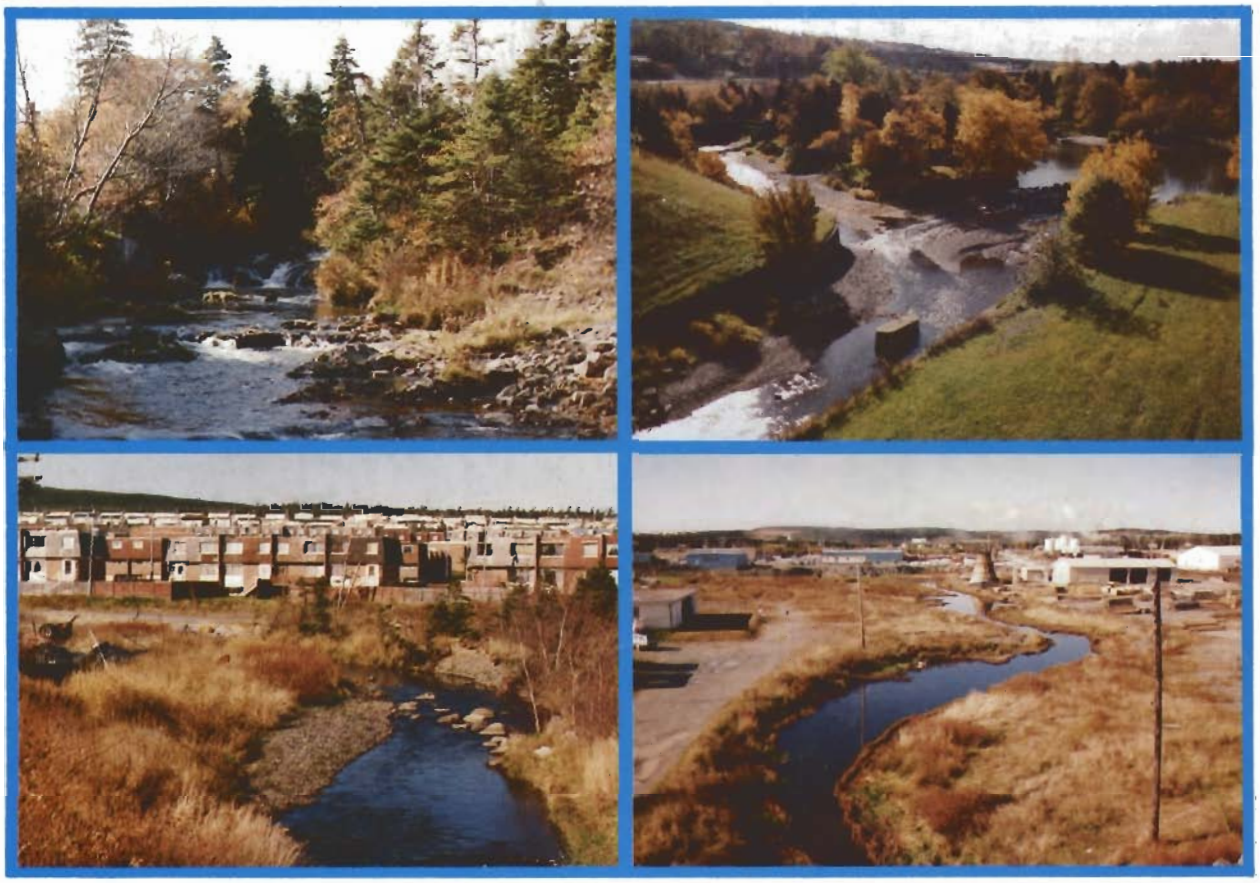
National Water Research Institute
Burlington, Ontario

BIOLOGICAL STUDY



Urban Hydrology Study of the Waterford River Basin

TECHNICAL REPORT No.

UHS-WRB 1.8

Biological Indicators of Water Quality in the Waterford River System

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St. John's, Newfoundland
November, 1986.

GOVERNMENT OF NEWFOUNDLAND AND LABRADOR
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November 27, 1986

Dr. Wasi Ullah, Chairman
Technical Committee
Waterford River Basin Urban
Hydrology Study
Newfoundland Department of Environment
Confederation Building West, 4th Floor
St. John's, Newfoundland.
A1C 5T7

Dear Dr. Ullah:

Please find attached a copy of the biological study final report entitled, "Biological Indicators of Water Quality in the Waterford River System", as partial fulfillment to the mandate of the Waterford River Basin Urban Hydrology Study.

Yours truly,

Ken Rollings

Ken Rollings,
Project Engineer

for: Working Group on Biological Study
Ms. S. Bonneyman
Mr. W. Pierce
Mr. D. Hansen
Mr. K. Rollings

ABSTRACT

The benthic invertebrates of the Waterford River System were sampled, using artificial substrates, in December 1984 and 1985, with the objective of identifying indicator communities for water quality, and to further assess the deterioration of water quality in that system. A diverse insect community, with a small number of individuals per species was found to be indicative of "clean water", while sites with poor water quality were observed to support communities dominated by a few species with large numbers of individuals. South Brook, the main tributary of the Waterford River was found to contain relatively clean water, while that of the Waterford River itself was only fair to poor. An unnamed tributary, running through the Canada Department of Agriculture experimental farm was found to have fair water quality upstream of the experimental farm and extremely poor water quality downstream.

RESUME

Les invertébrés benthiques du système de la rivière Waterford furent échantillonnés en décembre 1984 et 1985 à l'aide de substrats artificiels. L'objectif était d'identifier les communautés indicatrices de la qualité de l'eau afin de pouvoir évaluer la détérioration de la qualité de l'eau dans ce sytème. Une communauté d'insectes diversifiée, avec peu d'individus par espèce est indicatrice d'eaux propres, alors que des sites dont la qualité de l'eau était pauvre supportent une communauté dominée par peu d'espèces avec un grand nombre d'individus. Les eaux de South Brook, tributaire principal de la rivière Waterford, sont relativement propres alors que la qualité des eaux de la rivière Waterford est moyenne ou pauvre. La qualité des eaux d'un tributaire innommé traversant la ferme expérimentale du Ministère de l'Agriculture du Canada fut trouvée moyenne en amont de la ferme expérimentale et extrêmement pauvre en aval.

PREFACE

The Waterford River Basin Urban Hydrology Study, developed as a co-operative effort between the Governments of Canada and the Province of Newfoundland, was proposed by the Newfoundland Department of Environment in response to watershed management problems that had resulted from urbanization of the Waterford River Basin. Among such problems, negative effects of urbanization on both water quality and quantity were found to be so serious that the Newfoundland Department of the Environment identified the Waterford River Basin as a high priority area.

The five year study, begun in 1980, was completed in March, 1985. The primary objectives of the study were to develop environmentally acceptable criteria for urban development in Newfoundland and to utilize the study results directly in the urban planning process in the Province. The specific objectives of the study, as outlined in the report "Waterford River Basin - Urban Hydrology Study Plan" were as follows:

1. To examine the processes leading to changes in the hydrologic regime of the Waterford River watershed. This should include evaluation and monitoring of major hydrologic changes caused by urbanization, the study of precipitation- runoff processes, and the study of various forms of pollution originating in the urban areas of the watershed.

2. To provide a hierarchy of mathematical models describing hydrologic processes in the watershed. Such models should deal with both water quantity and quality, and should be capable of simulating the impact of urbanization on the water resources in the studied basin.

3. To recommend solutions to specific water management problems in the studied basin and to develop guidelines forimplementation of similar solutions elsewhere in Newfoundland. Furthermore, planning and management criteria should be developed for those aspects of the urban development which are related to the environmental protection of the affected water resources.

The complexity of the study called for a comprehensive approach which included hydrometric surveys, hydrological modelling, groundwater studies, biological surveys, water quality assessment, investigations of flooding and land use, and socio-economic analyses.

The study was administered by a Steering Committee appointed by the governments of Newfoundland and Canada. To implement the study plan, a Technical Committee consisting of two representatives, of each government, was established. Subsequently the Technical Committee appointed sub-committees and working groups to prepare and carry out the work plans for the various components of the study.

The report that follows deals with one such component - Biological Indicators of Water Quality.

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- Mr. Bruce Thompson, Environmental Assessment Division, Newfoundland Department of Environment.
- Dr. U.S. Panu, Water Resources Division, Newfoundland Department of Environment.
- Dr. W. Ullah, Water Resources Division, Newfoundland Department of Environment.
- Mr. T.W. Hennigar, Inland Waters Directorate, Environment Canada.
- Mr. J. Marsalek, National Water Research Institute, Canada Centre for Inland Waters.

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1 INTRODUCTION

The Newfoundland Department of Environment (NDOE) in 1978 designed the Waterford River Basin Urban Hydrology Study plan to study the effects of urbanization and development on the Waterford River system. The original objective of the biological component of the study was to examine the population characteristics of the stream bottom-dwelling animals and plants in order to determine the degree of pollution at any particular location in the Waterford River basin. In order to obtain pre-pollution base-line data on the composition of stream communities, a comparable natural stream would also be selected and surveyed. It was proposed to select five sampling stations in the Waterford River basin and sample each station seven times a year (May, June, July, September, and November). In order to obtain statistically acceptable data, it was proposed to obtain three samples simultaneously from each station. Two sampling stations were to be selected on a pristine stream close to Waterford River. The proposed survey would be done three times during the period of investigation. Data analyses and interpretation were to include designation of indicator species, use of biotic index to evaluate the composition of the stream bottom communities and evaluation of the degree of pollution in the Waterford River system. The terms of reference for this study are given in Appendix C.

The biological study was initiated in May, 1981 with a call for proposals. Five consulting firms, Beak Consultants Limited, Gavalin MacLaren Plansearch Limited, LGL Limited, Shawmont Newfoundland Limited, and Wildland Associates Limited submitted proposals according to the Terms of Reference (Appendix C) provided by the NDOE. A brief review of each proposal is presented in Appendix D. None of these proposals were accepted, as the costs of all exceeded the approved funding for the study.

In August, 1982, the Technical Committee decided that the biological component of the study be carried out on a reduced scale, and still be within budget. This was done, by using the services of in-house staff of the Newfoundland Department of Environment. The Steering Committee approved this proposal. A two-year study, involving only collecting and analyzing benthic invertebrate data was designed, to achieve some of the objectives of the original study concept. This study concept was discussed, and

subsequently approved by the Technical Committee, in May, 1983. The revised biological study program, as proposed by S. Bonnyman, is outlined in Appendix E.

An agreement was reached between the Newfoundland Department of Environment (NDOE) and Dr. D. Larson of the Department of Biology at Memorial University of Newfoundland (M.U.N.), under which the biological study would be completed. In each year of the study programme, one honours student in the Dept. of Biology would be employed by NDOE and supervised by Dr. Larson. The students hired to conduct the study were Mr. Tony Clemens in 1984 and Mr. Bill Stirling in 1985.

The two-year study was initiated in January, 1984, with the selection of the sampling stations, and placement of the artificial substrate bags in the streams. The first phase of the study was completed, with the interim report published in December, 1985. The second phase of the study was initiated in the fall of 1985. This is the first study of benthic invertebrate community structure conducted on the Waterford River system.

Studies done on other river systems in North America indicate a relationship between benthic invertebrate community structure and water quality (Tackett, 1964; Tackett, 1965; Gaufin, 1973; Whiting and Clifford, 1983). Generally, there is a direct correlation between poor water quality and reduced invertebrate diversity and/or abundance. These studies are reviewed in Appendix B.

1.1 Justification for study

The Waterford River system is an integral part of the lifestyle of area residents. For generations the river had been used as a source of fresh water, and for recreational purposes, such as swimming and trout fishing. In recent years, however, the continued development (residential and industrial) has resulted in serious deterioration of the water quality of this system. Previous studies on the Waterford River system, reviewed in Appendix A, have shown the need for an in-depth study to assess the effects of urbanization on the ecology of this system. These studies have all shown that the Waterford River and its tributaries are suffering major abuses, and as a result, have very poor water quality. It was decided to conduct this study, to determine the effects of

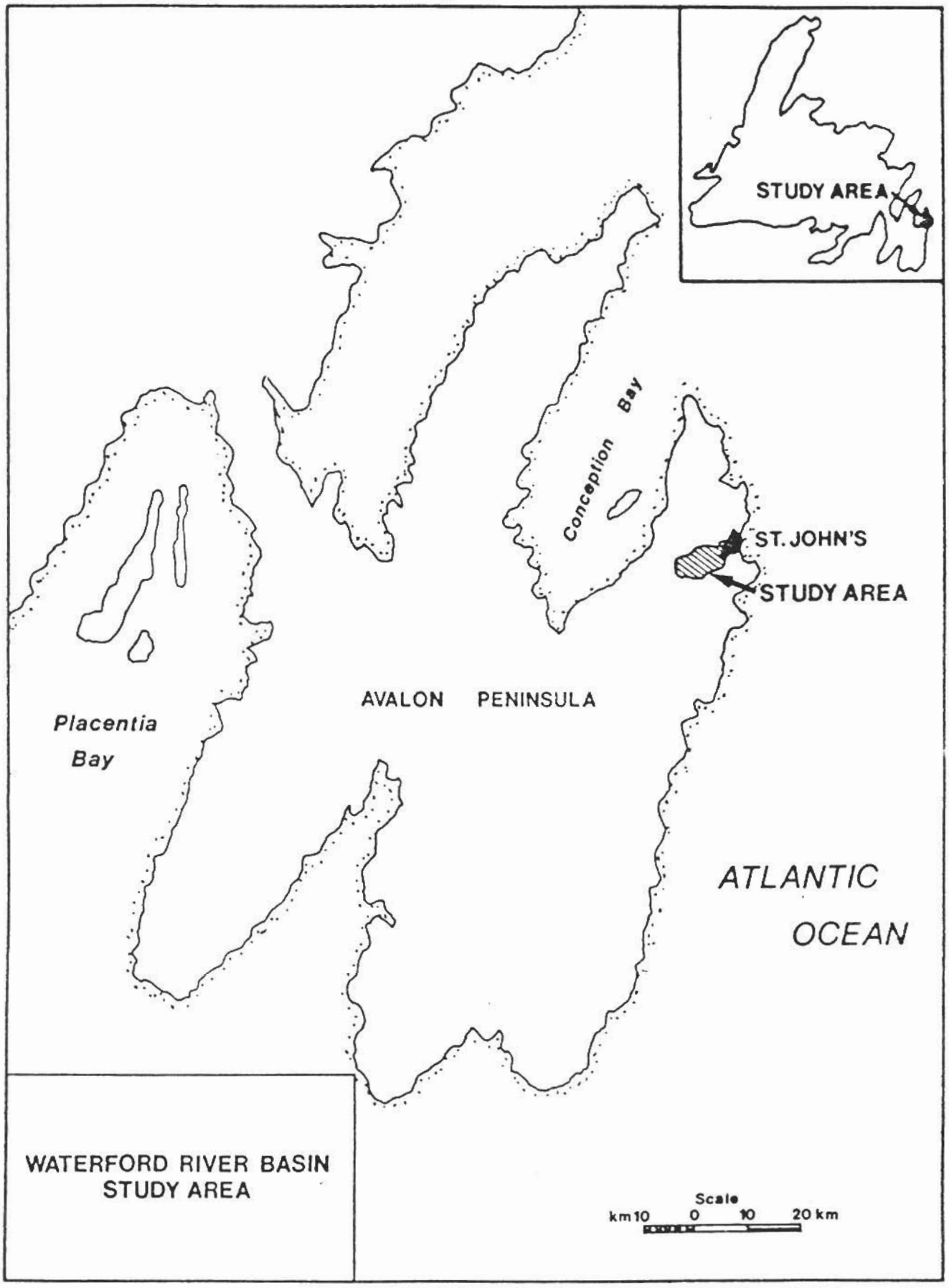


Figure 1. The location of the Waterford River Basin

urban development, and its associated stresses and problems, on the biological environment of the Waterford River system.

This study will provide greater resolution in terms of determining the extent to which the biological environment of the Waterford River system has been altered and affected by urban development. It will identify the major problem areas, and will provide baseline data to allow monitoring of environmental changes over time.

1.2 Objectives

Based on the above premises, a two-year study was conducted with the following objectives;

1. Examination of the benthic invertebrate community structure, to determine the effects of deteriorated water quality due to urbanization, on the biological activity in the Waterford River system.

2. Establish a dependable method for assessing the water quality of a river system.

3. Identify communities of invertebrates which are characteristic of "clean" and "unclean" water.

4. To identify problem areas, in terms of water quality and pollution, in the Waterford River basin.

5. To establish baseline data, which can be used to monitor environmental changes in the future.

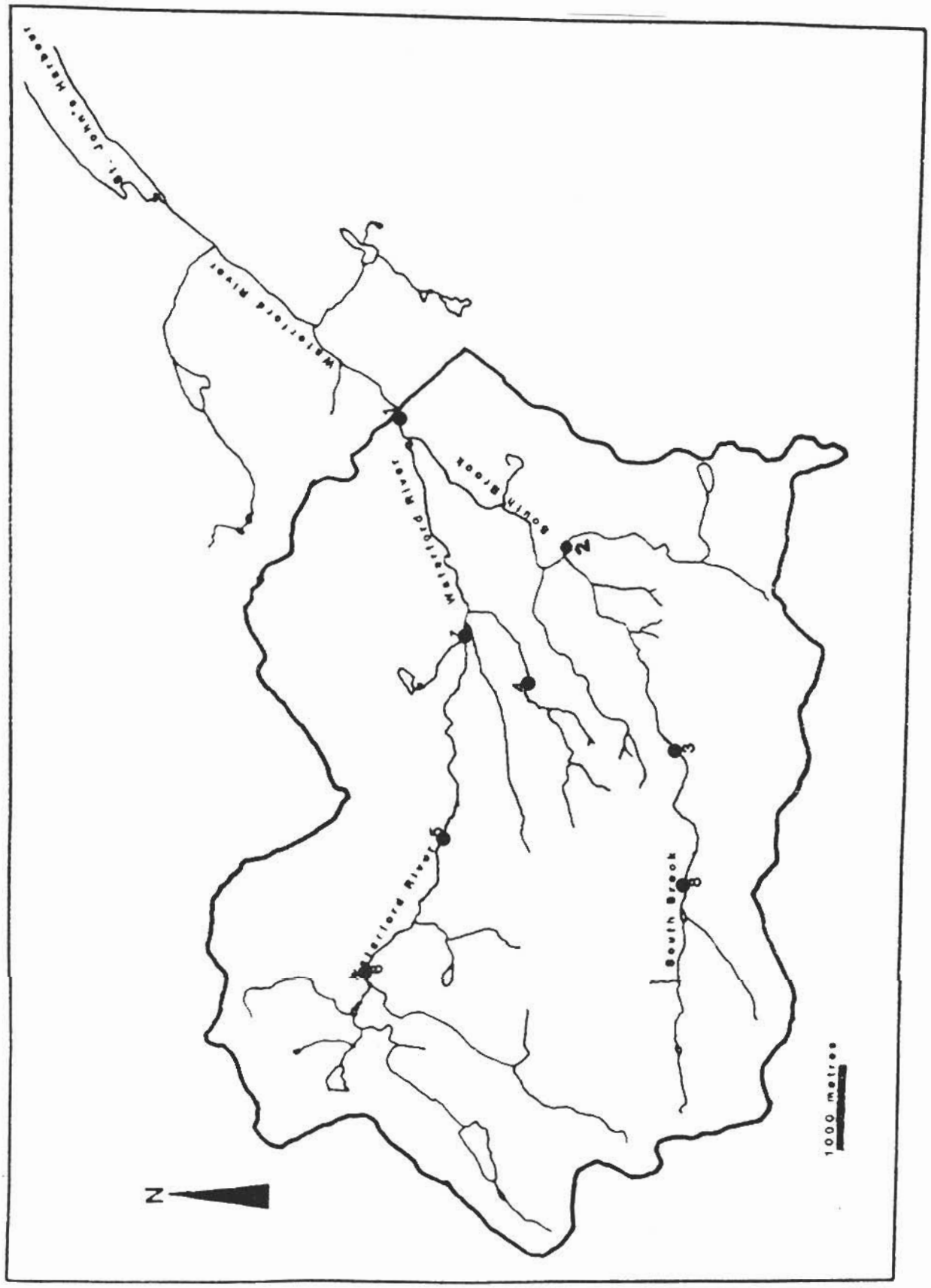


Figure 2. Locations of the water quality monitoring sites in the Waterford River Basin

2 LITERATURE REVIEW

This chapter contains a review of biological literature relevant to this study.

2.1 Natural variations in aquatic invertebrate populations.

Aquatic invertebrates have seasonal life cycles, with temporal variation exhibited among species. As a result the benthic fauna in a stream varies continuously throughout the year.

2.1.1 Seasonal effects

Seasonal differences in species richness have been observed and related to sediment changes with accompanying variations in flow rates which produce well-sorted, distinct sediment types. Species richness is highly correlated with mean particle size. As the interstices of the substrates were filled in with progressively finer particles during the summer, habitat heterogeneity is gradually reduced and mean particle size is less efficient in predicting the number of species (de March, 1976).

The changing supply of suitable food causes shifts in the invertebrate community structure. The invertebrates move from leaf packs to the underlying substrate as the leaves decompose in spring and summer (Reice, 1980).

Photoperiodic control of aquatic insects life histories has been reported. It is fairly important to insects due to its predictability and its reliable signalling of forthcoming seasonal changes, especially in temperate regions. Often life-histories can be disrupted in polluted lotic (i.e., flowing water) habitats due to dissolved humic substances, suspended sediment, ice and snow cover and other factors which significantly reduce the amount of light (Beak, 1980).

Aquatic plants and their distribution have also been shown to be important in invertebrate microhabitat selection. Benthos community structure has been shown to be related to spatial and seasonal abundance patterns of aquatic macrophytes (Gregg and Rose, 1985).

Seasonal fluctuations in temperature are important in regulating insect life histories.

Seasonal temperature fluctuations and growth seem to be linked, possibly because different physiological processes have different thermal optima. Seasonal temperature flucuations also may be effective in reducing competitive interactions between species, and hence promote coexistence (Ward and Stanford, 1979).

Biotic interactions will also change seasonally as populations of potential predators and competitors wax and wane.

2.1.2 Flow Variability and its Effects

Short-term flow fluctuations may modify aquatic invertebrate communities in several ways. Aquatic invertebrates may be stranded in pockets of standing water or along exposed shorelines due to fluctuating water levels. Differential tolerance to stranding eliminates certain groups while allowing others to predominate. Mayfly nymphs are particularly susceptible to stranding and are relatively intolerant of atmospheric exposure, whereas many chironomids tolerate considerable exposure, especially in cool weather. The majority of stream insects can apparently tolerate brief periods of atmospheric exposure, and some species migrate as the water level recedes. This, of course, also varies with the life cycle stage (e.g., larvae are generally more mobile than eggs or pupae). Other factors which influence the effects of stranding on aquatic invertebrates, besides duration of exposure and rate of change in water levels, are time of year (atmospheric conditions), channel configuration, and substrate heterogeneity (Ward, 1984).

Both increased and decreased flow rates will induce drift of aquatic invertebrates. Severe changes in flow rates may alter debris dams and remove accumulations of detritus, thereby modifying spatial and trophic characteristics. Reduction of physical retention structures will alter the ability of the stream ecosystem to withstand and recover from changes in flow rates (Webster *et al.*, 1975). Teague *et al.* (1985) report that water level and flow rate are important in invertebrate microhabitat selection.

2.1.3 Effects of Stream Order on Fauna

The relative dominance (as biomass) of the various trophic groups (shredders, collectors, grazers, and predators) is often influenced by stream order (Merritt *et al.*, 1984). Lower order streams usually have a large proportion of shredders and collectors but few grazers, while higher order streams have decreased shredder populations and increased grazer populations (Merritt *et al.*, 1980). The zone through which a stream shifts from a dominance of shredders to grazers is usually dependent on the degree of shading (Minshall, 1978). In deciduous forests and some coniferous forests, the shift from shredders to grazers is at stream order 3 (Merritt *et al.*, 1984).

2.2 Agricultural effects

About 11.2% of the Waterford River basin is used for agriculture. Almost every agricultural activity in a watershed changes the water chemistry by increasing the nutrient content of the water with regards to nitrates-nitrites and phosphates. The commonly cited effects of agriculture on aquatic environments are high bacterial and viral concentrations, soil erosion with its attendant high turbidity, eutrophication resulting from nutrient runoff, increased oxygen demand, pesticides, field drainage and the channelization of watercourses (Hynes, 1970; Hill, 1976; Dance and Hynes, 1980). Welch *et al.* (1977) observed that streams flowing through farmed basins had 64% less benthos than control streams.

2.2.1 Pesticides

The effect of pesticides on aquatic insects is harmful, especially to younger or smaller specimens which are more susceptible than larger ones (Eidt, 1975). It seems likely that insects are particularly susceptible during physiologically active periods, such as during molting and emergence (Jensen and Gaufin, 1964). Courtemanch and Gibbs (1980) showed that pesticide spraying increased insect drift up to 170 times in treated streams while populations of Plecoptera, Ephemeroptera and Trichoptera showed significant decline following spraying.

2.2.2 Animal waste and feed

The presence of abnormal amounts of animal waste and feed in rivers will increase the level of organic pollution (i.e., nitrates and phosphates) and result in abnormal numbers of bacteria and sewage-related fungi (Hynes, 1960). While some species of invertebrates may benefit from this increased food supply, and increase their numbers, overall species diversity generally declines (Hynes, 1960). Most species represented in unpolluted waters are also seen in rivers which receive mild organic pollution, however due to alteration of the stream characteristics (e.g., type of food available, or sedimentation of the substrate), the balance between species may be altered.

2.3 Industrial activities

Industrial activities account for about 5.5% of the land use in the Waterford River basin. Pollution often involves several potentially interacting agents. For example, industrial wastewater may contain some substances that promote plant growth and others that are toxic to plant or animal life. Industrial activities may increase nutrient levels in freshwater but may decrease habitat heterogeneity. Thus, it is often difficult to resolve and quantify the contribution from each factor or to predict the combined effect of several factors acting together. However, Warren (1971) described the effects of various pollutants or stresses on aquatic environments and concluded that in the area downstream from a point source of pollution there is: 1) a short zone where mixing occurs between clean water and the effluent, followed by 2) a zone of degradation, and after dilution and active decomposition, 3) a zone of cleaner unstressed water at some variable point downstream.

2.3.1 Chemical inputs

Heavy metals, along with pesticides, have been associated with morphological abnormalities in aquatic insects (Simpson, 1980; Petersen and Peterson, 1983; Wiederholm, 1984). Concentrations of heavy metals well below those that are acutely toxic to insects may reduce the incidence of molting and inhibit emergence. Bioaccumulation of heavy metals by many insects may mean that even low concentrations of heavy metals may alter community structure by modifying life span and inhibiting reproduction (Lehmkuhl, 1979). Hall *et al.* (1985) showed that addition of inorganic compounds (e.g., aluminium) can increase drift of benthic invertebrates.

Oil can enter the aquatic environment by accidental spills, chronic additions from industrial wastewater, and runoff from paved surfaces. The effect of oil pollution on benthic organisms is twofold; the toxicity of the lighter fractions, and the mechanistic dysfunctions caused by clogging of body parts and the substrate itself. The consequences of oil pollution have been less studied in freshwater than marine environments, and very little is actually known about its effects on freshwater organisms. Large amounts of oil, whether present as a surface slick shortly after a spill or as a sludge on the stream or lake bottom, will effectively eliminate most insects and other invertebrates. Similarly, low or moderate levels of contamination may result in decreased species richness and abundance of some insects, but these levels may also have the opposite effect. Increased insect abundance probably results from the proliferation of attached algae, which serves as food and substrate for some insects (Rosenberg and Wiens, 1976).

2.3.2 Thermal inputs

Alterations of the thermal regime in a water body occur in many ways. The effects of industrial discharges, particularly in the form of cooling water from power plants, are immediately obvious; however agricultural and forestry activities and urbanization probably are more important because they may influence large areas or even the entire watershed (Wurtz, 1969). Removal of riparian vegetation and the channelization and regulation of streams may change not only mean temperature, but also the diurnal and seasonal pattern of temperature fluctuations, and this may have profound biological consequences.

The sublethal effects of temperature on metabolism and growth may be more important than the absolute tolerance to high temperatures that an individual may exhibit, especially with regard to insect emergence. Increased temperatures may disrupt the seasonal emergence pattern of aquatic insects. Unseasonally high water temperature in winter can cause emergence to occur up to five months early and a greater time lag may occur between the emergence of males and females (Hynes, 1960). Temperature increases could have serious effects on natural communities because insects emerging too early in the season might be killed or inactivated by low ambient temperatures, and an extended period of time between emergence of males and females might prevent mating, thereby eliminating the species from the impacted habitat.

2.3.3 Sewage inputs

Sewage enters the Waterford River from residential areas (13.2% of the drainage area) and industrial areas (5.5%). Sewage, a general term used to indicate the dumping of effluents, can be categorized into six groups of pollutants; 1) inert suspensions (i.e., dirt, clay, dust, etc.), 2) poisons and toxic substances, 3) inorganic reducing agents (e.g., sulphides and sulphites), 4) hydrocarbons, 5) organic residues (i.e., domestic sewage) and 6) hot water inputs. Some of these categories have already been discussed above (e.g., hydrocarbon and thermal inputs).

Storm sewers which drain into the Waterford River, and its tributaries introduce road salts, hydrocarbons and their by-products, and street dust and dirt into the system.

Organic residues drain into the Waterford River system through sanitary and storm sewer cross-connections, and from the drainage fields of household septic tanks in the suburban areas.

These inputs result in increased turbidity, increased organic nutrient loading, and eventually deoxygenation of the water. The end result is a decrease in invertebrate diversity and abundance.

2.4 Habitat changes

Pollution from different sources can often cause similar changes in habitat. These habitat changes could be natural but more often are a result of various industrial, agricultural, and constructional practices. This section deals with how changes in the aquatic habitat can affect the aquatic ecosystem.

2.4.1 Clear cutting and channelization

Headwater streams are influenced strongly by the riparian vegetation which reduces autotrophic production by shading and contributes large amounts of allochthonous material (Vannote *et al.*, 1980). Clear-cutting reduces the amount of leaf detritus, the major energy source for benthic organisms, that the stream receives (Hynes, 1970) and thus there is a reduction in the abundance of shredder organisms, such as some

Ephemeroptera (Haefner and Wallace, 1981). Small streams are closely linked to the terrestrial environments so the surrounding forests exert very important influences on stream ecosystems (Cummins, 1974; Hynes, 1975; Vannote *et al.*, 1980; Haefner and Wallace, 1981). As stream size increases, the importance of terrestrial organic input as an energy source decreases while autochthonous primary production and the transport of organics from upstream become more important (Vannote *et al.*, 1980). The zone through which the stream shifts from allochthonous to autochthonous production is primarily dependent upon the degree of shading (Minshall, 1978). Haefner and Wallace (1981) concluded that: 1) terrestrial successional vegetation influences sequential changes in stream macrobenthos; and 2) following disturbance restoration of the stream benthos community with respect to trophic structure and function is a long-term process dependent on recovery of terrestrial vegetation.

Channelization destroys the habitat and is often linked to clear-cutting and siltation. Channelization has considerable impact on sediment load, water temperature, water chemistry, and aquatic biology. Keefer and Maughan (1985) observed much higher drift densities of benthic invertebrates in channelized versus unchannelized streams, and a corresponding drop in benthos standing crop (i.e., the total invertebrate biomass present in the system at any particular point in time).

2.4.2 Erosion and siltation

Erosion of undisturbed watersheds usually releases rather small amounts of particulate material while certain farming, forestry, or mining practices, along with dredging, industrial, or construction activities, often result in the introduction of substantial amounts of such solids into streams. Aquatic insects have, for the most part, evolved in clean water (Hynes, 1973), thus the effects, either direct or indirect, of siltation can be catastrophic. Silt can block the nets of filter-feeding insects, or can interfere with the normal respiratory processes of benthic animals. Increased silt can also cause a decrease in available food resources. The increased turbidity associated with siltation reduces light penetration and consequently causes a decrease in plant growth. Silt also smothers hard surfaces, and fills interstices within the substrate. Specific pollutants associated with sediment can create problems in addition to those caused by the inert material.

Perhaps the greatest single cause of water quality degradation in streams is siltation and turbidity from inorganic sedimentation. Sedimentation significantly degrades water quality with an increase in bed load and decrease of pH at the substrate-water interface. The most notable difference in the benthic fauna of silted waters, as compared to clear water, is the reduction in the abundance and diversity of filter-feeding Trichoptera and Diptera, predaceous Plecoptera, and certain Ephemeroptera of polluted waters (Lemly, 1982). Certain species are eliminated from the stream insect community through direct or indirect effects of sedimentation alone, but when siltation is combined with nutrient enrichment, a significantly greater number of taxa are eliminated.

Sediment addition to a stream initiates drift of animals at the affected site, and prolonged heavy sediment loads in streams reduces species richness and diversity, although some groups take advantage of the altered habitat conditions (Rosenberg and Wiens, 1978). Fredeen (1985) has observed that a decrease in sediment load in the Saskatchewan River caused a shift in species composition. Damming the river caused it to become shallow and clear, as the sediment settled in the resevoir. The larval black fly *Simulium articum* in sediment-rich rivers, was replaced by *S. luggeri* in clear water.

2.4.3 Eutrophication

Eutrophication is the enrichment of a water system by inorganic nutrients, primarily nitrates and phosphates, usually caused by agricultural, industrial, and/or residential activities. Higher order rivers are usually richer in dissolved nutrients than headwater streams with the load of suspended solids, and thus nutrient levels, increasing downstream (Hynes, 1967; Vannote *et al.*, 1980). In general, eutrophication influences aquatic insects by increasing the production of algae and other vegetation that provide these organisms with substrate, food, and shelter. Organic matter which enters the system from external sources (i.e., pollution) can be used similarly, although bacterial decomposition of nutrient rich organic material may reduce the amount of oxygen available to invertebrates for respiration. This has serious consequences for many species of aquatic insects. Organic pollution generally reduces the number of benthic species; however, since some species benefit from the increased supply of food, the total number of individuals may increase. La Valle (1975) states that a great deal of the increased

phosphate levels in streams running through urban areas can be directly attributed to household sewer systems (i.e., septic tanks) and that an integrated city sewer system would significantly reduce phosphorous loading.

Little is known about the effects of eutrophication on the fauna of flowing waters habitats (Hynes, 1967). It is possible that the effects of eutrophication are not perceived to be as disturbing in flowing waters as they are in lakes. Extensive plant production, characteristic of eutrophic conditions, is often prevented in rivers, due to shading and the attendant high turbidity caused by heavy sediment loads often associated with eutrophic conditions. Nutrients probably pass through a river almost as quickly as the water. Unlike lakes, which can be made eutrophic and then remain in that state, a stream or river has to be continuously enriched. Streams, therefore, can be rescued and restored more readily than can lakes.

2.4.4 Acidification

Acidification of water might not be as much of a problem in Newfoundland as elsewhere since Newfoundland waters are generally acidic and the native fauna is tolerant of low pH. On the other hand, the Island waters have little buffering capacity so that addition of even small amounts of acid could have serious effects (Scruton, 1983). Acidic waters typically have fewer species and lower abundance and biomass of benthic invertebrates than have non-acidic waters. The life cycle stages of molting and emergence are the most sensitive to acid stress. Bell (1971) found that the pH at which 50 per cent of tested aquatic insects emerged is from 0.52 to 2.10 pH units higher than the 30-day TL_{50} value (the pH at which 50% of the test species died after 30 days exposure). In general, Bell (1971) discovered that caddisflies are very tolerant of low pH, stoneflies are moderately tolerant, and mayflies are fairly sensitive. Mayflies may be somewhat limited in numbers and species composition under prolonged acid conditions. Species that live in areas that have snow and ice cover during winter are particularly vulnerable, as the spring thaw brings with it the most pronounced effects of acid precipitation, as the entire winter's accumulation is released over a short period of time.

Another possible source of acidification is acidic mine drainage. This usually is a result

of excavation within the drainage basin and exposure of certain bedrock units like sandstone, shale and coal. Surface run-off from this type of substrate generally has a low pH.

2.5 Urban Activities

Changes in the benthic invertebrate fauna can be a result of several urban-related activites. Streams running through population centres are often subject to many environmental disturbances including the effects of forestry (Newhold *et al.*, 1980), mining (Norris *et al.*, 1981) and agriculture (Dance and Hynes, 1980). City rivers may also suffer changes in water quality, resulting from several urban-related factors including introduction of potentially toxic substances such as road salts and petroleum hydrocarbons from surface runoff, especially in spring (Hynes, 1960; Hall *et al.*, 1985), the introduction of nutrients through sewage effluent (Lemly, 1982), and increased siltation from improper construction activity. In addition, drastic fluctuations in stream water levels may occur as a result of large impermeable surfaces (e.g., asphalt and concrete) that reduce the percolation of rain water into the soil and increase surface runoff.

Whiting and Clifford (1983) found that in an urban environment the benthic insect community structure changed greatly with many upstream insects absent or reduced in the city, but in contrast, the abundance of tubificids and chironomids (Diptera) increased. The diversity of taxa was lower within the city but the total density was much higher. Tubificids accounted for 72% of the invertebrate fauna within the urban areas, but only 14% in upstream sites, which generally fell outside the city limits. Whiting and Clifford (1983) concluded that the increased organic enrichment, due to urban runoff from storm sewer outlets, is the most important factor in determining diversity.

Under undisturbed environmental conditions, a well balanced benthic community is usually present and represented by a relatively great number of taxa. However, different groups and species of organisms within the community exhibit varying degrees of sensitivity to adverse changes in water quality. As stress (i.e., pollution) is placed on a

benthic community, the number of species decreases (i.e., lower diversity) and the distribution of individuals among these species becomes skewed. Partial environmental damage brought on by chemical or physical influences may kill or drive out the most sensitive organisms, while more tolerant forms may increase in numbers to fill this void. Further damage may result in a fauna comprised of very high numbers of a few tolerant forms, while total degradation results in the elimination of even these organisms. The varying degrees of sensitivity of different benthic species to environmental stresses and the poor migratory powers of most of these organisms, make benthic invertebrates an excellent choice for assessing water quality (Gaufin, 1973).

From the foregoing review, it is apparent that many habitat features related to benthic organism distribution are affected by human activity (e.g., water chemistry, temperature, sediment load, hydrological pattern). Although, some human impacts may appear to be temporal, the resulting effects on benthos are significant. Thus, the benthic fauna may present a relatively long lasting record of events occurring in a particular habitat and may provide a relatively easily observed index of change in habitat parameters. However, changes in benthic populations do not specify the event or process responsible for the change but rather the benthos acts as a general indicator of total conditions.

It is in this respect that benthos studies can be a powerful tool for elucidation and detection of changes in environmental conditions. It is from this perspective that benthic organisms are studied to assess the effect of urbanization on environmental conditions of the Waterford River Basin.

2.6 Sampling Methodologies

River beds are typically highly heterogeneous, and this variation in substrate characteristics makes quantitative sampling a difficult proposition (Hynes, 1971). Ideally, one would like to be able to choose one area in the stream, and remove a sample of the entire community. This is usually not easily done, if possible at all.

Several types of samplers have been developed to try to overcome the logistical problems associated with stream sampling. Most samplers are variations on three basic

themes. One of the most common types is the Surber sampler and its various relatives. This piece of equipment consists of a fine-mesh bag attached to a supporting frame. The base of the frame sits on the stream bed, and forms the boundaries of the sampling area; it is usually some standard unit of area measure, typically one square foot. The experimenter then manually cleans any big rocks, etc on the bottom, and agitates the remaining substrate; the principle being that anything dislodged is swept downstream into the net. This type of sampling has its disadvantages. Collections using Surber samplers are limited by depth of the water, inconsistent sampling effort among replicate samples, inconsistent depth penetration into the substrate and variability of substrate type sampled (Resh, 1979). In addition, the number of replicate samples from a site required to ensure suitable confidence that the numbers of taxa and individuals have been appropriately estimated can be as large as 70 (Dickson *et al.*, 1971).

Another common type of sampler is the kind in which a portion of the substrate is isolated. These usually take one of two forms, a drum sampler or a grab sampler. This type of apparatus works by isolating a standard measure of area on the stream bottom. The substrate within that area is then removed, or thoroughly examined, and the benthic organisms are separated. There are drawbacks with this type of sampling as well. Destruction of the substrate makes successive sampling impossible, unless the organisms are separated in the field, and the stream bottom restored to its original condition. The variability of the substrate makes this method of sampling unreliable (Hynes, 1970).

The third type of sampling unit is an artificial substrate sampler. This usually constitutes one of two types; ceramic (or similar) tiles, or rock-filled penetrable bags (or baskets). Artificial substrates have been shown to be useful in quantitative sampling of stream invertebrates (Dickson *et al.*, 1971; Beak *et al.*, 1973; Meier *et al.*, 1979). The duration of exposure of the substrate in the stream affects its colonization with respect to the number of individuals and taxa collected, and the observed diversity of benthic organisms (Meier *et al.*, 1979). Another important factor is that per unit volume, smaller diameter substrates support more individuals and more species than larger diameter substrates (Meier *et al.*, 1979). Total numbers of individuals and species on mixed size

substrate are between the numbers found on small and large substrate (Wise and Mulles, 1979). The advantages of using artificial substrate samplers are that sampling effort, surface area of sampled substrate and water depth can be standardized to provide uniformity of sampling among replicates and among stations (Beak *et al.*, 1973). Artificial substrate samplers do not collect fauna identical to that of the natural substrate; however, in evaluating water quality, the relative distribution of specimens among taxa is more important than the absolute value of either (Dickson *et al.*, 1971; Beak *et al.*, 1973).

Artificial substrates were chosen for this study, because of their relability, practicality and ease of replication.

3 STUDY AREA AND SAMPLING SITES

3.1 Study Area

The Waterford River basin covers an area of 52.7 km^2 and is located on the western outskirts of the City of St. John's (Fig. 1). The Waterford River stretches 13.5 km from where brooks from Bremigans Pond and Brazil Pond meet at an elevation of about 170 m in the west, to St. John's Harbour in the east. The major tributary of the Waterford River is South Brook which extends 11 km from the marshy headwaters, where flow is intermittent, to its confluence with the Waterford River at Bowring Park. Other smaller tributaries drain into either the Waterford River or South Brook.

3.1.1 Physical Features

The Waterford River is partially channelized below Bowring Park, its banks sustaining mostly grass interspersed with a few bushes. In some areas the natural banks have been replaced by concrete walls, which allow little, or no riparian vegetation. Above Bowring Park, the river runs through areas ranging from channelization, bare banks, and suburban developments to relatively undisturbed areas with healthy grass, bush and tree growth on the banks.

The substrate of the Waterford River is mostly gravel with less than half being rubble and a small portion being sand, mud or bedrock. A survey conducted by Arambarri and Haedrich (1983) showed that the river bed was 50.4% gravel, 34.4% rubble, 10.5% bedrock, 0.8% sand, and 3.9% mud along it's entire length. The river was composed of 69.4% riffle, 28.7% pool, and 1.9% falls. Instream vegetation occurred on 24.9% of the river bed.

South Brook flows through farm and forested terrain for its entire length. The brook was 79.7% riffle, 19.2% pool, and 0.7% falls. The substrate was composed of 64.7% gravel, 25.2% rubble, 5.1% bedrock, 0.4% sand, and 1.1% mud. Instream vegetation occurred on 5.9% of the river bed (Arambarri and Haedrich, 1983).

In a highly urbanized watershed, typically with a large number of point sources of

pollution, zones of clean water, degradation, and recovery may be ill-defined or non-existent. This is probably true for the Waterford River system, with less than 25% urbanization, since development has already proceeded along almost all of the watercourse. The Waterford River has been subjected to a variety of urban-related stresses including channelization, damming, and the pollutants associated with residential and industrial development. The stresses on the river system are probably cumulative and consequently, as the river winds towards downstream urban core areas, the water quality may not recover prior to flowing into the harbour.

The portion of the Waterford River basin (52.7 km^2) which comprised the study area extends upstream of the Kilbride site. The remaining portion of the basin, downstream of the Kilbride site, is highly urbanized and in a number of places the drainage system has been modified. Also, there are several unidentified cross connections of sanitary and storm sewers and domestic sewage outfalls in this section. This part of the basin was therefore not included in the study area.

3.1.2 Geology and Soils

Bedrock of the area consists largely of Precambrian materials, mostly sedimentary in origin but with some volcanic deposits. The principal rock types are Hydryrian siltstone, arkose, conglomerate, slate, and acidic to intermediate volcanic rocks. The most significant features which influence the river course in the basin are major plunging folds and fracture zones in low porosity rocks which generally slope toward the Waterford River and South Brook. Secondary growths of pyrite and pyrolusite are commonly altered to iron and manganese precipitates along fractures in some formations, including thinly bedded sandstones, which can adversely affect water quality (King, 1984).

Most of the study area is covered by materials of glacial origin which range in depth from 0 to 5m. The overburden is composed of a very compact, poorly sorted lodgement till with high silt-clay content overlain by a till deposit that is loose, coarser, and more permeable. The soils were formed largely by the action of weather and glacial deposits. The soils are coarse textured and contain stones, gravel, and boulders (Batterson, 1984). Water movement is largely determined by the overburden mantle. Soil capability for

agriculture and forestry in the area is low because of unfavourable topography, stoniness, and shallow depths of the soil (Yoxall, 1981).

3.1.3 Climate

The climate of the Avalon Peninsula is somewhat more temperate than the remainder of the Island, being modified by its proximity to the sea. The mean annual temperature is 4.0°C, mean relative humidity is 86%, average annual precipitation is 1600 mm and estimated annual evaporation is 380 mm. The distribution of precipitation is fairly uniform throughout the year except June and July which receive relatively less rainfall. Snowfall accounts for about one third of the total annual precipitation (Banfield, 1983).

3.1.4 Land Use and Land Use Changes

Land use maps of the study area were prepared by The Working Group on Land Use and the extent of various land uses in the study area, based on the 1981 and 1984 maps, is presented in Table 1.

The study area has not experienced any significant change in land uses during the period under study. There was only a small increase (2.1%) in urban land use. There has been sustained decrease in forestry and agricultural uses. This decline contributed mostly to unproductive land areas (cleared and open areas) which generally is a transitional stage between forest and developed land. These changes were observed to be generally similar in all sub-basins. The extent of urbanized area in any of the sub-basins did not exceed 20-25%.

3.2 Sampling Sites

For the first year of the program, seven sampling sites were selected. However, for the second year of the study, two new sites were added for the reasons noted below, bringing the total number of sampling sites to nine. The location of the sites is shown in Figure 3, and a description of each sampling site is given in Section 3.2.1.

Four sampling sites were located on the Waterford River from Donovan's to Kilbride, just below its confluence with South Brook. Two sites were located on an unnamed tributary of the Waterford River, which runs through the Agriculture Canada Brookfield

Table 1. Land Uses Changes During 1981-1984.

	1981		1984		
Land Use Category	Area km^2	% of Basin Area	Area km^2	% of Basin Area	% Change
Residential	6.6	12.5	7.3	13.2	+1.3
Commercial/ Industrial	2.5	4.7	2.9	5.5	+0.8
Agriculture	6.2	11.7	5.9	11.2	-0.5
Forest	19.7	34.3	17.2	32.6	-4.7
Unproductive Lands	13.2	25.0	15.3	29.0	+4.0
Other (Recreation, ponds lakes, etc.)	4.6	8.1	4.8	8.7	+0.6

Research Station. Three sampling sites were located on South Brook, with the farthest upstream site acting as the control site for the study. This site was in an almost undisturbed area, and as such it was decided that it could be taken as being representative of the natural condition. However, some habitat alteration occurred at this site over the course of the two-year study. These changes will be described in the next section. The samples from one site on the Waterford River were lost during the first year of the study, and one of the sites on the unnamed Waterford River tributary was added only for the second year of the study. The reason for this addition will also be discussed in the next section. The sites corresponded, approximately, to sites where water quality was monitored (Figures 2 and 3, Table 2). This was done to facilitate development of correlations between water quality differences and benthic invertebrate populations. The information on surface water quality in the basin was obtained from the Surface Water Quality Study Report.

3.2.1 Description of Sampling Sites

The following description of the sampling sites include factors that might affect the invertebrate fauna. Factors taken into account include sources of pollution, amount of vegetation and shading along the river bank and the size of the stream. All references to the right or left hand sides assume the observer to be facing downstream. Table 2 indicates the location of the sampling sites and the corresponding stream order and Figure 3 shows the locations of the sampling sites within the drainage basin. Figures 4-12 are photographs depicting each site. The placement of the artificial substrates in the streams is diagrammed in Appendix F.

Site 1

Site 1, located at Kilbride on the Waterford River, was the most downstream site. It is a fourth order stream about seven metres wide located one kilometre downstream from where South Brook enters the Waterford River.

About 250 m upstream from the sampling site a storm sewer outfall is installed in the retaining wall on the left bank; much of the river upstream had been channelized. Small

shrubs and grasses grew along the river banks but there were few tall trees and thus little shading. Water at the site was usually turbid. At the confluence of South Brook and the Waterford River a mixing of the relatively clear water from South Brook with the usually turbid water from the Waterford River was seen. Gravel bars moving downstream continuously shifted the configuration of the bed on the left side of the channel. This part of the bed was frequently exposed during low to medium flows but was inundated during high flows.

The river was located about three metres below the level of a road which ran parallel to it, and the river also flowed under a bridge about 50 m upstream from the sampling site.

Site 2

Site 2 was located on a tributary of South Brook, a second order stream about 2 m across. The site was in an area of some agriculture with cow pastures upstream of the site. Grass grew along the banks of the river for at least three kilometers upstream and thus there was no shading of the river.

The river passed under Old Bay Bulls Road about 20 m upstream from the sampling site, at which point a highway ditch drained into the brook. The water was very polluted due to poorly placed and managed local septic tank systems and surface run-off from local dairy cow pastures.

Site 3

Site 3 was the second most upstream site on South Brook where the stream was second order and about 3.5 m across. The river was completely shaded by a mixture of deciduous and coniferous trees. The rocks in the river were moss covered and the water was usually clear.

The river ran parallel to Heavy Tree Road for about 1 km and was only about 5 m from the edge of the pavement. The river ran through several small properties and farms before coming to the sampling site.

Table 2. Location of sampling sites within the study area.

Site	Location	Distance from Water Quality Study Site	Drainage Area (km^2)	Stream Order
1	Waterford River, Bowring Park, Kilbride	10m downstream	52.70	4
2	South Brook tributary, Old Bay Bulls Road	15m downstream	6.01	2
3	South Brook, Heavy Tree Road	20m upstream	7.40	2
4a	Waterford River tributary, Agriculture Canada Experimental Farm	25m upstream	3.16	2
4b	Waterford River tributary, Agriculture Canada Experimental Farm	400m upstream	3.02	2
5	Waterford River, Commonwealth Ave. Bridge, Mount Pearl	10m upstream	16.60	3
6	Waterford River, Donovan's	700m upstream	11.40	3
7	Waterford River, Dunn's Road Bridge, Mount Pearl	20m upstream	21.10	3
8	South Brook, Ruby Line	10m upstream	5.41	2

Note : Drainage area was obtained from Dept. of Environment. Stream order was determined from Dept. of Energy, Mines and Resources topographic maps (1:25 000 1N/10c Edition 3).

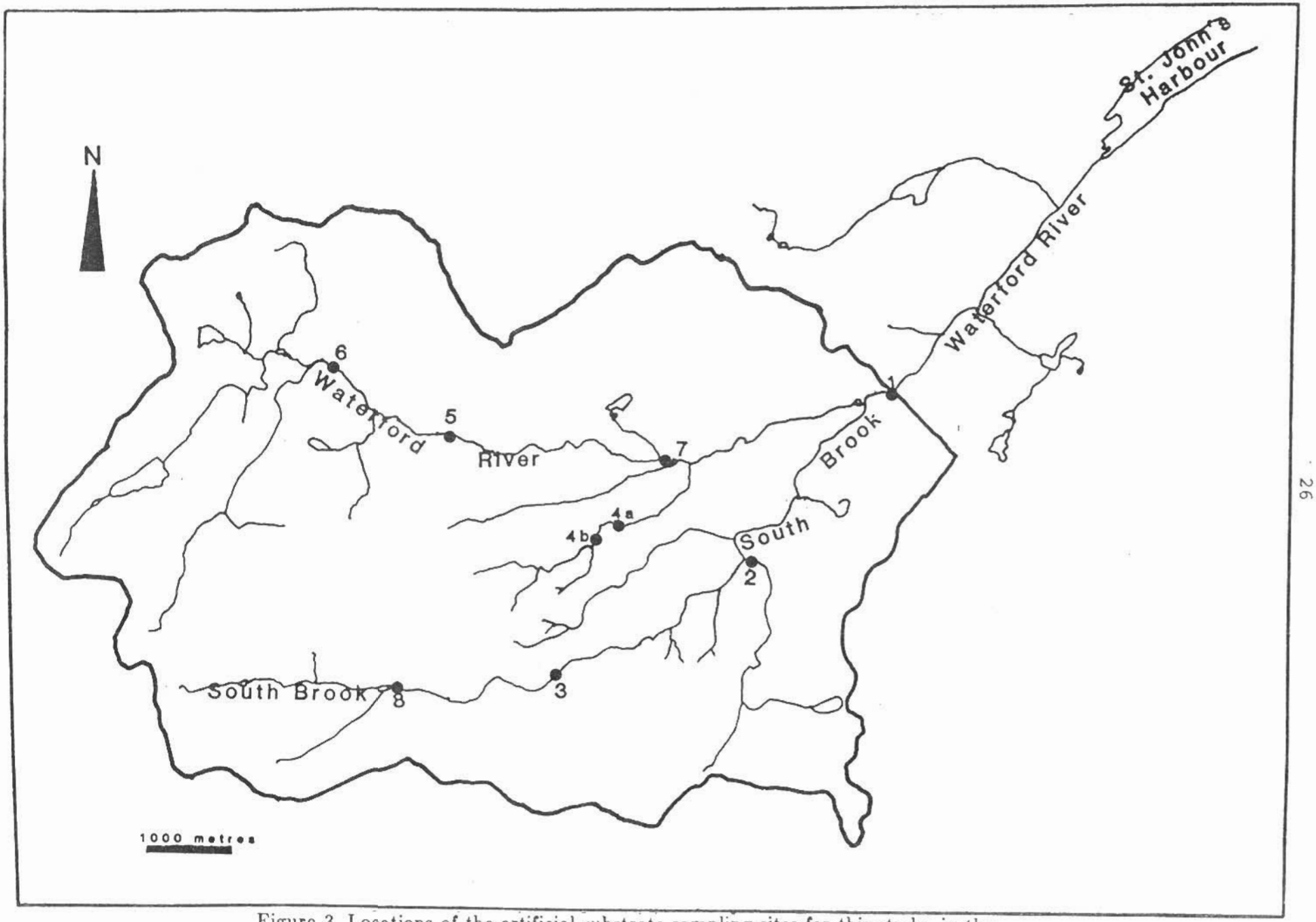


Figure 3. Locations of the artificial substrate sampling sites for this study, in the Waterford River Basin

Figure 4. Site 1.

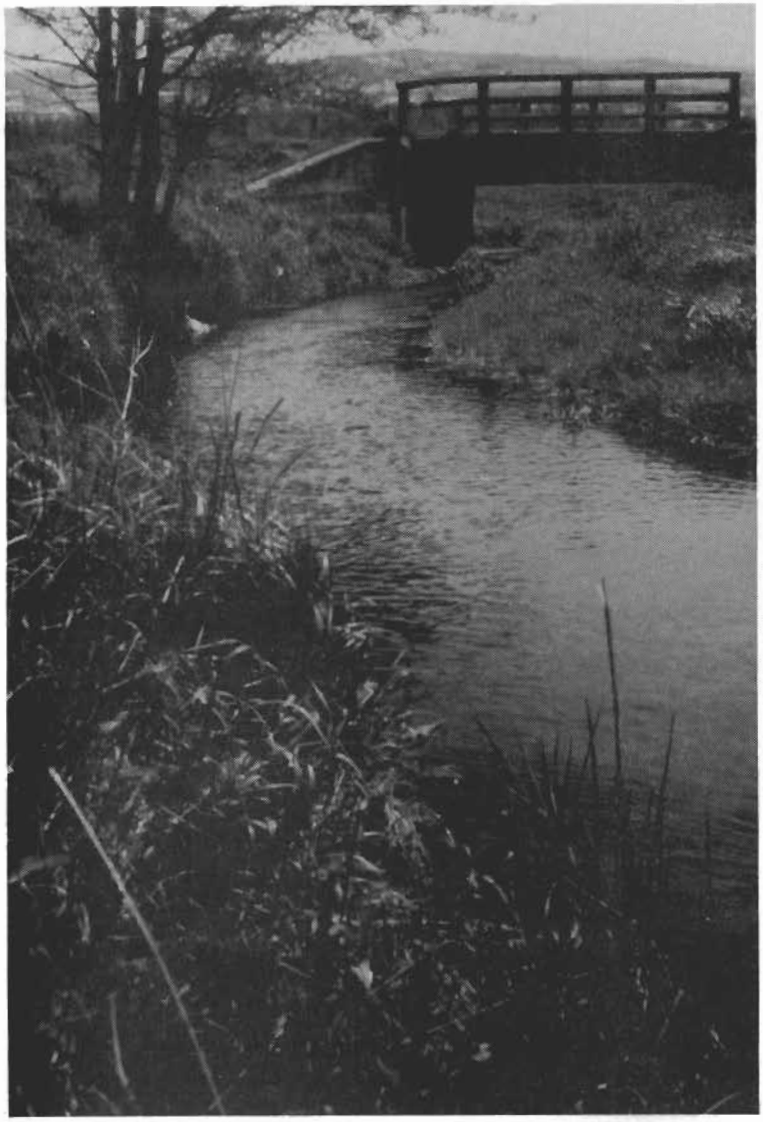


Figure 5. Site 2.

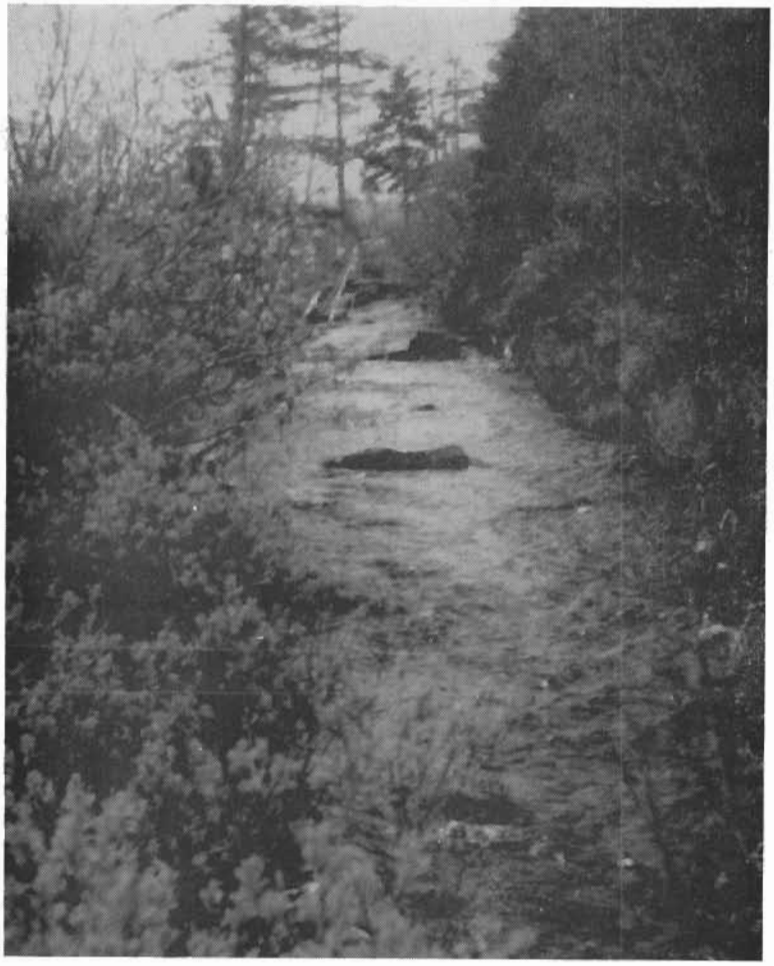


Figure 6. Site 3.

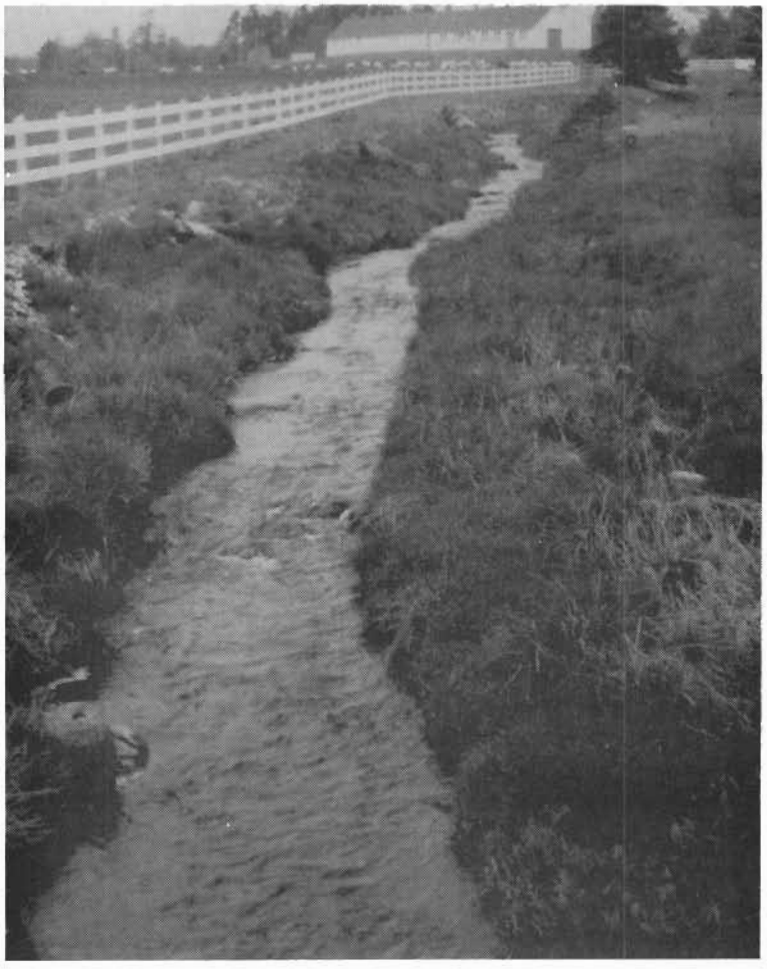


Figure 7. Site 4a.

Site 4a

Site 4a was located on a second order tributary of the Waterford River. The stream was less than 1 m across where it flowed through the Canada Department of Agriculture Experimental Farm.

There was no shading of the river and a sheep pasture existed just upstream of the site. The water was rarely if ever completely clear, even when not highly turbid it had a hazy quality to it. Approximately 300 m upstream of the sampling site there was a small man-made reservoir. There were usually geese on the reservoir.

Site 4b

Site 4b was located on the same tributary. The sample site was located about 20 m upstream of the reservoir. The area was almost completely shaded by conifer growth.

The stream was about 2 m across, and the water was generally clear. The brook was shallow and relatively fast flowing, with a loose pebble/ rocky substrate. This site was added after the first year, as only a small number of species (and individuals) were found at Site 4a. It was reasoned that the majority of invertebrates were entrapped in the reservoir. Therefore, it was decided to sample a site upstream of the reservoir (i.e., Site 4b).

Site 5

Site 5 was located on the Waterford River in Mount Pearl. The stream was a third order stream about 7 m across. The river was well shaded with many large trees growing along the banks.

The water was usually shallow and the bottom was rocky. The water was usually very turbid.

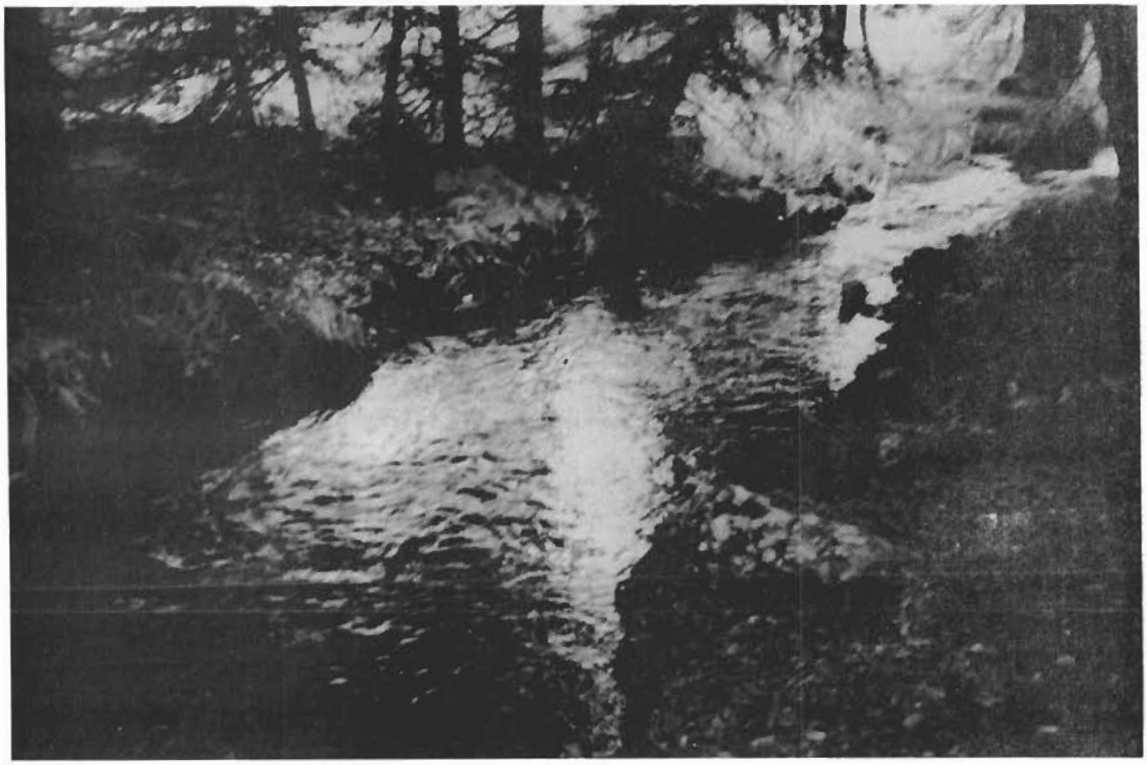


Figure 8. Site 4b.

Figure 9. Site 5.

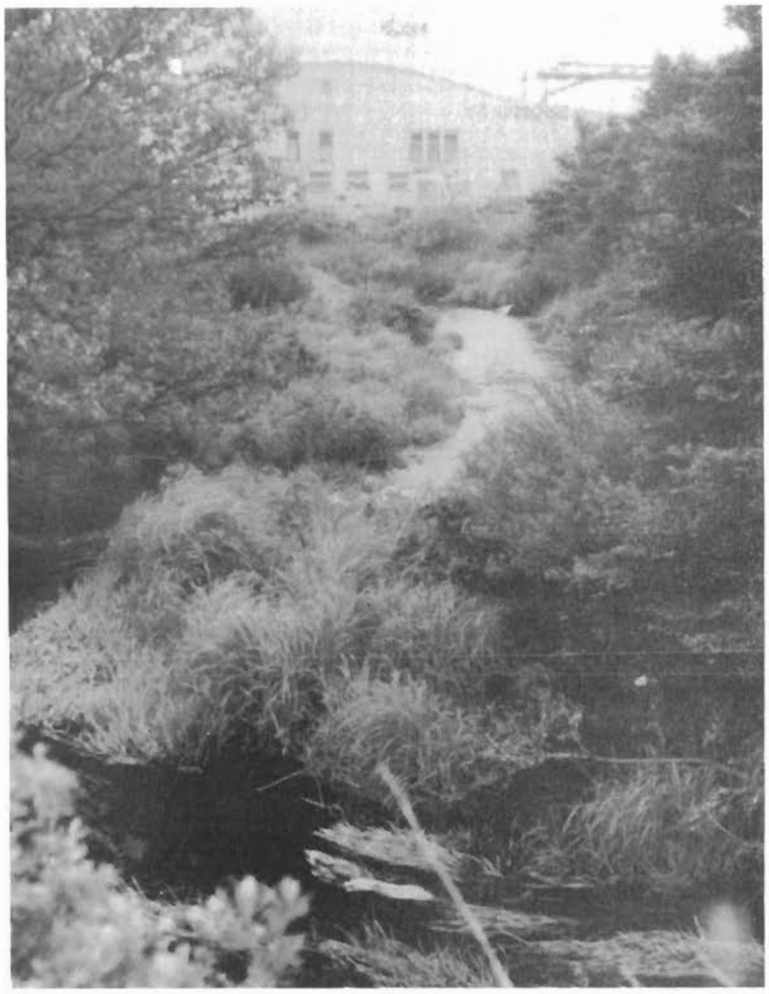


Figure 10. Site 6.

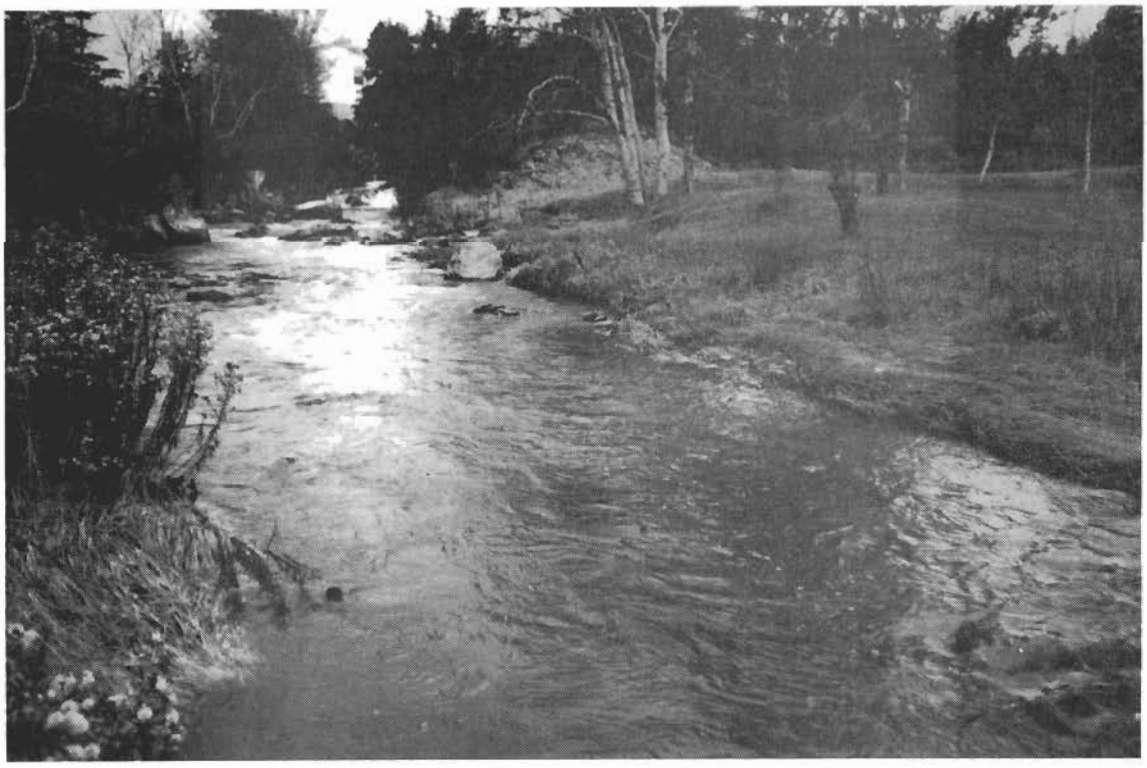


Figure 11. Site 7.

Site 6

Site 6 was located on the Waterford River in Donovans Industrial Park about 4 km upstream of Site 5. Although the river was a third order stream it was only about 3 m wide. The bottom was covered with aquatic plants and grasses.

The water was very turbid and the channel upstream was fairly broad with a bottom that was very muddy, having pockets of gas.

A railroad track runs near the river and at some points bridges it. Some industrial buildings, e.g., Newfoundland Hardwoods and Supersweet Feeds, were located close to the river. Water also flowed from a manhole which was connected to a storm sewer trunk.

Site 7

Site 7 was located on the Waterford River in Mount Pearl 20 m upstream of the Dunn's Road Bridge, about 2 km below Site 5. The river was third order at this site and was about 7 m across.

The stream lies in close proximity to a residential area and the site was fairly well shaded except for the right bank, which was partially bare. Another channel, which drained Sobey's Square and surrounding area to the north, entered the Waterford River about 30 m downstream of the site. Small waterfalls were located just upstream and downstream of the site. This site was not sampled in the first year of the study, as the rock bags had erroneously been removed during a river clean-up project.

Site 8

Site 8 was the uppermost site on South Brook. At this location, the river was a second order stream about 1.5m wide and was completely shaded by a mixture of deciduous and coniferous forests. The water was always clear even during rain storms.

Figure 12. Site 8.

It had originally been intended to use this site as a control, as there had been little human development in the area. Since the initiation of this study, however, an area upstream of the sample site has been clear-cut, a wide right-of-way has been cleaned out, and some waste materials (e.g., car wrecks) have been dumped in the area. The sampling site itself has been left relatively undisturbed, however some alteration of the stream characteristics has probably occurred.

3.2.2 Artificial Substrates

As described previously, in Section 1, the sampling unit was an artificial substrate sampler, consisting of a bag made of 0.7 cm Vexar's nylon mesh tubing, filled with 1 kg crushed rock (1 to 3 cm in size), and tied at both ends. At least fifteen rock bags were placed in the river at each sampling site. A tether wire was firmly fastened to a tree, or a steel rod embedded in the river bank. The rock bags were attached to this wire, and strung out into the river.

The bags were placed in the streams in such a way as to minimize the chances of being exposed during periods of low flow, and to maximize the potential of invertebrate colonization of each bag. In most cases they were placed in riffle areas, with fast moving current. Site 2 did not provide an area with these characteristics, thus the bags were placed in an area which was of comparable depth to the other sites, but the flow rate was lower.

3.2.3 Sampling Program Implementation

First Phase of Study - 1984-85

The artificial substrate samplers for the first phase of the study were placed in the rivers in April, 1983, but due to vandalism, four sets had to be replaced in December, 1983. To ensure an equal colonization period for all sites, the rock bags at the other four sites were thoroughly washed in the field at this time. The samplers were lifted from the streams in December, 1984.

Second Phase of Study - 1985-86

In September, 1985, the artificial substrates for the second phase of the study were

placed in the rivers, at the same locations, with the addition of Site 4b and Site 7. These samplers were removed in December, 1985. The sampling units were left in the streams for only four months, instead of the original twelve, to reduce or eliminate some problems encountered in the first year (e.g., siltation of the substrates, dislodgement of substrates during spates, vandalism, etc.). Studies have shown that four months is a more than adequate period for colonization (Dickson *et al.*, 1971; Spruce Budworm Spray Monitoring Program Report, 1984; Larson, 1985).

4 SAMPLING PROGRAM

The sampling program was set-up and conducted as follows.

4.1 Sampling Frequency:

The Waterford River system was sampled twice for benthic macroinvertebrates, at the sampling sites identified in Section 3.2.1. and shown in Figures 4-12. In December, 1984, samples were taken from seven locations. In December, 1985, the same seven locations, along with two additional locations, were sampled. For both sampling periods, a total of ten samples from each site were analyzed. The position of the bags in the streams is given in Appendix F.

4.2 Sampling Methodology/Preservation Procedure

4.2.1 Sampling unit

As noted earlier, the sampling unit was an artificial substrate sampler, which consisted of a mesh bag, containing 1 kg of crushed rock, ranging in size from 1 cm to 3 cm.

4.2.2 Sample Retrieval/Preservation Procedure

In December, 1984 and again in December, 1985, the samplers were lifted from the river. The bag removal procedure was as follows: The tether wire attached to the bag was cut, prior to the bag being disturbed. The loose bag was then "flipped" into a bucket, while the bucket itself was moved in and under the bag. With some practice, the entire procedure could be completed in one swift movement. It had originally been intended to have a plankton net held just downstream below the bag, to catch any drifting organisms, but the "quick flip" method is faster and was found to be effective in this study. Considering the problems associated with using a net in December, in sub-zero temperatures, this is perhaps a valid consideration. Debris attached to the bag, or to the wire, in close proximity to the bag, was also collected. This may introduce a slight positive bias in the samples for the study, however it was felt that this would be preferable to removing the debris, and possibly losing specimens as a result of the disturbance.

Once all the samplers at a site were removed, they were transported to the

Environmental Protection Service (E.P.S.) laboratory located in the Northwest Atlantic Fisheries Centre. Here, each rock bag was individually scrubbed, and the associated macroinvertebrates and debris were placed in a Mason jar, and preserved in Kahle's fluid (Wiggins, 1977). The contents of each jar constituted a sample for analysis. Kahle's contains formalin, ethanol and glacial acetic acid. To minimize the odor problem the specimens were transferred to 95% ethanol after being fixed in the solution for three weeks. Dickson *et al.* (1971) reported that four to six rock bag samples provided similar statistical confidence to that obtained from 70 surber samplers. Community structure analysis would require one or two samples per site to achieve 95% confidence limits that the mean diversity index values were within 25% of the true values (Dickson *et al.*, 1971). Based on such observations, and time constraints, ten samples from each site were cleaned and sorted for later identification. Any extra samples were also preserved for later use, if necessary. Because of time constraints only ten rock bags from each site were analyzed, these being selected randomly using a die, for the first year of the study. The bags for the second year of the study were selected using a random number generator on a Texas Instruments TI99/4A home computer.

4.3 Biomass Determination

Two possible methods of biomass determination could be used in this type of study (i.e., volumetric displacement in absolute ethanol, or weighing on an analytical balance). Volumetric displacement of organisms, accomplished by placing organisms in a graduated cylinder of 100% alcohol, was done for the first five samples from each site, in the first year of the study program. These samples were also weighed on an analytical balance. Both procedures yielded similar results, and as the volumetric displacement procedure was more time consuming, it was decided to proceed with the simple weighing method for the remaining samples in the first year and the entire second year program.

4.4 Identification of Benthic Invertebrates

All the samples were cleaned, sorted, and analyzed at the Biology Department of Memorial University of Newfoundland (M.U.N). The individually preserved samples were placed in white enamel pans, and the associated benthic organisms were removed by hand, under a 2X illuminated magnifying glass. The residues were retained in the

original Mason jars. A number of these were chosen at random, and examined by an independent experienced biologist to determine whether any individuals, or more importantly, any taxa, had been overlooked.

Several taxonomic keys were used for the identification of macroinvertebrates; Coleoptera (Leech and Chandler 1963), Diptera (Wirth and Stone 1963), Odonata (Walker, 1953; Walker 1958; Walker and Corbett, 1975) Plecoptera (Jewett 1963), Trichoptera (Wiggins 1977), and non-insect invertebrates (Pennak 1978). Species-specific data for Newfoundland insects was found in Larson and Colbo (1983) and Marshall & Larson (1982). Additional taxonomic references were obtained from Merritt and Cummins (1978).

Identification of macroinvertebrates was completed by two honours students at the Department of Biology at M.U.N., and were checked by an independent faculty member. The identification was conducted by Tony Clemens B.Sc. in the first year, and Bill Stirling B.Sc. in the second year. Dr. D. Larson checked the above-mentioned residues, and most insect identifications. Dr. J. Pickavance checked and/or conducted the annelid identifications.

Some of the specimens from the 1984 sampling program were kept at M.U.N., to be used as a reference collection. During the identification of one genus of caddisfly (Trichoptera) in the second year it was noticed that what had been identified as larval *Hydropsyche recurvata* in the reference collection was in fact *H. sparna*. All remaining specimens were re-examined, and were determined to be *H. sparna*. It was therefore considered that the same misidentification occurred throughout the results obtained in the first year, and it was decided to simply add the number of *H. sparna* and what had been identified as *H. recurvata* together, to come up with a single value for *H. sparna*. This was further supported by the observation that *H. recurvata* would not normally be expected to be collected in a river system such as the Waterford system. They are characteristically found in open, unshaded slow moving reaches, or in littoral areas of lakes (Wiggins, 1977). In Newfoundland, this species is usually found in streams at lake outlets (Genge, 1986).

Chironomidae (Diptera) are typically highly abundant in streams, and the Waterford River system is no exception. The ubiquitous midge larvae were found in varying abundance throughout the system, and could not be quantified accurately in the time-frame allowed by this study. No attempt was made to collect all chironomid larvae in each sample, as undoubtedly some samples contained several hundred and many were very small. Chironomid abundance was rated as an index value assigned as outlined below;

Number of Chironomidae	Abundance Index	
0-9	0	Rare
10-49	1	Few
50-99	2	Common
100+	3	Abundant

Organisms were identified to genus and to species, wherever possible, except Diptera and oligochaetes which were identified to family level. The total number of individuals in each taxon in a sample was determined.

A total of 69 different taxa were collected over the two-year period, however, in any one year of the study, only 60 different taxa were collected. A list of these taxa is given in Table 3. There were nine taxa collected in 1984 but not in 1985, and likewise, there were nine taxa collected in 1985, but not 1984. The faunal composition of the various sites, for both years of the study, is summarized in Tables A1 to A18, contained in Appendix G. Some taxa were very common, being found in relatively high numbers in most samples (e.g., *Hydropsyche spp.*, *Ephemerella subvaria* and *Baetis tricaudatus*). Other taxa were rare, often represented by one individual in one sample in only one year (e.g., *Mystacides sepulchralis*, some *Rhyacophila spp.* and *Hydroporus badiellus*).

Table 3. List of Taxa Collected During the Two-Year Study

Taxon	Collected in 1984	Collected in 1985
Phylum Nematoda	+	+
Phylum Annelida		
Class Hirudinea	+	-
Class Oligochaeta		
Order Lumbriculida		
Family Lumbricidae		
Lumbricus sp.	+	+
Family Lumbriculidae		
Lumbriculus variegatus Muller	+	+
Order Haplotaxida		
Family Enchytraeidae	+	+
Family Naididae	+	+
Nais communis		
Phylum Mollusca		
Class Gastropoda	+	+
Class Pelecypoda	+	+
Phylum Arthropoda		
Class Arachnida		
Order Acarida		
Family Hydracarinidae	+	+
Class Crustacea		
Order Amphipoda	+	+
Hyalella azteca		
Order Copepoda		
Family Calanoida	-	+
Class Insecta		
Order Coleoptera		
Family Dytiscidae		
Hydroporus badiellus Fall	+	-
H. paugus Fall	-	+
Family Elmidae		
Promoresia tardella Fall	+	+
Stenelmis crenata Say.	+	-

(continued)

Taxon	Collected in 1984	Collected in 1985
Family Hydrophilidae		
Hydrobia fuscus L.	-	+
Order Diptera		
Family Ceratopogonidae	+	+
Family Chironomidae	+	+
Family Empididae	+	+
Family Muscidae	+	+
Family Psychodidae	+	-
Family Simuliidae	+	+
Family Tabanidae	+	+
Family Tipulidae		
Antocha sp. Osten and Sacken	+	+
Limonia sp. Meighen	+	-
Tipula sp. L.	+	+
Order Ephemeroptera		
Family Baetidae		
Baetis flavistriga McD.	+	+
B. pygmaeus Hagen	+	+
B. tricaudatus Dodds	+	+
Centroptilum convexum Ide.	+	-
Family Ephemerellidae		
Ephemerella subvaria McD.	+	+
Eurylophella sp. McD.	+	+
Family Leptophlebiidae		
Habrophlebia vibrans Needham	+	+
Leptophlebia cupida Say	+	+
Paraleptophlebia adoptiva McD.	+	+
Order Odonata		
Suborder Anisoptera		
Family Aeshnidae		
Aeshna sp. Walker	-	+
Order Plecoptera		
Family Capniidae		
Paracapnia opis Newman	+	+

continued

Taxon	Collected in 1984	Collected in 1985
Family Leuctridae		
Leuctra ferruginea Walker	+	+
Family Perlodidae		
Isogenus frontalis Newman	+	+
Isoperla transmarina Newman	+	+
Order Trichoptera		
Family Brachycentridae		
Micrasema wataga Ross	+	+
Family Glossosomatidae		
Glossosoma sp. Curtis	+	+
Family Hydropsychidae		
Arctopsyche ladogensis Kolenati	+	+
Hydropsyche betteni Ross	+	+
H. slossonae Banks	+	+
H. sparna Ross	+	+
Family Hydroptilidae		
Hydroptila metoeca Bickle and Morse	+	+
Oxyethira sp. Eaton	+	+
Family Lepidostomatidae		
Lepidostoma sp. Rambur	+	+
Family Leptoceridae		
Mystacides sepulchralis Walker	-	+
Family Limnephilidae		
Hydatophylax argus Harris	-	+
Limnephilus sp. Leach	-	+
Pychnopsyche sp. Banks	-	+
Platycentropus sp. Ulmer	+	-
Family Philopotamidae		
Chimarra aterrima Hagen	+	+
Dolophiloides distinctus Walker	+	+
Wormaldia moesta Banks	+	+
Family Phryganeidae		
Oligostomis sp. Kolenati	+	+
Ptilostomis sp. Kolenati	+	+

continued

Taxon	Collected in 1984	Collected in 1985
Family Polycentropodidae		
Neureclipsis sp. McLachlan	+	-
Polycentropus sp. Banks	+	+
Family Rhyacophilidae		
Rhyacophila carolina Banks	+	+
R. fuscula Walker	+	+
R.invaria Walker	+	-
R. melita Ross	-	+
R. minora Banks	+	+
R. nigrita Banks	-	+
R. torva Hagen	+	+
R. vibox Milne	+	+

Note + collected
- not collected

5 ANALYTICAL PROCEDURES

The collected data was analyzed with the objective of identifying "clean water" and "unclean water" communities of macroinvertbrates. Analysis was conducted using the facilities of Computer Services at Memorial University. All data was entered into computer files, and upon completion of this study, these files were transfered to diskette, to be maintained for later use by the Newfoundland Department of Environment. These are available upon request.

5.1 Faunal Composition

The invertebrate data was examined on three levels; a) intra-site variation, b) inter-site variation and, c) between year variation. The data analysis procedures used at each level, and the results of these tests are discussed in the following sections.

5.1.1 Intra-site Variation

Faunal variation between samples at a site was examined by calculating coefficients of variation (C.V.) for the number of taxa at each site, and the number of individuals in each taxon with a statistical mean of at least one individual per sample. The values of coefficient of variation were calculated to give some idea about the spread of the data points, independent of the value of $\bar{X}$, using the formula:

$$\text{Coefficient of Variation} = \frac{\text{Standard Deviation}}{\text{Statistical Mean}}$$

The computed values of coefficients of variation are given in Tables 4 and 5. The values ranged from 0.30 for *Hydropsyche sparna* at Site 7 in 1985, to a high of 2.30 for *Wormaldia moesta* at Site 8 also in 1985. Variation in the position of the artificial substrate bags in the streams, particularly in the first year of the study, may account for some of the observed high values of coefficient of variation.

5.1.2 Inter-site Variation

The variation among sites in terms of taxon number, biomass, and number of individuals was examined by analysis of variance (ANOVA) using the statistical package SPSS-X (Nie, 1983). Taxa which were determined to have a mean of at least one individual per sample for at least one site were included in the analysis.

Table 4. Coefficients of Variation calculated for data from first year of study.

	Site Number						
Taxon	1	2	3	4a	5	6	8
Nematoda	-	-	1.25	-	-	-	-
Lumbricidae	-	-	-	0.76	-	-	-
Enchytraeidae	-	1.25	-	1.58	1.04	-	1.13
Naididae	-	2.16	-	-	1.13	-	1.31
Lumbriculus variegatus	0.80	1.18	2.25	-	0.99	-	0.90
Promoresia tardella	1.30	-	0.73	-	-	-	0.78
Paracapnia opis	-	-	-	-	-	-	0.52
Leuctra ferruginea	-	-	-	-	-	-	0.67
Isoperla transmarina	0.45	-	0.96	-	-	-	1.11
Baetis pygmaeus	-	-	2.18	-	-	-	-
Baetis tricaudatus	1.21	1.69	1.05		0.87	-	-
Ephemerella subvaria	0.68	1.51	-	-	0.59	1.29	-
Habrophlebia vibrans	-	-	-	-	-	-	1.53
Paraleptophlebia adoptiva	-	-	0.79	-	-	-	-
Ceratopogonidae	-	-	-	-	-	-	0.78
Chironomidae	1.16	0.96	0.46	0.66	0.95	1.13	0.56
Tipula sp.	-	-	1.45	1.08	-	0.84	1.05
Arctopsyche ladogensis	-	-	-	-	0.80	-	-
Hydropsyche betteni	-	1.07	-	-	1.27	1.65	-
Hydropsyche sparna	0.84	1.08	0.81	-	0.45	0.87	1.01
Hydropsyche slossonae	1.10	1.16	-	-	0.48	1.15	-
Rhyacophila carolina	-	-	-	-	-	-	1.17
Rhyacophila fuscula	-	-	-	-	0.84	-	-
Chimarra aterrima	-	-	0.94	-	-	-	1.97
Micrasema wataga	-	-	0.80	-	-	-	-

Table 5. Coefficients of variation calculated for data from the second year.

	Site Number								
Taxon	1	2	3	4a	4b	5	6	7	8
Lumbricidae	-	-	-	1.20	-	-	-	-	-
Nais communis	-	-	-	-	-	-	-	1.62	-
Promoresia tardella	-	-	106	-	-	-	-	-	0.54
Leuctra ferruginea	-	-	-	-	-	-	-	-	0.59
Isoperla transmarina	1.08	1.41	0.71	-	0.94	-	-	1.01	0.99
Isogenus frontalis	-	-	1.63	-	-	-	-	-	1.25
Chironomidae	0.64	0.47	0.91	0.51	0.71	0.80	0.52	0.51	0.42
Tipula sp.	-	-	-	0.73	-	-	-	-	0.52
Simuliidae	-	-	0.97	-	-	-	-	-	0.87
Baetis pygmaeus	-	-	1.09	-	-	-	-	-	-
Baetis tricaudatus	0.98	1.12	0.74	1.02	0.37	0.72	0.85	0.87	
Ephemerella subvaria	0.92	0.69	0.52	-	-	0.68	1.10	0.44	-
Paraleptophlebia adoptiva	1.36	-	1.55	-	-	-	-	-	-
Leptophlebia cupida	-	1.01	-	1.38	0.52	-	-	1.13	-
Hydropsyche betteni	-	1.67	1.46	1.47	0.46	-	2.12	-	-
Hydropsyche sparna	0.56	0.56	1.18	-	0.59	0.48	0.52	0.30	0.81
Hydropsyche slossonae	0.56	0.52	0.81	-	1.24	0.47	0.86	0.34	2.02
Chimarra aterrima	-	-	1.78	-	-	-	-	-	-
Wormaldia moesta	-	-	-	-	-	-	-	-	2.30
Rhyacophila fuscula	0.96	-	1.25	-	-	0.64	-	-	1.08
Hydatophylax argus	-	-	-	-	0.82	-	-	-	-
Lepidostoma sp.	-	-	1.79	-	-	-	-	-	-
Micrasema wataga	-	-	0.94	-	-	-	-	-	-
Oligostomis sp.	-	1.00	-	-	-	-	-	-	-

The results of ANOVA showed a number of taxa, in both 1984 and 1985, to have statistically significant (i.e., F-test result with $p<0.05$) differences in the mean number per sample among the sampling sites. To determine where these differences occur, multiple comparison tests were conducted on taxa which yielded a significant F-value. A multiple comparison test calculates a critical difference, and any two sites whose means differ by an amount greater than the critical value are declared significantly different.

The multiple comparison tests conducted were a Scheffe test and a Tukey test. The critical difference in the Scheffe test is calculated using the formula;

$$Q_{ij}=\sqrt{(k-1)\cdot F_{\alpha,k-1,N-k}}\cdot\sqrt{MSE\cdot(1/n_i+1/n_j)}$$

where k is the number of sampling sites, n_i is the number of samples from site i, n_j is the number of samples from site j, N is the total number of samples taken (i.e., n·k), and MSE is the mean square error from the ANOVA calculation

The critical difference in the Tukey test is calculated using the formula;

$$HSD=q(\alpha,k,N-k)\cdot\sqrt{MSE/n}$$

where k is the number of sampling sites, n is the number of samples taken per site, N is the total number of samples taken (i.e., n·k), MSE is the mean square error from the ANOVA calculation, and q is the appropriate Studentized Range Statistic corresponding to the values of α, k and N-k,

Both tests are similar, however the Tukey test is more appropriate in the analysis, such as in this study. It is more specialized than the Scheffe test, as it can only be used in the case of equal sample sizes (in this case, n=10). The Scheffe test can be used when the sample size differ between sites. Because the Tukey test uses a common sample size, the critical difference (HSD) which it calculates is a smaller value than the critical difference calculated by the Scheffe test. Therefore it is more precise in indicating where significant differences lie (Sokal and Rohlf, 1973).

The results of these tests are given in Tables 6 and 7. In these tables the sampling sites

are listed in order of increasing sample mean, and a bar is drawn over sites which are deemed to have similar means. For example, in 1984, the populations of *Promoresia tardella* are similar between Site 3 (mean = 13.40) and Site 8 (mean = 10.90) but numbers at these sites are significantly different from those of the other sites. Conversely, Sites 1, 2, 4a, 5 and 6 did not differ significantly in the number of *P. tardella*, eventhough Site 1 had a mean of 1.70, Site 2 had a mean of 0.10 and the remaining samples had no *P. tardella* at all. In this case, as in most cases, the Scheffe and Tukey tests yielded similar results, however in 30% of the cases, the Tukey test resulted in different groupings (e.g., *Hydropsyche betteni* and *Chimarra aterrima* in 1984, *Hydropsyche slossonae*, *Leptophlebia cupida* and *Paraleptophlebia adoptiva* in 1985).

Taxa which showed one site with a particularly higher mean number per sample in the multiple comparison tests were, for 1984; *Baetis tricaudatus* at Site 1, *Paraleptophlebia adoptiva* and *Micrasema wataga* at Site 3, Lumbricids (earthworms) at Site 4a, *Arctopsyche ladogensis* at Site 5, Ceratopogonidae, *Paracapnia opis* and *Leuctra ferruginea*, all at Site 8. For 1985, taxa which showed a significantly higher mean per sample were; *Ephemerella subvaria* and *Isoperla transmarina* at Site 1, *Oligostomis* sp. at Site 2, *Micrasema wataga* at Site 3, *Tipula* sp. and Lumbricidae at Site 4a, *Hydropsyche betteni* and *Hydatophylax argus* at Site 4b. Site 5 supported a significantly large population of *Rhyacophila fuscula*, while *Leuctra ferruginea* and *Dolophiloides distinctus* showed high populations at Site 8.

Other taxa were found in consistantly higher numbers at Sites 3, 8 and to a lesser extent in the second year of the program, Site 4b. These include *Promoresia tardella*, Simuliidae (Diptera), *Lepidostoma* sp., *Chimarra aterrima* and other Philopotamidae, and *Isogenus frontalis*.

Some taxa exhibited a very gradual change in mean number per sample, over the entire study area. For example, *Hydropsyche sparna* collected in 1985 showed this trend, as did *H. slossonae* and *Baetis tricaudatus*, 1985. In 1984, the following taxa showed this trend; *Rhyacophila fuscula*, *Isoperla transmarina*, *Hydropsyche slossonae* and *H. sparna*.

Table 6. Summary of ANOVA Multiple Comparison Tests for Data from First Year of Study.

Taxon	Scheffe Test Result[+] (α=0.05)	Tukey Test Result[+] (α=0.05)
Baetis tricaudatus	4a 6 8 2 3 5 1	4a 6 8 2 3 5 1
Rhyacophila fuscula	2 4a 6 8 1 3 5	2 4a 6 8 1 3 5
Rhyacophila carolina	1 2 4a 5 6 3 8	1 2 4a 5 6 3 8
Hydropsyche slossonae	8 4a 3 6 5 2 1	8 4a 3 6 5 2 1
Hydropsyche sparna	4a 3 8 1 2 6 5	4a 3 8 1 2 6 5
Hydropsyche betteni	3 4a 8 1 5 2 6	3 4a 8 1 5 2 6
Arctopsyche ladogensis	1 2 4a 6 3 8 5	1 2 4a 6 3 8 5
Ephemerella subvaria	4a 8 3 6 2 1 5	4a 8 3 6 2 1 5
Isoperla transmarina	4a 5 2 6 8 3 1	4a 5 2 6 8 3 1
Leuctra ferruginea	2 4a 5 6 1 3 8	2 4a 5 6 1 3 8
Paracapnia opis	2 4a 5 6 1 3 8	2 4a 5 6 1 3 8
Promoresia sp.	4a 5 6 2 1 8 3	4a 5 6 2 1 8 3
Paraleptophlebia adoptiva	2 4a 5 6 1 8 3	2 4a 5 6 1 8 3
Habrophlebia vibrans	1 2 4a 5 6 3 8	1 2 4a 5 6 3 8
Micrasema sp.	1 2 4a 5 6 8 3	1 2 4a 5 6 8 3
Tipula sp.	2 1 5 8 6 3 4a	2 1 5 8 6 3 4a

[+] - Sample sites are listed in order of increasing sample means. (continued)

Summary of ANOVA Multiple Comparison Tests on Data from First Year of Study (cont'd).

Taxon	Scheffe Test Result[+] (α=0.05)	Tukey Test Result[+] (α=0.05)
Chironomidae (Number collected)	6 4a 8 5 3 2 1	6 4a 8 5 3 2 1
Chironomidae (Abundance Indices)	4a 6 8 5 3 2 1	4a 6 8 5 3 2 1
Ceratopogonidae	1 2 3 4a 6 5 8	1 2 3 4a 6 5 8
Chimarra sp.	1 2 4a 5 6 8 3	1 2 4a 5 6 8 3
Nematoda	1 2 4a 5 6 8 3	1 2 4a 5 6 8 3
Earthworm	6 8 3 1 2 5 4a	6 8 3 1 2 5 4a
Lumbriculus variegatus	6 4a 3 1 2 5 8	6 4a 3 1 2 5 8
Enchytraeidae	6 1 3 2 4a 5 8	6 1 3 2 4a 5 8
Naididae	4a 3 1 6 2 8 5	4a 3 1 6 2 8 5
Acarida	2 4a 5 6 8 3 1	2 4a 5 6 8 3 1

[+] - Sample sites are listed in order of increasing sample means.

Table 7. Summary of ANOVA Multiple Comparison Tests on Data from Second Year of Study.

Taxon	Scheffe Test Result[+] (α=0.05)	Tukey Test Result[+] (α=0.05)
Baetis tricaudatus	7 4a 8 4b 3 5 6 1 2	7 4a 8 4b 3 5 6 1 2
Rhyacophila fuscula	2 4a 7 4b 6 3 8 1 5	2 4a 7 4b 6 3 8 1 5
Hydropsyche slossonae	4a 8 6 4b 3 7 1 2 5	4a 8 6 4b 3 7 1 2 5
Hydropsyche sparna	4a 8 7 3 5 1 4b 2 6	4a 8 7 3 5 1 4b 2 6
Hydropsyche betteni	1 5 7 8 4a 2 6 3 4b	1 5 7 8 4a 2 6 3 4b
Ephemerella subvaria	4a 4b 8 3 6 5 7 2 1	4a 4b 8 3 6 5 7 2 1
Isoperla transmarina	4a 6 5 2 7 4b 8 3 1	4a 6 5 2 7 4b 8 3 1
Leuctra ferruginea	1 2 4a 5 6 7 4b 3 8	1 2 4a 5 6 7 4b 3 8
Promoresia tardella	2 5 7 6 4b 4a 1 3 8	2 5 7 6 4b 4a 1 3 8
Paraleptophlebia adoptiva	4a 5 6 7 4b 8 2 1 3	4a 5 6 7 4b 8 2 1 3
Leptophlebia cupida	3 1 6 5 8 4a 7 4b 2	3 1 6 5 8 4a 7 4b 2
Micrasema sp.	2 4a 5 6 7 4b 1 8 3	2 4a 5 6 7 4b 1 8 3
Tipula sp.	2 6 4b 1 5 3 7 8 4a	2 6 4b 1 5 3 7 8 4a
Chironomidae (Number collected)	6 8 3 4a 4b 7 5 1 2	6 8 3 4a 4b 7 5 1 2
Chironomidae (Abundance indices)	6 8 3 4a 4b 5 7 1 2	6 8 3 4a 4b 5 7 1 2

[+] - Sample sites are listed in order of increasing sample mean. (continued)

Summary of ANOVA Multiple Comparison Tests on Data from Second Year of Study (Cont'd).

Taxon	Scheffe Test Result[+] (α=0.05)	Tukey Test Result[+] (α=0.05)
Simuliidae	1 4a 5 6 7 4b 2 8 3	1 4a 5 6 7 4b 2 8 3
Oligostomis sp.	1 4a 5 6 7 8 4b 3 2	1 4a 5 6 7 8 4b 3 2
Hydatophylax argus	1 3 5 6 8 7 2 4a 4b	1 3 5 6 8 7 2 4a 4b
Gastropoda[*]	1 3 6 7 8 4b 4a 5 2	1 3 6 7 8 4b 4a 5 2
Lepidostoma sp.	1 2 4a 5 6 7 8 4b 3	1 2 4a 5 6 7 8 4b 3
Dolophiloides distinctus[*]	1 2 3 4a 5 6 7 4b 8	1 2 3 4a 5 6 7 4b 8

[+] - Sample sites are listed in order of increasing sample means.

[*] - Taxon present with mean <1 per sample.

The relative abundance of a taxon at the various sites, not just taxa presence or absence, is also an important characteristic, in the present study. As species usually have wide ranging environmental tolerances, they could therefore be expected to be found over a wide gradient of conditions, in varying numbers. This was true for *Ephemerella subvaria*, *Baetis tricaudatus*, *Hydropsyche sparna* and *H. slossonae* for both years, and *Leptophlebia cupida* and *Isoperla transmarina* in 1985. *Promoresia tardella* also showed this pattern, but to a lesser extent.

Samples from Sites 3 and 8, the two most upstream sites on South Brook, characteristically contained a high number of taxa ($\bar{x}$=14.3, $\bar{x}$=16.6 for Sites 3 and 8 respectively, in 1984; $\bar{x}$=15.6, $\bar{x}$=14.4, in 1985), with low numbers of individuals per taxon. Most of these taxa were found only at one or both of these sites, or else in much lower numbers elsewhere (with the exception of *Isoperla transmarina*). Taxa which are characteristic of Site 3 and/or 8 include *Leuctra ferruginea*, *Paracapnia opis*, *Promoresia tardella*, *Rhyacophila carolina*, *Baetis pygmaeus*, *Micrasema wataga* and *Chimarra aterrima* in both 1984 and 1985. Ceratopogonidae, *Paraleptophlebia adoptiva* and *Habrophlebia vibrans* were characteristically high at these two sites in 1984, while *Lepidostoma* sp. and Simuliidae were in relatively high abundance in 1985. From the results of water quality sampling, these two sites can be categorized as "clean" water sites.

In 1984, samples from Sites 2 and 6 contained mainly a high number of *Hydropsyche* spp. and low numbers of *Baetis tricaudatus* and *Ephemerella subvaria*. In 1985, there were again high numbers of *Hydropsyche* spp., but the number of *B. tricaudatus* and *E. subvaria* were higher, and *Leptophlebia cupida* was also found in high numbers at Site 2 (This may be due to the shorter colonization period and/or the positioning of the artificial substrates in the stream at Site 2, as discussed later).

In summary, there appears to be a gradient of taxon number from many taxa at some sites (e.g., 3, 4b and 8) to a few taxa at other sites (e.g., 4a and 6). Sites 3 and 8 were the cleanest, according to results of the surface water quality study, while Sites 6 and especially 4a were considered to have poorer water quality. The "cleaner" sites tended to have the greatest diversity.

ANOVA is based on the assumptions that the populations are normally distributed, and that they have equal variances. It also assumes that the data has at least interval scale measurement, and as such it cannot be applied to the chironomid abundance indices. Another type of analysis of variance is the Kruskal-Wallis ANOVA. It is a non-parametric (or distribution-free) analysis of variance test, similar to ANOVA, however it makes no assumptions about the population distributions, nor does it require interval-scale measurement. All taxa examined by ANOVA, as well as the chironomid abundance indices, were also checked using a Kruskal-Wallis ANOVA. Scheffe-type and Tukey-type multiple comparison tests were applied to the results of the Kruskal-Wallis ANOVA of the chironomid indices. The results of these tests are given in Tables 8 and 9. In most cases, the inferences which can be drawn from the results of the Kruskal-Wallis ANOVA are similar to those of the parametric ANOVA.

5.1.3 Between-year Variation

To examine between-year site-specific differences, taxa which yielded a significant ANOVA result were examined using a paired comparison t-test. This test has similar assumptions as ANOVA, consequently a Wilcoxon matched-pairs signed-ranks test was also applied to the between-year data. The Wilcoxon test, a non-parametric test is similar to the t-test, and is almost equally as powerful without requiring the assumptions about the underlying distributions (Sokal and Rolhf, 1973).

The results of these tests are found in Table 10. In most cases (68.5%) there was no significant difference between the two years (i.e., $p>0.05$). However, 23 cases showed a significant difference between the first and second year, 91.3% (21 of 23) being a result of much higher numbers in 1985. Most of these differences are in the numbers of *Hydropsyche slossonae* and *Baetis tricaudatus*.

The large increase in number is probably a reflection of the shorter colonization period in the second year. The shorter exposure probably resulted in a reduction in the amount of silt and sediment in the artificial substrate bags. This could affect the composition of the collected fauna in two ways; firstly, there was probably more intersticial space available to the colonizing insects. Secondly, as discussed earlier, increased siltation

Table 8. Results of Kruskal-Wallis ANOVA on Significant Data from First Year of Study.

Taxon	H	p-value	Adjusted H	p-value
Nematoda	8.56	0.1999	21.44	0.0015
Earthworm	28.39	0.0000	41.94	0.0000
Lumbriculus variegatus	14.18	0.0277	15.76	0.0151
Enchytraeidae	9.04	o.1714	12.65	0.0490
Naididae	10.31	0.1123	16.24	0.0125
Acarida	7.04	0.3168	17.68	0.0071
Promoresia sp.	39.37	0.0000	53.66	0.0000
Ceratopogonidae	16.77	0.0102	38.99	0.0000
Chironomidae (Number collected)	26.82	0.0002	26.88	0.0002
Tipula sp.	15.19	0.0188	18.87	0.0044
Baetis tricaudatus	29.66	0.0000	32.75	0.0000
Ephemerella subvaria	45.12	0.0000	51.09	0.0000
Habrophlebia vibrans	9.23	0.1611	30.24	0.0000
Paraleptophlebia adoptiva	23.79	0.0006	38.92	0.0000
Leuctra ferruginea	25.41	0.0003	55.24	0.0000
Isoperla transmarina	30.53	0.0000	41.73	0.0000
Micrasema sp.	21.72	0.0014	47.25	0.0000
Arctopsyche ladogensis	18.58	0.0049	36.15	0.0000
Hydropsyche betteni	23.64	0.0006	36.08	0.0000
Hydropsyche slossonae	43.33	0.0000	47.54	0.0000
Hydropsyche sparna	35.24	0.0000	35.40	0.0000
Chimarra sp.	14.15	0.0280	32.82	0.0000
Rhyacophila carolina	6.31	0.3898	26.75	0.0002
Rhyacophila fuscula	18.28	0.0056	28.18	0.0001

Note: Kruskal-Wallis H (adjusted for ties) shows all above taxa significant at α=0.05.

Table 9. Results of Kruskal-Wallis ANOVA on Significant Data from Second Year of Study.

Taxon	H	p-value	Adjusted H	p-value
Baetis tricaudatus	45.04	0.0000	45.27	0.0000
Baetis pygmaeus *	13.05	0.1101	40.36	0.0000
Rhyacophila fuscula	34.52	0.0000	43.09	0.0000
Hydropsyche slossonae	62.25	0.0000	62.51	0.0000
Hydropsyche sparna	46.42	0.0000	46.49	0.0000
Hydropsyche betteni	35.09	0.0000	41.98	0.0000
Ephemerella subvaria	63.91	0.0000	66.06	0.0000
Isoperla transmarina	47.70	0.0000	52.86	0.0000
Isogenus frontalis *	11.08	0.1975	26.37	0.0009
Promoresia tardella	47.00	0.0000	68.41	0.0000
Paraleptophlebia adoptiva	15.40	0.0518	33.08	0.0000
Leptophlebia cupida	53.76	0.0000	62.29	0.0000
Micrasema sp.	17.94	0.0217	51.40	0.0000
Tipula sp.	35.42	0.0000	46.80	0.0000
Chironomidae (Number collected)	43.54	0.0000	43.57	0.0000
Chironomidae (Abundance indices)	25.69	0.0012	32.41	0.0001
Simuliidae	33.10	0.0001	62.53	0.0000
Oligostomis sp.	9.50	0.3220	44.04	0.0000
Chimarra sp. *	5.73	0.6777	19.26	0.0135
Lepidostoma sp.	4.22	0.8368	33.10	0.0001
Hydatophylax argus	13.69	0.0902	36.77	0.0000
Wormaldia moesta *	15.46	0.0508	47.83	0.0000

* - Taxon present with mean <1 per sample.

Note: Kruskal-Wallis H (adjusted for ties) shows all above taxa significant at $\alpha=0.05$.

Table 10. Comparison of Significant Taxa at all Sampling Sites.

Taxon	T-Test Result (p-value)	Wilcoxon Result (p-value)
Site 1		
Promoresia tardella	0.103	0.091
Chironomidae	1.000	1.000
Tipula sp.	0.168	0.180
Baetis pygmaeus	0.703	0.500
Baetis tricaudatus	0.951	0.799
Ephemerella subvaria	0.154	0.241
Paraleptophlebia adoptiva	0.162	0.225
*Isoperla transmarina**	0.032	0.005
*Hydropsyche slossonae**	0.035	0.047
Hydropsyche sparna	0.537	0.541
Rhyacophila fuscula	0.107	0.116
Site 2		
Chironomidae*	0.003	0.012
*Baetis tricaudatus**	0.028	0.017
*Ephemerella subvaria**	0.009	0.015
*Leptophlebia cupida**	0.013	0.005
Isoperla transmarina	0.084	0.080
Hydropsyche betteni	0.828	0.944
*Hydropsyche slossonae**	0.005	0.012
Hydropsyche sparna	0.295	0.285
Site 3		
*Promoresia tardella**	0.020	0.025
Chironomidae	0.443	0.463
Tipula sp.	0.185	0.173
Baetis pygmaeus	0.461	0.953
*Baetis tricaudatus**	0.003	0.008
Ephemerella subvaria	0.051	0.058
Paraleptophlebia adoptiva	0.436	0.236

continued

Site 3 (cont'd)		
Leuctra ferruginea	0.373	0.361
Isoperla transmarina	0.467	0.314
Micrasema sp.	0.088	0.0745
Hydropsyche betteni	0.059	0.028
*Hydropsyche slossonae**	0.005	0.007
Hydropsyche sparna	0.157	0.139
Chimarra aterrima	0.121	0.139
Rhyacophila fuscula	0.656	0.673
Isogenus frontalis	0.182	0.144
Site 4a		
Chironomidae*	0.001	0.012
Tipula sp.	0.252	0.185
Hydropsyche slossonae	0.168	0.138
Hydropsyche sparna	0.299	0.273
Site 5		
Chironomidae	0.053	0.063
Tipula sp.	0.343	0.345
*Baetis tricaudatus**	0.010	0.017
*Ephemerella subvaria**	0.002	0.009
Leptophlebia cupida	0.104	0.109
Hydropsyche betteni	0.111	0.106
*Hydropsyche slossonae**	0.000	0.005
Hydropsyche sparna	0.059	0.103
*Rhyacophila fuscula**	0.005	0.012
Site 6		
Chironomidae	0.104	0.142
*Baetis tricaudatus**	0.007	0.008
Ephemerella subvaria	0.088	0.076
Leptophlebia cupida	0.153	0.138
Hydropsyche betteni	0.773	0.541
*Hydropsyche slossonae**	0.007	0.009
*Hydropsyche sparna**	0.030	0.037

continued

Site 8		
Promoresia tardella	0.266	0.241
Chironomidae	0.081	0.109
Simuliidae*	0.007	0.008
Tipula sp.	0.758	0.753
Baetis pygmaeus	0.811	0.753
*Baetis tricaudatus**	0.012	0.005
Ephemerella subvaria	0.081	0.109
Leptophlebia cupida	0.560	0.686
Paraleptophlebia adoptiva	0.177	0.205
Leuctra ferruginea	0.725	0.767
Isoperla transmarina	0.074	0.063
Micrasema sp.	0.217	0.249
Hydropsyche betteni	0.096	0.109
Hydropsyche slossonae	0.153	0.109
Hydropsyche sparna	0.749	0.636
Chimarra aterrima	0.235	0.398
Wormaldia moesta	0.211	0.0277
*Rhyacophila fuscula**	0.042	0.043

* - Indicates taxa significant at α=0.05.

causes changes in microhabitat. As a result, the rock bags from 1984 were probably less suitable for the small Hydropsychidae and Baetidae, which were responsible for most of the between-year differences.

5.2 Biomass

The values of mean biomass are given in Tables 11 and 12 and are illustrated in Figures 13 and 14. Total biomass and biomass without large individuals (i.e., *Tipula* sp., *Aeshna* sp., large Lumbricids, etc.) are also given.

In 1984, there was no significant difference in mean total biomass per sample between any of the sites (Table 14). Intra-site variation was high (as evidenced by the relatively high standard deviations associated with the the values of the means given in Tables 11 and 12), which probably accounts for the inability of ANOVA to indicate inter-site differences. As discussed earlier, the placement of the rock bags in the rivers in 1984, or the changes in microhabitat due to siltation may explain a great deal of this variation.

In 1985, the only site with a detectable difference in total biomass was Site 4a (Table 14), which had relatively high populations of *Tipula* sp. and Lumbricidae (Tables A11 and A13), both of which are taxa whose individuals are disproportionately heavy.

Once the large individuals were removed, statistically significant differences in mean biomass were detected in the data for both years. In 1984, Site 1 was determined to be different from Site 4a, and Site 5 was declared different from Sites 3, 4a and 8 (Table 14). In 1985, Site 1 was declared significantly different from Sites 3, 4a and 8, while Site 2 was determined to be different from all other sites, with the exception of Site 1. The differences in the values of biomass could be a result of a number of factors influencing the water quality of the system (e.g., drainage from the sheep pasture, testing of insecticides, or other agricultural practices on the CDA farm, next to Site 4a, upstream pollution sources on the Waterford River, etc.).

5.3 Number of Taxa

The mean number of taxa ($\bar{x}$) at each site is given in Tables 11 and 12 and shown in Figures 13 and 14. The results obtained in 1984 and 1985 are presented below.

In 1984, the samples from Sites 3 and 8 ($\bar{x}$=14.3 and 16.6 respectively) possessed significantly higher mean number of taxa than samples from Sites 2, 4a and 6 ($\bar{x}$=8.2, 4.1, and 7.5) (Table 11). Also, Sites 1 and 5 ($\bar{x}$=11.0 and 11.9) were determined to be different from Site 4a.

In 1985, Site 3 ($\bar{x}$=15.6) was determined to be significantly different from Sites 4a, 4b, 5, 6 and 7 ($\bar{x}$=6.9, 9.7, 8.9, 6.6 and 9.4) (Table 12). Site 8 ($\bar{x}$=14.4) was determined to be different from Sites 4a and 6).

As noted earlier, the "clean" water sites tended to have more taxa (i.e., greater diversity) than the "unclean" sites. The decreased diversity, generally observed at sites located in the urbanized areas of the drainage basin, may be due to the effects of any or all of the urban-related factors discussed in Section 1.

5.4 Number of Individuals

The mean number of individuals per sample at each site is given in Tables 11 and 12 and also shown in Figures 13 and 14. Most sites showed a large (almost two-fold) increase in the number of individuals in 1985, with the exception of Site 8, which stayed almost exactly the same ($\bar{x}$=74.1 individuals/sample in 1984, and 75.2 individuals/sample in 1985). Most sites showed almost a doubling in mean number of individuals per sample (Tables 11 and 12).

In 1984, Site 1 contained the most individuals per sample ($\bar{x}$=144/sample), while in 1985, Site 2 supported the most ($\bar{x}$=268.5/sample). Site 1, in 1985 yielded an average of 258.5/sample. The lowest mean number of individuals was seen at Site 4a, in both 1984 and 1985, although the mean in 1985 was more than double the 1984 mean ($\bar{x}$=51.6 in 1985 vs 23.9 in 1984) (Tables 11 and 12).

Table 11. Biomass, number of individuals and number of taxa collected at each site in first year of study. (Mean, Std. Dev. and Range).

Site	Total Biomass (g)	Biomass without large individuals (g)	Total number of individuals	Number of taxa
	1.157	1.005	144	11.00
1	0.775	0.746	113	1.83
	0.26-2.65	0.26-2.65	25-363	8-14
	0.771	0.771	54.5	8.20
2	0.705	0.705	46.7	3.97
	0.04-1.87	0.04-1.87	7-140	2-15
	0.858	0.280	74.6	14.30
3	0.927	0.224	37.0	5.62
	0.03-3.12	0.03-0.61	15-136	5-24
	1.180	0.317	23.9	4.10
4a	1.21	0.269	13.9	4.10
	0.08-3.68	0.04-0.87	10-52	3-6
	1.466	1.185	115.6	11.90
5	0.779	0.551	54.6	2.08
	0.48-3.19	0.48-1.93	39-207	8-14
	1.037	0.583	40.7	7.50
6	0.576	0.469	31.5	2.32
	0.15-1.78	0.09-1.59	5-96	4-12
	0.860	0.276	74.1	16.60
8	0.956	0.200	34.9	4.67
	0.06-3.16	0.01-0.57	11-133	8-24

Table 12. Biomass, number of individuals and number of taxa collected at each site in second year of study. (Mean, Std. Dev. and Range).

Site	Total Biomass (g)	Biomass without large individuals (g)	Total number of individuals	Number of taxa
	1.269	1.053	258.5	10.6
1	0.956	0.639	117.565	3.836
	0.36-3.58	0.36-2.37	91-450	6-20
	1.320	1.260	268.5	10.7
2	.800	.738	139.656	3.529
	0.120-2.420	0.120-2.420	51-243	6-18
	0.428	0.225	128.0	15.6
3	0.359	0.195	89.727	4.575
	0.060-1.150	0.060-0.710	15-304	6-21
	3.842	0.089	51.6	6.9
4a	3.054	0.082	26.925	2.923
	0.040-9.920	0.020-0.220	17-98	2-11
	0.517	0.425	124.1	9.7
4b	0.295	0.259	54.643	2.163
	0.160-.930	0.160-0.930	74-242	6-13
	10.45	0.564	208.5	8.9
5	0.967	0.290	94.952	0.876
	0.110-3.260	0.110-1.060	64-361	8-10
	0.578	0.564	102.9	6.6
6	*0.366*	*0.381*	*39.088*	*1.350*
	0.140-1.310	0.050-1.310	30-169	5-9
	0.895	0.427	141.2	9.4
7	0.613	0.113	37.967	2.675
	0.330-2.140	0.280-0.640	88-222	6-14
	1.106	0.104	75.2	14.4
8	0.829	0.079	29.047	3.471
	0.070-2.930	0.020-0.250	33-114	10-20

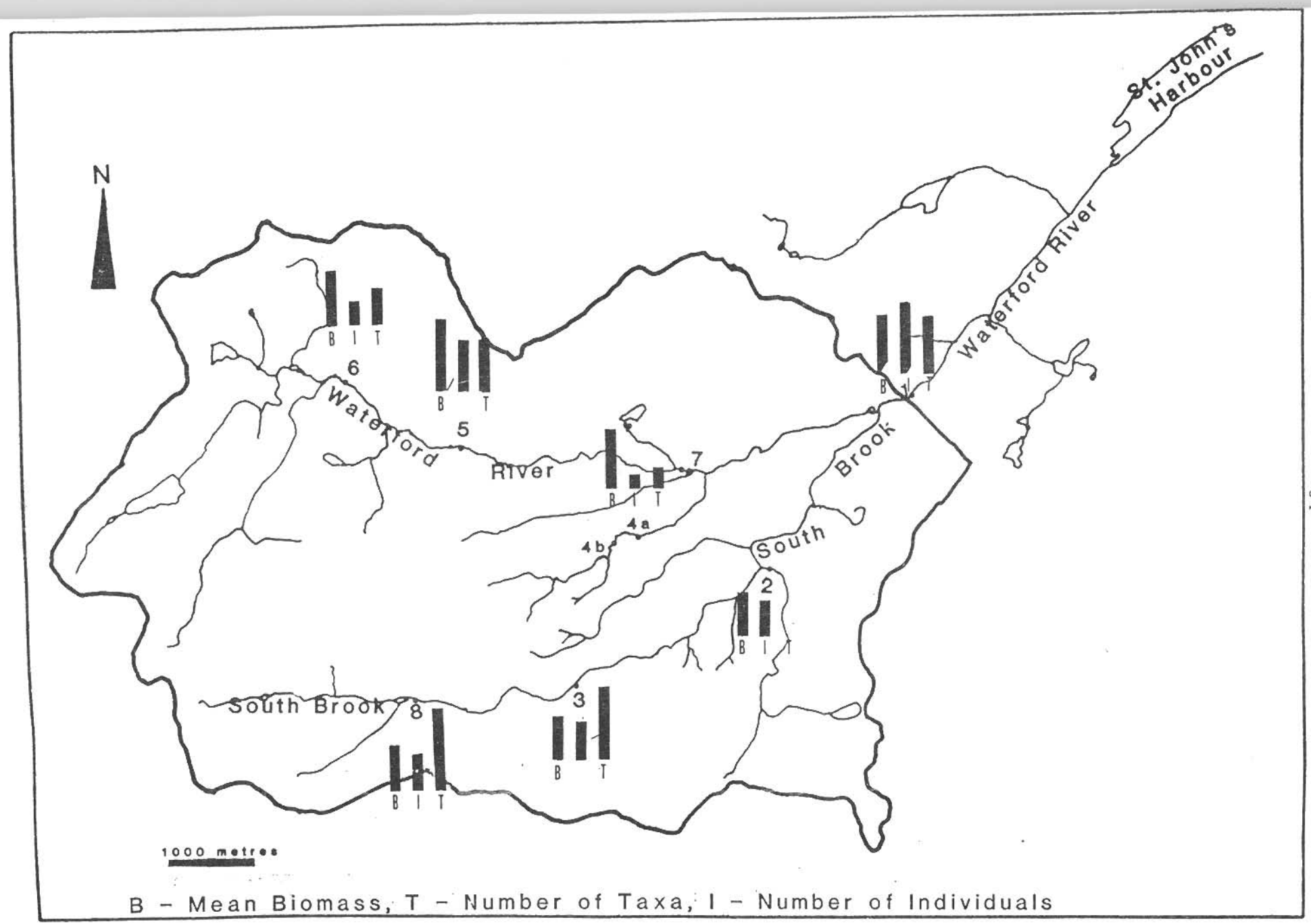


Figure 13. Relative distribution of biomass, number of individuals and number of taxa, 1984. 1 cm represents 1 g biomass, 100 individuals and 10 taxa.

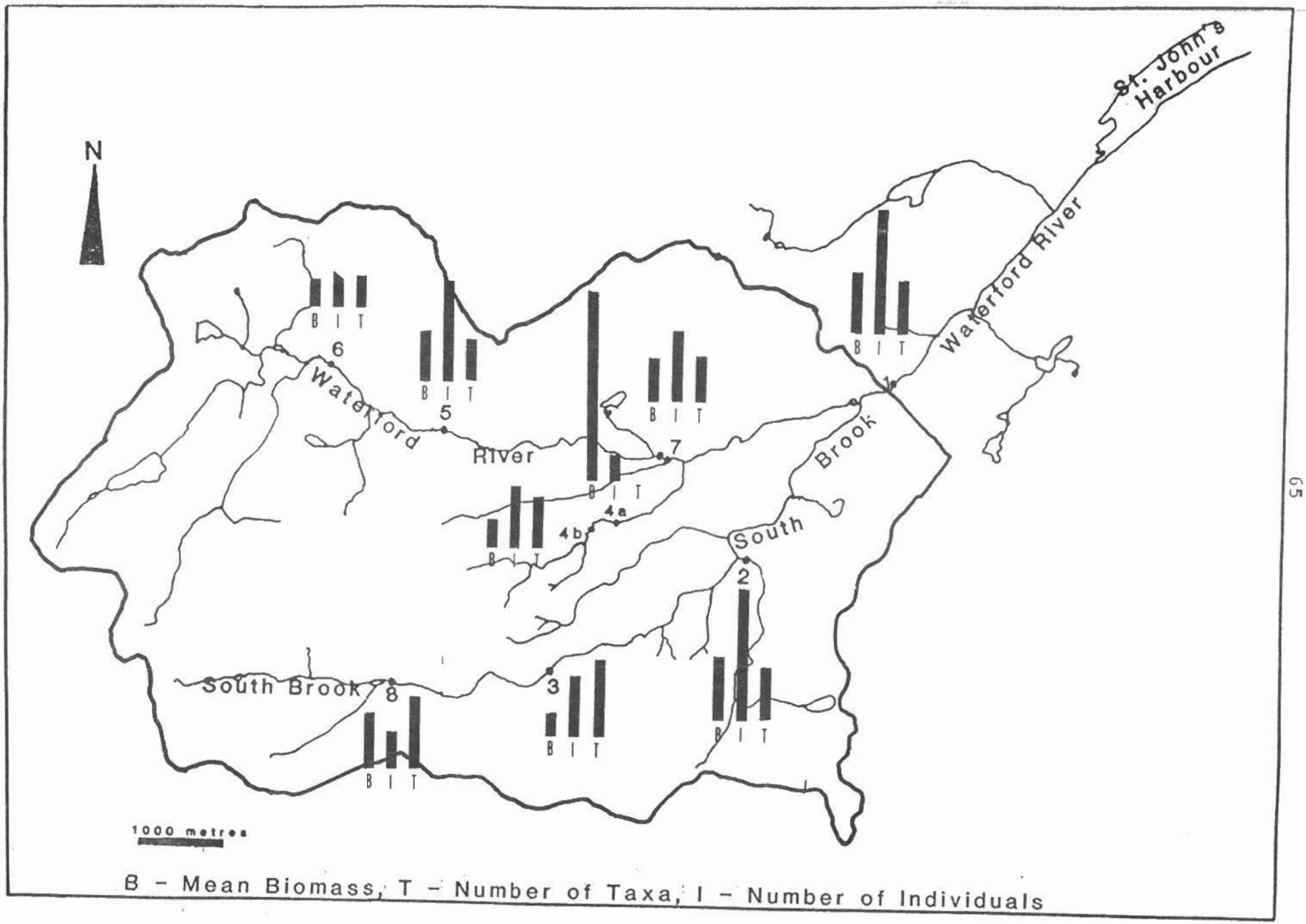


Figure 14. Relative distribution of biomass, number of individuals and number of taxa, 1985. 1 cm represents 1 g biomass, 100 individuals and 10 taxa.

The number of individuals collected at a site did not appear to be directly related to the biomass. For example, both Sites 3 and 8 had low sample biomasses yet the mean number of individuals per sample was fairly high. Sites 1 and 5 had both high biomass and high number of individuals per sample but while Site 5 had the highest biomass the number of individuals was less than at Site 1. Also notable is the fact that while the mean number of individuals at most sites showed a large increase in 1985, the mean biomass was much lower at most sites in 1985, than in 1984. Again, the effects of siltation may account for this difference, as most of the additional individuals in 1985 were very small hydropsychid caddisflies, baetid mayflies and chironomids. The bags from the 1984 study were heavily silted, therefore these small individuals were present in reduced numbers.

At Site 1, in 1984, chironomids made up 41.8% of the individuals while *Hydropsyche* spp. made up 25.7%. *Baetis tricaudatus* and *Ephemerella subvaria* both made up 12.2% while the annelids made up 2.6% and the stoneflies made up 2.5% of the individuals. In 1985, chironomids made up 37.4%, *Hydropsyche* spp. comprised 30.8%. *B. tricaudatus* made up only 6.6%, with *E. subvaria* staying about the same, at 12.6%. *Isoperla transmarina* accounted for 10.1% in 1985. Site 1 had an average of 144 individuals/sample in 1984, 258.5 in 1985.

In 1984, Site 2 had an average 54.5 individuals per sample. In 1985, this had increased to 268.5. Chironomids made up 32.1% of the individuals while *Hydropsyche* spp. made up 47.9% and the annelids, 11.4%, in the first year of the study. In 1985, chironomids were at 36.4%, *Hydropsyche* spp. 36.8%. Annelids accounted for less than 1% of the individuals in 1985.

Site 3 had an average of 74.6 individuals per sample with chironomids accounting for 32.7% of the individuals, in 1984. *Promoresia tardella* was the next most common taxon accounting for 18.0% of the individuals. In the second year of the study, Site 3 yielded a mean of 128.0 individuals per sample. Chironomids accounted for only 22.7%, *P. tardella* only 3.8%. *Hydropsyche* spp. (largely *H. slossonae*) accounted for 42.3% of the individuals in samples from Site 3.

Site 4a, as previously stated, had the lowest mean number of individuals per sample, during both years of the study (23.9 in 1984 and 51.6 in 1985). Terrestrial earthworms (lumbricids) accounted for 49.0% of the individuals while chironomids accounted for 31.4% and tipulids 7.1% of the individuals in the first year. In 1985, lumbricids and *Tipula* sp. were equal, at 5.4% each. Chironomids constituted 71.5% of the number of individuals. *Leptophlebia cupida*, which was not found at this site in 1984, comprised 4.7% of the individuals.

Site 4b was only sampled in 1985, and it showed a mean of 124.1 individuals per sample. Chironomids comprised 30.9%, *Hydropsyche* spp. contributed 48.4%. Other common taxa were *Leptophlebia cupida* at 9.4% and *B. tricaudatus* at 6.6%. This site showed a much more diverse fauna than that which was present at Site 4a. There were a number of cleaner water organisms present (e.g., *L. cupida*). This indicates that the water quality is better upstream of the pasture, and that the impoundment of water in the reservoir is probably blocking the downstream drift of invertebrates.

Site 5 had the second highest counts of organisms, in the first year of the study, with a mean of 115.6 individuals per sample. Chironomids accounted for 19.2% of the individuals while the hydropsychid caddisflies, including *Arctopsyche ladogensis*, contributed 34.5% of individuals. *A. ladogensis* on its own comprised only 2.3%. *Ephemerella subvaria* accounted for 32.2% of the individuals while the small *Tipula* sp. population accounted for 0.52%. *Baetis tricaudatus* accounted for 2.5%. In 1985, chironomids accounted for 30.4% of the mean of 208.5 individuals per sample. Hydropsychidae constituted 52.6%, *A. ladogensis*, less than 1%. *E. subvaria* comprised only 3.6%, while *B. tricaudatus* made up 6.3%.

In 1984, Site 6 yielded 40.7 individuals per sample of which hydropsychid caddisflies accounted for 70.8% while chironomids accounted for 13.5%. Tipulids accounted for 3.4% of the individuals. In 1985, Site 6 had an average of 102.9 individuals per sample. *Hydropsyche* spp. accounted for 64.3%, and chironomids had increased to 16.3%. As mentioned earlier, *B. tricaudatus* increased at this site in 1985, and accounted for 13.3% of the individuals.

Site 7, is another site that was sampled only in 1985, and it yielded a mean of 141.2 individuals per sample. *Hydropsyche* spp. accounted for 46.5%, chironomids made up 37.5%. Of the remaining taxa, the most common were *E. subvaria* at 6.6% and the annelid *Nais communis* (Oligochaeta: Naididae) at 3.7%.

Site 8, with an average of 74.1 individuals per sample, in 1984, had a chironomid population which accounted for 14.4% of the individuals while *Promoresia tardella* accounted for 14.7%. Stoneflies (*Leuctra ferrugina*, *Paracapnia opis*, *Isogenus frontalis* and *Isoperla transmarina*) made up 19.2% *Hydropsyche sparna* made up 11.0% and tipulids comprised 1.3% of the individuals in the site's fauna, in the first year of the study. In 1985, chironomids accounted for 22.7% of the mean 75.2 individuals per sample. *P. tardella* made up 9.4%, while the stoneflies comprised 20.1%. *Hydropsyche* spp. accounted for 12.1%, and *Tipula* sp. constituted 1.4% of the individuals present. Simuliidae showed an increase in the second year of the study, comprising 5.8%.

As can been seen in Tables 11 and 12 and from the above discussion, the most densely populated sites were those with only fair to poor water quality. The sites with the lower number of individuals were the cleanest sites (3 and 8) and the worst water quality site (4a). This may be due to the presence of generalist taxa (i.e., taxa which are not specialized in environmental requirements), which can move into areas of stressed water and either displace more sensitive taxa, or simply replace those lost due to the increased pollution. Some taxa will actually increase their numbers in the presence of increased organic pollution. This has been observed by Hynes (1960) and by Whiting and Clifford (1983).

The Scheffe's test results (Table 14) showed a continuous gradient in mean number of individuals per sample between sites in both 1984 and 1985. In 1984, Sites 1 and 5 were determined to be significantly different from Site 4a. In 1985, Sites 1 and 2 were determined to be significantly different from Sites 4a, 6 and 8, while Site 5 was detected as being different from Site 4a only.

5.5 Benthic Community Structure

Community structure was examined using two common methods; diversity indices and a cluster analysis.

5.5.1 Diversity Indices

The diversity of the benthic fauna was examined at each site, by calculating Shannon-Weaver diversity indices (H) for each sample. This index is a measure of the relative diversity of the community, and is calculated by the formula;

$$H = -[\sum_{i=0}^{s} p_i \ln p_i]$$

$$\text{where, } p_i = \frac{\text{number of individuals in taxon}_i}{\text{number of individuals in all taxa}}$$

and $\ln p_i$ is the natural log of p_i

The values of H are given in Table 13. Site 4a exhibited the lowest diversity in both years ($\overline{H}$ =1.07 and 0.98 in the first and second years respectively). Site 8 showed the highest diversity ($\overline{H}$ =2.32 and 2.15). Because the Shannon-Weaver diversity index takes both species (or taxa) number and number of individuals into account, it should not be taken as an index of abundance. As is shown by the results of the Scheffe test (Table 14), the significance pattern of the diversity indices is more similar to that of taxon number rather than the number of individuals.

Vannote, *et al.* (1980) stated that maximum diversity should occur in fourth order streams. Site 1, which was the only site at which the river was order four, had mean diversity indices ($\overline{H}$) lower than those of some of the upper stream sites (Table 13). Sites 3 and 8, in both 1984 and 1985, had higher $\overline{H}$ values than Site 1, eventhough they are both on a second order stream. Site 5 had $\overline{H}$ higher than that for Site 1, in 1984, but lower than Site 1 in 1985. Although Site 1 has a relatively high diversity, it should theoretically be the most diverse site.

Table 13. Shannon-Weaver diversity indices for both years of study. (Mean, Std. Dev. and Range)

site	First Year	Second Year
	1.762	1.586
1	0.237	0.293
	1.216-2.033	0.943-1.899
	1.435	1.589
2	0.572	0.146
	0.325-1.857	1.309-1.847
	1.938	2.089
3	0.475	0.247
	1.081-2.583	1.617-2.408
	1.065	0.981
4a	0.173	0.363
	0.847-1.296	0.406-1.459
	-	1.741
4b	-	0.237
	-	1.207-2.010
	1.847	1.385
5	0.170	0.156
	1.639-2.171	1.098-1.627
	1.435	1.354
6	0.254	0.250
	1.136-2.074	0.930-1.830
	-	1.459
7	-	0.214
	-	1.217-1.850
	2.324	2.150
8	0.208	0.166
	2.020-2.649	1.916-2.445

- Sample site not included in first year of study.

Table 14. Summary of Scheffe tests on diversity indices, biomass, and counts of taxa and individuals, in both years of study.

Source	First year result	Second year result
Diversity Indices	4a 2 6 1 5 3 8	4a 6 5 7 1 2 4b 3 8
Total Biomass	2 3 8 6 1 4a 5	3 4b 6 7 5 8 1 2 4a
Biomass without large individuals	4a 8 3 6 2 1 5	4a 8 3 4b 7 5 6 1 2
Total number of individuals	4a 6 2 8 3 5 1	4a 8 6 4b 3 7 5 1 2
Total number of taxa	4a 6 2 1 5 3 8	6 4a 5 7 4b 1 2 8 3

Note - Sites are listed in order of increasing sample mean.

An ANOVA was performed on the calculated H values, with a Scheffe multiple comparison test to show which sites had significantly different H values. This test showed that, in 1984 and 1985, Sites 3 and 8 had similar H values, while these two sites were declared significantly different from Sites 1, 2, 4a and 6. The addition of Sites 4b and 7, in 1985, yielded a result showing 4b similar to Sites 3 and 8, and Site 7 being similar to all sites except that with the most impoverished fauna (Site 4a) and those with the richest diversity (Sites 3 and 8) (Table 14).

5.5.2 Cluster Analysis

Community structure was also examined through a cluster analysis, using Clustan-2 (Wishart, 1978), a Fortran-coded program available on the VAX/VMS system at M.U.N. The results of a cluster analysis give an indication as to which samples (and therefore which sites) are similar to each other. By comparing these results with the water quality data, it is possible to outline communities of "clean" and "unclean" invertebrates. A cluster analysis groups the samples together, based on the number of taxa they have in common, independent of the site from which the samples were taken. The groups are combined together at a given levels of similarity. Samples are compared on the basis of shared attributes, in this case, presence or absence of a taxon, and are clustered together, based on a Jaccard Index of Similarity which is calculated as follows;

$$S_j = \frac{a}{a+b+c}$$

where S_j=Jaccard index of similarity,
a=number of taxa found in both samples,
b=number of taxa in sample 1 but not in sample 2, and
c=number of taxa in sample 2 but not in sample 1.

The results of these calculations are ordinated by group average clustering.

This type of cluster analysis is based solely on presence or absence of taxa (i.e., binary data). It does not account for the relative abundance of taxa. All taxa encountered (not just those with mean >1 per sample) were considered in the cluster analysis.

Dendrogram tables created by the cluster analyses on the data from both years are shown in Figures 15 and 16, and are discussed in the following sections. The numbers on the left side of the table are the values of the Jaccard index of similarity (S_j).

5.5.3 Cluster Analysis on 1984 Data

In the data from the first year of the sampling program, there were several clusters representing samples from the same site, and samples from different sites. For example, samples from Site 1 clustered together with S_j= 0.477, while five samples of Site 3 clustered with samples from Site 1 (S_j=0.401). The other five samples of Site 3 clustered with Site 8 samples (S_j=0.260) (at p = 0.05 (d.f. = 60), S_j=0.250). Site 1 samples and Site 8 samples did not significantly cluster together even though Site 3 samples clustered with both sites. This indicates that Site 3 is faunistically similar to Sites 1 and 8. Site 1 contained several species which were generally found at Sites 3 and 8 so it was expected that there would be similar grouping of samples. There were some taxa which were common to both Sites 1 and 3, while others were common to Sites 3 and 8. The fauna of Site 8 was probably influencing the fauna of Site 3 which would be expected as Site 3 was downstream of Site 8. Site 1, which was located on the Waterford River about one kilometre below the confluence of South Brook, had a fauna which was influenced by both rivers. It appeared however, that the greater influence was by South Brook. The cleaner, unstressed water from South Brook probably afforded some relief to the polluted water of the Waterford River. The existence of the Bowring Park duck pond, upstream of Site 1, also may have had a beneficial effect on the downstream course of the river. A great deal of the silt being carried by the river probably settled in the pond, and the falls both at the outlet, and especially upstream of the pond, would increase the dissolved oxygen content of the water flowing downstream. The fauna of Site 1 was fairly diverse and contained some 'clean water' species. It was not as diverse as Sites 3 or 8, but it was more diverse than most other sites.

Table 13 shows that Sites 2 and 6 had the same diversity even though the sites were located on different rivers. As explained before this does not imply that the two sites had similar fauna composition just a similar distribution of species and numbers, but the cluster analysis revealed that Sites 2 and 6 also had similar faunas since the two sites'

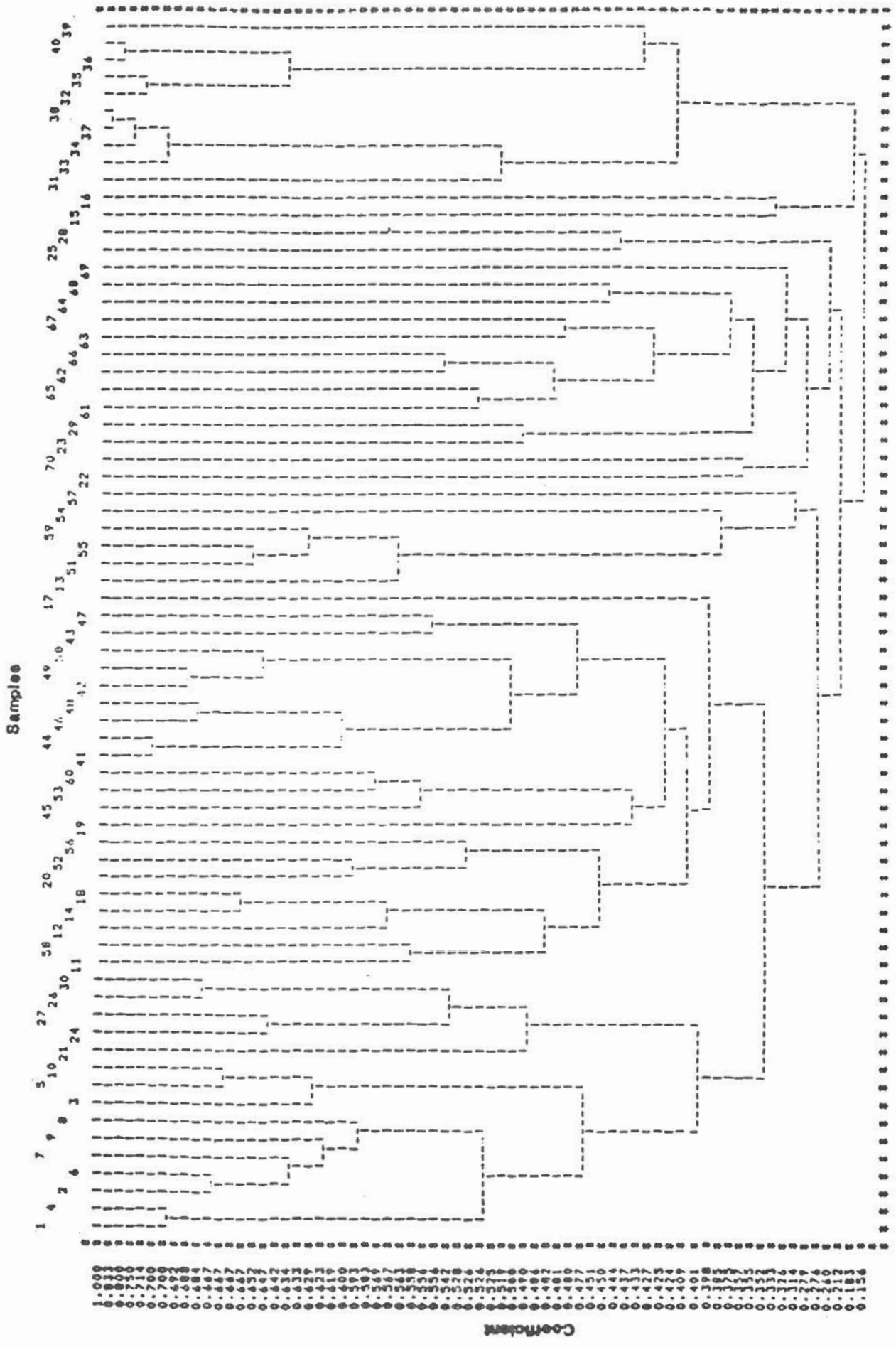


Figure 15. Dendrogram of Cluster Analysis on 1984 Data.

samples clustered together. Nine samples from Site 5 clustered at S_j=0.480. Site 5 samples clustered with those of Sites 2 and 6 (S_j=0.425) probably because Site 5 was located downstream of Site 6, and therefore under some influence of that site. As this shows, Site 5 samples clustered together more tightly than they did with samples of Sites 2 and 6, thereby implying the community structure of Site 5 was more similar within itself than with samples from other sites. Samples from Sites 2, 5, and 6 clustered with Site 1 samples at S_j=0.352 This is understandable as Site 1 was the most downstream site and was probably influenced by all upstream communities

Samples from Site 4a clustered together with S_j=0.424 but did not correlate significantly with samples from any other sites. Therefore the community composition of Site 4a was significantly different from all other sites. This was expected as Site 4a is located downstream of the sheep/cow pasture at the CDA Experimental Farm, and as such would be expected to show signs of serious water quality deterioration. The samples from this site characteristically contained few taxa; the only taxa usually present in significant numbers were chironomids and *Tipula* sp.

5.5.4 Cluster Analysis on 1985 Data

As seen in the previous year, there was a tight cluster comprised of samples from Sites 3 and 8 (S_j=0.419). In this case, however, there was no significant clustering with samples of Site 1. The samples from Site 1 clustered most closely with samples from Site 5 and some Site 6 samples (S_j=0.495). There was also a cluster of the remaining Site 6 samples with some from Sites 2, 4b and 7. As explained below, this may be due to inputs of headwater streams at these sites.

Again, as with the first year, Site 4a stood on its own, and only clustering poorly with the remaining sites (S_j=0.140).

The samples from site 2 were grouped together with S_j=0.392. Samples from Site 5 were clustered together at S_j=0.563. The samples from Site 6 were scattered, having S_j=0.439.

All the remaining sites (1, 3, 4a, 4b, 7 and 8) had samples scattered throughout the

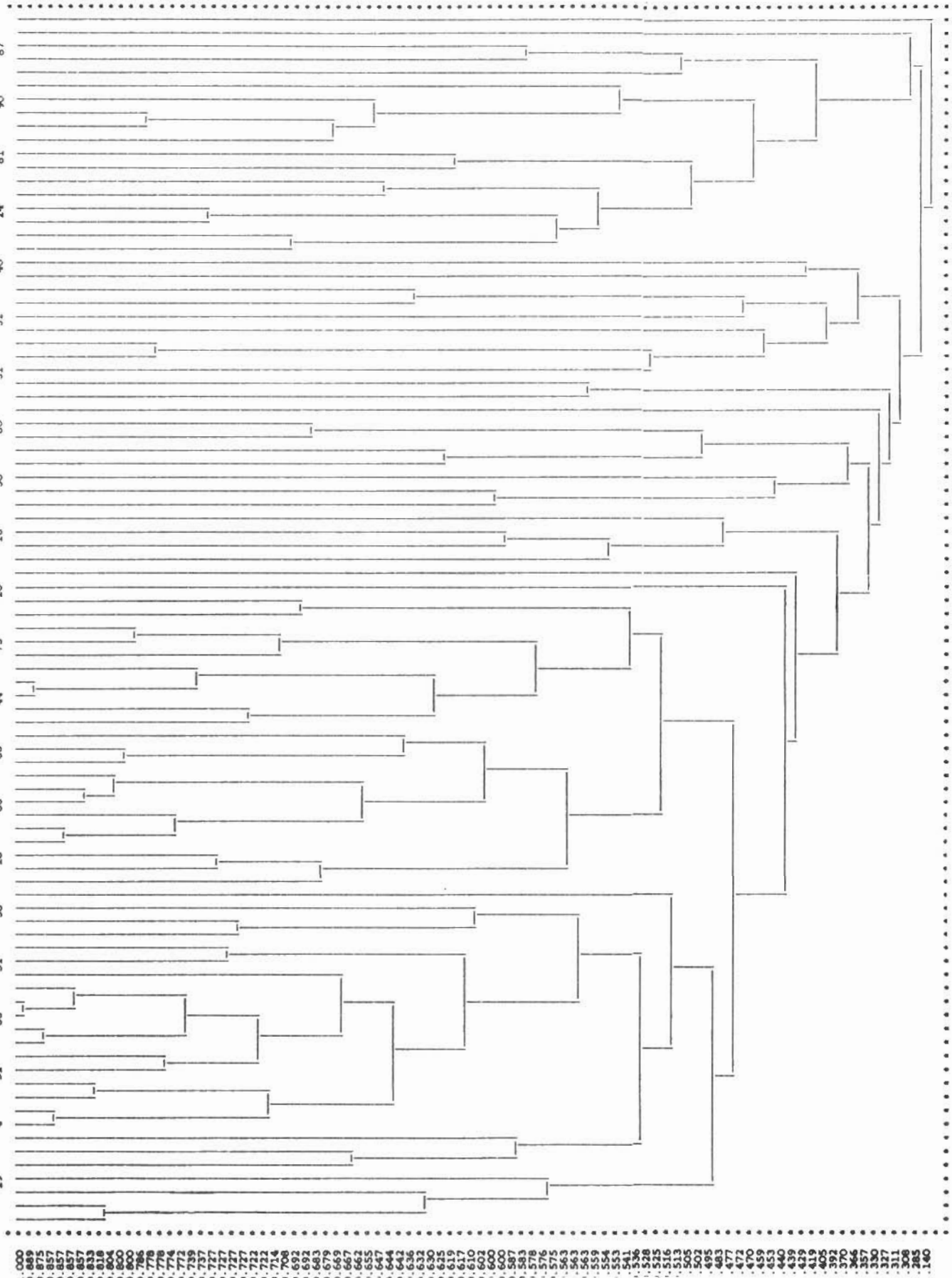


Figure 16. Dendrogram of Cluster Analysis on 1985 Data.

dendrogram, but had some samples clustered together tightly. There was a cluster of eight samples from site 1 grouped together with S_j=0.495. All ten samples were clustered with S_j=0.357. Site 3 had overall S_j=0.285, but had a group of seven samples with S_j=0.505. Site 4a had a tight cluster of nine samples, at S_j=0.366, while the overall index of similarity was 0.140. It had one sample which stood completely on it's own in the dendrogram, not clustered with any other sample. This sample contained only two taxa (chironomids and *Tipula* sp.) therefore did not cluster with any other sample. Site 4b had a cluster of eight samples with S_j=0.525, the entire site was grouped at S_j=0.357. This site was not included in the first year sampling program, therefore was not present in the 1984 cluster analysis. As stated above, the samples from this site were most closely grouped with those from Sites 2 and 7 and some from Site 6. All four of these sites (i.e., 2, 4b, 6 and 7) were either located on, or receiving input not far upstream from, a low order tributary (i.e., first or second order). It is possible that these inputs are affecting the community composition in the river itself. Headwaters typically have higher pH and relatively more allocthonous input, as well as numerous other physico-chemical differences, as a result of their close association with the terrestrial environment. The input of these waters into the main river may explain some of the similarities between these sampling sites. Site 7 was also absent from the cluster analysis of the data from 1984. It had on overall S_j of 0.357. A small cluster of Site 7 samples (four samples) was grouped at S_j=0.502. At the time of removal, the rock bags at this site were washed up along the bank of the river, and some of them may have been in unsuitable habitats in the preceding days or weeks. The river edges were frozen and snow-covered at the time of removal, and the actual position of the bags in the water (above or below the water level) could not readily be determined. No bags were taken that were obviously in unsuitable areas, however some of the heterogeneity in Site 7 samples could be attributed to this situation.

There was a tight cluster of nine samples from Site 8, which had an index of similarity of 0.419; the entire site had S_j=0.308.

5.6 Water Quality and Benthos Distribution

The values of Pearson correlation coefficients (C.C.) were calculated for various combinations of stream characteristics, water quality, and the invertebrate data. All correlations were calculated using the computer statistics package MINITAB (Ryan, 1981). Water quality parameter values used in Sections 5.6.2. and 5.6.3. were obtained from the Surface Water Quality Study, conducted by NDOE, 1984, and are summarized in Table 15.

The total number of individuals, the number of taxa, diversity index, total biomass, stream order and drainage area were correlated with several water quality measurements to determine which parameters were significantly correlated with the distribution of the invertebrate fauna in the Waterford River system. Tests of association between stream order, drainage area and the invertebrate data were also performed.

Between-taxa correlations were also performed to determine whether any consistent co-occurence patterns could be identified. Highly correlated species were then compared to water quality and diversity indices to determine whether there were any "clean water" and "unclean water" trends.

The purpose of the correlations was to check if patterns in water quality data corresponded to patterns in invertebrate data. Matrices of correlation coefficients between mean benthos numbers, biomass, and diversity per station, and selected stream parameters are presented in Tables 16 and 17. It is noted that values of correlation coefficients greater than 0.754 are significant at $p = 0.05$ (d.f.$= 5$) that is, the two sets of data show a statistically significant correlation.

5.6.1 Stream Characteristics and Invertebrate Data

As expected, stream order, drainage area, and flow rate are all highly positively correlated with each other. Figures 17 and 18 show the relationship of stream order to drainage area and of flow rates to drainage area. Regression equations were calculated using SPSS-X by the Least Squares method. Both combinations show a strong linear relationship, with high R^2 values. Regressions were calculated on the 1984 data only, as

flow rates were measured only in the autumn of 1985, and a regression on this data would not be appropriate. The relationships between drainage area, stream order and flow rates are well known to hydrologists, and are now used, in conjunction with other stream parameters, by biologists to predict the type of benthic community present in a river (Vannote *et al.*, 1980; Merritt *et al.*, 1984).

In 1984, the total number of individuals was highly correlated with stream order, drainage area, and flow rate (Table 16). High flow rates generally decrease the diversity of the fauna (Hynes, 1970) but they appeared to have little effect in the Waterford River basin; flow rates were not significantly correlated with diversity (Table 17). None of the other invertebrate data (e.g., biomass, number of taxa, etc.) are significantly correlated with stream characteristics. It is interesting to note that even though the number of individuals did show a significant change over the course of the river system, the biomass did not.

In 1985, there were no significant correlations between any of the stream characteristics and the invertebrate data (Table 17). Neither stream order, flow rate, nor drainage area showed a significant relationship to the number of individuals, taxa, total biomass or biomass without large individuals.

5.6.2 Stream Characteristics and Water Quality Data

The only water quality parameter measured in the Surface Water Quality study, which exhibited a significant correlation with the stream characteristics was the dissolved chloride content (Table 17). No other parameter showed a significant relationship with either stream order, flow rate or drainage area at α=0.05.

As the three stream characteristics were all highly inter-related (Figures 17 and 18), the water quality parameters tended to correlate with each one in a similar pattern (Table 17).

able 15. Chemical and bacterial water quality measurements taken during first year of tudy (Mean and Range).

Site	Total Organic Carbon (mg/L)	Nitrite-Nitrate (mg/L as N)	Total Nitrogen (mg/L)	Total Phosphorus (mg/L)	Dissolved Oxygen (mg/L)	Total Alkalinity (mg/L as $CaCO_3$)	Dissolved Chloride (mg/L)	Total Coliforms (#/100 mL)
1	8.3 2.5-22.5	0.390 0.01-0.90	0.66 0.31-1.30	0.084 0.008-0.300	11.4 8.8-16.8	8.0 1.5-15.2	50.0 12.5-350.0	17,692 1600-110,000
2	9.7 4.0-20.0	0.260 0.01-0.67	0.66 0.20-2.20	0.096 0.008-0.650	11.1 8.6-15.6	7.8 0.5-16.5	23.0 5.6-170.0	18,668 80-72,000
3	7.1 3.0-15.0	0.068 0.01-0.63	0.22 0.01-0.72	0.035 0.001-1.300	11.3 8.6-14.6	2.8 0.5-9.4	13.5 4.8-33.0	552 0.2-8600
4a	9.3 2.5-85.0	0.380 0.01-0.60	0.58 0.29-1.10	0.051 0.003-0.700	11.4 8.7-14.8	9.5 2.7-17.0	40.5 6.2-130.0	1698 200-8600
5	11.5	0.270	0.50	0.120	11.5	5.3	39.1	6198
6	9.2 3.5-31.5	0.260 0.01-0.70	0.54 0.17-1.80	0.068 0.008-0.470	10.8 6.8-14.4	5.6 0.6-10.8	37.3 7.0-100.0	24,865 520-560,000
8	7.7 2.5-15.4	0.080 0.01-1.10	0.25 0.05-1.40	0.015 0.001-0.230	11.2 8.3-14.7	1.7 0.5-15.1	7.6 4.4-27.0	479 0.2-5600

lote - Data obtained from NDOE Surface Water Quality Report, unpublished report.

Table 16. Correlations of benthos numbers and biomass with stream parameters.

	Invertebrate Data			
Stream Parameter	Diversity Indices	Biomass	Number of Individuals	Number of taxa
		1984		
Stream Order	0.031	0.550	0.720	-0.033
Drainage Area	0.131	0.356	0.820	0.092
		1985		
Stream Order	-0.233	0.323	0.331	-0.218
Drainage Area	-0.075	0.361	0.399	-0.035

Table 17

Correlation coefficients between selected stream characteristics, invertebrate numbers, biomass and water quality data.

Source	pH	Specific Conductivity	Temperature	Turbidity	Flow Rate	Organic Carbon	Nitrate-Nitrite	Dissolved Nitrogen	Dissolved Phosphorus	Dissolved Oxygen	Total Alkalinity	Dissolved Chloride	Total Coliforms
Stream Order	0.466	0.683	-0.082	0.565	0.875	0.206	0.551	0.479	0.513	0.115	0.271	0.756	0.550
Drainage Area	0.466	0.541	0.180	0.340	0.959	-0.040	0.496	0.422	0.368	0.297	0.281	0.627	0.391
Diversity Indices	-0.733	-0.642	-0.361	-0.303	-0.021	0.383	-0.715	-0.707	-0.324	0.154	-0.859	-0.572	-0.269
Biomass	0.460	0.737	-0.232	0.945	0.495	0.671	0-559	0.315	0.562	0.543	0.316	0.729	-0.056
Number of Individuals	0.158	0.205	0.053	0.355	0.726	0.051	0.109	0.058	0.390	0.525	-0.121	0.289	0.049
Number of taxa	-0.730	-0.687	-0.337	-0.358	-0.061	-0.434	-0.766	-0.753	-0.348	0.172	-0.874	-0.623	-0.411

Water quality data taken from NDOE Surface Water Quality Report, unpublished report.

Table 18. Summary of Physical Parameters Measured During the First Year of the Study (Mean and Range).

Site	Temperature (°C)	Specific Conductivity (μmho)	pH	Turbidity (J.T.U.)	Flow Rate (m^3s^{-1})
1	7.6 0-19.8	190 69-1100	6.8 5.8-7.7	17.4 0.7-128	1.249 0.558-1.920
2	8.0 0-21.6	106 56-345	6.6 5.8-7.6	7.9 0.6-50	0.103 0.004-0.176
3	7.1 0-20.0	70 32-194	6.1 5.1-7.0	2.2 0.3-32	0.213 0.126-0.394
4a	7.5 0-19.1	175 79-380	6.7 5.9-7.7	16.9 0.4-285	0.389 0.085-1.040
5	7.1 0-18.5	164 59-600	6.5 5.8-8.9	33 0.6-520	0.461 0.322-0.683
6	6.8 0-18.0	155 73-360	6.3 5.5-6.9	14.7 0.8-280	0.293 0.206-0.472
8	7.1 0-19.3	40 27-104	5.6 4.8-6.5	1.4 0.3-6	0.156 0.092-0.288

Note - Data obtained from NDOE unpublished report.

Table 19. Summary of Physical Parameters Measured During the Second Year of the Study (Mean and Range).

Site	Temperature (°C)	Specific Conductivity (μmho)	pH	Turbidity (J.T.U.)	Flow Rate (m^3s^{-1})
1	4.4	259.8	6.82	2.63	0.988
	0-12	130-510	6.8-6.9	1.4-3.3	0.300-2.670
2	4.3	511.3	6.72	3.85	0.113
	0-12	55-1600	6.6-7.0	1.9-8.3	0.034-0.304
3	4.0	571.7	6.19	1.33	0.138
	0-12	55-1600	5.8-6.7	0.62-2.0	0.042-0.374
4a	4.1	201.3	6.72	2.01	0.059
	0-12	170-280	6.2-7.1	1.4-3.0	0.018-0.160
4b	4.5	167.5	6.66	1.76	0.056
	0-12	100-215	6.4-6.8	1.2-2.6	0.017-0.152
5	4.9	373.8	6.8	12.19	0.448
	0-13.5	175-920	6.4-7.3	2.1-34	0.156-1.120
6	4.6	381.3	6.50	23.15	0.307
	0-13	125-1100	6.3-6.6	2.0-49	0.107-0.769
7	5.0	426.3	6.89	10.18	0.569
	0-14	160-1150	6.3-7.1	1.9-26	0.198-1.424
8	4.1	353.8	5.85	1.91	0.101
	0-12	40-1250	5.4-6.3	0.95-3.6	0.031-0.272

Note - Measurements were taken monthly, from October, 1985 to January, 1986.

Figure 17. Regression of Stream Order on Drainage Area.

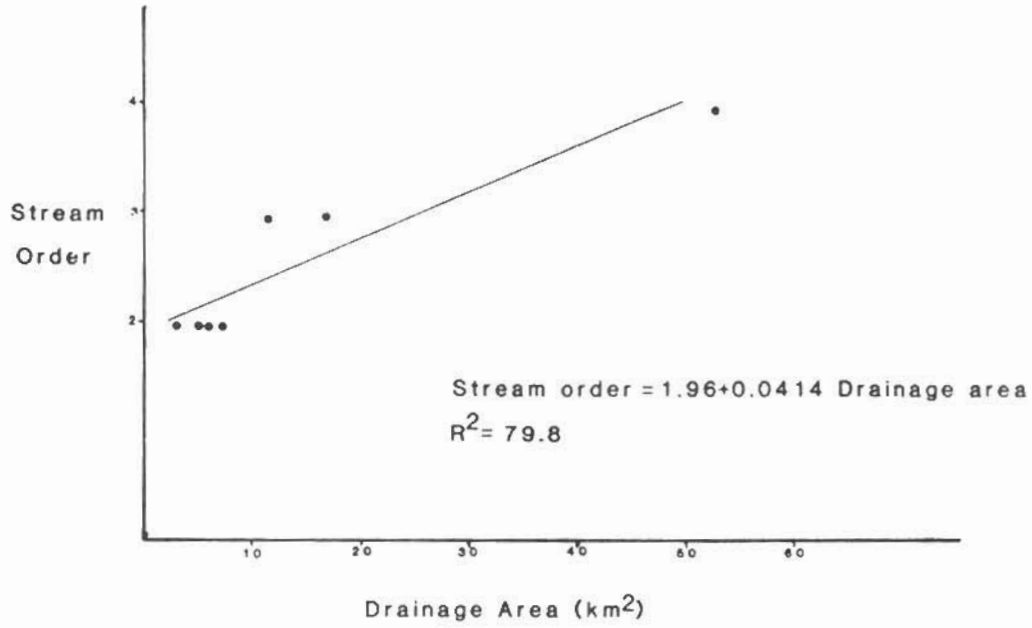
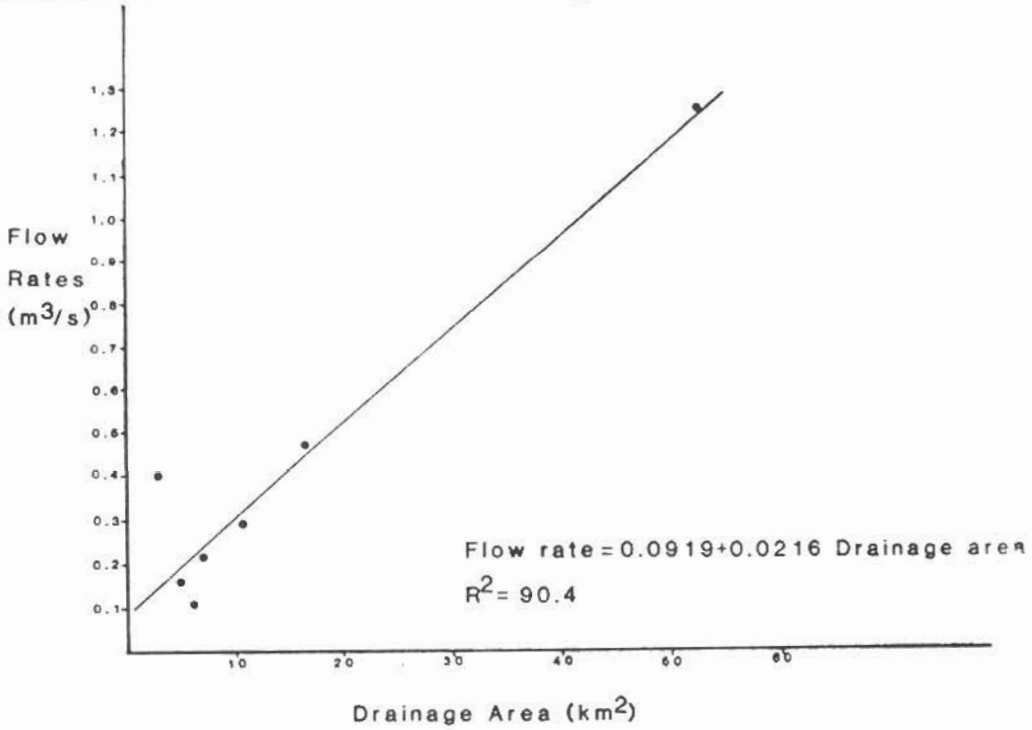


Figure 18. Regression of Flow Rate on Drainage Area.



5.6.3 Water Quality and Invertebrate Data

The diversity indices and the number of taxa were significantly negatively correlated with total alkalinity and nitrate-nitrite content (Table 17). Diversity and taxa number did not significantly correlate with the other water quality data.

Biomass was significantly positively correlated with turbidity but did not significantly correlate with the other water quality parameters (Table 17).

No water quality parameter was significantly correlated with the total number of individuals (Table 17).

The results of the analytical procedures indicate that the composition of the benthic community changed in response to the chemical and physical conditions present at the sample sites. Wide ranging differences were found in; community diversity, the number of individuals, and the number of taxa. No significant differences in total biomass were detected, with the exception of Site 4a. Biomass without large individuals did show a pattern similar to taxon number and number of individuals.

The community composition analysis indicated that some sites showed similarities to others, while one was similar to no other (i.e., Site 4a).

Stream characteristics were shown to be highly correlated, while they were not correlated with the invertebrate data (with the exception of the number of individuals).

The results presented in the section will be discussed in the next chapter.

6 DISCUSSION

6.1 Invertebrate Data

The ensuing discussion is primarily based on a site-by-site comparison. Clean water and unclean water sites will be identified, as well as a discussion of which taxa tend to be found in clean and unclean water. As stated earlier, stream order affects the composition of the benthic community, in terms of the proportion that each trophic group represents (Vannote *et al.*, 1980; Merritt *et al.*, 1984).

Site 1, a fourth order stream, contained a large number of grazers (eg., *Ephemerella subvaria*, *Baetis tricaudatus* and *Promoresia tardella*), especially in 1984, but also had a large number of shredders (eg., stoneflies). Many shredder species become increasingly more predaceous as they mature, and the size of their mouthparts increases (Lamberti and Morre, 1984). Such changes have been reported for species of both Trichoptera and Plecoptera. Collectors made up well over half of the community at Site 1 (*Hydropsyche* spp. alone, made up over half of the individuals present). Other collectors (e.g., chironomids and annelids) were also found in large numbers at this site.

Site 8, a second order stream and the control site, contained a diverse fauna, with both shredder and grazer taxa but populations of most taxa were low. Few collectors (e.g., an occasional small hydropsychid) were observed at this site. The community at this site was the most diverse. There were more taxa per sample at this site, and fewer individuals per taxon. No one taxon dominated the samples. Site 3 was the most like the control site (i.e., 8). It, too, was located on South Brook, and supported a diverse fauna.

The flow regime below dams may be considerably different from that of unregulated streams. As well, populations of some running water taxa may be depleted in streams below dams, as illustrated by Site 4a. Drift from upstream areas is unavailable to replenish the individuals lost from the regulated stream segment (Ward, 1984). The results of the first year sampling program suggested that this was the case at Site 4a, so Site 4b, located upstream of the dam was added to the sampling program. This site

yielded a much more diverse fauna, having $\bar{H}=1.741$. This site supported some taxa which were common to Sites 2 and 6 (e.g., *Leptophlebia cupida* and *Aeshna* sp.) and some common to, or in similar numbers to Sites 3 and 8 (e.g., *Rhyacophila fuscula*, *Baetis tricaudatus* and Chironomidae).

Removal of vegetation along streams can result in ecological conditions that originally characterized lower reaches of rivers (e.g., nutrient levels) but have now been shifted upstream. In addition, because downstream conditions are dependent upon upstream functions (Vannote, *et al.*, 1980), alteration of the riparian habitats of headwater streams could be reflected in lower reaches. The poor diversity of Site 1 could be a reflection of the lack of vegetation upstream of the site. All sites had vegetation growing along the banks and Sites 3, 4b, 5, 6, and 8 were well shaded. Shading can effect the occurrence and abundance of certain species. Hughes (1966) found that many species are indifferent to shade, however *Hydropsyche* spp. were more abundant in open places. Both Sites 1 and 2 were not shaded and both contained large populations of *Hydropsyche* spp.(Tables A7 and A16) although well shaded sites, such as 5 and 6, both contained high *Hydropsyche* spp. populations (Tables A7 and A16).

Although Sites 3 and 8 had low biomasses, the number of individuals was fairly high and more importantly the number of species and the diversity indices were the highest of the sites, in both 1984 and 1985. Both sites contained taxa only found at these two sites (Tables A1 to A18). The results of the Scheffe's test conducted on the different taxa indicated that unique populations of *Promoresia tardella*, Ceratopogonidae, *Paraleptophlebia adoptiva*, *Habrophlebia vibrans*, *Paracapnia opis*, *Leuctra ferruginea*, *Micrasema wataga*, *Chimarra aterrima*, and *Rhyacophila carolina* were found at either Site 3 or 8 or, more often, both, in at least one of the two sampling years. Usually, these taxa were in significantly greater numbers at these two sites, as compared to the other sites (Tables 4 and 7). The fauna was more diverse and balanced and no taxon dominated the samples of these sites.

In 1984, samples from Sites 1 and 5 contained more individuals per sample, on average,

than the other sites, however the diversity indices were lower than those of Sites 3 and 8. The same pattern is true for 1985, but Site 2 supported more individuals per sample than either of Sites 1 or 5 (Table 12). Except for a few taxa in which only one specimen was found, due to its rarity in the community, or to an accidental occurrence there were no unique taxa at these sites. The distinguishing factor at these sites was the local abundance of some widely distributed species (e.g., *Ephemerella subvaria*, *Baetis tricaudatus*, *Hydropsyche slossonae*, and *H. sparna*). Site 5 also contained the most *Arctopsyche ladogensis* and *Rhyacophila fuscula* while Site 1 contained many species which were also found at Sites 3 and 8, probably because of some influence from South Brook.

Sites 2 and 6 had similar diversity indices and similar faunas in both years, but Site 2 had more individuals, more taxa, and a higher biomass (without large individuals) per sample than Site 6. Although there were no statistically distinct populations of benthic taxa at either site, they both had high populations of *Hydropsyche betteni*, *H. slossonae*, *H. sparna*, and *Ephemerella subvaria*.

The most depauperate site was 4a, which contained mostly *Tipula* sp. and lumbricids, in relatively large populations. Samples from this site in 1984 completely lacked mayflies, stoneflies, and water beetles, and contained only four caddisflies in total (3 *H. sparna*, and 1 *H. slossonae*). The samples in 1985 did yield an occasional mayfly or beetle larva, but the numbers collected were far below those collected at the other sites. Some chironomids were found at that site but also in low numbers. Larvae of chironomids, *Tipula* sp. and muscids were the only diptera collected at Site 4a, in 1984. In addition to these taxa, one tabanid was collected there in 1985. Enchytaeidae were also found at Site 4a but in very low numbers. Some *Lumbriculus variegatus* were also collected at this site in 1984. The low numbers of individuals and species might be a reflection of sampling, since the artificial substrate samplers at Site 4a were found partially washed up on the stream bank in 1984 (S. Bonnyman, personal communication). The poor water quality of this site, and the observed poor fauna, is most likely a result of the combination of run-off from the pasture, with its high fecal content, and the entrapment of invertebrates in the reservoir. A flock of geese is

maintained at the reservoir, which is also undoubtedly affecting the quality of the water immediately downstream. It is unlikely that the poor water quality is a result of any upstream perturbation or input, as Site 4b showed a much higher diversity, and supported a much more "healthy" community, indicating a cleaner environment.

At many sites, a few taxa contributed a major portion to the sites biomass and/or number of individuals. *Hydropsyche spp.* formed a dominant component of the samples of Sites 1, 2, 5, 6 and 7 and therefore these sites may have experienced some type of environmental perturbation. Site 8, the control site, had no one taxon dominate the site, except tipulids which were found in low numbers (but has high biomass/individual). Site 4a was dominated almost exclusively by tipulids and earthworms and therefore this appeared to be the most stressed site. A well-balanced benthic community is an indication of undisturbed environmental conditions but a community that is dominated by one or a few taxa is an indication of a disturbed environment (Gaufin, 1973). Site 1 contained many individuals of a few taxa, namely chironomids and hydropsychid species, indicating a stressed environment. Sites 2, 5, 6 and 7 all had a similar pattern of a few species dominating both biomass and number of individuals. Site 4a also contained very low numbers of individuals and few taxa with the terrestrial earthworms and tipulids dominating the site in numbers. Almost no other taxa were collected at that site, also indicating a stressed environment, probably more stressed than the other sites. Gaufin (1973) found that similar results indicate a stressed environment, and this is likely the case in this study.

Based on the above observations, the sites can be divided into four categories on the basis of the faunal data;

1) "Clean" Sites - 3 and 8.

2) "Fair" Sites - 1, 4b and 5.

3) "Poor" Sites - 2, 6 and 7.

4) "Extremely Poor" Site - 4a.

Dominant populations of chironomids and annelids are often characteristic of polluted conditions (Gaufin, 1973b; Newbold *et al.*, 1980; Whiting and Clifford, 1983), however, in this study the distribution of these taxa did not reveal that pattern. In fact, there were no distinctive populations of chironomids or annelids. Significant populations of chironomids and *Lumbriculus variegatus* were found but revealed no patterns since all types of water conditions had varying populations of these organisms. For example, numbers of chironomids were found to be similar at Site 3 (a "clean water" site) and Site 2 (a "poor water" site) and *L. variegatus* numbers, in 1984, were similar at Site 8 (a "clean water" site) and Site 2 (a "poor water" site). It is possible that artificial substrate samplers do not collect these organisms or the cleaning and sorting techniques used lost smaller chironomids and annelids (e.g., *Tubifex* sp.) from the samples. While the numbers of chironomids collected in 1985 had increased over the number collected in 1984, the numbers of annelids collected had decreased. No *L. variegatus* were collected at all in 1985. This is again perhaps due to the shortened colonization period. The artificial substrate bags had not been subjected to the same degree of siltation and sediment loading in 1985 as in 1984, so it is probable that the microhabitat required for colonization by some annelid taxa (e.g., *L. variegatus*) was not provided. If the artificial substrate did not provide the correct microhabitat, those taxa would not be collected.

Mayflies have generally been considered to be very sensitive to pollutants but this is not always true. *Ephemerella subvaria* has been collected in a polluted stream in St. John's (Larson and Colbo, 1983). The results of this study indicate that this species is tolerant of slightly polluted water, as the most dense populations were found at Sites 1 and 5. The more polluted sites (2 and 6) also had high populations of *E. subvaria*. In cleaner waters the species was less common, perhaps due to greater competition or a decreased availability of food. *Baetis tricaudatus* appeared to do well in slightly polluted waters, as the highest populations were found at Sites 1 and 5, but did not do well in more polluted waters (Sites 2 and 6). Newbold *et al.* (1980) found higher densities of *Baetis spp.* in polluted waters. *Baetis pygameus*, *Habrophlebia vibrans*, and *Paraleptophlebia adoptiva* appear to be less tolerant of polluted water; the largest populations were found at Sites 3 and 8, the "clean water" sites. *Leptophlebia cupida*, Newfoundland's most common mayfly species (Larson and Colbo, 1983), was fairly

evenly distributed at all sites with the exception being that none were found at Site 3, a "clean water" site. The extremely poor site (4a) also showed a lack of *L. cupida*, but only in 1984. The population was much higher at most sites in 1985 than in 1984. *L. cupida* is usually found in lentic habitats, therefore it is unusual to collect it in streams in the high numbers exhibited in 1985. The site with the highest mean number of *L. cupida* per sample was Site 2 ($\bar{X}$=22.6/sample). The samplers at this site were located in a slow flowing pool, which could account for the high population.

Stoneflies (Plecoptera), although not too sensitive to high pH, are sensitive to most other water quality parameters (Roback, 1974). Twelve species of stoneflies occur in Newfoundland but only six are found on the Avalon Peninsula (Larson and Colbo, 1983). Four species were found in the Waterford River basin although *Isoperla transmarina* was the only species which was widely distributed throughout the system. *Isogenus frontalis*, of which only two individuals were collected in 1984 (both at Site 3), was collected at five sites in 1985 (two of which were the newly added Sites 4b and 7). The number collected at each site was low, the highest being the "clean water" sites (3 and 8), each with a mean of 1.0 *I. frontalis*/sample (Table A15). The distribution pattern of the stoneflies indicates that they are "clean water" organisms since most specimens were found at Sites 3 and 8. Three species were also found at Site 1, including the highest density of *Isoperla transmarina*, an indication that Site 1 was a fairly clean site, probably because of the South Brook influence.

The water beetle *Promoresia tardella* is usually associated with aquatic mosses (Larson and Colbo, 1983), which need a firm unsilted object on which to attach themselves. Aquatic mosses are generally characteristic of clean water, but can survive in mildly polluted streams (Hynes, 1970). *P. tardella* was found in large numbers at Sites 3 and 8 and in smaller numbers at Site 1. The presence of "clean water" taxa, such as stoneflies and *P. tardella* at Site 1 indicated that site was fairly clean, probably due to the mixing of South Brook water with Waterford River water.

Most of the caddisflies seem to prefer "clean water", especially *Micrasema wataga*, *Chimarra aterrima*, and *Rhyacophila carolina* but some species appear to be more

pollution tolerant. Hydropsychid species favour some degree of pollution (Nielsen, 1974) and the results of this study support this finding. *Hydropsyche betteni* was not found at either Sites 3 or 8 in 1984, while in 1985, they occurred at these sites in low numbers. *H. slossonae* was found in higher numbers at Site 3 but not as high as at some of the "unclean" sites. *Arctopsyche ladogensis* was found in both "clean water" sites but was most abundant at Site 5, a "fair water" site. The largest hydropsychid populations were found at the "fair water" and "poor water" sites. Roback (1974) concluded that net-building caddisflies seem to be tolerant of organic loadings, but not of toxic pollutants. This was apparently true of the hydropsychid species but not of *C. aterrima*, which were found only at the "clean water", and occasionally the "fair water" sites. Nielsen (1974) found *Rhyacophila* species to be among the most resistant caddisflies. Most *Rhyacophila* species were found at Sites 3, 5 and 8. *R. carolina* seems to prefer "clean water" while *R. fuscula* was found at both the "clean" and "fair" water sites. The results indicate that *Rhyacophila* spp. appeared to be tolerant of slightly polluted water but not badly polluted water.

6.2 Water Quality and Benthos Distribution

The correlations presented in Tables 16 and 17 do not necessarily imply cause and effect, they are simply measures of association between two factors.

6.2.1 Stream Characteristics and Invertebrate Data

Stream order, drainage area, and flow rates can affect the type of fauna found in a stream. Figures 17 and 18 show the relationship of stream order to drainage area and of flow rates to drainage area. The number of individuals collected at a site had a significant positive correlation with stream order, drainage area, and flow rates but diversity and biomass were not significantly correlated with the stream characteristics.

The Waterford River basin is probably too small to permit detection of major natural changes in fauna due to changes in the stream order. Although differences in fauna between first, second and third order streams occur, the most noticable differences (e.g., the shift from heterotrophy to autotrophy and the accompanying shifts in trophic group proportionality) occur in even higher order streams. It is possible to detect induced changes in functional group proportionality, as is evidenced by the fauna of Site 1.

6.2.2 Stream Characteristics and Water Quality

The one water quality parameter which correlated significantly with the stream characteristics was dissolved chloride. All other water quality parameters examined did not correlate significantly with the stream characteristics.

The ocean is a major source of atmospheric chloride. The high chloride content in salty air from the sea results in increased dissolved chloride content in streams. The proximity of the stream to the ocean should affect the dissolved chloride content. Results of the Surface Water Quality Study showed that downstream areas contained higher levels of dissolved chloride than upstream areas. The positive significant correlation observed can be explained by the shorter distance to the sea and to the reduction in the amount of forested area surrounding the lower portion of the drainage basin. The coniferous cover could act as a filter removing chloride and other ions from the sea-air. Another likely source of the high dissolved chloride content of the river is run-off from winter application of road-salts, and the concentration would probably increase downstream as urbanization increased. One would expect this to have a pronounced seasonal effect, as the concentration of salt in run-off waters should decrease over the summer months, as the salt residues are washed away and diluted. The Surface Water Quality study showed that this pattern did indeed occur in the Waterford River.

The lack of significant correlations of the other water quality parameters with stream size indicates that the river is probably suffering from environmental stress. Natural rivers usually show a general pattern of changes in the stream characteristics, both chemical and physical, from headwaters streams to the higher order downstream reaches (Vannote *et al.*, 1980). Inorganic nutrients (e.g., nitrates and phosphates) and suspended solids usually show an increasing trend over the course of a river system (Hynes, 1967). Soluble organic compounds are generally found with high concentrations at headwater streams because it is here that the maximum interface with the terrestrial environment occurs (Vannote *et al.*, 1980).

Headwaters tend to have higher organic nutrient levels than are observed downstream, again due to the close association with the surrounding landscape. Leaching of nutrients

from the soil provides an important portion of the production-base of the aquatic community (Cummins, 1974). Communities below the headwaters contain a large proportion of collectors, organisms which capitalize on the inefficiencies of the processing by upstream individuals (Vannote *et al.*, 1980).

The Waterford River does not show the above-noted patterns of high organic carbon levels in headwater areas and decreased levels downstream or an increased level of inorganic nutrients at higher stream orders. Site 8 did show physical characteristics and a diverse community, typical of a headwater tributary, however the typical downstream patterns were not observed. Because of this, the river can be assumed to be under environmental stress. Pollution has probably caused a shift in the water quality patterns.

6.2.3 Water Quality and Invertebrate Data

The diversity indices and the number of taxa were significantly negatively correlated with nitrate-nitrite and total alkalinity. An increase in such a water quality parameter will result in a decrease in diversity and the number of taxa.

One of the possible main sources of nitrates is human and animal wastes, high concentrations of which may reflect unsanitary conditions. As the Waterford River system drains large areas of unserviced residential and pastural land, it is quite likely that the increased nitrate-nitrite levels observed are due to waste material. Increased pollution usually results in decreased diversity in the fauna (Gaufin, 1973) and thus the negative correlation between diversity and nitrate-nitrite and dissolved nitrogen could be expected.

Water hardness, of which total alkalinity is a measure, affects the type of fauna (Hynes, 1960). As the hardness of the water declines, a number of changes occur. Worms, shrimp, molluscs and finally chironomids tend to decline in importance to the functioning of the ecosystem, and are replaced by various insects, particularly mayflies, and stoneflies (Hynes, 1960). The results of this study indicated a decrease in diversity and number of taxa as the alkalinity increased. Softer waters had a more diverse fauna

especially a more diverse stonefly fauna, the most sensitive taxa to hard water (Roback, 1974). Since few worms, shrimp, and molluscs were collected definite conclusions on the effects of alkalinity on diversity can not be made, although the pattern observed by Hynes (1960) appeared to exist in the Waterford River basin.

Biomass was significantly positively correlated with turbidity. Turbidity measures the suspended particles in water which are usually held in suspension by turbulent flow and Brownian motion. High turbidity values can often be related to urbanization and other human activities. As turbidity increased so did biomass, probably due to the elimination of sensitive taxa and the enhancement of more resistant taxa, as illustrated by a comparison between Sites 6 and 8. Both sites are the most upstream sites on their respective segments of the system, however Site 6, which runs through an industrial area was highly turbid. The fauna of Site 6 was characterized by a large number of *Hydropsyche* spp., and few of anything else. Site 8, on the other hand, had very clear water, and the community was diverse, not dominated by any one taxa. The species generally enhanced by turbidity were *Hydropsyche* spp. and *Ephemerella subvaria*. Siltation and turbidity from inorganic sedimentation is perhaps the greatest single cause of water quality degradation (Lemly, 1982)

When all taxa collected in this study are taken into consideration, (i.e., including large individuals, such as *Tipula* sp., *Aeshna* sp. and lumbricids) there were no significant differences in mean sample biomass between any sites in 1984, and only Site 4a was considered different in 1985. If decreased water quality had any effect on total biomass, it could not be detected in this study. The low biomass at Site 4a in 1985, could probably be directly attributable to the combined effects of the dam just upstream, and the pollutants and/or fertilizers from the sheep pasture.

Water quality appeared to have a lesser effect on the number of individuals at a site than did stream characteristics.

In conclusion, habitat changes and pollution in the Waterford River system from its many and varied sources resulted in a trend of decreasing faunal diversity, and

increasing faunal biomass due to functional changes over the course of the river. Further, it was observed that the pollution eliminated the sensitive taxa while enhancing the more pollution resistant taxa. South Brook was determined to be in much better condition than the Waterford River. The two sites identified as clean sites were both on South Brook. The upstream reaches of the Waterford River were determined to have poor water quality, as does the tributary of South Brook in which Site 2 was located. Sites 1 and 5, both on the Waterford River, and Site 4b, upstream of the reservoir at the CDA farm, were judged to have fair water quality, and Site 4a, downstream of the reservoir and the sheep pasture at the CDA farm was determined to have the worst water quality.

7 CONCLUSIONS AND RECOMMENDATIONS

7.1 Conclusions

Using the benthic invertebrates as pollution indicators and patterns in invertebrate community structure and population as a method to predict the degree of pollution, several patterns can be detected.

The most downstream site on the Waterford River (Site 1), was in fair condition. The diversity indices were fair and some "clean water" taxa (e.g., *Isoperla transmarina* and *Promoresia tardella*) were found at that site. The cluster analysis of the 1984 data indicated the fauna of Site 1 was similar to Site 3 which was one of the "clean water" sites. The fair condition of Site 1 was probably due to the influence of South Brook, a relatively clean water stream, which enters the Waterford River about one kilometre upstream of that site.

The South Brook sites were in excellent condition with high diversity and good populations of "clean water" species. The high diversity was expected as these sites are not located in urbanized areas. The control site (8) had the highest diversity but the water quality deteriorated downstream, as urbanization increased (as indicated by decreased diversity downstream at Site 3). A tributary of South Brook (Site 2) had poor diversity with only a few taxa dominating the fauna. At this site, urbanization, especially poorly placed septic tanks, and a lack of shade are probably the major factors contributing to the decreased diversity.

Sites 4a and 4b were located on the unnamed tributary of the Waterford River. The results of both sampling programs indicate that Site 4a has a relatively impoverished fauna. Since the water quality results are not the worst at this site (other sites had poorer water quality conditions but richer faunas), the likely reason for the poor fauna could either be sampling techniques, lack of information on a water quality parameter which was not included in water quality sampling procedure, or the lack of drifting invertebrates due to entrapments in the reservoir upstream from this sampling site. However, it is noted that the Site 4a is located on the CDA farm, where pesticides are

being tested on crops. On the other hand, Site 4b, located upstream of the reservoir created by the dam on the tributary, was found to have a relatively diverse fauna, with representatives of several "clean" and "fair" water taxa. The poor fauna of Site 4a, was probably a result of entrapments in the reservoir, causing a reduction in invertebrate drift, but further investigation of the water quality at Site 4a should probably be undertaken.

The relatively more urbanized Waterford River had a lower invertebrate diversity than South Brook. Site 6, the uppermost site on Waterford River, was in poor faunal shape although the river appeared to clean itself, as downstream, Site 5 had a better invertebrate diversity. The water quality deteriorated downstream of Site 5, as evidenced by Site 1 having a lower diversity in 1984 and only marginally higher in 1985. Site 1 would probably have lower diversity than it currently does, however, South Brook appeared to influence the faunal composition. Several "clean water" species were found at that site, which probably drifted downstream from South Brook.

Site 7, which is located between Sites 1 and 5, was only sampled in 1985. It showed a mean diversity index which was roughly halfway between that of Site 1 and that of Site 5.

7.2 Recommendations

The following recommendations suggest ways of improving the benthic community in the Waterford River system. Any improvement in the macrobenthos represents an improvement in the water quality of a river system. Since this study used the macroinvertebrate community as an indicator of water quality, the recommendations are a reflection of this method of environmental monitoring.

1. For future studies it is recommended that the samplers should be left in the water for only 2 months at the maximum.

2. Future studies should have the artificial substrate samplers located both about 10 m above and below the water quality monitoring sites.

3. Monthly measurements of significantly correlated water quality parameters (e.g., nitrate-nitrite content, dissolved chloride, etc.) should be taken for the duration of the sampling program.

4. Enhance the area around Site 2 by planting trees to increase shade and nutrient input; Ward (1984) stated that buffer strips, which are strips of mixed forest next to a stream, can improve the diversity of the stream fauna. Find a solution for the septic tank problem.

5. Further investigation of Site 4a to help explain the poor results obtained from that site. Were the results caused by the sampling technique or some unexamined environmental condition (e.g., pesticides)?

6. The Waterford River was in fair condition but solutions should be found to enhance the river, especially areas upstream of Site 6. Buffer strips should be planted along the river and any external sources of pollution should be pinpointed and eliminated.

7. An investigation to explain why the Waterford River faunal composition deteriorated in the downstream reaches should be undertaken. The probable reason is the lack of shade along the river banks and the presence of industrial areas located in the lower reaches of either South Brook or Waterford River. Some kind of habitat restoration (e.g., planting buffer strips, reducing the amount of channelization) on a large scale would probably be required for any significant increase in benthic community diversity to be realized (i.e., for a significant improvement in water quality).

8. South Brook was in excellent condition so any future development in that area should be carefully monitored and environmental standards either strictly enforced or improved.

9. The area upstream of Site 8 should be cleaned up, and further dumping in the area should be discouraged. Any serious reduction in water quality at this site will undoubtedly have drastic effects on both South Brook, and the downstream reaches of the Waterford River (i.e., below the confluence of the two).

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APPENDIX A

A Review of Previous Studies on the
Waterford River System

1. Surface Water Quality Study (1980-1984). A joint study, conducted by Newfoundland Department of the Environment and Environment Canada, found the following;

    a. differences in water quality between developed and undeveloped areas of the Waterford River system.

    b. developed areas had consistently higher dissolved ion content, organic nutrient, and trace metal concentrations, than did the undeveloped areas.

    c. high coliform counts, throughout the river system.

    d. a significant reduction in water quality over the course of South Brook.

2. Waterford River Ecological Study (1983). This was a study completed by Arambarri and Haedrich (1983) for the Salmon Association of Eastern Newfoundland, which found that;

    a. the Waterford River System was suffering major abuses, including sewage discharges, dumping of cement and chemicals, and the accumulation of garbage and debris.

    b. increased siltation had clogged many potential trout spawning sites, but populations of brown trout, brook trout, sticklebacks and eels were present.

3. Water Quality Survey of South Brook (1985). This study, conducted by the St. John's Metropolitan Area Board and the Environmental Protection Service of Environment Canada, studied South Brook and its tributaries. Its results showed that;

    a. 31% of sample sites had coliform levels exceeding the environmental standards, as outlined in Environmental Control (Water and Sewage) Regulations, 1980.

4. Waterford River Bacterial Survey (1981-1984). The Waterford River was sampled over a four year period, to determine coliform levels. Sampling was conducted by K. Snelgrove (1981, 1982), S. Dyke (1983) and P.J. Marrie (1984), all students hired by NDOE. The following was observed in this study;

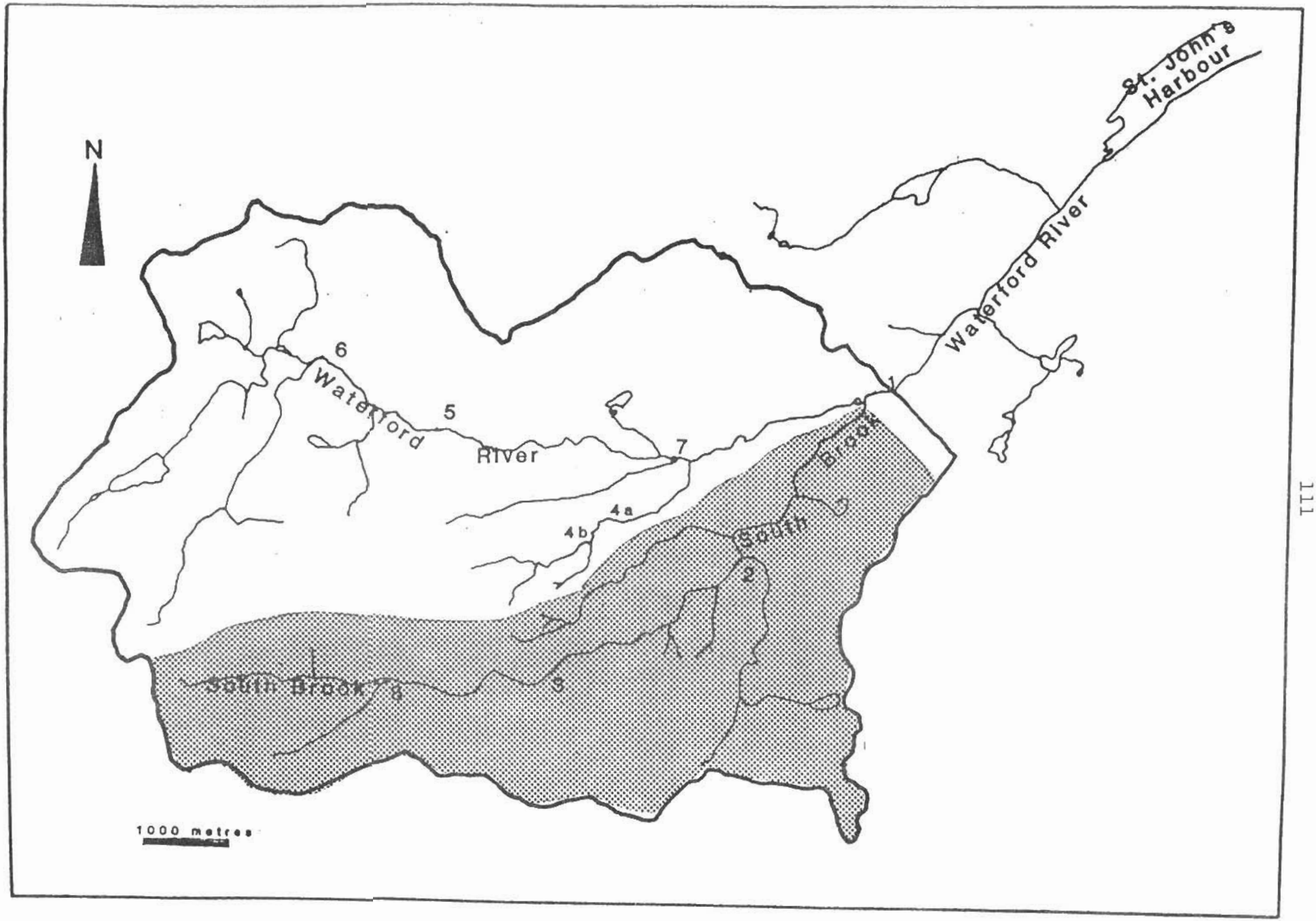

Figure 19. Drainage Area for the Water Quality Survey of South Brook.

a. 1981 - The lower reaches (Bowring Park and below) were sampled and showed excessively high coliform contamination at 18 of 24 sampled outlets.

b. 1982 - The same survey was repeated, with similar results.

c. 1983 - The Waterford River, was sampled between Kilbride (i.e., above Bowring Park) and Commonwealth Avenue, with bacterial analysis conducted at 43 of 62 identified outfalls. 51% showed high coliform contamination, with another 9% listed as borderline cases.

d. 1984 - Sampling was conducted above Commonwealth Avenue, with 10 outfalls being identified. Five (5) of these showed high coliform counts.

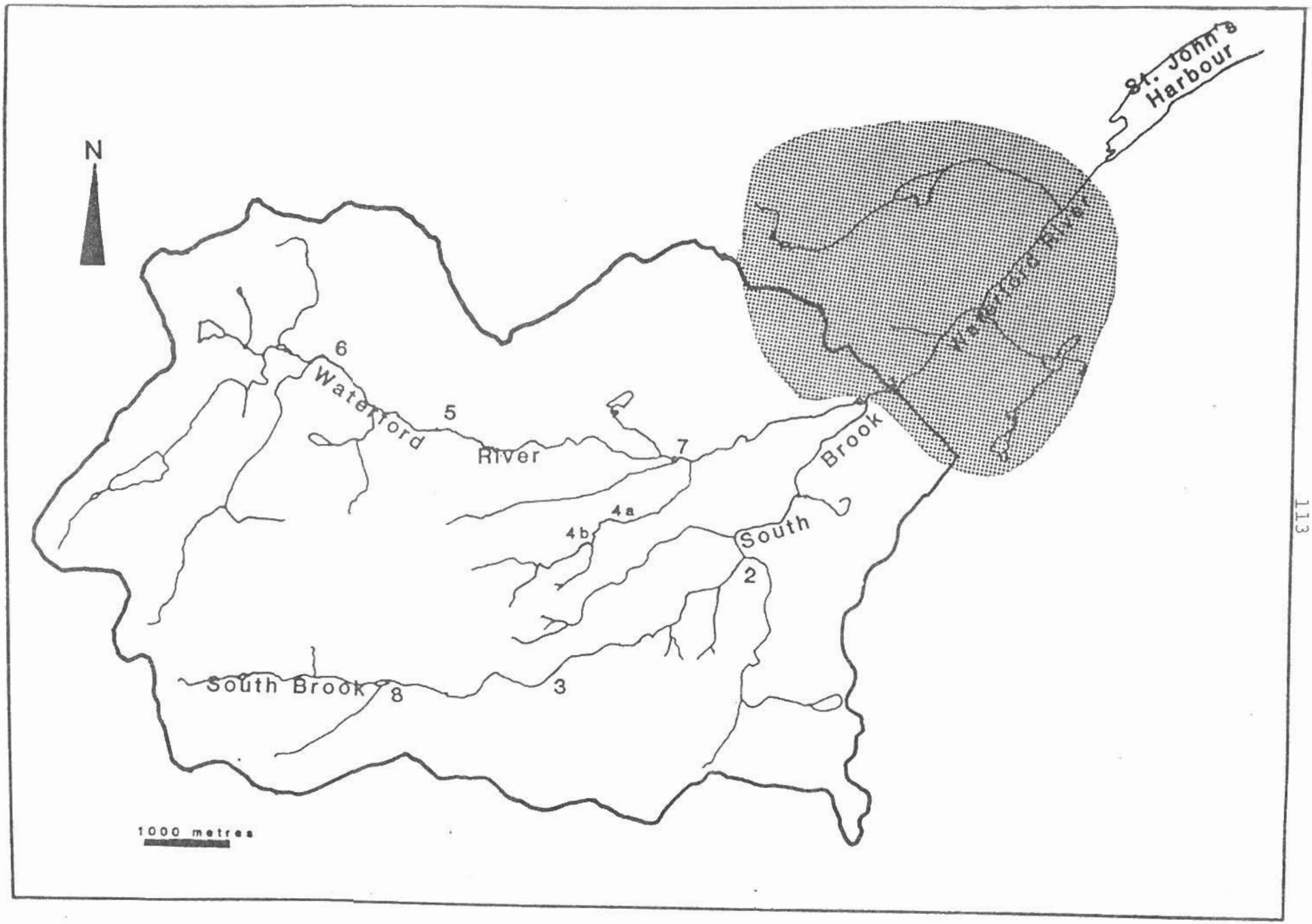

Figure 20. Area of Study for the Waterford River Bacterial Survey, summer 1981 and 1982.

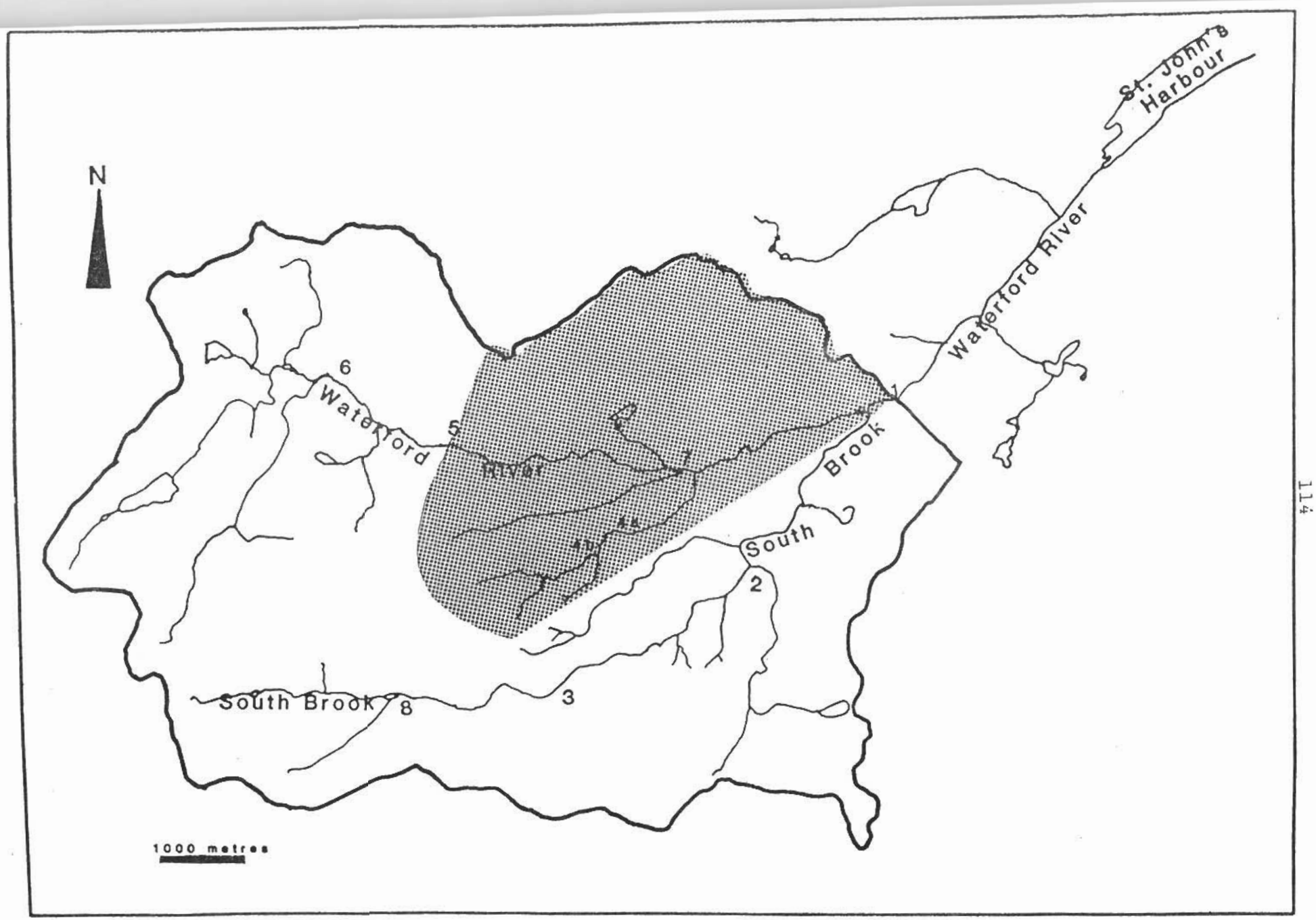

Figure 21. Area of Study for the Waterford River Bacterial Survey, summer 1983.

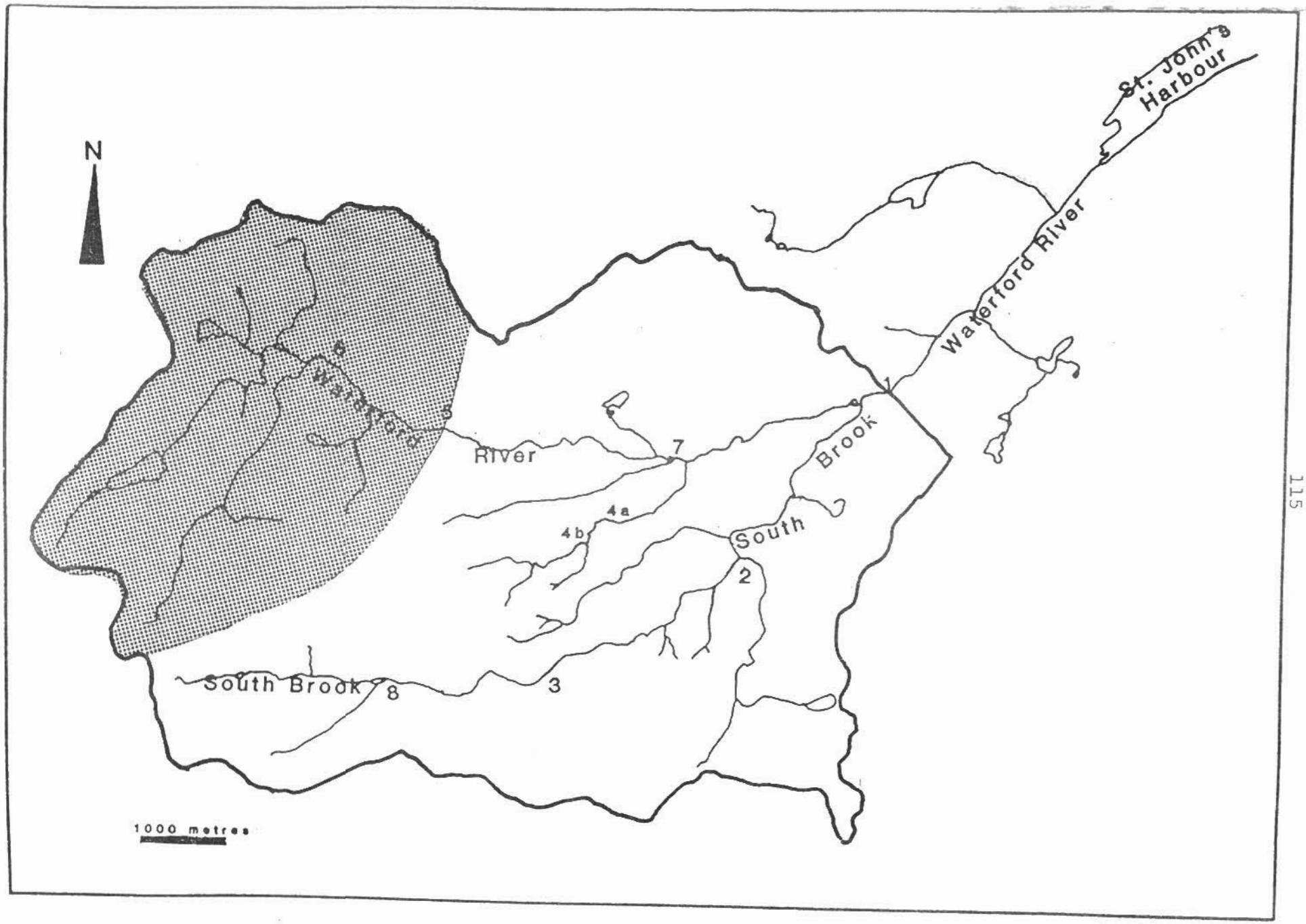

Figure 22. Area of Study for the Waterford River Bacterial Survey, summer 1984.

APPENDIX B

Review of Similar Studies on
Other Rivers in North America

1. A two year study done on the Shenandoah River basin in Virginia by Tackett in 1962 and 1963 concluded that residential and industrial waste was eliminating all but the most "hardy" species of invertebrates. This study used 23 sampling sites, selected for all the sewage and industrial waste discharges, along with some upstream control sites. The results of the study showed no improvement in water quality during the two years and the slight improvement in benthic invertebrate populations was probably due to the lower flow rates thus less scouring of the river bottom. The control sites had good populations of invertebrates but at the polluted sites even the "hardy" organisms were found in low numbers.

2. In 1965, Tackett of the Virginia State Water Central Board assessed the water quality of 'Roanoke River - Tinker Creek'. The study found that 1.4 miles of the 1.7 miles of Tinker Creek surveyed was significantly degraded and that 17.6 miles of the 22.6 miles of Roanoke River surveyed was significantly degraded. Many of the samples taken below pollution sources were completely devoid of invertebrates. Those site which contained invertebrates usually only had annelids and air-breathing snails.

3. The Lytle Creek drainage basin in Ohio was surveyed by Gaufin (1973). Unlike other studies Gaufin found even sensitive species in dirty water occasionally and thus concluded that for identifying deteriorating water quality the invertebrate community structure and population was more important than just the presence or absence of indicator species. Even slight changes in environmental conditions, he further concluded, if persistent, can lead to a significant change in community composition and diversity.

4. The effects of urban runoff in Whitemud Creek in Edmonton, Alberta was studied by Whiting and Clifford (1983). There were no sewer outlets entering the river just storm sewer outlets and the river was completely shaded. Within the city only chironomids and annelids were found in high numbers while outside the city the invertebrate community structure was very diverse. The cause of the deterioration in water quality and invertebrate diversity, they concluded, was excess nutrient enrichment from organic matter and silt from run-off.

As can be seen from the above reviews, the invertebrate community structure can be very useful in investigations of water quality, and monitoring of environmental changes over time.

APPENDIX C

Terms of Reference
of Original Study Concept

Proposed Biological Study Program
to assess Stream Pollution in the
Waterford River System

The following biological study is proposed as part of a larger, 5-year study to determine the effects of urban development on aquatic environment of the Waterford River System. Other components of the study will include physical, chemical, and bacteriological analyses of the water and the determination of biological oxygen demand.

1. Basic Design of Study

The study will involve regular sampling of benthic invertebrates and riverine flora at eight sites on the Waterford River system, and two sites on a suitable "control" river in the general vicinity of St. John's. Sampling will be done over three years: 1981, 1982, 1984, but a 2-year model (1981, 1984 only) is being considered as an alternative. The major emphasis will be on benthic invertebrates, for which quantitative sampling will be carried out at 10 sites, on seven designated dates for each of three years. Quantitative sampling of algal flora and qualitative sampling of aquatic macrophytes will occur twice per year. Quantitative sampling of fish populations (twice per year at 3 sites) will be considered as an optional component of the study.

2. Sampling Techniques and Schedule

2.1 Benthic Invertebrates

Sampling of benthic invertebrates to be carried out using Surber samplers (Surber, 1936) in standard fashion. At each site on each sampling date, ten (10) Surber samples to be taken. The exact locations of the samples to be planned in advance so that the effects of site disturbance from previous collection activities are minimized. All benthic invertebrates to be preserved in a manner appropriate for long term storage, regardless of the intended date of sorting.

All benthic invertebrates to be identified where feasible to the species level; where this is not possible, due to a lack of taxonomic information, every effort should be made to carry out identification to generic level. Counts for each sample are to be made for all taxa.

Sampling dates, for 1981, 1982, and 1984 are as follows;

- May 1
- May 15
- June 1
- June 15
- July 1
- September 1
- November 1

The eight sites on the Waterford River to be sampled on each of the above dates are shown in the accompanying map. The two sites on the control stream are in the process of being chosen, but the intention is that they will be of relatively easy access, within no more than about 30 minutes drive from any of the Waterford Site.

2.2 Aquatic Flora

Algae

Relative abundance of attached algae (identified to generic level) to be estimated from collections made twice per year (May 1 and July 1) of 1981, 1982, and 1984 at all sites.

Sampling to be carried out be removal (e.g., brushing) of periphyton from natural substrata (e.g., a standardized number of rocks of standardized size). Relative abundance of taxa determined at each site by counting (see Sladeckova, 1962, pp. 308-309). Where this is not appropriate, relative abundance on natural substrate may be assessed by estimating relative cover (Sladeckova, 1962, pp. 309-310).

Aquatic Macrophytes

Qualitative survey (i.e., species list) of aquatic macrophytes to be made at all sites on September 1 1981, 1982, 1984.

2.3 Fish

Electroseining (the use of barrier nets in conjunction with electrofishing apparatus) to be carried out at Site 1 and Site 3 of the Waterford, and at one site on the control stream on June 1, September 1 of 1981, 1982, and 1984. Counts and individual fork lengths to be given for all species encountered.

2.4 Other Information

At all sites on all sampling dates, the following information to be recorded: stream water temperature, pH and dissolved oxygen content. At the beginning of the study, a single point in mid-stream to be chosen so that on each sampling the stream depth and water current velocity (taken 60% of depth below surface) can be recorded. Current velocities should be recorded also in the immediate vicinity of the Surber sampling activites. Observations such as spates, ice-cover, turbidity, etc. should be made in a general way on each sampling occasion.

On one occasion, stream width and a description of substrate, particularly substrate size, to be made at each site.

3. Data Analysis and Interpretation

Analysis of data should include:

1. Designation of indicator species.

2. Use of biotic index which evaluates the compostion and diversity of the stream bottom communities with particular reference to indicator species.

Interpretation of the results should include:

1. Evaluation of the degree of pollution in the Waterford system, relative to the estimated conditions prior to urban development. The latter estimate is to be made using data from the control stream and any other pertinent information which may be available.

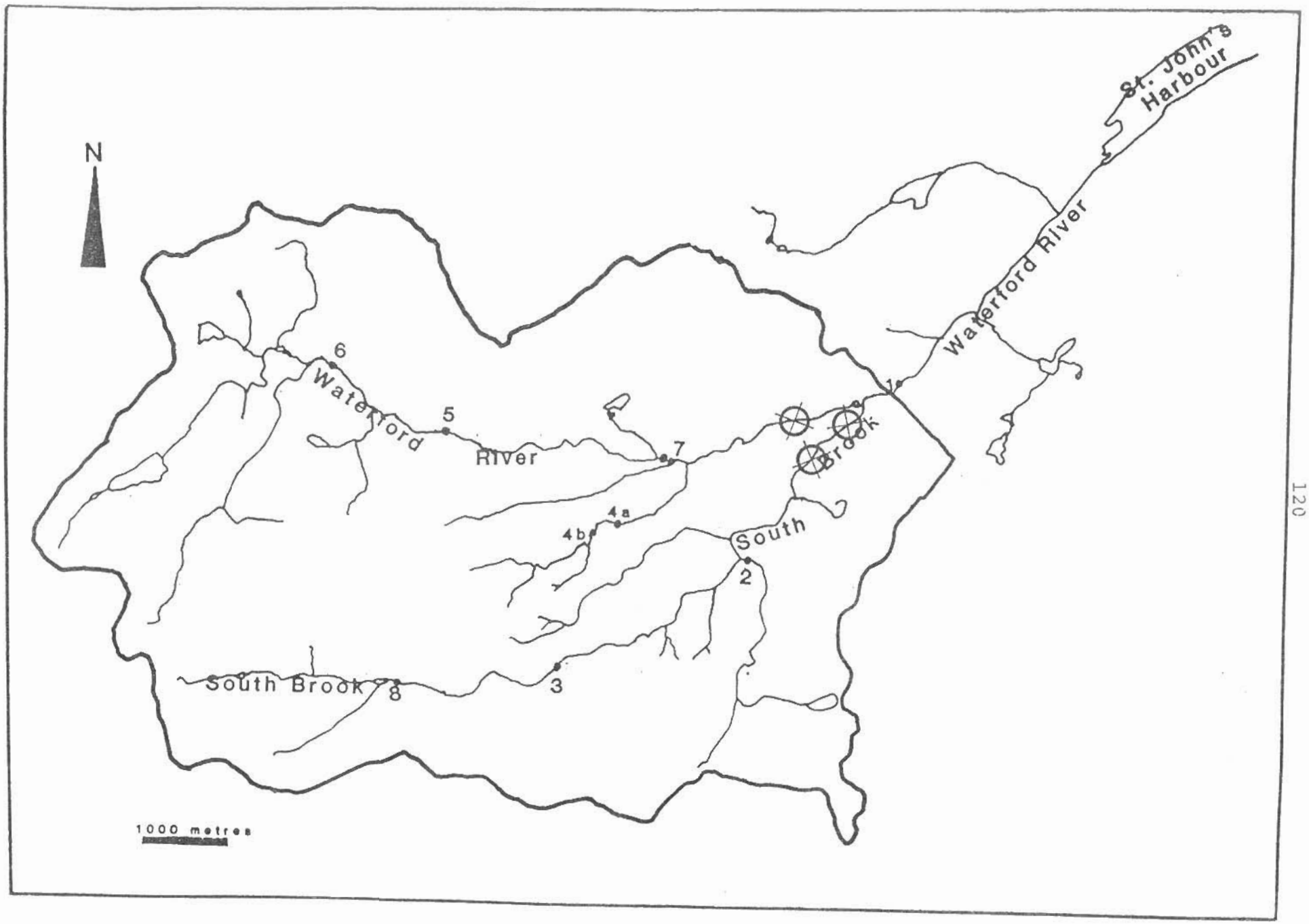

Figure 23. Proposed fish sampling locations.

2. Description of any changes in the stream environment which become apparent during the course of the study, and which may be the result of changing pollution conditions.

4. Options

For budgetary purposes, we request the following costing scenarios;

1.

   a. if only the processing and analysis of samples were done (in this case, all sampling activities per se would be carried out by Government).

   b. all work including sampling activites, processing and data analysis, as described.

2.

   a. total costs omitting one year of the study (1982); i.e., 2 years only.

   b. total costs for all three years.

3.

   a. including work on fish sampling described in section 2.3.

   b. excluding same

5. References

1. Sladeckova, A. 1962. Botanical Review 28:286-350.

2. Surber, 1936.

APPENDIX D

Review of Consultants Proposals:

The biological study was initiated in May 1981 with a call for proposals. Five consulting firms, Beak Consultants Limited Lavalin MacLaren Plansearch Limited, LGL Limited, Shawmont Newfoundland Limited, and Wildland Associates Limited submitted proposals according to the Terms of Reference (Appendix B) provided by the NDOE. A brief review of each proposal is presented below.

Proposal by Beak Limited

Although the terms of reference called for 10 surber samples per site, Beak proposed that only four surber samples be taken per site. The reason for reducing the number of Surber samples was that a large number of Surber samples would disrupt a large proportion of the stream substrate near each sampling site and would affect the abundance of animals collected during future sampling periods, and also to minimize the cost, especially identification cost. The collected invertebrates were to be preserved in 70% ethanol.

Beak suggested a restructuring of the sampling schedule because, firstly, the May 1 and May 15, 1981 sampling periods would have passed prior to awarding of the contract and, secondly, the benthic community changes seasonally in response to annual occurrences, such as spring run-off, and the emergence of aquatic insects.

Beak suggested that only a pre-determined number of samples should be identified to species level and that the other samples should be microscopically scanned for species not found in the completely processed samples. Counts of each sample were to be made for all taxa.

Beak proposed collecting algae using one sample per site by pooling the periphyton removed from several rocks (approximately five). The collected periphyton would be identified to species level.

The collecting schedule of aquatic macrophytes would be reorganized to coincide with

the flowering stages since the taxonomic keys rely heavily on floral parts. Detailed notes on the extent of growth, apparent health and the species dominance would be provided.

Fish sampling was to be carried out as outlined in the terms of reference using a Smith-Root type VIII electrofisher. Fish population to be estimated using the DeLury method or mark-recapture techniques. Assessment of the species present, size classes and relative abundance. Fish populations per unit area, fork lengths and weights will be determined for each species encountered.

Beak recommended, along with the other environmental variable outlined in the terms of reference, that the depth and current velocity be recorded three (3) times annually (spring and fall peak flows and summer low flow).

Data analysis will include evaluation of diversity indexes and doing a cluster analysis so that significant differences can be found due to sampling site and date.

Beak submitted several cost proposal options which ranged from a low of $280,000 for a two year study omitting the 1982 study, to a high of $441,027 for a complete three-year study including a fisheries study. The cost break-down was approximately 7% of the cost for sampling, 78% for analysis, and 15% for report writing.

Proposal by Lavalin MacLaren Plansearch Limited

Lavalin also proposed the use of four surber samples per site instead of ten. The collected invertebrates were to be preserved in 95% ethanol. The invertebrates were to be identified to species whenever possible.

Lavalin proposed collecting algae by using five rocks of uniform size treating each rock as a distinct sample. The algae was to be identified to the generic level, a quantitative survey (ie., species list) of the aquatic macrophytes was to be made at all sites on September l, 1981, 1982, and 1984.

Fish sampling would be done as outlined in the terms of reference along with counts and individual fork lengths for all species encountered.

Along with the various environmental variables measured as outlined in the terms of reference, Lavalin also proposed to survey the sampling sites and make maps showing the riffle and pool areas along with the contours of substrate types, current veolocity and water depth. There would be a qualitative assessment of substrate particle size on the Westworth scale. Lavalin was proposing to re-survey the stream each year of the study.

The data would be analyzed for in-site and between site differences using multivariate statistics and cluster analysis.

Lavalin cost proposals ranged from $113,208 for a two year study which omitted 1982, to $116,246 for the complete study as outlined in the Terms of Reference. Sampling would account for approximately 25% of the budget, analysis for 69% of the budget, and report writing would be 6% of the budget.

Proposal by LGL Limited

LGL proposed the use of artificial substrate samplers to both reduce within site variation and to preserve the stream bed. Three artificial substrate replicates per monitoring site were to be used. The exposure period for the samples should be four weeks and, as well, two annual sampling periods, one in later spring (early May to early June) and one in early fall (September to October), so as to maximize the number of species and individuals in recoverable, early age-class groups. LGL also proposed that South Brook site 3 be replaced for an additional site on the Waterford River downstream of site 1. Invertebrates collected to be identified to species whenever possible.

It was proposed to use artificial samplers or glass slides suspended in the water column to collect algae. Three samples were to be collected per site and the algae identified to the generic level. Aquatic macrophytes and fish were to be sampled as outlined in the terms of reference.

Environmental variables were to be measured as outlined in the terms of reference and the data was to be analyzed. LGL indicated that indicator species were of little use.

LGL Limited submitted budget proposals ranging from $73,478 for a two year study

which omits the 1982 program, to $103,092 for the complete three-year program. Approximately 27% of the budget would be for sampling, 59% for analysis, and 14% for report writing.

Proposal by Shawmont Newfoundland Limited

Shawmont proposed the use of only 2 surber samples per site instead of the original ten and the collected invertebrates to be preserved in 10% formalin and identified to the generic level. Genus is the largest taxonomic level for which reliable and reasonably priced analysis could be provided, analysis to species level would be somewhat invalid due to the ongoing restructuring of the taxonomy.

Algae was to be collected by scraping a 15 cm^2 section of 3 separate rocks per site and identified to the generic level. Aquatic macrophytes were to be qualititatively surveyed (ie., species list).

Shawmont presented the most detailed proposal for fish sampling. A 200m^2 or more area was to be blocked off using barren nets and an electrofisher (A Smith-Root electrofisher which has proven effective in the low conductivity waters of Newfoundland) used to stun the fish which were to be then anaesthetised using either MS222 or tertiary amyl alcohol. Shawmont proposed to sample at 6 sites once annually.

Environmental variables and data analysis was to be done as outlined in the terms of reference.

Shawmont Newfoundland Limited's budget for their proposal ranged from $91,370 for a two year study which excluded the fish sampling (processing and analysis of samples only), to $166,910 for a three year study with fish sampling (sampling activities, processing and analysis included). Sampling would account for 19% of the budget, sample and data analysis for 74% of the budget, and report writing for 7% of the budget.

Proposal by Wildland Associates Limited

Wildland's proposal was the consultants proposal which most closely matched the terms of reference with the exception that Wildland proposed a classification of each site as per the land/water intergration system and, that not only would the data be analyzed as outlined in the terms of reference but it would also be analyzed to check for in site and between site differences using multivariate statistics and cluster analysis.

The budget proposed by Wildland ranged from $38,490 for a two year study which did not include sampling activities, to $66,675 for the complete three study including sampling activities. Approximately 18% of the budget would be for sampling, 81% would be for sample and data analysis, and 1% would be for report writing.

Overview of proposals

Four of the five consultant firms suggested that 10 surber samples per site was too much for a variety of reasons. Three of the firms proposed reducing the number of surbers while another firm proposed the use of artificial substrate samplers. Invertebrates and algae would be identified to either genus or species level depending on the firm.

The different consultant firms proposed varying methods of data analysis from simply diversity indices to complex multivariate statistical analysis.

All the proposals would have achieved the objectives set out in the Terms of Reference. The proposal by Wildland Associates most closely matched the Terms of Reference for the study, with the exception of the sampling program. All the consulting firms, however, designed very grandiose proposals, both in scope and cost requirements.

APPENDIX E

Revised Biological Study Program
to Assess Stream Pollution in the
Waterford River System[1]

Overview

The following revised biological study is proposed as part of a larger, 5-year study to determine the effects of urban development on the aquatic environment of the Waterford River system. Effects will be determined by focussing on one component of the aquatic ecosystem, resident invertebrates.

Basic Study Design

The study will involve the placement of artificial substrates (rock bags) at eight (8) sites on the Waterford River system and two (2) sites on a suitable control river in the vicinity of St. John's. The substrates will remain in the river for 12 months and will be available for colonization by benthic invertebrates. They will then be retreived, all benthic invertebrates removed and preserved, and a new set of substrates replaced to be retreived the following year at the same time.

Artificial substrates have been used frequently in the assessment of water pollution. While samples collected do not truly represent the resident in number and diversity, they do provide an adequate variable for evaluation of changes in water pollution. The technique allows for the production of replicate samplings and statistical comparison of results from site to site, and between river systems.

Methods

1. Artificial Substrates

   - Rock Bags - The artificial substrates will consist of rock bags, made with 1/4" nylon tube mesh, containing one kilogram of washed, graded, 1/2"-3/4" crushed rock. The rock bags will be placed on the river bed

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[1]The sampling program was designed by S. Bonneyman, Environmental Biologist, Environmental Assessment Division, Newfoundland Department of Environment, on the recommendation of the Technical Committee of the Waterford River Basin Study.

and tethered to a stake to prevent downstream movement during freshets. Thirty bags will be placed at each sampling location, to insure that fifteen bags will survive to be collected after one year in the river.

- Bricks - Several clay bricks containing crevices suitable for benthic invertebrate colonization will also be placed at the sampling sites. The bricks will provide a back-up sampling system, in case the rock bags are washed away. The relatively high bed movement of the Waterford River necessitates this precaution.

2. Sample Preservation - All samples will be preserved in 70% ethanol in quart mason jars.

3. Sample Analysis - Preserved samples will be analyzed on a contract basis. The estimated cost of sorting and identification to the family level is $50./sample.

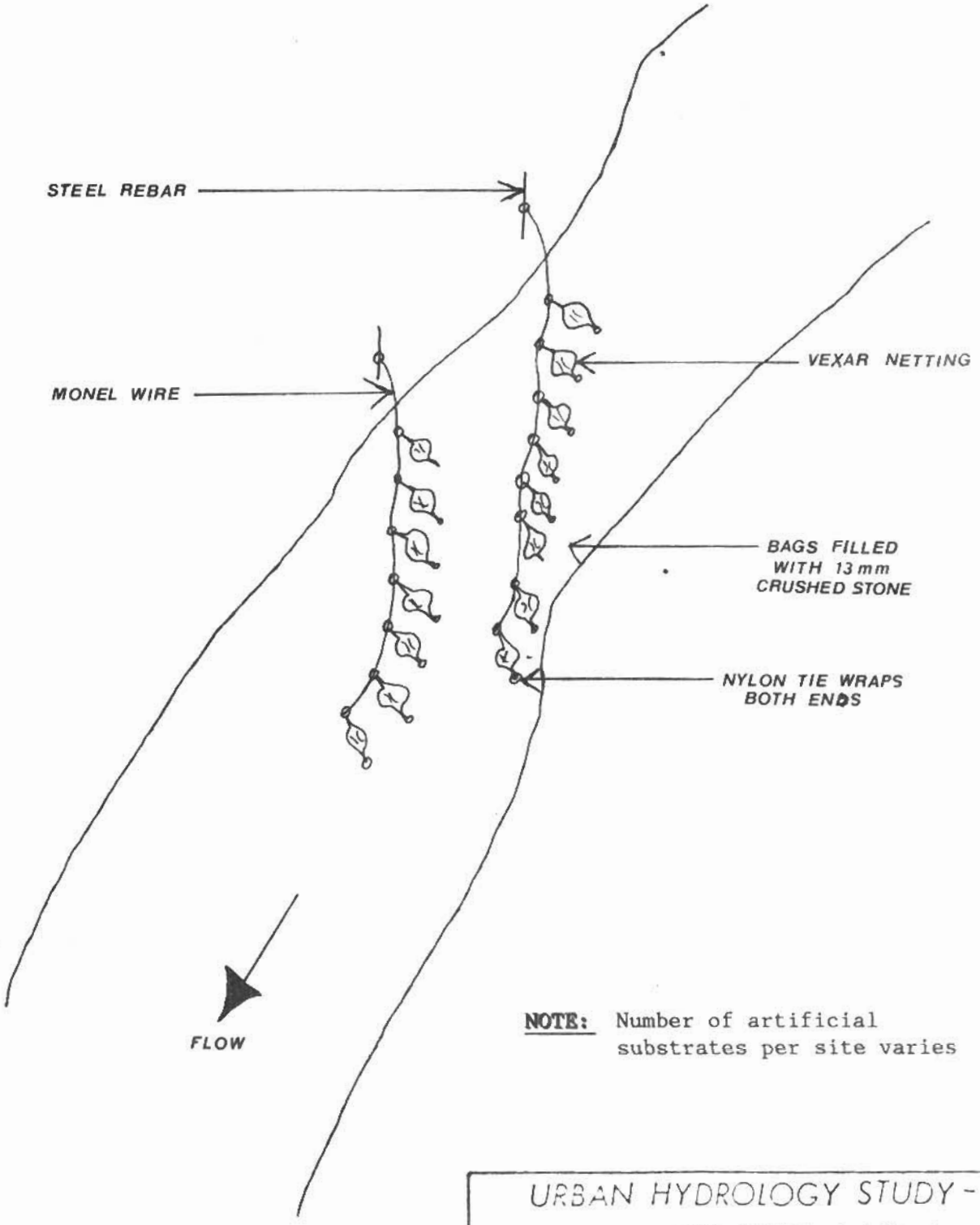
4. Data Analysis and Interpretation - Data analysis will include calculation of diversity indices, and their comparison from site to site, and with the two control rivers. Percent similarity indices will also be calculated. Invertebrates will be divided into "clean water species" and "unclean water species" based on available literature. This will allow for a qualitative evaluation of pollution levels down the river system. The same comparisons will be made between the within year data and the different years data.

APPENDIX F

Placement of Artificial Substrates at the Sites Sampled in this Study

The following diagrams (Figures 24 to 32) show the placement of the artificial substrate bags in the streams at the various sampling sites. Diagrams show important physical characteristics as well as the direction of flow of the stream, and the number of rock bags placed at each site.

Figure 24. Typical Placement of Artificial Substrates.



NOTE: Number of artificial substrates per site varies

URBAN HYDROLOGY STUDY - WATERFORD RIVER BASIN BIOLOGICAL SUBSTUDY

TYPICAL INSTALLATION OF ARTIFICIAL SUBSTRATES

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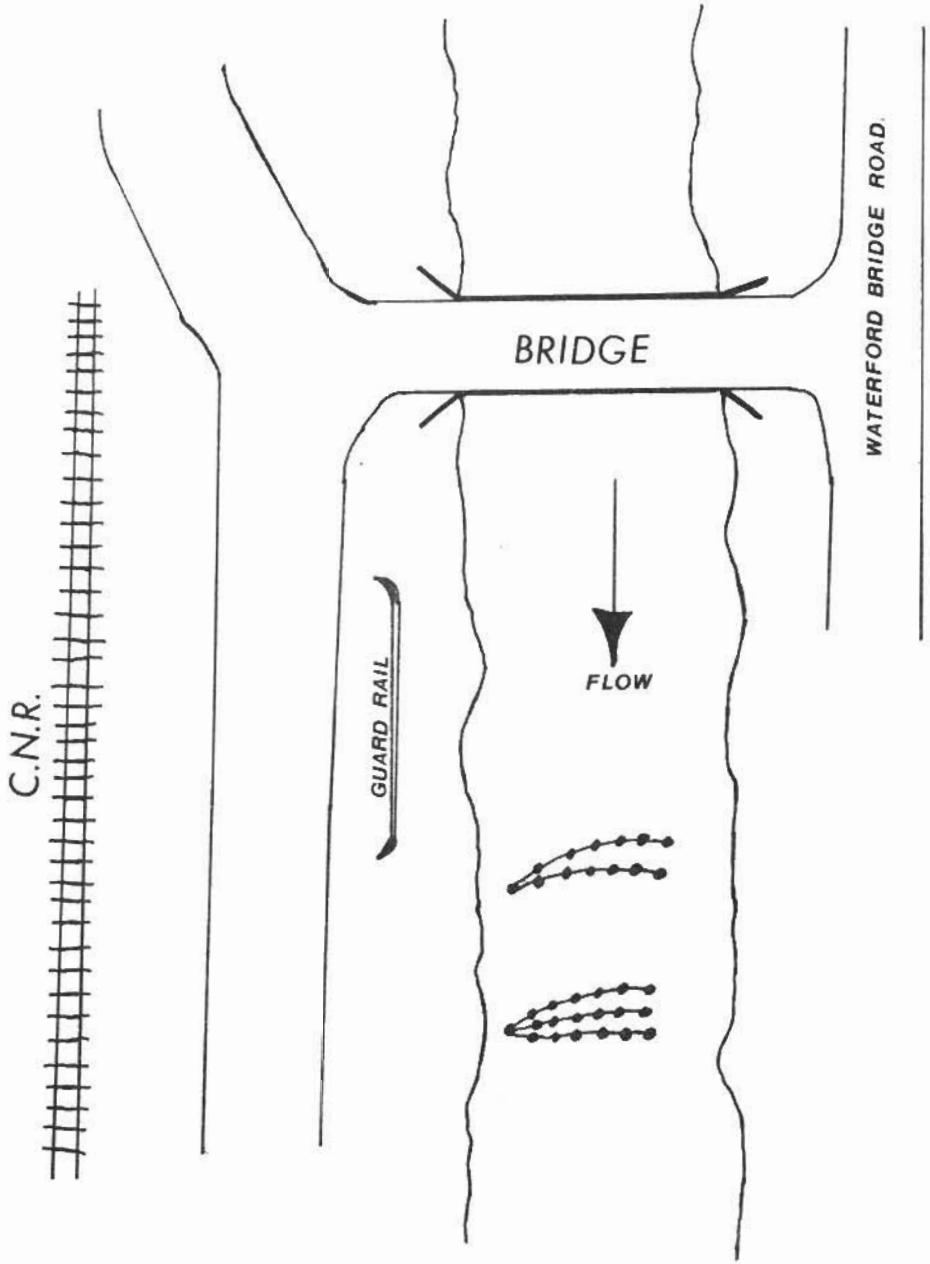
Figure 25. Placement of Artificial Substrates at Site 1.



NOTES:

- Date installed - Aug. 26, 1985
- No. of substrates - 30
- No. of rebar stakes - 2
- Naquadat No. - 00NF02ZM0009

URBAN HYDROLOGY STUDY - WATERFORD RIVER BASIN BIOLOGICAL SUBSTUDY

SITE 1

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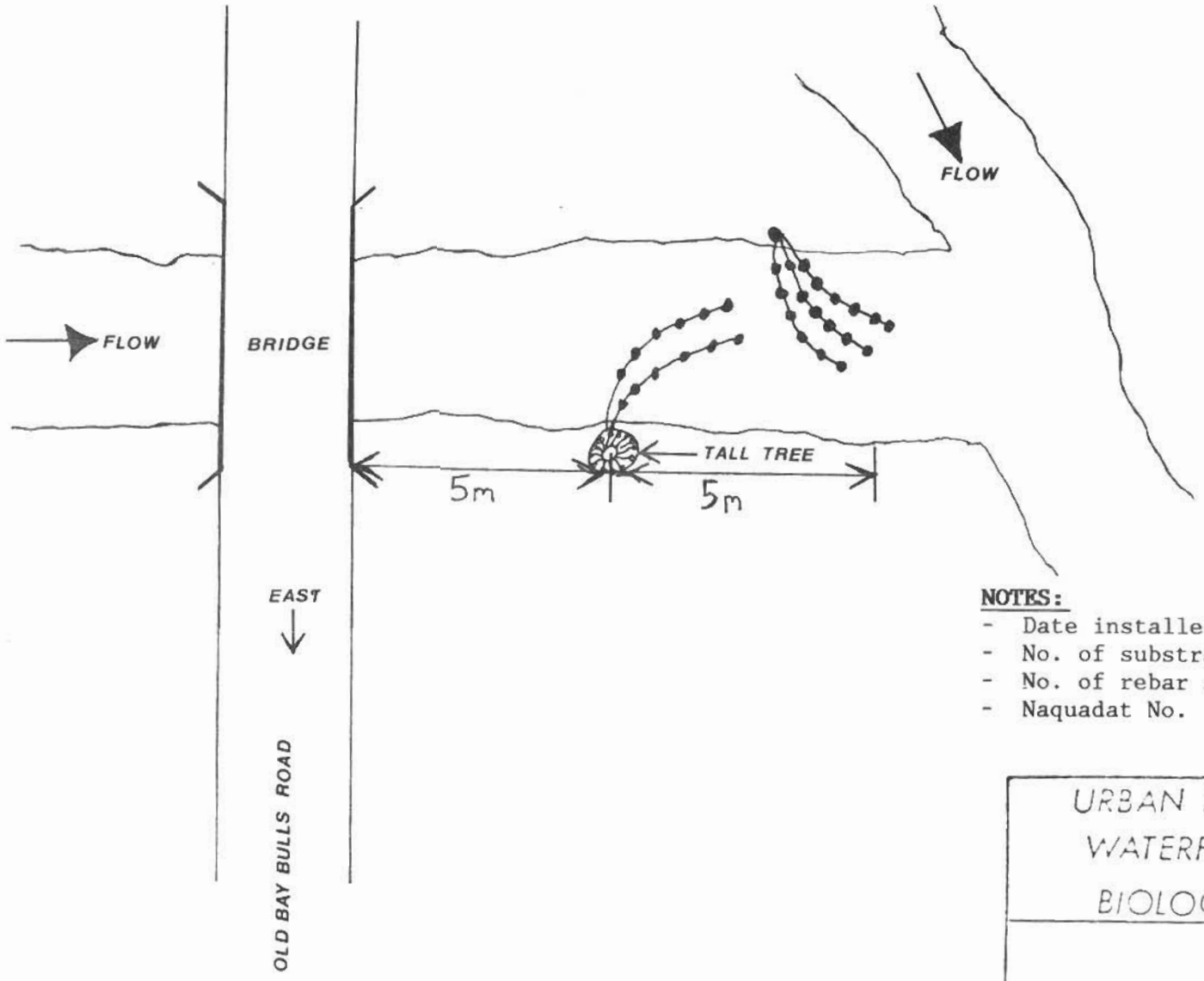
Figure 26. Placement of Artificial Substrates at Site 2.



NOTES:

- Date installed - Aug. 28, 1985
- No. of substrates - 30
- No. of rebar stakes - 1
- Naquadat No. - 00NF02ZM0008

URBAN HYDROLOGY STUDY - WATERFORD RIVER BASIN BIOLOGICAL SUBSTUDY

SITE 2

Scale:	Date	Drawn by:	Checked by:
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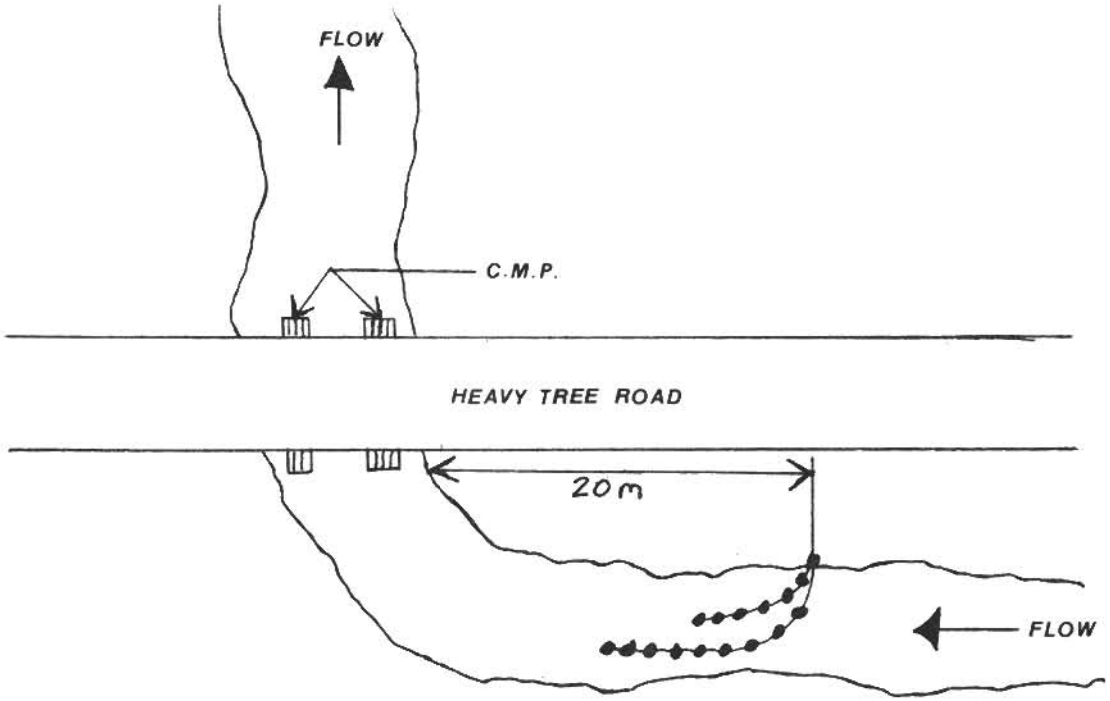
Figure 27. Placement of Artificial Substrates at Site 3.



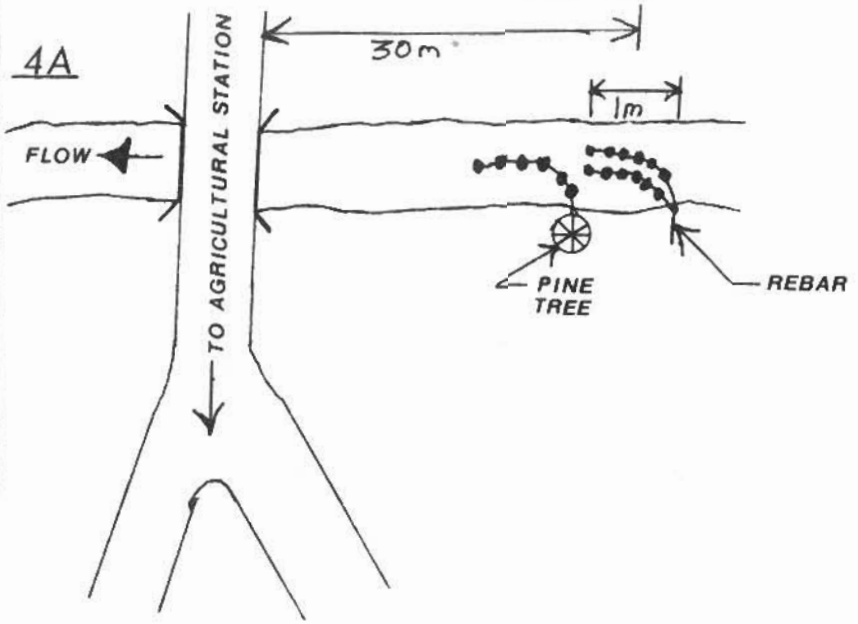
NOTES:

- Date installed - Aug. 28, 1985
- No. of substrates - 15
- No. of rebar stakes - 1
- Naquadat No. - 00NF02ZM0007

URBAN HYDROLOGY STUDY - WATERFORD RIVER BASIN BIOLOGICAL SUBSTUDY

SITE 3

Scale:	Date:	Drawn by:	Checked by:
NTS	28 Aug 85		



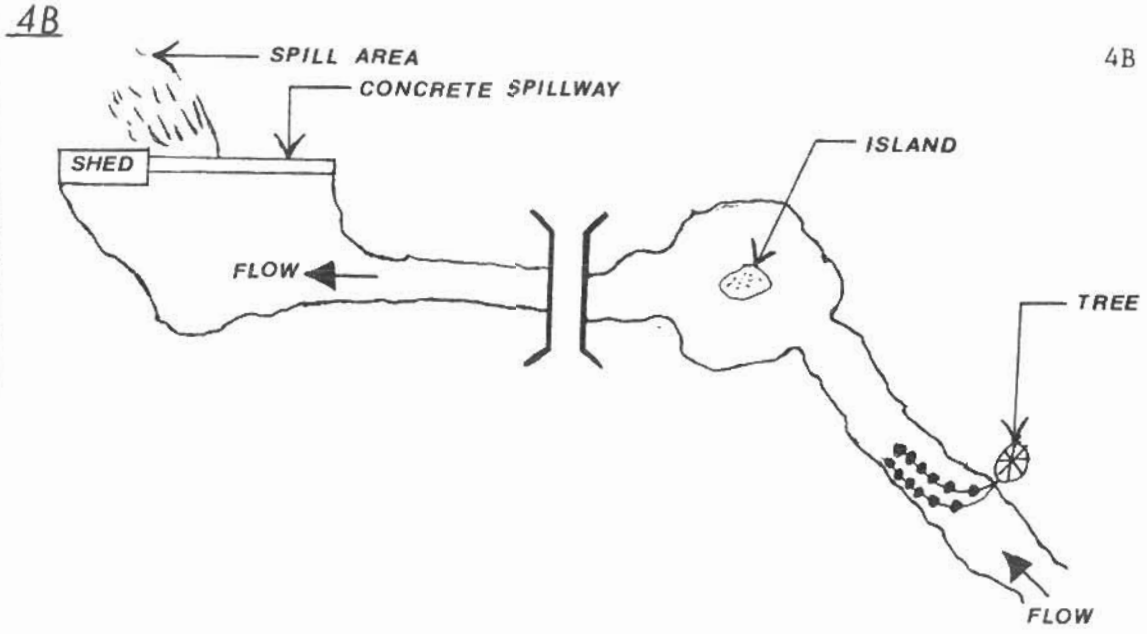


Figure 28. Placement of Artificial Substrates at Site 4A and 4B.

4A **NOTES:**

- Date installed - Aug. 28, 1985
- No. of substrates - 18
- No. of rebar stakes - 1
- Naquadat No. - 00NF02ZM0006

4B **NOTES:**

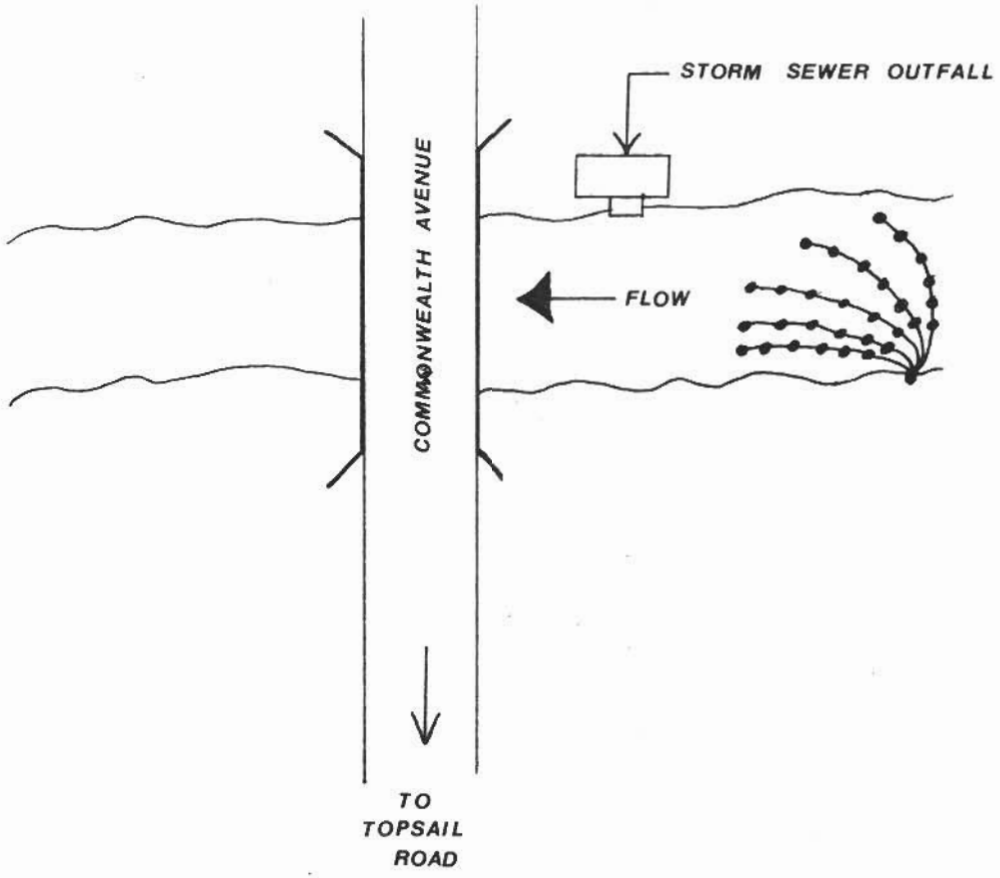
- Date installed - Aug 28, 1985
- No. of substrates - 12
- No. of rebar stakes - 0
- Naquadat No. - N/A

URBAN HYDROLOGY STUDY - WATERFORD RIVER BASIN BIOLOGICAL SUBSTUDY

SITE 4A & 4B

Scale:	Date	Drawn by:	Checked by:
NTS	28 Aug 85		

Figure 29. Placement of Artificial Substrates at Site 5.



NOTES:

- Date installed - Aug. 26, 1985
- No. of substrates - 30
- No. of rebar stakes - 1
- Naquadat No. - 00NF02ZM0004

URBAN HYDROLOGY STUDY - WATERFORD RIVER BASIN BIOLOGICAL SUBSTUDY

SITE 5

Scale:	Date:	Drawn by:	Checked by:
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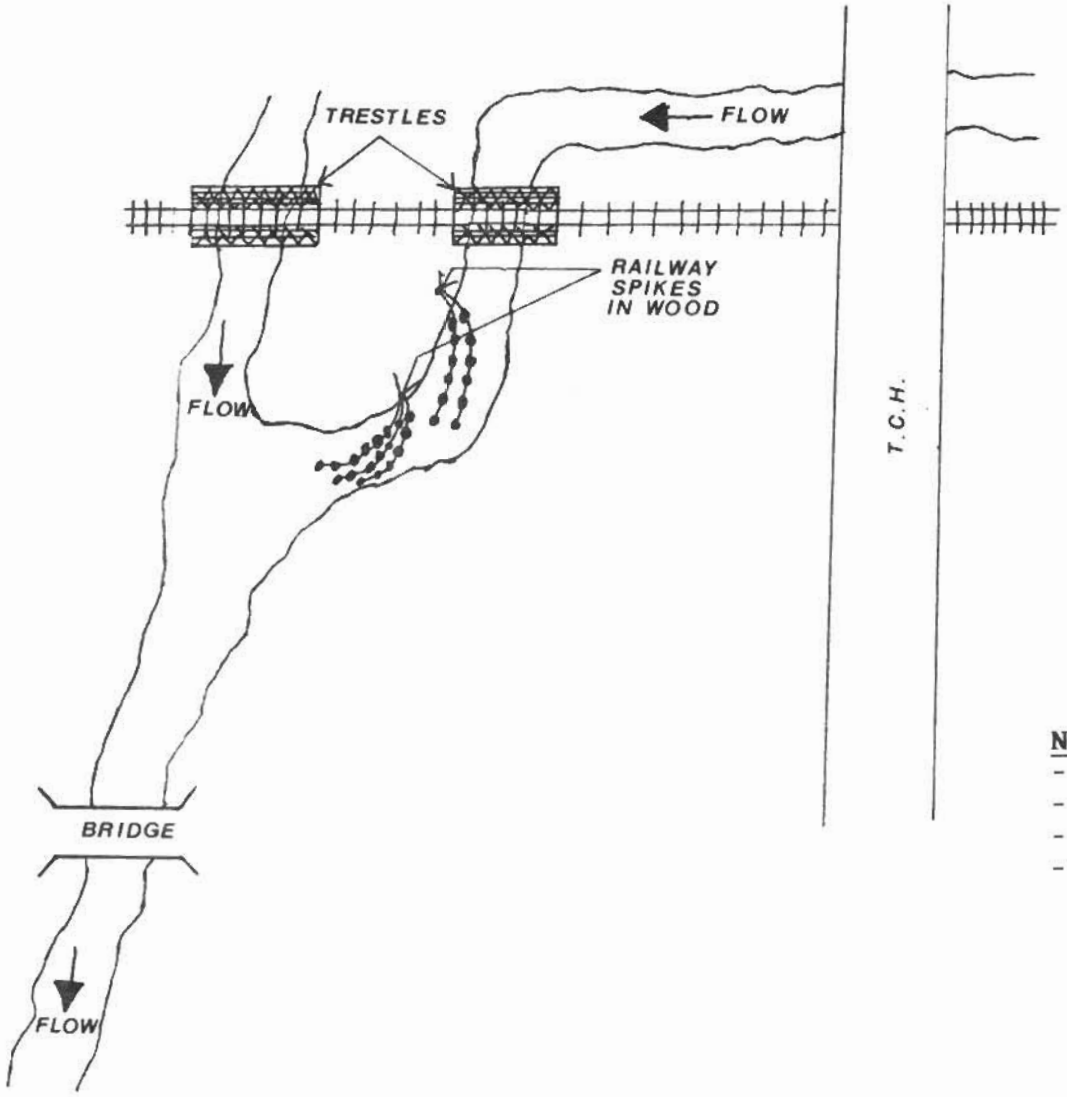


Figure 30. Placement of Artificial Substrates at Site 6.

NOTES:

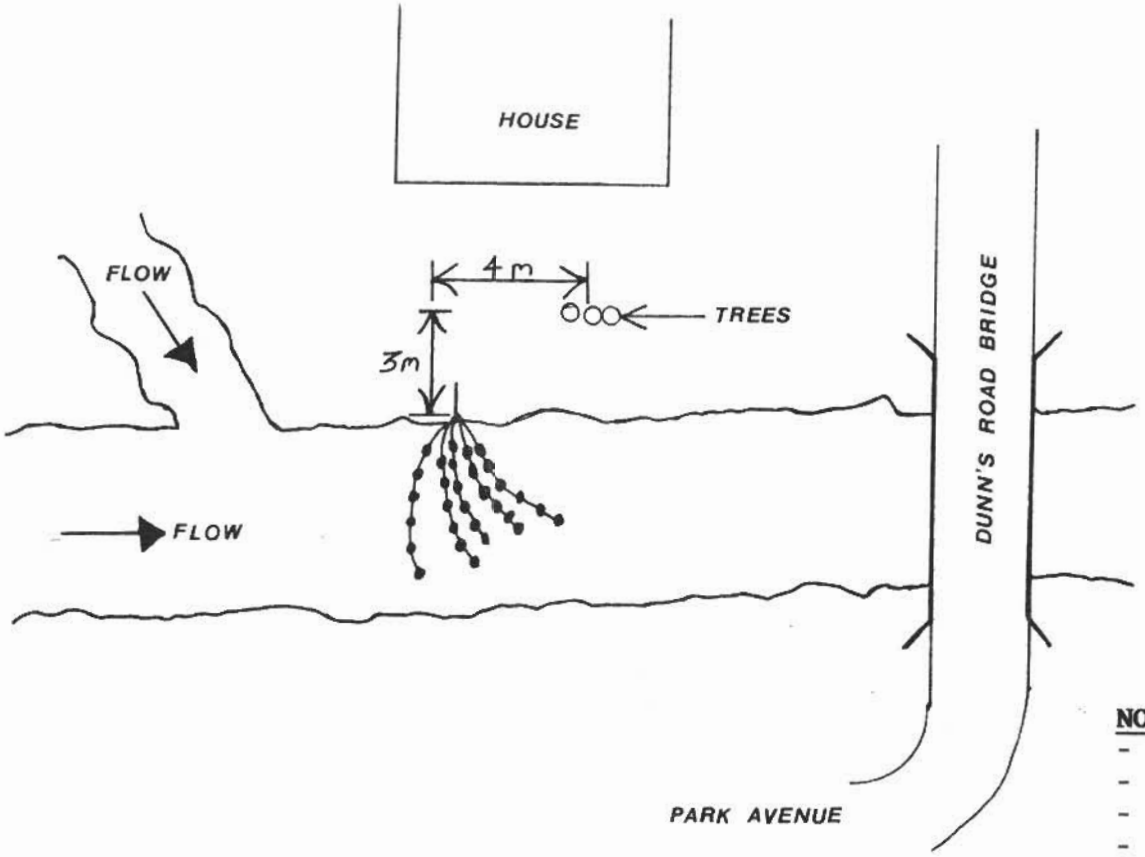
- Date installed - Aug. 26, 1985
- No. of substrates - 30
- No. of rebar stakes - 0
- Naquadat No. - 00NF02ZM0003

URBAN HYDROLOGY STUDY - WATERFORD RIVER BASIN BIOLOGICAL SUBSTUDY

SITE 6

Scale.	Date	Drawn by:	Checked by:
NTS	28 Aug 85		

Figure 31. Placement of Artificial Substrates at Site 7.



NOTES:

- Date installed - Aug. 26, 1985
- No. of substrates - 30
- No. of rebar stakes - 1
- Naquadat No. - 00NF02ZM0012

URBAN HYDROLOGY STUDY - WATERFORD RIVER BASIN
BIOLOGICAL SUBSTUDY

SITE 7

Scale:	Date	Drawn by:	Checked by:
NTS	28 Aug 85		KMR

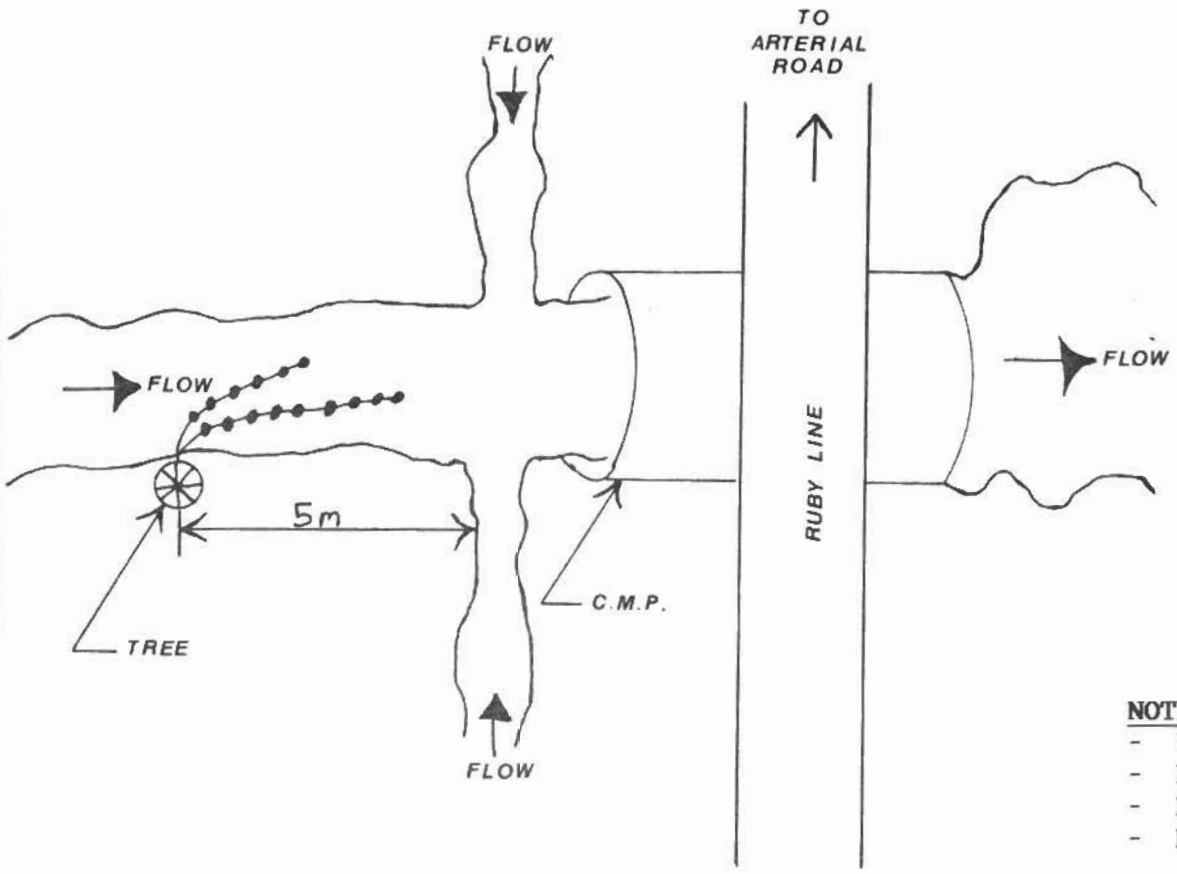


Figure 32. Placement of Artificial Substrates at Site 8.

NOTES:

- Date installed - Aug. 28, 1985
- No. of substrates - 15
- No. of rebar stakes - 0
- Naquadat No. - 00NF02ZM0001

URBAN HYDROLOGY STUDY - WATERFORD RIVER BASIN BIOLOGICAL SUBSTUDY

SITE 8

Scale:	Date	Drawn by:	Checked by:
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APPENDIX G

Results of Invertebrate Sampling in
the Waterford River system

Table A1. Non-insect/non-annelid taxa collected in the first year of study. (Mean, Std. Dev. and Range)

Site	Acarida	Amphipoda	Gastropoda	Pelecypoda	Nematoda
1	*	0	*	0	0
2	0	*	0	0	0
3	*	0	0	0	1.10 1.37 0-4
4a	0	0	0	0	0
5	0	0	0	*	*
6	0	0	0	0	*
8	*	0	0	0	*

* - Taxon present with mean <1 per sample.

Table A2. Annelida taxa collected in first year of study.
(Mean, Std. Dev. and Range)

Site	Earthworm	Enchytraeidae	Naididae	*Lumbriculus variegatus*	Hirudinea
1	*	*	*	2.30 1.83 0-5	0
2	*	1.00 1.25 0-3	1.70 3.68 0-12	2.60 3.06 0-9	*
3	*	*	*	1.20 2.70 0-8	*
4a	11.70 8.87 3-32	1.30 2.06 0-6	0	*	0
5	*	1.90 1.97 0-5	2.90 3.28 0-8	3.60 3.57 0-11	*
6	0	0	*	*	*
8	0	2.00 2.26 0-6	2.80 3.68 0-11	7.30 6.60 0-18	0

* - Taxon present with mean <1 per sample.

Table A3. Coleoptera taxa collected in first year of study.
(Mean, Std. Dev. and Range)

Site	*Hydroporus badiellus*	*Promoresia tardella*	*Stenelmis crenata*
1	0	1.70 2.21 0-6	0
2	0	*	0
3	0	13.40 9.73 2-31	*
4	0	0	0
5	0	0	0
6	0	0	0
8	*	10.90 8.50 0-25	0

* - Taxon present with mean <1 per sample.

Table A4

Diptera taxa collected in the first year of study.
(Mean, Std. Dev. and Range)

Site	Ceratopogonidae	Chironomidae	Empididae	Muscidae	Psychodidae	Simuliidae	Tabanidae	*Antocha* sp.	*Limonia* sp.	*Tipula* sp.
1	0	60.2 70.1 9-241	*	0	0	0	0	*	0	*
2	0	17.5 16.8 0-55	0	*	*	0	*	*	0	0
3	0	24.4 11.2 7-42	*	*	0	0	0	0	*	1.50 2.17 0-7
4a	0	7.50 4.95 1-18	0	*	0	0	0	0	0	1.70 1.83 0-5
5	*	22.2 21.1 2-72	*	*	0	0	0	0	0	*
6	*	5.50 6.24 0-18	0	0	0	0	0	0	0	1.40 1.17 0-3
8	1.70 1.34 0-4	10.7 6.02 1-22	*	0	0	*	*	0	0	1.00 1.05 0-3

* - Taxon present with mean less than 1 per sample.

Table A5

Ephemeroptera taxa collected in the first year of study.
(Mean, Std. Dev. and Range)

Site	Baetis flavistriga	Baetis pygmaeus	Baetis tricaudatus	Centroptilum convexum	Eurylophella sp.	Ephemerella subvaria	Habrophlebia vibrans	Leptophlebia cupida	Paraleptophlebia adoptiva
1	0	*	17.6 21.3 0-68	0	0	17.5 11.9 1-32	0	*	*
2	0	0	1.80 3.05 0-8	0	0	1.60 2.41 0-6	0	*	0
3	0	2.40 5.23 0-17	2.20 2.30 0-7	0	*	*	*	0	3.80 3.01 0-8
4a	0	0	0	0	0	0	0	0	0
5	0	0	2.90 2.51 0-9	0	0	37.2 22.0 4-68	0	*	0
6	0	0	*	0	0	1.20 1.55 0-4	0	*	0
8	*	*	*	*	0	0	3.70 5.66 0-16	*	*

* - Taxon present with mean less than 1 per sample.

Table A6. Plecoptera taxa collected in first year of study.
(Mean, Std. Dev. and Range)

Site	*Paracapnia opis*	*Leuctra ferruginea*	*Isogenus frontalis*	*Isoperla transmarina*
1	*	*	0	3.30 1.49 1-5
2	0	0	0	*
3	*	*	*	3.00 2.87 0-9
4	0	0	0	0
5	0	0	0	0
6	0	0	0	*
8	3.80 1.99 1-6	8.60 5.75 1-17	0	1.80 1.99 0-6

* - Taxon present with mean <1 per sample.

Table A7

Hydropsychidae and Rhyacophilidae collected in the first year of study.
(Mean, Std. Dev. and Range)

Site	Arctopsyche ladogensis	Hydropsyche betteni	Hydropsyche sparna	Hydropsyche slossonae	Rhyacophila carolina	Rhyacophila fuscula	Rhyacophila invaria	Rhyacophila minora	Rhyacophila torva	Rhyacophila vibox
1	0	*	16.6 14.1 1-46	20.3 22.3 1-76	0	*	0	0	0	0
2	0	1.60 1.71 0-4	19.5 21.0 0-61	13.5 15.6 0-47	0	0	0	0	0	0
3	*	0	6.10 4.95 1-18	*	*	*	*	*	0	0
4a	0	0	*	*	0	0	0	0	0	0
5	2.70 2.16 0-7	1.50 1.90 0-5	34.4 15.6 12-57	11.9 5.70 6-21	0	1.40 1.17 0-4	*	0	*	0
6	0	3.40 5.60 0-19	23.7 20.62 2-65	1.30 1.49 0-4	0	0	0	0	0	0
8	*	0	8.00 8.06 0-28	0	1.10 1.29 0-3	*	*	*	0	*

* - Taxon present with mean less than 1 per sample.

Table A8

Philopotamidae, Phryganeidae and Polycentropodidae collected in the first year of study. (Mean, Std. Dev. and Range)

Site	Chimarra aterrima	Dolophiloides distinctus	Wormaldia moesta	Oligostomis sp.	Pstilostomis sp.	Polycentropus centralis	Polycentropus cinereus	Neureclipsis sp.
1	0	0	0	0	0	0	0	0
2	0	0	0	0	0	0	0	0
3	6.30 5.96 0-14	*	0	0	0	*	*	0
4	0	0	0	0	0	0	0	0
5	0	0	0	*	*	0	0	*
6	*	0	0	0	0	0	0	0
8	2.30 4.54 0-14	*	*	0	*	0	*	*

* - Taxon present with mean less than 1 per sample.

Table A9

Remaining Trichoptera collected in first year of study.
(Mean, Std. Dev. and Range)

Site	Micrasema wataga	Glossosoma sp.	Hydroptila meteoca	Oxyethira sp.	Lepidostoma sp.	Platycentropus sp.	Pstilotreta frontalis
1	0	0	0	0	0	0	0
2	0	*	0	0	0	0	0
3	2.20 1.75 0-6	0	0	0	*	0	*
4a	0	0	0	0	0	0	0
5	0	0	0	0	0	0	0
6	0	0	0	*	0	*	0
8	*	*	*	*	*	0	*

* - Taxon present with mean less than 1 per sample.

Table A10. Non-insect/non-annelid taxa collected in the second year of study.

Site	Acarida	Gastropoda	Pelecypoda	Copepoda	Amphipoda	Nematoda
1	+	-	-	-	-	-
2	-	+	-	+	+	+
3	+	-	-	-	-	-
4a	+	+	-	-	-	-
4b	+	-	-	-	+	-
5	-	+	+	-	-	-
6	-	-	-	-	-	-
7	+	-	-	-	-	-
8	+	-	-	-	-	-

+ Present with mean <1 per sample
- Not present

Note: No non-insects were found with mean =>1 per sample.

Table A11. Annelid taxa collected in second year of study.
(Mean, Std. Dev. and Range)

Site	Lumbricidae	Enchytraeidae	*Nais communis*	*Stylodrilus heringianus*
1	*	*	*	0
2	0	0	*	0
3	0	0	*	0
4a	2.70 3.234 0-8	*	0	0
4b	0	0	0	0
5	*	0	0	0
6	*	0	0	0
7	0	*	5.20 8.404 0-25	*
8	0	0	*	0

* - Taxon present with mean <1 per sample.

Table A12. Coleoptera and Odonata taxa collected in the second year of study. (Mean, Std. Dev. and Range)

Site	*Promoresia tardella*	*Hydroporus parsus*	*Hydrobia fuscus*	*Aeshna sp.*
1	*	0	0	0
2	0	0	0	*
3	5.30 5.926 1-17	0	0	0
4a	*	0	0	0
4b	*	0	0	*
5	0	0	0	0
6	*	0	0	*
7	0	*	*	0
8	7.50 4.089 1-12	0	0	0

* - Taxon present with mean <1 per sample.

Table A13. Diptera taxa collected in the second year of study.
(Mean, Std. Dev. and Range)

Site	*Tipula* sp.	*Antocha* sp.	Chironomidae	Empididae	Muscidae	Ceratopogonidae	Simuliidae	Tabanidae
1	*	*	96.1 61.954 32-237	*	*	*	0	0
2	0	0	97.4 46.275 24-157	0	0	0	*	0
3	*	0	29.1 26.350 4-73	*	0	*	5.0 4.830 0-14	0
4a	2.8 2.044 0-7	0	36.9 18.823 11-67	0	*	0	0	*
4b	0	0	38.4 27.285 16-91	0	0	0	*	0
5	*	0	63.4 50.794 20-173	*	*	0	0	0
6	0	0	16.8 8.741 9-33	*	0	0	0	0
7	*	*	52.9 27.111 19-108	*	*	0	0	*
8	1.1 0.568 0-2	0	17.1 7.264 2-26	*	0	*	4.4 3.806 0-13	*

* - Taxon present with mean <1 per sample.

Table A14. Ephemeroptera taxa collected in second year of study.
(Mean, Std. Dev. and Range)

Site	Baetis tricaudatus	Baetis pygmaeus	Baetis flavistriga	Ephemerella subvaria	Paraleptophlebia adoptiva	Leptophlebia cupida	Eurylophella spp.	Habrophlebia vibrans
1	17.0 16.613 2-46	*	0	32.3 29.721 8-90	1.7 2.316 0-5	*	0	0
2	33.8 37.732 0-110	0	0	9.7 6.717 0-21	*	22.6 22.911 3-76	*	0
3	8.3 6.129 2-22	1.1 1.197 0-4	0	2.2 1.135 0-4	2.3 3.561 0-12	0	*	0
4a	1.2 1.229 0-4	0	0	*	0	2.4 3.307 0-10	0	0
4b	8.2 3.048 5-14	0	0	*	*	11.7 6.075 2-23	0	0
5	13.2 9.496 3-33	0	0	7.5 5.083 1-15	0	*	0	0
6	13.7 11.710 1-33	0	*	4.5 4.972 0-14	0	*	*	0
7	*	0	0	9.3 4.057 6-20	0	2.7 3.057 0-10	0	0
8	6.9 5.990 2-20	*	0	0	*	*	*	*

* - Taxon present with mean <1 per sample.

Table A15. Plecoptera taxa collected in second year of study.
(Mean, Std. Dev. and Range).

Site	*Isoperla transmarina*	*Isogenus frontalis*	*Paracapnia opis*	*Leuctra ferruginea*
1	25.9 28.081 5-98	*	0	0
2	1.2 1.687 0-4	0	0	0
3	4.0 2.828 0-8	1.0 1.633 0-5	*	*
4a	0	0	0	0
4b	1.6 1.506 0-4	*	0	0
5	*	*	0	0
6	0	0	0	0
7	1.5 1.509 0-4	0	*	0
8	3.8 3.765 0-11	1.0 1.247 0-4	*	9.4 5.562 2-18

* - Taxon present with mean <1 per sample

Table A16 Hydropsychidae, Hydroptilidae and Philopotamidae collected in the second year of study. (Mean, Std. Dev. and Range)

Site	Arctopsyche ladogensis	Hydropsyche betteni	Hydropsyche sparna	Hydropsyche slossonae	Hydroptila meteoca	Oxyethira sp.	Chimarra aterrima	Dolophiloides distinctus	Wormaldia moesta
1	0	*	21.4 12.176 6-40	57.7 32.375 11-116	0	0	0	0	0
2	0	1.8 3.011 0-10	31.2 17.574 4-61	65.5 33.752 13-108	0	0	*	0	0
3	*	3.3 4.832 0-16	17.4 20.495 0-66	33.5 27.464 2-79	*	*	1.9 3.376 0-12	*	*
4a	0	1.7 2.497 0-8	*	*	0	0	0	0	0
4b	0	17.6 8.168 5-33	25.3 14.967 7-51	17.2 21.348 0-64	0	*	0	0	0
5	*	*	20.3 9.719 2-33	88.2 41.360 26-146	0	0	0	0	0
6	0	2.6 5.522 0-18	47.7 24.789 3-76	15.8 13.522 1-46	0	*	0	0	0
7	0	*	15.2 4.590 8-22	50.0 17.988 25-76	0	0	*	0	0
8	*	*	6.9 5.607 0-17	1.6 3.239 0-10	0	*	*	*	5.9 13.568 0-44

* - Taxon present with mean less than 1 per sample

Table A17

Rhyacophilidae and Phryganeidae collected in the second year of study.
(Mean, Std. Dev. and Range)

Site	Rhyacophila carolina	Rhyacophila fuscula	Rhyacophila melita	Rhyacophila minora	Rhyacophila nigrita	Rhyacophila torva	Rhyacophila vibox	Hydatophylax argus	Limnephilus sp.	Pychnopsyche sp.
1	*	1.5 1.434 0-4	0	0	0	0	0	0	0	0
2	0	0	0	0	0	0	0	*	0	0
3	*	1.1 1.370 0-3	0	*	0	*	0	0	0	0
4a	0	0	0	0	0	0	0	*	*	0
4b	0	*	0	0	0	0	0	1.0 0.816 0-2	0	*
5	*	5.0 3.197 1-10	*	0	0	0	0	0	0	0
6	0	*	0	0	0	0	0	0	0	0
7	*	0	0	0	0	*	0	*	0	0
8	0	1.4 1.506 0-5	0	*	*	*	*	0	0	0

* - Taxon present with mean less than 1 per sample.

Table A18. Other Trichoptera collected in second year of study.
(Mean, Std. Dev. and Range)

Site	*Lepidostoma* sp.	*Micrasema* sp.	*Mystacides sepulchralis*	*Glossosoma* sp.	*Oligostomis* sp.	*Pstilostomis* sp.	*Polycentropus*
1	0	*	0	*	0	0	0
2	0	0	0	*	1.1 1.101 0-3	*	*
3	1.1 1.969 0-6	5.6 5.275 0-18	0	0	*	0	0
4a	0	0	0	0	0	0	0
4b	0	0	0	0	0	0	*
5	0	0	0	0	0	0	0
6	0	0	0	0	0	0	0
7	0	0	*	0	0	0	0
8	0	*	0	0	0	0	*

* - Taxon present with mean <1 per sample.