Fish-Benthos Correlations and Effects on Benthos that Reflect Significant Effects on Fish Communities in Southern Ontario Streams

by

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ABSTRACT

Fish-Benthos Correlations and Effects on Benthos that Reflect Significant Effects on Fish Communities in Southern Ontario Streams

This thesis attempts to determine the types of effects on benthic community composition that can be considered to have ecological consequence. The approach was based on the philosophy that changes in benthos are of little significance unless they coincide with unacceptable changes in fishery resources. The derivation of critical benthic community effect sizes required four components. First, a definition of unacceptable change in a fishery resource. Second, selection of some aspect of fishery resources that could be considered important to prevent damage to. Third, a demonstration that benthic community composition was inherently related to relevant descriptors of the condition of a fishery. And finally, specific relationships between benthic community composition and the fishery descriptor, from which critical benthic effect sizes could be determined.

To define an unacceptable change in a fishery resource, I adopted the use of normal ranges which are inherent in most goal-setting forums. Historically, two problems with the use of normal ranges has been in the application of normal ranges to multivariate descriptors, and in the application of appropriate statistical tests for comparing "impact" locations against the normal range. I therefore provide an operational definition for the normal range that simplifies the use of normal ranges in hypothesis testing. I also demonstrate three statistical tests that can be used to determine when a point observation falls outside of the normal range. The consequences of using conventional two-sample contrasts in relation to one-sample non-central equivalence and interval tests are determined. For sites that are truly outside of the normal range of variation for reference locations, equivalence tests will lead to erroneous conclusions of no impact at most 5% of the time. Two-sample contrasts will lead to erroneous conclusions of no impact about 50% of the time with low sample sizes (i.e., 10-20 reference locations). In contrast to both the two-sample contrast and the equivalence test, interval tests fail to recognize sites as being impacted unless impacts are in excess of about 3 σ from the reference population average (with a reference sample size of 20). Finally, the penalty for using the equivalence test is that

it will fail up to 26% of sites that are truly members of the reference population. Practitioners contrasting non-random impacted locations against a set of reference locations should consider these characteristics of the various potential tests when deriving conclusions of impact.

Fish community data from 37 streams, variously affected by agriculture, urbanization and impoundments, were used to determine the statistical power of various descriptors of fish community composition. In general, single-pass estimates of biomass of all species, and using a multivariate approach to describing the community, provided more modelling and statistical power. In contrast, measuring only the abundances or biomasses of individual (preferred) species like brook trout, was a less powerful method (statistically) for characterizing the fishery of a stream, and was less useful for modelling purposes.

To demonstrate consistent associations between fish and benthos, I used three independent data sets in which fish and benthos were collected across stream size and temperature gradients. These data showed consistent and strong associations between stream fish and benthos community composition. The data also suggested that more detailed benthic taxonomy resulted in stronger associations between fish and benthos, and provided evidence that the fish-benthos association is strongly driven by coincident association of fish and benthos with environmental conditions.

Finally, critical benthic effect sizes were determined for 11 combinations of data sets to explore sources of variation in derived critical values. In general, benthic critical values that coincided with ecologically relevant effects on fish were less than the normal range of variation for benthic communities, but varied with factors such as the type of benthic sampling apparatus, the microhabitat benthos were collected from, and the nature of the study design (i.e., whether it incorporated regional-reference or site-specific reference locations). Although the fish-benthos correlations were significant regardless of the level of benthic identification, identifying benthos to species resulted in more accurate prediction of the condition of fish communities. In addition, the reliability of benthos was higher when the reference fish community was a brook trout community, or when the assessment was site specific. In both situations, changes in fish communities from reference to impacted conditions were abrupt.

The results from this study have obvious application to aquatic environmental assessments, particularly stream assessments in southern Ontario. Benthic criteria, such as those derived here,

should be applied in association with relevant statistical procedures, primarily non-central interval or equivalence tests. These tests will provide exact probabilities that benthic community composition in impacted locations exceeds the benthic critical value.

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CHAPTER 1

General introduction

1.0 Overview of Monitoring with Benthos

Aquatic Environmental Effects Monitoring (AEEM) programs are now in widespread use in Canada (Hodson et al., 1996; DFO and EC, 1995a,b) to monitor and assess aquatic environmental conditions and their responses to anthropogenic activities. Such AEEM programs typically include surveys of fish, benthic and algal communities, as well as any of several abiotic measurements of system condition. Benthic macroinvertebrates are a preferred monitoring tool for assessing the condition of aquatic resources because they: (1) are relatively sedentary and thus exposed continuously to local conditions (Klemm et al., 1990); (2) have lifespans long enough to provide a record of environmental quality (Pratt and Coler, 1976); (3) are differentially sensitive to pollutants of various types, react to them relatively quickly (Cook, 1976), and provide graded responses to various stressor intensities (Pratt and Coler, 1976); (4) are relatively easy to collect and can usually be found in large numbers (Plafkin et al., 1989); taxonomy is well established; and (6) expertise in this area is common (Plafkin et al., 1989). Also given the high number of taxa that might be found at any one location or time, the probability that at least some taxa will respond to a stressor is high (France, 1990).

Given these considerations, it is not surprising that surveys of benthos are a common means of characterizing ecosystem condition in Canadian waters (Rosenberg and Resh, 1993; Griffiths, 1993; DFO and EC, 1995a,b). However, the use of benthos as part of AEEM programs is not without some conceptual difficulties. Of primary importance to the objective use of surveys of benthos is an understanding of the magnitude of change in benthic community composition that can be tolerated or accepted. In the design of environmental monitoring programs, it is difficult to justify sample allocation unless some specific acceptable effect size (i.e., degree of impact) has been specified *a priori* (Mapstone, 1995). The answer to this question of critical effect sizes, can in part be answered if we know why it is we monitor benthos, and what it is about benthos we are really interested in.

Although benthos are integral in energy flows (and other processes) of aquatic ecological systems, it is unclear whether the majority of practitioners or academics consider impacts on benthos to be of concern (e.g., CPPA, 1993). Since surveys of benthos are often rationalized on the basis that they are surrogate measures for ecosystem health (Kerans and Karr, 1994; DeShon, 1995; Yoder and Rankin, 1995) or surrogate measures for the condition of fish or fish habitat (Kreutzweiser, 1990; EC and DFO, 1992a,b; Hodson et al., 1996; Stanfield et al., 1996), it is apparent that a significant fraction of aquatic scientists do not consider effects on benthos to be of primary interest. Instead, the protection of ecosystem health and fish condition appear to be of more primary concern.

Cairns et al. (1993) recognize two types of environmental indicators and suggest that the term *compliance indicator* be used to describe those ecosystem characteristics that are of ultimate interest, while the term *early-warning indicator* be used to describe those characteristics that are merely surrogate responses for those ecosystem characteristics that are of ultimate interest. Herein, I use the term compliance indicator to refer to those characteristics of the aquatic ecosystem that are protected by legislation, while I use the term *surrogate* to refer to those ecosystem characteristics, like benthic communities, that are of secondary interest, but may be of significant value in terms of providing early warning of impending non-compliance through monitoring. In terms of using benthos in AEEM programs, a timely exercise would therefore be to determine how benthos can be related to compliance indicators and to determine the types of effects on benthos that correspond with unacceptable effects on relevant compliance indicators.

1.1 The *Fisheries Act*

As briefly mentioned above, surveys of benthic community composition are often used because they are implied surrogate measures of ecosystem health and the condition of fisheries resources, with ecosystem health and the condition of a fishery considered the compliance indicators. If this is true, then calibration of benthic community composition to either descriptors of ecosystem health or to measures of the condition of fisheries, would seem a logical step. Doing this would confirm that surveys of benthos can be used to monitor either ecosystem health or the condition of a fishery.

Of these two endpoints, ecosystem health is a more ambiguous term that has been variably

defined (Karr et al., 1986; Rapport, 1990; Schaeffer et al., 1988; Costanza, 1992) and indexed (e.g., Karr, 1981, 1993a,b; Costanza, 1992; DeShon, 1995). Such indices, were they defensible, could be used to calibrate the responses of benthic communities. Suter (1993), one of the main critics of the use of ecosystem health as an operational term, however, argues that because ecosystem health and associated indices are derived using circular logic, they are of limited practical application. Indices of ecosystem health are usually derived through calibration of the index at a series of sites deemed either "healthy" (i.e., acceptable) or "unhealthy" (unacceptable). Usually such designations of health status of a site are imposed by the investigator rather than by having an independent measurement of health status. In addition, the indices themselves are usually derived using subjective criteria (e.g., Karr, 1981, 1993a,b; Steedman, 1988). During the calibration process, it is inevitable that index scores reflecting good condition will occur at sites which are deemed by the investigator to be good sites (and *vice versa*).

In contrast, calibrating benthic community responses to descriptors of the condition of fishery resources has much more obvious value for two reasons: First, fisheries-related endpoints are, in themselves, generally considered to be of at least societal value (Karr, 1981; Scrimgeour and Wicklum, 1996). Second, fisheries-related endpoints have obvious relationship with the Canadian *Fisheries Act*. By calibrating benthic community responses to relevant fisheries-related endpoints, one could effectively use surveys of benthic community composition to make statements regarding the condition of fish or fishery resources, i.e., the degree of compliance with the *Fisheries Act*.

The *Fisheries Act* has a general goal of having "*no net loss of the productive capacity of fish habitat*" (Stoneman et al., 1996). Several endpoints have been proposed as potential monitors of the condition of fishery resources, including: (1) toxicity tests of potentially deleterious compounds (Environment Canada, 1992a,b); (2) habitat surveys (Lanno and Wren, 1992; Stoneman et al., 1996); (3) fish community surveys (Ontario Ministry of Transportation [MOT], 1994); (4) fish population (i.e., adult fish surveys proposed by EC and DFO, 1992a,b); and (5) benthic community surveys (EC and DFO, 1992a,b). Of these, surveys of fish community composition have the most obvious relationship with the *Fisheries Act* because they actually result in characterizations of the condition of fish resources. The other endpoints are

used primarily as surrogates, with some exceptions. Fish population surveys have obvious application to the *Fisheries Act* when they focus on species of commercial, game or societal value. However, most of the applications of fish population surveys (i.e., adult fish surveys) focus on species of limited (or no) commercial, game or societal value (e.g., Munkittrick and Dixon, 1989; Goede and Barton, 1990; Munkittrick, 1992; Gibbons and Munkittrick, 1994; DFO and EC, 1995a). As a result, demonstrating that there has been a change in the demography of such non-valued species is often questioned (e.g., CPPA, 1993). To be of general use, such surrogate measures require calibration with fisheries-related compliance indicators (endpoints considered more interesting) (Cairns et al., 1993). Such calibration allows one to make predictions on the condition of the compliance indicator after obtaining information on the surrogate response only. Generally, fish habitat models are used for assessing *Fisheries Act* violations only after it has been demonstrated that they can reliably predict fish biomass or productivity (Lanno and Wren, 1992; Stoneman et al., 1996). Because toxicity testing, and fish population and benthos surveys have been incorporated into *Fisheries Act*-related AEEM programs (e.g., EC and DFO, 1992a,b), they could presumably, on their own, be used to demonstrate violations of the act. However, unless these surrogate endpoints are calibrated with more specific fish-related endpoints such as fish biomass or production, or fish community composition, interpreting changes in such surrogate measures in relation to the goals of the *Fisheries Act* becomes problematic (Jones et al., 1996).

1.2 Fishery Endpoints

In this thesis, my main objective is to determine the types of changes in benthic community composition that correspond with unacceptable effects on relevant fishery-related endpoints. There are, however, several fish community endpoints that one might choose, including: (1) presence/absence, abundance or biomass of selected species (Jones and Stockwell, 1995); or, (2) indices of community composition based on presence/absence, abundance, biomass or production of all members of the community (e.g., Portt, 1980; Karr, 1981; Mahon, 1985, Mahon et al., 1979; MOT, 1994; Wichert, 1994a,b; Minns et al., 1996; Stanfield et al., 1996). When assessing the condition of a fishery, measuring <u>total</u> fish biomass or production is not very informative since two locations could have the same overall fish production while having completely different

assemblages (i.e., one may be desirable while the other may not). In contrast then, measuring the abundances, biomass and/or productivity of valued members of a fish community at least allows us to assess whether valued species are present in acceptable numbers (or biomass/production levels).

Apart from the *Fisheries Act*, being able to predict changes in fish community composition can be considered important because changes in fish communities, or of selected valued species, has obvious societal value (Karr, 1981) which can be either economic or sociological in nature. Game fish species have economic value because anglers spend money on their sport. The loss or gain of a game fishery can, therefore, result in a change in the economic value of an aquatic system (Kreutzweiser, 1984). Changes in the abundance or biomass of commercial species have even more obvious consequences. Whether the game or commercial fishery is situated in lotic, lentic or marine environments, change in the status of a fishery is normally a reflection of change in overall fish community composition (Leach et al., 1977; Mills and Schiavone, 1982; Bowlby and Roff, 1986a,b; McQueen et al., 1986; Carpenter et al., 1987; Olver et al., 1995; Sherman and Busch, 1995). Consequently, there should be value in attempting to relate benthic community composition to the overall composition of fish communities.

1.3 Benthos as a Potential Surrogate Measure for Fishery Endpoints

Being able to use a surrogate measure of fish community composition has several advantages over the measurement of the fish community itself. First, measurement of a surrogate response (i.e., the benthic community) avoids the possibly destructive effects of sampling on the fishery resource (compliance indicator). All forms of fish capture are destructive to fish in some way. Seining results in the loss of scales at the least, while gill netting and electrofishing can result in significant mortality depending on environmental conditions (Horak and Klein, 1967; Reynolds and Koltz, 1995). Secondly, fish can be difficult to collect in some locations, particularly in large waters (e.g., Mah et al., 1989; Hodson et al., 1992; Gibbons et al., 1995; Brown et al., 1996), whereas surveys of benthos are generally more practical. Thirdly, surrogates such as benthos that respond more quickly to environmental disturbances can provide early warning of impending non-compliance prior to degradation of the fishery resource. Finally, the joint use of

quick responding surrogates and slow responding fish community endpoints can diagnose the nature of a disturbance. For example, if the slow-responding compliance indicator demonstrated evidence of a deleterious effect while the faster-responding surrogate did not, then one could infer that a pulse-event impact (Underwood, 1989) had occurred in the past. Given that the surrogate has obviously recovered from the event, the probability of the compliance indicator recovering is high, given enough time. In contrast, if a fast-responding surrogate had demonstrated evidence of a deleterious effect while the slower-responding compliance indicator had not, then this would provide evidence that a press (continuous) impact (Underwood, 1989) had been occurring. If allowed to continue, such press impacts would likely affect the compliance indicator. Finally, if we know that a system has been unable to support a specific desirable fish community in the past, it could be important to determine whether conditions have improved enough at a site to consider stocking of that more desirable fish community. In such situations, the use of surrogate measures of the condition (or potential condition) of the fish community are required.

1.3.1 Known Linkages Between Benthos and Fish

Benthic macroinvertebrates are often rationalized as components of AEEM programs because of a presumed relationship between fish and benthic invertebrates (EC and DFO, 1992a,b; DFO and EC, 1995b, Hodson et al, 1996). Because most fish eat benthic organisms during at least part of their life cycle, it is sensible to assume that changes in benthic production will affect production of fish through *bottom-up* effects. Waters (1988) and others have demonstrated relationships between benthic and fish production for stream ecosystems, while similar relationships have been demonstrated for lake (Northcote and Larkin, 1956; Hayes, 1957; Rawson, 1960; Matuszek, 1978; Boisclair and Leggett, 1989; Pierce et al., 1994) and marine systems (MacKinnon, 1973; Diaz and Schaffner, 1990). In contrast, fish can have influences on benthic community composition through "top-down" feeding effects in relatively controlled (closed) systems like lakes (Ball and Hayne, 1952; Anderson et al., 1978; Chess et al., 1993), experimental cages (Gilinsky, 1984), or modified stream sections (Allan, 1982; Elliott, 1986; Morgan and Ringler, 1994). These findings provide considerable evidence that benthos and fish are linked at least energetically. However, as suggested earlier, total fish biomass or productivity can be the same at two locations that have different fish faunas. Consequently, being able to

predict total fish biomass (or production) at a site may tell us little about the condition of the fishery, whereas being able to predict the fish assemblage at a site provides much more information (e.g., Jackson and Harvey, 1993). Unfortunately, none of the fish-benthos models currently available allow for such predictions.

1.4 Overview of Thesis Objectives

As specified earlier, the main objectives of this thesis are to:

- establish the relationship between descriptors of benthic community composition and relevant fishery-related descriptors, and,

- determine the types of effects in benthic community descriptors that correspond with unacceptable effects on relevant fishery-related endpoints.

The first objective is addressed using surveys of benthos and fish in first- to fourth-order streams in southern Ontario. The studies are limited to stream fisheries for several reasons. First, there are currently several monitoring programs in southern Ontario which have the objective of determining the effects of anthropogenic activities on smaller, wadeable stream systems. These monitoring activities are being conducted by various government agencies, including the Ontario Ministry of Environment & Energy (Griffiths, 1993; K. Somers, pers. comm.), the Ontario Ministry of Transport (W. Snodgrass, pers. comm.), the Ontario Ministry of Natural Resources (Stoneman et al., 1996; M. Jones, pers. comm.), Credit Valley Conservation Authority (H. Breton, pers. comm.), and the Metro Toronto and Region Conservation Authority (S. Meek, pers. comm.). Each of these programs incorporates surveys of benthic community composition, while most (except the MOEE) have the objective of protecting fish and fish habitat. Consequently, there is a substantial need to understand the fish-benthos relationship in small, wadeable streams. Second, the relationships between fish and invertebrates can be expected to be stronger in smaller streams than in larger rivers because larger systems support larger fish species with larger home ranges (Minns, 1995). Larger home ranges for fish would result in fish integrating environmental conditions over a broader spatial scale. Due to the more limited spatial movement of benthos

in either small or larger river systems, it seems likely that any fish-benthos association would be stronger in smaller rivers/streams. Future work on fish-benthos associations in larger systems may be warranted if associations are strong in smaller streams. A final consideration for conducting this work in small lotic systems is because associations between fish and benthos communities have already been fairly well documented in lentic systems (Jackson and Harvey, 1993; Pierce et al., 1994), although the derived models require refinement before they can be used in the setting of AEEM.

Both the first and second objectives require that the fishery resource is described in a fashion that has relevance to the goals of the *Fisheries Act*. Although the federal *Fisheries Act* has a guiding policy of maintaining the productive capacity of fish habitat, several approaches to characterizing fish communities have been used (Wichert, 1994a,b, Jones and Stockwell, 1995). Surveys that result in the quantification of the production or biomass of all community members is most time consuming, but logically most closely related to the goals of the act (Randall et al., 1995; Minns et al., 1996). There is however, little work demonstrating the effect of choice of community descriptor on our ability to demonstrate an impact (Minns et al., 1996). Consequently, a third objective of this thesis is to examine the relationships among stream-fish community descriptors.

The second objective also requires an understanding of the types of effects on relevant fishery-related endpoints that are unacceptable. As before, the Canadian federal *Fisheries Act* provides little advice on what should be considered a significant impact on fisheries-related endpoints. Several jurisdictions, have however, attempted to develop *criteria* for fish communities based on various indices of composition, and on various definitions of normal or acceptable reference conditions (see references in Woodley et al., 1993, and Davis and Simon, 1995). The approaches taken so far generally involve the characterization of a number of acceptable reference locations, derivation of an index, and calculation of critical index values that denote acceptable and unacceptable conditions. There are two general difficulties with this approach to developing criteria. First, there is the impression that a very large number of reference locations must be described in order to characterize the reference conditions (e.g., Yoder and Rankin, 1995). Secondly, since most indices are developed for specific regional applications (e.g., the IBI, Karr, 1981; Steedman, 1988), they are difficult to apply in other

locations. This became apparent during development of the Canadian EEM program for the pulp and paper industry. Each mill across the country was required to conduct an assessment of the effects of their mill effluent on receiving water quality. There were over 120 mills at the time, discharging effluent into a wide variety of lotic, lentic and marine habitats. The application of conventional bio-criteria for adult fish survey, fish community collections, and benthic community collections was not possible because none of the conventional criteria was appropriate to all sites. This presented an obvious requirement for the development of a generic criterion that could be applied to any parameter, regardless of the units of measurement or the scale at which the assessment was being conducted (i.e., regional or site-specific assessment).

1.5 Thesis Outline

Each of the objectives listed above is addressed in a separate chapter. Chapter 2 develops the argument for use of normal ranges as a generic decision criterion for compliance parameters. The chapter also outlines univariate and multivariate analyses of variance that can be used to determine the probability that effects truly fall outside of the normal range. The concepts presented in this chapter have obvious application in environmental assessments, but can also be applied to more pure, academic, research.

Chapter 3 evaluates the relationships among descriptors of fish community composition. I evaluate single- and three-pass electrofishing data as multivariate descriptors of composition based on presence/absence, abundance and biomass data. In addition, I contrast all of these descriptors with estimates of abundance and biomass of brook trout and brook + brown trout abundances and biomass to determine which series of descriptors would be most protective if the goal was to detect impacts on stream-fish communities dominated by brook and brown trout.

The correlations among benthos, fish and environmental descriptors are evaluated in Chapter 4. For this exercise, I relied on data from three independent studies: (1) Barton et al.'s (1985) buffer strip study with added benthic data; Farrara and Reid's (1995) fish and benthic survey for the City of London, Ontario; and (3) a survey I conducted during the summer of 1995 in conjunction with the Ontario Ministry of Natural Resources. Each study had somewhat different objectives, and therefore a moderately different suite of environmental variables against which to test for fish-environment and benthos-environment correlations. These studies also

employed different collection methods for fish and benthos. Consequently, these three independent data sets provided an opportunity to establish the robustness of fish-benthos-environment correlations. With these data, I was specifically interested in demonstrating that fish and benthic communities are correlated, (i.e., changes in one correspond with changes in the other), as well as demonstrating that fish and benthos are controlled by (correlated with) similar suites of environmental variables.

Chapter 5 examines in more detail the fish-benthos correlation. More specifically, this chapter establishes relationships between descriptors of the two groups that allow us to predict the degree of impact on a fish community given the degree of impact on a benthic community. I assessed such relationships for studies that encompassed what I consider to be two scales of sampling. First, I examine the fish-benthos relationship across large spatial scales where investigators compared regional-reference (Hughes, 1995) stream sites against a number of test stream sites. In such studies, investigators are usually more interested in knowing if a stream site is impacted, but not necessarily the cause. For this spatial scale, I used the same three data sets that were used in Chapter 3. After selecting a set of streams which I considered to be relevant target reference streams, I established the relationship between degree of difference of non-reference fish communities and the degree of difference of non-reference benthos communities. Second, I examined the fish-benthos relationship across small-spatial scales where investigators use site-specific (Hodson et al., 1996) reference sites to determine whether a specific point source has significantly altered the condition of a fish community. For this spatial scale, I used fish and benthos data that I collected from Laurel and Canagagigue Creeks in 1994, each of which had fish communities impacted by impoundments. I used stream stations upstream of these impoundments as reference locations against which to compare downstream stations. Again, with these data sets, I examined the correlation between impacts measured by fish and impacts measured by benthos. Finally, Chapter 6 presents an overview of the findings of the thesis and presents a view for future endeavour.

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CHAPTER 2

Using the normal range as a criterion for ecological significance in environmental monitoring and assessment

2.0 Abstract

Environmental monitoring programs typically measure some component of the environment considered worth protecting. Data from monitoring programs are often used to compare potentially impacted locations with unimpacted reference locations. The use of normal ranges to define (or set) ecological criteria has logical and precedented use. In this Chapter, I define ecologically relevant effects as being observations at impact locations that fall outside the normal range of variation at reference locations as determined by the region enclosing 95% of reference observations. Simple ways to calculate normal ranges (95% regions) for univariate and multivariate responses are demonstrated. I also demonstrate three statistical tests that can be used to determine when a point observation falls outside of the normal range. The consequences of using conventional two-sample contrasts in relation to one-sample non-central equivalence and interval tests are determined. For sites that are truly outside of the normal range of variation for reference locations, equivalence tests will lead to erroneous conclusions of no impact at most 5% of the time. Two-sample contrasts will lead to erroneous conclusions of no impact about 50% of the time with low sample sizes (i.e., 10-20 reference locations). In contrast to both the two-sample contrast and the equivalence test, interval tests fail to recognize sites as being impacted unless impacts are in excess of about 3 σ from the reference population average (with a reference sample size of 20). Finally, the penalty for using the equivalence test is that it will fail up to 26% of sites that are truly members of the reference population. Practitioners contrasting non-random impacted locations against a set of reference locations should consider these characteristics of the various potential tests when deriving conclusions of impact.

2.1 Introduction

Ecological monitoring programs are commonly incorporated into regulatory programs to monitor the impacts of industrial or other anthropogenic activities on natural resources (Rees et al., 1990; USEPA, 1990; Environment Canada and Department of Fisheries and Oceans, 1992; Davis and Simon, 1995). The currencies of these programs are biological measures that describe some aspect of the ecosystem considered worth protecting (Schindler, 1987; Courtemanch, 1989; Courtemanch et al., 1989; Karr, 1991; Suter, 1993; Department of Fisheries and Oceans and Environment Canada, 1995). Generally, observations from presumably impacted locations are compared with observations from either regional-reference or site-specific unimpacted locations (Hughes, 1985, 1995; Hughes et al., 1986, 1990; USEPA, 1990).

Regardless of the particular design, biologists are increasingly being asked to interpret the importance of observed effects on measured biological endpoints (Yoder, 1989; Bode and Novak, 1995; Courtemanch, 1995; Department of Fisheries and Oceans, and Environment Canada, 1995; Mapstone, 1995; Power et al., 1995; Southerland and Stribling, 1995; Skalski, 1995; Yoder and Rankin, 1995). Consequently, the development of *biocriteria* to denote what is, or is not, an acceptable value for an impacted location, has become an important issue. In most goal-setting (biocriteria development) forums, the use of normal ranges of variation in unimpacted locations to denote what is an acceptable or unacceptable value for a biological entity is common (e.g., Yoder and Rankin, 1995). Observations outside that normal range are considered evidence that a stressor is having undue influence on the environment. The use of normal ranges is particularly commonly used in the medical industry, to determine when individuals are sick. Anyone with a measured physical condition that is outside of the normal range of variation for a healthy population is usually considered sick. The use of normal ranges in biological applications is also precedented (e.g., Kersting, 1984, 1988, 1991; Bloom, 1980; Yan et al., 1996; Findlay and Kasian, 1996).

Typically, the normal range of variation is considered to be the range of values enclosing 95% of the reference values (Thompson, 1938, Leffler, 1978; Resh et al., 1988; Rode and Chinchilli, 1988; Montgomery, 1991). For single descriptors that are normally distributed, the normal range of variation (i.e., the region enclosing 95% of the data) is defined by $\mu \pm 1.96\ \sigma$.

If the number of observations describing the reference condition is very large, then one could make the assumption that the estimated normal range ($\bar{X} \pm 1.96$ standard deviations), is in fact *the* normal range (i.e., measured without error). If, however, there is only a small number of observations to characterize the reference condition, then our estimate of the normal range will have some (considerable?) error. Consequently, using our estimate of the normal range as the criterion against which to judge a value from an impacted location could easily result in an erroneous conclusion if there has been only limited effort put towards characterizing the reference condition. Although this problem would suggest a need to put considerable effort into characterizing reference conditions whenever an assessment is being conducted, there are statistical procedures that can be used to determine when a point falls outside of the normal range of reference location observations, based on only a sample of reference location observations.

This chapter has three main objectives. The first is to provide an operational definition of normal ranges. Doing this helps to formulate appropriate null hypotheses that can be tested with analysis of variance. The operational definition is also applied to multivariate responses. This is important because many biological responses are multivariate (e.g., length-weight relationships in fish populations). The second objective is to compare three different ANOVA tests that could be applied to designs in which a single impacted location is contrasted against a set of reference locations. With each model, I examine their basic assumptions and the effects of those basic assumptions on our ability to make the correct inference; i.e., how often we will say that an impacted location falls outside of the normal range of variation, when in fact it really does fall outside. The third objective is to demonstrate use of proposed statistical techniques using a simulated, but realistic data set in which benthic invertebrate species richness and abundance at a single impact location are compared to richness and abundance estimates from 10 reference locations. To simplify this presentation, I focus on parametric procedures that assume a normal distribution of observations from reference locations. I do however, recognize that many biological attributes are not normally distributed in reference locations, even with appropriate transformations (Yoder and Rankin, 1995; Bailey, 1996). At some point, it would be beneficial for someone to demonstrate non-parametric procedures that can be applied for the same purpose. I also focus on simple study designs that use variation among spatially separated

reference locations to characterize the normal range of variation for reference conditions. There are several designs that might incorporate variation over time, both before and after the putative impact, in both reference and impact locations. However, for simplicity, they are ignored. The approach of using the temporal variations to create the normal range of variation would be an obvious extension of the ideas presented here.

2.2 Defining The Normal Range

If the region enclosing 95% of the observations from a set of reference locations defines the normal operating range for any variable (Thompson, 1938; Rode and Chinchilli, 1988; Kersting, 1984, 1988, 1991; Resh et al., 1988), then observations falling outside this region are unusual by definition. By extension, observations from potentially impacted locations are not overly unusual if they fall within this normal range. This use of normal operating ranges is thus proposed as a generic criterion for ecological significance.

For single variables, the scatter of points is determined by the population mean (μ) and standard deviation (σ). If the data are normally distributed, then the region defined by $\mu \pm 1\ \sigma$ incorporates about 67% of the population, and $\mu \pm 1.96\ \sigma$ incorporates about 95% of the population. Irrespective of the variable of interest and its original units of measurement (assuming that the variable is either a ratio-scale or continuous variable), normal ranges can be re-expressed in units of standard deviations by dividing by the standard deviation (Cohen, 1977). This simple transformation puts all variables on a common scale and provides a convenient way of standardizing the degree of impact across studies (Hunter et al., 1982).

Normal ranges for multivariate responses can be defined in a similar way. When the response involves two variables (Y1 and Y2), the normal range is defined by an ellipse that accounts for correlation between the variables (Sokal and Rohlf, 1981). For multi-variable responses, the normal range is still expressed in standardized units analogous to the univariate normal ranges described above, and are referred to as generalized distances (or Mahalanobis distances; i.e., D) from the grand mean or centroid. For single variables, we could also use the term generalized distances to express the normal range because $1.96\ \sigma$ and 1.96 generalized distances are equivalent.

Because of the inter-relationships between different statistical distributions, generalized distances associated with the normal operating range can be expressed as a function of the appropriate t, F, or X^2 distributions. If the response is univariate, the normal operating range ($\mu \pm 1.96\ \sigma$) is simply determined by $\mu \pm t_{(\alpha=0.05)}\ \sigma$, where $t_{(\alpha=0.05)} = 1.96$. Rather than t, we could also use the square-root of $F_{(1,\alpha=0.05)}$, which is also 1.96. Generally though, for normally distributed descriptors, the normal range is expressed as $\sqrt{X^2_{(0.95,p)}}$ generalized distances about the population mean (univariate) or centroid (multivariate), where $X^2_{(0.95,p)}$ is the 95th percentile of a chi-square distribution for p variables (Rode and Chinchilli, 1988). As before, for univariate tests we still obtain a 95% region of $\sqrt{X^2_{(0.95,1)}} = \sqrt{3.841} = 1.96$ generalized distances (or 1.96 σ).

2.3 Tests for Effects Exceeding the Normal Range

2.3.1 Two Sample Contrasts

The conventional assessment of a single observation involves a two sample contrast. For univariate responses, the two-sample test statistic is (Zar, 1984):

$$F = \frac{(\bar{X}_r - Imp)^2}{\left(S\sqrt{1+\frac{1}{n_r}}\right)^2} \qquad [1]$$

where $\bar{X}_r$ is the mean of the reference location responses, *Imp* is the impact location observation and S_r is the estimated standard deviation of the reference location observations. The test statistic has 1 and n_r-1 degrees of freedom and is compared to a critical F value derived from the central F distribution.

The multivariate equivalent of this test statistic is (Morrison, 1967):

$$F = \frac{(n_r - p)\, n_r D^2}{(n_r - 1)(n_r + 1)\, p} \qquad [2]$$

where p is the number of variables, D is the generalized distance

$$D = \sqrt{(\bar{X}_r - Imp)' S_r^{-1} (\bar{X}_r - Imp)}\ , \qquad [3]$$

$\bar{X}_r$ is the centroid (vector of mean responses) of the reference locations, *Imp* is the vector of response at the impact location, and S_r^{-1} is the inverse of the variance-covariance matrix for the reference location observations (Owen and Chmielewski, 1985). This multivariate F statistic has p and n_r-p degrees of freedom.

This particular formulation of the test statistic is designed to test the null hypothesis

- H_{OI}: the single impact site is a member of the reference population *vs* H_{AI}: that it is not (Table 1).

2.3.1.1 Testing the Performance of Two-Sample Contrasts

Most environmental assessments are generally concerned with knowing whether a particular non-random site is impacted (Green, 1979). For example, downstream of a domestic sewage outfall there is a site at which we can collect a fish community. Our question is whether the fish community downstream of the outfall is outside of the normal range of variation for a set of presumed randomly sampled reference fish communities. Reference fish communities would probably have come from upstream reference locations, or from regional-reference locations with similar physical features to the site we are testing. The impact site, however, cannot be replicated and is not considered a random observation. We cannot replicate the impacted location. By definition, the fish community would be collected from within something like a 100

m^2 area that covers the whole stream from bank to bank. Taking additional collections immediately upstream or downstream (i.e., extra 100 m^2 locations) would not be appropriate as replicates because those areas would be exposed to different effluent concentrations. Moreover, they would not be independent because of spatial/temporal autocorrelation.

Further, if we were measuring something else like a benthic community downstream of the sewage outfall, we might be able to take "replicate" samples if the technique allows for it. Barton (1996) and others use a travelling kick technique that collects benthos from the same spatial unit as fish are collected from. As with the fish sampling technique, there is no way to replicate the impact site. Sometimes, benthic ecologists will use a quantitative technique that allows for multiple collections from a single riffle within the 100 m^2 that the fish were collected from. Because of the spatial proximity, the multiple collections from the same riffle are not independent. Even if we were to assume that they were independent, we would then have to collect multiple samples from within riffles at each of the reference locations that we had sampled. If it could be assumed that the samples within riffles were independent, then the difference between reference and impact would still be judged against the among reference-location variability.

Since we are only interested in evaluating the one impact site, we do not care to make inferences about a larger population of potentially impacted sites. Rather, we are only interested in knowing whether the site we are studying is inside or outside the 95% region for reference sites. Because we know that sewage is being discharged by the treatment facility, we already know that there is at least some amount of effect downstream, no matter how trivial that effect may be.

What, then, would happen if we used the two-sample contrast to test that the non-random impacted site was a member of the reference population? To find out, I conducted a simulation experiment using RESAMPLING STATS (Bruce, 1993) in which I collected 10 random samples from a reference population that had a mean value of 0 and standard deviation of 1. The value for the impact location was fixed at varying distances from the reference population mean. For example, to determine the probability that H_{01} will be rejected when the impact location is exactly 2 σ from the population mean, I ran the simulation 10,000 times. For each iteration of the simulation, the impact location was given a value of 2 while the reference location samples

were randomly selected from a universe of samples with a mean of zero and a standard deviation of one. This represents the situation in which the impacted location is non-random, while the reference locations are a random sample of a large universe of potential reference locations. If we refer to the example where we wish to know whether the fish community downstream of sewage outfall is outside of the normal range of variation for reference fish communities, we could not change our characterization of the impacted fish community. We could, however, have selected any number of combinations of 10 reference locations against which to make our comparison. For the 10,000 simulated sampling events, I counted the number of times the test statistic from [1] exceeded the critical value for the appropriate degrees of freedom (i.e., for 1 and 9 degrees of freedom the critical F value was 5.12). I repeated this procedure for reference sample sizes of 20 also.

Figure 2.1 shows that the probability of rejecting H_{O1} and concluding that the impacted site is outside of the normal range of variation for reference locations is just over 30% when the impact observation is just outside the normal range (i.e., at about 2 σ) and with a sample size of n_r=10 reference locations. With n_r=10, the impact location would have to be about 3 σ or more from the mean of the reference locations before there would be a reasonable chance of rejecting H_{O1} and correctly concluding that the impact location had been impacted. Consequently, the two-sample test will allow many significant effects to go unnoticed.

2.3.2 One-Sample Contrasts

Well, if the two-sample contrast for small sample sizes does not have a high enough power to tell us when an observation from a site falls outside of the normal range of variation, what can we do? We can try a one-sample contrast with the null hypothesis restated as either

- H_{O2}: $|\mu_r - Imp| \leq \delta$ *vs* the alternate hypothesis H_{A2}: $|\mu_r - Imp| > \delta$, or
- H_{O3}: $|\mu_r - Imp| \geq \delta$ *vs* the alternate hypothesis H_{A3}: $|\mu_r - Imp| < \delta$,

where μ_r is the average response of the reference population, *Imp* is the response or value at the impacted location, and δ is a deemed important effect size (Table 1). In typical one-sample

contrasts, δ is set to zero, thereby testing for a difference between the sampled population and the fixed value (standard). However, for the question being posed here, setting δ to zero would be meaningless since any of the reference locations could be shown to be different than the mean of the reference population if enough reference-location samples were collected.

To make these one-sample contrasts test that a point falls at or beyond the 95% range for the reference population, we set δ equal to the limit of the normal range of variation. For single variables, limits for the 95% region are at $\mu \pm 1.96\ \sigma$. The critical effect size (δ) can, therefore, be set to $1.96\ \sigma$. If the biological response was bivariate, the limit for 95% of the data would be $\sqrt{X^2_{2,.95,2}}$ or $\sqrt{5.991} = 2.45$ generalized distances from the reference population centroid. Other δ's would be set in a similar way if the number of variables was increased.

The second null hypothesis (H_{O2}) is an interval test (Lehman, 1959; Mood and Graybill, 1963, Ferguson, 1967; McBride et al., 1993). If H_{O2} is rejected, then there is reasonable confidence in concluding that the difference between the mean reference response and the response at the impact location exceeds the critical effect size (δ). If we wish to test the null hypothesis that the impact location has a value in excess of the limits for the 95% region, then we could set δ to $1.96\ \sigma$. Acceptance of the second null hypothesis (H_{O2}) infers that the impact location is less or equal to δ (or $1.96\ \sigma$) from the mean reference response.

The third null hypothesis (H_{O3}) is an equivalence test (Patel and Gupta, 1984; McBride et al., 1993). Rejection of H_{O3} infers that the difference between the mean reference response and the response at the impact location is less than δ ($1.96\ \sigma$), whereas acceptance of the null hypothesis infers that the difference is at least as large as δ ($1.96\ \sigma$). Both the interval and equivalence tests provide information on the magnitude of the difference between reference and impact responses.

When δ is set to anything other than zero (i.e,. a non-zero effect size), we compare test statistics against non-central distributions of the test statistic under our specified null hypothesis (McBride et al., 1993). In the sections that follow immediately (2.3.1, 2.3.2), I discuss derivation of the test statistic as well as non-central distributions against which to compare the test statistic.

2.3.2.1 Calculating the Test Statistics

For the null hypotheses described above, I propose using the absolute value of the difference from the reference-population mean because the direction of change in environmental assessments is usually unimportant (i.e., it is often more important to know whether things have changed, not so much in what direction the change has occurred). For a one-sample univariate contrast with multiple reference observations, we can test the two new null hypotheses (H_{O2} and H_{O3}) with the following observed F statistic (Zar, 1984):

$$F = \frac{(\bar{X}_r - Imp)^2}{\left(S_r\sqrt{\frac{1}{n_r}}\right)^2}, \qquad [4]$$

which has 1 and n_r-1 degrees of freedom. If the data are multivariate, the observed F statistic is (Morrison, 1967):

$$F = \frac{(n_r - p)\, n_r D^2}{p(n_r - 1)} \qquad [5]$$

which has p and n_r-p degrees of freedom.

2.3.2.2 Estimating Critical Values to Evaluate Observed F Values

Evaluating the significance of the various test statistics presented involves comparing the observed F statistic with critical F values. As before, if the null hypothesis of no difference was specified (i.e., H_{O1}), we would compare the observed F statistic with a critical F value derived from a central F distribution. The central F distribution is the distribution of F statistics when the null hypothesis of no difference is true (Figure 2.2). The 95th percentile of this central F distribution is the critical $F_{0.05:HO1}$ value against which we would judge the significance of the observed F statistic for an impacted location (Figure 2.2). These critical $F_{0.05:HO1}$ values are the

traditional values provided in statistical texts.

In contrast, if we specify either the interval or equivalence null hypotheses, the critical F value from the central F distribution is no longer appropriate. Because we have specified null hypotheses with non-zero effect sizes, the distribution of F statistics follows a non-central distribution (Figure 2.2). For H^{O2}, if we observe an F statistic that is greater than the upper 95th percentile ($F_{0.05:HO2}$) of the non-central F distribution, we have some certainty that impact observations falls at or beyond δ. $F_{0.05:HO2}$ is therefore compared to the observed F statistic when testing the interval null hypothesis. In contrast, if we observe an F statistic that is greater than the lower 5th percentile ($F_{0.05:HO3}$) of the non-central F distribution, then we have some certainty that the impact observation falls at or beyond δ. $F_{0.05:HO3}$ is compared to the observed F statistic when testing the equivalence null hypothesis.

Probability density functions of non-central F distributions can be calculated using the equations given in Patnaik (1949), Tiku (1965), Guenther (1979), Norton, (1983) or Narula and Weistroffer (1986). Based on these equations, Narula and Weistroffer (1986) provide a Fortran algorithm that can give critical values, while Pearson and Hartley (1972) and Odeh and Fox (1991) provide simplified formulae and tables for determining critical F values. Finally, there are a variety of commercial software packages (reviewed by Goldstein, 1989) that can also be used to determine non-central critical F values.

All approaches to determining non-central critical F values require input of a non-centrality parameter (often defined as λ) which determines the position of the non-central distribution. Zar (p 172; 1984) illustrates how to calculate non-centrality parameters for simple single-factor designs, while Odeh and Fox (1991) present equations for the non-centrality parameter for both univariate and multivariate ANOVA designs. For one-sample contrasts, the non-centrality parameter is:

$$\lambda = \frac{n_r \times (\mu_r - Imp)^2}{\sigma^2} \qquad [6]$$

With μ_r - Imp = 1.96 σ, this reduces to:

$$\lambda = n_r \cdot 3.841. \qquad [7]$$

For multivariate one-sample contrasts, the non-centrality parameter is

$$\lambda = n_r(\mu_r - Imp)' \sum_r^{-1} (\mu_r - Imp), \quad [8]$$

More simply, this can be written:

$$\lambda = n_r D^2 \quad [9]$$

When the response involves two variables, the normal range is D or $\sqrt{X^2_{0.95,2}} = \sqrt{5.991} = 2.45$ generalized distances. By setting the square of the generalized distance (D^2) to 5.991, the non-centrality parameter for bivariate contrasts reduces to:

$$\lambda = n_r \cdot 5.991. \quad [10]$$

2.3.2.3 Testing the Appropriateness of the One-Sample Non-Central Tests

To determine the performance of the one-sample, non-central tests, I examined the probability of rejecting both the interval and equivalence tests. The probability of rejecting the interval null hypothesis can be estimated by determining the proportion of F statistics that will exceed the critical value ($F_{0.05;HO2}$) under assumed true differences between the average reference response and the impact location response. The probability of rejecting the equivalence test is estimated by determining the proportion of observed F values that will be less than the critical F value ($F_{0.05;HO3}$) under assumed true differences. Determining these proportions can be accomplished through simulation or through the use of non-central F distributions, the results are equivalent.

When the effect size is 1.96 σ, the use of the equivalence test will result in inferences that a site is outside of the normal range, 95% of the time (Figure 2.1). The penalty for this test formulation is that sites that are truly just inside the limits of the normal range will have a high probability of leading to acceptance of H_{O3} and a conclusion that the site is outside the normal range. If the impact location truly falls at 1.6 σ (i.e., is within the normal range), the probability of rejecting the equivalence null hypothesis (and concluding no significant impact) is only about 20% with 10 reference locations, and 30% with 20 reference locations (Figure 2.1). To ensure

a 95% chance of rejecting the null hypothesis, the impact location would have to fall at 0.5 σ if 10 reference locations were surveyed, and at 0.7 σ if 20 reference locations were surveyed (Figure 2.1).

In contrast, the interval test requires that impacted locations be considerably impacted before there is a high probability of concluding that the impacted site is outside of the normal range of variation for reference locations (Figure 2.1). When the impact location is truly 2 σ from the reference population mean response, the probability of rejecting H_{O2} is about 5% (i.e., the nominal Type I error rate, Figure 2.1). With n_r=10 reference locations, and a true effect size of 3 σ, the probability of rejection increases to about 40%. Increasing the number of reference locations to 20 would increase this probability to about 68%. To ensure a 95% chance of rejecting this null hypothesis, the true effect size would have to be close to 5 σ when 10 reference locations are sampled, and 3.5 σ when 20 reference locations are sampled (Figure 2.1).

Finally, to determine the effect of using non-central tests if the impact location truly is a member of the reference population, I conducted a second simulation experiment. As with the first simulation experiment, I sampled 10 observations from a reference population with a mean of zero and a standard deviation of one. The single impact observation was also sampled *randomly* from the reference population. I calculated the one-sample test statistic, and counted the number of times out of 10,000 iterations that the test statistic exceeded the critical values for the equivalence and interval null hypotheses. As above, with n_r=10, the non-centrality parameter is 38.41. Consequently, the critical value for the equivalence test is 16.2 while the critical value for the interval test is 115.8. With this simulation, 26% of the test statistics exceeded the critical value of 16.2 which would lead to the incorrect conclusion that the impact location was truly in excess of the normal range of variation of reference locations. Only 1% of the test statistics exceeded the critical value for the interval test. These results demonstrate that use of the one-sample equivalence contrast could result in failing 26% of reference locations, while only 1% would be so misclassified by the interval test. With two-sample contrasts, the probability of misclassifying a true reference sample would be 5% (i.e., the nominal Type I error rate).

2.4 Application and Demonstration of the 95% Rule

To demonstrate the 95% rule and associated tests, data representing species richness and

logarithmically transformed abundance of an aquatic benthic community were simulated for 10 reference locations and one potentially impacted location (Figure 2.3). Richness is the number of taxa, whereas abundance is the number of individuals collected from each location. The data were simulated to represent benthic communities with true differences in abundance and richness of 2 σ, and a correlation between abundance and richness in the reference locations of 0.8 (Figure 2.3c). These data are used to demonstrate how both the one-sample univariate and multivariate null hypotheses are specified and tested. Finally, I demonstrate the use of the conventional two-sample test statistic to demonstrate that there are circumstances when its use will result in erroneous conclusions.

For the interval and equivalence tests, critical F values are estimated using non-central distributions. Because the non-zero effect sizes were chosen to be 1.96 σ for the univariate contrasts and 2.45 generalized distances for the bivariate contrasts, the appropriate non-centrality parameter for the non-central tests are 3.841 n_r or 3.841 x 10 = 38.41 (univariate) and 5.991 n_r or 5.991 x 10 = 59.91 (bivariate). Based on these non-centrality parameters, the lower 5% and upper 95% tails of the non-central F distribution fall at 16.2 and 115.8 for the univariate tests, and 13.67 and 95.12 for the multivariate test (Table 2.2). The upper critical values (i.e., 115.8 for the univariate and 95.12 for the bivariate) are used to evaluate the interval test null hypothesis.

The lower 5% critical values from the non-central distribution are used to test the equivalence null hypothesis. Observed F statistics less than the lower critical value suggest that the true absolute difference was < 1.96 σ (or < 2.45 generalized distances). As a result, if we reject the null hypothesis we then accept the alternate hypothesis that no significant effect had occurred (i.e., that the impact location had a value that fell inside the normal range of variation). Alternatively, observed F values greater than the lower critical values suggest that the true absolute difference between the average reference response and the impact location was ≥ 1.96 σ (or ≥ 2.45 generalized distances), and would imply that a significant effect had occurred.

Observed differences between the reference and impact samples were -2.194 for log-transformed abundance, and 18.8 for richness. When expressed as standardized differences, these values were 2.875 S_r and 2.331 S_r respectively. The estimated bivariate generalized distance was

7.69. First, let us look at the performance of the two-sample contrast. The observed univariate F statistic associated with the difference in abundance (7.526) was greater than the two-sample critical value of 5.12. The observed bivariate F statistic (23.88) associated with the two-sample test was also greater than the bivariate critical value of 4.46. Using either of these tests, we would have come to the correct conclusion that the impacted observation was outside of the normal range of variation for reference locations. However, the observed univariate F statistic associated with the difference in species richness (3.449) was less than the critical value of 5.12. Had only species richness been tested, we would have come to the erroneous conclusion that the impact observation truly lay inside the normal range of reference locations. Given the observed standardized difference in abundance (2.19 S) and in standardized the bivariate difference (7.69 generalized distances), it is not surprising that the two-sample tests rejected the null hypothesis of no difference between reference and impact location responses. However, it should be noted that Figure 2.1 demonstrated that there is just over a 30% chance of detecting such effects if the true effect is just beyond the 95% region. In other words, with this particular simulation, we were lucky to have rejected the null hypothesis of a difference between the impact and reference populations. In contrast, had the test only been used to assess species richness, the two-sample test would have resulted in an acceptance of the null hypothesis with the result that we would have incorrectly concluded that the difference was less than the normal range of variation for reference locations.

The equivalence tests however, always gave results leading to the correct conclusion. To test the equivalence null hypothesis (Table 2.1), we compare the observed F values with the lower non-central critical $F_{0.05;HO3}$ values. We find that the observed univariate F values and the observed bivariate F value were larger than the lower non-central critical F values (Table 2.2). As a result, H_{O3} not rejected in any of the cases with the correct conclusion that the true difference was $\geq 1.96\ \sigma$ (or ≥ 2.45 generalized distances), and that the impact observation was at or beyond the normal range for reference locations.

To test the interval null hypothesis (see Table 2.1), we compare the observed F statistic with the upper critical $F_{0.05;HO2}$ value from the non-central F distribution (Figure 2.2). In this case, the observed univariate F statistics were less than the upper critical values (Table 2.1),

whereas the observed bivariate F statistic (262.7) was much greater than the upper critical value of 95.12. Based on these results, the univariate null hypotheses are not rejected and the conclusion is that the true univariate differences were $\leq 1.96\ \sigma$. The bivariate comparison, however, was significant with the conclusion that the bivariate difference was > 2.45 generalized distances (i.e., that the normal range of the reference locations did not include the observation from the impacted location). Rejection with the bivariate test occurred as a result of the high correlation between abundance and richness. In Figure 2.3, the ellipse about the reference data portrays an estimate of the true 95% region (i.e., normal range). The impact observation falls at approximately the upper limit of the normal range for abundance and the lower limit of the normal range for richness. However, because of the correlation between abundance and richness, the impact observation falls well outside of the normal range (Figure 2.3).

2.5 Considerations for Choosing a Test

If our concern as environmental managers is to know when there are aberrant occurrences, we should choose a test that has an adequate chance of leading us to the correct conclusion. When we are comparing a single non-replicated and non-random impacted site against a set of random and replicate reference locations, there are three tests to choose from: (1) the two-sample contrast; (2) the one-sample non-central interval test; and (3) the one-sample, non-central equivalence test. The benefit of using the equivalence test is that it will provide a signal 95% of the time when an impact location has a value that is truly at the limits of the normal range of variation for reference locations. The penalty for using this test is that it will fail locations that are truly part of the reference population 26% of the time. To protect against failing impact sites that are truly part of the reference population, one could resort to the two-sample contrast. However, when impacts are truly just in excess of the normal range of variation, the two-sample test has a poor chance of providing a signal with a small sample size (Figure 2.1). Of the three tests, the interval test is the least protective of the environment because it provides a signal only if the true effect is substantial. Both the equivalence and two-sample tests would provide a signal of a problem well in advance of the interval test. The various benefits and penalties for use of the various tests should be considered by those assessing single impacted locations against a set of reference locations.

Choosing between the various tests could depend on the costs of inferring there is an effect when there is no effect, and the cost of inferring no effect when there is one (Mapstone, 1995). If we infer that there is an effect when there is no effect, we will impose remedial measures in situations where they are not required. If we infer that there is no effect when there really is one, we essentially fail to detect (or report) significant effects when they occur, and lose the value of the environmental resources being protected. If the costs of unnecessary remediation are equal to the costs associated with unmitigated environmental damage, Power et al. (1995) demonstrate that environmental management is cost efficient if tests such as the two-sample contrast and interval tests (i.e., those that err in favour of development) are incorporated into decision making processes. However, because it is difficult to estimate the costs of environmental damage, the costs of Type II errors are difficult to quantify (Peterman, 1990). As a result, Peterman (1990) recommends that a responsible approach to environmental management, is to assume that the costs of environmental damage are greater than the costs of unnecessary remediation. We should therefore use equivalance tests that are more likely to err on the side of environmetal protection because of the inherent assumption (i.e., H_0) that significant effects have occurred.

2.6 Summary

There are many proponents for the use of normal ranges (e.g., 95% regions) to evaluate the condition of ecosystem characteristics (Kersting, 1984, 1988, 1991; DFO and EC, 1995; Findlay and Kasian, 1996; Yan et al., 1996). The 95% rule provides a generic criterion to assess the ecological relevance of environmental impacts through direct comparison to the normal range of variation at reference locations. This criterion can be used for any parameter, irrespective of the scale of measurement. Moreover, no prior information is required, although a representative set of reference-location samples is critical. Some hesitation to incorporate normal ranges into monitoring programs is anticipated because of the perception that a great deal of effort is required to characterize the normal range (e.g., Yoder, 1989; Hughes, 1995; Reynoldson et al., 1995; Yoder and Rankin, 1995; Wright, 1995). However, it has been shown here that using non-central tests, a modest number of reference locations (which can be calculated *a priori*) will provide reasonable power for detecting locations that are truly outside the normal range. Three tests for

comparing an impacted location against a set of reference location observations include two-sample contrasts, and one-sample non-central equivalence and interval tests. Using equivalence tests, we can determine the probability that the observation at an impacted location falls exactly at the limits of the normal range. One penalty of the test is that impact locations just inside the normal range have a high probability of leading to an incorrect conclusion of impact. A second potential penalty of the test is that it could fail sites that are truly part of the reference population. Both two-sample and interval tests require that the impacted location be considerably in excess of the normal range of variation before a site would be failed. However, sites that are truly part of the reference population would be failed only 5% of the time with the two-sample test and 1% of the time with the interval test. Choosing a particular test could be based on the costs of making incorrect conclusions.

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Table 2.1. Summary of statistical tests and associated null and alternate hypotheses discussed in the text. The critical effect size is 1.96 σ for single-variable responses, 2.45 generalized distances for two-variable responses, and more generally $\sqrt{X^2_{(0.95,p)}}$ for multivariable responses, where p is the number of variables.

Type of test	Number of response variables	Null Hypothesis (H_0)	Alternate Hypothesis (H_A)
tradition test	1	the impact location is a member of the reference population	the impact location is not a member of the reference population
	2	the impact location is a member of the reference population	the impact location is not a member of the reference population
	>2	the impact location is a member of the reference population	the impact location is not a member of the reference population
interval test	1	$\lvert\mu_r\text{-}Imp\rvert \leq 1.96$	$\lvert\mu_r\text{-}Imp\rvert > 1.96$
	2	$\lvert\mu_r\text{-}Imp\rvert \leq 2.45$	$\lvert\mu_r\text{-}Imp\rvert > 2.45$
	>2	$\lvert\mu_r\text{-}Imp\rvert \leq \sqrt{X^2_{(0.95,p)}}$	$\lvert\mu_r\text{-}Imp\rvert > \sqrt{X^2_{(0.95,p)}}$
equivalence test	1	$\lvert\mu_r\text{-}Imp\rvert \geq 1.96$	$\lvert\mu_r\text{-}Imp\rvert < 1.96$
	2	$\lvert\mu_r\text{-}Imp\rvert \geq 2.45$	$\lvert\mu_r\text{-}Imp\rvert < 2.45$
	>2	$\lvert\mu_r\text{-}Imp\rvert \geq \sqrt{X^2_{(0.95,p)}}$	$\lvert\mu_r\text{-}Imp\rvert < \sqrt{X^2_{(0.95,p)}}$

Table 2.2. Results from an example two-sample, interval and equivalence contrast. In each of 10 reference locations and a single impacted location, abundance and richness of an ecological community were quantified. Univariate and multivariate test statistics for the comparison between reference and impact are calculated, while critical upper and lower F values from a non-central F distribution, and critical values from a central F distribution are determined. See Figure 2.3 for data.

Statistics	Log Abundance	Richness	Bivariate Contrast
reference average $(\bar{X}_r)$	0.806	38.8	
difference $(\lvert \bar{X}_r - Imp \rvert = \delta)$	2.194	18.8	
standard deviation of reference data (S_r)	0.763	8.066	
generalized distance (δ/S_r)	2.875	2.331	7.69
squared generalized distance	8.279	5.434	59.11
2-Sample Test			
test statistic	7.53	3.45	23.88
degrees of freedom	1, 9	1, 9	2, 8
two-sample test critical F value ($F_{0.05;HO1}$)	5.12	5.12	4.46
1-Sample Tests			
test statistic	82.79	54.3	262.7
degrees of freedom	1, 9	1, 9	2, 8
interval test critical F value ($F_{0.05;HO2}$)	115.8	115.8	95.12
equivalence test critical F value ($F_{0.05;HO3}$)	16.2	16.2	13.67

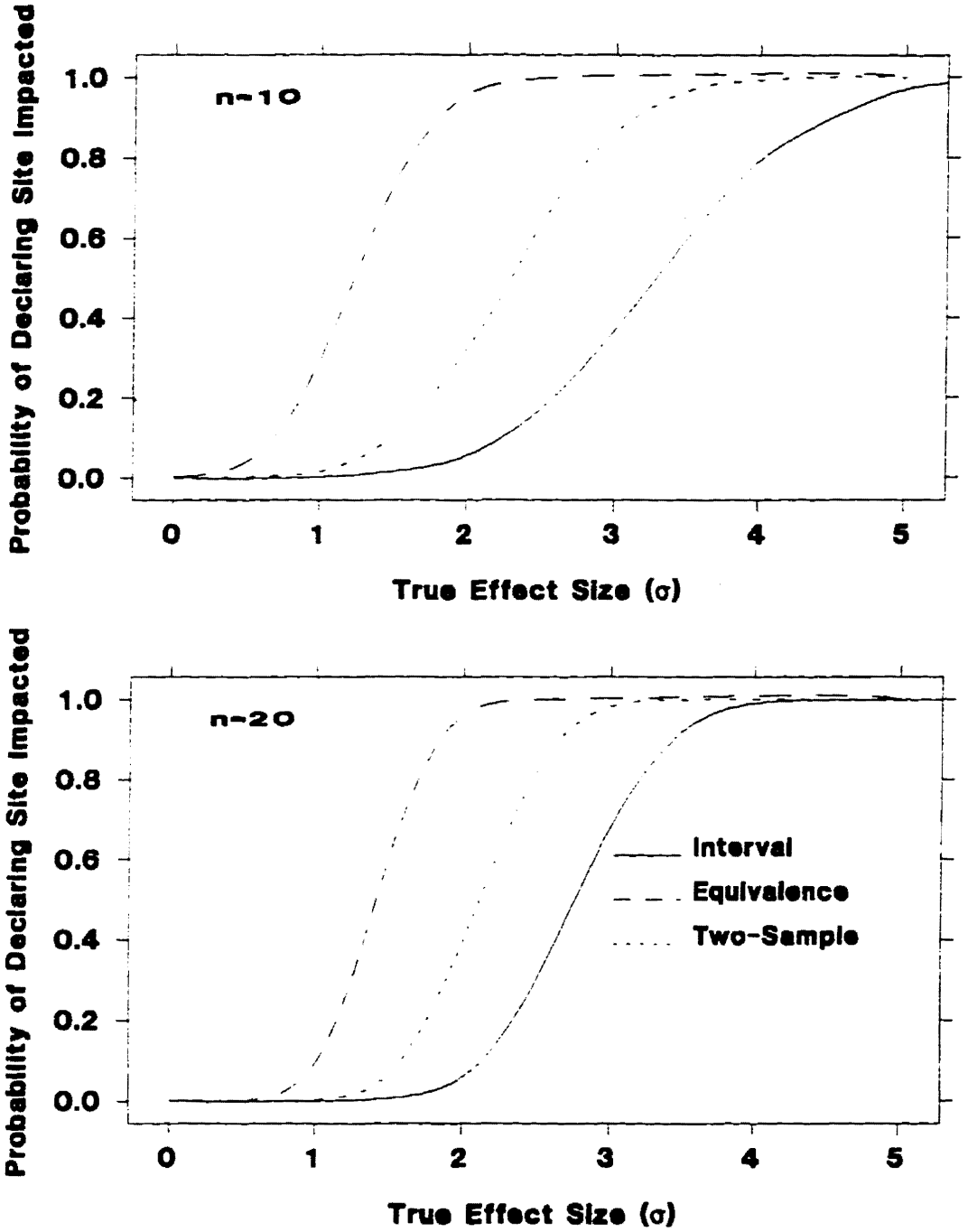


Figure 2.1. Power curves showing the probability of declaring a site impacted when the differences between the reference population mean and the impact observation are some specific (magnitude expressed as standard deviations, σ) for three different tests. Power curves are given for sample sizes n_r = 10 or 20.

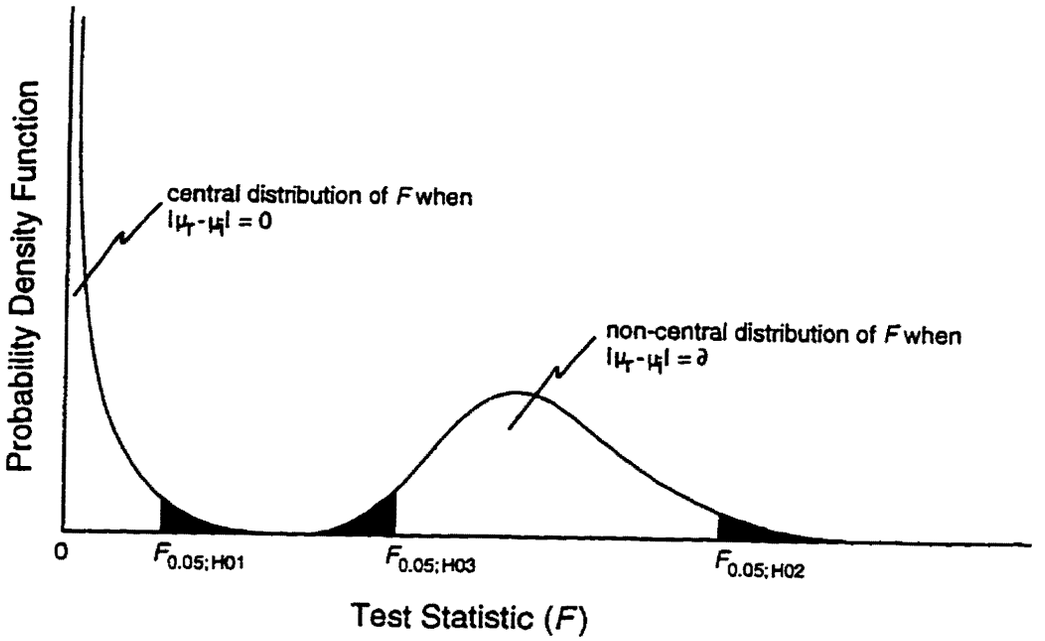


Figure 2.2. Probability density function of test statistics when the difference between a mean of a reference population and an impact location is 0 or when the difference is some true value (δ). The shaded regions of each distribution represent 5%. $F_{0.05:HO1}$, $F_{0.05:HO2}$ and $F_{0.05:HO3}$ are the critical F values used to test each of the first (i.e., two-sample), second (interval) and third (equivalence) null hypotheses (see text) respectively.

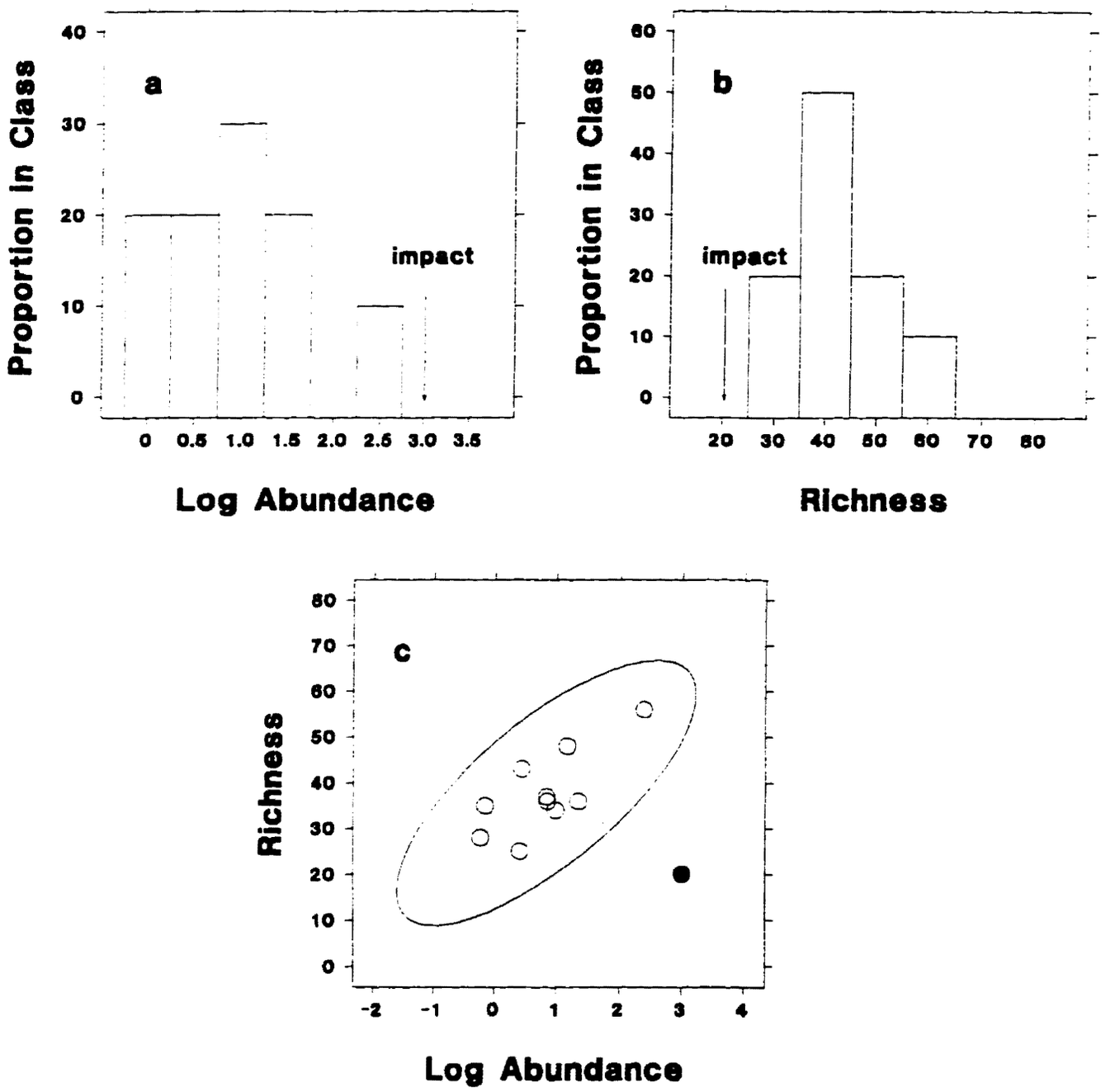


Figure 2.3. Presentations of the simulated log abundance and richness data used in the example. (a) and (b) are histograms showing the impact observation relative to the distribution of reference data. (c) is a bivariate plot of the raw data. In (c), a 95% confidence ellipse encloses the reference observations (open circles) while the impact observation is indicated by the filled circle.

CHAPTER 3

Estimating effect sizes with electrofishing surveys of stream fish communities: comparisons among single- and multi-pass data and community descriptors

3.0 Abstract

I evaluated the sensitivities of one- and three-pass electrofishing data and various fish community descriptors to determine combinations of methods that could reliably detect the effects of urbanization/reservoirs and agriculture on stream fish communities. In this analysis, six brook trout streams were used as regional reference streams against which to evaluate 31 other streams variously impacted by agricultural practices, urbanization and reservoirs. Estimated impacts were greatest when impacts were measured using percent-model affinity (PMA) on one- or three-pass biomass data. In contrast, estimated impacts were much less for estimates based on species richness, or PMA based on presence/absence information. Surveys conducted in the fall resulted in marginally larger differences between reference and impacted fish communities. Because of the increased sensitivity of biomass data, fisheries assessments are more likely to detect ecologically relevant impacts when surveys quantify biomasses of individual species using either single- or three-pass sampling, than when abundances or presence/absence are recorded.

3.1 Introduction

Surveys of lotic-fish communities are commonly used to evaluate and quantify the effects of anthropogenic disturbance on aquatic systems. Typically, surveys of stream fish incorporate a set of either regional-reference (Hughes, 1995) or site-specific reference streams (Hodson et al., 1996) against which other test streams are judged. In Canada, the Federal *Fisheries Act* has a policy of "no net loss of the productive capacity of fish or fish habitat". In general, this policy has been interpreted liberally to imply that any change in any descriptor of individual fish species, fish communities, or fish habitat, is a deleterious change, and is a change worth some form of action. Unfortunately, such loose interpretations can be costly either to the environment when we have failed to measure an appropriate ecosystem endpoint and miss ecologically relevant effects on fish when they occur, or in terms of dollars spent on remediation of an ecosystem when we have measured a significant change in an endpoint that has little relationship to the *Fisheries Act*. The objective of this paper is to determine how well a variety of methods for the collection and description of fish data quantify the degree of impact on stream communities affected by agriculture, urbanization and reservoirs (impoundments).

Generally, the goals of the *Fisheries Act* are interpreted to mean that the productive potential of game or commercially important species should be protected (Stoneman et al., 1996; Stanfield et al., 1996). This approach is logical for obvious economic reasons, while some (Edwards et al., 1990) add that the protection of game or commercial species (e.g., lake trout) results in the protection of overall ecosystem condition. Such goals generally lead to field surveys that quantify trout abundance or biomass in streams (Jones and Stockwell, 1995) for either assessment or modelling purposes, even though the quantification of all species at a site has more obvious ecological relevance. For single-species or whole-community surveys, abundance, biomass and presence/absence have all been used to characterize the fish fauna of streams (Wichert, 1994a,b; Jones and Stockwell, 1995; Fitzgerald, 1996) with little consideration of relevance to management goals or potential interpretational implications. Surveys that result in quantification of biomass of all community members are most time consuming, but logically more related to the overall goals of the *Fisheries Act*. Our ability to prevent and detect changes in the productive capacity of trout or other fisheries resources depends on our ability to perform surveys that accurately, and cost-effectively, depict the fishery resource we are interested in

protecting. For assessment purposes, field methods and data analyses should result in endpoints or descriptors that have adequate statistical power for detecting effects (Green, 1989) of some relevant magnitude (McBride et al., 1993; Mapstone, 1995; Chapter 2). Selected endpoints should also have an obvious relationship to the management goals of the assessment (Cairns et al., 1993). Statistical power refers to the probability of saying that a stream-fish community is impacted when it truly is. Power is therefore of critical importance when designing impact assessments for stream-fish communities. Power is dependent on: (1) the true degree of difference between the reference condition and the impacted community; (2) the degree of variation of the community descriptor within the reference condition; and (3) the number of replicate locations used to characterize the reference condition (Green, 1989). Investigators can therefore influence the power of a survey by: (1) increasing the number of reference locations at which the fish community descriptor is estimated; (2) selecting a community descriptor that has low variance in the reference condition; and (3) selecting a community descriptor that is responsive to the imposed stressor. Together, low variance and responsiveness result in large effect sizes (i.e., standardized differences between two treatments, Cohen, 1977). By incorporating powerful descriptors into study designs and analyses, investigators have greater potential to identify problem areas before effects become too severe.

Although several stream-fish investigators have explored the effects of various samplers and sampling regimes on descriptors of fish resources (e.g., Wiley and Tsai, 1983; Angermeier and Karr, 1986; Fisher, 1987; Vadas and Orth, 1993; Angermeier and Smogor, 1995; Jones and Stockwell, 1995), few have examined the resolution of different methodologies or community descriptors between reference and impacted locations. (except see Jones and Stockwell, 1995; Peterson and Rabeni, 1995). For stream fishes, only Peterson and Rabeni (1995) has evaluated different community endpoints associated with stream-fish surveys. However, their interest was in determining the number of stream sections required to attain certain precision in estimates of fish community descriptors for a whole reach. For lake systems, Minns et al. (1996) examined temporal variations in fish community endpoints including species richness, total abundance and total biomass and production.

The objective of this study, therefore, is to determine the types of stream fish community descriptors that would provide high statistical power for evaluating effects. I focus on

community descriptors related to the assessment of the condition of trout streams because strong populations of trout are common goals for southern-Ontario watersheds (e.g., Jones and Stockwell, 1995; Stoneman et al., 1996; Stanfield et al., 1996). Rather than estimate the statistical power of each descriptor (since power is in part dependent on the number of reference locations described and the study design), I focus on the other parameter influencing power, the estimated effect size (Cohen, 1977). Under similar sample sizes and study designs, those variables that result in larger effect sizes will be more powerful (i.e., more likely to provide evidence of an impact when in fact an impact has occurred).

Biomass-related whole-community endpoints are ecologically relevant because whole communities are described. Because of the strong relationship between standing-crop biomass and annual production (Downing et al., 1990; Kelso and Johnson, 1991; Randall et al., 1995), whole community characterizations that incorporate standing-crop biomass are also relevant to the goals of the *Fisheries Act*. I therefore use biomass-related whole-community endpoints as a yardstick against which to compare other characterizations.

For this analysis, I combine electrofishing data from Bowlby and Roff (1986) with some new data collected in 1994. I considered six of 37 streams as reference trout streams, while the remainder were variously impacted by agriculture, urbanization and reservoirs. I compared sensitivities for estimates of brook trout and total-trout abundance and biomass, species richness and a multivariate descriptor of composition (percent-model affinity, PMA, Novak and Bode, 1992). For each of these descriptors, I compared single- vs multi-pass data as well as spring vs fall collections. I also compared the sensitivities of surveys for detecting effects associated with agricultural practices and combined urban/reservoir effects. In this chapter, I intentionally avoid comparisons of the PMA-derived indices with other indices such as the Index of Biotic Integrity (IBI, Karr, 1981), primarily because such indices are appropriately applied only to the region in which they were derived (e.g., Fausch et al., 1984). In addition, affinity models make no assumptions regarding the environmental sensitivities or the intrinsic value of the species found. Rather, those species characteristic of the reference locations build the model of expectation for acceptable conditions.

3.2 Methodology

3.2.1 Study Area

Fish communities of 37 1st-4th-order streams in southern Ontario (Figure 3.1; Appendix A) were surveyed. Nine of these streams were surveyed by Bowlby and Roff (1986) in 1982; the remaining streams were surveyed in 1994 as part of a study specifically designed to compare invertebrate and fish responses to impoundment, urbanization and agricultural activities (Kilgour et al., 1996). The majority of streams were tributary to Lake Erie through the Grand River watershed, while two of Bowlby and Roff's (1986) streams were tributary to Lake Ontario via Black Creek and the Credit River. A final six streams were tributary to Lake St. Clair via the Thames River watershed.

3.2.2 Fish Collections

At each stream, 6-mm square-mesh blocking nets were used to isolate riffle-pool-run sequences and an electrofisher was used to capture fish within the blocked sections. After each fishing episode, fish were counted, weighed and released downstream of the lower blocking net. Collections were made in May and September of 1982 (Bowlby and Roff, 1986) and May and September of 1994. Total abundances and biomasses for each species at each stream site were estimated using Carle and Strub's (1978) equations for each collection.

3.2.3 Fish Community Descriptors

Bowlby and Roff's (1986) data were incorporated into this study because their stations were primarily good trout streams and would serve as relevant reference fish communities for this exercise. Of the 30 stream sites that they surveyed, nine were surveyed in both May and September of 1982, so they were included in this exercise. Of those, six were numerically dominated by brook trout in both spring and fall surveys and were considered representative reference streams against which to judge the degree of impact of the streams surveyed in 1994. Principal Coordinates Analysis (PCo-A) was used to portray the similarities in species composition among stations, and to demonstrate that the six streams selected as reference locations, did in fact represent a cluster of sites that was distinct (Figures 3.2 and 3.3). PCo-A is an ordination technique that summarizes, in two or three dimensions the associations among

communities (Rohlf, 1993). Communities close together in the ordination diagram have a similar assemblage of organisms, while communities far apart in the ordination diagram have different assemblages. In this case, I used PCo-A to ordinate Bray-Curtis (Rohlf, 1993) distances between pairs of fish communities for both spring and fall collections. The Bray-Curtis distances between pairs of sites were based on $\log_{10}$ transformed estimated total biomasses (g/m^2) for each species.

Brook and total trout abundance and biomass, as well as overall species richness were estimated for each site based on single- and three-pass electrofishing data. In addition, I also estimated the affinity of impacted locations to the average reference location using Novak and Bode's (1992) percent model affinity (PMA) approach. Novak and Bode (1992), and later Barton (1996) and Bailey (1996) used a percent similarity of community (PSC) coefficient to measure the similarity of impacted communities to the average reference community for their impact assessments. In this analysis, I estimated PMA of impact locations to the average reference location using Bray-Curtis coefficients on $\log_{10}$ transformed species biomasses and abundances as well as percentage biomasses and abundances. The Bray-Curtis coefficient is a commonly used community coefficient for quantitative or proportional data (Legendre and Legendre, 1993; Rohlf, 1993). The coefficient is calculated using the formula:

$$d_{ij} = \frac{\sum_k |x_{ki} - x_{kj}|}{\sum_k (x_{ki} + x_{kj})} \qquad [1]$$

where d_{ij} is the Bray-Curtis distance between two samples i and j, k refers to the kth species, x_{ki} refers to the abundance of species k in sample i and x_{kj} refers to the abundance of species k in sample j (Rohlf, 1993). Two samples with identical species composition will result in a Bray-Curtis distance of zero, while two samples with no species in common will result in a Bray-Curtis distance of 1. Bray-Curtis calculations are very similar to those of the PSC coefficient used by Novak and Bode (1992) and Barton (1996).

I also used Jaccards coefficient (JC) to measure the similarity of impacted locations to the average reference location based on presence/absence data for fish species. The Jaccard coefficient is

$$JC=\frac{a}{a+b+c} \quad [2]$$

where a represents the number of species in common, b is the number of species found in sample j but not i, and c is the number of species found in sample i but not j (Rohlf, 1993). The Jaccard coefficient was used to estimate PMA with the presence/absence data because it is one of the more commonly used coefficients for that type of data (Legendre and Legendre, 1993; Rohlf, 1993). With Jaccard's coefficient, samples with all species in common will result in a value of one, whereas samples with no species in common will result in a value of zero. In order for the Jaccard's coefficient to complement the Bray-Curtis coefficient, I used the inverse of the Jaccard's coefficient such that small values indicated similarity while large values indicated dissimilarity.

As a multivariate approach, PMA (i.e., use of raw similarity or dissimilarity measures) is becoming more commonly used in ecological assessments (e.g., Smith et al., 1990; Novak and Bode, 1992; Barton, 1996; Bailey, 1996), primarily because of its simplicity. In this case, the average reference fish community was the average of Bowlby and Roff's six reference streams. PMA for each fish community to the average reference community was based on single- and three-pass abundance, biomass and presence/absence data (Table 3.1). Species abundances were expressed as catch-per-unit-effort (CPUE) for single- and three-pass catches, as well as % of total catch for single-pass catches. Effort was expressed as the area (m^2) of stream shocked. For % and CPUE data, Bray-Curtis distances estimated the degree of similarity of sites with the average reference community, whereas for presence/absence data, Jaccards' similarity measure was used.

3.2.4 Estimating Effect Sizes

Because each community descriptor calculated in this analysis has the potential to be affected/influenced by a different suite of anthropogenic stressors, there may be variations in what the community descriptors are sensitive to (e.g., brook trout biomass may be more sensitive than a percent-affinity model when the stressor is urbanization, whereas the reverse may be true when the stressor is agricultural activity). However, regardless of the stressor, the ultimate insult on

a fish community will result in complete elimination of fish. I therefore, calculated the maximum possible impacts on each community descriptor for a stream with no fish (i.e., obviously seriously degraded) and compared the probability of detection of such an effect when the number of reference sites was six. The degree of impact was expressed as:

$$Effect\ Size = \frac{(\bar{X}_r - Imp)}{SD_r} \quad [2]$$

where $\bar{X}_r$ is the estimated average reference community response, *Imp* is the observed response of the community at an impact location, and SD_r is the estimated standard deviation of responses at the reference location (Chapter 2). If one were interested in testing the null hypothesis that there was no difference in the value at an impact site and the mean of the reference location responses (or either of the interval or equivalence null hypotheses described in Chapter 2), those descriptors with large effect sizes would be more likely to result in inferences that significant impacts had occurred. For example, in Table 3.2, the maximum effect size for CPUE biomass PMA for a community with no trout was estimated to be 4.55 *SD*s, whereas for brook trout abundance the estimated maximum effect size was 0.56 *SD*s. Whether one tested a conventional, interval or equivalence null hypothesis, one is more likely to reach the correct conclusion that there has been a significant (i.e., one exceeding the normal range) impact on the fish community if one used the CPUE biomass PMA as the community descriptor.

In Figures 3.2 and 3.3, each stream was categorized according to whether the immediate and dominant landuses within its watershed was forest, agricultural or urban, and whether or not it was downstream of a major reservoir. These categorizations were based on maps at 1:50,000 scale. Trout reference streams were those forested streams dominated by trout in both spring and fall collections. Trout non-reference streams were those forested streams in which trout were absent during either a spring or fall collection. Based on the configuration of sites in Figures 3.2 and 3.3, it appeared that urbanization and reservoirs had effects on fish communities that were distinct from those of agriculture or urbanization. Since these were the main stressors affecting fish in this collection of streams, I also determined the average effect size for streams in

urban/reservoir systems and in agricultural systems.

3.2.5 Associations Among Descriptors

To more fully understand why effect sizes estimated using one survey method were more or less sensitive than others, I then examined the relationships among the various community descriptors. This was done in two steps. First, a correlation matrix that summarized the associations among descriptors was constructed. This correlation matrix was based on Mantel correlations (Mantel, 1967; Rohlf, 1993) of distance/similarity matrices of the various community descriptors. Mantel correlations are a measure of the concordance between the off-diagonal elements of similarity/dissimilarity matrices. For example, Bray-Curtis distances were used to summarize similarities of fish communities based on abundances and biomasses, while Jaccards coefficient was used to summarize presence/absence information. The Mantel correlation between a Bray-Curtis distance matrix and a Jaccard coefficient matrix was used to represent the association between the two descriptors. For univariate descriptors like brook and total trout abundances and biomass, and species richness, euclidean distances between pairs of stations were used to create the dissimilarity matrix. Mantel correlations between these euclidean distance matrices and the Bray-Curtis and Jaccard matrices were similarly determined.

The second step in demonstrating (lack of) concordance among community descriptors was to portray the similarities among community descriptors graphically. To do this, PCo-A was used to summarize the Mantel correlation matrix and to portray the similarities in two dimensions (Figure 3.4). Community descriptors close together in this PCo-A diagram were highly correlated (i.e., they tended to consider similar sites similar, and dissimilar sites dissimilar). In contrast, community descriptors far apart in the PCo-A diagram can be considered to be poorly correlated. This particular ordination diagram (Figure 3.4) shows that all of the trout-related descriptors were highly correlated, but that they had only low correlations with the PMA descriptors (i.e., Mantel correlations of $\approx$ 0.4). Similar ordination diagrams were obtained for the spring and fall collections, so only the spring diagram is given.

3.3 Results

The analyses conducted demonstrate that there were differences in the nature of responses of the

fish community descriptors to the various stresses related to urbanization/reservoirs and agriculture. Biomass-related percent-affinity models demonstrated consistently greater sensitivity than any of the other descriptors (Tables 3.2-3.4). Trout-related endpoints were also poorly correlated with PMA endpoints (Figure 3.4). There were some seasonal changes in community composition at some sites. Some of the urban/reservoir sites clustered with agricultural streams during spring, whereas there were fewer similarities during fall (Figure 3.3). Table 3.5 gives the average composition of fish communities during spring and fall. The major seasonal changes included higher biomasses of northern hog suckers and golden redhorse suckers in agricultural streams during the fall, more northern redbelly dace, bluntnose minnow, fathead minnow and rock bass in reference streams during the spring, and more common shiner in reference streams during fall. Estimated effect sizes were marginally lower for the spring surveys, particularly for the trout-related abundance responses (Tables 3.2-3.4).

Biomass-related percent-affinity models resulted in consistently larger observed effect sizes than did the other community descriptors (Table 3.2-3.4). Further they provided more of a graded response from reference to impact locations than did descriptors of only the trout community. Table 3.5 shows that there were no trout in any of the urbanized streams, meanwhile, similarities between urbanized streams and reference streams were clearly graded (Figures 3.2 and 3.3). Consequently, use of similarity/dissimilarity measures would provide more information that could be used in modelling the association between a fish community and any environmental variable, than would the abundances or biomass of a single target species such as brook trout.

As a result of the PMA endpoints being more sensitive to the various stressors, and because they provide more potential for modelling, I conducted a final analysis to determine which of the PMA endpoints were as sensitive as the biomass-related percent-affinity models. In this last analysis, I wanted to determine if the estimated effect sizes for a given whole community descriptor were consistently more or less than effect sizes estimated using an intensive three-pass method that characterized biomasses of each of the individual species. First, however, I compared effect sizes estimated between spring and fall surveys by examining the relationship between effect sizes estimated using three-pass CPUE biomass PMA in spring and fall. I used the three-pass biomass community characterization because it is more obviously

related to the goals of the federal fisheries act. To make the comparison, I determined the linear equation that described the relationship between the two endpoints. If the two community descriptors (i.e., spring and fall, three-pass CPUE biomass PMA) had similar sensitivities to the various stressors, then the equation describing the relationship between the two descriptors would not be significantly different from a 1:1 line of expectation (i.e., the slope would not be significantly different from one, and the intercept would not be significantly different from zero). After spring and fall had been compared using three-pass CPUE biomass data, I contrasted all other descriptors to the three-pass CPUE biomass community descriptor within both spring and fall sampling periods. For those descriptors for which there was a significant deviation from the line of expectation, I estimated the observed effect size in the community descriptor that coincided with an effect of 1.645 σ in the three-pass CPUE biomass community descriptor. An effect of 1.645 σ represents a fish community that is at the limit of the normal range of variation for reference fish communities using a percent affinity model such as the Bray-Curtis or Jaccard coefficient (Chapter 2).

The relationship between effect sizes estimated using three-pass CPUE biomass collections from spring and fall was described by a relationship that was significantly different than a 1:1 line (Table 3.6, Figure 3.5). In general, when the impacts were large, surveys conducted in the fall were more sensitive (Figure 3.5). The only PMA descriptors to have a relationship with three-pass CPUE biomass that was not different than the line of expectation was single-pass CPUE biomass (both spring and fall) and single-pass CPUE abundance in the spring. All other descriptors were less sensitive than three-pass CPUE biomass (Table 3.6). Of the first pass descriptors, the biomass descriptors were most highly correlated with three-pass CPUE biomass PMA (Figure 3.4, Table 3.6). Snedecor and Cochran (1989, p 188) provide a procedure for determining if two correlation coefficients are significantly different. I used that procedure to determine that the correlation coefficients for the relationships between single-pass biomass percent affinity models and three-pass CPUE biomass PMA were larger ($p<0.05$) than the correlation coefficients for the single-pass abundance related percent affinity models. Consequently, the regression equations and associated statistics suggest that single-pass CPUE biomass and single pass % biomass PMA were more highly correlated with the fully quantitative three-pass CPUE biomass PMA than were the single-pass abundance PMA descriptors.

3.4 Discussion

The streams included in this assessment of fish community descriptors ranged from non-degraded cold-water streams dominated by brook trout, to streams heavily affected by impoundments, agriculture and urbanization, and dominated by more tolerant cyprinids such as common shiner, creek chub and carp (Figure 3.2 and 3.3). The effects of impoundment/urbanization were marginally greater than the effects of agriculture on all fish-community responses with the exception of species richness (Tables 3.4). Impoundments and urbanization are associated with several stressors that could potentially alter a fish community including alterations in oxygen and thermal regimes (Lehmkuhl, 1972; Mackie et al., 1983) and increased suspended sediments (Spence and Hynes, 1971). In addition, altered hydrologic regimes (Leopold, 1968; Lazaro, 1979) can result in changes in water chemistry (Pederson and Perkins, 1986) and in-stream physical habitat characteristics (Fox, 1974; Booth, 1990). Two streams primarily affected by urbanization clustered with the streams primarily affected by agricultural practices suggesting that the streams classified as being affected by urbanization/reservoirs were primarily affected by the reservoirs, not necessarily the urban centers. Both agriculture and urbanization are associated with physical and chemical changes in streams because of alterations in hydrology due to removal of forest cover, tile drains, and paving. The differences in fish communities between agriculture and urbanization, and those found below reservoirs, suggest that the specific causes of changes in fish community composition vary with landuse.

This analysis of different fish community sampling approaches demonstrates that the sensitivity of a study can depend on the methods used (i.e., single- vs multi-pass), the fish community endpoints measured (i.e., PMA vs abundance or biomass of selected species) and, for some descriptors, season. In general, biomass-related percent-affinity models were consistently more sensitive than percent-affinity models based on presence/absence or abundance data and more sensitive than trout-related descriptors. The high concordance among the abundance-related percent-affinity models (Figure 3.4) indicates that biomass-related percent-affinity models do characterize a different community than abundance-related percent-affinity models. For lentic systems, Minns et al. (1996) recommend characterizing the biomass of species rather than abundances because biomass is more stable temporally than abundances.

The biomass-related percent-affinity models have two significant advantages over

biomass-related trout responses. First, the increased sensitivity of the percent-affinity models will result in investigators identifying significant alterations in fish community composition more reliably than when trout-biomasses are estimated. This seems counter-intuitive, particularly if we consider that trout may be the species we are attempting to model, predict or protect. Multivariate descriptors of community attributes are generally more sensitive than single descriptors of composition (Clarke and Green, 1988; Yan et al., 1996). More sensitive multivariate responses suggests reduced within-reference variability in estimating condition of the fish community. The higher variability of single-variable trout-related responses is probably not due to sampling inappropriate stream sections since most trout have home ranges that are about the same size as the average stream section sampled (i.e., 20-30 m; Heggenes et al., 1991; Bridcut and Giller, 1993). Rather, the increased sensitivity of surveys that quantify all fish species is a direct result of having quantified the abundances or biomasses of several species, each of which has a set of environmental tolerances. Every time a new species is quantified, one can more precisely determine the conditions at a given site (Jongman et al., 1995).

The second advantage of percent-affinity models over the trout-related responses is the graded response to anthropogenic stressors. Trout responses and PMA responses were generally poorly correlated because trout demonstrated an "all-or-none" response: i.e., either trout were dominant in a stream or were absent (e.g., Table 3.5). Using the presence or absence of trout to assess the degree of impact would not be very useful. In this analysis, three of Bowlby and Roff's (1986) nine sites had trout during one season, but not the other, and were therefore not considered to be reference streams. However, had an assessment of the degree of impact of these three streams been based on the absence of trout during the one sampling season that they were not found, they would have been shown to be as poor as the other urbanized streams that do not have trout at any time of the year. This simple example shows that the information gained from sampling the whole community is very valuable in allowing us to more precisely determine how similar to reference conditions the site is. Further, without a graded response to disturbance, our ability to determine critical environmental variables is limited. The percent-affinity models provide both graded responses for modelling purposes as well as high sensitivity that gives reasonable confidence that assessments will lead to correct conclusions regarding the degree of impact.

In contrast to the biomass-related percent-affinity models, those percent-affinity models based on abundances and presence/absence had less resolution between reference and impacted locations (i.e., smaller effect sizes). The lower resolution with presence/absence survey data is unfortunate since this type of fisheries data is commonly used to document the condition of stream-fish communities (Wichert, 1994a,b; Fitzgerald, 1996). Presence/absence survey data are inherently less reliable than species counts or biomass for interpreting condition of a site: two sites with substantial differences in environmental condition can have the same suite of fish species, although in different proportions (Jongman et al., 1995).

Season appears to have some effect on the resolution between reference and impact locations with fall surveys, on average, giving greater resolution between reference and impacted locations (Figure 3.5; Tables 3.2-3.4). Some (Carey et al., 1987; Fitzgerald, 1996) report that the overall composition of fish at a site can vary markedly because of reproductive events. Although most stream species have limited home ranges as adults (Gerking, 1953, 1959; Greenberg and Holtzman, 1987; Hill and Grossman, 1987; Heggens et al., 1991; Bridcut and Giller, 1993), some species like white sucker have significant spawning-associated movements (Scott and Crossman, 1973). Changes in species composition in reference and reservoir influenced streams (Table 3.5) suggested that northern hog sucker, northern redbelly dace, brassy minnow, common shiner and rock bass may have made at least moderate movements in these wadeable streams. Such movements for some species are probably responsible for the relatively low correlation between spring and fall estimated effect sizes ($R = 0.736$; Figure 3.5, Table 3.6). In general, however, the average contributions of the majority of species in reference, agricultural and urban/reservoir influenced streams did not change substantially between spring and fall (Table 3.5) supporting the idea that most stream fish have limited movements. Although there were at least some obvious changes in species composition due potentially to movements by a few species, changes within a land use were clearly smaller than differences in composition between land uses (Figures 3.2 and 3.3). Given these results, surveys conducted during spring or fall should generally result in similar conclusions regarding the degree of impact of a site, although estimated effect sizes from fall surveys may be larger than estimates from spring surveys.

Finally, my analyses suggest that single-pass electrofishing data are as sensitive as three-

pass data, particularly when CPUE biomass is measured. This has significant implications for stream fish surveys because it suggests that full quantitative surveys are not required to achieve reasonable sensitivity: more rapid single-pass surveys can be used for assessment purposes. Since less effort is required to quantify the community at a single site, more sites can be quantified for the same budget. However, if CPUE is not recorded, and biomass or abundance are expressed as relative terms (i.e., abundances are expressed as % composition), estimated effect sizes are not as great. Consequently, the resolution between reference and impact locations will be reduced as will statistical power (i.e., the probability of detecting the impact). Although most fishery biologists record the effort (length of stream, fishing time) that has been expended, proportional catch data are often the only data available from older archived information (e.g., Johnson and Owen, 1966; Reed, 1968). Such data are often used as a baseline against which to assess the degree of change that has occurred in a watershed over time (e.g., Wichert, 1994b).

3.5 Summary

This study demonstrates that there are differences in the resolution of different stream survey methodologies. Surveys that measure CPUE biomass resulted in greater differences between reference and impacted fish communities in contrast to surveys that measured either abundance or presence/absence, or proportional biomass or abundance. Surveys that result in single-pass CPUE biomass data resulted in effect sizes that were as large as those for three-pass CPUE biomass data. Season (spring or fall) had some effect on the resolution of the survey, with fall surveys providing marginally larger effect sizes.

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Table 3.1. Community descriptors examined in this analysis. Percent model affinity (PMA) descriptors were based on biomass, abundance and presence/absence of fish species.

Descriptor	Characteristic	# of Passes	Description	Code
PMA	Biomass	1	Catch per unit effort (CPUE) of each species (g/m^2).	b1
		1	Relative (%) contribution of each species to total biomass.	%b1
		3	CPUE for each species based on Carle and Strub (1978) estimates (g/m^2).	b3
		3	Relative (%) contribution of each species to total biomass	%b3
PMA	Abundance	1	CPUE for each species ($\#/m^2$).	a1
		1	Relative (%) contribution of each species to total numbers.	%a1
		3	CPUE for each species based on Carle and Strub (1978) estimates ($\#/m^2$).	a3
		3	Relative (%) contribution of each species to total abundance	%a3
PMA	Presence/Absence	1	Presence or absence of species based on one pass.	p/a1
		3	Presence or absence of species based on three passes.	p/a3
	Brook Trout Abundance	1	CPUE for brook trout after one pass.	brooka1
		3	CPUE for brook trout based on Carle and Strub (1978) estimates ($\#/m^2$)	brooka3
	Brook Trout Biomass	1	CPUE for brook trout after one pass.	brookb1
		3	CPUE for brook trout based on Carle and Strub (1978) estimates (g/m^2).	brookb3
	All Trout Abundance	1	CPUE for all trout after one pass.	trouta1
		3	CPUE for all trout based on Carle and Strub (1978) estimates ($\#/m^2$)	trouta3
	All Trout Biomass	1	CPUE for all trout after one pass.	troutb1
		3	CPUE for all trout based on Carle and Strub (1978) estimates (g/m^2).	troutb3
	Species Richness	1	Total number of species after one pass.	rich1
		3	Total number of species after three passes.	rich3

Table 3.2. Mean and standard deviation (SD) for reference fish community descriptors, and maximum observable effect sizes.

Number of Passes	Community Descriptor	Spring			Fall		
		Mean	SD	Maximum Effect Size	Mean	SD	Maximum Effect Size
Single Pass	CPUE abundance PMA	0.56	0.12	3.67	0.46	0.15	3.60
	% abundance PMA	0.54	0.19	2.42	0.45	0.19	2.89
	presence/absence PMA	0.57	0.29	1.98	0.56	0.21	2.66
	CPUE biomass PMA	0.50	0.11	4.55	0.40	0.14	4.29
	% biomass PMA	0.51	0.16	3.06	0.43	0.19	3.00
	species richness	6.50	4.02	1.63	6.17	2.99	2.06
	brook trout abundance	0.27	0.24	1.15	0.34	0.45	0.76
	brook trout biomass	1.04	0.66	1.57	1.10	0.87	1.26
	all trout abundance	0.28	0.24	1.18	0.36	0.44	0.81
	all trout biomass	1.12	0.61	1.81	1.13	0.85	1.33
Three Pass	CPUE abundance PMA	0.51	0.11	4.45	0.45	0.17	3.24
	% abundance PMA	0.48	0.19	2.74	0.43	0.21	2.71
	presence/absence PMA	0.55	0.25	1.80	0.54	0.20	2.30
	CPUE biomass PMA	0.46	0.13	4.15	0.37	0.14	4.50
	% biomass PMA	0.47	0.18	2.94	0.40	0.18	3.33
	species richness	7.67	4.32	1.78	6.83	3.06	2.23
	brook trout abundance	0.42	0.27	1.57	0.47	0.49	0.94
	brook trout biomass	1.42	0.60	2.37	1.47	0.68	2.14
	all trout abundance	0.43	0.26	1.64	0.49	0.48	1.02
	all trout biomass	1.48	0.57	2.60	1.52	0.63	2.41

Table 3.3. Within-reference-location standard deviation, as well as average difference in raw and standardardized terms (i.e., effect size [ES]) for agricultural streams.

Community Descriptor	Spring			Fall		
	SD	Average Difference	Average ES	SD	Average Difference	Average ES
Single Pass						
CPUE abundance PMA	0.116	0.189	1.63	0.151	0.265	1.75
% abundance PMA	0.185	0.083	0.45	0.193	0.185	0.96
presence/absence PMA	0.294	0.22	0.75	0.214	0.191	0.89
CPUE biomass PMA	0.108	0.282	2.61	0.140	0.351	2.51
% biomass PMA	0.159	0.205	1.29	0.185	0.222	1.20
species richness	4.42	1.75	0.40	2.99	1.167	0.39
brook trout abundance	0.236	0.261	1.11	0.451	0.333	0.74
brook trout biomass	0.663	0.957	1.44	0.874	1.016	1.16
all trout abundance	0.235	0.267	1.14	0.441	0.347	0.79
all trout biomass	0.614	1.028	1.67	0.850	1.047	1.23
Three Pass						
CPUE abundance PMA	0.112	0.234	2.09	0.174	0.209	1.20
% abundance PMA	0.191	0.122	0.639	0.206	0.207	1.00
presence/absence PMA	0.254	0.206	0.810	0.204	0.199	0.98
CPUE biomass PMA	0.128	0.311	2.43	0.135	0.332	2.46
% biomass PMA	0.184	0.227	1.23	0.176	0.25	1.42
species richness	4.32	1.92	0.44	3.06	1.25	0.41
brook trout abundance	0.267	0.405	1.52	0.493	0.442	0.90
brook trout biomass	0.596	1.287	2.16	0.684	1.339	1.96
all trout abundance	0.263	0.418	1.59	0.476	0.463	0.97
all trout biomass	0.570	1.355	2.37	0.631	1.394	2.21

Table 3.4. Within-reference-location standard deviation, as well as average difference in raw and standardardized terms (i.e., effect size [ES]) for streams affected by urbanization and reservoirs.

Community Descriptor	Spring			Fall		
	SD	Average Difference	Average ES	SD	Average Difference	Average ES
Single Pass						
CPUE abundance PMA	0.116	0.376	3.24	0.151	0.479	3.17
% abundance PMA	0.185	0.323	1.74	0.193	0.440	2.28
presence/absence PMA	0.294	0.253	0.86	0.214	0.294	1.37
CPUE biomass PMA	0.108	0.360	3.33	0.140	0.427	3.05
% biomass PMA	0.159	0.336	2.11	0.185	0.395	2.14
species richness	4.42	0.07	0.02	2.99	0.26	0.09
brook trout abundance	0.236	0.272	1.15	0.451	0.343	0.76
brook trout biomass	0.663	1.044	1.57	0.874	1.099	1.26
all trout abundance	0.235	0.278	1.18	0.441	0.357	0.81
all trout biomass	0.614	1.115	1.87	0.850	1.130	1.33
Three Pass						
CPUE abundance PMA	0.112	0.375	3.35	0.174	0.469	2.69
% abundance PMA	0.191	0.322	1.69	0.206	0.420	2.04
presence/absence PMA	0.254	0.199	0.78	0.204	0.302	1.48
CPUE biomass PMA	0.128	0.357	2.79	0.135	0.412	3.05
% biomass PMA	0.184	0.330	1.79	0.176	0.381	2.16
species richness	4.32	0.26	0.06	3.06	1.02	0.33
brook trout abundance	0.267	0.420	1.57	0.493	0.465	0.94
brook trout biomass	0.596	1.415	2.37	0.684	1.467	2.14
all trout abundance	0.263	0.433	1.65	0.476	0.486	1.02
all trout biomass	0.570	1.480	2.60	0.631	1.522	2.41

Table 3.5. Average biomass (g/m^2) of fish in reference, agricultural and urban reservoir influenced streams.

Species	Spring			Fall		
	Ref	Agr	Urb/Res	Ref	Agr	Urb/Res
Brook lamprey		0.14			0.08	
Rainbow trout	4.50			1.18		
Brook trout	50.88	0.79		73.0	1.46	
Northern pike	1.71			1.02		
Central mudminnow	3.16	0.05		8.99	0.08	
White sucker	4.43	2.42	10.9	6.04	1.53	12.3
Northern hog sucker			0.02			1.62
Golden redhorse sucker			0.07			0.24
Northern redbelly dace	7.88	0.02		0.16	0.04	
Carp			5.19			2.51
Brassy minnow	0.26	0.03	0.33		0.01	0.20
Hornyhead chub						0.01
Golden shiner			0.02			
Emerald shiner			0.01			
Common shiner		0.66	0.83	1.05	0.30	1.74
Bluntnose minnow	0.37	0.25	0.06		0.41	0.66
Fathead minnow	0.69	0.03	0.02		0.04	
Blacknose dace	26.7	3.73	0.32	21.8	5.83	0.29
Longnose dace			0.14			0.19
Creek chub	4.90	4.26	1.83	6.33	6.57	1.89
Pearl dace	2.67			3.20		
Central stoneroller		0.33			0.57	
Nocomis sp.	0.34			0.26		
Yellow bullhead			0.16			
Brown bullhead			0.04			
Stonecat						0.25
Brook stickleback	0.59	0.16	0.13	0.36	0.15	0.01
Rock bass	4.16	0.03	0.15		0.18	0.26
Pumpkinseed			0.17			0.66

Species	Spring			Fall		
	Ref	Agr	Urb/Res	Ref	Agr	Urb/Res
Smallmouth bass			0.60			0.17
Largemouth bass			0.49			2.33
Rainbow darter	0.16	0.02	0.02	0.03	0.02	0.02
Iowa darter		0.09	0.10		0.11	0.31
Fantail darter		0.07		0.10	0.07	
Johnny darter		0.16	0.01		0.77	0.04
Mottled sculpin	3.75	0.55		1.58	0.17	

Table 3.6. Regression equations describing the relationships among effect sizes estimated from PMA community descriptors. Also given are effect sizes corresponding with effect sizes of 1.65 standard deviations (SDs) in three-pass CPUE biomass PMA, as well as correlation coefficients (R) and mean-squared-error terms (MSE) for the regressions.

Regression Model	MSE	R	Effect Size (SDs)
Spring vs Fall			
$b3_{spring}$=0.644+0.688 $b3_{fall}$	0.633	0.736	1.78
Spring			
b3			1.65
%b3 = 0.190 + 0.517 b3	0.147	0.846	1.04
b1 = 0.048 + 1.115 b3	0.254	0.933	1.65
%b1 = 0.197 + 0.581 b3	0.207	0.832	1.15
a3 = 0.556 + 0.549 b3	0.149	0.858	1.46
%a3 = 0.272 +0.314 b3	0.330	0.540	0.79
a1 = 0.623+ 0.741 b3	0.857	0.685	1.65
%a1 = 0.143 + 0.417 b3	0.570	0.545	0.83
p/a3 = 0.485 + 0.150 b3	0.112	0.467	0.73
p/a1 = 0.461 + 0.159 b3	0.074	0.565	0.72
Fall			
b3			1.65
%b3 = 0.220 + 0.559 b3	0.188	0.851	1.14
b1 = 0.246 + 0.923 b3	0.154	0.947	1.65
%b1 = 0.336 + 0.485 b3	0.220	0.793	1.14
a3 = 0.609 + 0.610 b3	0.213	0.857	1.61
%a3 = 0.111 + 0.436 b3	0.239	0.857	0.83
a1 = 0.383 + 0.761 b3	0.495	0.806	1.63
%a1 = 0.271 + 0.510 b3	0.387	0.718	1.11
p/a3 = 0.580 + 0.234 b3	0.130	0.632	0.96
p/a1 = 0.524 + 0.227 b3	0.141	0.605	0.90

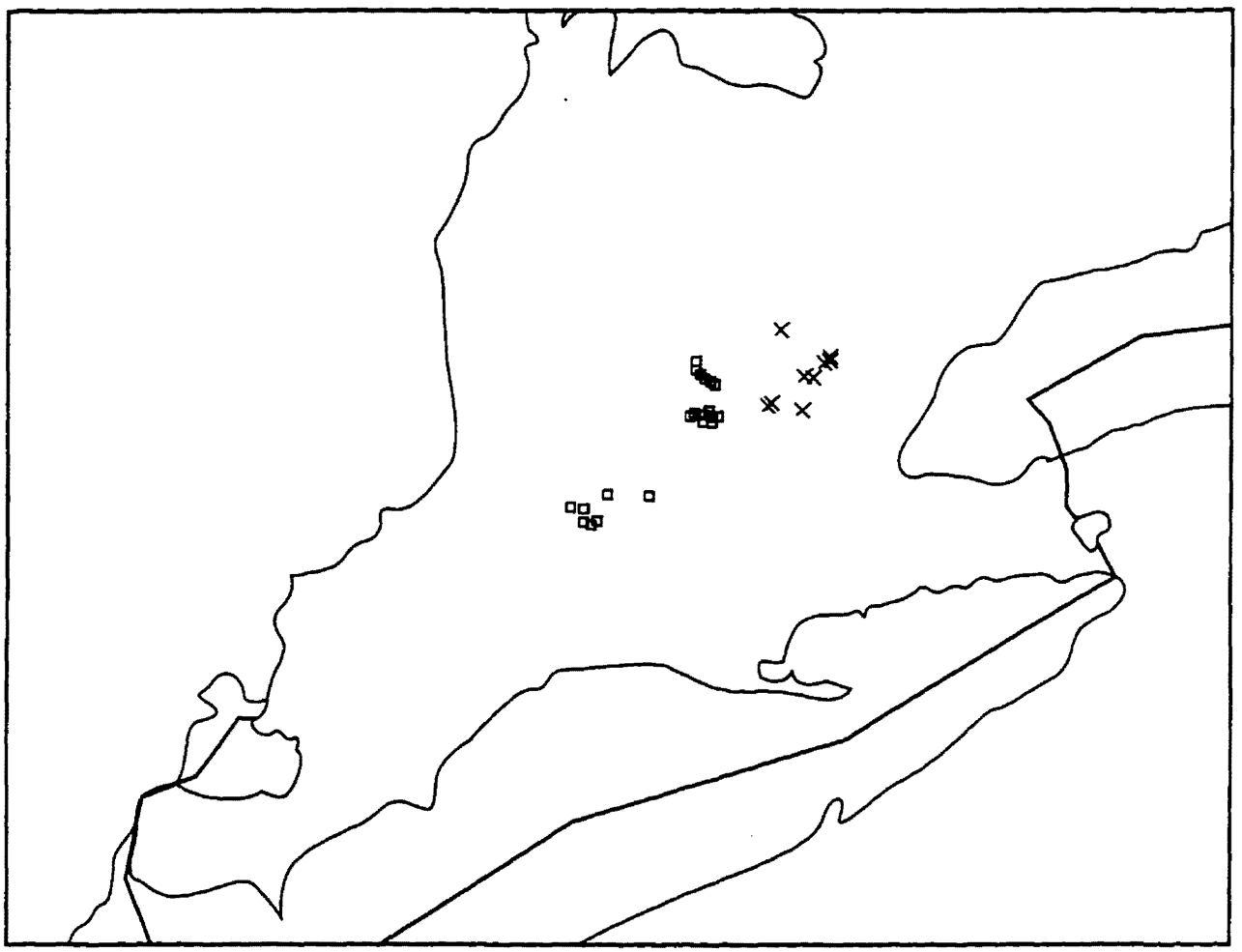


Figure 3.1. Map of southern Ontario showing locations of study sites. Those stations surveyed by Bowlby and Roff (1986) are depicted by x's, while those surveyed in 1994 are depicted by □.

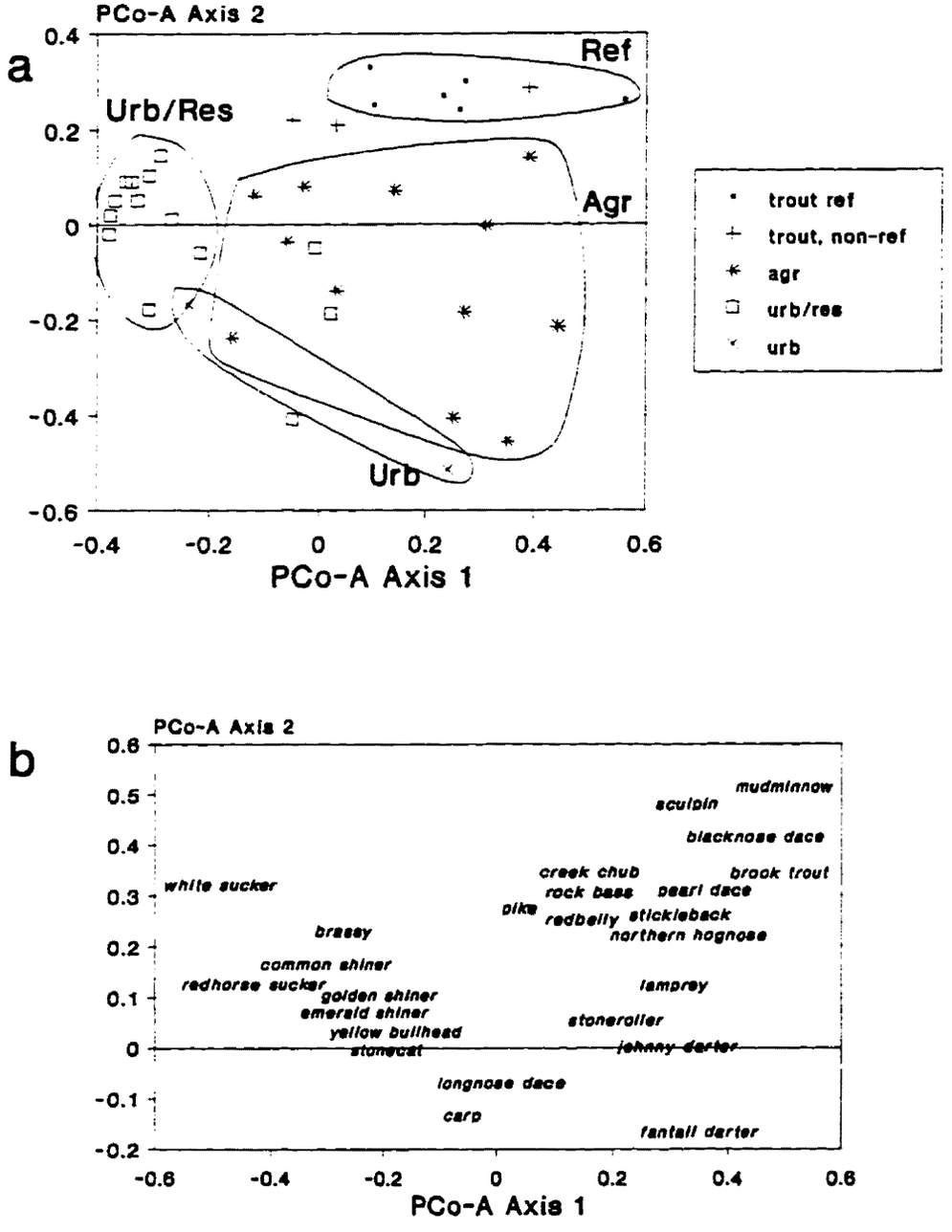


Figure 3.2. Principal coordinates analysis of three-pass-CPUE biomass data from spring electrofishing surveys of 37 streams in southern Ontario. (a) gives sample scores for reference (ref) and non-reference (non-ref) trout streams, as well as agricultural (agr) urban/reservoir (urb/res) and urban (urb) stream sites, while (b) gives taxa correlations with the PCo-A axes. See Appendix B for latin names for species.

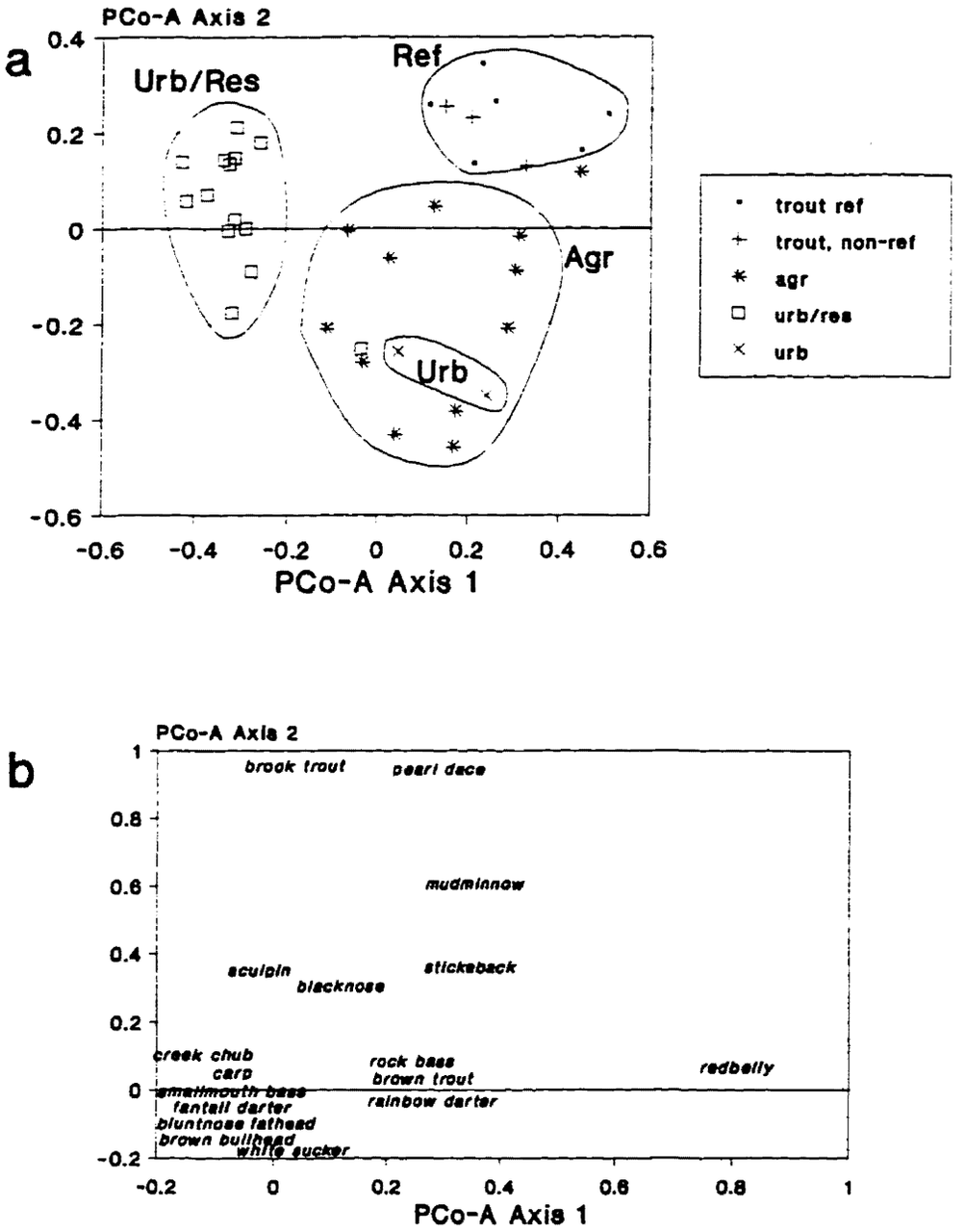


Figure 3.3. Principal coordinates analysis of three-pass-CPUE biomass data from fall electrofishing surveys of 37 streams in southern Ontario. (a) gives sample scores for reference (ref) and non-reference (non-ref) trout streams, as well as agricultural (agr) urban/reservoir (urb/res) and urban (urb) stream sites, while (b) gives taxa correlations with the PCo-A axes. See Appendix B for latin names for species.

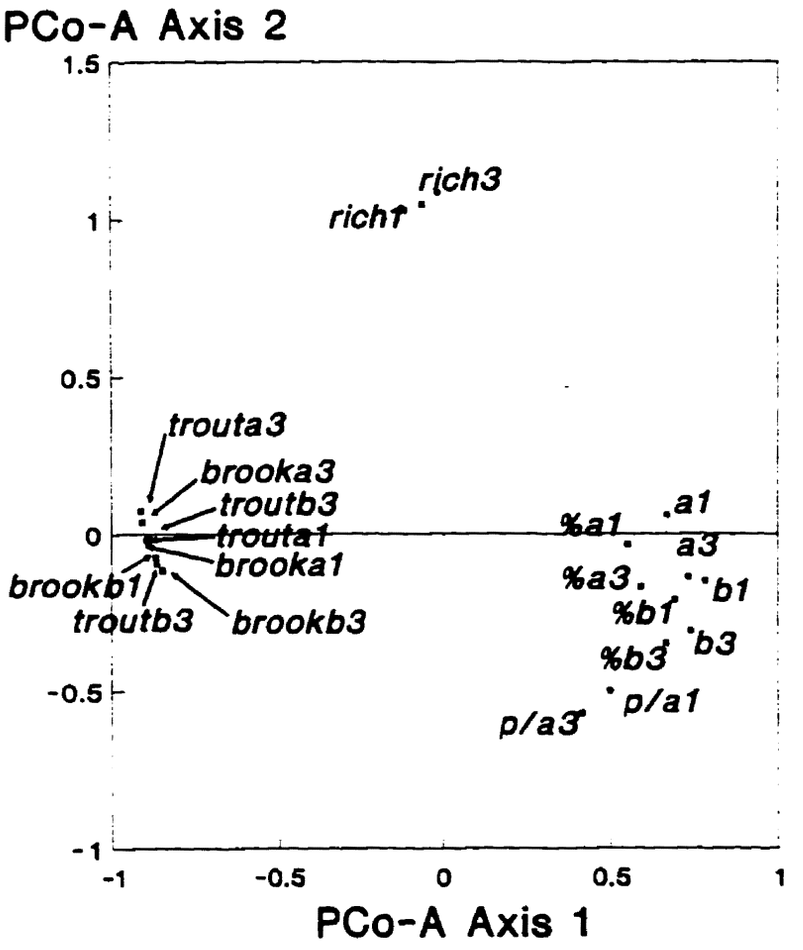


Figure 3.4. Principal coordinates analysis of correlations between fish community descriptors during spring electrofishing surveys of 37 streams in southern Ontario. See Table 3.1 for explanation of codes for the community descriptors.

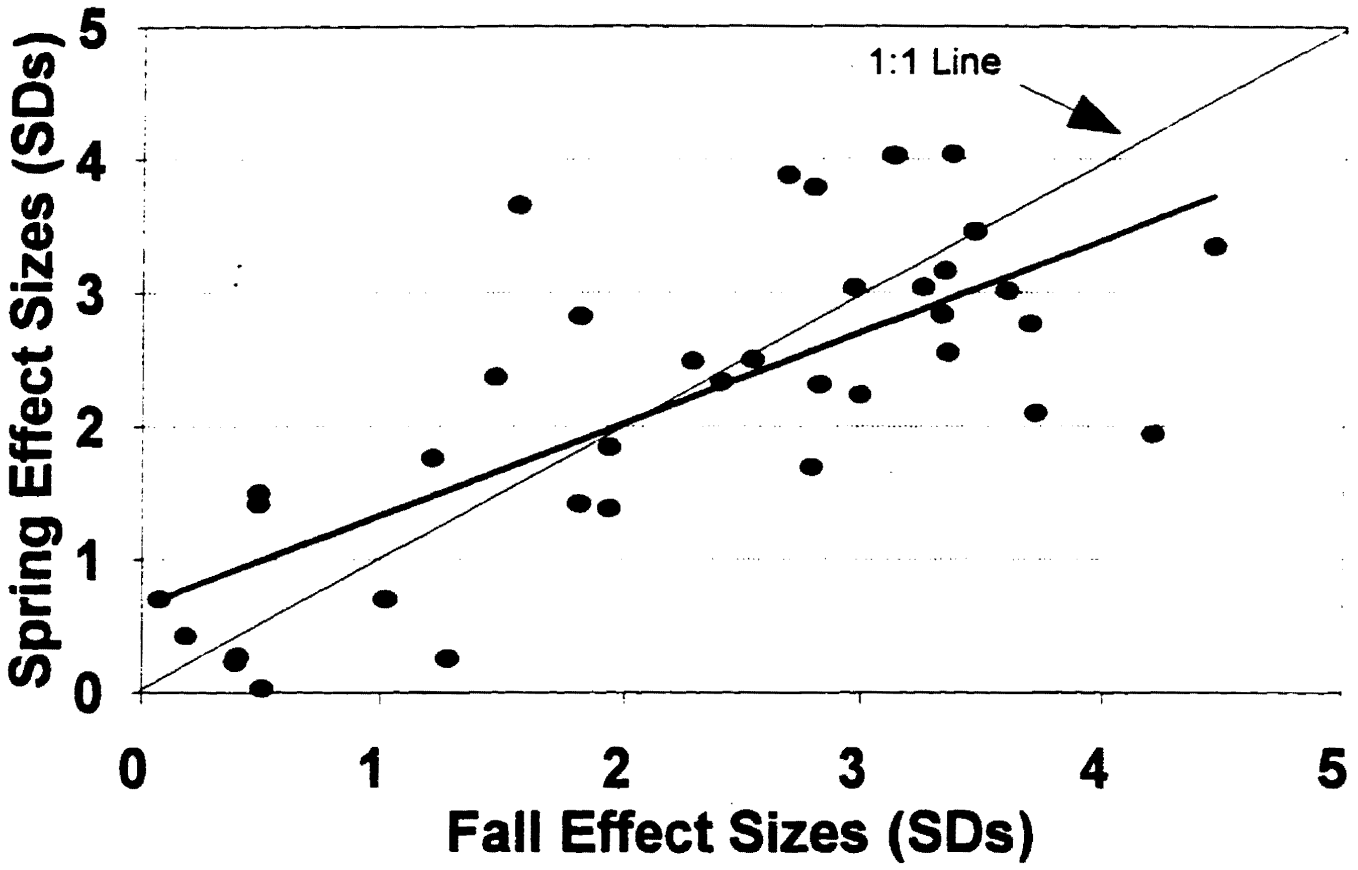


Figure 3.5. Relationship between effect sizes estimated in the spring and fall using three-pass catch per unit effort biomass data. The equation describing the relationship between the two is given in Table 3.6, and is significantly different from the line of expectation (1:1 line).

CHAPTER 4

Fish-benthos-environment correlations in southern Ontario streams

4.0 Abstract

The relationship between fish and benthic invertebrate communities in wadeable streams in southern Ontario was examined using three independent and spatially distinct data sets. For two of the data sets, fish and invertebrates had been collected across a thermal/enrichment gradient, while the third data set was based on collections of fish and invertebrates across a land-use and stream size gradient. For all three data sets, correlations between fish and benthos were higher when benthos were identified to lower taxonomic levels (i.e., species and lowest practical levels). Correlations were low for one study that utilized a "rapid" bioassessment protocol involving field sorting. In two surveys, stream temperatures were important to the distributions of both fish and benthos, while in a third survey, fish and benthos were primarily influenced by the size of the stream (watershed size). In all three studies, fish and benthos were associated with similar suites of environmental variables, suggesting that the fish-benthos association in these streams was driven by corresponding environmental tolerances. In addition, the data suggest that fish and benthos respond to environmental cues at similar spatial scales. Although there was significant variation in the fish-benthos correlation that could be attributed to differences in sampling methodologies, the findings from this study confirm that stream fish and benthos are significantly associated and that surveys of benthos can be used to make inferences on the condition of fish community composition.

4.1 Introduction

Surveys of both fish and benthic invertebrate communities are commonly used tools to assess the effects of human activity on the condition of stream resources. Surveys of fish are somewhat easier to justify because of their more obvious societal value (Karr, 1981) and relationship with legislation (i.e., the Canadian *Fisheries Act*; Chapter 1; Minns et al., 1996). Surveys of benthos, however, are a preferred means of characterizing the condition of aquatic systems for several reasons (Pratt and Coler, 1981), but primarily because: (1) they are less mobile than fish, thereby reflecting mainly local conditions; (2) they are considerably easier to sample (although they take longer to process); (3) they have relatively short life cycles (generally about 1 year) such that the composition of a community is expected to change fairly quickly when a stressor is present in a system; and (4) because the condition of benthic communities is presumed to have some relationship with the condition of fish communities and resources while being a non-destructive surrogate of that resource (Department of Fisheries and Oceans and Environment Canada, 1995; Hodson et al., 1996).

The fourth rationale for conducting benthic surveys relies upon an understanding of the fish-benthos relationship. Based on a substantial body of work, we now know that productivity of fish resources is directly linked to the production of benthic communities in lentic (Northcote and Larken, 1956; Hayes, 1957; Rawson, 1960; Matuszek, 1978; Hanson and Leggett, 1982; Boisclair and Leggett, 1989; Pierce et al., 1994) lotic (Bowlby and Roff, 1986a; Waters, 1988) and marine systems (MacKinnon, 1973; Diaz and Schaffner, 1990). We also know that changes in the trophic structure (feeding guilds) of a fish community can have effects on standing crop and size-class structure of benthic communities in fairly controlled situations like enclosed lakes (Ball and Hayne, 1952; Anderson et al., 1978), experimental cages (Gilinsky, 1984), or experimental stream sections (Allan, 1982; Elliott, 1986; Morgan and Ringler, 1994).

Although such demonstrations confirm that benthos and fish are inherently linked when most major environmental factors are controlled, there are few demonstrations that the composition of the benthic community can be used to predict the composition of a fish community across ecological gradients. Demonstrating that fish and benthos do respond to similar environmental factors is crucial if we are to assume that the condition of a fish community can be predicted from some descriptor of a benthic community. In lake systems in

central Ontario, Jackson and Harvey (1993) demonstrated a relatively strong correlation between the composition of fish and benthic communities across a fairly long pH gradient. A strong relationship might be expected in closed lake systems where both trophic groups are forced to respond to the same environmental conditions. In lotic systems without such obvious barriers to fish movement, fish presumably have the potential to move in and out of stressed locations. The benthic community, in contrast, remains more confined to localized areas within a stream whether a stressor is present or not. As a consequence, stream fishes are more likely to reflect watershed-level characteristics, and benthic communities are more likely to reflect local conditions. As a result, there is no guarantee that strong and useful relationships between fish and benthic community structure will exist in natural lotic systems. Recent attempts to demonstrate correlations between fish and benthic communities in lotic systems have been inconclusive. Kerans et al. (1992) showed that there was a strong correlation between the composition of benthos and the composition of fish communities using only three sites. In contrast, Berkman et al. (1986) were unable to demonstrate concordance between fish and benthos communities at eight sites in an agricultural region. Consequently, there is a need to examine more fully the association between fish and benthos.

The primary objective of this chapter was to examine the nature and degree of association between fish and benthic invertebrate communities in 2nd-4th order streams in southern Ontario. For this exercise, three independent data sets, each with at least 35 stream sites, were analyzed. The first data set included Barton et al.'s (1985) fish community data as well as a previously unpublished benthic community survey from the same year and the same locations as the fish survey. Barton et al. surveyed fish and benthic communities across a continuum from cool, highly forested streams, to warm agricultural streams. The second data set was collected by Farrara and Reid (1995) for the City of London (Ontario) as part of a series of sub-watershed studies. Most of their streams were highly influenced by agriculture and/or urbanization. Finally, the third data set was obtained through a study conducted in association with the Ontario Ministry of Natural Resources (M. Jones and L. Stanfield) and involved quantitative electrofishing of half-meander wave-length stream sections. The MNR survey included streams in forested catchments, as well as some heavily urbanized systems. In each of these data sets, both fish and invertebrates were surveyed at the same spatial scale (i.e. within the same stream

reaches). Because the methodologies used to survey both benthos and fish varied among studies, this analysis also allows evaluation of the robustness of the underlying fish-benthos relationship.

A secondary objective of this chapter was to evaluate the effect of taxonomy on the strength of the fish-benthos relationship. This is in response to a growing interest in determining optimal taxonomic levels for benthic identifications (Warwick, 1988a,b; Ferraro and Cole, 1990; Somerfield and Clarke, 1995; Wright et al., 1995; Barton, 1996). Identifications to the species level are tedious and costly (Rosenberg et al., 1986), but are often recommended (Resh and Unzicker, 1975; DFO & EC, 1995). For those benthic surveys being conducted as part of evaluations of the condition of fish habitat (DFO & EC, 1995; Stanfield et al., 1996; Hodson et al., 1996), understanding the taxonomic level at which assessments should be made is an important consideration.

4.2 Methodology

4.2.1 The Data Sets

The sampling methods used to collect fish and invertebrates varied among the three studies used in this overall analysis: (1) Barton et al.'s (1985) buffer strip study; (2) Farrara and Reid's (1995) London sub-watershed study; and (3) a study conducted in conjunction with the Ontario Ministry of Natural Resources (OMNR) (Figure 4.1). Methods used to collect fish, benthos and environmental data for each study are therefore described in detail.

Barton et al. (1985) - Buffer Strip Study

Fish were collected during August 1980 using a backpack electroshocker. Blocking nets (6 mm mesh) were used to enclose 25-50 m reaches. The original study by Barton et al. (1985) recorded total catches (abundances) of all fish species from either two or three passes through the blocked sections, as well as additional seining where there were pools. This analysis of the fish data utilizes only the catches from the first pass. In addition, because biomass data provide better resolution between impacted and non-impacted sites, and because biomass data are more closely related to the goals of the Federal *Fisheries Act* (i.e., no net loss of the productive capacity of fish and fish habitat) (Randall et al., 1995; Minns et al., 1996; Chapter 3), I converted species abundances into species biomasses by multiplying the abundances of each species by their

average weight. Average weights for species in this data set were determined from collections made as part of Chapter 3 (Appendix C). Making this conversion from abundance to biomass gives larger and less abundant fish species such as trout more weight in the analysis and leads to greater separation of trout streams with cyprinids and cyprinid dominated streams (Chapter 3). The statistical analyses of the fish community data described below were performed using both species abundances and biomasses, with little change in the interpretation of the results. Consequently, I present the results of the analysis on biomass data only.

Benthic collections were not reported by Barton et al. (1985) but consisted of a travelling kick and sweep technique that attempted to collect representative benthos from all microhabitats within the stream section that was shocked. Mesh size of the kick net was 200 μm. Single collections were made at each site during both May and August of 1980. Samples were preserved in the field using buffered formalin (5-10%) and sorted in the laboratory using 6-12x magnification. Generally, over 400 organisms were removed from each sample and identified to the lowest practical taxonomic level. Hirudinea, Naididae and Tubificidae (Oligochaeta), Gastropoda, Amphipoda, Isopoda, Decapoda, most Ephemeroptera, Plecoptera, Hemiptera, most Hydropsychidae (Trichoptera), adult Elmidae (Coleoptera) and Chironomidae (all pupae, and larvae of some genera) were identified to species. Other animals were identified to genus or family.

The environmental variables measured by Barton et al. (1985) are given in Table 4.1. Barton et al. (1985) also determined the ratio of the median to minimum discharge and the ratio of maximum to median discharge. These ratios were not included in this analysis (1) because there were missing values for one site, (2) because neither ratio explained variation in fish community composition (Barton et al., 1985), and (3) because part of the information incorporated into these ratios is explained by median discharge which is examined in detail here. Prior to analysis, all variables were $\log_{10}$ transformed with the exception of the discharge ratio and stream temperature.

Farrara and Reid (1995) - London Subwatershed Studies

Farrara and Reid (1995) surveyed 110 3rd-4th order stream sites within the boundaries of the City of London from which I selected 40 for which there was complete environmental, fish

and benthic data. Fish were collected during May and June of 1994 by electrofishing stream sections that were at least 30x the bankful width. Numbers of each species caught were recorded. As with the Buffer-Strip data, abundances of each species were converted to biomass using data on average biomasses from additional streams surveyed in 1994 (Chapter 3). At riffle areas of each surveyed stream station, two or three Surber samples were collected during December 1994, and sieved in the field with 500 μm mesh. In addition to the quantitative samples, a single qualitative sample was collected throughout the study reach at the same time. Each qualitative sample was collected over a 30-minute period using a travelling kick-and-sweep technique (following Griffiths, 1993) with a 500 μm mesh dip net. Both qualitative and quantitative samples were preserved in the field to 10% buffered formalin. Samples were viewed under 10x magnification and organisms were removed and identified to the lowest practical taxonomic level (species when possible, but usually genus, Farrara and Reid, 1995). The environmental variables included in the analyses are listed in Table 4.2. In-stream measurements (dissolved oxygen, temperature, depth, width) were made at the same time as the December benthic survey. Planimeters were used to measure the area of each land use within subwatersheds from Ontario Ministry of Agriculture and Food (OMAF) land use maps. Of the environmental variables included in the analysis, only conductivity was $\log_{10}$ transformed.

MNR - Habitat Suitability Index (HSI) Data

In association with the Ontario Ministry of Natural Resources (OMNR), I obtained data on fish, benthos and environmental variables at 54 3rd-4th order stream locations across southern Ontario. Of those, 35 stream sites had complete fish, benthos and habitat information. At each station, half-meander wavelengths (crossover point to crossover point) were enclosed using blocking nets. These sections were electrofished once, and fish were counted and weighed. Prior to fishing, benthos were collected at crossover points using a kick net (Plafkin et al., 1989) and 1 minute of kicking (Stanfield et al., 1996). On site, 100+ randomly selected benthic organisms was removed from the debris, preserved in 70% ethanol and later identified to genus. Environmental variables measured at each site are listed in Table 4.3. Streams were classified as either warm (≈ 23°C), cool (≈ 18°C) or cold (≈ 14°C) based on the differential between actual stream temperatures at ≈16:00 h and maximum daily air temperatures on the same day (Stoneman

and Jones, 1996). Solar window (a measurement of the percentage canopy) was measured following Stoneman and Jones (1995) while the percentage of the stream as pools, flats and chutes, the percentages of stream cover as rock, wood and undercut banks, and the ratio of the average pavement to sub-pavement particle sizes was determined following Stanfield et al. (1996). Habitat measurements were determined using a point-transect method that involved measurement of stream depth, slope, particle size and flows at each of ten equispaced points along 10 transects equally spaced between two thalweg crossover points (riffles). The percentages of stream habitat as pool, chute or riffle was determined using the dichotomous key in Stanfield et al. (1996). Watershed size as well as the proportion of the watershed as forest, agriculture and urban areas were determined from LandSat™ scenes from southern Ontario, interpreted by the Provincial Remote Sensing Office. Using PC-ARC/INFO, land use coverages were intersected with catchment boundaries on digital Ontario Base Maps to produce the landuse information for each site (Mee, 1996).

4.2.2 Statistical Analyses

The analysis of fish-benthos-environment correlations was conducted in two steps. The first step involved performing Mantel (1967) tests to determine whether fish and benthic communities were significantly correlated. With Mantel's test concordance between two matrices is measured as a summation of the cross-products between the off-diagonal elements of the two matrices. The degree of association of the two matrices is then judged against a null distribution determined via permutation. According to K.M. Somers (pers. comm., Ontario Ministry of Environment) the Mantel statistic operates like a coefficient of determination (i.e., r^2). Consequently, Mantel r values indicate, approximately, the amount of variation in one matrix that can be explained another.

Distance matrices for fish and benthos were made by calculating Bray-Curtis distances (Rohlf, 1993) between all pairs of stream stations. The Bray-Curtis coefficient is a commonly used community coefficient for quantitative or proportional data (Legendre and Legendre, 1993; Rohlf, 1993). The coefficient is calculated using the formula:

$$d_{ij} = \frac{\sum_k |x_{ki} - k_{kj}|}{\sum_k (x_{ki} + x_{kj})} \qquad [1]$$

where d_{ij} is the Bray-Curtis distance between two samples i and j, k refers to the kth species, x_{ki} refers to the abundance of species k in sample i and x_{kj} refers to the abundance of species k in sample j (Rohlf, 1993). Two samples with identical species composition will result in a Bray-Curtis distance of zero, while two samples with no species in common will result in a Bray-Curtis distance of one.

Fish and benthos distance matrices were calculated for each of the three data sets. For the qualitative (presence/absence) kick-and-sweep survey conducted by Farrara and Reid (1995), Jaccard's measure of similarity was calculated. The Jaccard coefficient (JC) is calculated using the formula:

$$JC = \frac{a}{a+b+c} \qquad [2]$$

where a represents the number of species in common, b is the number of species found in sample j but not i, and c is the number of species found in sample i but not j (Rohlf, 1993). The Jaccard coefficient was used to estimate the similarity among communities using presence/absence data because it is a commonly used coefficient for that type of data (Legendre and Legendre, 1993; Rohlf, 1993). With Jaccard's coefficient, samples with all species in common will result in a value of one, whereas samples with no species in common will result in a value of zero. In order for the Jaccard's coefficient to complement the Bray-Curtis coefficient, I used the inverse of the coefficient such that small values indicated similarity while large values indicated dissimilarity.

For qualitative abundance data, numbers (for benthos) or biomass (for fish) of taxa were transformed into proportional abundances (i.e., % of total number or total biomass) after $\log_{10}$ transformation, while for quantitative data, numbers or biomasses were simply $\log_{10}$ transformed prior to calculation of Bray-Curtis coefficients. To determine whether taxonomic level of a benthic survey influenced the observed correlation between benthos and fish, benthos distance matrices were calculated for counts at the species, genus, family, order and phylum levels.

The second stage of the analysis used canonical correspondence analysis (CCA) to summarize fish-environment and benthos-environment correlations. CCA is a direct ordination technique, which constrains the ordination of taxa distributions to the set of environmental variables measured by the investigator (Ter Braak, 1986, 1991, 1994; Jongman et al., 1995). CCA elegantly portrays the relationships between species distributions and environmental variables, and can assist in determining primary and secondary gradients controlling the distributions of ecological communities.

Each of the three fish and benthic data sets were analyzed separately using CCA for a total of six separate analyses. Because the correlations between fish and benthos distance matrices were nearly as strong at family as at genus or species (Table 4.4), only family-level benthos data were analyzed with CCA for simplicity. Each data set had several taxa (fish species and benthic families) that were found in only a few samples. Since rare taxa can cause distortions of underlying pattern in correspondence analysis (CA; Ter Braak, 1986) taxa found in fewer than three samples in a given data set were removed prior to running CCA.

To determine if the environmental variables measured in each of the studies accounted for a significant fraction of the community distributions, CA and detrended CA (DCA) were also used to analyze these same community data sets. Ordinations by CA and DCA can be thought of as determining how much non-random variation is present in a community data set (Gauch, 1982). When an ordination by CA or DCA accounts for much more variation in community data than does CCA, then it can be inferred that the measured environmental variables are not the dominant factors governing the taxa distributions. CANOCO (Ter Braak, 1991) was used to perform all constrained and un-constrained ordinations.

4.3 Results

4.3.1 Mantel Correlations

Correlations between fish and benthos were significant for all three studies (Table 4.4) but the strength of the correlations varied between studies and depended on the timing of the benthic surveys, the level to which benthos were identified, and the type of benthic survey conducted. Correlations were highest for the buffer strip study with Mantel correlations > 0.5 for surveys conducted in August. For the London subwatershed study, fish-benthos correlations were higher

for quantitative than qualitative benthic data, while for all three studies there was generally little reduction in the Mantel correlation at the family level.

4.3.2 Canonical Correspondence Analysis

Figures 4.2-4.4 give the CCA biplots for the fish and benthos community data for each of the three studies. In these biplot diagrams, lines connected to the origin represent the magnitude of the influence of an environmental variable on the distribution of organisms. Environmental variables connected to the origin with long lines are more important, while those connected by short lines are less important. The direction of the line from the origin represents the direction of change in that particular environmental gradient (Ter Braak, 1986). The position of a species in the biplot represents the location of the species' optimum in relation to each of the environmental gradients portrayed in the biplot. Species positioned at an extreme end of an environmental gradient are more influenced by that gradient than species positioned close to the origin (Ter Braak, 1986).

Ordinations by CCA generally accounted for significant fractions of the non-random variation in community composition. Table 4.5 provides the % of variation accounted for by a given CCA axis for a given data set. Measured environmental variables generally accounted for $\approx$ 70% of the non-random variation in taxa distributions in each of the fish and benthos data sets, and generally accounted for more of the non-random variation in fish distributions than benthic family distributions for all three studies (Table 4.5).

This analysis of the buffer-strip data also confirmed the findings of Barton et al. (1985) that colder streams with a higher proportion of the watershed as forested were dominated by brook, brown and rainbow trout. Colder streams also had less suspended fine particulates and higher discharge than the warmer streams that supported primarily cyprinids (Figure 4.2). In addition, the colder streams were also characterized by mayfly (Siphlonuridae), stonefly (Leuctridae, Perlidae) and caddisfly (Psychomyidae, Phryganeidae) families whereas warmer streams were dominated by several snails (Planorbidae, Valvatidae), glossiphoniids (leeches) and planariids. Fish distributions were secondarily related to riffle width (or stream width, CCA axis 2), whereas benthos distributions were secondarily related to overall stream gradient (CCA axis 2) (Figure 4.2).

In the London sub-watershed study, the primary gradients in the distributions of fish and benthos were associated with a suite of environmental variables primarily reflecting stream/watershed size. In this study, fantail darters, northern hog sucker and blacknose shiner were associated with higher-order streams, as were several benthic families including the Philopotamidae, Helicopsychidae, Heptageniidae and Psephenidae (Figure 4.3). In contrast, northern redbelly dace, brook stickleback, central stoneroller, pumpkinseed sunfish and least darter were associated with lower-order streams, as were several benthic families including Tabanidae, Erpobdellidae, and Perlodidae (Figure 4.3). Second axes for fish and benthos were also correlated with the percentage of the watershed as intensive agriculture and winter stream temperatures. Streams surrounded by landuses with a higher proportion as intensive agriculture were colder during the winter and were dominated by hornyhead chub, river chub, rosyface shiner and *Hybognathus* sp (brassy/eastern silvery minnow), as well as the benthic families Leptophlebiidae and Limnephilidae. In contrast, streams with less intensive agriculture and higher winter water temperatures were more dominated by central stonerollers and pumpkinseed as well as the benthic families Coenagrionidae, Tricorythodae and Ancylidae. Winter stream-water temperatures ranged from 1 to 8°C (Table 4.2) with the warmer stream temperatures indicating a higher proportion of flows originating as ground water.

Finally, as with the London subwatershed study, the primary gradients in the distributions of fish and benthos in the MNR-HSI data were associated with maximum temperatures and % forest cover (Figure 4.4). However, in contrast to the previous two studies, this analysis of the MNR-HSI study showed that other additional environmental factors were highly correlated with the primary distributions of fish (i.e., % of the stream as flats, % of the instream cover as undercut banks), and benthos (i.e., watershed size and % of the watershed as urban). Generally though, colder streams with greater forest cover were more dominated by brook, rainbow and brown trout, as well as the benthic families Perlodidae, Tricorythodae and Nemouridae, while streams with high temperatures and a lower % forest cover in the watershed were more dominated by white sucker, largemouth bass, northern redbelly dace and pumpkinseed sunfish, as well as several benthic families including Erpobdellidae, Tubificidae and Planorbidae (Figure 4.4). The secondary distributions of fish and benthos were associated with apparently different environmental factors. Secondary fish distributions were related to the % of the stream as pools

and chutes, while the secondary benthos distributions were related to % of the stream as pools and flats, and the % of the instream cover as rock.

4.4 Discussion

This analysis of fish, benthos and environmental data suggests that across environmental gradients, fish and benthic communities in 2nd to 4th order streams in southern Ontario are highly correlated. This was demonstrated by relatively high and significant Mantel correlations (Table 4.4). Moreover, higher correlations between fish and benthos with benthos identified to lower taxonomic levels (i.e., genus and species) provides support for the idea that the fish-benthos correlation is due to fish and benthos changing in concert across environmental gradients. Further support for this idea comes from demonstrating that the primary distributions (and secondary distributions for the London Subwatershed study data) of fish and benthos taxa were generally related to similar environmental factors in each of the three independent data sets.

In the buffer-strip study, fish and benthos responded in concert to a change in stream type from cool streams with low concentrations of suspended fine particulates and a high proportion of forest in the catchment, to warmer streams with high concentrations of suspended fine particulates and a low proportion of the catchment as forested (Figure 4.2). The changes in both fish and benthos were characteristic of a change from relatively pristine (undisturbed) stream conditions to relatively degraded. As fully described by Barton et al. (1985), colder streams had fish species such as brook, brown and rainbow trout that are relatively intolerant of high temperatures and degraded (loss of) habitat, whereas the warmer streams were dominated by more tolerant cyprinds, ictalurids and darters (Wichert, 1994). The changes in the benthic community were also indicative of a general change in water and habitat quality. The cold-water streams were dominated by relatively sensitive stonefly, caddisfly and mayfly taxa, while the warmer streams were dominated by more tolerant snails, leaches, isopods, planariids, and damselflies (Bode, 1988).

Demonstrating that fish and benthos distributions are correlated with the same environmental factors implies that fish and benthos respond to the same environmental cues at similar spatial scales. This probably results from stream fishes having limited mobility or home ranges. Although stream dwelling species such as rainbow trout and white sucker may make

spawning migrations (Scott and Crossman, 1973), several studies confirm that the majority of individual fish within populations have adult home ranges less than 100 m (e.g., Gerking, 1953, 1959; Greenberg and Holtzman, 1987; Hill and Grossman, 1987; Heggenes et al., 1991; Bridcut and Giller, 1993). As a consequence, a survey of fish or benthos in a section of stream will reflect either local or watershed-level influences, whichever is more important at that specific location or time. Most of the environmental factors measured in this study reflect whole-watershed activities.

As Bowlby and Roff (1986a,b) surmise, fish and benthos may be related through top-down influences of the fish community on the benthic community. They suggested that when top-level piscivorous fish were present in a stream, benthic feeding fishes were reduced in biomass resulting in a release of larger benthic taxa. Although that hypothesis has been supported by several controlled experiments in both lentic (Gilinsky, 1984; Post and Cucin, 1984; Hanson and Legget, 1986; Bronmark et al., 1992; Hayes and Taylor, 1992) and lotic environments (Cooper et al., 1990; Power, 1992; Wiseman et al., 1993), the findings from this study suggest that environmental cues are more important in determining the composition of the fish and benthic communities than top-down or bottom-up effects. Stronger correlations between fish and benthos with benthos identified to the species (or lowest practical) level suggest that environmental tolerances of benthos are more important to the correlation than body size (since body size tends to be fairly uniform at higher taxonomic levels, Dolédec and Statzner, 1994). In addition, further evidence that top-down effects are not as important comes from the analysis of the buffer strip data. In that data set, large benthos were found at the extremes of the temperature gradient while piscivorous fish were present at only the cold end of the same temperature gradients (Figure 4.2). Large Megaloptera (Sialidae, Corydalidae) and stoneflies (Leuctridae) were dominant in colder streams in association with brook trout in the buffer strip study, while large Glossiphoniidae and Coenagrionidae were more dominant in warmer streams in association with benthivorous cyprinids in the same study (Figure 4.2). More likely than top-down effects, concurrent (and independent) changes in fish and benthos community compositions are probably responsible for the observed associations between fish and benthos. Similar conclusions were reached by Cross et al. (1985) in a study of fish food habits across a pollution gradient in the coastal waters of California. As my analyses confirm, temperature is a major

influencing factor in stream communities, and there are classic fish and benthic communities associated with cold, cool and warm environments (Scott and Crossman, 1973; Beck, 1977; Harris and Lawrence, 1978; Hubbard and Peters, 1978; Surdick and Gaufin, 1978). Top-down effects of fish feeding on benthos are probably important within both cold and warm systems, but large environmental gradients appear to be more important in determining the complex of species (both fish and benthos) that may occur at a site.

Demonstrating that benthic communities identified to generic and species levels results in a higher correlation with fish community composition also has applied consequences. This finding infers that benthic surveys at lower levels will more accurately predict the condition of fish communities. Several authors have previously recommended the use of family-level identifications in surveys of stream benthos (International Organization for Standardization, 1979; Armitage et al., 1983; Hilsenhoff, 1988; Wright et al., 1995) because surveys at the family level are generally as successful at identifying impacts as are surveys at the species or generic level (but see Barton, 1996). Within lower taxonomic groupings (i.e., genus and species) taxa tend to respond more to micro-habitat (e.g., the presence or absence of macrophytes, interstitial substrate particle size, etc.,) influences, while family-level characteristics have been considered more related to macro-habitat factors like overall water quality (Buikema et al., 1979; Warwick, 1988b). The analyses I conducted with the three data sets suggest that habitat and water quality factors control benthos because of family level characteristics. The minor increase in correlation between benthos and fish when genus and species-level identifications are performed is probably a reflection of differences in tolerances within families. The overall importance of the strength of the fish-benthos relationship (and thus of the level of effort required to identify benthos) for making inferences of the condition of fish condition based on surveys of benthos, is further explored in Chapter 5.

Although there were obvious differences in the magnitude of the correlations between fish and benthos within and between studies, the confounding effects of season and samplers with studies makes it difficult to specifically determine whether some samplers collected benthic communities that were more highly correlated with fish communities than other methods. In general, however, higher correlations between fish and benthos were observed for the buffer strip study which incorporated a qualitative travelling-kick method that sampled benthos from all

microhabitats within streams. That study also incorporated a late summer (August) survey, and processing of a high number of organisms (i.e., generally > 400) into the benthos sampling protocol. In contrast, the poorer correlations between fish and benthos in the MNR-HSI and the London subwatershed study may have been due to differences in microhabitats that benthos were sampled from, season that benthos collections were made, and differences in benthic processing techniques. Both the London subwatershed and MNR-HSI studies collected benthos from riffles only, contrasting the travelling kick technique of the buffer-strip study. Data in Chapter 5 demonstrates that correlations between fish and benthos are higher when benthos are collected from runs. Riffle habitats have the potential to be more influenced by cleaner and colder groundwater than runs or pools (Godbout and Hynes, 1982). Runs and pools are also more likely to collect anthropogenically derived contaminants (Kerans et al., 1992). Data in Lenat (1988) suggests that the travelling kick technique provides a more accurate characterization of water quality at a site than do other methods that focus on only riffle habitats. According to the species descriptions in Scott and Crossman (1973), many fish species spend most of their feeding and resting periods in runs and pools. Consequently, stronger associations between run/pool benthos and fish communities may be expected.

Differences in sorting techniques between the various studies may also explain some of the observed differences in the fish-benthos correlations. For example, the field sorting technique used in the MNR-HSI study is known to be a biased technique that generally misses smaller benthos (Dukerschein et al., 1996). In addition, the "rapid" MNR-HSI study protocol required the collection and identification of only 100 organisms in the contrast to the > 400 organisms collected and identified in the buffer-strip data set. In such qualitative sampling techniques, more organisms result in more accurate characterization of benthic community composition (Plafkin et al., 1989).

Finally, based on the assumption that fish and benthos are associated because they both respond to the same environmental cues, collecting benthos during different seasons would be expected to affect the strength of the relationship. Low flow periods in mid to late summer (i.e., mid July to mid September, Stanfield et al., 1996) tend to limit the distributions of taxa because surface water temperatures are highest, anthropogenic contaminants are at peak concentrations (because of the low flows) and dissolved oxygen concentrations are low (because of high

temperatures and high nutrient concentrations). Distributions of fish in small streams tends to be a function of these low-flow conditions, primarily because of limiting temperatures (Li et al., 1994; Schlosser, 1995). Barton (1996) recently demonstrated that benthos collected during mid-summer low-flow periods show greater differences between reference-forested and impacted-agricultural streams than other times of the year including spring and fall. This is because some species of typically sensitive groups like Plecoptera, Limnephilidae and Diamesinae can be found in impacted sites during winter when conditions are less limiting (Barton, 1996). Consequently, finding that the fish-benthos correlation was lower in the London sub-watershed study, in which the benthic survey was conducted during December, is not surprising.

Regardless of the differences in magnitude of the fish-benthos correlations and potential effects of season and sampling methods, the findings from this study confirm that stream fish and benthos are significantly associated and that surveys of benthos can be used to make inferences on the condition of fish community composition. Further studies are required to determine more specifically how much change in benthic community composition corresponds with ecologically relevant changes in fish community composition (Chapter 5).

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Table 4.1. List of environmental variables used to describe stream physical characteristics for 39 streams in southern Ontario as part of Barton et al.'s (1985) buffer strip study, and the variable codes used in Figure 4.2.

Variable	Code	Average	Minimum	Maximum
Watershed Area (ha)	WATERSHED	4363	1440	8560
% of catchment forested	FOREST	40	0	97
Reach gradient (m/km)	GRAD1	3.9	0.9	15.2
Stream gradient (m/km)	GRAD2	5.2	1.0	25.5
Riffle width (m)	RIFWD	3.6	0.9	7.3
Riffle depth (cm)	RIFDEP	22.2	4.8	41.6
Trimean weekly maximum temperature (°C)	WMT	24	17	29
Median suspended fine particulate matter (mg/l)	FPM	33.6	10.1	128.6
Median suspended coarse particulate matter (mg/l)	CPM	10.4	3.6	32.2
Median discharge (l/s)	DISCH	108.9	4.9	420.1

Table 4.2. List of environmental variables used to describe stream physical characteristics for 40 streams in southern Ontario as part of Farrara and Reid's (1995) London sub-watershed study, and the variable codes used in Figure 4.3.

Variable	Code	Average	Minimum	Maximum
Watershed Area (ha)	WATERSHD	1 783	98	19 476
% of watershed as forest	FOREST	10	0	25
% of watershed as intensive agriculture	AGR_INT	64	0	85
% of watershed as moderate agriculture	AGR_MOD	10	0	34
% of watershed as pasture	PAST	4	0	23
% of watershed as urbanized	URBAN	12	0	100
dissolved oxygen (mg/l)	DO	12.6	9.4	15.2
temperature (°C)	TEMP	4.2	1	8
pH	PH	7.9	7.6	8.3
conductivity (μmhos/cm)	COND	724	415	1230
water velocity (m/sec)	VELOCITY	0.29	0.05	0.80
stream depth (cm)	DEPTH	28	10	50
stream order	ORDER	3	1	4
stream width (m)	WIDTH	4.1	0.8	20.0

Table 4.3. List of environmental variables used to describe stream physical characteristics for 35 streams in southern Ontario as part of the MNR study, and the variable codes used in Figure 4.4.

Variable	Code	Average	Minimum	Maximum
Watershed Area (ha)	WATERSHD	2504	139	9511
% of watershed as agricultural	AGR	64	16	91
% of watershed as forest	FOREST	26	2	63
% of watershed as urban	URBAN	9	0	81
Temperature (°C)	TEMP	23	17	28
Stream width (m)	WIDTH	4.7	1.0	15.0
% of stream section as pools	POOLS	31	1	9
% of stream section as flats	FLATS	41	6	8
% of stream section as riffles	RIFFLES	16	0	66
% of stream section as chutes	CHUTES	7	0	24
% of cover as wood	WOOD	31	0	98
% of cover as rock	ROCK	55	0	100
% of cover as undercut banks	UNDERCUT	1	0	5
% of canopy as open	WINDOW	58	14	77
stream depth (cm)	DEPTH	13	6	21
log of the ratio of pavement to subpavement particle sizes	P/SP	1.24	0.19	2.68

Table 4.4. Correlations between fish and Bray-Curtis distance matrices as well as the probability of a non-significant correlation for each of three fish-benthos data sets.

	Data Set				
	Barton		London		MNR
	May Qualitative	August Qualitative	December Quantitative	December Qualitative	July/August Qualitative
Species	**0.444 (<0.01)**	**0.504 (<0.01)**	**0.286 (<0.01)**	**0.148 (0.04)**	
Genus	**0.482 (<0.01)**	**0.506 (<0.01)**	**0.271 (<0.01)**	**0.155 (<0.01)**	**0.198 (0.014)**
Family.	**0.386 (<0.01)**	**0.507 (<0.01)**	**0.231 (<0.01)**	**0.154 (0.029)**	**0.149 (0.047)**
Order	**0.367 (<0.01)**	**0.394 (<0.01)**	**0.229 (0.02)**	0.108) (0.096)	0.041 (0.315)
Phylum	**0.146 (0.049)**	**0.161 (0.038)**	0.113 (0.16)	**0.144 (0.021)**	0.132 (0.057)

Table 4.5. Percent of variation associated with the first two axes for each of the constrained (CCA) and unconstrained (CA) ordinations used to evaluate the fish and benthic distributions in each of the three data sets. The sum of variation explained by the first two axes, as well as the % of the unconstrained variation explained by CCA are also given.

	Fish				Benthos			
Ordination	Axis 1	Axis 2	sum	% of unconstrained	Axis 1	Axis 2	sum	% of unconstrained
Buffer Strip Study								
CCA	10.5	6.4	16.9	54	11.5	5.9	17.4	75
CA	18.0	13.5	31.5		14.4	8.8	23.2	
London Subwatershed Study								
CCA	12.1	8.2	20.3	70	9.4	5.9	15.3	66
CA	16.1	12.7	28.8		13.9	9.1	23.0	
MNR-HSI Study								
CCA	14.7	10.3	25.0	84	11.6	8.6	20.2	79
CA	17.5	12.4	29.9		14.2	11.2	25.4	

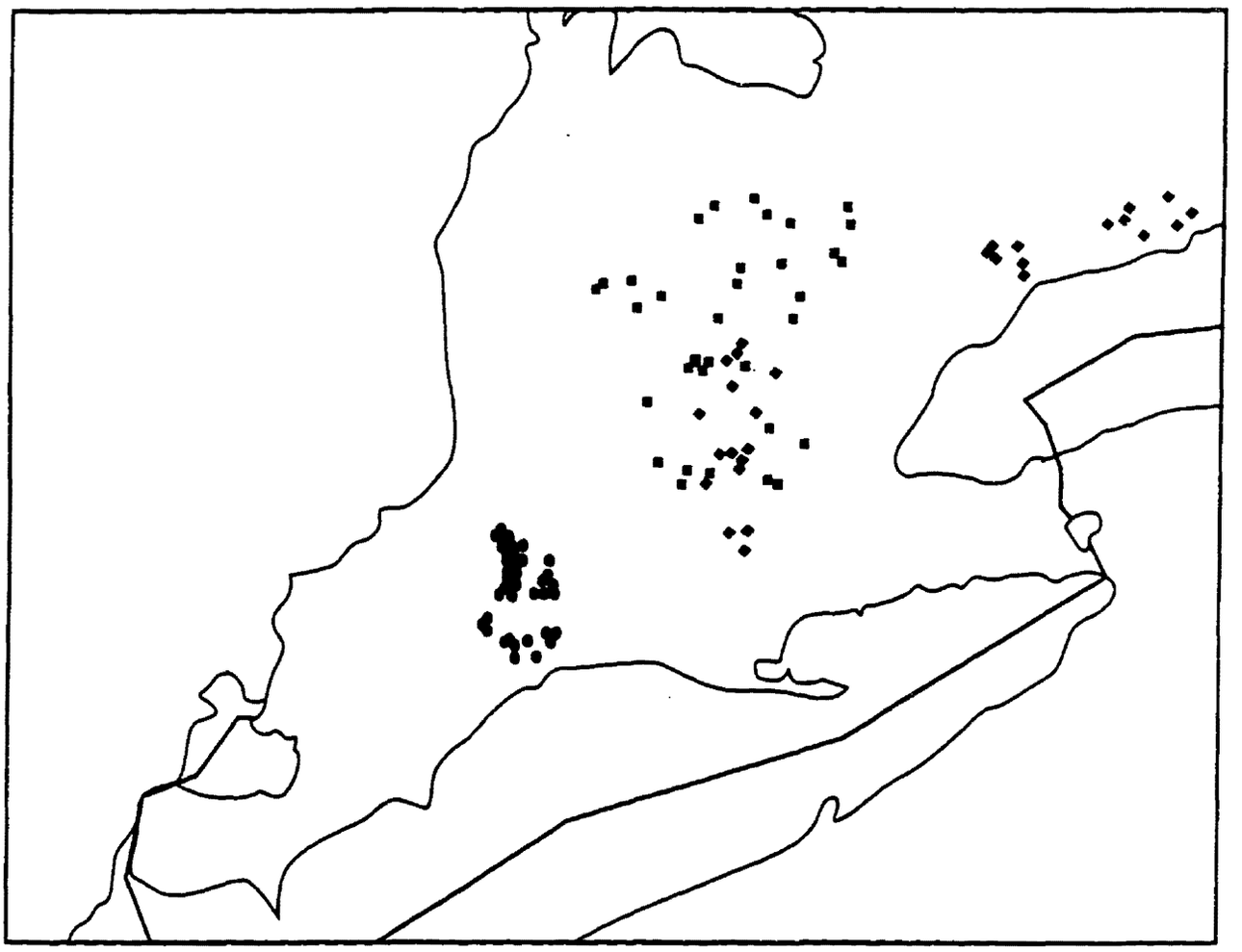
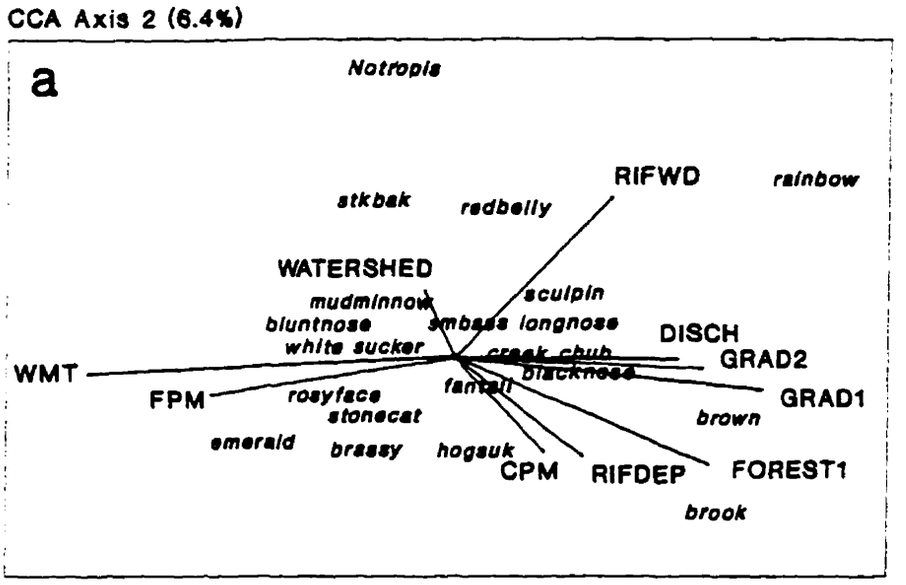


Figure 4.1. Map of study stream locations for Barton et al.'s (1985) buffer strip study (■), Farrara and Reid's (1995) London sub-watershed study (•) and the MNR-HSI study (♦).



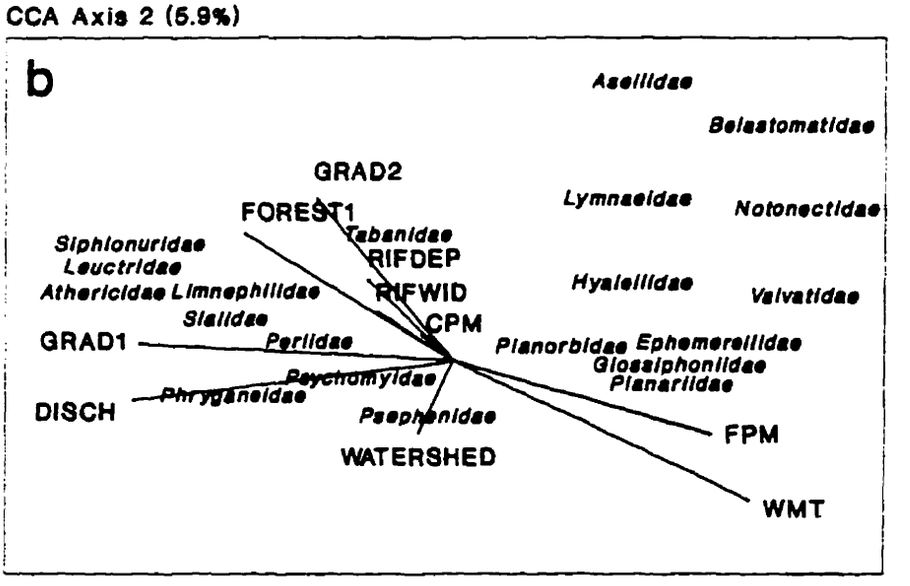
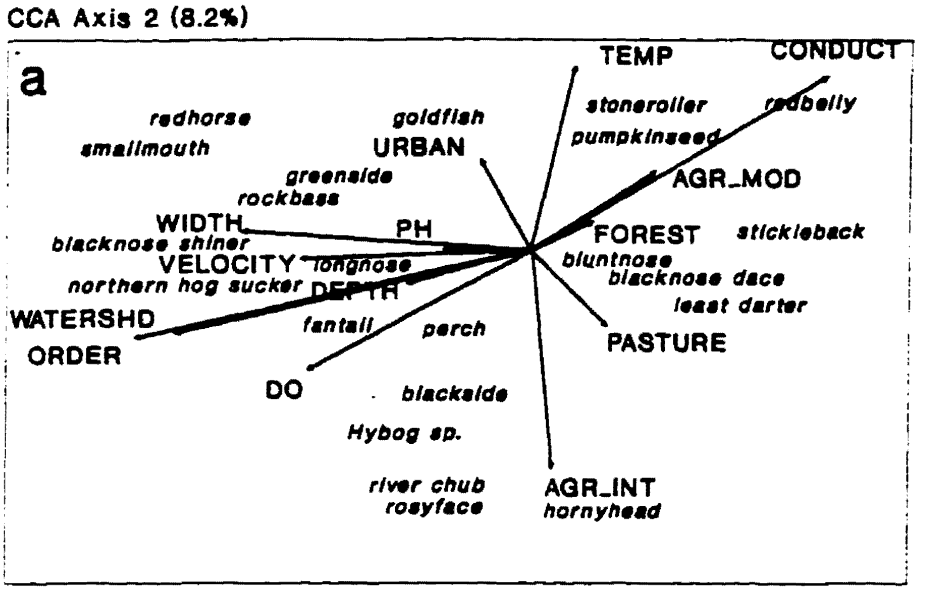


Figure 4.2. Canonical correspondence analysis ordination diagram of (a) fish species and (b) benthic families in relation to environmental variables for the buffer-strip study. Only a partial set of fish and benthic families are given for clarity.



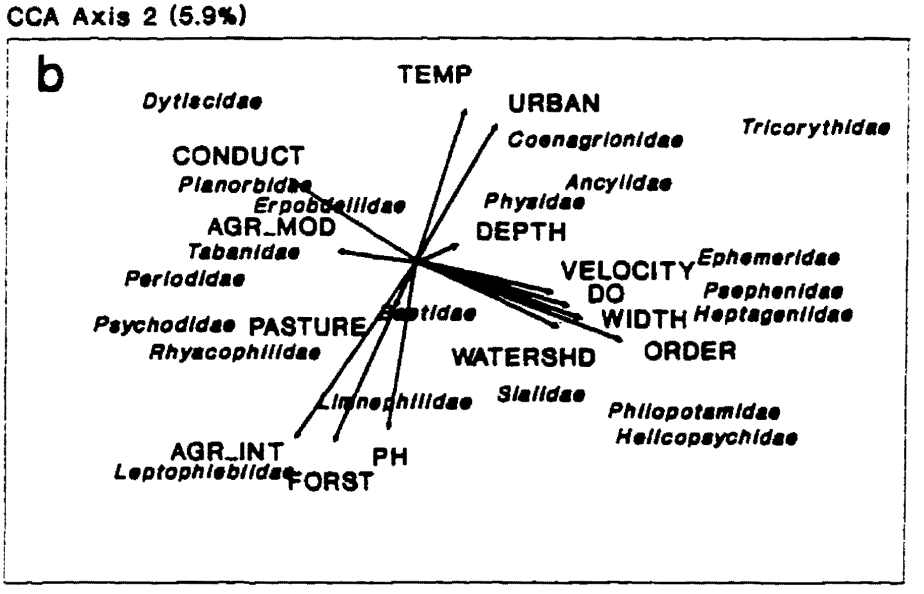
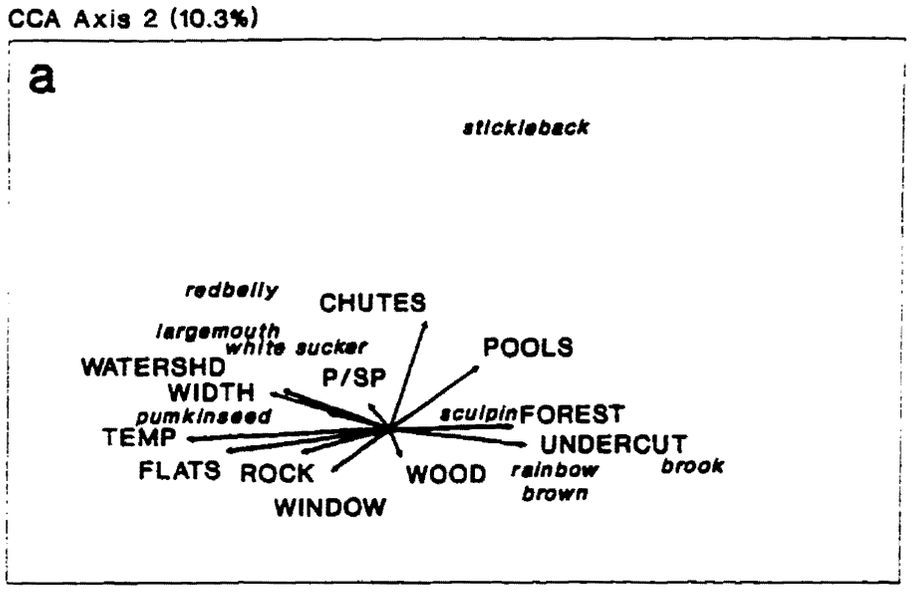


Figure 4.3. Canonical correspondence analysis ordination diagram of (a) fish species and (b) benthic families in relation to environmental variables for Farrara and Reid's (1995) London sub-watershed study. Only a partial list of fish and benthic families are given for clarity.



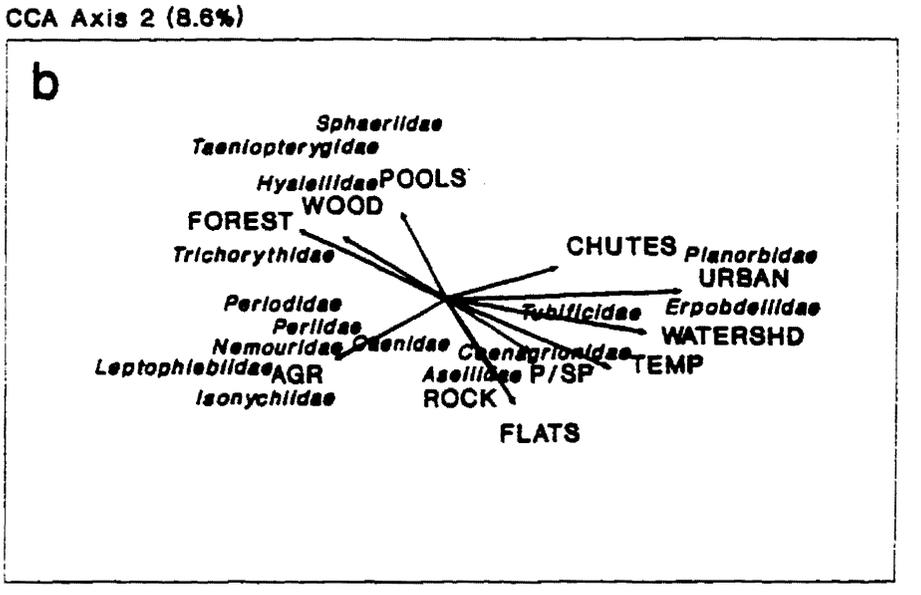


Figure 4.4. Canonical correspondence analysis ordination diagram of (a) fish species and (b) benthic families in relation to environmental variables for the study done in conjunction with the MNR in 1995. Only a partial list of fish and benthic families are given for clarity.

CHAPTER 5

Effects on stream benthos that correspond with ecologically significant effects on fish

5.0 Abstract

Using five independent data sets (broken down further into 11 combinations of data types), I examined the relationships between measured impacts on benthos and fish to determine how much change in the composition of benthic communities one would need to observe in order to observe an ecologically significant effect on fish. Ecologically significant effects on fish are here defined as those impacts which exceed the region enclosing 95% of reference location observations. The degree of impact of both fish and benthos was described using the percent model affinity approach (PMA), where affinity was the percentage difference in composition of the "impacted" community relative to the average reference community. On the basis of regressions describing the relationship between PMA for fish (i.e., PMA_{fish}) and PMA for benthos (i.e,. $PMA_{benthos}$), I determined critical benthic effects that coincided with ecologically significant effects on fish. With the exception of site-specific surveys of stream reaches classified as runs, changes in benthic communities that coincided with ecologically significant effects on fish communities were generally less than estimated 95% confidence regions for $PMA_{benthos}$ (or less than the mean reference community response + 1.65 standard deviations). The reliability of benthic surveys for making the correct inference of the degree of impact on fish communities varied with the scale of the survey and the abruptness of the changes in fish communities. Surveys conducted at small spatial scales (i.e., those using site-specific reference areas), tended to be more reliable at inferring true-positive effects on fish communities (i.e., 70-100% of the time inferences based on benthos made the correct conclusion that fish community effects exceeded the estimated 95% region). Surveys incorporating regional-reference areas were moderately less reliable (50-80%) unless there were abrupt changes in fish communities (e.g.,

from brook-trout streams to marginal trout or cyprinid streams) in which case inferences made using benthos were also highly reliable at inferring true-positive effects (90-100%).

5.1 Introduction

The previous four chapters have: (1) demonstrated the importance of being able to specify critical effect sizes for environmental indicators (Chapter 2); (2) argued that benthos are often considered a surrogate endpoint for other aspects of ecosystem condition that have more apparent value (Chapter 1); and (3) demonstrated that benthic community composition is correlated with the composition of fish communities, apparently because both fish and benthos in streams respond to similar environmental factors (Chapter 4). A primary objective of this chapter is to demonstrate a procedure for determining effect sizes in benthos that correspond with critical effects in descriptors of fish community composition. This involves demonstrating a relationship between relevant descriptors of fish and some descriptor of benthic community composition. Once the relationship is determined, it can be used to determine critical benthic effect sizes that coincide with what one might consider to be ecologically significant effects on fish communities. As in Chapter 2, I assume that effects on fish communities exceeding the normal range of vaiation in composition for reference communities can be considered ecologically significant. Effects less than that magnitude can be considered acceptable because the effects are within the range that is normal given the reference communities that are being considered.

A second objective of this chapter is to determine the reliability (Murtaugh, 1996) of derived benthic critical values for making the correct inference that a fish community would, in fact, be impacted. The reliability of any critical value as an early-warning indicator depends on the strength of the relationship between the compliance and early-warning indicators, and on the nature of that relationship. Similar studies have attempted to document the reliability of toxicity tests for inferring instream biological impacts (Eagleson et al., 1990; Marcus and McDonald, 1992). Yoder and Rankin (1995) have recently determined the reliability of a benthic community index for inferring impacts on a fish community index. In general, they found that a site deemed to be impacted according to a benthic community index was generally (about 80% of the time) also deemed to be impacted according to a fish community index (and *vice versa*). The Yoder and Rankin (1995) study was based on the philosophy that an index (in this case the Index of Biotic Integrity for fish, and the Invertebrate Community Index for benthos) is an appropriate means of characterizing a community. Both of these indices have subjective components in that the environmental tolerances assigned to the various taxonomic groups are subjective, and are

not based on empirical data. In addition, the indices are specific to Ohio and to regional-reference study designs (Hughes, 1995). They would require calibration in order to be applied elsewhere, or to be applied site-specifically (Hodson et al., 1996). Finally, the Yoder and Rankin (1995) approach used benthic decision criteria that were derived independently of any fish community data; their criterion for impact is the 75th percentile of the normal range of reference community observations. This is a significant variance from the approach that I have proposed throughout this thesis.

In this chapter (as with previous chapters) I use a percent-affinity model (PMA, Novak and Bode, 1992) to characterize the degree of similarity of an "impacted" fish or benthic community to a reference. This approach is taken to determine if more "objective" means of characterizing a benthic community can be used to predict the condition of a fish community. Objective methods are more appropriate when the nature of the response of the community to the stressor is unknown. This generally occurs when surveys are conducted for the first time in a new ecoregion, and when the reference condition is anything other than pristine.

As a third objective, I examine the effects of several methodological factors on both derived benthic critical values as well as reliability estimates. These methodological factors include sampling techniques (quantitative Hess and Surber vs qualitative travelling and stationary kick), sampling locations (run vs riffle), spatial scale (regional-reference vs site-specific reference) and taxonomic level (species, genus, family, order). The taxonomic level to which benthos are identified, has cost implications (Rosenberg et al., 1986) although Barton (1996) indicates that identifications to genus or species improves our ability to detect impacts when they occur. Riffle and run habitats generally support different benthic fauna (Brown and Brussock, 1991) because of differences in hydrodynamics (Brussock and Brown, 1991). Monitoring programs in Canada, therefore, generally recommend sampling only one of the habitats, or at least stratifying designs by such habitat characteristics (Department of Fisheries and Oceans and Environment Canada [DFO & EC], 1995). Finally, both regional-reference and site-specific reference locations can be incorporated into study designs. In this chapter, I use five independent data sets to establish relationships between impacts measured with benthos and impacts measured with fish. Three of the data sets involved collections of fish and benthos across broad-spatial scales (i.e., with regional-reference locations). Two other data sets are used to examine

relationships between measured impacts on fish and invertebrates at small spatial scales with site-specific reference locations.

5.2 Methodology

5.2.1 Description of Data

Five data sets were used to explore the correlation between the degree of impact estimated by fish and benthic communities. Each of these data sets was collected independently (either by different investigators, or in different systems) so they are described in detail below. These descriptions are broken down into data sets that examined the fish-benthos correlation at (1) broad-spatial scales, where a set of test streams is contrasted against a set of presumed regional-reference (Hughes, 1995) streams, and (2) small-spatial scales, where a set of streams affected by a point source are contrasted against site-specific (Green, 1979) reference streams. Summaries of key similarities and differences among data sets are given in Table 5.1.

5.2.1.1 Large Spatial Scale Data

Data from three large-spatial scale studies were used: (1) Barton et al.'s (1985) buffer strip study, (2) a study I conducted in conjunction with the Ontario Ministry of Natural Resources during the summer of 1995 as part of a habitat suitability index modelling exercise (MNR-HSI study), and, (3) Farrara and Reid's (1995) survey of fish and benthos conducted as part of a sub-watershed study for the City of London, Ontario. These data sets are described in detail in Chapter 4, while the site locations are given in Figure 5.1.

5.2.1.2 Small Spatial Scale Data

Fish and benthos were collected from Laurel and Canagagigue Creeks above and below major reservoirs (Figure 5.2 a,b) during fall (late September to early October) of 1994. Fish were collected by electroshocking riffle-pool-run sequences (approximately half meander wavelengths) enclosed with 6-mm blocking nets. After each of three fishing episodes through each enclosed section, fish were counted and weighed to the nearest 0.01 g. Total biomasses for each species at each site were estimated using Carle and Strub's (1978) equations.

At each site, two benthic samples were collected from both riffle and run (Jowett, 1993)

habitats. Samples were collected using a plastic Hess sampler (Hess, 1941) with 240 μm mesh. Samples were washed on site with 500 μm mesh and preserved in 5-10% buffered formalin. In the laboratory, rose-bengal dye was added to the samples 24 h prior to sorting, to improve sorting efficiency (Williams, 1974). Just prior to sorting, excess formalin and dye were washed from the samples using 200 μm mesh. Samples were sorted under binocular microscopes at 6-12 x magnification. With the exception of oligochaetes which were identified to family, and nematodes and turbellarians, all benthos were identified to genus using conventional taxonomic literature (Morton and Gale, 1996).

5.2.2 Statistical Analyses

The specific analyses performed varied slightly depending on the scale at which the fish and benthic surveys had been conducted. In general, however, I was interested in determining, first, the nature of the fish-benthos relationship. This involved estimating the degree of impact of both fish and benthic communities, and the use of simple regression analyses and inspection to assess the relationships. For the broad-scale spatial data, regional-reference locations, with presumably unaltered (or moderately altered) fish communities were used as reference locations. For small-scale spatial data, stream locations upstream of the major impoundments were used as reference locations for downstream "impacted" locations.

5.2.2.1 Percent Model Affinity PMA

I used percent-model affinity (PMA, Novak and Bode, 1992) to measure the degree of difference between reference and impacted fish and benthic communities. The Bray-Curtis coefficient (Rohlf, 1993; Chapters 2 and 4) was used as the community distance measure for the PMA calculations. This approach was used for several reasons. First, PMA results in a single value which summarizes the absolute similarity or dissimilarity between two communities. Other methods that summarize similarities among communities (e.g., ordination) result in multiple derived variables. With ordination, whether one chooses all of the derived variables, or a sub-set, is subjective. Second, the measure of affinity, in this case the Bray-Curtis coefficient, is the metric that is used as raw data by ordination procedures such as non-metric multidimensional scaling (NMDS) and principal coordinates analysis (PCo-A). Such ordination procedures simply

attempt to summarize the original distance matrix, but often distort it (Gauch, 1982; Kenkel and Orloci, 1986; Minchin, 1987). Consequently, use of the original distance matrix avoids possible distortion associated with ordination. Using the PMA approach has one conceptual difficulty in that changes in fish or benthic communities may be in any of several directions from the reference community, and it could be that variations in benthic and fish communities are only in concert if changes are in a certain direction from the reference. However, Chapter 4 demonstrated that major variations in fish and benthos do tend to covary across environmental gradients.

5.2.2.2 Specifying Groups with Large-Scale Spatial Data

With the broad-scale spatial data, there were no *a priori* groups of stream stations that one might consider reference locations. Consequently, preliminary assessment of the streams was required in order to select relevant reference or target streams. Figures 5.3-5.5 show ordinations of the broad spatial scale stream-fish communities. These principal coordinates analyses (PCo-A) portray in two dimensions the similarities in fish community composition determined using Bray-Curtis coefficients calculated between pairs of stations. Streams dominated by brook trout and those marginal-brook-trout streams dominated more by cyprinids are denoted for the buffer-strip (Figure 5.3) and MNR (Figure 5.4) studies. Designating brook-trout streams as a regional-reference condition is appropriate since brook trout were historically the dominant salmonid in southern-Ontario watersheds (Martin, 1984). The use of marginal trout streams as a secondary target also has value since most of our watersheds are, to some extent, impacted by anthropogenic activity, and marginal trout streams may be considered acceptable in many areas. For these two data sets, then, the affinity of each anthropogenically altered fish community to these average reference communities (brook trout and marginal-brook trout) was calculated.

In both the buffer-strip study and the MNR study, stream temperature was the dominant environmental gradient, primarily influenced by land use (Barton et al., 1985; Chapter 4). For the purposes of this chapter, I make the assumption that the differences between streams in these two data sets were due to anthropogenic activities. Some of the variation in stream temperatures, the major environmental gradient, may also be due to natural differences in underlying physiography.

In the London-subwatershed study, the dominant environmental gradient influencing both fish and invertebrates was watershed size (Chapter 4). Figure 5.4 shows that streams dominated by creek chub were distinct from all other streams. One of these creek-chub dominated streams also had brown trout, so these may approach the marginal-trout condition in the previous data sets.

5.2.2.3 Reference and Impact Stations for Small-Scale Spatial Data

For the Laurel and Canagagigue Creek small-scale spatial data, I calculated PMA of fish and benthos communities to the average of the upstream (of impoundments) reference communities. In each system, several downstream fish and benthos communities were described at increasing distances from the reservoirs (Figure 5.2). This sampling strategy was incorporated in an attempt to provide a graded response in fish and benthos communities. Those communities nearest the reservoirs were expected to be the most impacted, while those farthest from the impoundments were expected to show some degree of recovery from the effects of impoundment. Since duplicate benthos samples were collected from both riffle and run habitats at each station, I calculated correlations between fish and benthos PMAs for each sampling strategy.

5.2.2.4 Portraying Variations in Fish and Benthos Communities with Principal Coordinates Analysis (PCo-A)

In order to confirm that the observed effects in this study make biological sense, I ordinated both fish and benthos for all of the data sets (Figures 5.3-5.14). Principal coordinates analysis (PCo-A) was used to ordinate Bray-Curtis distance matrices calculated from $\log_{10}$ transformed data. For those data sets in which the taxa abundances were proportional (i.e., non-quantitative, Table 5.1), Bray-Curtis distances were determined for proportional $\log_{10}$ transformed data. Only the first two ordination axes are shown for simplicity.

5.2.2.5 Deriving Critical Benthic Values

In this study, it is assumed that impacts on fish community composition in excess of the normal range of variation (or in excess of the 95% region for the reference community response) are effects worth preventing and/or detecting. With percent-affinity models, calculation of normal

ranges depends upon whether one uses a similarity or dissimilarity measure of affinity. In this study, the Bray-Curtis coefficient was a dissimilarity measure such that small values indicated high similarity with the reference community and large values indicated low similarity with the reference community. Table 5.2 provides the average PMAs for reference fish and benthic communities for each of the 11 data sets. For percent affinity models, 95% of the observations are enclosed by the mean reference response + 1.65 standard deviations (Barton, 1996). Estimated 95% regions are also given in Table 5.2 for fish communities. Based on relationships between fish and benthos PMAs to reference communities (Figures 5.15-5.25), I estimated the benthos community PMAs that coincided with 1.65 standard deviation effects on fish communities. Critical benthic PMAs were then re-expressed as the number of standard deviations from the mean reference community. For example, based on the relationship between species-level-benthic PMA (SP_{PMA}) and single-pass-%biomass-fish PMA ($FISH_{PMA}$) for the buffer strip data using brook-trout streams as the reference, the critical benthic PMA was estimated as 0.55 Bray-Curtis units (Table 5.3). The average affinity of the reference communities was 0.49 with SD of 0.037. The critical benthic PMA re-expressed in standard deviation units was therefore

$$\frac{0.55-0.49}{0.04} = 1.7\ SDs.$$

5.2.2.6 Determining the Reliability of Benthos

The reliability of benthos as a surrogate measure of the condition of fish depends upon the likelihood of making correct inferences. These probabilities will be a function of the nature and degree of correlation between fish and benthos PMAs. Based on methods from signal-detection theory (Swets, 1988; Murtaugh, 1996), I determined the probability of making correct and incorrect inferences of the ecological relevance of effects on fish communities given the results of a benthic survey. To do this, fish communities from each data set which fell within the estimated 95% region (i.e., mean PMA + 1.65 SDs) were deemed acceptable, while those with PMAs outside of the estimated 95% region were deemed to exhibit ecologically significant effects. Thus, each stream site was classified as being either within or outside the acceptable

range. Based on these classifications, I determined the number of: (1) TP = true positives, where both the fish and benthos surveys agreed that the fish community was outside the acceptable 95% region; (2) FP = false positives where benthic community surveys indicated that the fish community should have been outside the range, but was not; (3) TN = true negatives where both the fish and benthic surveys agreed that fish community was within the acceptable 95% region; and (4) FN = false negatives where benthic community surveys indicated that fish communities were within the acceptable range, when in fact they were outside that range. The probability of a true positive (%TP) result was then estimated as

$$\%TP = \frac{TP}{TP + FN}, \qquad [1]$$

whereas, the probability of a false positive (%FP) result was estimated as

$$\%FP = \frac{FP}{FP + TN}. \qquad [2]$$

Ninety-five percent confidence limits for these proportions were calculated based on equations for binomial distributions given in Zar (p 378, 1984). Murtaugh (1996) recommends this overall approach for assessing the reliability of ecological indicators.

5.2.2.7 Determining Sources of Variation in Derived Critical Values and Reliability

One of the objectives of this work was to determine if one could predict, *a priori*, what the benthic critical value would be without having to calibrate fish and benthos responses. Two approaches to examining sources of variation were used. First, I used principal components analysis to examine variations and covariations of fish-benthos regression statistics (i.e., slopes, intercepts and correlation coefficients, Table 5.3), derived benthic critical values (Table 5.3), measures of reliability (i.e., %TP and %FP, Table 5.4), spatial scale, and taxonomic level for the 11 combinations of studies (Table 5.1). Spatial scales were classified as 1 (small-spatial) or 2

(large-spatial). Taxonomic levels were classified as 1 (species), 2 (genus), 3 (family) or 4 (order). The PCA was based on the correlation matrix of these variables.

The second approach used analysis of variance (ANOVA) to specifically test whether habitat (run or riffle), sample type (quantitative or qualitative) or spatial scale (large or small) caused variation in the size of the benthic critical value. This ANOVA approach was also used to test the various sources of variation in measures of reliability. To test for differences between spatial scales, I contrasted the Laurel and Canagagigue data against all other data sets. To test for differences due to quantitative or qualitative sampling, I contrasted the Laurel, Canagagigue and London data sets against all others. To test for differences between riffle and run data, I contrasted riffle and run habitats within Laurel and Canagagigue Creeks. Each ANOVA assessed variation in effect sizes, and true and false positives estimated using genus-, family- and order-level taxonomy. Since identifications were made to species level only with the buffer strip data, no comparisons were possible for species-level identifications.

5.3 Results

5.3.1 Community Descriptions

In this section, I describe the major trends in fish and benthos communities observed in the various data sets. Generally, the objective here is to illustrate the nature and degree of observed effects in both the fish and benthos data for each of the data sets used in this overall analysis.

5.3.1.1 Broad-Scale Spatial Studies

This section provides a brief overview of the general trends in fish and benthos distributions for each of the five data sets. In general, those clusters of stations designated as reference streams according to the distributions of trout species (Figures 5.3-5.4), also tended to have discrete communities of benthos. In the buffer-strip study, streams with trout (with the exception of two that appeared to be outliers, Figure 5.8a) tended to have a higher predominance of taxa such as several stonefly families (Leuctridae, Perlodidae, Perlidae) that are considered sensitive to thermal and nutrient enrichment (Hilsenhoff, 1988). In contrast, the cyprinid streams (and two marginal trout streams) had a higher predominance of taxa tolerant of thermal and nutrient enrichment such

as the Tubificidae, Physidae and Erpobdellidae. In the buffer-strip data set, there was no obvious separation of good trout streams from marginal trout streams, suggesting that these streams had generally similar water quality.

Similar patterns were found in the MNR-HSI data set. Streams dominated by trout tended to have a greater predominance of cold-water benthic taxa such as Perlidae, Perlodidae and Taeniopterygidae. In contrast, the cyprinid dominated streams were dominated by taxa tolerant of thermal and nutrient enrichment such as Tubificidae, Erpobdellidae and Planorbidae (Hilsenhoff, 1988). In contrast to the buffer strip data, the MNR-HSI benthos data less obviously separated trout streams from cyprinid streams (Figure 5.9).

In the London subwatershed study, the benthic communities associated with creek chub streams appeared to be distinct (i.e. clustered together, Figure 5.10). However, in contrast to the patterns seen with the buffer-strip and MNR-HSI benthos (Figures 5.8 and 5.9), the non-reference streams in the London data set were not separate from the reference-group along any single dimension. Rather, the non-reference stations surrounded the reference stations (Figure 5.10). In Chapter 4 I showed that the fish-benthos correlation for this London subwatershed study was lower than for the buffer-strip and MNR surveys. This seeming lack of concordance among stations based on the fish PCo-A and the benthos PCo-A appears to be reflective of this poorer correlation.

5.3.1.2 Small-Scale Spatial Studies

In both Laurel and Canagagigue Creeks, differences in fish and benthic communities above and below major reservoirs are obvious from the PCo-A ordinations (Figures 5.6 and 5.7, and 5.11-5.14). Fish communities upstream of Laurel reservoir were dominated by benthic-oriented fishes including rainbow and johnny darters and blacknose dace. In contrast to these reference stations, downstream stations were dominated by fish species generally typical of lentic conditions including brassy minnows, smallmouth bass and pumpkinseed sunfish (Figure 5.6). Upstream stations had benthic communities that are generally more associated with less anthropogenic activity, than were the downstream fauna. Regardless of whether samples were collected from runs or riffles, downstream benthos were dominated by tubificid and naidid worms, as well as planariids, glossiphoniid leaches, and isopods (Figures 5.11, 5.12). All of these taxa tend to be

fairly tolerant of high oxygen demands and temperatures (Hilsenhoff, 1988). In contrast, upstream reference stations were more dominated by Ceratopogonidae, Baetidae, Elmidae and Tipulidae (among others) which are generally less tolerant of high oxygen demands (Hilsenhoff, 1988).

In Canagagigue Creek, large differences in fish and benthos communities were observed above and below the major reservoir (Figures 5.7, 5.13 and 5.14). Upstream of Woolwich reservoir, Canagagigue Creek was dominated by cold- and cool-water fishes such as brook trout, sculpin, blacknose dace and rainbow darter. Downstream of the reservoir, there was a greater biomass of species more commonly associated with warmer, lentic conditions such as carp, rock bass, smallmouth bass, and white sucker. Below the reservoir, there was no obvious downstream trend in fish community composition (Figure 5.7). Benthic communities above and below Woolwich Reservoir were also different, particularly those collected from run habitats (Figure 5.13). In contrast, riffle benthos from upstream station 1 were more similar to downstream assemblages (Figure 5.14). In general, stations both above and below the reservoir were characterized by benthos that are tolerant of moderately enriched conditions. For example, the dominant benthos in upstream runs were naidid and tubificid worms, chironomids and erpobdellid leaches, all of which are tolerant of moderate to severe enrichment (Hilsenhoff, 1988). Downstream of the reservoir, numbers of simuliids, planariids, *Hydra*, and ancylids increased. These groups are also tolerant of enriched conditions (Bode, 1988).

5.3.2 Derived Benthic Critical Values and Sources of Variation

The Principal Components Analysis (PCA) demonstrated that derived benthic critical effect sizes were moderately positively correlated with the slope and correlation coefficient, and negatively correlated with the intercept of the fish-benthos relationship. These variables were all correlated with the first PCA axis (Table 5.5). In addition, there was a considerable amount of variation in critical values that was independent of the other variables included in the PCA (see PCA axis 3, Table 5.5). Based on the ANOVA (Table 5.6) some of this variation in critical values was associated with sample type, spatial scale and possibly habitat type. Effects of scale and sample type were clearly significant depending on the taxonomic level considered (Table 5.6). Although

the effects of habitat type were not significant at the 5% level (i.e., $p \approx 0.06$), the probability of rejecting the habitat contrast was very low with only two data sets. Consequently, the effects of habitat type may be significant. Liberating the Type I error rate to 10% for this contrast does result in a signficant habitat effect (Table 5.6). Critical values were generally higher for benthic surveys based on quantitative methods, higher for surveys incorporating site-specific reference areas, and were higher when benthos were collected from runs (Figure 5.26). In the ANOVA (Table 5.6), the effects of sampling scale and sampling type (i.e., quantitative vs qualitative) were confounded such that they shared a significant amount of the variation in estimated critical values. This was because the Laurel and Canagagigue Creek studies provided all of the small-scale data, and most of the quantitative data (only the London subwatershed study provided additional quantitative data). There was no apparent effect of taxonomic level on derived critical values (Figure 5.26).

5.3.3 Reliability of Benthos for Inferring Ecologically significant Effects on Fish Communities, and Sources of Variation

Based on these derived benthic critical values, the probability of saying that there is an ecologically significant effect on fish communities when in fact there is (i.e., a true positive assessment) or is not one (i.e., a false positive assessment) is given in Table 5.4. The effects of scale on true-positive probabilities were marginal for genus ($p = 0.097$) and significant for order-level taxonomy (Table 5.7; Figure 5.27). True-positive probabilities were also high for those data sets that considered good trout streams to be the reference condition (i.e., buffer strip and MNR-HSI, 50-80%; Table 5.4). There was no apparent difference between riffle and run habitats in terms of true-positive probabilities (Table 5.7; Figure 5.27). The probability of a true-positive statement decreased with an increase in taxonomic level (Figure 5.27). For example, the probability of a true-positive statement fell from 100% at the species and generic levels in the buffer-strip study (brook trout streams as reference) to 83% at the order level (Table 5.4; Figure 5.27).

The association between taxonomic level and the probability of a true-positive result could have occurred for two reasons. First, higher correlations between fish and benthos would result in a higher proportion of true-positive results. In the buffer-strip study, reductions in true-

positive results corresponded with reductions in the fish-benthos correlation (Table 5.5). However, consistent reductions in correlations did not occur for all data sets such as the MNR-HSI data set. Consequently, an increase in true-positive results with increasing taxonomic effort was not explained on the basis of improvements in the fish-benthos correlation. A second explanation for higher true-positives with increased taxonomic effort may have been related to increased resolution between reference and impact streams. Greater differences between impact and reference sites at lower taxonomic levels were confirmed by measuring the length of the ecological gradients. Benthic community gradient lengths were measured using CANOCOs (Ter Braak, 1991) detrended correspondence analysis (DCA). DCA scales the ordination of sites by the tolerance range of species abundances, which are expressed as the number of standard deviations (SDs) from a species' modal abundance. Site scores are therefore scaled according to these SDs which are a common means of expressing the relative change in community composition across an ecological (environmental) gradient (Gauch, 1982; Ter Braak, 1991; Jongman et al., 1995; Van Wijngaarden et al., 1995). Calculated gradient lengths (Table 5.9) were longer for lower-level taxonomy than higher-level taxonomy, confirming that higher-level taxonomy resulted in generally greater differences between reference and impacted communities.

False-positive results were generally about 30% for all data sets with the exception of the August benthos survey in the buffer-strip study and the small-scale spatial surveys in Laurel and Canagagigue Creeks (both riffle and run benthos) which had no false-positives. There was no apparent relationship between the proportion of false-positive results and taxonomic level among data sets (Figure 5.28). There was, however, a consistent effect of scale on false-positive probabilities (Table 5.8) with small-scale surveys having no false-positives, and large-scale studies having between 0 and 45% false positives (Figure 5.28). There was no significant effect of sample type on the probability of a false-positive statement.

Inspection of scatterplots of the fish-benthos relationships (Figures 5.15-5.25) suggests that the proportion of true and false positives also tends to reflect the nature of that relationship. Data sets with discrete groups of impacted fish communities (e.g., buffer strip and MNR-HSI studies using trout streams as the reference, Figures 5.15 and 5.17; Laurel Creek, Figures 5.19-5.20) tended to yield more true-positive and fewer false-positive results than studies in which impacted fish communities were less discretely separated from the reference fish communities.

In Chapter 3, I recommended that three-pass-CPUE-biomass data be considered as compliance indicators because of an obvious relationship with the goals of the *Fisheries Act*, and because of increased statistical power for detecting effects. Of the data sets examined for this portion of my study, only the Laurel Creek and Canagagigue Creek data sets incorporated three-pass-CPUE-biomass fish data. All other studies used single-pass-biomass data. To determine if the separation of small-scale spatial studies from large-scale spatial studies (Figure 5.26) was due to this methodological difference, I re-calculated fish-benthos regressions for the Laurel and Canagagigue data sets using both single-pass biomass and single-pass-% biomass (Table 5.10). In all cases, these new fish-benthos regressions resulted in larger estimated critical benthic effect sizes than those estimated from regressions based on three-pass-CPUE-biomass fish community data (c.f. Tables 5.3 and 5.8). Consequently, the observed separation between the small- and large-scale spatial studies in Figure 5.26 is not likely due to having used different fish community endpoints when building the fish-benthos regressions.

5.4 Discussion

These results suggest that several factors influence the strength of the fish-benthos relationship, estimated benthic critical values, as well as the overall utility of benthos for predicting the degree of impact exhibited by a fish community. Critical benthic effect sizes appear to be a function of the specific micro-habitat from which benthos are collected, the scale at which samples are collected, and/or the type of sample collected. The reliability of benthos for correctly predicting when fish communities will exhibit an ecologically significant effect appears to increase when benthos are identified to lower taxonomic levels, and when changes in a fish community are abrupt.

5.4.1 Variations in Critical Benthic Effect Sizes and Fish-benthos Correlations

Derived critical effect sizes for benthos exhibited significant variation and ranged from 0 - 3.8 SDs. Effect sizes < 1.65 SDs indicate that benthos had changed less than fish, whereas effect sizes > 1.65 SDs indicate that benthos had changed more than fish to the given stressors. Under

an *a priori* assumption that benthos are more sensitive to change than are fish, one would have predicted that benthos would change more to environmental stressors than fish. The observed variation in derived critical effect sizes observed here suggest that whether benthos change more or less than fish depends on the nature of the impact and the specific methodological factors associated with the benthic portion of the study.

Larger critical benthic effect sizes were generally associated with stronger fish-benthos correlations as observed in the Laurel and Canagagigue systems, particularly when benthos were collected from runs (Figure 5.26). High correlations were also observed with the buffer-strip data, August benthos collections. These data sets with high correlations all include benthos collected from run habitats. In contrast, those data sets with lower correlations had benthos collected only from riffles. In the Laurel and Canagagigue surveys, benthos were collected specifically from run habitats, while in the buffer-strip study, benthos were collected from runs as well as other microhabitats within the stream sections (i.e., riffles and margins). High correlations between fish and benthos collected from run habitats makes some sense when one considers that run (and pool) habitats are the micro-habitats within streams where contaminants are more likely to settle, and if one assumes that the fish-benthos relationship is driven more by common environmental tolerances of fish and benthos than by other top-down or bottom-up events (see Chapter 4). Riffle habitats, although having more diverse invertebrate faunas than other stream microhabitats (Brown and Brussock, 1991), have the potential to be more influenced by cleaner and colder groundwater (Godbout and Hynes, 1982). Although many stream fishes rely on riffle habitats for completion of reproductive cycles (e.g., trout, darters), most fishes spend much of their feeding and resting time in run and pool habitats (Scott and Crossman, 1973). Historically, fisheries ecologists attempting to understand fish-benthos relationships focused on estimating numbers of benthos from riffle regions because that was the part of the stream considered to have the most diverse and abundant fish food (i.e., benthos) (Waters, 1988; Bowlby and Roff, 1986a,b). However, the results from my analyses suggests that effects on fish community composition can be more accurately predicted if samples of benthos are collected from run locations.

In addition, larger benthic critical values were obtained in small-scale spatial studies in which benthos were collected using quantitative methods. These two factors (i.e., scale and

sampling method) were generally confounded with most of the quantitative studies being part of the small-scale spatial studies; the exception was the quantitative work done in the London subwatershed study. The Laurel and Canagagigue fish-benthos relationships were characterized by steeper slopes, which was probably a function of having lower within-reference average percentage affinity than the other studies (i.e., lower intercepts, Tables 5.2 and 5.5). Low within-reference average percentage affinity indicates greater similarity among sampling areas for benthos (i.e., less variation). The cause of this reduced variation in the Laurel and Canagagigue studies was probably a function of the scale at which sampling was conducted, rather than the sampling method used. Lenat (1988) found that qualitative methods give less variable descriptions of reference communities. Based on his findings, one would have predicted that the quantitative sampling as conducted in the Laurel and Canagagigue studies would have resulted in greater within-reference variation than the qualitative studies conducted in the buffer strip study. Alternatively, one could have predicted *a priori* that the small-scale spatial studies would have resulted in smaller within-reference percent affinity since there is more control over extraneous environmental variables (e.g., substrate) that can cause extra (non-impact related) variation in the benthic community.

Finally, moderately larger critical benthic effect sizes associated with benthos collected from runs in the Laurel Creek and Canagagigue studies also resulted from a steeper slope in the fish-benthos relationship. Steeper slopes to the fish-benthos relationship imply that the benthic communities from runs are responding more to environmental change than are benthos from riffles. Such variation in effect sizes between run and riffle benthos may be a function of the nature of the impact. As discussed earlier, benthos from runs and riffles are likely to vary in their response because of inherent differences in hydrodynamics between riffle and run habitats. Barton and Metcalfe (1986) confirmed that the degree of response of a benthic community to a particular stressor can depend on the micro-habitat benthos are collected from. Consequently, a change in the stressor may change the nature of the relationship between fish and benthos, and hence, the critical benthic effect size corresponding with ecologically significant impacts on fish.

5.4.2 Variations in Reliability

Reliability of benthos, measured as the probability of a true-positive or false-positive inference

of an ecologically meaningful effect on fish, varied primarily with taxonomic level, and apparently with the nature of the impact and the nature of the reference.

First, reliability varied with taxonomic level because lower taxonomic levels resulted in greater separation of reference and impact sites in terms of composition (Table 5.9). This is important because it not only implies that reliability of a benthic survey will be improved with lower-level taxonomic work, but it also implies that there will be more statistical power for detecting effects that exceed the critical benthic effect sizes. Much of the work on taxonomic levels in benthic ecology has attempted to argue that higher levels such as family are suitable for impact assessment purposes (Buikema et al., 1979; Hilsenhoff, 1988; Warwick, 1988a,b, 1990; Ferraro and Cole, 1990; Somerfield and Clarke, 1995; Wright, 1995). However, much of this work is based on assessments in marine environments (Warwick, 1988a,b, 1990; Ferraro and Cole, 1990; Somerfield and Clarke, 1995). In contrast there is increasing evidence from freshwater systems, that identifications to species levels (or lowest practical level, see Barton, 1996) results in more accurate depiction of conditions at a site (Furse et al., 1984; Rosenberg et al., 1986; Barton, 1996). Identifying organisms only to higher levels assumes uniformity within these broader taxonomic groupings. As Barton (1996) argues, many species or genera within higher taxonomic groups do respond in similar fashions to a variety of factors. However, there are other groups such as chironomids, caddisflies, mayflies and stoneflies which have several species that can respond differently to different stressors (Rooke and Mackie, 1982; Bode, 1988).

Secondly, reliability was high whenever the reference community consisted of good trout streams, as in the buffer strip or MNR-HSI studies. For these two studies, as well as for the Laurel and Canagagigue Creek studies (i.e., small-scale studies), there were large and discrete differences between reference and impacted fish communities. Such distinctive reference fish communities could be a result of: (1) not having sampled a complete sequence from reference to highly impacted. However, stream temperatures were the dominant environmental factor associated with the fish community in the buffer-strip study (Chapter 4), and there was no gap in temperatures from cold to warm streams (Barton et al., 1985). Such large changes in the fish community, then, apparently represents a catastrophic change in state (Saunders, 1980) that occurred at some critical temperature (probably $\approx 22°C$ for trimean weekly temperatures, Barton et al., 1985). For the Laurel Creek study, there were no intermediate communities to survey.

In that study, representative stream sections throughout the reference and reservoir-altered reaches were surveyed. Depending on the fish community, then, changes may be abrupt depending on the reference community being sampled (i.e., brook trout and/or site-specific reference locations), and the reliability of benthos may be quite high for inferring ecologically significant impacts on fish in those situations.

With the exception of the London Subwatershed study, reliability estimates in this study were generally moderately higher than estimates given by Yoder and Rankin (i.e., ≈ 80%, 1995), particularly when identifications were taken to genus or species (c.f. Table 5.4). However, given that the Yoder and Rankin (1995) reliability estimates were fairly similar to those reported here provides further support for the use of benthos as surrogate monitors of the condition of fish communities. The lack of complete agreement between benthos and fish can probably be attributed to secondary environmental factors acting independently on fish and benthos. Historically, one of the main reasons workers have been reluctant to attempt to make inferences of fish condition based on benthic surveys is because fish were presumed to respond to different environmental factors (primarily habitat related) than benthos (primarily water chemistry related). The findings in Chapter 4 suggest that whether habitat or water chemistry changes are dominant in a system, fish and benthos will generally respond in concert. However, findings from Chapter 4 also demonstrated that benthos tended to be more responsive to secondary environmental factors than were fish. Consequently, some of the noise in the fish-benthos relationships observed in this analysis may be due to benthos responses being somewhat modified by extraneous factors (e.g., substrate particle size). An alternative explanation may be that benthos and fish respond at different rates to local phenomena. Because benthos have shorter life cycles (≈ 2 wks to 1-2 y, in general) than fish (2-5 y for most stream fishes), benthic community composition can presumably change more quickly when environmental conditions change. This means that when stressors are increased at a site, benthos will react more quickly, whereas when stressors are removed from a site, benthos will also recover to normal (reference) conditions more quickly. This study has ignored such possible temporal lags and assumed the communities were in equilibrium with their local environmental conditions.

5.5 Summary

This chapter demonstrates an approach for determining not only critical effects in benthos that correspond with unacceptable effects in fish community composition, but a technique for determining the reliability of benthos as a surrogate indicator. The approach is based on demonstrating significant relationships between a descriptor of the condition of the fish community and a descriptor of benthic community composition. Based on a regression that summarizes this relationship, benthic critical values that correspond with significant effects on the fish community are determined. We can also use the calibration data to estimate the reliability of benthos for making correct and incorrect predictions concerning the likely status of a fish community at a site. The analyses here demonstrated that reliability varied with the strength of the correlation between benthos and fish (higher correlations resulted in higher reliability), and apparently with the nature of the effect on fish communities. In those situations in which the changes from reference to impacted fish communities were catastrophic (as in site specific assessments above and below point sources, or from cold to warm-water systems), the ability of benthos to predict when a fish community was impacted was quite high (up to 100% in some cases). Reliability also varied with taxonomic level, with increasing detail in benthic taxonomy resulting in greater predictability. Because decision criteria varied significantly with various methodological and study design factors, standardization of collection methods would be of obvious benefit.

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Table 5.1. Summary of data sets used to evaluate the correlation between measured impacts in fish and benthos.

Data Source	Watershed(s) included	Fish Collection Method	Benthos Collection Method	Year of survey
Buffer Strip Study				
Barton et al. (1985) for fish, benthos from Barton (pers. comm.)	Credit, Grand	single pass, relative biomass	kick + sweep, relative abundance, May and August surveys	1980
London Subwatershed Study				
Farrara and Reid (1995)	Thames	single pass, relative biomass	quantitative surber	1994
MNR-IISI exercise				
this study	Wilmot, Grand	single pass, quantitative biomass	kick+sweep in riffles	1995
Impoundment studies				
this study	Laurel Creek	three pass, quantitative biomass	hess in riffles and runs	1994
this study	Canagagigue Creek	three pass, quantitative biomass	hess in riffles and runs	1994

Table 5.2. Mean, standard deviation (SD) and estimated 95% region (mean + 1.65·SDs) expressed as Bray-Curtis distances from the average reference community. Values are given for fish and benthic community composition, for each of six data sets described in the text.

Data Set & #	Variable	Fish PMA	Benthos			
			Species PMA	Genus PMA	Family PMA	Order PMA
Buffer Strip Study						
1. trout streams as reference, August benthos	mean	0.234	0.486	0.413	0.325	0.232
	SD	0.064	0.037	0.060	0.024	0.059
	95% region	0.339				
2. trout streams as reference, May benthos	mean	0.234	0.476	0.442	0.337	0.186
	SD	0.064	0.112	0.116	0.119	0.060
	95% region	0.339				
3. marginal trout streams as reference, August benthos	mean	0.400	0.583	0.488	0.368	0.244
	SD	0.048	0.105	0.057	0.097	0.042
	95% region	0.479				
4. marginal trout streams as reference, May benthos	mean	0.400	0.585	0.471	0.350	0.201
	SD	0.048	0.076	0.080	0.089	0.058
	95% region	0.479				
MNR-HSI exercise						
5. Trout streams as reference	mean	0.454		0.533	0.551	0.227
	SD	0.098		0.072	0.131	0.084
	95% region	0.615				
6. Marginal trout streams as reference	mean	0.486		0.527	0.442	0.259
	SD	0.078		0.120	0.101	0.072
	95% region	0.614				
London Subwatershed Study						
7. Creek chub streams as reference	mean	0.393	0.476	0.463	0.334	0.249
	SD	0.171	0.040	0.039	0.075	0.085
	95% region	0.674				

Data Set & #	Variable	Fish PMA	Benthos			
			Species PMA	Genus PMA	Family PMA	Order PMA
Laurel Ck.						
8. benthos from runs	mean	0.474		0.307	0.266	0.176
	SD	0.072		0.079	0.084	0.064
	95% region	0.592				
9. benthos from riffles	mean	0.474		0.290	0.227	0.195
	SD	0.072		0.127	0.091	0.097
	95% region	0.592				
Canagagigue Ck.						
10. benthos from runs	mean	0.660		0.237	0.223	0.163
	SD	0.120		0.072	0.049	0.054
	95% region	0.857				
11. benthos from riffles	mean	0.660		0.211	0.180	0.143
	SD	0.120		0.078	0.062	0.043
	95% region	0.857				

Table 5.3. Linear regressions relating benthic community percent-model affinity (PMA) with reference communities and fish PMA ($FISH_{PMA}$) with reference communities for five data sets, at four levels of benthic taxonomy (species = SP_{PMA}, genus = GEN_{PMA}, family = FAM_{PMA}, order = ORD_{PMA}). Correlation coefficients (r), mean-squared errors (MSE), and probability (p) values for the models are given, as are the critical fish effect sizes (PMA_c-fish) from Table 5.2. Based on these models, critical benthic effect sizes are estimated and expressed in terms of affinity (PMA_c-benthos), as well as the estimated number of standard deviations (SDs) from the average within-reference PMA.

Data Set	Model	n	r	MSE	p	PMA_c -fish	PMA_c- benthos	# SDs
Buffer-Strip Study								
1. trout streams as reference, August benthos	$SP_{PMA}=0.43+0.35 \cdot FISH_{PMA}$	39	0.71	0.005	<0.000	0.34	0.55	1.73
	$GEN_{PMA}=0.37+0.29 \cdot FISH_{PMA}$	39	0.58	0.008	<0.000	0.34	0.47	0.95
	$FAM_{PMA}=0.28+0.25 \cdot FISH_{PMA}$	39	0.60	0.005	<0.000	0.34	0.37	1.88
	$ORD_{PMA}=0.20+0.29 \cdot FISH_{PMA}$	39	0.36	0.026	0.025	0.34	0.30	1.13
2. trout streams as reference, May benthos	$SP_{PMA}=0.43+0.35 \cdot FISH_{PMA}$	39	0.71	0.006	<0.000	0.34	0.55	0.66
	$GEN_{PMA}=0.36+0.293 \cdot FISH_{PMA}$	39	0.58	0.008	<0.000	0.34	0.46	0.16
	$FAM_{PMA}=0.28+0.25 \cdot FISH_{PMA}$	39	0.59	0.005	<0.000	0.34	0.37	0.28
	$ORD_{PMA}=0.14+0.20 \cdot FISH_{PMA}$	39	0.50	0.006	0.001	0.34	0.30	0.20
3. marginal trout streams as reference, August benthos	$SP_{PMA}=0.47+0.33 \cdot FISH_{PMA}$	39	0.52	0.007	0.001	0.48	0.63	0.45
	$GEN_{PMA}=0.49+0.08 \cdot FISH_{PMA}$	39	0.11	0.103	0.488	0.48	0.53	0.74
	$FAM_{PMA}=0.25+0.32 \cdot FISH_{PMA}$	39	0.47	0.009	0.012	0.48	0.40	0.33
	$ORD_{PMA}=0.15+0.18 \cdot FISH_{PMA}$	39	0.35	0.005	0.027	0.48	0.24	0.00
4. marginal trout stream as reference, May benthos	$SP_{PMA}=0.44+0.34 \cdot FISH_{PMA}$	39	0.69	0.006	<0.000	0.48	0.60	0.20
	$GEN_{PMA}=0.39+0.31 \cdot FISH_{PMA}$	39	0.64	0.006	<0.000	0.48	0.54	0.86
	$FAM_{PMA}=0.28+0.32 \cdot FISH_{PMA}$	39	0.56	0.010	<0.000	0.48	0.43	0.90
	$ORD_{PMA}=0.11+0.23 \cdot FISH_{PMA}$	39	0.36	0.096	0.023	0.48	0.21	0.28
MNR-HSI Exercise								
5. trout streams as reference	$GEN_{PMA}=0.31+0.47 \cdot FISH_{PMA}$	37	0.39	0.013	0.017	0.60	0.66	0.38
	$FAM_{PMA}=0.51+0.13 \cdot FISH_{PMA}$	37	0.54	0.013	0.001	0.62	0.58	0.22
	$ORD_{PMA}=0.09+0.34 \cdot FISH_{PMA}$	37	0.31	0.013	0.059	0.29	0.40	0.75
6. marginal trout streams as reference	$GEN_{PMA}=0.32+0.47 \cdot FISH_{PMA}$	37	0.73	0.008	<0.000	0.61	0.61	0.69
	$FAM_{PMA}=0.28+0.49 \cdot FISH_{PMA}$	37	0.22	0.010	0.193	0.61	0.58	1.46
	$ORD_{PMA}=0.24+0.24 \cdot FISH_{PMA}$	37	0.58	0.010	<0.000	0.61	0.28	0.38

Data Set	Model	*n*	*r*	MSE	*p*	PMA_c -fish	PMA_c- benthos	# SDs
London Subwatershed Study								
7. creek chub streams as reference	SP_{PMA}=0.52+0.11·$FISH_{PMA}$	47	0.25	0.005	0.096	0.67	0.59	2.85
	GEN_{PMA}=0.51+0.10·$FISH_{PMA}$	47	0.21	0.005	0.148	0.67	0.58	3.25
	FAM_{PMA}=0.36+0.14·$FISH_{PMA}$	47	0.28	0.006	0.056	0.67	0.45	1.54
	ORD_{PMA}=0.29+0.02·$FISH_{PMA}$	47	0.05	0.005	0.711	0.67	0.30	0.60
Laurel								
8. fish as 3-pass biomass, benthos from runs	GEN_{PMA}=-0.03+0.89·$FISH_{PMA}$	10	0.76	0.019	0.011	0.59	0.49	2.32
	FAM_{PMA}=0.01+0.69·$FISH_{PMA}$	10	0.74	0.012	0.014	0.59	0.42	1.83
	ORD_{PMA}=-0.06+0.63·$FISH_{PMA}$	10	0.73	0.012	0.017	0.59	0.31	2.09
9. fish as 3-pass biomass, benthos from riffles	GEN_{PMA}=0.08+0.60·$FISH_{PMA}$	10	0.64	0.017	0.048	0.59	0.43	1.10
	FAM_{PMA}=-0.01+0.66·$FISH_{PMA}$	10	0.72	0.013	0.020	0.59	0.38	1.68
	ORD_{PMA}=0.03+0.47·$FISH_{PMA}$	10	0.65	0.010	0.044	0.59	0.31	1.18
Canagagigue								
10. fish as 3-pass biomass, benthos from runs	GEN_{PMA}=-0.16+0.64·$FISH_{PMA}$	10	0.79	0.005	0.007	0.86	0.39	2.13
	FAM_{PMA}=-0.22+0.73·$FISH_{PMA}$	10	0.80	0.006	0.006	0.86	0.41	3.82
	ORD_{PMA}=-0.14+0.50·$FISH_{PMA}$	10	0.67	0.007	0.035	0.86	0.29	2.35
11. fish as 3-pass biomass, benthos from riffles	GEN_{PMA}=-0.13+0.54·$FISH_{PMA}$	10	0.85	0.003	0.002	0.86	0.33	1.52
	FAM_{PMA}=-0.12+0.47·$FISH_{PMA}$	10	0.83	0.002	0.003	0.86	0.28	1.61
	ORD_{PMA}=-0.05+0.30·$FISH_{PMA}$	10	0.69	0.002	0.028	0.86	0.21	1.56

Table 5.4. Probabilities of true and false hits: i.e., the likelihood of making a statement that fish communities are outside of the normal range of reference conditions when they truly are (true hit), or when they truly are not (false hit), using surveys of benthos. Data are for five separate studies as described in the text. Values are percentages with exact 95% confidence regions enclosed in brackets.

Data Set	Taxonomic Level	True Hits (%)	False Hits (%)
buffer-strip study			
1. brook trout streams as reference, August benthos	species	100	0
	genus	100	0
	family	94.4 (78.4, 99.3)	0
	order	83.3 (57.8, 97.8)	0
2. brook trout streams as reference, May benthos	species	100	33.3 (9.6, 89.5)
	genus	100	33.3 (9.6, 89.5)
	family	94.4 (78.4, 99.3)	33.3 (9.6, 89.5)
	order	50.0 (23.2, 90.0)	0
3. marginal brook trout streams as reference, August benthos	species	70.0 (40.2,95.4)	44.4 (17.6, 89.5)
	genus	56.7 (27.9, 92.2)	44.4 (17.6, 89.5)
	family	70.0 (40.2, 95.4)	44.4 (17.6, 89.5)
	order	56.7 (27.9, 92.2)	44.4 (17.6, 89.5)
4. marginal brook trout streams as reference, May benthos	species	83.3 (57.1, 97.8)	44.4 (17.6, 89.5)
	genus	63.3 (33.6, 93.9)	22.2 (7.4, 78.5)
	family	66.7 (36.8, 94.7)	22.2 (7.4, 78.5)
	order	43.3 (18.8, 87.5)	11.1 (3.4, 68.1)
MNR-HSI Exercise			
5. brook-trout streams as reference	genus	93.5 (78.6, 99.2)	0
	family	64.5 (34.8, 94.2)	20.0 (6.0, 81.0)
	order	83.9 (58.1, 97.9)	20.0 (6.0, 81.0)
6. marginal trout streams as reference	genus	66.7 (35.9, 94.8)	43.8 (18.3, 88.4)
	family	76.2 (46.0, 96.7)	37.5 (14.8, 85.6)
	order	52.4 (24.2, 91.1)	43.8 (18.3, 88.4)

Data Set	Taxonomic Level	True Hits (%)	False Hits (%)
buffer-strip study			
London subwatershed study			
7. creek chub streams as reference	species	55.6 (26.2, 92.1)	27.6 (10.4, 78.5)
	genus	55.6 (26.2, 92.1)	27.6 (10.4, 78.5)
	family	44.4 (18.9, 88.5)	27.6 (10.4, 78.5)
	order	55.6 (26.2, 92.1)	31.0 (12.0, 81.0)
Laurel Creek			
8. run habitats	genus	100	0
	family	100	0
	order	100	0
9. riffle habitats	genus	100	0
	family	100	0
	order	100	0
Canagagigue Creek			
10. run habitats	genus	85.7 (42.1, 99.0)	0
	family	85.7 (42.1, 99.0)	0
	order	85.7 (42.1, 99.0)	0
11. riffle habitats	genus	100	0
	family	71.4 (29.1, 97.7)	0
	order	71.4 (29.1, 97.7)	0

Table 5.5. Principal components analysis of fish-benthos regression statistics, characteristics of benthic critical values, taxonomic level and spatial scale of fish and benthos surveys. Correlations between these variables and the principal component axes are given.

	Principal Component Axis		
Variable	1	2	3
Intercept	-0.850	0.438	-0.127
Slope	0.893	0.159	0.175
Correlation	0.810	0.281	0.318
Critical Benthic Effect Size	0.534	0.031	-0.812
Taxonomic Level	0.118	-0.993	0.182
Spatial Scale	-0.922	0.133	0.084
% True Positives	0.639	0.509	0.188
% False Positives	-0.770	0.137	0.198

Table 5.6. Results of analysis of variance (ANOVA) testing sources of variation in the size of (1) critical benthic values. *df* = degrees of freedom, *SS* = sum of squares, *MS* = mean square, *F* = test statistic, *P* = probability that the source does not affect the dependent variable.

Taxonomic Level	Source	*df*	*SS*	*MS*	*F*	*P*
Genus	Scale	1	1.483	1.483	1.833	0.209
	Error	9	7.282	0.809		
Family	Scale	1	4.241	4.241	6.139	0.035
	Error	9	6.216	0.691		
Order	Scale	1	4.421	4.421	23.445	0.001
	Error	9	1.697	0.189		
Genus	Sample Type	1	5.608	5.608	15.991	0.003
	Error	9	3.156	0.351		
Family	Sample Type	1	4.268	4.268	6.207	0.034
	Error	9	6.189	0.688		
Order	Sample Type	1	3.296	3.296	10.512	0.010
	Error	9	2.822	0.314		
Genus	Habitat Type	1	0.837	0.837	15.760	0.058
	Error	2	0.106	0.053		
Family	Habitat Type	1	1.392	1.392	1.405	0.358
	Error	2	1.983	0.991		
Order	Habitat Type	1	0.723	0.723	13.632	0.066
	Error	2	0.106	0.053		

Table 5.7. Results of analysis of variance (ANOVA) testing sources of variation in true-positive probabilities. *df* = degrees of freedom, *SS* = sum of squares, *MS* = mean square, *F* = test statistic, *P* = probability that the source does not affect the dependent variable.

Taxonomic Level	Source	*df*	*SS*	*MS*	*F*	*P*
Genus	Scale	1	1006	1006	3.42	0.097
	Error	9	2645	293		
Family	Scale	1	683	683	2.55	0.145
	Error	9	2410	267		
Order	Scale	1	2072	2072	8.72	0.016
	Error	9	2139	237		
Genus	Sample Type	1	184	184	0.479	0.506
	Error	9	3467	385		
Family	Sample Type	1	19	19	0.057	0.817
	Error	9	3074	341		
Order	Sample Type	1	1195	1195	3.57	0.091
	Error	9	3015	335		
Genus	Habitat Type	1	51	51	1.00	0.423
	Error	2	102	51		
Family	Habitat Type	1	51	51	0.20	0.698
	Error	2	511	255		
Order	Habitat Type	1	51	51	0.20	0.698
	Error	2	511	255		

Table 5.8. Results of analysis of variance (ANOVA) testing sources of variation in false-positive probabilities. *df* = degrees of freedom, *SS* = sum of squares, *MS* = mean square, *F* = test statistic, *P* = probability that the source does not affect the dependent variable.

Taxonomic Level	Source	*df*	*SS*	*MS*	*F*	*P*
Genus	Scale	1	1524	1524	6.655	0.030
	Error	9	2061	229		
Family	Scale	1	1777	1777	12.78	0.006
	Error	9	1251	139		
Order	Scale	1	1173	1173	4.92	0.054
	Error	9	2146	238		
Genus	Sample Type	1	926	926	3.14	0.110
	Error	9	2659	295		
Family	Sample Type	1	1170	1170	5.66	0.041
	Error	9	1859	206		
Order	Sample Type	1	510	510	1.64	0.233
	Error	9	2809	312		
Genus	Habitat Type	1	no variation in false positives			
	Error	2				
Family	Habitat Type	1				
	Error	2				
Order	Habitat Type	1				
	Error	2				

Table 5.9. Gradient lengths of benthic communities for each of the studies listed in Table 5.1.

Data set	Taxonomic level	Gradient length (standard deviations)
buffer strip, August benthos	species	3.08
	genus	2.77
	family	2.29
	order	1.39
MNR-HSI	genus	2.91
	family	2.89
	order	2.01
London subwatershed	species	1.64
	genus	1.66
	family	1.61
	order	1.43
Laurel Creek - run benthos	genus	2.29
	family	1.98
	order	1.61
Laurel Creek - riffle benthos	genus	2.23
	family	1.86
	order	1.63
Canagagigue Creek - run benthos	genus	1.36
	family	1.55
	order	1.10
Canagagigue Creek - riffle benthos	genus	1.27
	family	1.20
	order	0.87

Table 5.10. Linear regressions relating benthic and fish community percent-model affinities (PMAs) for Laurel and Canagagigue data sets. Fish PMAs were based on one-pass-CPUE-biomass and one-pass-% biomass. Benthos were collected from riffles and runs, while PMAs were calculated at three levels of benthic taxonomy (genus = GEN_{PMA}, family = FAM_{PMA}, order = ORD_{PMA}). Sample sizes (n), correlation coefficients (r), mean-squared errors (MSE), and probability (p) values for the models are given, as are the critical fish effect sizes (PMA_c-fish) from Table 5.2. Based on these models, critical benthic effect sizes are estimated and expressed in terms of affinity (PMA_c-benthos), as well as the estimated number of standard deviations (SDs) from the reference PMA. Sample sizes are also given.

Data Set	Model	n	r	MSE	p	PMA_c-fish	PMA_c-benthos	# SDs
Laurel								
fish as 1-pass biomass, benthos from runs	GEN_{PMA}=-0.11+0.89·$FISH_{PMA}$	10	0.85	0.013	0.002	0.706	0.518	2.67
	FAM_{PMA}=-0.04+0.68·$FISH_{PMA}$	10	0.83	0.009	0.003	0.706	0.440	2.07
	ORD_{PMA}=-0.12+0.64·$FISH_{PMA}$	10	0.83	0.008	0.003	0.706	0.332	2.44
fish as 1-pass biomass, benthos from riffles	GEN_{PMA}=0.01+0.62·$FISH_{PMA}$	10	0.74	0.013	0.016	0.706	0.448	1.24
	FAM_{PMA}=-0.06+0.64·$FISH_{PMA}$	10	0.79	0.010	0.007	0.706	0.392	1.81
	ORD_{PMA}=-0.00+0.46·$FISH_{PMA}$	10	0.72	0.008	0.018	0.706	0.324	1.33
fish as 1-pass-% biomass, benthos from runs	GEN_{PMA}=-0.11+0.90·$FISH_{PMA}$	10	0.86	0.012	0.001	0.727	0.544	3.00
	FAM_{PMA}=-0.05+0.69·$FISH_{PMA}$	10	0.84	0.008	0.002	0.727	0.452	2.21
	ORD_{PMA}=-0.12+0.64·$FISH_{PMA}$	10	0.84	0.007	0.003	0.727	0.345	2.64
fish as 1-pass-% biomass, benthos from riffles	GEN_{PMA}=0.03+0.63·$FISH_{PMA}$	10	0.75	0.013	0.012	0.727	0.488	1.56
	FAM_{PMA}=-0.06+0.65·$FISH_{PMA}$	10	0.81	0.010	0.005	0.727	0.413	2.04
	ORD_{PMA}=-0.01+0.47·$FISH_{PMA}$	10	0.74	0.008	0.014	0.727	.332	1.41
Canagagigue								
fish as 1-pass biomass, benthos from runs	GEN_{PMA}=0.06+0.41·$FISH_{PMA}$	10	0.85	0.004	0.002	0.840	0.404	2.32
	FAM_{PMA}=0.06+0.43·$FISH_{PMA}$	10	0.79	0.006	0.006	0.840	0.421	4.04
	ORD_{PMA}=0.04+0.31·$FISH_{PMA}$	10	0.70	0.006	0.024	0.840	0.300	2.54
fish as 1-pass biomass, benthos from runs	GEN_{PMA}=0.06+0.34·$FISH_{PMA}$	10	0.91	0.001	0.000	0.840	0.346	1.73
	FAM_{PMA}=0.06+0.28·$FISH_{PMA}$	10	0.84	0.002	0.002	0.840	0.295	1.85
	ORD_{PMA}=0.06+0.19·$FISH_{PMA}$	10	0.72	0.002	0.020	0.840	0.220	1.79
fish as 1-pass-% biomass, benthos from runs	GEN_{PMA}=0.11+0.37·$FISH_{PMA}$	10	0.85	0.004	0.002	0.868	0.431	2.69
	FAM_{PMA}=0.10+0.38·$FISH_{PMA}$	10	0.79	0.007	0.006	0.868	0.430	4.22
	ORD_{PMA}=0.07+0.28·$FISH_{PMA}$	10	0.71	0.006	0.023	0.868	0.313	2.78

Data Set	Model	n	r	MSE	p	PMA_c-fish	PMA_c-benthos	# SDs
fish as 1-pass-% biomass, benthos from riffles	$GEN_{PMA}=0.09+0.30 \cdot FISH_{PMA}$	10	0.90	0.002	0.000	0.868	0.350	1.78
	$FAM_{PMA}=0.09+0.24 \cdot FISH_{PMA}$	10	0.81	0.002	0.004	0.868	0.298	1.90
	$ORD_{PMA}=0.08+0.16 \cdot FISH_{PMA}$	10	0.70	0.002	0.025	0.868	0.219	1.77

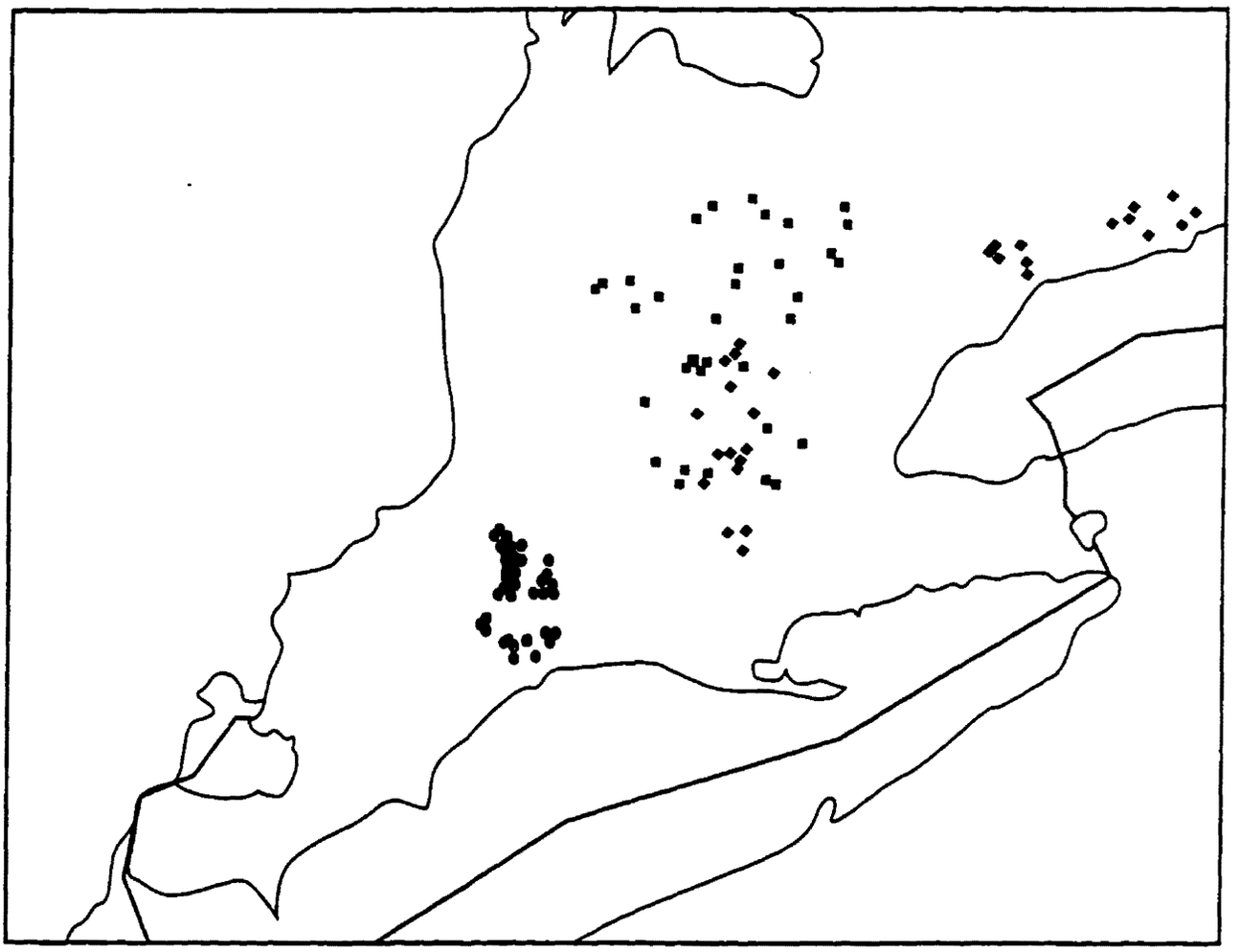
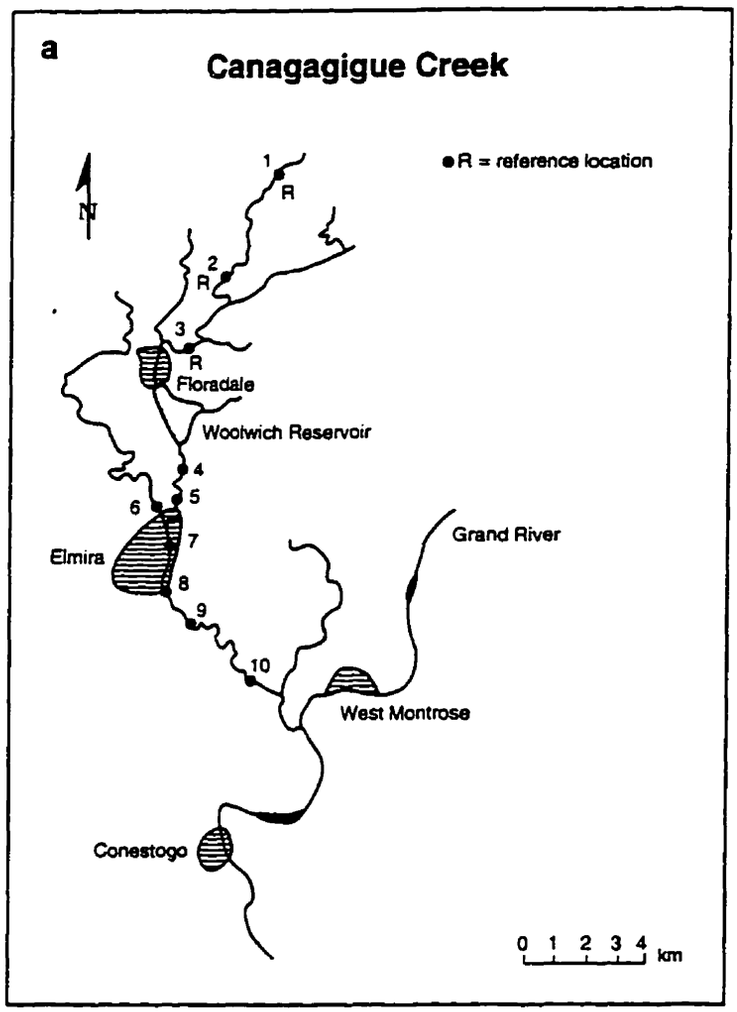


Figure 5.1. Map of large-scale spatial study locations in southern Ontario. Studies are:•=London subwatershed study; ■=Barton et al. (1985); and, ♦=MNR-HSI study.



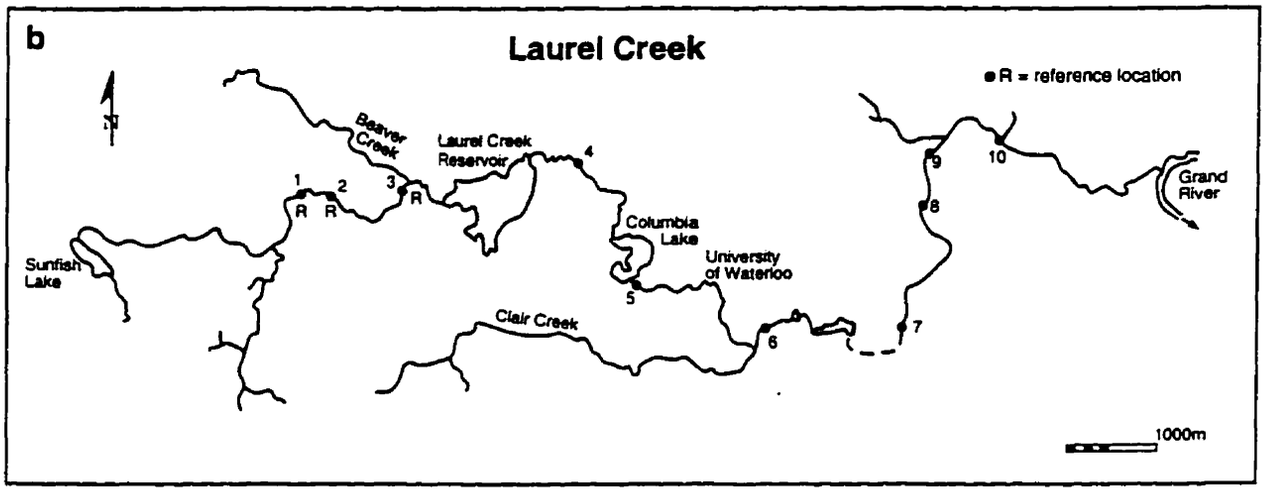


Figure 5.2. Map of small spatial scale study locations in (a) Canagagigue and (b) Laurel Creeks.

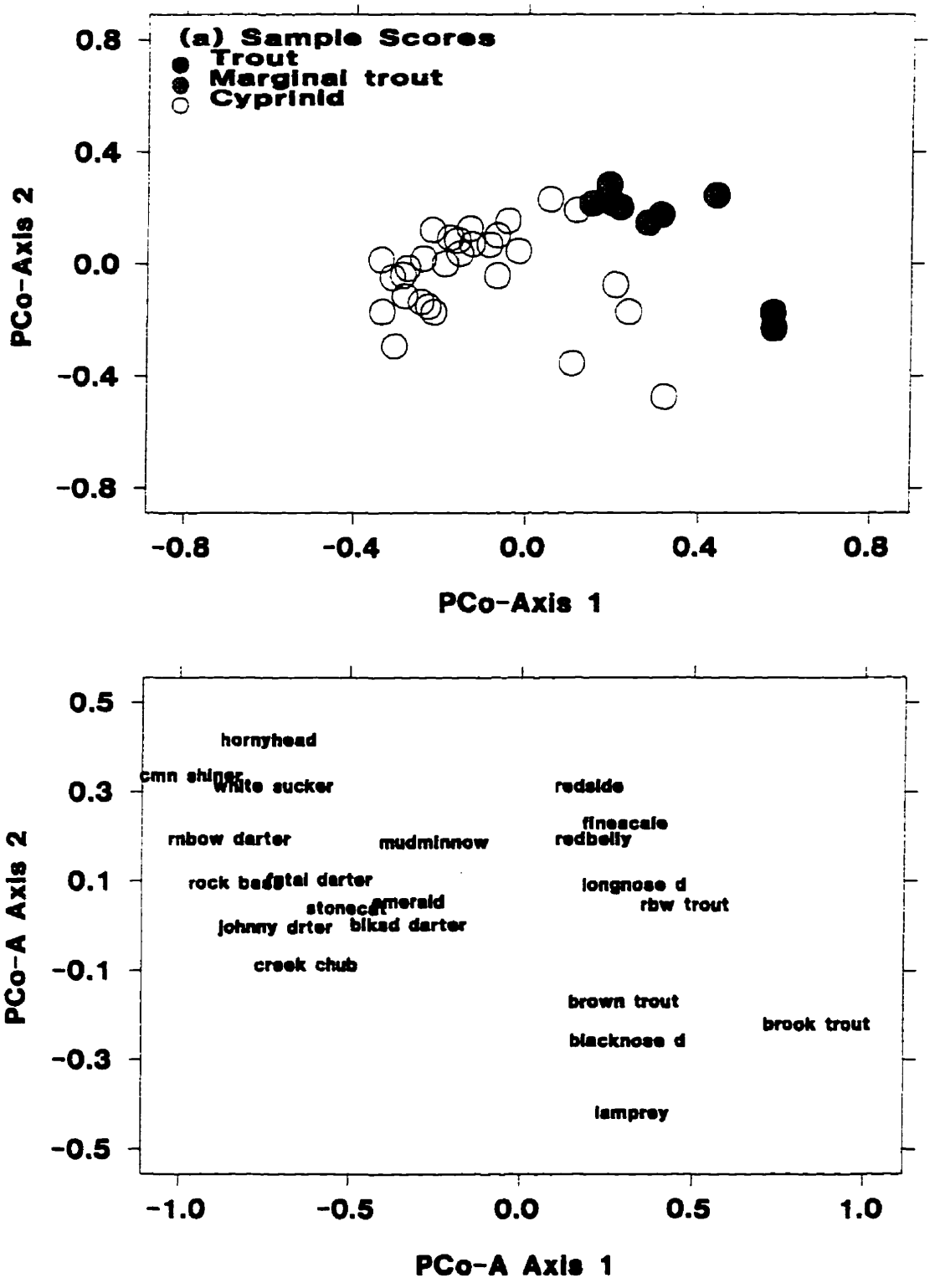


Figure 5.3. Scatterplots of (a) sample scores along principal coordinate analysis (PCo-A) axes, and (b) taxa correlations with the same PCo-A axes for the buffer-strip fish community data. See text and Table 5.1 for details on the data set and the ordination. See Appendix A for latin names for fish species.

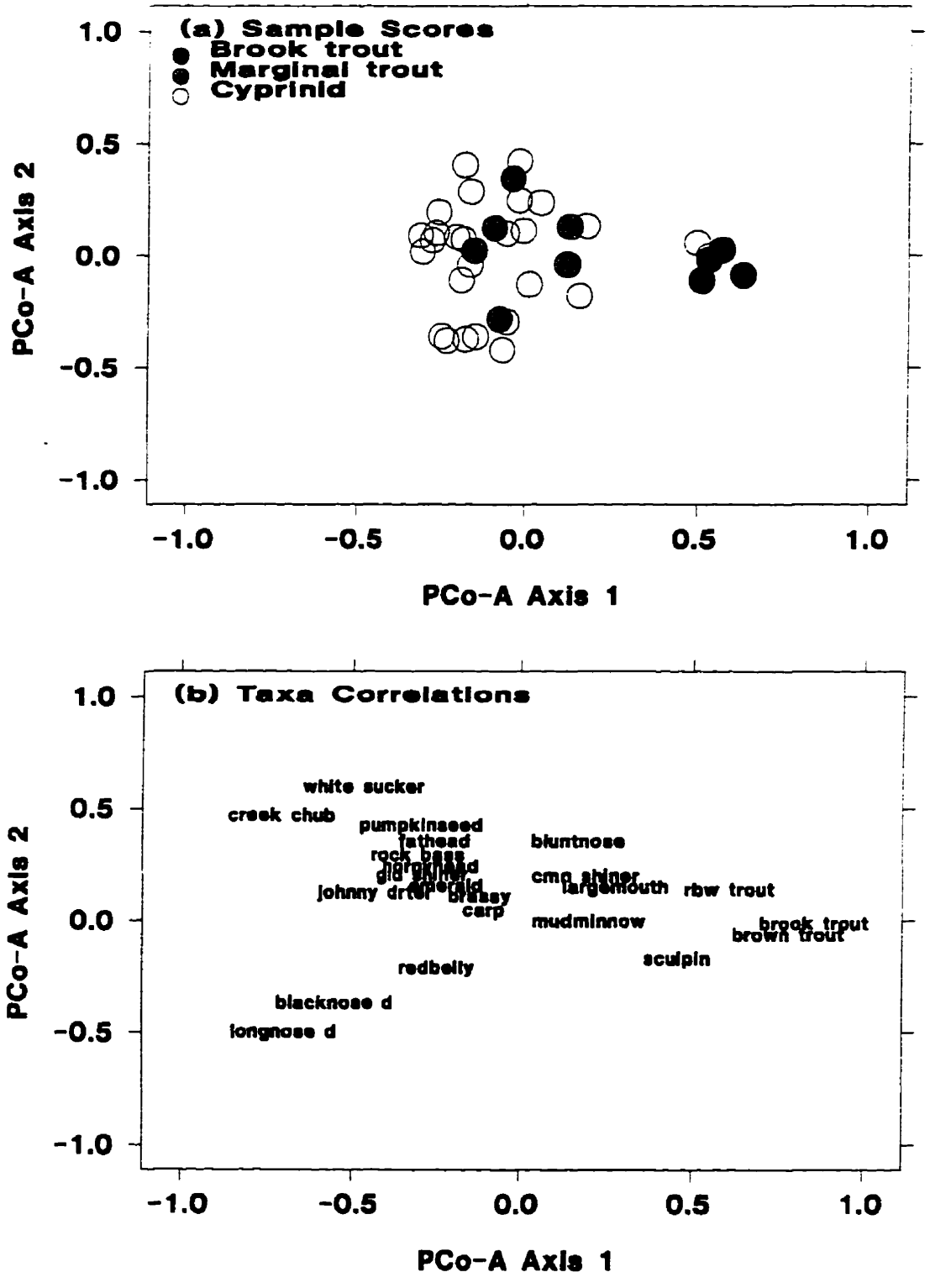


Figure 5.4. Scatterplots of (a) sample scores along principal coordinate analysis (PCo-A) axes, and (b) taxa correlations with the same PCo-A axes for the MNR-HSI fish community data. See text and Table 5.1 for details on the data set and the ordination. See Appendix A for latin names for fish species.

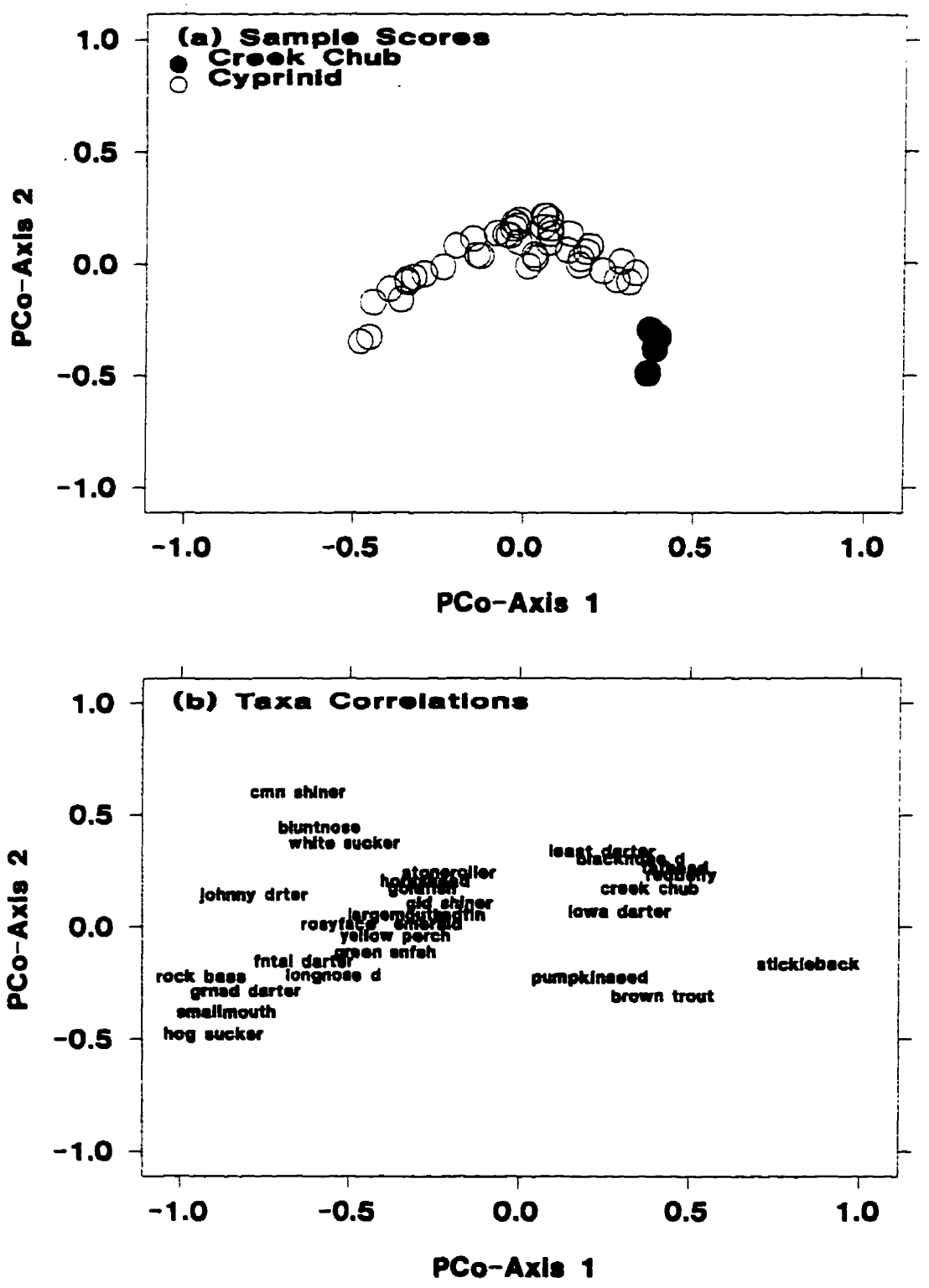


Figure 5.5. Scatterplots of (a) sample scores along principal coordinate analysis (PCo-A) axes, and (b) taxa correlations with the same PCo-A axes for the London subwatershed fish community data. See text and Table 5.1 for details on the data set and the ordination. See Appendix A for latin names for fish species.

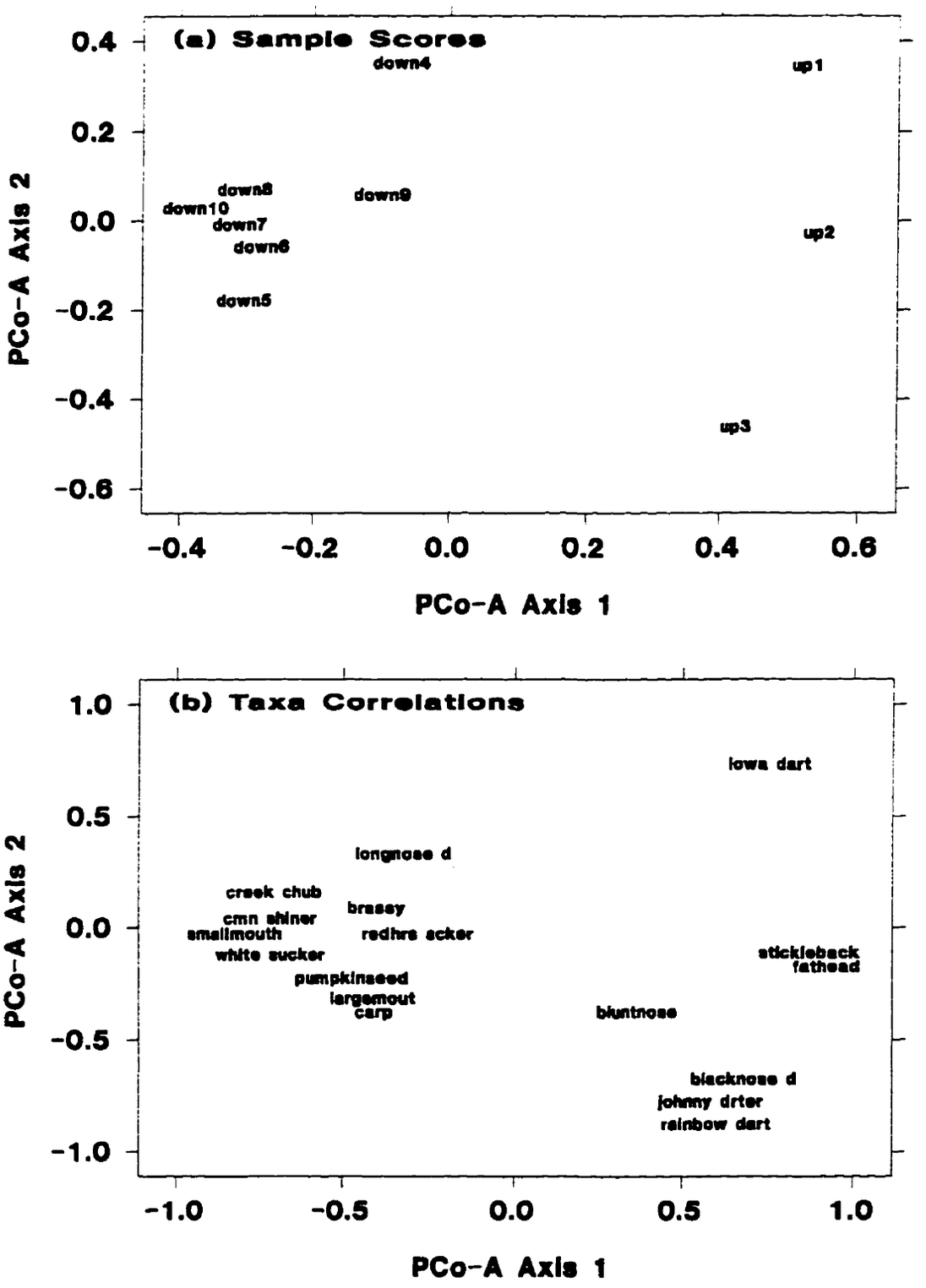


Figure 5.6. Scatterplots of (a) sample scores along principal coordinate analysis (PCo-A) axes, and (b) taxa correlations with the same PCo-A axes for the Laurel Creek fish community data. See text and Table 5.1 for details on the data set and the ordination. See Appendix A for latin names for fish species.

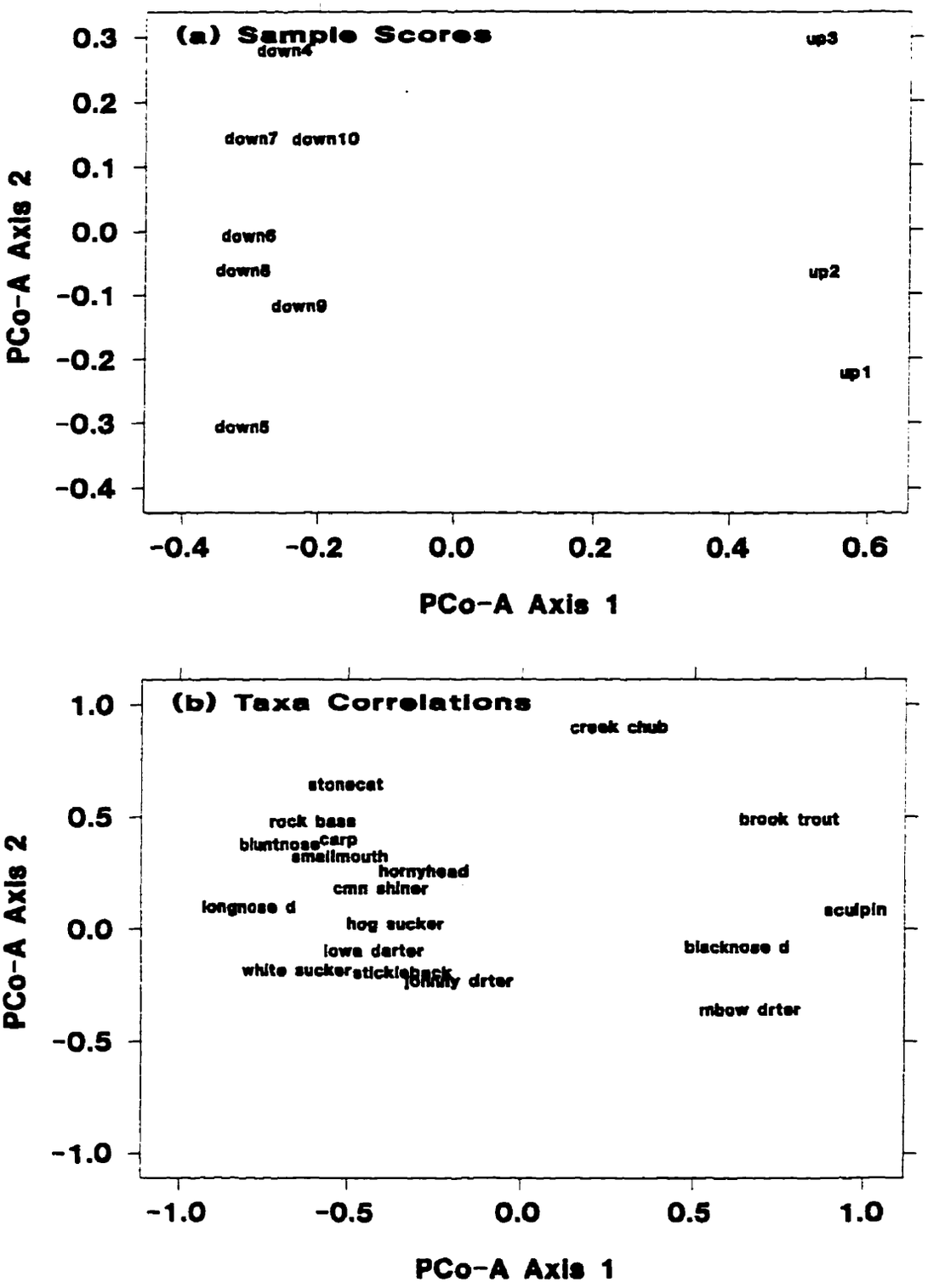


Figure 5.7. Scatterplots of (a) sample scores along principal coordinate analysis (PCo-A) axes, and (b) taxa correlations with the same PCo-A axes for the Canagagigue Creek fish community data. See text and Table 5.1 for details on the data set and the ordination. See Appendix A for latin names for fish species.

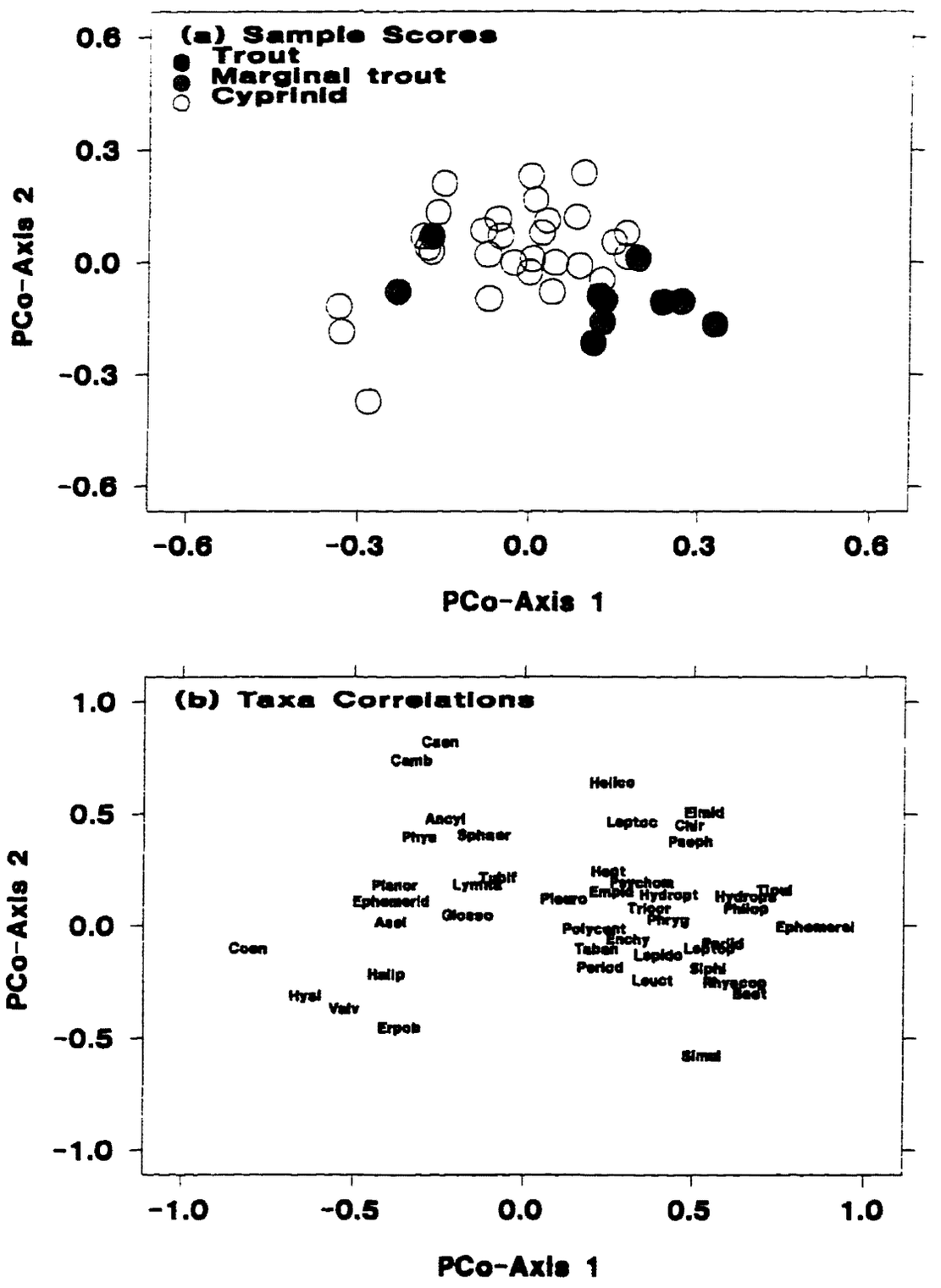


Figure 5.8. Scatterplots of (a) sample scores along principal coordinate analysis (PCo-A) axes, and (b) taxa correlations with the same PCo-A axes for the buffer-strip August benthic community data. See text and Table 5.1 for details on the data set and the ordination. See Appendix D for full benthos family names.

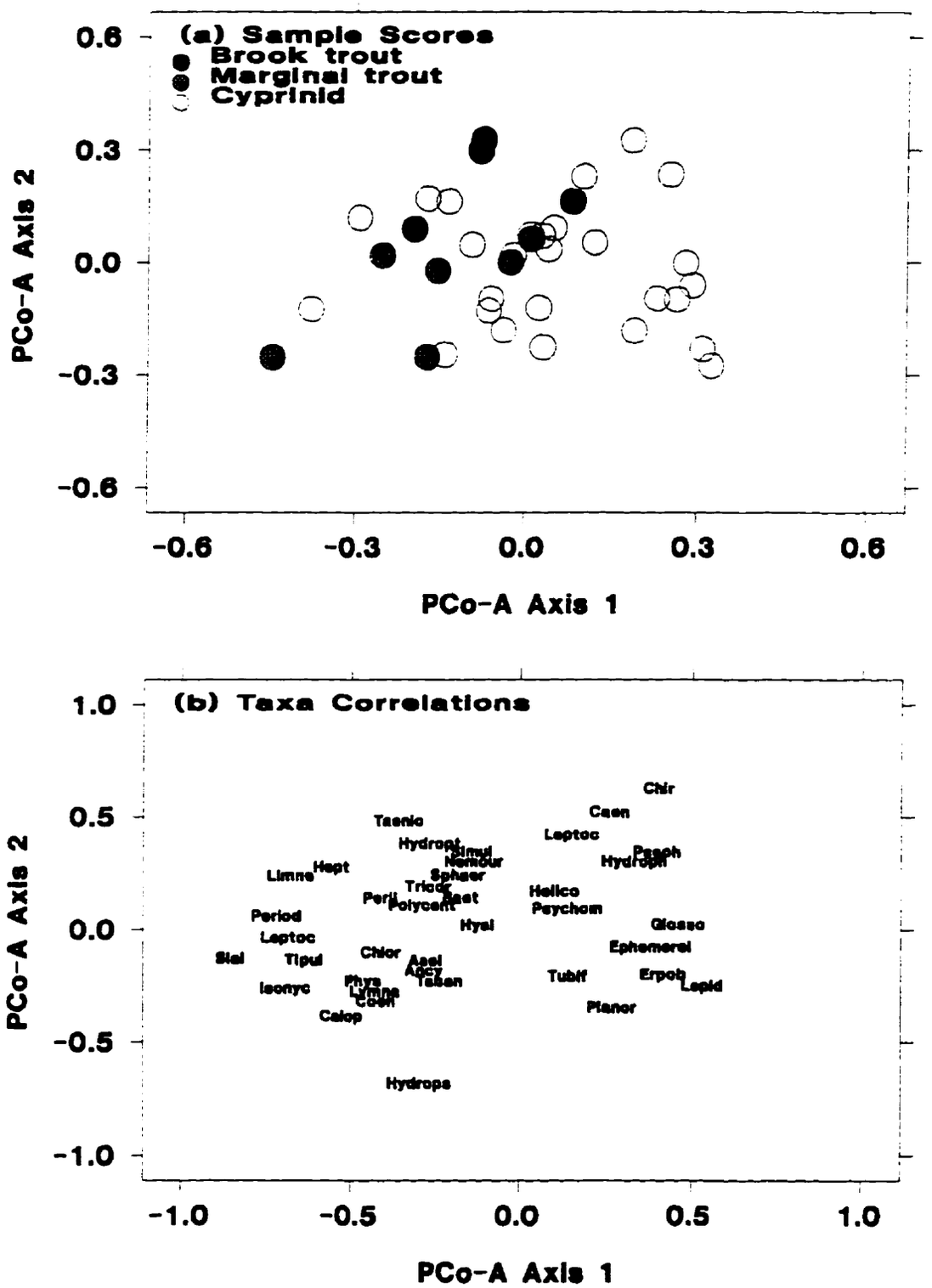


Figure 5.9. Scatterplots of (a) sample scores along principal coordinate analysis (PCo-A) axes, and (b) taxa correlations with the same PCo-A axes for the MNR-HSI benthic community data. See text and Table 5.1 for details on the data set and the ordination. See Appendix D for full benthos family names.

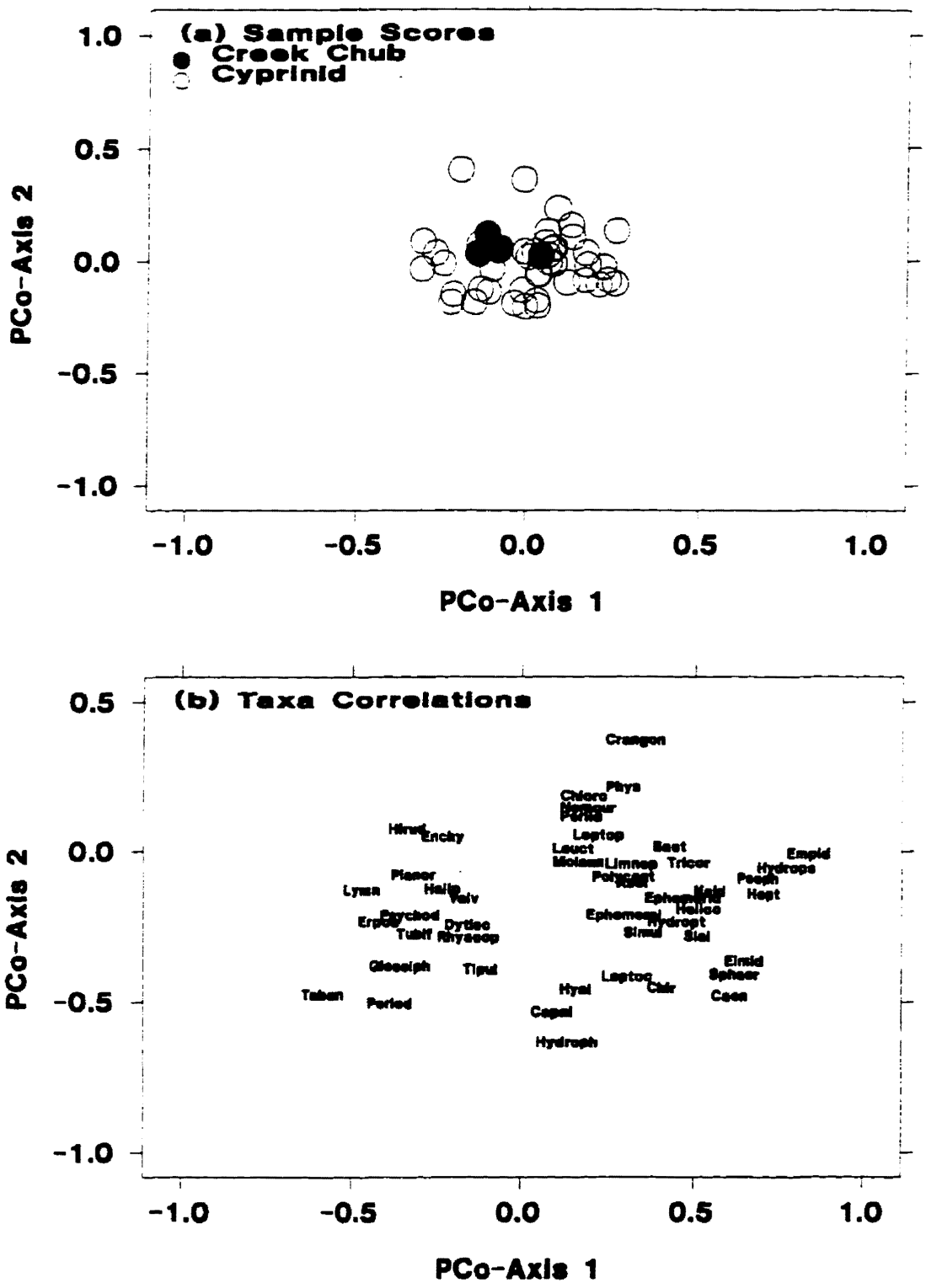


Figure 5.10. Scatterplots of (a) sample scores along principal coordinate analysis (PCo-A) axes, and (b) taxa correlations with the same PCo-A axes for the London subwatershed benthic community data. See text and Table 5.1 for details on the data set and the ordination. See Appendix D for full benthos family names.

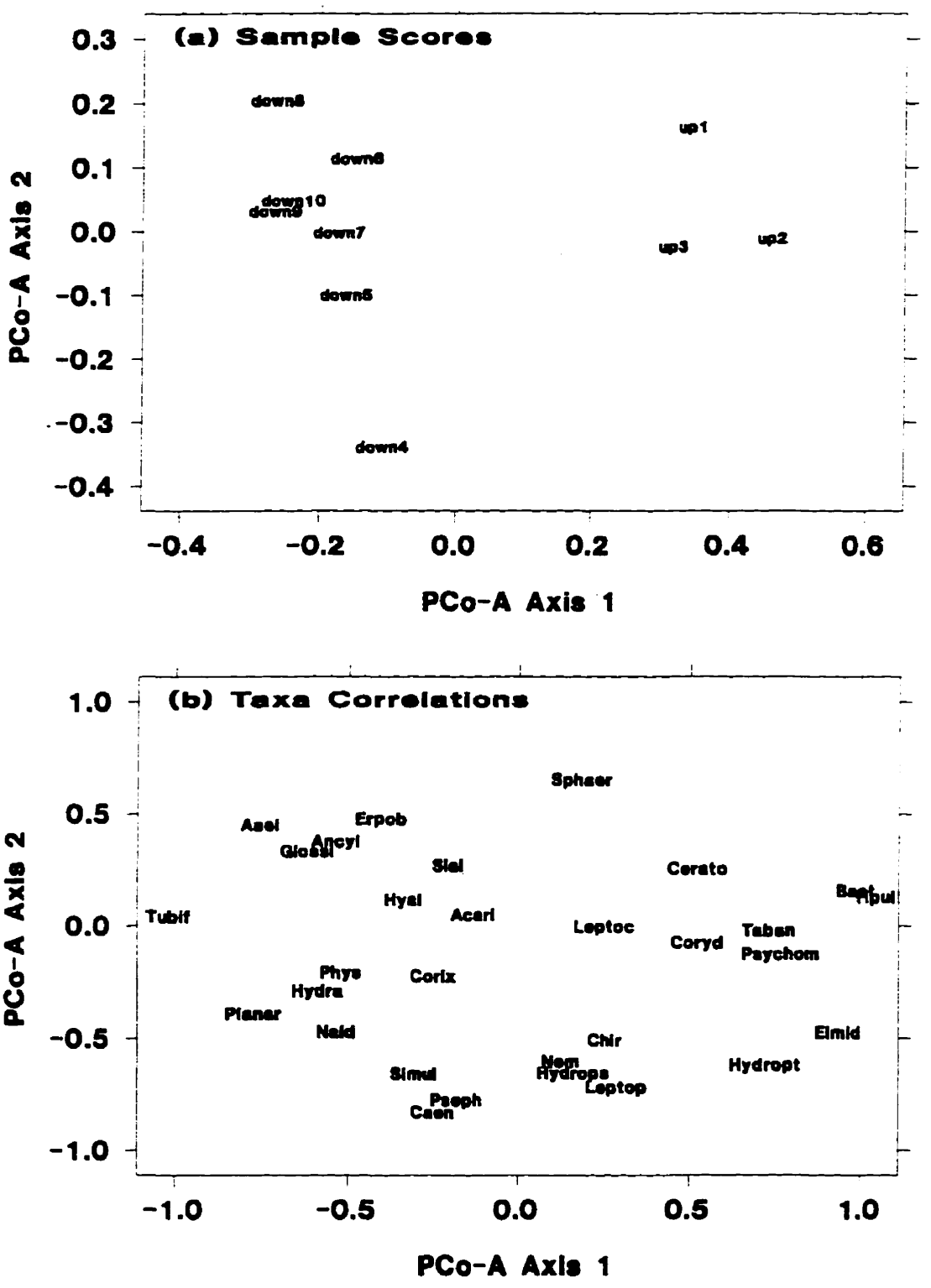


Figure 5.11. Scatterplots of (a) sample scores along principal coordinate analysis (PCo-A) axes, and (b) taxa correlations with the same PCo-A axes for the Laurel Creek benthic community run data. See text and Table 5.1 for details on the data set and the ordination. See Appendix D for full benthos family names.

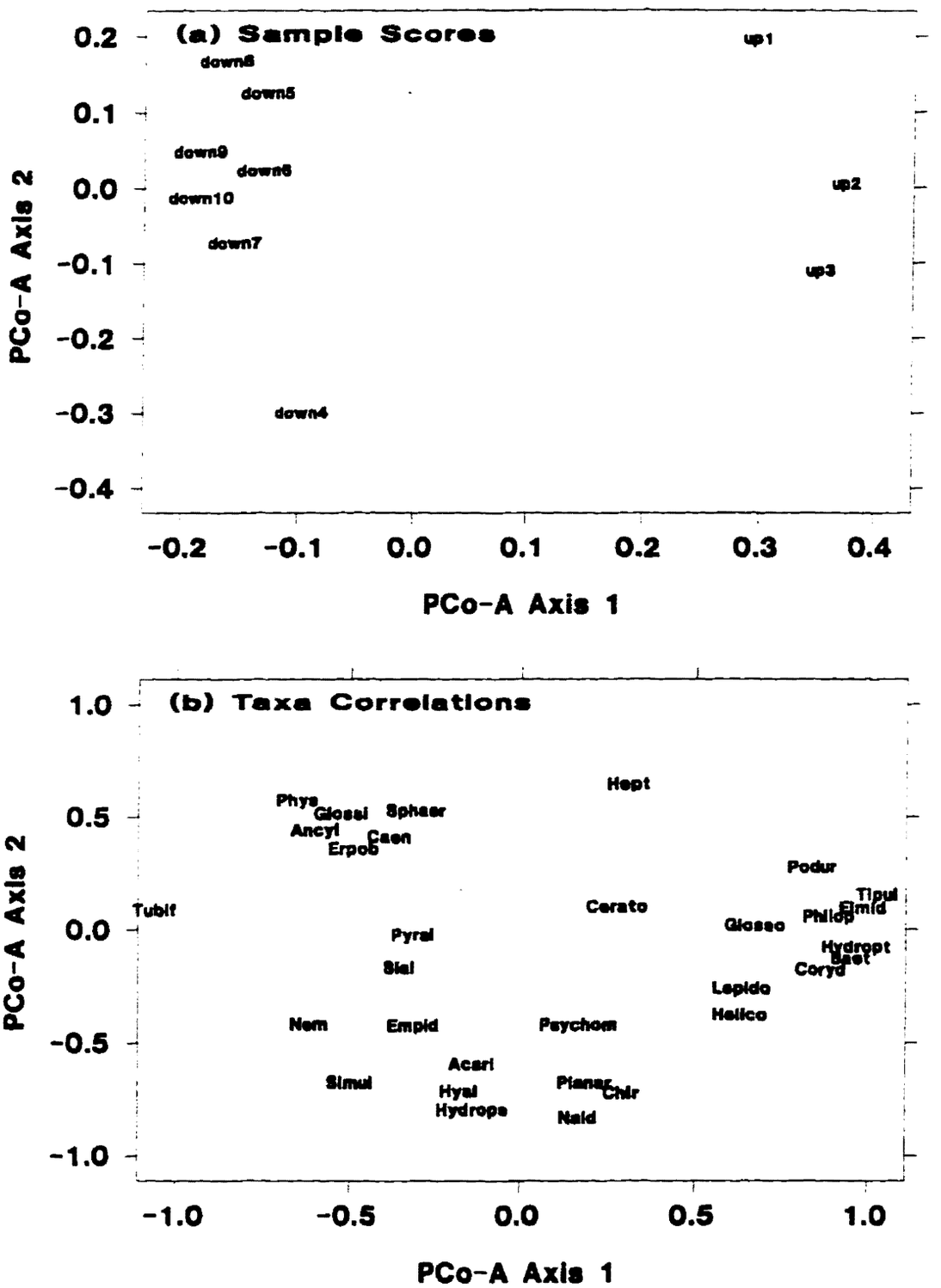


Figure 5.12. Scatterplots of (a) sample scores along principal coordinate analysis (PCo-A) axes, and (b) taxa correlations with the same PCo-A axes for the Laurel Creek benthic community riffle data. See text and Table 5.1 for details on the data set and the ordination. See Appendix D for full benthos family names.

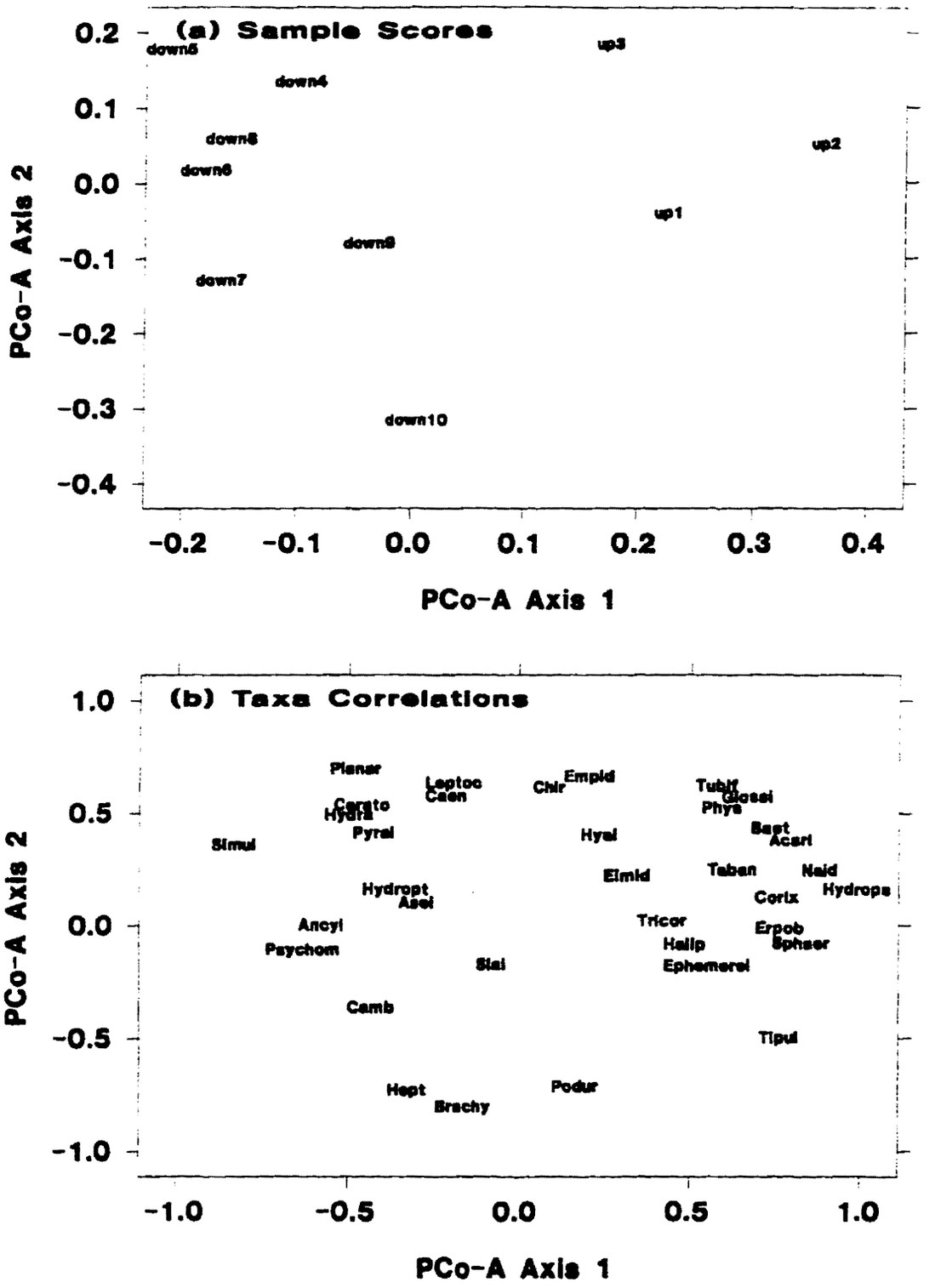


Figure 5.13. Scatterplots of (a) sample scores along principal coordinate analysis (PCo-A) axes, and (b) taxa correlations with the same PCo-A axes for the Canagagigue benthic community run data. See text and Table 5.1 for details on the data set and the ordination. See Appendix D for full benthos family names.

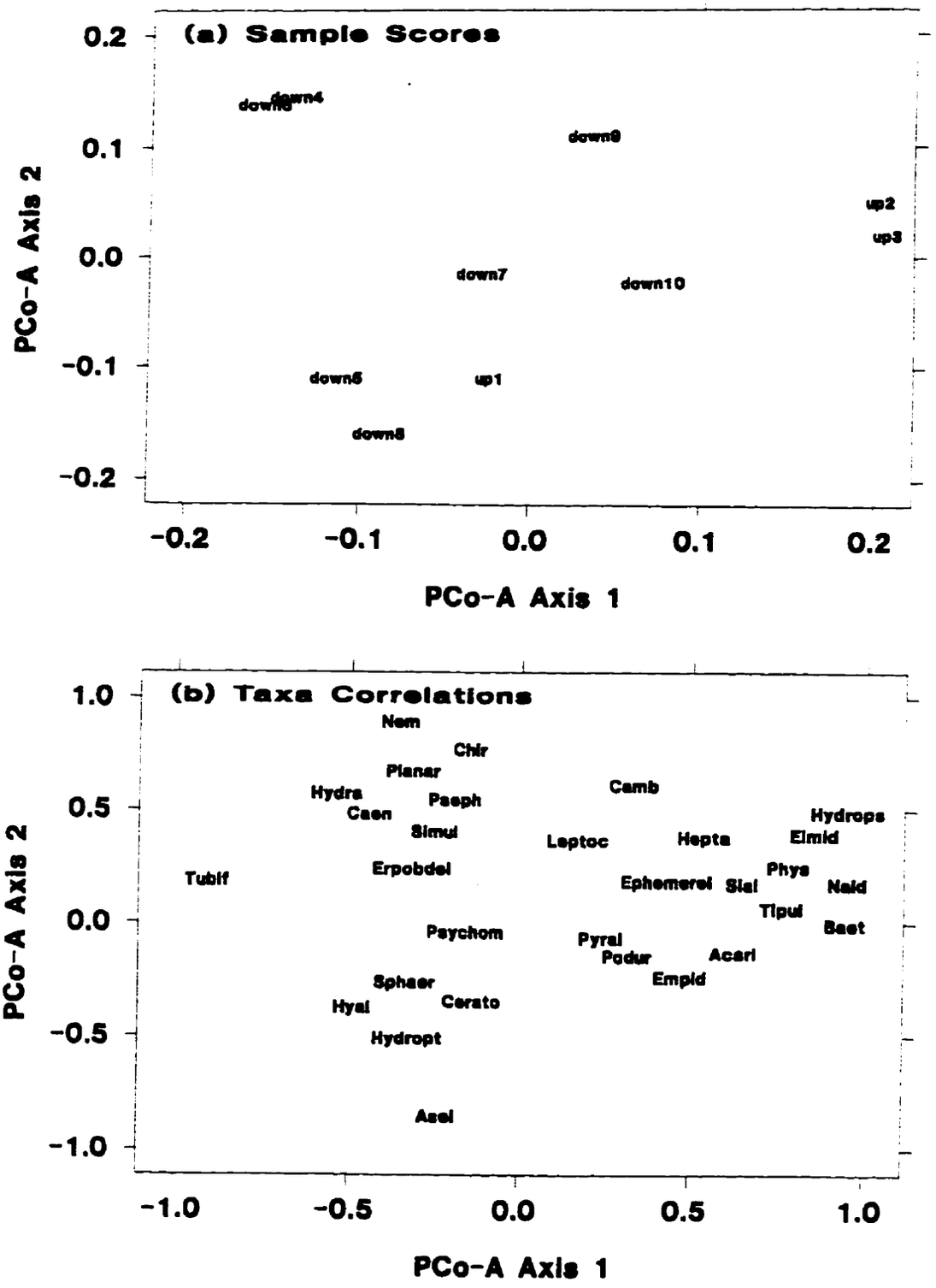


Figure 5.14. Scatterplots of (a) sample scores along principal coordinate analysis (PCo-A) axes, and (b) taxa correlations with the same PCo-A axes for the Canagagigue Creek benthic community riffle data. See text and Table 5.1 for details on the data set and the ordination. See Appendix D for full benthos family names.

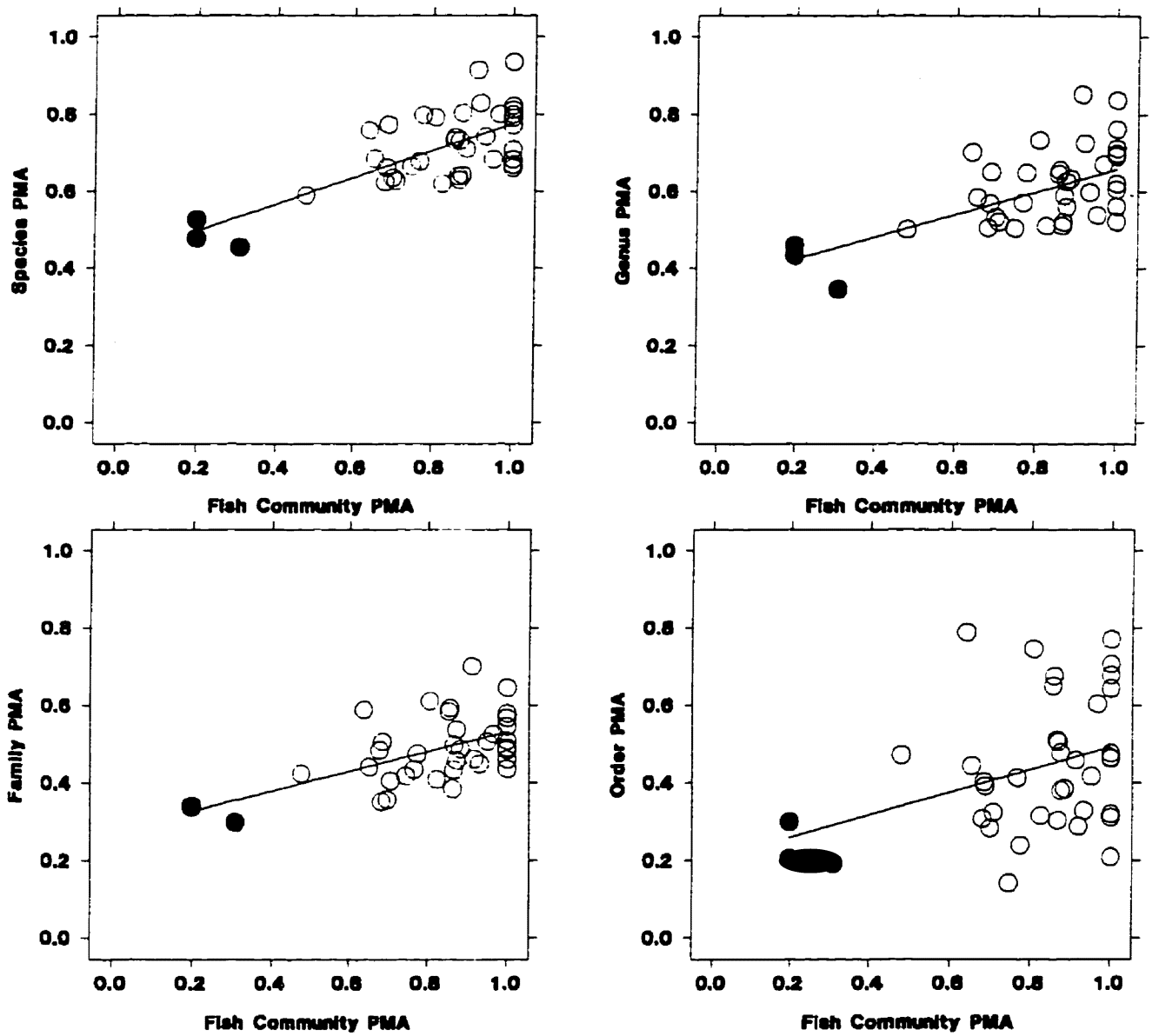


Figure 5.15. Relationship between $BENTHOS_{PMA}$ and $FISH_{PMA}$ for the buffer-strip data using brook-trout streams as the reference and August benthic surveys. Relationships for benthos identified to species, genus, family and order are given. PMA is based on Bray-Curtis (BC) distances. Regression equations and diagnostics are given in Table 5.3.

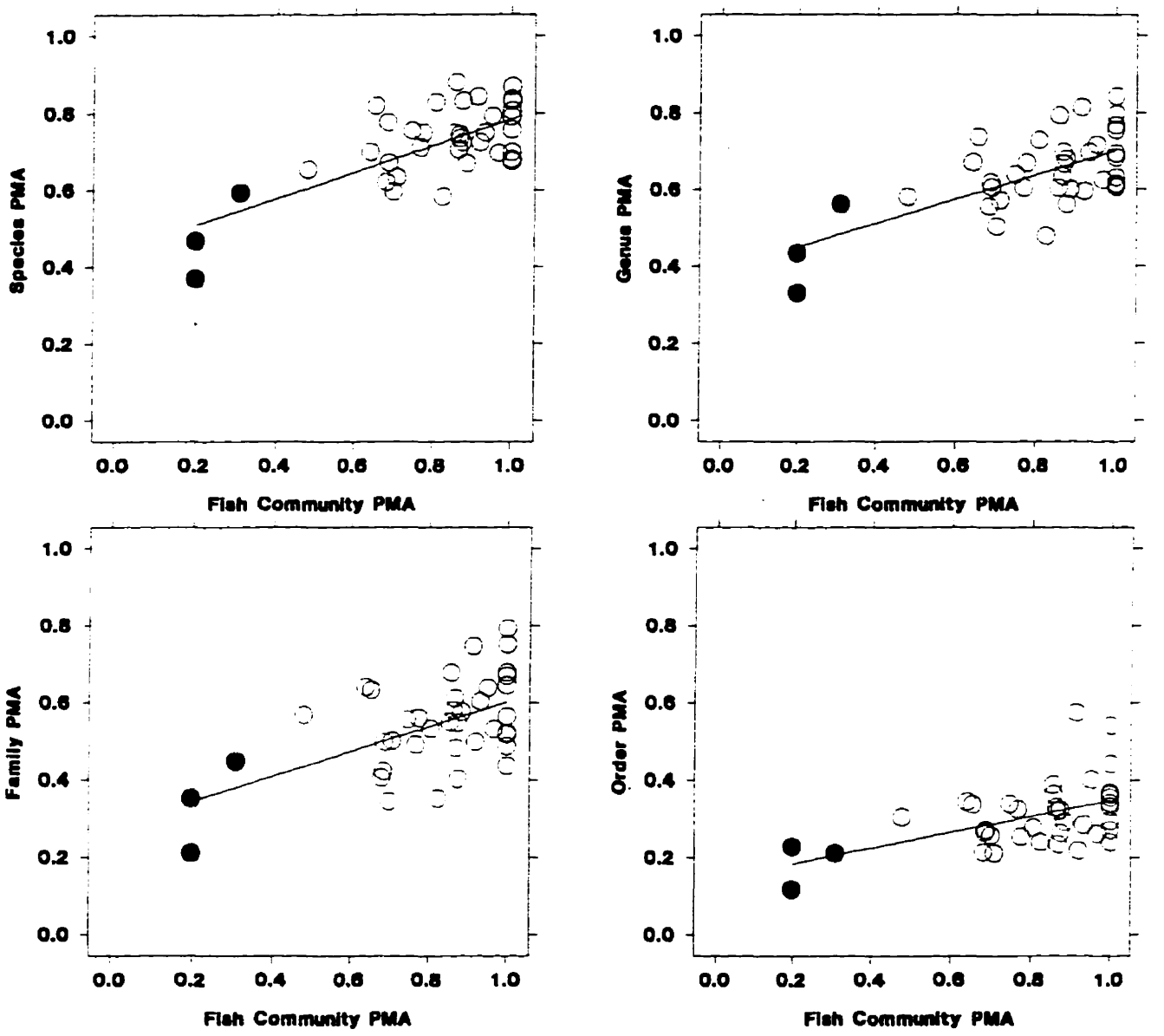


Figure 5.16. Relationship between $BENTHOS_{PMA}$ and $FISH_{PMA}$ for the buffer-strip data using brook-trout streams as the reference and May benthic surveys. Relationships based on benthos identified to species, genus, family and order are given. PMA is based on Bray-Curtis (BC) distances. Regression equations and diagnostics are given in Table 5.3.

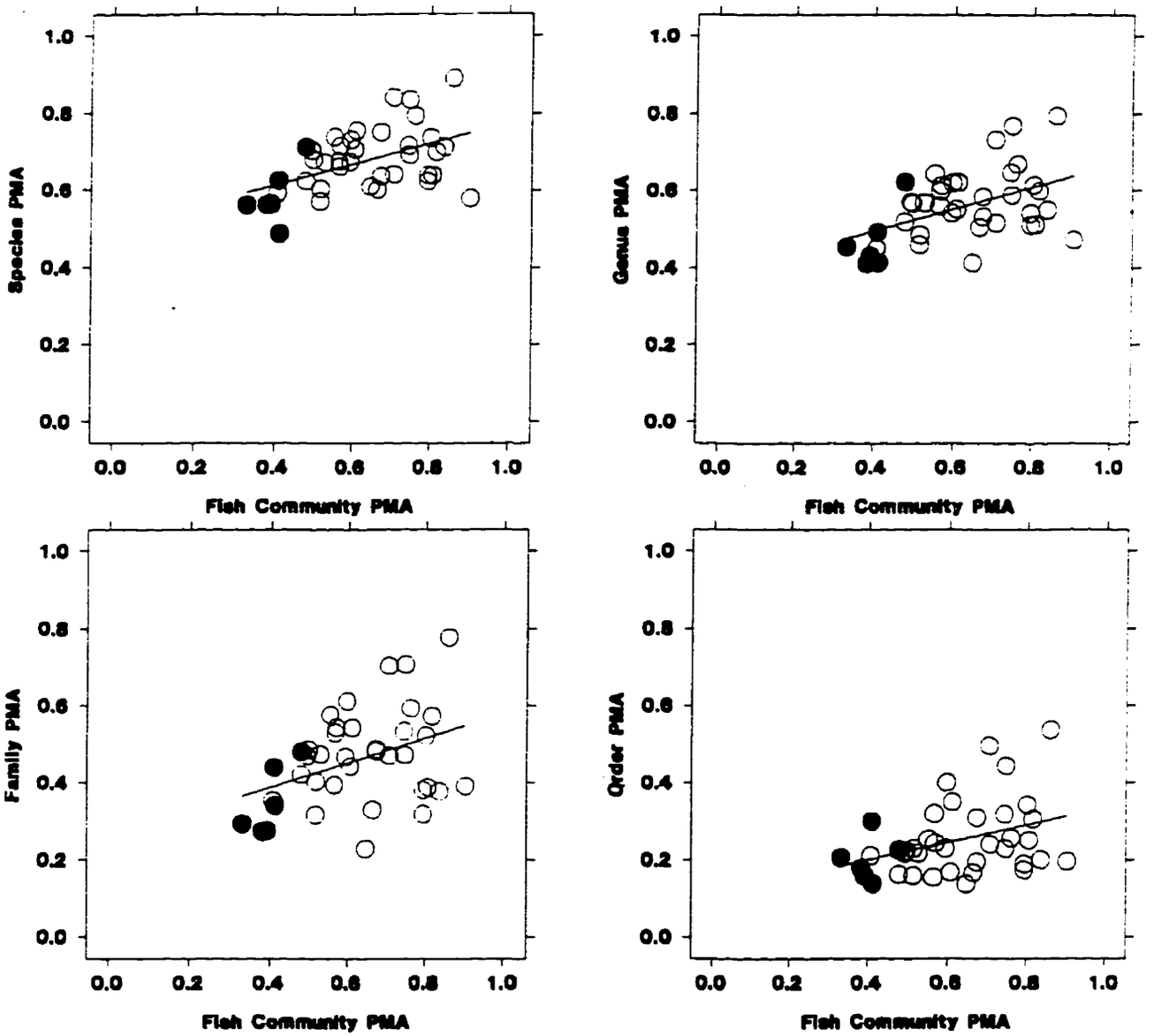


Figure 5.17. Relationship between $BENTHOS_{PMA}$ and $FISH_{PMA}$ for the buffer-strip data using marginal-trout streams as the reference and August benthic surveys. Relationships for benthos identified to species, genus, family and order are given. PMA is based on Bray-Curtis (BC) distances. Regression equations and diagnostics are given in Table 5.3.

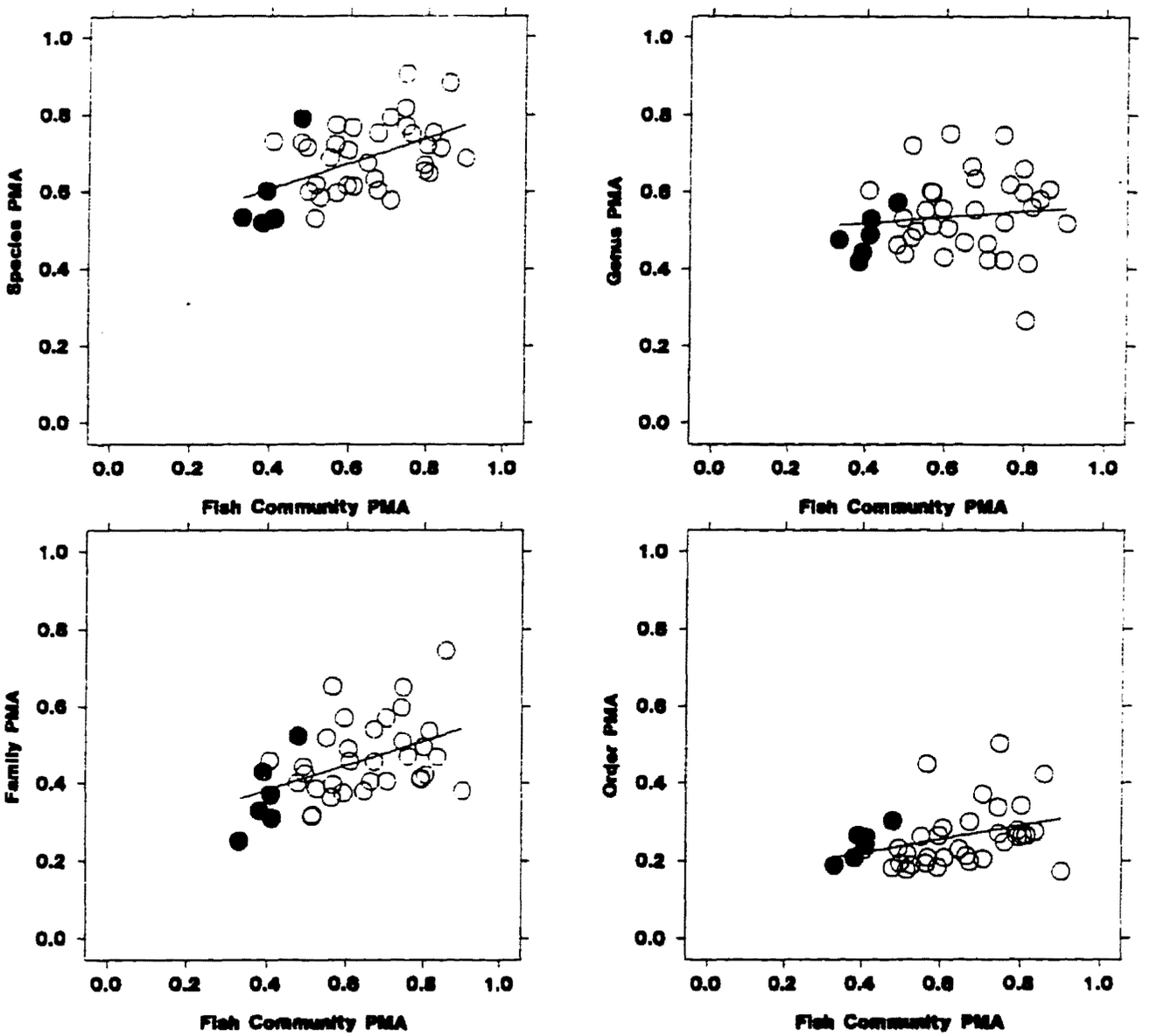


Figure 5.18. Relationship between $BENTHOS_{PMA}$ and $FISH_{PMA}$ for the buffer-strip data using marginal-trout streams as the reference and May benthic surveys. Relationships for benthos identified to species, genus, family and order are given. PMA is based on Bray-Curtis (BC) distances. Regression equations and diagnostics are given in Table 5.3.

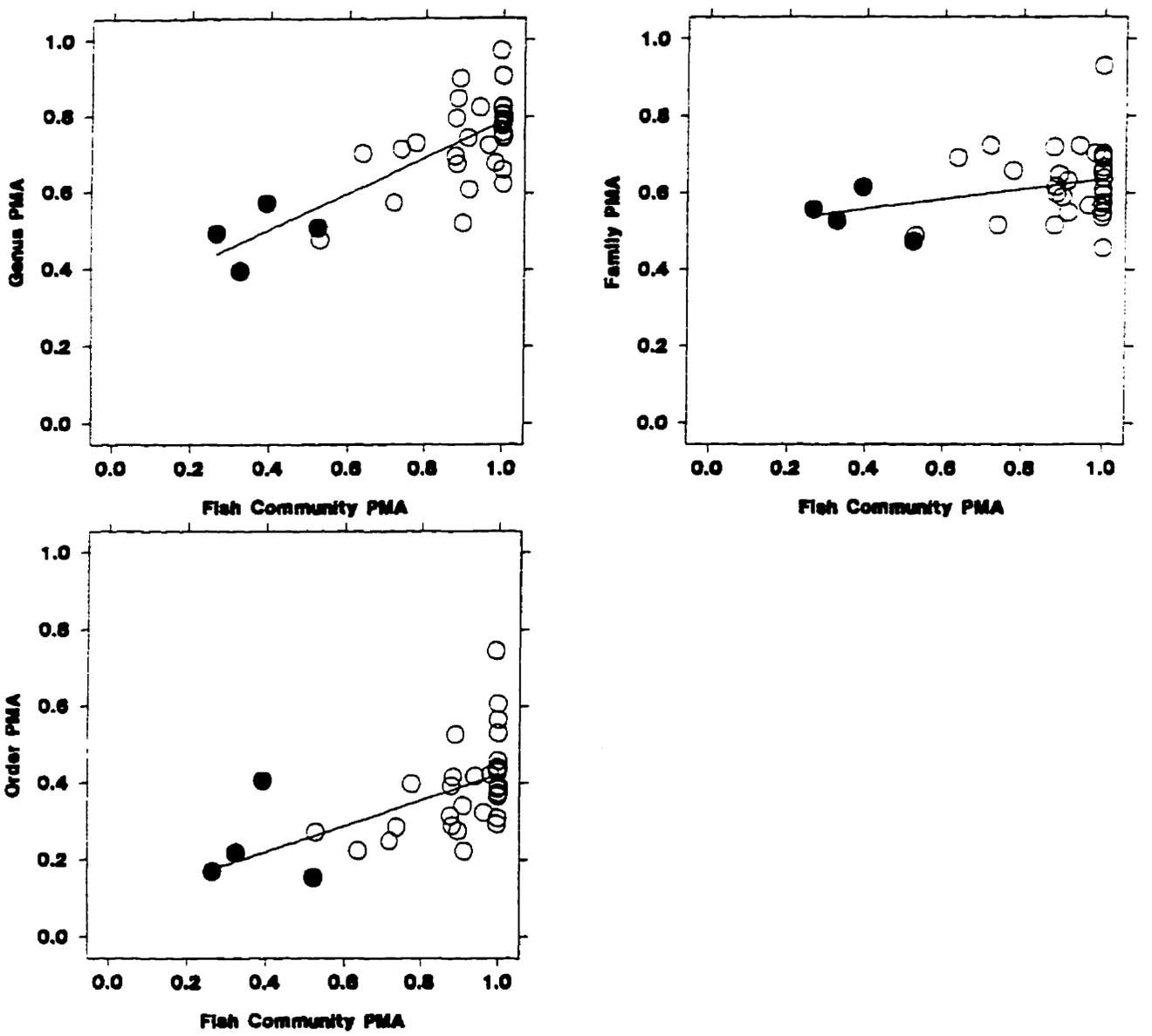


Figure 5.19. Relationship between $BENTHOS_{PMA}$ and $FISH_{PMA}$ for the MNR-HSI data using brook-trout streams as the reference. Relationships for benthos identified to genus, family and order are given. PMA is based on Bray-Curtis (BC) distances. Regression equations and diagnostics are given in Table 5.3.

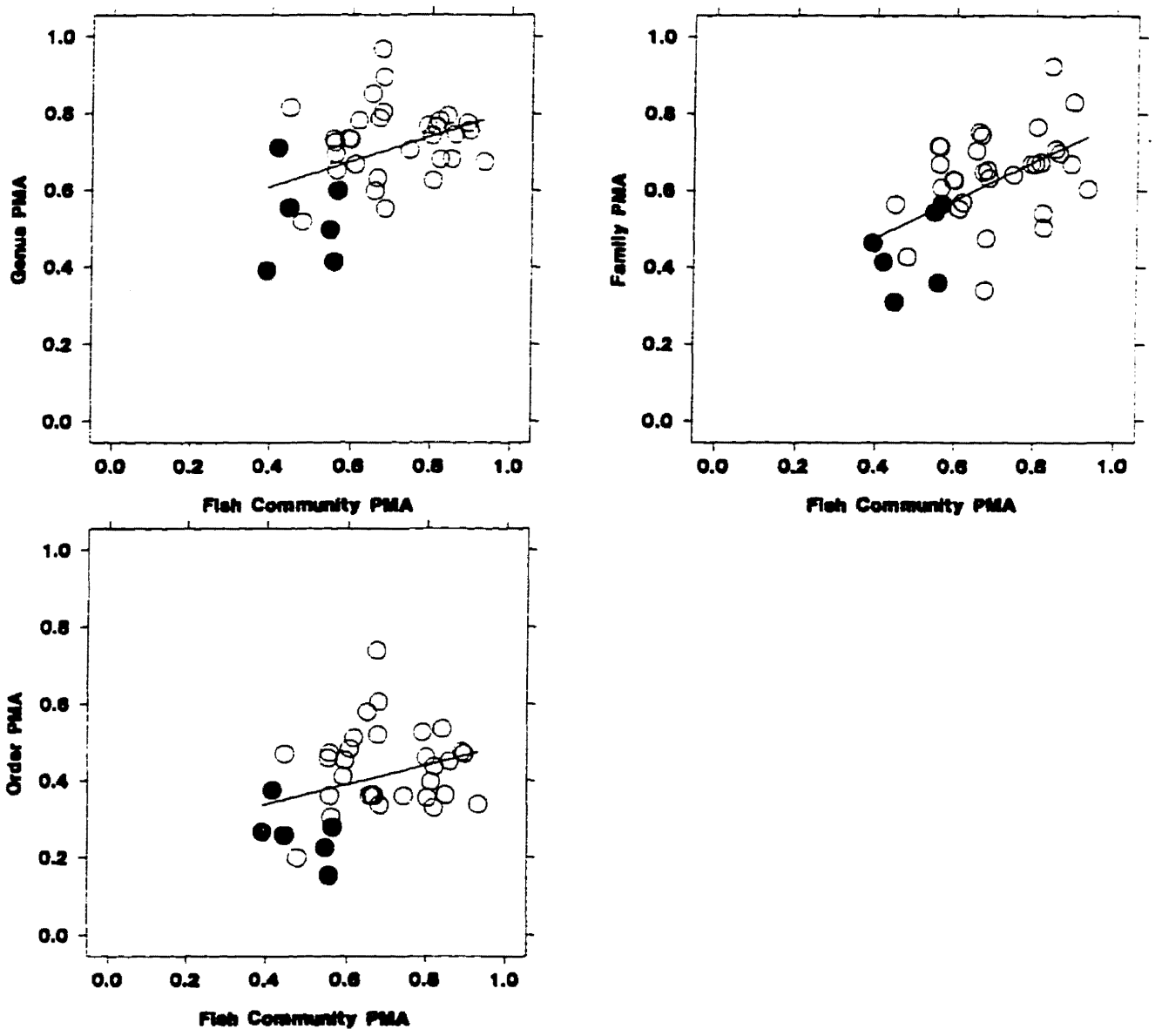


Figure 5.20. Relationship between $BENTHOS_{PMA}$ and $FISH_{PMA}$ for the MNR-HSI data using marginal trout streams as the reference. Relationships for benthos identified to genus, family and order are given. PMA is based on Bray-Curtis (BC) distances. Regression equations and diagnostics are given in Table 5.3.

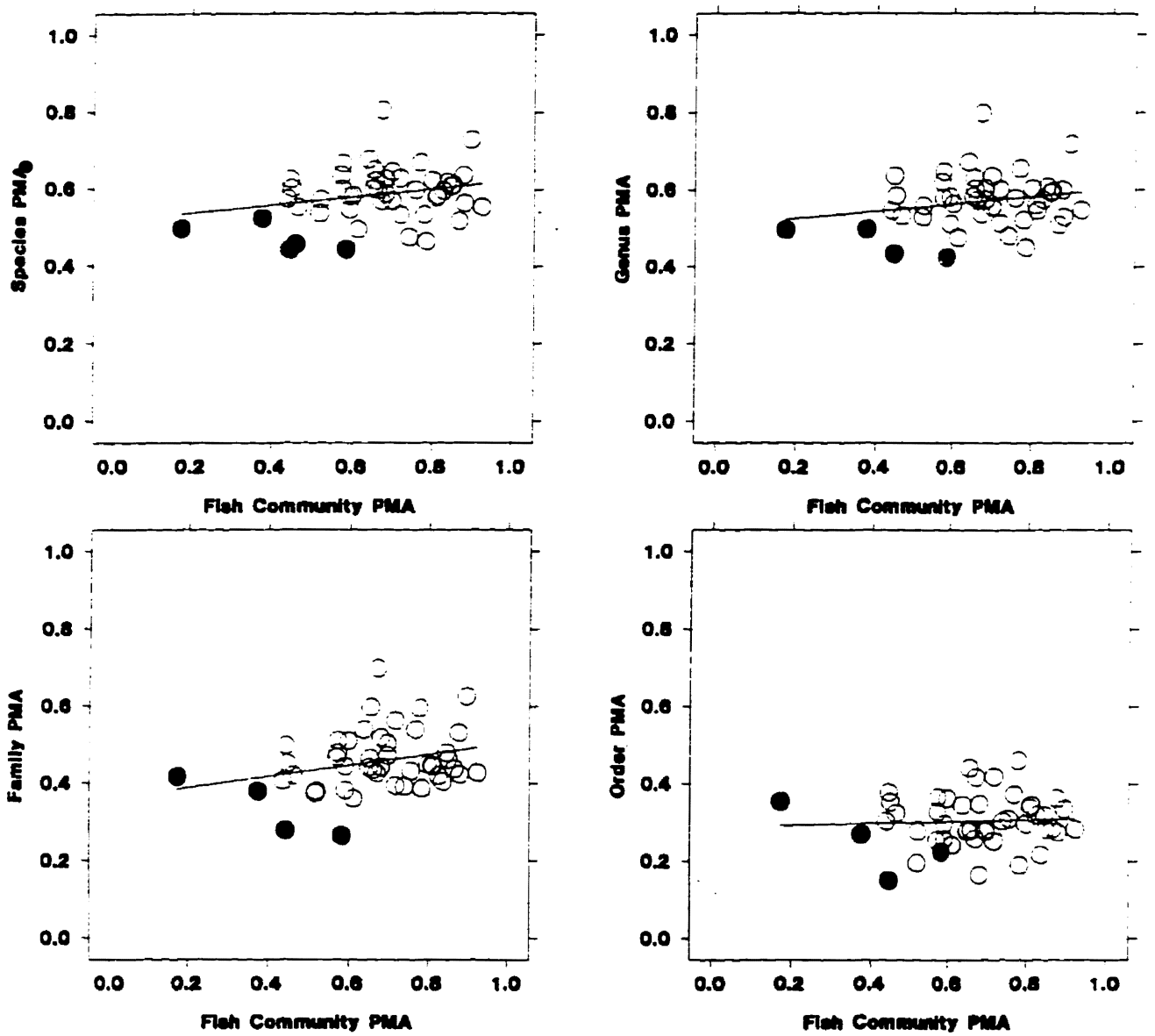


Figure 5.21. Relationship between $BENTHOS_{PMA}$ and $FISH_{PMA}$ for the London subwatershed data using creek chub streams as the reference. Relationships for benthos identified to species, genus, family and order are given. PMA is based on Bray-Curtis (BC) distances. Regression equations and diagnostics are given in Table 5.3.

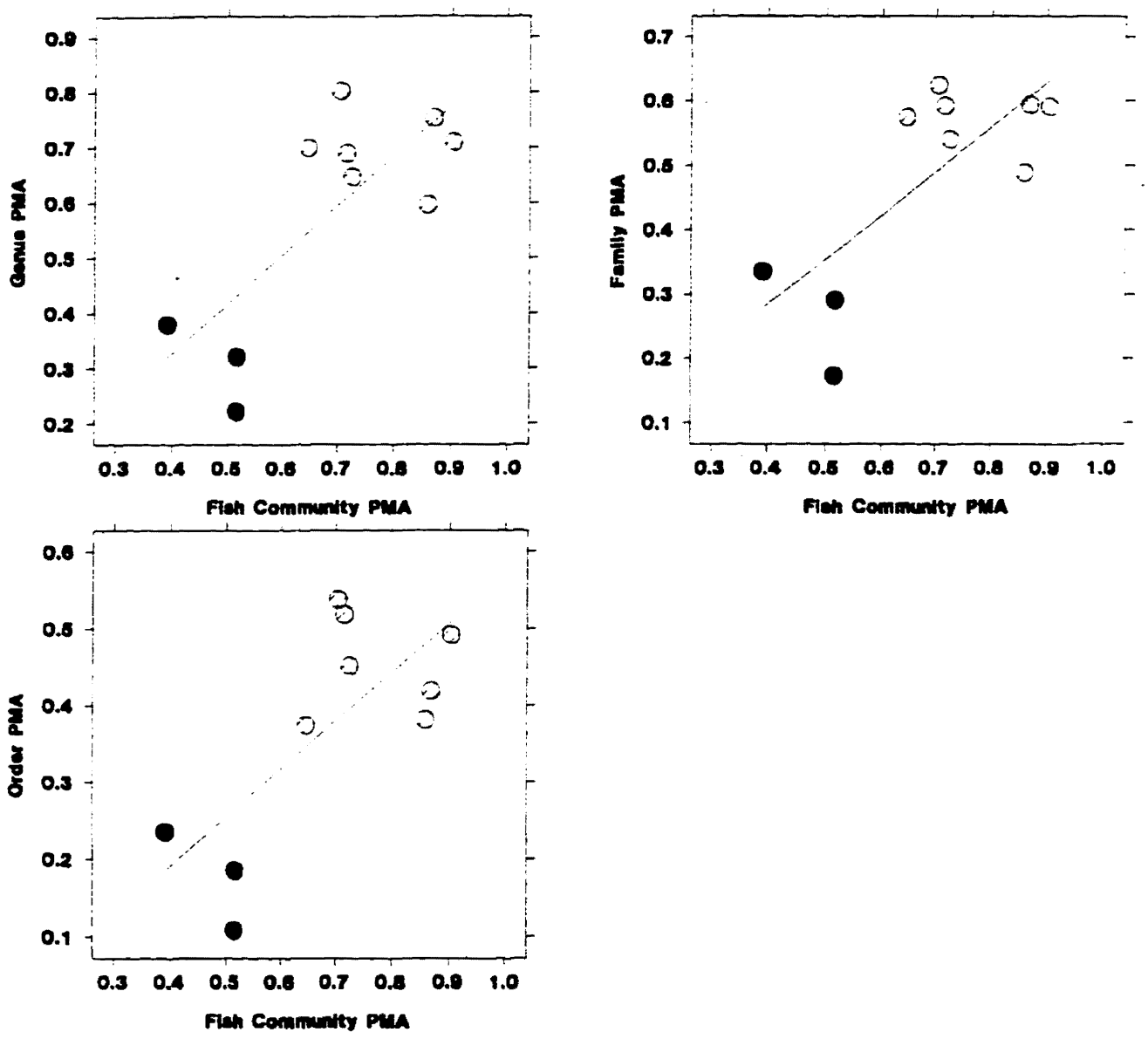


Figure 5.22. Relationship between $BENTHOS_{PMA}$ and $FISH_{PMA}$ for the Laurel Creek data. Relationships for run benthos identified to genus, family and order are given. PMA is based on Bray-Curtis (BC) distances. Regression equations and diagnostics are given in Table 5.3.

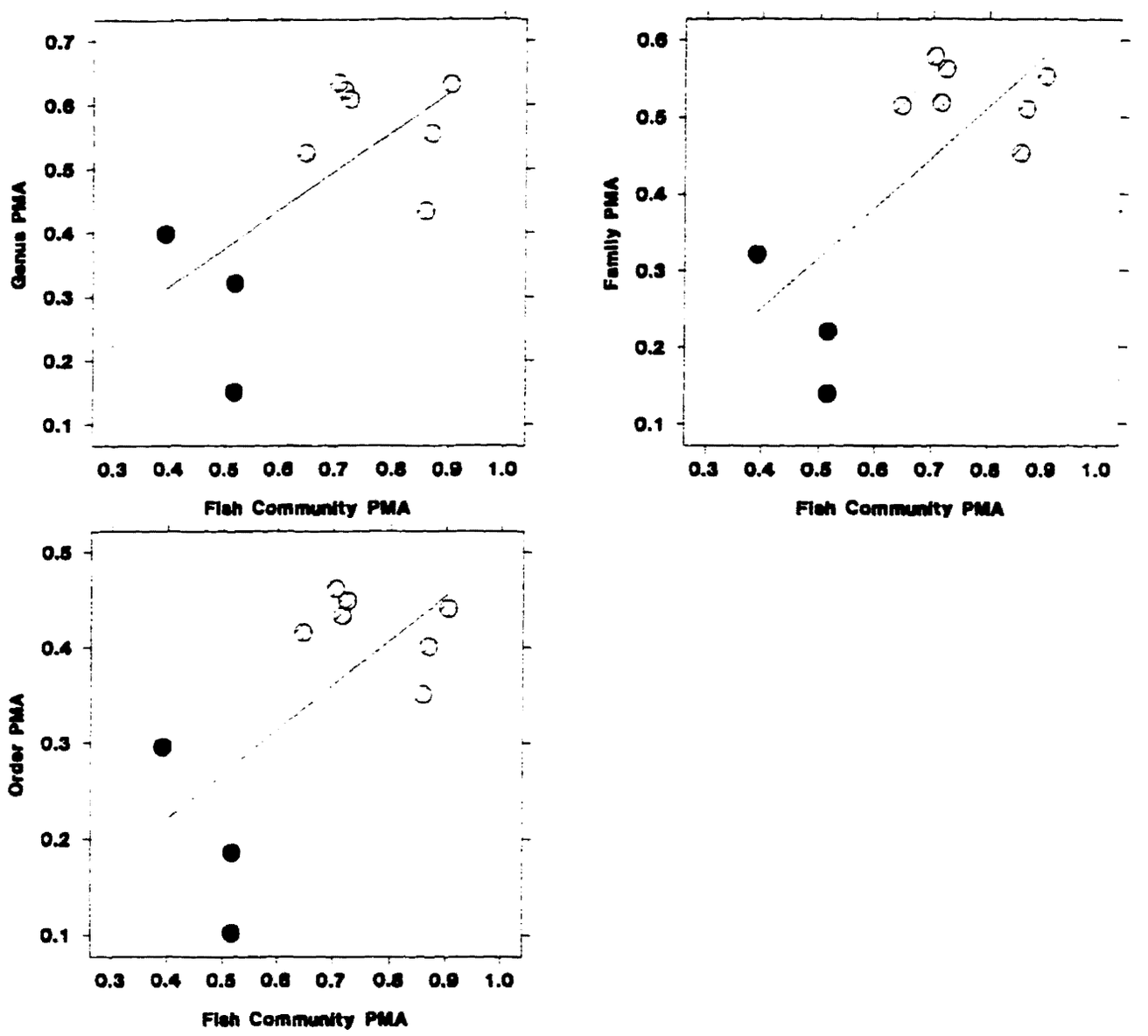


Figure 5.23. Relationship between $BENTHOS_{PMA}$ and $FISH_{PMA}$ for the Laurel Creek data. Relationships for riffle benthos identified to genus, family and order are given. PMA is based on Bray-Curtis (BC) distances. Regression equations and diagnostics are given in Table 5.3.

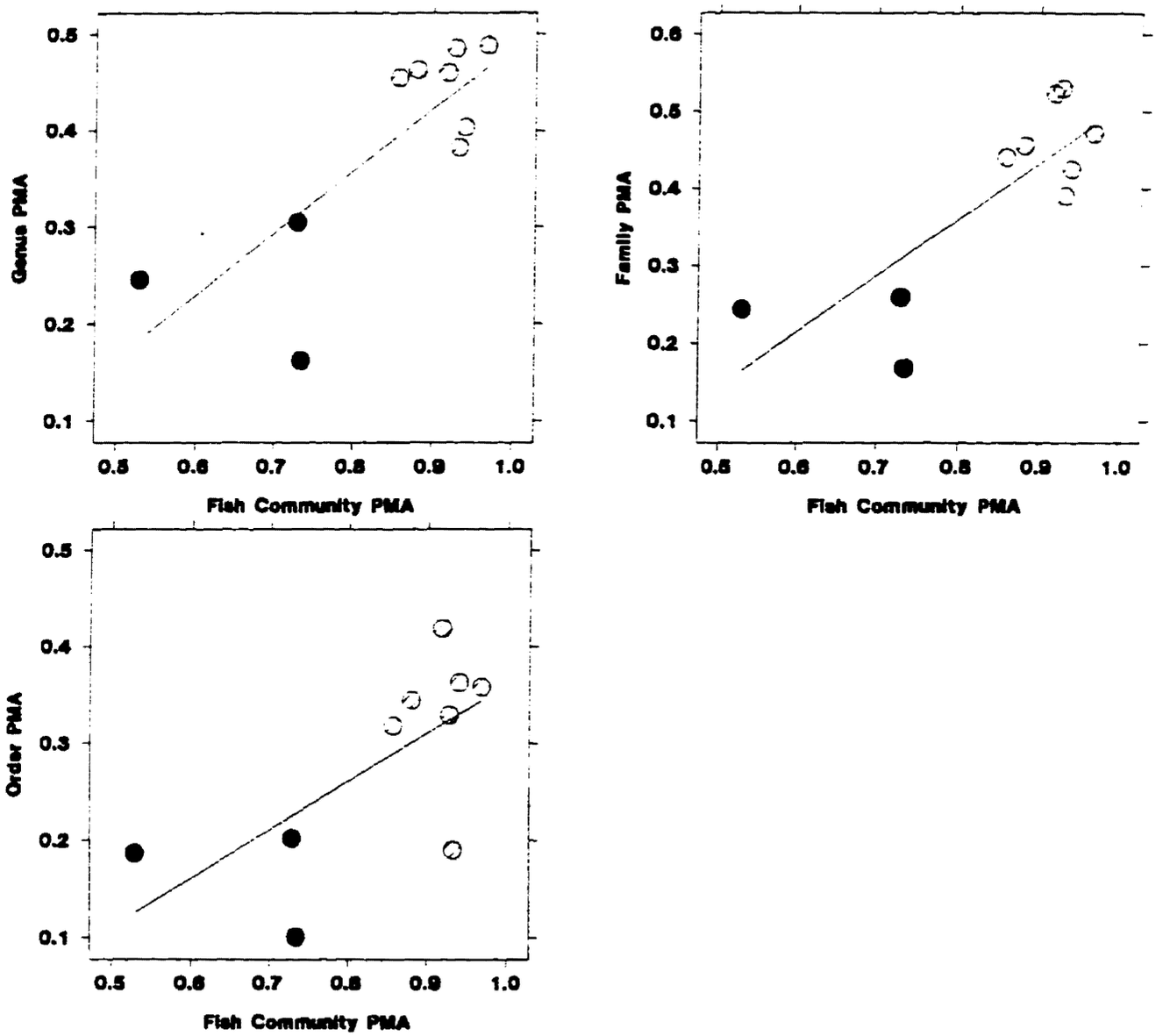


Figure 5.24. Relationship between $BENTHOS_{PMA}$ and $FISH_{PMA}$ for the Canagagigue Creek data. Relationships for run benthos genus, family and order are given. PMA is based on Bray-Curtis (BC) distances. Regression equations and diagnostics are given in Table 5.3.

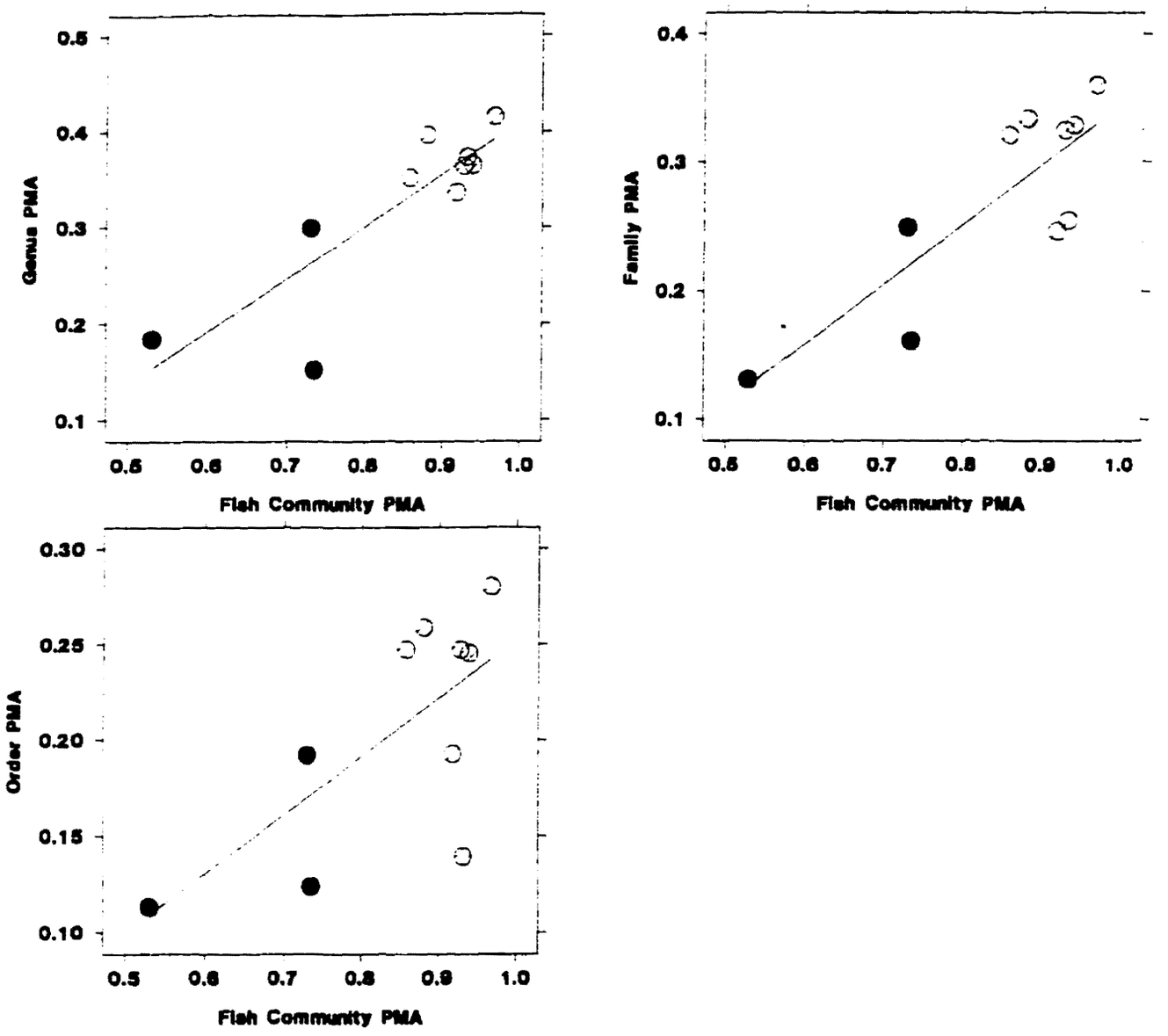


Figure 5.25. Relationship between $BENTHOS_{PMA}$ and $FISH_{PMA}$ for the Canagagigue Creek data. Relationships for riffle benthos identified to genus, family and order are given. PMA is based on Bray-Curtis (BC) distances. Regression equations and diagnostics are given in Table 5.3.

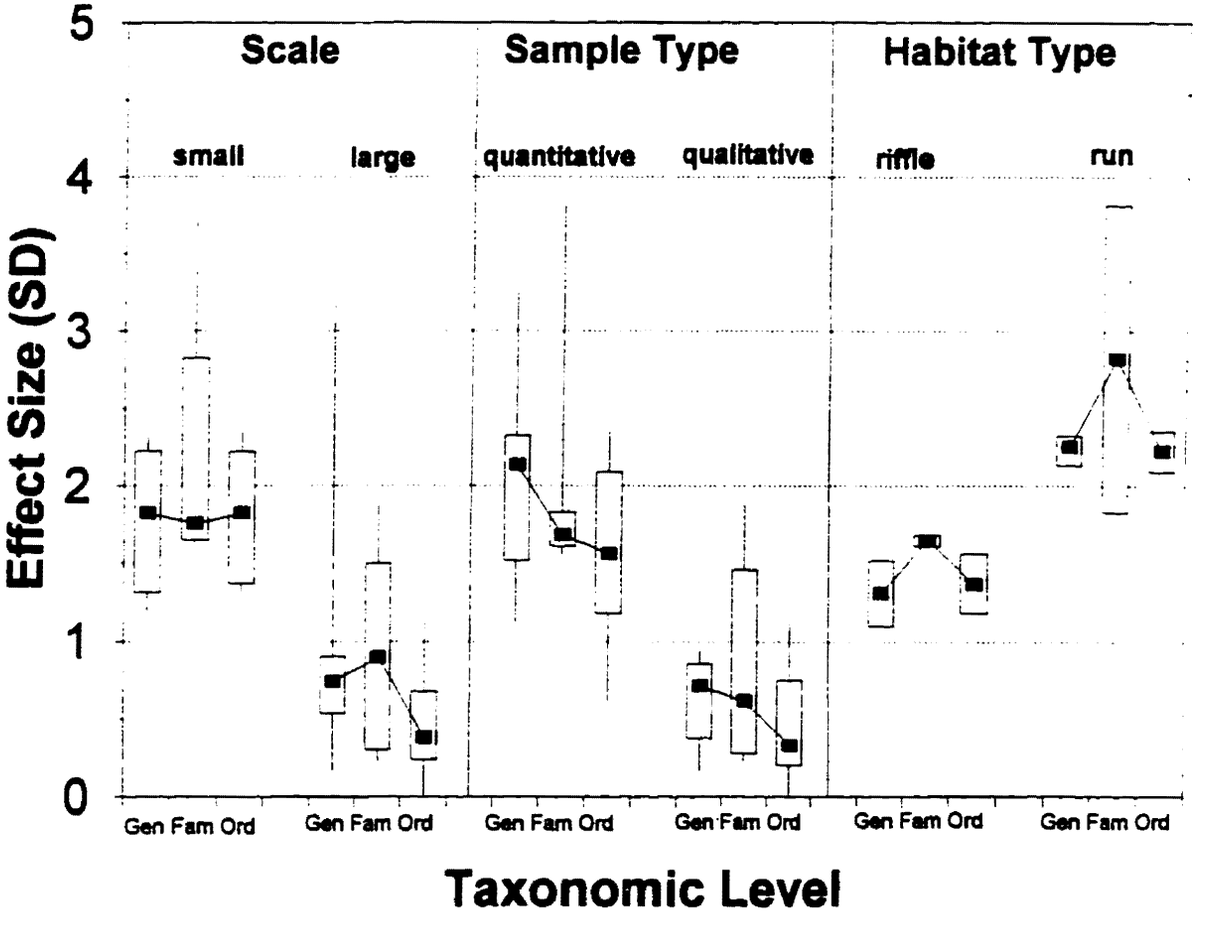


Figure 5.26. Box plots showing the effects of sampling scale, sample type, and habitat type on derived critical benthic effect sizes. The upper and lower limits of the boxes denote 25th and 75th percentiles, minimum and maximum values are denoted by whiskers, and median values are indicated by checkered boxes. Gen=Genus, Fam=Family, Ord=Order-level taxonomy.

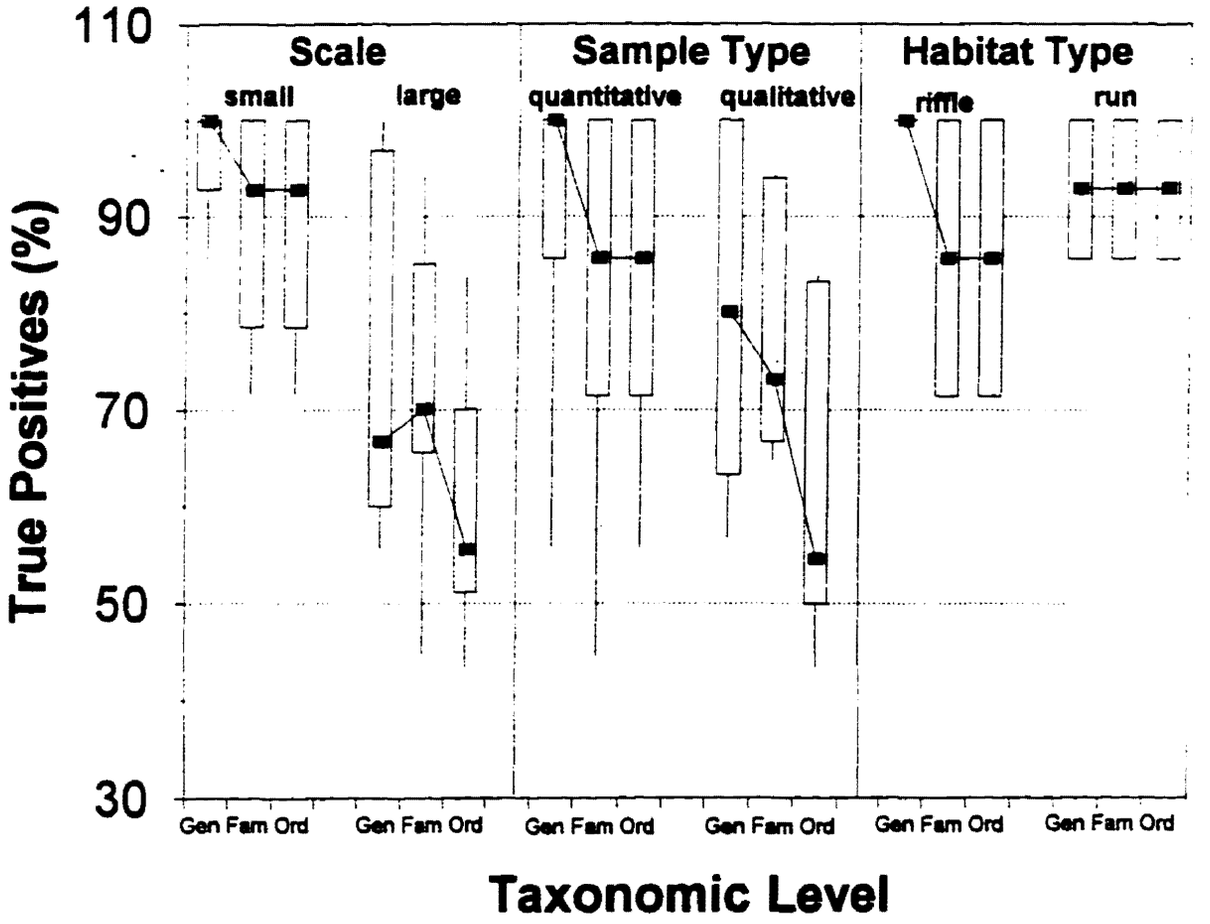


Figure 5.27. Box plots showing the effects of sampling scale, sample type, and habitat type on true-positive probability statements. The upper and lower limits of the boxes denote 25th and 75th percentiles, minimum and maximum values are denoted by whiskers, and median values are indicated by checkered boxes.Gen=Genus, Fam=Family, Ord=Order-level taxonomy.

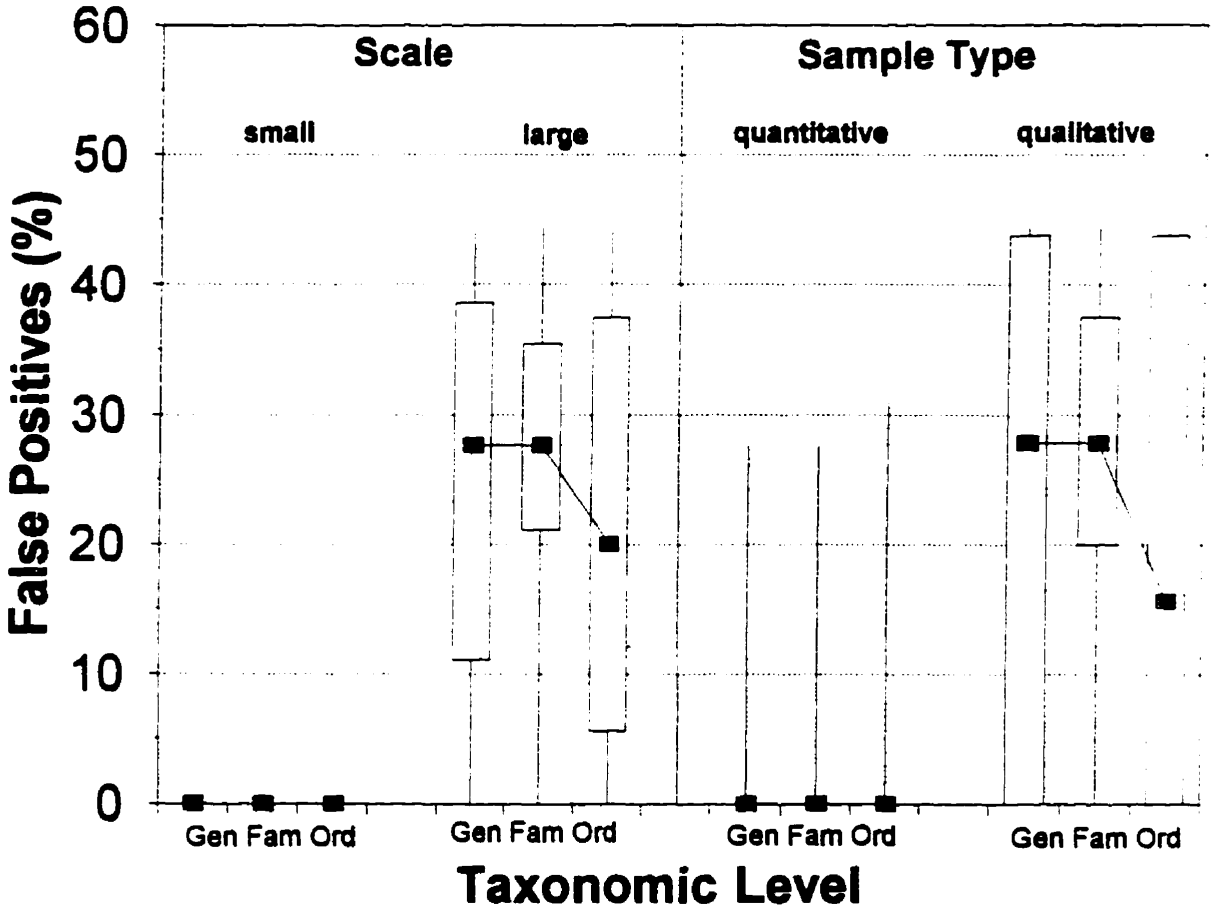


Figure 5.28. Box plots showing the effects of sampling scale, sample type, and habitat type on false-positive probability statements. The upper and lower limits of the boxes denote 25th and 75th percentiles, minimum and maximum values are denoted by whiskers, and median values are indicated by checkered boxes. Gen=Genus, Fam=Family, Ord=Order-level taxonomy.

CHAPTER 6

Epilogue

This thesis determined the types of effects in benthic community composition that can be considered to have ecological consequence in wadeable streams in southern Ontario. The approach taken was based on the philosophy that changes in benthos are of little significance unless they coincide with unacceptable changes in fishery resources. Benthos have historically been used as environmental monitors of the condition (health?) of aquatic systems, often with the justification that benthos could forecast impending effects on ecosystem endpoints considered more important. The derivation of critical benthic community effect sizes required four things. First, a definition of unacceptable change in a fishery resource. Second, selection of some aspect of fishery resources that could be considered important to protect. Third, a demonstration that benthic community composition was inherently related to relevant descriptors of the condition of fisheries. And finally, specific relationships between benthic community composition and the fishery descriptor, from which critical benthic effect sizes could be determined.

The second chapter is a discussion of the potential use of normal ranges for setting generic ecological criteria. The use of normal ranges is inherent in most goal-setting forums, however, appropriate statistical tests for comparing "impact" locations against the normal range have not been put forward. The second chapter, therefore, provides an operational definition for the normal range and demonstrates appropriate statistical tests for determining when impact locations truly fall outside of the range. For sites that are truly outside of the normal range of variation for reference locations, one-sample contrasts with equivalence tests will lead to erroneous conclusions of no impact at most 5% of the time. Two-sample contrasts will lead to erroneous conclusions of no impact about 50% of the time with low sample sizes (i.e., 10-20 reference locations). In contrast to both the two-sample contrast and the equivalence test, interval tests fail to recognize sites as being impacted unless impacts are in excess of about 3 σ from the reference population average (with a reference sample size of 20). Finally, the penalty for using the equivalence test is that it will fail up to 26% of sites that are truly members of the reference

population. Practitioners contrasting non-random impacted locations against a set of reference locations should consider these characteristics of the various potential tests when deriving conclusions of impact.

The third chapter examined the sensitivity of various descriptors of fish community composition. Data from 37 streams, variously affected by agriculture, urbanization and impoundments, were used to determine the effect of single or three pass electrofishing, characterizations based on presence/absence, abundance or biomass, or surveys conducted in the spring or fall, on statistical power. In general, single-pass estimates of biomass of all species, and the use of a multivariate approach to describing the community, was more sensitive (would provide more statistical power) and would be more useful for modelling. In contrast, measuring only the abundances or biomasses of individual (preferred) species like brook trout, was less sensitive (would provide less statistical power) for characterizing the fishery of a stream, and would be less useful for modelling purposes.

Based on these findings, Chapter 4 was used to examine the association between fish and benthos in southern Ontario streams. Using multivariate descriptors of fish community composition (with species quantified using biomasses), I demonstrated that there were consistent and strong associations between stream fish and benthos. This chapter also suggested that more detailed benthic taxonomy resulted in stronger associations. This provided evidence that the fish-benthos association is strongly driven by coincident association of fish and benthos with environmental conditions. This finding somewhat contradicts the previous assumption that fish and benthos were associated because of strong top-down predatory influences on benthos. There are, undoubtedly, top-down effects of fish on invertebrate community structure, as many have demonstrated. However, along long environmental gradients, the major factor influencing benthic community composition appears to be environmental conditions.

Finally, Chapter 5 specifically examined the reliability of benthic surveys for predicting the degree of impact in fish communities. The chapter also estimated and examined sources of variation in derived critical benthic effect sizes. Based on the discussion in Chapter 2, ecologically relevant impacts on fish were defined as those effects that exceed the normal range of variation (or the 95% region) of reference observations. In general, critical benthic effect sizes that coincided with ecologically relevant effects on fish were less than 2 standard deviations, but

varied with factors such as the type of benthic sampling apparatus, the microhabitat benthos were collected from, and the nature of the study design (i.e., whether it incorporated regional-reference or site-specific reference locations). Although the fish-benthos correlations were significant regardless of the level of benthic identification, identifying benthos to species resulted in more accurate prediction of the condition of fish communities. In addition, the reliability of benthos was higher when the reference fish community was a brook trout community, or when the assessment was site specific. In both situations, changes in fish communities from reference to impacted conditions were abrupt.

The results from this thesis have obvious application to aquatic environmental assessments, particularly stream assessments in southern Ontario. Most of the streams in this study were smaller wadeable streams that are tributary to larger systems such as the Grand and Thames Rivers that support game fisheries (i.e., brown trout, walleye). These smaller systems are of some interest to the Ontario Ministry of Transportation, Ontario Ministry of Natural Resources, and several municipalities (Metro Toronto and Region Conservation Authority, Credit Valley Conservation Authority). All of these groups hope to demonstrate that such small streams, within their jurisdictions, are in relatively good condition (i.e., support, or could support, a relatively unaltered fish fauna). Since the relationships between fish and benthos (and associated benthic decision criteria) varied with benthic sampling approaches (i.e., whether benthos were sampled from riffles, runs or a combination) and study design (i.e., whether regional reference or site-specific reference locations were used), the use of benthos as a predictor of the condition of fish communities would require calibration for each application (i.e., new locations). Alternatively, standardization of data collection protocols would make the derivation of critical benthic effect sizes a more feasible possibility.

Benthic criteria, such as those derived in Chapter 5, should be applied in association with relevant statistical procedures, primarily non-central interval or equivalence tests. These tests will provide exact probabilities that benthic community composition in impacted locations exceeds the benthic critical value. Exceedence of these critical values would imply that fish communities at those impacted locations either are currently significantly impacted, or will be in the future. In contrast, non-detection of a significant impact in benthos would imply either that the fish community is not currently significantly impacted, or will at some point recover. Measures of

reliability are further required to provide some estimate of the likelihood that the inference based on benthic community composition is incorrect for that location and time.

Throughout this thesis, it has been suggested that macrobenthos are merely a surrogate measurement for the condition of fish community composition. The use of benthos as monitors has advantages over fish community work in that it is less damaging to the fishery resource itself, and benthos may provide early warning of impending impact on (or recovery of) a fish community. There are however, several other ways to monitor aquatic ecological condition, including surveys of adult fish populations, algae, plankton, toxicity, and physical and chemical properties of the environment. Understanding how each of these are related to our ultimate ecological goals (i.e., compliance endpoints) is essential if we are to conduct environmental assessments in a clear and logical fashion.

Appendix A

Stream Locations

The following table lists each of the streams included in the various chapters of the thesis. Study numbers are (1) = Farrara and Reid (1995); (2) = Barton et al. (1985); (3) = the study I conducted with the Ontario Ministry of Natural Resources in 1995; (4) = Bowlby and Roff (1986); and (5) = the study I conducted in 1994. Geographic locations for stations are given either in degrees, minutes and seconds latitude/longitude, or in Universal Transverse Mercator Grid (UTMG) coordinates (meters northing and easting). In some cases, both are given. UTMGs were taken from topographic maps at 1:50,000. Stream codes were taken from the original publications when available, or created for the purposes of this study.

Site No.	Data base	Stream Name	Code	Study	Latitude (°)	Longitude (°)	Northing (m)	Easting (m)
1	Beak, 1994	Crumlin	c2	1	425919	810757		
2	Beak, 1994	Crumlin	c6	1	425958	810852		
3	Beak, 1994	Crumlin	c7	1	430006	810824		
4	Beak, 1994	Crumlin	c9	1	430045	810938		
5	Beak, 1994	Dodds	d1	1	424705	811247		
6	Beak, 1994	Dodds	d2	1	424743	811708		
7	Beak, 1994	Dodds	d4	1	424856	811658		
8	Beak, 1994	Dodds	d5	1	424931	811916		
9	Beak, 1994	Dodds	d6	1	424923	811832		
10	Beak, 1994	Dodds	d7	1	425043	811474		
11	Beak, 1994	Kettle	k1	1	424933	810854		
12	Beak, 1994	Kettle	k3	1	425051	810805		
13	Beak, 1994	Kettle	k4	1	425132	810847		
14	Beak, 1994	Kettle	k5	1	425134	810626		
15	Beak, 1994	Medway	m1	1	430056	811640		
16	Beak, 1994	Medway	m2	1	430044	811823		
17	Beak, 1994	Medway	m3	1	430114	811814		
18	Beak, 1994	Medway	m4	1	430151	811850		
19	Beak, 1994	Medway	m5	1	430311	811746		
20	Beak, 1994	Medway	m6	1	430425	811628		

Site No.	Data base	Stream Name	Code	Study	Latitude (°)	Longitude (°)	Northing (m)	Easting (m)
21	Beak, 1994	Medway	m7	1	430437	811751		
22	Beak, 1994	Medway	m10	1	430622	811538		
23	Beak, 1994	Medway	m12	1	430633	811712		
24	Beak, 1994	Medway	m14	1	430717	811432		
25	Beak, 1994	Medway	m15	1	430722	811926		
26	Beak, 1994	Medway	m17	1	430808	811622		
27	Beak, 1994	Medway	m20	1	430933	812044		
28	Beak, 1994	Medway	m21	1	431032	811829		
29	Beak, 1994	Medway	m22	1	431038	812042		
30	Beak, 1994	Medway	m23	1	431150	811917		
31	Beak, 1994	Mud	md1	1	425843	811720		
32	Beak, 1994	Mud	md2	1	425857	811753		
33	Beak, 1994	Pottersburg	p1	1	425828	811031		
34	Beak, 1994	Pottersburg	p3	1	430015	810938		
35	Beak, 1994	Pottersburg	p4	1	430027	811035		
36	Beak, 1994	Pottersburg	p5	1	430101	810941		
37	Beak, 1994	Pottersburg	p6	1	430132	810929		
38	Beak, 1994	Pottersburg	p8	1	430238	810731		
39	Beak, 1994	Pottersburg	p10	1	430407	810718		
40	Beak, 1994	Stanton	st1	1	425856	812030		
41	Beak, 1994	Stanton	st2	1	425858	812045		
42	Beak, 1994	Sharon	s1	1	425226	812358		
43	Beak, 1994	Sharon	s2	1	425302	812406		
44	Beak, 1994	Sharon	s4	1	425417	812229		
45	Beak, 1994	Stoney	sy1	1	430120	811509		
46	Beak, 1994	Stoney	sy2	1	430208	811513		
47	Beak, 1994	Stoney	sy3	1	430222	811526		
48	Beak, 1994	Stoney	sy6	1	430316	811404		
49	Barton (1980)	Amaranth	Amar	2	435100	801600	4860600	558800
50	Barton (1980)	Bailey	Bail	2	440600	795000	4884400	596800

Site No.	Data base	Stream Name	Code	Study	Latitude (°)	Longitude (°)	Northing (m)	Easting (m)
51	Barton (1980)	Barlow	Barl	2	431600	800900	4791300	568400
52	Barton (1980)	Beeton	Beet	2	440400	794800	4879200	597000
53	Barton (1980)	Bethel	Beth	2	435600	804400	4864300	521900
54	Barton (1980)	Black	Blak	2	431700	803400	4792800	535800
55	Barton (1980)	Boomer	Boom	2	433200	804100	4821300	525900
56	Barton (1980)	Boyne	Boyn	2	440600	800800	4883300	569000
57	Barton (1980)	Bronte	Bron	2	432300	800100	4804200	579600
58	Barton (1980)	Canagagigue	Cana	2	433900	803400	4829100	535500
59	Barton (1980)	Carroll	Carr u/s	2	434000	803000	4832600	541600
60	Barton (1980)	Carroll	Carr d/s	2	434100	803100	4833400	540500
61	Barton (1980)	Cedar	Cedr	2	431800	802600	4794800	545900
62	Barton (1980)	Cox	Cox	2	433600	802400	4825700	544700
63	Barton (1980)	E. Credit	E.Crdt	2	434700	800300	4847300	576800
64	Barton (1980)	W. Credit	W.Crdt	2	434700	800300	4854500	578700
65	Barton (1980)	Ellis	Ells	2	434700	802100	4812400	553400
66	Barton (1980)	Fairchild	Fair	2	431700	801300	4792400	566300
67	Barton (1980)	Four-mile	4mile	2	435200	803500	4857400	532600
68	Barton (1980)	Galt	Galt	2	432500	801400	4811200	566900
69	Barton (1980)	Grand	Grand	2	440800	802200	4887600	551300
70	Barton (1980)	Hopewell	Hope	2	433000	802400	4815700	542700
71	Barton (1980)	Humber	Humb	2	435800	795200	4868500	591900
72	Barton (1980)	Humber trib	Hmbtrb	2	435600	795000	4865500	593100
73	Barton (1980)	Hunsberger	Huns	2	432100	803900	4800500	528500
74	Barton (1980)	Irvine	Irvin	2	434800	802200	4849300	550600
75	Barton (1980)	Lutteral	Lutt	2	434000	801500	4833000	559500
76	Barton (1980)	N. Maitland	N.Mtlnd	2	435500	805000	4962700	513100
77	Barton (1980)	S. Maitland	S.Mtlnd	2	435400	805100	4860500	511600
78	Barton (1980)	Mallet	Mall	2	435000	804300	4854100	523300
79	Barton (1980)	Nottawasaga	Nott	2	435700	800500	4866800	573200
80	Barton (1980)	Pine	Pine	2	440900	801100	4889600	565400

Site No.	Data base	Stream Name	Code	Study	Latitude (°)	Longitude (°)	Northing (m)	Easting (m)
81	Barton (1980)	Saugeen	Saug	2	440600	802900	4883800	546800
82	Barton (1980)	Sheldon	Sheld	2	440400	800300	4880500	576700
83	Barton (1980)	E. Speed	E.Spd	2	434200	801600	4839500	558800
84	Barton (1980)	W. Speed	W.Spd	2	434100	801700	4837200	557200
85	Barton (1980)	Swan	Swn u/s	2	433900	802500	4832000	545600
86	Barton (1980)	Swan	Swn d/s	2	434000	802200	4833700	548300
87	Barton (1980)	Washington	Wash	2	431800	803400	4796400	538900
88	Barton (1980)	Willow Brook	Will	2	435600	801600	4865800	459400
89	MNR, 1995	Rest Acres	AMCA	3			4775344	550774
90	MNR, 1995	Cedar	ANSD	3			4792447	544387
91	MNR, 1995	Laurel	BPC	3			4814266	540757
92	MNR, 1995	Duffins	DUF01	3			4868665	653959
93	MNR, 1995	Duffins	DUF02	3			4868023	653960
94	MNR, 1995	Duffins	DUF03	3			4868036	645830
95	MNR, 1995	Duffins	DUF04	3			4867263	643458
96	MNR, 1995	Duffins	DUF05	3			4867304	643624
97	MNR, 1995	Duffins	DUF06	3			4864574	646946
98	MNR, 1995	Duffins	DUF07	3			4863158	655062
99	MNR, 1995	Duffins	DUF08	3			4858076	654633
100	MNR, 1995	Mill	FBBU	3			4802166	556969
101	MNR, 1995	Blair	FBU	3			4803583	549411
102	MNR, 1995	Moffat	FRD	3			4799778	557509
103	MNR, 1995	Hanlon	GAPT	3			4817118	561036
104	MNR, 1995	Laurel	GHPU	3			4814401	541865
105	MNR, 1995	Marden	GTPF	3			4825267	555678
106	MNR, 1995	Laurel	HPRF	3			4814606	539353
107	MNR, 1995	Hanlon	HRU	3			4816898	562333
108	MNR, 1995	Swan	HW6U	3			4834754	553416
109	MNR, 1995	Richardson	HW7D	3			4829009	569532
110	MNR, 1995	Schneiders	MDBD	3			4806691	544662

Site No.	Data base	Stream Name	Code	Study	Latitude (°)	Longitude (°)	Northing (m)	Easting (m)
111	MNR, 1995	Ellis	MGBD	3			4809095	553984
112	MNR, 1995	Mt. Pleasant	MPBD	3			4768795	556504
113	MNR, 1995	Devils	MRGU	3			4802401	553926
114	MNR, 1995	Ganaraska	ORM01	3			4882409	705912
115	MNR, 1995	Orono	ORM02	3			4875404	690542
116	MNR, 1995	Soper's Creek	ORM03	3			4874375	686109
117	MNR, 1995	Harris	ORM04	3			4878380	692321
118	MNR, 1995	Port Britain	ORM05	3			4873299	708973
119	MNR, 1995	Graham	ORM06	3			4870231	697049
120	MNR, 1995	Colonel Giles	ORM08	3			4875740	713652
121	MNR, 1995	Patterson	PWRD	3			4796343	555424
122	MNR, 1995	Mill	SMCA	3			4803558	558475
123	MNR, 1995	Mill	SPAU	3			4801472	556434
124	Bowlby, 1982	Blue Springs	BS0	4			4828200	577100
125	Bowlby, 1982	Blue Springs	BS1	4			4827600	573400
126	Bowlby, 1982	Black Creek	B1	4			4830900	579700
127	Bowlby, 1982	Black Creek	B2	4			4831500	580500
128	Bowlby, 1982	Hanlon	H1	4			4816600	560400
129	Bowlby, 1982	Hanlon	H3	4			4817000	561700
130	Bowlby, 1982	Lutteral	L1	4			4841100	562800
131	Bowlby, 1982	Mill	G0	4			4813900	568600
132	Bowlby, 1982	Galt	G1	4			4811200	566900
133	Kilgour, 1994	Canagagigue	c1	5			4832800	534500
134	Kilgour, 1994	Canagagigue	c2	5			4831700	534000
135	Kilgour, 1994	Canagagigue	c2.5	5			4829100	535500
136	Kilgour, 1994	Canagagigue	c3	5			4828200	535900
137	Kilgour, 1994	Canagagigue	c4	5			4827900	535400
138	Kilgour, 1994	Canagagigue	c5	5			4827400	536300
139	Kilgour, 1994	Canagagigue	c6	5			4826600	536400

Site No.	Data base	Stream Name	Code	Study	Latitude (°)	Longitude (°)	Northing (m)	Easting (m)
140	Kilgour, 1994	Canagagigue	c7	5			4825600	537600
141	Kilgour, 1994	Canagagigue	c8	5			4825100	539600
142	Kilgour, 1994	Kintore	k1	5			4779400	498500
143	Kilgour, 1994	Kintore	k2	5			4778800	498300
144	Kilgour, 1994	Kintore	k3	5			4778600	498200
145	Kilgour, 1994	Kintore	k4	5			4782700	495100
146	Kilgour, 1994	Waubuno	K5	5			4782700	495100
147	Kilgour, 1994	Gregory	k5	5			4784200	491500
148	Kilgour, 1994	Phelam	k7	5			4787400	416300
149	Kilgour, 1994	Trout Ck trib	k8	5			4788200	403300
150	Kilgour, 1994	Laurel	l1 u/s	5			4814000	532100
151	Kilgour, 1994	Laurel	l1 d/s	5			4814300	532000
152	Kilgour, 1994	Laurel	l1.5	5			4814100	533000
153	Kilgour, 1994	Laurel	l2	5			4814400	535200
154	Kilgour, 1994	Laurel	l3	5			4812500	535900
155	Kilgour, 1994	Laurel	l4	5			4812600	537500
156	Kilgour, 1994	Laurel	l5	5			4812400	538800
157	Kilgour, 1994	Laurel	l6	5			4812800	539200
158	Kilgour, 1994	Laurel	l7	5			4814600	539300
1659	Kilgour, 1994	Laurel	l8	5			4814700	540000
160	Kilgour, 1994	Schneiders	s1	5			4806300	544800
161	Kilgour, 1994	Schneiders	s2	5			4805600	545700

Appendix B

Fish Species Abbreviations

The table below lists the fish species that were members of the data sets for this study. MNR species codes are given, as are latin names, common names and abbreviations used in the figures. For some species, common names were short enough not to require abbreviation.

MNR Species Code	Latin Name	Common Name	Abbreviation
12	Ichthyomyzon fossor	northern brook lamprey	lamprey
76	Oncorynchus mykiss	rainbow trout	rbw trout
78	Salmo trutta	brown trout	brown trout
80	Salvelinus fontinalis	brook trout	brook trout
131	Esox lucius	northern pike	pike
141	Umbra limi	central mudminnow	mudminnow
163	Catostomus commersoni	common white sucker	white sucker
165	Hypentelium nigricans	northern hog sucker	hog sucker
170	Moxostoma erythrurum	golden redhorse sucker	redhrs scker
181	Carassius auratus	goldfish	goldfish
182	Phoxinus eos	northern redbelly dace	redbelly
183	Phoxinus neogaeus	finescale dace	finescale
184	Clinosotomus elongatus	redside dace	redside
186	Cyprinus carpio	common carp	carp
189	Hybognathus hankinsoni	brassy minnow	brassy
192	Nocomis buguttatus	hornyhead chub	hornyhead
194	Notemigonus crysoleucas	golden shiner	gld shiner
196	Notropis atherinoides	emerald shiner	emerald
198	Notropis cornutus	common shiner	cmn shiner
200	Notropis heterolepis	blacknose shiner	

MNR Species Code	Latin Name	Common Name	Abbreviation
201	Notropis hudsonius	spottail shiner	spottail
202	Notropis rubellus	rosyface shiner	rosyface
203	Notropis spilopterus	spotfin shiner	spotfin
204	Notropis stramineus	sand shiner	sand
205	Notropis ubratilis	redfin shiner	redfin
206	Notropis volucellus	mimic shiner	mimic
208	Pimephales notatus	bluntnose minnow	bluntnose
209	Pimephales promelas	fathead minnow	fathead
210	Rhinichthys atratulus	blacknose dace	blacknose d
211	Rhinichthys cataractae	longnose dace	longnose d
212	Semotilus atromaculatus	creek chub	creek chub
214	Semotilus margarita	pearl dace	pearl dace
216	Campostoma anomalum	central stoneroller	stoneroller
217	Notropis chrysocephalus	striped shiner	striped
232	Ictalurus natalis	yellow bullhead	
233	Ictalurus nebulosus	brown bullhead	
235	Noturus flavus	stonecat	stonecat
281	Culea inconstans	brook stickleback	stickleback
311	Ambloplites rupestris	rock bass	
312	Lepomis cyanellus	green sunfish	
313	Lepomis gibbosus	pumpkinseed	
314	Lepomis macrochirus	bluegill	
315	Lepomis megalotis	longear	
316	Micropterus dolomieu	smallmouth bass	smallmouth
317	Micropterus salmoides	largemouth bass	largemouth
331	Perca flavescens	yellow perch	perch

MNR Species Code	Latin Name	Common Name	Abbreviation
336	Etheostoma blennioides	greenside darter	greenside
337	Etheostoma caeruleum	rainbow darter	rnbow drter
338	Etheostoma exile	iowa darter	
339	Etheostoma flabellare	fantail darter	fantail
340	Etheostoma microperca	least darter	
341	Etheostoma nigrum	johnny darter	johnny drter
344	Percina maculata	blackside darter	blksd darter
381	Cottus bairdi	mottled sculpin	sculpin

Appendix C

Average Weights of Fish Species

The table below lists fish species for which average wet body weights (g) were determined at 37 locations in southern Ontario during the spring and fall of 1994. These average weights were used to transform abundances to biomasses for analyses in Chapters 4 and 5. Average wet weights for other species encountered in data sets for Chapters 4 and 5 not listed here, were obtained from data given in Scott and Crossman (*Freshwater Fishes of Canada*, Bulletin 194 of the Fisheries Research Board of Canada, Ottawa, 1973) and/or Carlander (*Handbook of Freshwater Fishery Biology*, Volume 2, The Iowa State University Press, Ames, Iowa, 1973).

Species	**Fall Weights (g)**	**Spring Weights (g)**
northern brook lamprey	9	9.5
brook trout	25	29
central mudminnow	3.5	2
white sucker	45	57
northern hog sucker	491	11
golden redhorse sucker	136	198
northern redbelly dace	2	2
carp	134	1518
brassy minnow	3	2.5
hornyhead chub	3	

Species	Fall Weights (g)	Spring Weights (g)
golden shiner		8
emerald shiner		6
common shiner	8	14
bluntnose minnow	2	2
fathead minnow	2	2
blacknose dace	3	3
longnose dace	6	4
creek chub	13	12
central stoneroller	7	5
yellow bullhead		102
brown bullhead		25.5
stonecat	43	
brook stickleback	1	1
rock bass	38	19
pumpkinseed	3	9
smallmouth bass	8	346
largemouth bass	126	338
rainbow darter	2	3.5
iowa darter	1	1

Species	Fall Weights (g)	Spring Weights (g)
fantail darter	2	2
johnny darter	1.5	2
mottled sculpin	5	5

Appendix D

Benthic Family Abbreviations

The table below lists the benthic families that were members of the data sets for this study. Abbreviations given were used in some of the figures.

Benthic Family	Abbreviation
Ancylidae	Ancyl
Asellidae	Asel
Baetidae	Baet
Brachycentridae	Brachy
Caenidae	Caen
Calopterygidae	Calop
Cambaridae	Camb
Capniidae	Capni
Ceratopogonidae	Cerato
Chironomidae	Chir
Chloroperlidae	Chlor
Coenagrionidae	Coen
Corixidae	Corix
Corydalidae	Coryd

Benthic Family	Abbreviation
Crangonyctidae	Crangon
Dytiscidae	Dytisc
Elmidae	Elmid
Empididae	Empid
Enchytraidae	Enchy
Ephemerellidae	Ephemerel
Ephemeridae	Ephemerid
Erpobdellidae	Erpob
Glossiphoniidae	Glossi
Glossosomatidae	Glosso
Haliplidae	Halip
Helicopsychidae	Helico
Heptageniidae	Hept
Hirudiniidae	Hirud
Hyalellidae	Hyal
Hydridae	Hydra
Hydrophilidae	Hydroph
Hydropsychidae	Hydrops
Hydroptilidae	Hydropt
Lepidostomatidae	Lepido

Benthic Family	Abbreviation
Leptoceridae	Leptoc
Leptophlebiidae	Leptop
Leuctridae	Leuct
Limnephilidae	Limne
Lymnaidae	Lymna
Molannidae	Molann
Naididae	Naid
Nemouridae	Nemour
Perlidae	Perlid
Perlodidae	Perlod
Philopotamidae	Philop
Phryganeidae	Phryg
Physidae	Phys
Planariidae	Planar
Planorbidae	Planor
Pleuroceridae	Pleuro
Poduridae	Podur
Polycentropidae	Polycent
Psephenidae	Pseph
Psychodidae	Psychod

Benthic Family	Abbreviation
Psychomyiidae	Psychom
Pyralidae	Pyral
Rhyacophilidae	Rhyacop
Sialidae	Sial
Simuliidae	Simul
Siphlonuridae	Siphl
Sphaeriidae	Sphaer
Tabanidae	Taban
Taeniopterygidae	Taenio
Tipulidae	Tipul
Tricorythodae	Tricor
Tubificidae	Tubif
Valvatidae	Valv