

**IMPACT OF RIP RAP ARMoured SHORELINES ON LITTORAL ZONE FISH HABITAT ON THE
WINNIPEG RIVER, CANADA**

BY

JEFFREY MICHAEL LONG

A Thesis presented to
the Faculty of Graduate Studies
in partial fulfillment of the requirements for the Degree of

DOCTOR OF PHILOSOPHY

Natural Resources Institute
University of Manitoba
Winnipeg, Manitoba

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DOCTOR OF PHILOSOPHY

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ABSTRACT

This project sought to uncover the impact rip rap shoreline armouring may have on fish habitat. I assumed that fishes most likely to be affected by the physical alterations to the shoreline were those fishes associated with the shallow waters of the littoral zone. The study was limited to an examination of granite rip rap, used in a boreal river context, with application at a project scale. Over 43 000 small-bodied and juvenile large-bodied fishes were captured from the littoral zone along the Winnipeg River, MB along with measures of environmental variables. Information from these fishes and environmental variables was used to evaluate sites and site types, three different ways.

Firstly, I used a two-pronged approach to evaluate sites, and simultaneously, to determine fish community health. Using PCA analysis, I found that Karr's Index of Biotic Integrity - IBI (1981) was highly correlated to species composition. I also found that Karr's IBI was able to uncover differences in shoreline habitat, even at a small scale (Long and Walker, 2005). Higher IBI scores were associated with armoured sites.

Secondly, I evaluated sites and site types using surrogate measures of production. Although production and species composition (especially abundance) may be collinear, they are not necessarily so. Although these measures are not fully independent of one another, they do provide important insights into habitat function. Notwithstanding, higher production values were associated with armoured sites.

Lastly, I found that despite *a priori* differences in habitat at each sample site, a constrained analysis of the species composition information indicated that several environmental variables played a role in the quality of fish habitat. Among these variables, rip rap shoreline armouring was found to be associated with better quality habitat due to its causal effects on turbidity, among other things.

The observational and multivariate nature of this study leads to specific conclusions about the ecological value of rip rap application. At the same time, these conclusions should be limited to boreal riverine systems until they have been examined for other systems.

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GENERAL INTRODUCTION

Rip rap armouring is anthropogenically placed rock primarily used to protect shorelines from eroding (Fig 1.1a-c). This study seeks to uncover and assess any relationship between the use of rip rap shoreline armouring and near shore fish habitat on a large (Simon and Emery 1995) regulated boreal river. Overall, this dissertation is divided into two parts. In part one I provide background and context. Chapter one – the introduction – situates the study within its intellectual and historical framework. Chapter two describes and critically evaluates the study design, the study area and the sample sites. Chapter three is a detailed examination of the sampling methods, focussing on fish capture methodology. In part two, I examine specifically the impact of rip rap on fish habitat using three approaches. In chapter four, I use species composition information to assess the impact of rip rap application on littoral zone fish habitat. In chapter five, I examine the use of stored energy (in the form of fish biomass and its proxies) as an instrument to detect human induced impacts from rip rap use and I compare these results to those found in chapter four. Chapter six assesses site type (armoured or unarmoured) fish habitat after adjusting for the contributions of local environmental variables. A final concluding chapter summarises and interprets the results to provide an overall assessment about the impact of rip rap on fish habitat.



Fig 1.1a. “New” rip rap (site 10). Granite stone installed on shoreline *c* 2000-2001. Average size of rock approx. 30-60 cm



Fig 1.1b “Old” rip rap (site 6). Granite stone installed *c*. 1970-1980s. Note smaller size of rock (compare to toe of rubber boot at extreme bottom edge of photo) and presence of large gravel “scree” at water’s edge forming the toe of the armouring.



Fig 1.1c Unarmoured shoreline (site 4). Note turbidity plume adjacent to shoreline.

CHAPTER 1
THE INTELLECTUAL AND HISTORICAL CONTEXT
RIP RAP SHORELINE ARMOURING: ECOLOGICAL DAMAGE OR RESTORATION?

This chapter introduces the field of natural resource management as a discipline in which applied natural resource-based problems posed in the every day world can be addressed within the context of regular scientific pursuit. To do this, first I discuss the practice of natural resource management and some of its theoretical underpinnings. Then, I provide the ecological and historical context for this study and place it within the relevant literature. Lastly, I lay out the fundamental question in the form of a null and two alternative hypotheses.

1.1 THE INTELLECTUAL ROLE OF NATURAL RESOURCE MANAGEMENT

According to Minns *et al.*, (1996), scientific assessment methodology takes one of either two forms: “one designed to enable decision-making now and another to provide peer-acceptable conclusive evidence of cause and effect.” Traditionally, the field of natural resource management followed the former course, and this issue was played out at the species level in the form of bag limits, catch limits, or the like. However, with the steady shift towards ecosystem based management, modern resource management is bringing these two dichotomous positions together. Now, the resource manager must develop, understand and evaluate decision support systems at a systems level (which implies understanding of causal links or at least of habitat and species interconnectedness) rather than exclusively at a species level (Minns *et al.*, 1996). Since all ecosystems exist in

space (and time), the modern resource manager must view place as potential habitat, and it, rather than a target species, is the primary resource to be managed. Furthermore, habitat management must consider the spatial use of habitat within the framework of temporal variation (e.g. diurnal or seasonal fluctuation). According to Langton *et al.*, (1996), the role for resource managers is to assess the spatio-temporal effects of anthropogenic events on habitat and to determine appropriate means for mitigation – (Table 1.1). However, resource managers are often practitioners who must make these determinations without adequate methods (for example, see chapter 5) or information (questions 1- 4, Table 1.1). Indeed, Fränzle (2000) argues that one of the greatest challenges to environmental science is quite simply determining the impact of human activity on the environment. For instance, with regard to aquatic ecosystems, Minns *et al.*, (1996), say that the effects of human alterations to habitat may become manifest in several ways (1) changes in total fish biomass (chapter 5); (2) shifts in species composition (chapter 4) and changes in the distribution of species over space and or time (chapter 4). Nevertheless, if we allow and equip the resource manager to make informed decisions regarding space (i.e. habitat), we can leave the biologist to determine the effects of these decisions on individual species (Langton *et al.*, 1996). While this statement demonstrates a separation between resource management and other related disciplines (Table 1.1), the distinction lies only with the study focus, rather than with the methods used or with the importance of the contribution. Indeed minimising future perturbation and reversing deleterious impacts upon space as habitat resonates with Wilson (1998), who states that habitat destruction is “the leading cause of [species]

extinction”.

Table 1.1 Priorised questions and actions to determine management of habitat(s). Questions 1-4 belong to the domain of biologists / ecologists; 5-6 to natural resource managers. Modified from Langton *et. al.*, (1996).

	<u>Priorised Sequential Questions:</u>	<u>Information and /or research action</u>
1	Is there a critical phase in the life cycle?	Literature review - complete habitat matrix
2	Is there an essential habitat?	Literature review & field and /or lab research
3	Can the habitat be characterised?	Modify habitat characterisation
4	Can the areal extent of the essential habitat be characterised?	Conduct mapping projects to determine location of habitat(s)
5	Is the essential habitat vulnerable?	Conduct experiments to determine human impacts on habitat
6	Develop means to protect, conserve, restore and/or enhance essential habitat(s)	Determine effectiveness of effort or plan

1.1.1 ECOSYSTEM MANAGEMENT - THEORY AND HISTORY

Typically, modern natural resource management utilises ecosystem management as an overarching philosophy in which to study the effects of anthropogenic impacts on ecosystems and essential habitats (Grumbine 1994; Haufler *et. al.*, 1996; Haufler *et. al.*, 2002). Meshed within this philosophy are discipline-based theories which provide an intellectual framework in which empirical research can be positioned by offering broad understanding to specific phenomena. However, with respect to ecological inquiry, the possibility for contributions to new theories are limited. Instead, contributions typically support existing theory. Indeed, Breckling and Dong (2000) argue that ecology has not been able to contribute universal laws and principles the way other scientific disciplines

are able to do – chiefly they say, because most ecological studies are highly site specific. Jørgensen and Müller (2000) contend that the opportunity to develop ecological theory has been limited due to the problem-solving focus of many ecologists (and environmental scientists) and thereby defend the idea that ecological theory is not as well developed as theory within other disciplines. Notwithstanding the overall lack of ecological theory, both Breckling and Dong (2000) and Fränzle (2000) recognise the importance of hierarchy theory. Indeed, while I utilise several theories to provide intellectual context and explanation, I rely on hierarchy theory as a method of understanding general ecosystem organisation. Therefore, I discuss the ideas of ecosystem based management and hierarchy theory in greater detail.

Ecosystem (based) management can be traced to its infancy with the postulation of the ecosystem as an intellectual unit in the 1930s, (O'Neill *et. al.*, 1986; Slocombe 1993; Grumbine 1994) but its real acceptance and widespread use by scientists has only emerged since the 1980s (Grumbine 1994). Perhaps owing to its recent development, ecosystem (based) management has been ascribed neither common nomenclature nor a particularly firm definition. For instance, whereas Grumbine (1994) uses the term 'ecosystem management', Jensen and Bourgeron (2001) use 'ecosystem based management' and Van Eeten and Emery (2002) use both terms, where ecosystem management is simply an abbreviated form of ecosystem based management.

I contend the term 'ecosystem management' suggests that entire ecosystems are

manipulated, and that they are done so for human or societal purposes. In contrast, I suggest that 'ecosystem based management' implies that management is conducted primarily from the ecological /ecosystem perspective and therefore is possibly at odds with modern human and industrialised societal demands. However, these implied distinctions in terminology are not supported within the literature. In fact, the opposite is more accurate. For example, Slocombe (1993) and Heissenbuttel (1995) advocate public resource use for social purposes and they include "economic and social needs" as inherent goals of ecosystem based management. Similarly, Jensen and Bourgeron (2001) state that "ecosystem based management is an evolving philosophy that many government agencies have adopted in the multiple-use, sustained-yield management of federal lands". They claim that the main goal of ecosystem based management is to ensure sustained integrity¹ of ecosystems for the future while at the same time, continuing to produce goods and services for the present. Implicit within this statement is that future use – both rate and amount – (1) can be predicted, and (2) will be available if we manage ecosystems under the currently acceptable concept and level(s) of integrity. In contrast, scholars such as Grumbine (1994) and Wilcove and Blair (1995), who hold an ecological bias, focus objectives on the preservation of biodiversity and ecosystem health and stability, while placing less emphasis on the accommodation of "human use

1

The term "integrity" is used by the authors here to mean ecological health – that is a properly functioning ecological unit rather than the more rigid idealised definition attributed to ecological integrity of an ecosystem which is not unduly influenced by modern, industrialised human activity - arguably an impossible circumstance in the modern era (see for instance glossary entries in Draper and Reed 2005).

and occupancy” (Grumbine 1994). Indeed, Wilcove and Blair (1995) state specifically that “human activities are to be accommodated within these constraints.” While Haufler *et. al.*, (1996) recognise that many agencies implement ecosystem management to meet targets such as the “maintenance or enhancement of biodiversity”, they state that generally ecosystem management strives to balance ecological, social and economic objectives (see also Haufler *et. al.*, 2002). Noss (2000) agrees, claiming that prevailing use and understanding of ecosystem management places primary emphasis on human needs. However, he goes on to argue that “any ecosystem management project that operates on this assumption will ultimately fail, as it will not maintain ecological integrity.”

Although this lack of agreement over primacy in ecosystem (based) management terminology makes specific use difficult as a rigid set of guidelines, it does not prevent researchers and others from adopting some form of ecosystem thinking for their work. For instance, Wilcove and Blair (1995) state that “the fact that no one really knows what ecosystem management means has not diminished enthusiasm for the concept.” Pahl-Wostl (1998) agrees, stating that the term ecosystem management is vague and this ambiguity gives authors the opportunity to “apply these terms any way they want” (Pahl-Wostl 1998). Wagner (1995) presents a third option by simply stating that ecosystem management is “the skilful manipulation of ecosystems to satisfy specified societal values.” Despite these variations, there is some common ground underpinning the essentials of ecosystem management philosophy.

Regardless of the failure of scholars to agree on specific nomenclature and to adhere to a strict definition of ecosystem management, these differences are matters of degree.

Currently, there is no clear, inherent meaning in one term that utterly negates use of the other. Some scholars include the role of “human occupancy and use” within the ecosystem to be a primary consideration entrenched within ecosystem management (or its derivatives) while for others this role is secondary. Nevertheless, there is unanimity that an examination of the ecosystem and its management recognises the inclusion of human involvement, at some level, either explicitly or implicitly.

There is also general consensus about the nature of ecosystems themselves. Costanza *et al.*, (1993) state that “ecosystems are groups of interacting, interdependent parts linked to each other by exchange of energy, matter, and information” (*in* Turner and Johnson 2000). Schopf and Ivany (1998) use a similar definition and consider an ecosystem as a “reference to the organismal systems tied to particular portions of time and space...”. Noss (1990) concurs, arguing that the “term ecosystem includes abiotic aspects of the environment with which the biotic community is interdependent.” Essentially, the ecosystem is an intellectual unit encapsulating otherwise independent components which are bound together in a systematic way. By meshing these points of consensus (human involvement and ecosystem definition), ecosystem management can be usefully defined. Conceptually, and in the short term, providing that ecosystem management includes an examination of the interaction of humans and their environs, the terms ecosystem management and ecosystem based management can be used interchangeably without

undue confusion. Nevertheless, I envision the distinction I have outlined above will grow rather than diminish with time. Harper (2004) argues that humans have a dual role in the environment: 1) as biological organisms forming a part of nature and 2) as a species with an inordinate capacity to alter our environment and one which lives apart from the natural world. The human disconnection from the natural world is compellingly argued by Pollan (2006) in his description of the industrialised manufacture of food. As humans continue to develop our increasing capacity to alter ecosystems through industrialised means, we may correspondingly diminish our role as merely another species among many. Therefore, the role of humans within ecosystem management is not merely a matter of semantics. Instead, our role belongs to a larger philosophical debate hinged upon deciding whether we have abrogated our “right” to be considered just another species and thus permitted to modify our habitat for our own purposes as might any other species. Or, whether we are obliged to accept a transcendental role, unlike any other species, as a means of mitigating some of our extraordinary power. Therefore, when we speak of ecosystem management, if not now then in the future, perhaps we should qualify what we mean by ‘the human role’, using Harper’s rubric. *In this study, ecosystem management acts as a conceptual framework which examines the role of the industrialised human on the linkages between biotic and abiotic factors within an ecologically and anthropogenically meaningful described space.* Specifically, this study reflects the human manipulation of ecosystems through policy according to a hierarchy of human demands. Finally, there is agreement that ecosystem management is predicated on the ecological principle of hierarchical theory (Slocombe 1993; Haynes *et. al.*, 1996

in Jensen and Bourgeron 2001; Lessard *et. al.*, 1999; Boureron *et. al.*, 2001; Van Eeten and Emery 2002).

1.1.2 HIERARCHY THEORY

Hierarchy theory provides an intellectual vehicle for applying ecosystem management philosophy to real world problems by explaining how ecosystem linkages are to be understood. Under this theory, individual ecosystem components, each with a specific set of spatial and temporal scales, are nested within larger layers, forming a hierarchy (Overton 1972; Allen and Starr 1982; Salthe 1985; O'Neill *et. al.*, 1986; O'Neill 1988; Messer 1992; Parker and Pickett 1998; Turner and Johnson 2001). The relationship between layers is not simply a sum total proposition, but rather each successive level up the hierarchy is enabled by its constituents and these parts, integrated together as a whole, form more than the sum of the parts (Messer 1992). Additionally, each level bears a particular focus which is not readily observed at adjacent levels (Messer 1992). This structure has important considerations for the indicators which are used to measure each level; indicators must be appropriate to the hierarchy being measured or else they will not reflect changes at that focal level (Messer 1992). In other words, the indicator framework must be linked in a spatial-temporal sense to the scale of the issue.

Accordingly, this framework has practical application in selection of sample area and sample unit size (see chapter 2) as well as in sampling frequency (see chapter 3).

Minimally, hierarchy theory requires the identification of three distinct levels within an

ecosystem. Unfortunately, the nomenclature for these levels abounds with synonyms, making for distracting reading. The area of focus is the “N” level (“clade”, Salthe 1985; “focal level”, Messer 1992; “scale of the investigation”, Parker and Pickett 1998; or “phenomenon”, Turner and Johnson 2001). Within this level is nested the “N⁻¹” level (“lineage”, Salthe 1985; “-1”, Messer 1992 or “mechanisms”, Turner and Johnson 2001). Components in this level are independent and like those in the N level, have their own space and time. As a rule, components at the “N⁻¹” level operate faster (by an order of magnitude – O’Neill *et. al.*, 1986; O’Neill and King 1998) than do components at the “N” level (Messer 1992; Parker and Pickett 1998). This can be difficult to assess, especially if units of measurement are not identical or otherwise commensurable.

Nevertheless, the reasoning is intuitive; smaller units can operate within the sphere of the larger, slower, [upper] level, but not vice versa. On the other side of the N level, is the “N+1” level (“context”, Salthe 1985; “+1”, Messer 1992 or “controls”, Turner and Johnson 2001). Components of this level are larger, and usually operate slower (by an order of magnitude – O’Neill and King 1998) than components of the “N” level (Messer 1992). As with the “N” and “N⁻¹” levels, components at the “N+1” level also have a particular time and scale, and within this level is nested the “N” level. Hierarchy theory makes sense intuitively since larger parts are made up of smaller ones. While larger parts are consequently more complex, they are also slower, since they are limited by the process of aggregation as well as slowed to the maximum speed of the slowest sub-component. Furthermore, this level also contains any delay associated with the interaction of the various sub-components.

Ecosystems are dynamic and in response to disturbance, they demonstrate either normal variation (resilience) or change. Unlike discontinuous change which indicates a qualitative difference between two states (say “before” and “after”), continuous change describes the incremental stages of change which exist between the “before” and “after” states (Turner and Johnson 2001). Whereas discontinuous change can only describe that a change has occurred, continuous change implies scale-appropriate measurements that demonstrate the process of change. Hierarchy theory implicitly demands a scaled approach to the measurement of variables at each of the ‘N’ levels and therefore facilitates better understanding of the causal relationships and the nature of change within ecosystems. Without scale-appropriate measures, components within ecosystems may be erroneously described. Thus the ecosystem might be misunderstood. For example, the circumference a single heavily barked tree measured by a forester and by an hypothetical 10 mm insect would yield discrepant values (Kenkel and Walker 1996); neither of which would be of much use to the other for explaining taxon-specific, distance-related interactions with the tree. Thus hierarchy theory provides an important intellectual framework for the research design.

1.2 STUDY CONTEXT

This section describes the physical features of the ecosystem under examination, the specific historical and jurisdictional context necessary to understand the study and a review of the relevant literature.

1.2.1 SHORELINE EROSION AND THE LITTORAL ZONE

The shoreline is the nexus between the aquatic and terrestrial environments and plays a disproportionately larger role in aquatic community development than does the pelagic zone (Wetzel 2001). Indeed, in many large rivers the centre channel is hydraulically efficient but biologically poor in contrast to the rich river margins (Stalnaker *et. al.*, 1989). Fishes are of special interest to humans for their sport, food and cultural value and, as a result, their interaction with the littoral zone is well established within the literature. While species-specific spawning (Geiling *et. al.*, 1996; Hamel *et. al.*, 1997) and nursery/ juvenile habitat (Casselman and Lewis 1996; Chiasson *et. al.*, 1997) continue to be important issues in fisheries research, attention is shifting to the examination of the total fish assemblage of the littoral zone (Floyd *et. al.*, 1984; Boisclair and Leggett 1985; Benson and Magnuson 1992; Weaver *et. al.*, 1993; Minns *et. al.*, 1994; Pierce *et. al.*, 1994; Beja 1995; Jurajda 1995; Jones *et. al.*, 1996; Jennings *et. al.*, 1996; Fischer and Eckmann 1997a, b; Fago 1998; McInerney and Cross 2000; Pierce *et. al.*, 2001).

Land use adjacent to the shoreline can influence the littoral zone (sec. 2.2.2.1.3) by providing (or not) allochthonous energy inputs (vegetative matter; woody vegetation; insects; etc.) and / or disturbances (particle loading through excessive erosion; fertilisers; other associated run-off pollution). As part of the littoral environment, both the *epi* and *supra* littoral zone (always above water, not influenced by wave spray; always above water, but influenced by wave spray, respectively) act as the transition zone between the

terrestrial and the aquatic environments. These zones can act either as barriers to, or conversely, as pathways for terrestrial inputs to aquatic systems. On the other hand, the *eulittoral* zone (i.e. a relative term indicating the water / shore interface at any point through the seasonal range) is by definition exposed to chronic impact, such as wave action (Wetzel 2001), and thus it makes autochthonous contributions (beneficial or deleterious as the case may be) to the aquatic system. In regulated systems, where water levels are generally less seasonally dynamic (though not necessarily less erratic) than in unregulated systems, the tendency for the eulittoral to shift in absolute space for any prolonged period of time is reduced. Thus, the opportunity for the eulittoral to recover from chronic disturbance through vegetative regeneration is similarly reduced. In addition, the usual circumstance of a regulated river system results in a rise in mean water level, leading to a prolonged period of chronic disturbance on a previously unexposed (or rarely exposed) portion of the shoreline. When shorelines which are susceptible to erosion form the new eulittoral zone, they can contribute particulate matter to the aquatic system beyond the dosage limits of some aquatic resident species.

1.2.2 SHORELINE EROSION AND SUSPENDED SOLIDS

The literature regarding the effects of suspended solids on the aquatic community is extensive. A comprehensive, although not exhaustive, annotated bibliography was compiled by Kerr (1995), consisting of almost 1200 references. While an analysis of those sources is neither appropriate nor tenable within the scope of this study, several studies bear review in order to understand the effect of suspended solids on fish habitat

in general. Effects occur at several levels, including (1) physiological damage to fish and their eggs as a result of degraded water quality; (2) impacts to prey; (3) indirect impacts on aquatic vegetation, and (4) physical changes to the abiotic habitat.

1.2.2.1 DIRECT IMPACTS TO FISH AS A RESULT OF DEGRADED WATER QUALITY

The Fisheries Act [Sec. 34(1a)] addresses water quality under the umbrella of deleterious substances; it reads: "any substance that, if added to any water, would degrade or alter or form part of a process of degradation or alteration of the quality of that water so that it is rendered or is likely to be rendered deleterious to fish or fish habitat or to the use by man of fish that frequent that water." According to this definition, fish which exhibit impairment as a result of poor water quality are therefore considered to have been subject to deleterious substances which therefore, in turn, constitutes damage to fish habitat.

Alabaster and Lloyd (1982) list a suite of negative impacts that suspended solids manifest upon freshwater fisheries including: outright mortality (Ward 1992), increased susceptibility to disease, reduced growth rate, modified movement patterns, reduced spawning area, egg mortality and affected food sources. Ward (1992) notes that suspended solids can enter the gut of fish, leaving the fish filled with indigestible material. Although Nolen *et. al.*, (1985) simply state that turbidity² has a negative effect

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It should be noted that all seston (all particulate material in the free water including bioeston (plankton) and abioeston (non-living particulate matter)) can contribute to

on sport fishes, there is general evidence suggesting that every fish species has a turbidity threshold, values above which result in increasing susceptibility to its damaging effects. (Alabaster and Lloyd 1982; Newcombe and Jensen 1996). Through extensive literature review, Newcombe and Jensen (1996) developed a numeric measure for assessing effects of suspended solids on various lentic, lotic and estuary species. They found that harmful dosages (concentration x exposure duration) to individuals were lower than those at which significant responses in populations could be detected (Newcombe and Jensen 1996). These results are in keeping with traditional ecotoxicological benchmarks such as LC_{50} (concentration which is lethal to 50% of the test population) and suggest that PNEC values (probable no effect concentration) are considerably lower than many stated threshold values. Therefore, even in water with acceptable turbidity values (a standard benchmark <25 NTU's Nolan *et. al.*, 1985 – values for State of Oklahoma; Kerr 1995); populations may not show visible harm, yet individuals may suffer ongoing effects and thus populations can be subject to chronic, systematic impacts. Alternatively, habitats that experience excessive abioseston derived turbidity may simply become species poor as a result of species-specific avoidance behaviours, or due to disruptions associated with reproduction (Berkman and Rabeni 1987). In either situation, abioseston derived turbidity can degrade fish habitat quality. Schnick *et. al.*, (1982) described the settling time for disturbed sediments from four mid-western rivers at 30-150 minutes. Thus, ecosystems which experience heavy littoral disturbance (e.g. boat or wind generated

turbidity. However, turbidity (measured as the inverse of water clarity) is usually considered to be comprised of abioseston – that is inorganic material, or suspended solids, see Wetzel (2001).

waves) may endure prolonged settling times to the point of continuous disturbance over the course of the day.

1.2.2.2 IMPACTS TO PREY

The Fisheries Act [Sec. 34(1e)] includes areas of food supply within its definition of fish habitat. Consequently, impacts to prey are a direct threat to fish habitat. In general, suspended solids pose a threat to prey species. For instance, the common prey, *Daphnia ambigua* were found to have reduced maximum body length (1.19 mm control; 1.09 mm impact), reduced age-specific survivorship and both reduced age-specific fecundity and cumulative fecundity in the presence of suspended clay (mean particle size 1 μ m; [50mg/l]) (Kirk 1992). Exposed *Daphnia* juveniles were also shown to have reduced body size and reduced age-specific survival (Kirk and Gilbert 1990). Presumably, increased aquatic turbidity results in reduced body mass for juvenile fishes, since these fishes (1) consume smaller prey and (2) will be forced to expend more energy in search of food; (Ward 1992) leaving less energy for growth.

1.2.2.3 IMPACTS TO AQUATIC VEGETATION

Light attenuation as a result of suspended solids is negatively correlated with algal photosynthesis (Blanch *et al.*, 1998), macrophyte photosynthesis (Harding *et al.*, 1986), plant growth and floral diversity (Gardiner 1991), and thus theoretically, summer CO₂ uptake and O₂ production. The littoral zone is generally considered to extend to the depth of water to which visible light can penetrate, thus fuelling macrophyte growth (Wetzel

2001). Aquatic vegetation plays a crucial role in fish habitat. Without overlooking the important contribution made by algae (for O₂ production and as a food source), macrophytes make a critical contribution to fish habitat, due in large part to the physical structure they provide – especially for juveniles in the presence of predators (Hayes *et al.*, 1996). As an example, fish species segregation in the littoral zone of two small Michigan lakes could be accounted for, in part, by “vegetational structure” (Werner *et al.*, 1977). Weaver *et al.*, (1997), qualified the importance of macrophytes, finding that macrophyte patchiness plays a more important role than abundance for some fish species and age classes. For instance, whereas they found *Perca flavescens* (yellow perch) was most abundant and dominant where vegetation was most complex and abundant, yearling and older *Ameiurus nebulosus* (brown bullhead) and *A. melas* (black bullhead) preferred patchy macrophytes to overall macrophyte density (Weaver *et al.*, 1997). Similar results were found by Casselman and Lewis (1996) who found that *Esox lucius* (northern pike) were most abundant under moderate (31-70%) macrophyte cover. Consequently, macrophyte species composition and distribution is an important consideration in the assessment of fish habitat. Shoreline erosion which results in increased turbidity may limit macrophyte development to sub-critical values and consequently reduce fish habitat in the littoral zone (see chapter 6).

1.2.2.4 IMPACTS TO THE ABIOTIC ENVIRONMENT

In general, fish demonstrate avoidance behaviour to suspended solids in the water column (Mulkey and Falco 1977 citing Sorensen *et al.*, 1977; see also Berkman and

Rabeni 1987). Apart from the water itself however, other abiotic features of fish habitat are susceptible to impact from suspended solids. Mulkey and Falco (1977) argued that sedimentation creates changes to aquatic habitat and therefore changes to community succession, irrespective of sediment types. Snell (1984) repeated the concern that sedimentation from terrestrial systems may be damaging the aquatic system through siltation and loading of potential toxicants. According to Ward (1992), sediments settle and prove harmful to fish by smothering eggs and destroying spawning beds by filling interstitial spaces. Cunjak (1996) investigated overwintering habitat for fishes and found that fine sediments introduced to the water course can reduce habitat heterogeneity by filling spaces under boulders, cobbles and other course substrate.

1.2.3 RIP RAP SHORELINE ARMOURING

Traditionally, shoreline erosion techniques have been developed for the protection of terrestrial property, with little consideration given to the impact on the aquatic environment. To date, the preferred method of shoreline protection has been human-placed stone, commonly referred to as rip rap (ref. Figs 1.1a & 1.1b).

Rip rap is a common and frequently recommended engineering approach to shoreline armouring (USA Highway Research Board 1973; The Manitoba Water Commission 1975; Boysen 1977; Highfill and Kimberlin 1977; US National Research Council 1980; Bowie 1982; Schnick *et al.*, 1982; Goldman *et al.*, 1986; Pilarczyk 1990; Gray and Sotir 1996; City of Wpg. 2000). Indeed, the US Dept of Agriculture (1989) states: "rock rip rap

is the most widely used and considered the most desirable type of revetment in the United States". As part of "good practice", DFO and MB DNR, (1996)³ specifically regard rip rap as an appropriate "long-term erosion control technique" and an important streambank protection strategy.

Apart from its beneficial engineering properties, some authors contend that rip rap armouring enhances aquatic habitat. However, within much of the literature, the assessment of rip rap with respect to fish habitat is secondary or casual, rather than examined specifically. For example, Lubinski *et al.*, (1981) posit that rock revetments *may* create superior habitat compared to natural shorelines. Gore (1985) recommends rip rap to increase velocities, reduce suspended sediment and therefore increase macroinvertebrate habitat. Gardiner (1991) concurs, arguing that rock filled gabions and other granite rubble provides suitable holes for organisms such as crayfish, and DFO (1999)⁴ state that rip rap includes "interstitial spaces [which] provide cover and rearing habitat for fry and juvenile fish." Cairns (1995) adds that rip rap adds substrate for periphyton and benthos, as well as spawning beds for fish. Some authors simply report that rip rap shoreline forms a specific shoreline type off which sampling is conducted but is otherwise left unqualified (Dionne and Karr 1992; Fago 1998). Still other scholars

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Document prepared by a consultant and was not subject to peer review – perhaps accounting for why this DFO source stands in disagreement with DFO regulatory practice *viz* rip rap.

⁴ *ibid.* – although a different consulting firm.

have reported benefits specifically to fish stocks. For instance, Gore and Bryant (1988) indicate that rip rap treatment in restored rivers and streams enhances fish production. Similarly, Jurajda (1995) observed that rip rap shorelines figured among the best nursery grounds in a channelized river in the Czech Republic. However, this conclusion should be treated as a guarded endorsement given that no comparison could be made to natural shorelines. Still, Kempinger (1996 citing Folz and Meyers 1985) reports that rip rap protected streambanks provided an increase in spawning habitat for *Acipenser fulvescens* (lake sturgeon). Working in Lake Constance, Germany, Fischer and Eckmann (1997a, b) noted a substantial increase in fish biomass at rip rap-like sample sites. They concluded that *Noemacheilus barbatulus* (stone loach) and *Lota lota* (burbot) showed “a strong preference for areas with a high percentage of stone coverage.”

At least one study has explicitly examined rip rap among other perturbations. However, the results were ambiguous. For instance, in an examination of 14 compensation case studies, DFO authorities in Ontario determined that stone revetments and rip rap shorelines were categorised as both constituting a HADD and providing HADD (see sec. 1.2.6) compensation (Azimuth Consulting 2001). These conflicting results may have arisen due to the use of a model developed primarily for assessing large scale projects (Azimuth Consulting 2001; Minns 1995; see chapter 5). Perhaps most importantly, these results indicate the importance of developing appropriately scaled assessment methods for small scale projects.

Opponents to rip rap use focus primarily on the physical change-to-space at the littoral-terrestrial interface (the eulittoral zone). For example, Environment Canada (1999) is concerned that rip rap isolates the terrestrial from the aquatic environment. DFO (1999) provides a catalogue of advantages and disadvantages attributed to rip rap use. Although nowhere among the disadvantages are listed the loss of aquatic habitat, health or integrity, there is the stipulation that channel width should not be narrowed as a result of rip rap use. DFO (2002) repeats the concern and warns against the use of rip rap where there is risk of reducing natural channel width. Implicit within this provision is that spatial quantity can be equated with habitat quality and therefore loss of littoral zone due to rip rap in-fill must necessitate a net loss of habitat. Taken at face value, this argument is intuitively sound, since removal of critical littoral habitat can be a serious concern for some fish species. However, installation of rip rap seldom removes all littoral habitat. Consequently, the more important issue is whether rip rap installation removes important or significant habitat. This begs the fundamental question of this thesis: is the "loss" (i.e. alteration) of physical space compensated for by an increase in habitat quality, resulting in a net improvement of fish habitat or not?

There is a relatively small body of technical reports and peer reviewed work that specifically addresses the impact of rip rap on fish habitat. The most detailed approach to date has been by Jennings *et al.*, (1996) who used a large scale approach, (354 samples from 20 different Wisconsin lakes) during a single sampling season. They found that rip rap enhanced (statistically significant) fish habitat, irrespective of variations in large

scale parameters such as differing land use patterns (urban, forested and agricultural) and lake type. However, while there is some evidence to support rip rap as a habitat-enhancing technique, this position is not unanimous. For instance, Knight and Cooper (1991) found both fish number and biomass gradually declined (over 3 years) following placement of stone toe revetments in small Mississippi streams. Their findings suggest assessment over more than one field season is important. Shields *et al.*, (2000) who sampled a small warm-water Mississippi stream (average width 35 m), found that species diversity was lower at treatment sites than at natural sites. However, the authors recognise that bank protection treatments were linked to specific stream morphology (riffles, bends or pools) and therefore sampling results likely reflect some blend of both stream morphology and bank protection method (Shields *et al.*, 2000). Shields *et al.*, (1995a) reviewed various technical and peer reviewed riparian studies, and concluded that rip rap effects could be assessed on three separate scales, with three different results. At the micro scale (areas roughly equivalent to the mean stone diameter), rip rap was found to be overwhelmingly beneficial to invertebrates and compared favourably with natural shorelines as benthic habitat. At the meso-scale (areas roughly equivalent to the square of the channel width), results were mixed, in part depending upon the basis of assessment. For instance, some investigators measured biomass change, whereas others compared species richness. Out of ten studies measuring species richness, eight direct comparisons were made between natural banks sites and rip rap sites. Of these eight, three studies found that species richness was greater at natural bank sites, four studies showed it was greater at rip rap sites and the same in one study. Lastly, at the macro scale

(areas roughly equivalent to 10 times channel width), negative effects of rip rap were determined by commercial harvest returns. However, the temporal scale (e.g. “since 1915”; or “from 1950-59”) leaves room for doubt that the observed changes were due exclusively to rip rap and not due, in part, to a suite of other environmental variables over these periods. Indeed, in these cases, the causal link appears related to removal of backwaters and the reduction of flooding rather than to rip rap presence or absence. Consequently, poor commercial harvest returns appear to be more closely linked to channelisation, rather than the material used to create channels.

Despite this work, some gaps remain. Firstly, Mathews and Robinson (1998) found that large scale analysis did not explain faunal composition at smaller scales and that local assemblages were best explained by local factors or at a fine scale. By fine scale, I mean a smaller grain size (i.e. unit size -see sec. 2.1.2.2) rather than fine filter (which tends to focus at the species or guild level (Haufler *et al.*, 1996). Admittedly, while the accumulation of many small scale impacts may result in a unique type of disturbance at the landscape scale (*sensu* Shields *et al.*, 1995a), since many disturbances to the littoral zone occur in small, discrete blocks, understanding the ecology of disturbance at a fine scale is necessary for its own sake. This issue is revisited in the discussion of sample site parameters (sec. 2.3 *et seq.*). Secondly, Shields *et al.*, (2000) and Jennings *et al.*, (1996), among others, assume a positive correlation between fish species composition or fish biomass and fish habitat quality. However, while Angermeier and Karr (1986) and Gammon and Simon (2000) show these are important contributing measures of aquatic

health, a suite of metrics within the Index of Biotic Integrity - IBI (Karr 1981) framework may provide a more comprehensive understanding of aquatic health. To date, there is no quantitative comparison of species composition information with IBI scores at a fine scale in boreal river ecosystems (see chapter 4). Similarly, there is no quantitative comparison of fish biomass with species composition information at a fine scale in boreal river ecosystems (see chapter 5). Lastly, Yount and Niemi (1990) argue that ecosystem recovery is slowest after disturbance to the physical environment. However, there has been little attempt to examine the effect of rip rap age on fish habitat (but Knight and Cooper 1991) and none in boreal river ecosystems, or where time frames are larger by at least an order of magnitude than life-spans of organisms within the community assemblage (see chapter 6).

1.2.4 ENFORCEMENT OF THE FISHERIES ACT IN MANITOBA

Shortly after joining confederation in 1870, the Province of Manitoba (among others) undertook jurisdictional negotiations with the Federal Government. The results of these negotiations culminated in the December 14, 1929 *Manitoba Natural Resources Transfer Agreement*, which was confirmed in Parliament, July 10, 1930 under *The Constitution Act*. Among the rights and responsibilities transferred to the western provinces including Manitoba, was control over fishery resources:

Except as herein otherwise provided, all rights of fishery shall, after the coming into force of this agreement, belong to and be administered by the Province, and the Province shall have the right to dispose of all such rights of fishery by sale,

licence or otherwise, subject to the exercise by the Parliament of Canada of its legislative jurisdiction over sea –coast and inland fisheries. (*The Constitution Act*, 1930).

For the next 53 years, Manitoba's fishery (i.e fish stocks) was administered by the Province of Manitoba's Department of Natural Resources, while responsibility for fish habitat remained within the Federal jurisdiction. However, following a public discussion paper in 1983 and a policy and procedures paper in 1985, it became apparent that a new approach to the management of the *Fisheries Act* in general and of fish habitat specifically, was desirable (DFO 1986). In October 1986, the Department of Fisheries and Oceans presented to Parliament, a policy entitled: *Policy for the management of fish habitat* (DFO 1986). This policy outlined the department's new approach to fish habitat management. The policy was considered, at least internally, as the "first national example of a workable environmentally sustainable approach to resource management in Canada" (DFO 1986). Notwithstanding, the policy still recognised the day-to-day provincial administration of all fisheries in Ontario, Manitoba, Saskatchewan and Alberta, as well as freshwater anadromous and catadromous fisheries in Quebec and British Columbia – an arrangement that continued through 1991 (DFO 1991) and beyond.

In the 1996 *Throne Speech*, the Federal Government stated its intentions to delegate enforcement of the *Fisheries Act* to the Provinces, thus integrating management of both fish stocks and habitat at the provincial level. However, both Ontario and Quebec

rejected this proposed transfer and responsibility for the act remained with the Federal government's Department of Fisheries and Oceans – DFO – (MB Heavy Construction Assoc. *et al.*, 2002). Thus, from that point forward, DFO resumed its primary authority over all projects which may affect fish habitat outside provincial park boundaries. Since 1996, DFO has significantly increased focus on protection of fish habitat, according to the guiding principles established in the 1986 policy. The subsequent search for better understanding of anthropogenic impacts to watercourses has been in direct consequence of this revived concern for fish habitat.

1.2.5 RIP RAP AS A BEST MANAGEMENT PRACTICE - A CURRENT RESOURCE

MANAGEMENT CONTROVERSY

Manitoba Hydro is Manitoba's predominant energy provider, acting as a producer and seller of hydroelectricity (and seller of natural gas). This utility relies extensively on hydro electricity generation to meet legislated energy service to residents of the Province of Manitoba, and for surplus power for export. Along the Winnipeg River alone, Manitoba Hydro owns and operates six hydro electricity producing stations, with facilities (listed from upstream to downstream) at Point du Bois, Slave Falls, Seven Sisters Falls, MacArthur Falls, Great Falls and Pine Falls. Although these dams are considered run-of-the-river stations, each site influences water levels immediately upstream from the dam site, in the area known as the fore-bay and for some distance beyond. In general, each dam is operated according to water levels controlled largely through the Lake of the Woods Control Board (LWCB). LWCB macro control and site

specific local fore-bay control together have created a scenario whereby post development river levels are elevated above historic levels (K. B. Burnett per. comm; Winnipeg River Power Surveys period maps 1911-1913 – these maps show land slated for inundation after hydroelectric development) and pre-development seasonal water level fluctuations have been reduced in the post development era. These elevated water levels have resulted in potential and actual riverbank destabilisation and bank failures both within communities and for individual land holders along the Winnipeg River. Therefore, Manitoba Hydro considers bank stabilisation to be an important mechanism for reducing shoreline erosion and the resulting damage to public and private land. Typically, this utility uses rip rap armouring for shoreline protection. Recently, however, this practice has come under scrutiny from the Federal regulator (DFO), over concerns that the practice may constitute a HADD.

1.2.6 HADD - HARMFUL ALTERATION, DISRUPTION OR DESTRUCTION

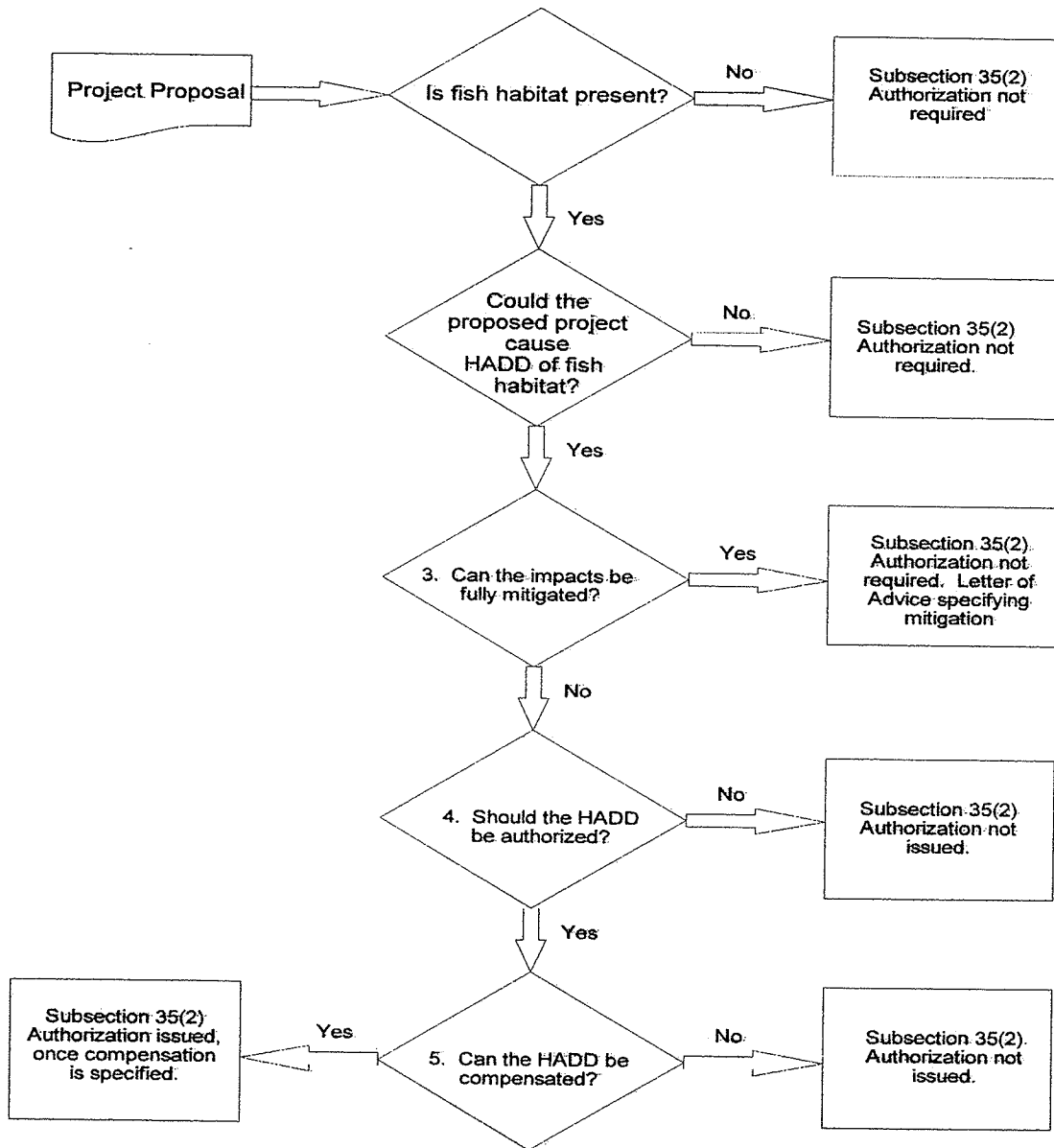
Under its renewed mandate, DFO operates its habitat protection policy on a hierarchy of preferred options. The regulator prefers project relocation or redesign to avoid fish habitat altogether (DFO 1995; DFO 1998). However, if these two preferences are unmanageable then further project assessment is necessary. In order to establish a national process for projects which *may* lead to habitat alteration, the department produced a decision framework document, outlining definitions, process and mitigation points (Fig. 1.2) to fulfill the 1986 no-net-loss guiding principle (DFO 1998) and section 35 of the *Fisheries Act*. Within this document a HADD (Harmful Alteration, Disruption

or Destruction) is defined as: “any change in fish habitat that reduces its capacity to support one or more life processes of fish” (DFO 1998). A HADD can occur when, as the result of [new]⁵ anthropogenic impact, “there is a change to the physical, chemical or biological attributes of fish habitat which is of a type and magnitude sufficient to render the habitat less suitable, or unsuitable, for supporting one or more life processes of fish” (DFO 1998). Furthermore, and with particular importance for rip rap use owing to its permanency, the document indicates that harmful alteration is regarded as “any change to habitat that *indefinitely* reduces its capacity to support one or more life process of fish but does not completely eliminate the habitat” (DFO 1998). The decision framework sets out in a series of five questions, a process which determines whether any project constitutes a HADD, along with subsequent required action (Fig. 1.2). Although these questions are clear and simple, the answers may be complex. In anticipation, built within the framework is the DFO position on uncertainty: “in cases where there is doubt about the impact of a project on fish habitat and if sufficient information is not provided to enable for a conclusion that a HADD is not likely to result, reviewers should adopt a precautionary approach and conclude that a HADD is likely to result” (DFO 1998). Since there is uncertainty about whether rip rap constitutes simply an alteration (i.e. not a

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It should be noted that although the decision framework document does not state that its process is specifically limited to new impacts, this position is implicit in the language of the conditional future used in the decision tree for the “project proposal” (second diamond, Fig. 1.2). Secondly, the document contains no provision for retroactive HADD assessment. Consequently, an existing dam across a river is not subject to HADD assessment, regardless of any ongoing impact to fish habitat that it might cause, whereas construction of a new dam (or any other alteration, regardless of the magnitude of its impact) is subject to the HADD assessment protocol.

Figure 1.2 Decision framework for HADD, (DFO 1998)



HADD) or, a *harmful* alteration (HADD), the rule of uncertainty applies. Therefore rip rap use is currently considered a HADD.

1.2.7 CONCERNS ABOUT HADD AND NO-NET-LOSS (NNL)

The DFO decision framework is built upon the understanding that section 35 of *The Fisheries Act* is designed to protect fish habitat due to its importance to fisheries (or potential fisheries) rather than to fishes (DFO 1998). Furthermore, this human-use perspective is limited to extractive uses: “ ‘fishery’ includes the area, locality, place or station in or on which a pound, seine, net, weir or other fishing appliance is used, set, placed or located, and the area, tract or stretch of water in or from which fish may be taken by the said pound, seine, net, weir or other fishing appliance.....” (*The Fisheries Act*, 2(c)). In short, section 35 of the fisheries act preserves fish habitat in order to produce fishes for human extraction. There is no provision within this section for fish *qua* fish, nor for other non-extractive human uses. As a result, there is a link between extractive capacity and productive capacity.

Underlying the no-net-loss principle is the assumption that fish habitat and fish production are positively correlated (DFO 1998; Jones *et al.*, 1996). This assumption has come under two criticisms. Firstly, productive capacity (and ultimately extractive capacity) is not reflective of modern valuations of ecosystems beyond pure utility (Jones *et al.*, 1996). For instance, Shields *et al.*, (1998) suggest that well-designed rehabilitation schemes of degraded physical habitat can yield major gains to *ecosystem recovery*

(demonstrated by a compositional shift from small cyprinids to larger centrarchids, catostomids and ictalurids), rather than simply to fish stocks or productive capacity. Cairns (1995) recognises the role of a no-net-loss policy, but he does so only as an interim phase between what exists at present and future rehabilitation. He argues that ecological rehabilitation must outpace degradation simply to keep pace with burgeoning human population growth and the commensurate pressure on ecosystems. The second criticism is that productive capacity is itself vague (Jones *et al.*, 1996), especially when assigning quantifiable value to habitat. For example, DFO (1998) states that “a project which destroys or degrades a small area of high value habitat could be more harmful than one which impacts a larger area of habitat of very low value”. The implication is that fish habitat value and therefore HADD assessment is linked exclusively to production values, but not to ecosystem stability, especially over time. By extension, it could be suggested that habitat which produces few, rare species is less valuable than habitat which produces more and of abundant species. Furthermore, HADD assessment which results in nil alteration may not result in no-net-loss in the future, particularly if a system experiences chronic stress as a result of HADD-like conditions set in place prior to the establishment of HADD assessment and regulation. The ramifications to long term ecosystem health and production are obvious. Notwithstanding the anthropogenic bias potentially associated with HADD enforcement, in this study I take the larger view of fish habitat, and include an examination of all fishes, irrespective of their direct value to humans, as outlined within the *Fisheries Act*.

1.3 THE ROLE OF THIS STUDY - GENERAL HYPOTHESES

Ecosystem based management expects that resource managers examine the role of human impact on the natural world and its systems. Hierarchy theory provides an intellectual framework to examine ecosystem structure. The littoral zone is an important ecosystem, both from ecological and human use perspectives, which is susceptible to the effects of erosion. Rip rap shoreline armouring has been used as a *de facto* best management practice (BMP) to curb shoreline erosion, but has recently come under scrutiny regarding the legitimacy of this designation. By examining information embedded in resident fish assemblages associated with both armoured and unarmoured shorelines, I test the general null hypothesis:

H_0 Use of rip rap shoreline armouring does not affect fish habitat.

However, at the same time, the DFO decision framework discussed above suggests an alternative hypothesis as its default position:

H_1 Use of rip rap shoreline armouring constitutes a negative effect on fish habitat.

On the other hand, a two-tailed approach (Legendre and Legendre 1998; Norman and Streiner 1999) to the null hypothesis offers the second alternative hypothesis:

***H*₂ Use of rip rap shoreline armouring constitutes a positive effect on fish habitat.**

Positing null and alternative hypotheses generally seems to imply requirement for tests of statistical significance. However, I examine these general hypotheses from a variety of perspectives, about which I say more in chapter two.

CHAPTER 2
RESEARCH DESIGN, SCALE AND ASSESSMENT OF THE STUDY AREA
AND EVALUATION OF THE SAMPLE UNITS

GENERAL INTRODUCTION

This chapter is divided into three sections. In section one, I describe the research design and the basis for the associated analytical methods. In section two, I discuss the study area and critically evaluate it for its representativeness as a sample of the larger river ecosystem. In section three, I discuss and assess the sample units from which data were collected.

2.1 RESEARCH DESIGN

2.1.1 ASSUMPTIONS

There are three fundamental assumptions made within my research design. The first assumption is that an *in situ* field assessment is the most appropriate method for understanding the relationships between species and environment in any given ecological community. Although an experimental, laboratory-style simulation of a rip rap armoured shoreline could be undertaken which would facilitate univariate statistical testing methods, an artificial streambed is not necessarily a good analogue for natural ecosystems (Jones *et al.*, 2003). This assumption is supported by Jørgensen and Müller (2000) who state:

The entire ecological network plays a role for all the processes in an ecosystem. If we isolate a few components and their interacting processes by a laboratory or an *in situ experiment* [my emphasis], we will exclude the indirect effects of components interacting through the entire network.

As the indirect effects are often more dominant than the direct ones ...our experiments will not be capable to uncover the results of the relations as they are observed in nature (Jørgensen and Müller 2000 p.13).

The second assumption is that the best approach to the analysis of my data is found within the methods of numerical ecology. Legendre and Legendre (1998) divide ecology into theoretical and quantitative subsets. Within the latter set they make subsequent divisions, which include modelling, ecological statistics and numerical ecology. They define numerical ecology as the subset of quantitative ecology which uses numerical analytical methods to examine ecological data, primarily by community ecologists using multivariate methods. The third assumption is that one mechanism for measuring fish habitat is through analysis of the information contained in the resident fish assemblage (Karr 1981; Karr *et al.*, 1986; *sensu* ter Braak 1987; Fausch *et al.*, 1990).

2.1.2 STUDY DESIGN

2.1.2.1 STUDY TYPE - CONTROLLED PROSPECTIVE STUDY

Several research models are available for the design of ecological studies (Table 2.1) and broadly speaking, they are either manipulative or mensurative (Hurlburt 1984). While the inferential power of properly designed manipulative experiments is high, the explanatory properties of variables within a controlled prospective study should not be discounted (Jager and Looman 1987); this despite criticism attributed to the problem of the “cohort”. Norman and Streiner (1999) define the cohort where “two groups (or ‘cohorts’) of subjects are identified, one of which, by choice, luck or chance, has been exposed to the

clinical intervention or a putative causal agent, and another which has not.” The major disadvantage of a cohort study is that we cannot be sure that the subjects are otherwise the same, notwithstanding the treatment. Similarly, this concern may also hold true for mensurative studies. Although it might be fair to speculate that rip rap armoured sites were different from unarmoured sites prior to treatment, it is not certain that these sites were ecologically different *a priori*. Indeed there is the distinct possibility that ecologically irrelevant (i.e. socio-political) drivers forced treatment of some sites over others. Thus, Hurlburt’s (1984) recommendation that we engage in study replication is well advised; particularly when study explanatory power is not necessarily statistically inferential, based on analytical methodology (see below).

Table 2.1 Summary of ecological study types modified from Jager and Looman (1987) matched to Hurlburt’s (1984) study types (in parenthesis).

Study type	Design	Explanatory power
Experiments - (Manipulative)	controlled experiment with random assignment of treatments	high
Pure observational studies (Mensurative)	investigator has no control over data	low
Sample surveys (Mensurative)	a random sample population is described	low
Controlled prospective studies (Comparative mensurative)	researcher selects units in which variables expected to be explanatory are measured.	High but regarding measured variables only
Controlled retrospective studies (Mensurative)	history of a response is traced	potentially high

Both Hurlburt (1984) and Johnson (2002) advocate “metareplication” – that is the

replication of a study focus, but under varying conditions – as a means of enhancing the overall confidence in general findings. Following that suggestion, this study seeks to replicate other work, while offering its own unique contributions. Specific work on the impact of rip rap revetments on fish habitat has been carried out by Knight and Cooper (1991), Shields *et al.*, (1995b), Jennings *et al.*, (1996) and Shields *et al.*, (1998, 2000). These investigations have typically focussed on small, warm-water streams. The exception is Jennings *et al.*, (1996), who used a large scale approach over a single year. Although the current study investigates much the same phenomenon as its antecedents, it does so with a unique blend of circumstances, hitherto unexamined. In particular, I conducted a fine scale, intensive examination in the context of a large boreal river. The chief advantage of this type of study is that it may generate greater completeness in species composition, and greater accuracy in measuring environmental factors than in large scale studies (Gauch 1982a). As with previous studies, including this one, rip rap armouring is in place prior to, and temporally independent of, study commencement - thus forming two cohorts or site types (armoured and unarmoured), by independent allocation. Therefore, comparisons are made principally between site-type since any disruption to the measured variables associated with a series of removal protocols to create a before-and-after experimental model⁶ would far outweigh the explanatory qualities that hypothetical model might offer. On the other hand, since rip rap installation

6.

That is to say: if rip rap armoured sites were sampled and then stripped of armour and sampled again (the reverse for unarmoured sites), it would be unclear if results were due to the result of the alteration, or to the act of disturbance itself.

has occurred over a period of years, I examined rip rap age as an environmental variable (sec. 6.2.2.4). So, this study is strictly mensurative; it is a controlled prospective study focussing primarily on two site types.

2.1.2.2 STUDY SIZE - SMALL SCALE, FINE RESOLUTION STUDY

In chapter one (sec. 1.2.3), I indicated that many in-stream projects occur at a small scale and individually they cover only a very small percentage of the total length of river shoreline. To assess projects of this size, we require appropriately scaled analytical tools which themselves must first be evaluated for use at the appropriate scale (see chapter 4). Scale establishes the observational relationship among data and refers to the specific spatial and temporal dimensions on which observations are made. Observation scale is comprised of grain, interval and extent (Legendre and Legendre 1998). While grain describes the smallest spatial and temporal unit within an observation set (O'Neill and King 1998), extent refers to the parameters, or total limits of the observation set at the established grain. Interval refers to the distance between sample sites (Legendre and Legendre 1998). As an example, sampling might include: *500 m of shoreline, from May to October, 2002-2004* [extent] *every 50m* [interval] *using 1m² plots once per week* [grain].

Hierarchy theory contains four features necessary to understand ecosystem linkages and dynamics (Jensen *et al.*, 2001 citing Allen *et al.*, 1984; O'Neill *et al.*, 1986) and for the consideration of study scale. 1) The whole-part duality recognises that every component

of a given ecosystem is simultaneously both a whole and a part depending upon the focal level at which it is assessed. From this tenet, it is clear that 2) “patterns, processes and their interactions can be defined at multiple spatial and temporal scales” (Jensen *et al.*, 2001). Meaningful observation requires scales appropriate to the issue under investigation. Consequently, 3), no single scale is appropriate for all purposes. Lastly, 4) the definition of the ecological hierarchy [assignment of the N levels] must be determined by critical issues (e.g. policy) within the ecosystem (Jensen *et al.*, 2001).

With respect to this last point, HADD policy deals with fish habitat on a project basis. Consequently, the focal level (N level) for this project is scaled to the size of the “impacted” site. Clearly this area is part of a much larger riverine ecosystem (N+1 level), while sample units (N-1 level) are a subset of the study area.

Therefore, although specific discussion of the study area and the sampling units are discussed below (secs. 2.2 - 2.3), it is sufficient to say here that the study occurs at fine scale. Similarly, sampling frequency (sec. 3.2) was conducted at a rate which is conceptually consistent with a fine-scale study.

2.1.3 MULTIVARIATE APPROACHES TO ECOLOGICAL DATA

This investigation fits within the discipline of environmental science, inasmuch as I examine the ramifications of human alteration to ecosystems. Nevertheless, it is essentially an ecological study, where I examine the interactions of biotic and abiotic

components within an ecosystem. The data collected from sample units during 2002-2004 resulted in varying abundances of 28 species of fishes (Appendix A) along with multiple observations of several environmental variables (chapter 6). Multivariate analysis of data sets is appropriate when species or object variables are minimally in the 10-15 range (Gauch 1982*a*). Indeed, ecological investigations are well served by multivariate methods given that most ecological data are multivariate in nature; where each sampling unit displays numerous properties which may influence data (Jongman 1987). Furthermore, rarely do the all the variables in an ecological study meet the assumptions of independence necessary to utilise properly univariate methods, sometimes resulting in spatial autocorrelation among objects (i.e sample sites - see sec 2.3.3.4.1) or multicollinearity among descriptors (e.g. water clarity and macrophyte development). Typically however, lack of full independence is more problematic for determining degrees of freedom used to determine statistical significance (Legendre and Legendre 1998) than it is for understanding objects conceptually, based on the many descriptors (perhaps some related and others perhaps not) associated with any given object. Thus, some community ecologists have relied on recognising and interpreting the informal patterns generated by multivariate methods (Gauch 1982*a*) rather than the standard tests of significance associated with univariate statistical methods. Therefore, multivariate techniques are the correct analytical approach for this project.

The general techniques for handling ecological data fall into three categories: 1) direct gradient analysis (regression analysis); 2) indirect gradient analysis (ordination methods);

and 3) classification (cluster analysis) (Jongman 1987). Irrespective of method, all multivariate methods seek to find shared information (redundancy) among variables, which can be summarised and interpreted. Here, *redundancy* indicates similarity among variables and represents the *coordinated* response of species (Gauch 1982a; Gauch 1982b). In contrast, *noise* (including outliers) is not easily interpreted, and it represents a sample of species composition that has low similarity and is *not coordinated* to either other samples or, to all other sampled species (Gauch 1982a). Unlike the linear regression model which looks for mathematical prediction of the dependent variable according to a specified value of the independent variable (i.e. for a given x there is a predictable y), most multivariate methods seek to illuminate the coordinated (i.e. correlation) response among variables. While these methods do not attempt to provide causal certainties, the inherent structure elucidated by multivariate techniques cannot be discounted.

2.1.3.1 DIRECT GRADIENT ANALYSIS (REGRESSION ANALYSIS)

We understand an environmental gradient as an identifiable length of a definable environmental unit. Direct gradient analysis (i.e. multiple and linear regression analysis as well as the multivariate canonical methods, Legendre and Legendre (1998)) is simply the graphing of species composition and abundance along this gradient. This method is considered a specific technique for assessing the direct relationship between species and a measured [and understood, *a priori*, to be ecologically relevant] environmental variable(s) (Gauch 1982a; ter Braak 1987). While direct gradient analysis has the

advantages of simplicity and ease of interpretation, the primary difficulty with its use is understanding the environmental variable itself. The unpredictable use of physical habitat by any given species makes it difficult to assign explicit species' response to any specific environmental variable by direct means. Secondly, when using direct gradient analysis, the investigator is never certain that all, or indeed any, of the ecologically significant environmental features have been measured. The practical ramifications include the possible collection of meaningless, or at least diluted, data and the associated waste of time and money. Furthermore, unless active management for a specific species is the aim of a project, it is presumably more valuable to understand the effect of an environmental variable on an assemblage, rather than on an individual species (ter Braak 1987). This is an important point with respect to this study, since I examined the impact of rip rap on fish habitat rather than on the stocks of one or several select species. Nevertheless, direct gradient analysis is used within this study when species composition information is constrained by a series of measured environmental variables hypothesised *a priori* to be important components of fish habitat in general (chapter 6), and after indirect analysis has been utilised to find initial trends in the data.

2.1.3.2 INDIRECT GRADIENT ANALYSIS (ORDINATION)

In those circumstances where environmental variables are not clear, cannot be measured directly, or if they are to be "*assayed or scaled in terms of their impact on plant and animal communities*, [my emphasis] the appropriate methods include ordination and classification" (Gauch 1982*a*). Ordination is a collective term used to describe the

multivariate methods of plotting multidimensional data in reduced (usually two or three dimension) space. Typically, data points are the aggregate species richness and abundance values (i.e. descriptors) collected from the sample unit, although ordination can also be performed using object (site) information. Each ordination technique carries slightly different emphasis and consequently each method provides a particular option for data analysis, depending upon the intent of the investigation. Regardless of method, all ordination techniques situate the data in such a way as sites that are similar are plotted close together while those that are different are positioned far apart (ter Braak 1987; Legendre and Legendre 1998). All ordination methods are predicated on the assumption that the ecological significance of an environmental feature can be derived from the species assemblage recorded at the sample site. This assumption is based on the idea that the data follow some underlying structure, where there is a hitherto uncovered, yet comprehensible, response to environmental conditions. Ordination attempts to recover this latent structure (ter Braak 1987). Since ordination assumes that the species assemblage is a more stable indicator of the effect of an environmental variable on a general habitat than is any individual species, ordination methods are not prone to the criticisms levied at direct gradient analysis (above). On the other hand, ordination methods typically offer a general picture and *may not* provide the resolution necessary to facilitate final interpretation and, ultimately, decisions about specific environmental data. This study examines the role of a known environmental variable (rip rap), but one which has no clear *a priori* analogue that can be measured. Therefore, ordination techniques form the primary analytical methods.

2.1.3.2.1 SELECTING THE ORDINATION METHOD

There are several ordination techniques designed to illustrate differences between sample sites according to species composition information. Among these approaches, principal component analysis (PCA) and correspondence analysis (CA) are both well suited to the analysis of ecological data sets⁷ (Legendre and Legendre 1998). Nevertheless, there are differences between the two methods. I outline briefly here those differences which are of importance in the choice of analytical method in subsequent chapters.

Arch effect

While both PCA and CA methods are similar to one another in procedure and closely related in end product, PCA is prone to an involuted arch effect (in this case, data are plotted in a horseshoe shape with the feet turned inwards rather than outwards) whereas CA is only susceptible to the simple arch effect (Gauch 1982a; Jongman *et al.*, 1987). When the arch effect occurs, objects which are actually far apart (and therefore dissimilar) are plotted close together, rendering metric concepts of distance meaningless and creating the appearance of object similarity where none exists, resulting in confusion during interpretation. The involuted arch makes this data appear even more similar than it would in under a simple arch effect. The arch effect can be eliminated from CA, but not from PCA, through the process of detrending (thus detrended correspondence

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Gauch (1982a), suggests that CA may be the better method for ecological data sets. However, this assessment is largely made on the basis of the process of detrending and the subsequent DCA method – see text for discussion.

analysis - DCA (Gauch1982a; Jongman *et al.*, 1987). The downside to DCA may be the unwanted loss of important ecological information (Pielou 1982; Jongman *et al.*, 1987; Legendre and Legendre 1998). Indeed, theoretical support for detrended analysis still appears to be wanting (Legendre and Legendre 1998) and detrending may not be the panacea it was first thought to be.

Rare species and data set heterogeneity

The arch effect aside, CA does not handle rare species well, placing them at sites with low abundances and in distinctive positions. This limitation is usually handled by deleting rare species data prior to analysis (Gauch1982a). However, if the retention of rare species within the data set contributes to our understanding of the environment⁸, then PCA is the more appropriate method. Furthermore, PCA is preferred over DCA when data set heterogeneity is low (Gauch 1982a). In a fine grain, small scale study, we might expect rare species to contribute to data set heterogeneity necessary to show differences between sites.

Distortion on the second axis

Lastly, CA is susceptible to the plotted information on the second axis becoming

8

One difficulty of a small scale study is the treatment of rare species. I contend that species which are rare to the watershed should be removed since they share little coordinated response with other species. On the other hand, species that are rare to the sample area but not to the watershed, help define and distinguish individual sample units from one other.

distorted; a distortion that can continue into the remaining axes, but which is most problematic on this particular axis (Gauch 1982a). If the environment is better understood by examination of the output on the second axis, CA may compromise this important information. Furthermore, if much of the variation of the uncorrelated data can be described in the first few variables⁹, then PCA is well suited to the data (Manly 1994). Overall, the conditions of this study suggest that PCA should be the default ordination method (unless object constellations demonstrate signs of the arch effect) with any corroboration sought through CA, rather than vice versa.

2.1.3.3 CLASSIFICATION AND CLUSTER ANALYSIS

Classification (hierarchical) and cluster analysis (non-hierarchical) are specific grouping methodologies. These techniques attempt to find similarities between some objects (or alternatively, species) while simultaneously, creating separation among other objects / species (Legendre and Legendre 1998). Clustering is most appropriate either when the data can be demonstrated to be discontinuous or where there is a “practical need to divide a continuous swarm of objects into groups” (Legendre and Legendre 1998; also Gauch 1982a). Otherwise, ordination techniques usually suffice. Clustering is used to examine (dis)similarity among study areas, below (sec. 2.2.41.2).

⁹

That is to say if the first few PCA axes contain a large aggregated portion of explained variance.

2.2 THE STUDY AREA

2.2.1 TERMINOLOGY

Several meanings have been applied to the term sample (see Pielou 1982). In general, I observe the following usage: *study area* means the general location in which the study occurs whereas the *sample site* or *sample unit* refers to one of several units identified within the study area on which measurements and observations are made. *Sample* is primarily used in the verb form, indicating the process of collecting data and secondarily as a noun, indicating a particular measurement or observation. Together, the collection of data from various, or all sample units within the sample area comprises a data set.

2.2.2 SELECTING THE STUDY AREA

Choosing a sample area presents a unique set of challenges. On the one hand, the selection of any random area which contains the conditions of research interest is presumably as likely to include meaningful information as any other sampling area. Indeed, this is a basic assumption on which statistical testing relies. On the other hand, any given sampling area may consist of unique qualities that are uncharacteristic of the larger ecosystem to which the selected area belongs. This concern *may* be particularly germane when the selected study area has been allocated some treatment not endemic to all other parts of an ecosystem. Therefore, some criteria must accompany the choice of sample area. I selected one primary sample area¹⁰ for three interconnected reasons. This

¹⁰

Two secondary study areas were used as part of the assessment of this sample area. They are described below (sec 2.2.4).

part of the chapter 1) outlines the selection criteria; 2) describes the primary study area; and 3) discusses a series of tests which assess study-area-representativeness of the larger ecosystem.

2.2.2.1 STUDY AREA SELECTION CRITERIA

2.2.2.1.1 Study scale relevance

Application of rip rap armouring can occur in small doses and in specific areas of interest (e.g. urban shorelines, cottage development shorelines, marinas and boat launches, etc.) where aquatic ecological considerations may not have played any role in the decision to allocate such armouring. In these circumstances, impact assessment should be conducted at a scale commensurable with the area of impact – generally, that is at a small scale.

This intuitive approach is supported by Mathews and Robinson (1998) who found that a large scale analysis did not explain faunal composition at smaller scales and that local assemblages were best explained by local factors or at a fine scale. Consider: If rip rap armouring has an effect on fish habitat, it is more likely that the resident fish assemblage will be affected than is the non-resident assemblage. However, unless the variation in species composition between site types is dramatic, large scale studies may overwhelm the nuances which occur at the scale of the impacted site; the reasons for which are listed below.

2.2.2.1.2 Inherent structural change along the river gradient

The variation between site types at the focal (N) level may be overshadowed by larger

shifts in species composition according to either of two views about inherent structural change along the river gradient. Vannote *et al.*, (1980) developed the river continuum concept to explain the changing balance between biotic (especially macroinvertebrates) and abiotic interactions along the downstream river gradient. They assume that rivers exhibit a (more or less) predictable biological community as a function of location along a river's length. They contend that downstream communities become increasingly well-adapted to securing energy lost due to the capture inefficiencies of the upstream communities in synchronicity with concomitant changes in adjacent land morphology, resulting in successive changes to downstream communities (i.e. upstream communities derive energy predominantly from allochthonous sources while downstream communities rely chiefly on autochthonous sources). In somewhat similar fashion, Rice *et al.*, (2001) developed the link discontinuity concept to describe the "discontinuous nature of lotic ecosystem gradients". They argue the importance of hydrological and sedimentological regimes in arranging unique community structure at "intermediate scales" (10^0 - 10^2 kms). Although both Vannote *et al.*, (1980) and Rice *et al.*, (2001) focussed on invertebrate community structure, work by Santos *et al.* (2004) suggests that community structure from other trophic levels may behave in a similar manner. Indeed, they found that distance-from-river-source was among the primary variables explaining differences in river fish assemblages; findings that are consistent with basic riverine ecological theory (see Townsend, 1980). In addition to these inherent characteristics of rivers, Santos *et al.*, (2004) also reported that river obstructions, such as dams, had a significant impact upon fish assemblage structure; results in keeping with Rice *et al.*, (2001) who

found the link discontinuity concept applicable to both regulated and unregulated river systems. Together, or apart, any of these perspectives suggest that potential sample areas substantially distant from one another may possess different species composition simply due to their position along the length of the river. Presumably the inherent variation of two or more discontinuous sample areas would overshadow the smaller variation expected at the local level.

2.2.2.1.3 Adjacent land use

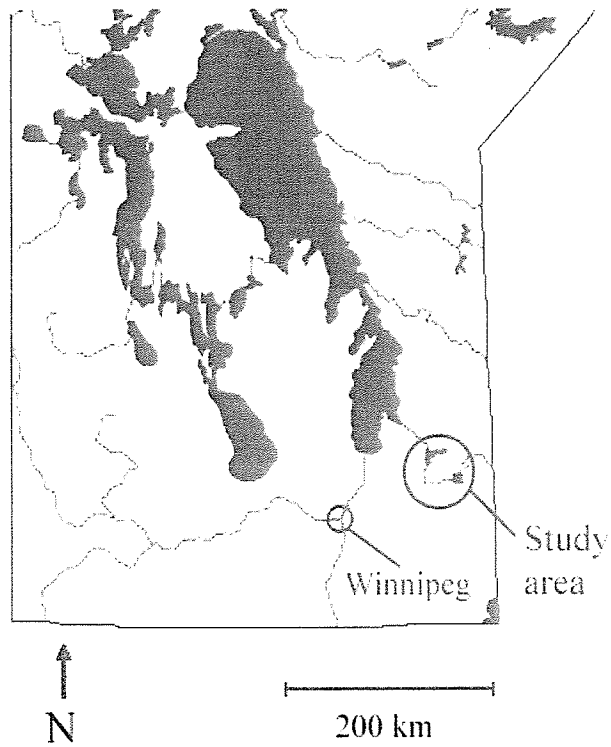
In a related point, land use patterns have been shown to influence fish communities in adjacent aquatic systems. Schlosser (1991) argues that boundaries between the terrestrial and the aquatic environments are important sources for nutrients and organic matter. As a result, anthropogenic alterations to landscape (deforestation, agriculture, urbanisation, etc.) can result in significant alterations to fish habitat and thus to fish communities in the adjacent aquatic system. For instance, O'Neill *et al.*, (1997) reported that water quality suffers with a decrease in natural vegetation on adjacent land. Richards *et al.*, (1996), found that while channel morphology was the greatest variable for explaining macroinvertebrate assemblages, where homogeneous channels exist, the effects of land-use is the next most important variable. Roth *et al.*, (1996), found that land use affected stream ecosystem integrity when assessed by both an Index of Biotic Integrity (IBI) and a Habitat index (HI). They reported significant negative correlations between stream integrity and agricultural land use, and positive correlations between forested cover and index scores, particularly at the catchment scale rather than at local levels. Therefore,

study areas which contain heterogeneous land use types may generate confounded results.

2.2.2.2 SELECTION OF STUDY AREA

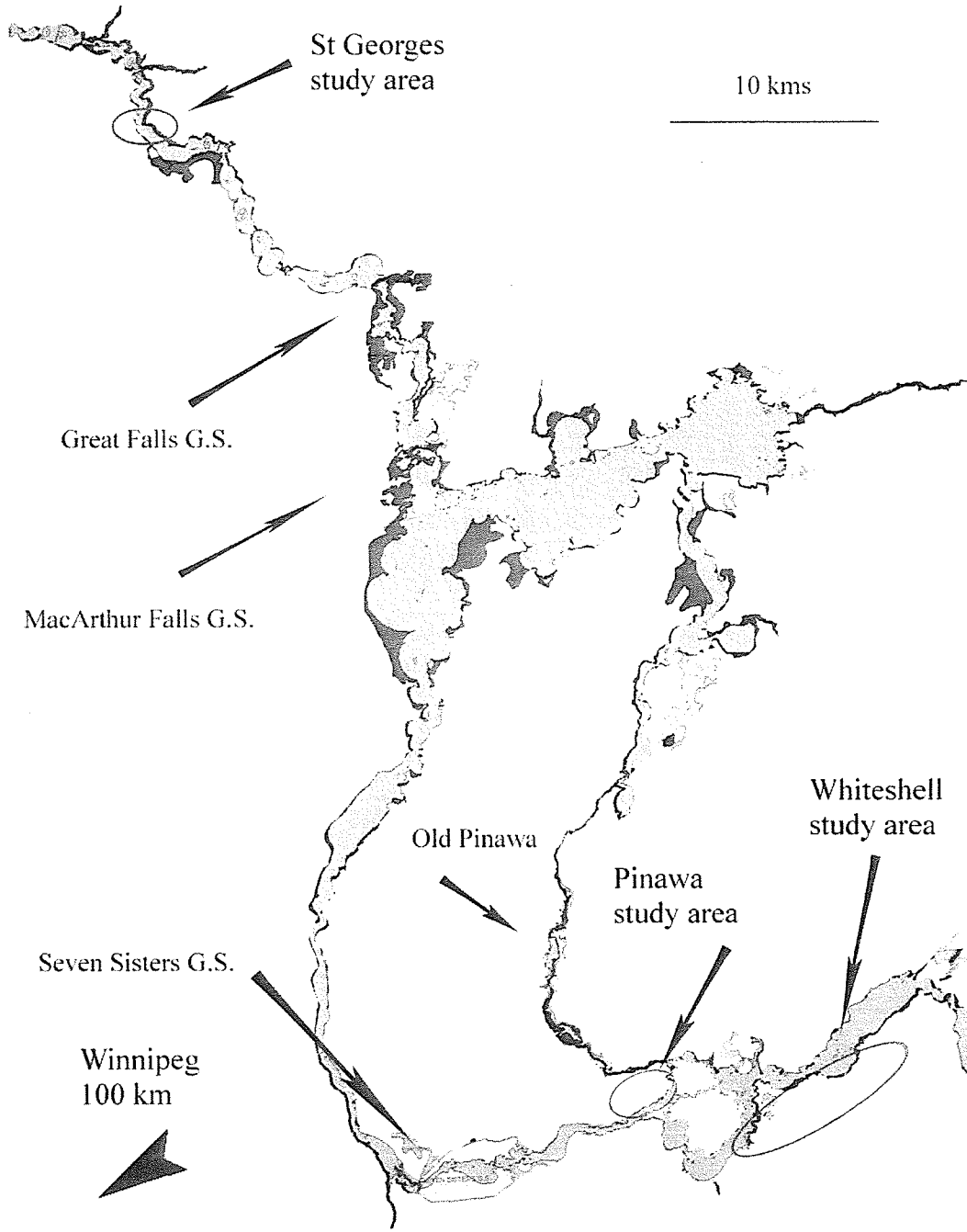
In accordance with the points outlined above, in 2002, I selected a single study area at Pinawa, MB (Figs 2.1a & 2.1b) where the study extent matched the parameters of the impact area and where land use (residential) was essentially homogeneous. A single study area was selected since no pair or multiple set of impacted areas were found proximate to one another with similar, relatively homogeneous, land use. In 2004, two additional study areas were sampled, primarily to determine the representativeness of the fish assemblage at the primary study area compared with a larger segment of the river. One of these two areas was selected slightly upstream of the Pinawa study area at Whiteshell Provincial Park (Fig 2.1b). Although it is proximate to Pinawa (~5-15 km upstream), land use within the park is varied, being a mixture of recreational, residential, campground, and wilderness activity; thus making it different from that at Pinawa. While this study area contains both armoured and unarmoured shorelines, I selected it primarily as a point of comparison with the Pinawa study area. That is, notwithstanding variation in adjacent land use between the two study areas, I used Whiteshell to determine whether or not the Pinawa study area fish community was representative of the general area. The second additional study area was selected at a distance of approximately one order of magnitude ($N+1$ level) greater than the mean distance from Pinawa to Whiteshell. This study area is located about 70-80 kms downstream from Pinawa at St Georges, MB (Fig

Fig 2.1a - Physical map of southern Manitoba indicating the general study area.



2.1b). Here, land use is rural-residential, recreational and agricultural. Rationale for selecting this study area was the same as for selecting Whiteshell, but with emphasis on determining fish species composition for a much larger reach of the middle part (i.e. non headwaters; non-mouth / estuary) of the river.

Fig 2.1b Map of Pinawa, Whiteshell Provincial Park and St Georges, MB study areas - designated by ovals. Dark shading indicates areas most susceptible to shoreline erosion. Approximate scale, 1:500,000. Modified from MB Hydro "Discussion Soils Map" Sept. 8, 2003.



2.2.3 STUDY AREA DESCRIPTION

The Winnipeg River basin is >107,000 km² (StatsCan 2004) and the river itself can be classified as a Great River using criteria set out by Simon and Emery (1995) and Simon and Sanders (1999). Physiography of the river basin is predominantly Precambrian Shield landscape. In general, soil type varies depending upon topography and can include peat, sand and clay dominated soils. Within the Pinawa study area, substrate is predominantly classified as Zone 4; land described as including steep slopes, exposed bedrock, sand dunes and wetlands (MB Water Stewardship 2005). Interspersed within the study area however, are substantial pockets of Zone 1 land, primarily located adjacent to major water bodies (MB Water Stewardship 2005). Zone 1 is highly productive agricultural land (MB Water Stewardship 2005) – in this case, clay based substrate overlain with topsoil. Accordingly, vegetative patterns match those associated with this range of soil types and relief, and include both coniferous and mixed coniferous and deciduous forests. The Winnipeg river is fed by substantial (683 mm yr⁻¹) annual precipitation (StatsCan 2004), contributing to a mean monthly flow of 983 m³ sec⁻¹ (Lake Winnipeg Stewardship Board 2005). The river may be considered meso-oligotrophic with (1994-2001) mean total phosphorus (TP) reported at 788 tonnes yr⁻¹ – [TP] not given (Bourne *et al.*, 2003). Local water chemistry measurements from the Pinawa area are discussed in chapter six.

The Manitoba portion of the Winnipeg River has been dammed in six locations for hydroelectric power generation and has experienced at least two discrete periods of

hydrological impact in the Pinawa study area as a result. The first event occurred at the turn of the 20th century, when river water was directed into the Pinawa Channel by a series of in-stream dams to supply Manitoba's newly constructed (opened 1906) and first year-round hydro electric facility at (Old) Pinawa. As part of this initial development, the 1912 era inlets, known collectively at the time as "Loon River" (Lower Goldeneye Pass - Buchner 2000), were flooded creating a new through-channel on the west side of the newly formed Hind Island (Winnipeg River Power Survey 1913 map sheets 24 & 25).

The increasing energy demand within Manitoba following World War I and beyond, led to the construction of a new hydro electric installation at the Seven Sisters Falls site to augment output from (Old) Pinawa. The first stage of the Seven Sisters facility was constructed between 1929-31. Although it is unclear if river water levels changed significantly during this period, a second development phase which completed the facility between 1948-1952 included raising mean water levels of the forebay (Natalie Lake) by 2.4 m to present levels, in 1948. During this time, the site at (Old) Pinawa was closed (1951) and the water supply to the Pinawa channel was ostensibly blocked to increase flows to the Seven sisters facility (1952), essentially reversing the 1906 flow alteration. Since 1952, water levels in the study area have operated at a new elevated equilibrium (1956-2007) water level range, 274-276.8m (except for a 1979-81 dam repair draw down to \approx 273m). The 1948-52 flooding created a new littoral zone on this stretch of the shoreline, most of which is clay substrate and susceptible to erosion. The development of the planned town of Pinawa (built by Atomic Energy of Canada Ltd in

1963) on the shores of the Winnipeg River, brought an urban presence to the area and with it, heightened awareness of shoreline erosion. Such public concern prompted the armouring of several stretches of riverbank within the town site with rip rap stone armouring. Local rip rap shoreline protection began in the 1970's and has been installed by Manitoba Hydro as recently as 2001.

I hypothesised that significant parameters associated with residential land use within the Pinawa study area were: 1) effects of storm sewer drains into the river course, 2) fertilizer and herbicide effluent from an upstream golf course and urban lawn care regimes, and 3) a regular (weekly) municipal lawn mowing regime which extends to within a small distance (<1m) of the shoreline in the green belt adjacent to the shoreline along the entire length of the sampling area. The study area covers approximately 2.5 km of urban shoreline, with varying river channel widths (200-700m), morphology and orientation. Individual sample site descriptors are listed in Appendix B.

The Whiteshell Provincial Park study area is relatively close (~5-15 km upstream) to the Pinawa site but experiences different land use. For example, although the Park is heavily populated during the summer, all unarmoured sample sites were substantially less exposed to shoreline mowing / vegetation control than in the Pinawa sample area and none of the sites were subject to the systemic paved street / storm sewer infrastructure, or permanent human population bias found in the Pinawa site. Individual sample site descriptors are listed in Appendix B.

The St Georges study area is approximately 70 km downstream from the Pinawa sample site. Three hydro electric dams are situated between St Georges and the Pinawa / Whiteshell sites. They are located at Seven Sisters Falls, MacArthur Falls and Great Falls (Fig 2.1b). This region is a rural-urban mix, with a blend of agricultural land and a less well developed urban infrastructure than in the Pinawa area. Individual sample site descriptors are listed in Appendix B.

In general, rip rap armoured sites in both St Georges and Whiteshell occurred where road washout was otherwise imminent (especially sites 17-18 and 25-26) or near boat launches, boat ramps and other human use infrastructure. This distinction is not so obviously the case in Pinawa, where general bank stabilisation appears to have been the rationale for application.

2.2.4 STUDY AREA TESTING

For the purpose of determining the representativeness of the Pinawa study area to the larger river ecosystem, fish species composition information was compared between the three study areas. Both the link discontinuity concept (Rice *et al.*, 2001) and the river continuum concept (Vannote *et al.*, 1980) might suggest that the fish assemblage from the three study areas are discontinuous. If such is the case, then it might not be tenable to infer any trend determined by analysis of the Pinawa-only-based data to the larger river ecosystem. Conversely, if the total composite species composition information is not clearly segregated by study area, then the fish assemblage of the Pinawa sample area

could be considered representative of the Winnipeg River (but not headwaters or estuary) ecosystem. From this logic, I defined terms *a priori*, that if the study areas were to be determined as discontinuous, then fish species composition information within each area would be most similar to itself and most dissimilar to information from the other study areas. On the other hand, if sites within study areas showed similarity across study areas, then study areas *would not* be considered discontinuous, and therefore, the Pinawa study area could be considered an analogue for a large reach of the Winnipeg River.

2.2.4.1 Methods

2.2.4.1.1 Field Methods

Analysis of study area (dis)similarity was conducted on littoral zone fish species composition information, derived from samples taken during the 2004 field season. Six sample sites (3 armoured; 3 unarmoured) were selected from each of the Whiteshell and St Georges study areas (for site description, see Appendix B; sites 17-28). Fishes were captured at these sites, as well as at 16 Pinawa study area sites (for site description, see Appendix B; sites 1-16) using a modified beach seine, described below (sec.3.1.1.1).

2.2.4.1.2 Analytical Methods

Analysis of species composition data followed two courses; one assumed the study areas were discontinuous and the other assumed they were not. In the first case, on the basis of both the link discontinuity and river continuum concepts, a data set comprised of the 2004 fish species composition information from all study areas were compared using

clustering analytical methods (Legendre and Legendre 1998). While they caution that not all ecological data should be handled with clustering methods, Legendre and Legendre (1998) recognise that discontinuous data is well separated by these techniques. Pielou (1982) adds that procedurally, nonhierarchical [i.e. cluster] methods should be used prior to hierarchical methods. In the second case, distribution information (Stewart and Watkinson 2004) suggests the Pinawa study area fish assemblage is not dissimilar to the fish assemblage of the larger river ecosystem. Indeed, specific species distribution data (Stewart and Watkinson 2004) suggests that the Winnipeg River fish assemblage could be quite homogeneous. Therefore, the 2004 data set was analysed through both classification and ordination methods (Legendre and Legendre 1998).

Cluster analysis

Fish species composition data from the three study areas (Table 2.2) were assembled into a single data matrix. Site singletons were removed, resulting in a 19 species x 28 site data matrix. All data were natural log transformed ($\ln+1$) prior to analysis. Using a non-hierarchical technique (fuzzy C-Means clustering methodology, clusters formed by centroids of objects; coefficient of fuzziness = 1.25), all data were obliged to conform to one of three groups (matching the number of study areas).

Classification

Under the alternative assumption, that study sites are *not* discontinuous, classification (hierarchical grouping) was used to determine the degree of (dis)similarity between all

Table 2.2 Summary of 2004 fish species composition (by common name), by sample site and study area. For study area, (1) = Pinawa; (2) = Whiteshell; 3= St Georges.

Species Composition (by site)				Presence / Absence (by study area)	Study Area		
Study Area	Site #	Site Abundance	Site Richness	Species present	(1)	(2)	(3)
(1)	13	3642	11	Blackchin shiner	Y	--	--
(1)	3	1248	12	Blacknose shiner	Y	Y	--
(1)	9	971	9	Brook stickleback	Y	Y	Y
(2)	18	940	12	Emerald shiner	Y	Y	Y
(2)	21	683	8	Fathead minnow	Y	Y	--
(2)	20	643	14	Golden shiner	Y	Y	--
(2)	19	581	13	Iowa Darter	Y	Y	Y
(1)	2	568	8	Johnny darter	Y	Y	Y
(1)	8	457	8	Logperch	Y	--	Y
(1)	15	394	7	Mimic shiner	Y	Y	Y
(1)	6	376	12	Northern pike	Y	Y	Y
(1)	1	372	12	Rainbow trout	Y	Y	--
(1)	14	303	12	River darter	--	Y	--
(3)	23	285	10	Rock bass	Y	Y	--
(1)	16	277	7	Sculpin spp.	--	Y	Y
(1)	4	274	8	Spottail shiner	Y	Y	Y
(1)	10	271	7	Troutperch	--	--	Y
(3)	25	269	9	unknown fry	Y	Y	Y
(3)	26	264	10	Weed shiner	Y	Y	Y
(1)	5	226	8	White sucker	Y	Y	Y
(2)	17	226	10	Yellow perch	Y	Y	Y
(3)	27	210	8				
(1)	7	157	10	Total Richness	19	19	14
(3)	28	135	9				

Table 2.2 (con't)

Study	Site	Site	Site
Area	#	Abundance	Richness
(1)	12	104	10
(3)	24	90	9
(2)	22	43	10
(1)	11	13	5

sample sites within all study areas. Average link classification was performed on the same data matrix used for cluster analysis.

2.2.4.2 Results and Discussion

2.2.4.2.1 Cluster analysis

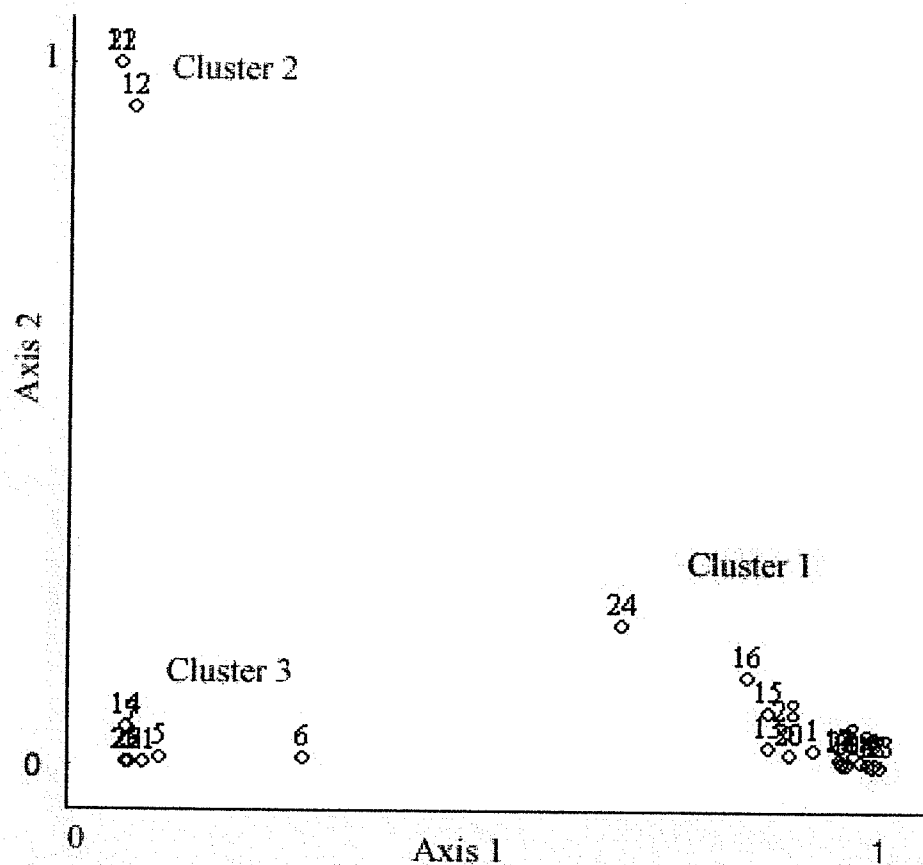
Results of the analysis indicate that although distinctive clusters form from the sample site data when forced to do so (for separation coefficients see Table 2.3), there is strong homogeneity among study areas. A scatterplot of the clusters (Fig. 2.2) shows that cluster

Table 2.3 Pairwise separation coefficients for clusters.

Cluster	Coefficient		
1	0.0000	3.0198	1.6332
2	3.0198	0.0000	3.5109
3	1.6332	3.5109	0.0000

one contains 64.3% of all sample sites, separated out on the first axis. More importantly, the among-cluster balance of sample sites within each study area is undeniably similar. Indeed, 62.5% of the Pinawa sample sites, and 66.7% of both the Whiteshell and St Georges sample sites were grouped together to form the first cluster (Table 2.4). In and

Fig 2.2. Non-hierarchical cluster analysis of sample sites using 2004 field season fish species composition data. Scatterplot shows 64.3% of sites from all study areas grouped into cluster one, with equal site contribution balance from each study area. Clusters two and three show similar balance.



of itself, this large cluster suggests that the study areas are not discontinuous. Clusters two and three show similar balance (Table 2.4).

Table 2.4 Cluster membership and distribution of 28 sample sites across study areas.

Cluster membership	Object (sample site)	Study area	% of total number of sites per study area
1	1 2 3 4 8 9 10 13 15 16	Pinawa	62.5
	17 18 19 20	Whiteshell	66.7
	23 24 27 28	St Georges	66.7
2	11 12	Pinawa	12.5
	22	Whiteshell	16.6
3	5 6 7 14	Pinawa	25
	21	Whiteshell	16.6
	25 26	St Georges	33.3

2.2.4.2.2 Classification

Since cluster analysis did not show dissimilarity among the study areas, it was expected that this result would be reinforced through a classification analysis and that hierarchical similarity measures would be correspondingly high. Cophenetic correlation measures similarity between the resemblance matrix and the cluster result, where a perfect cophenetic correlation = 1, (Legendre and Legendre 1998). Cophenetic correlation was found to be 0.8299, indicating that all sample sites, irrespective of the study area to which they belong, are similar. Overall, St Georges sites were most similar to each other. Nevertheless, the sites with low dissimilarity listed among the first cycle in the output (24-28; 25-26; 27-24/28; 2-23) include those found at Pinawa and St Georges (sites 2 and

23) - where overall distances between sites were the greatest (i.e. 70 km). What is most interesting is that these two sites were more similar to each other than were sample sites 8 and 9 – both of which are from the Pinawa study group and are <25 m apart from one another - see Fig 2.3 and Table 2.4 for concordance of sample site to study area.

Fig 2.3. Hierarchical analysis of study areas using 2004 field season fish species (n=19) composition data from 28 sample sites. Dendrogram shows low dissimilarity between sample sites among study areas. Sites which are most similar include those with both least (sites 8 & 9) and greatest (sites 2 & 23) physical distance between them.

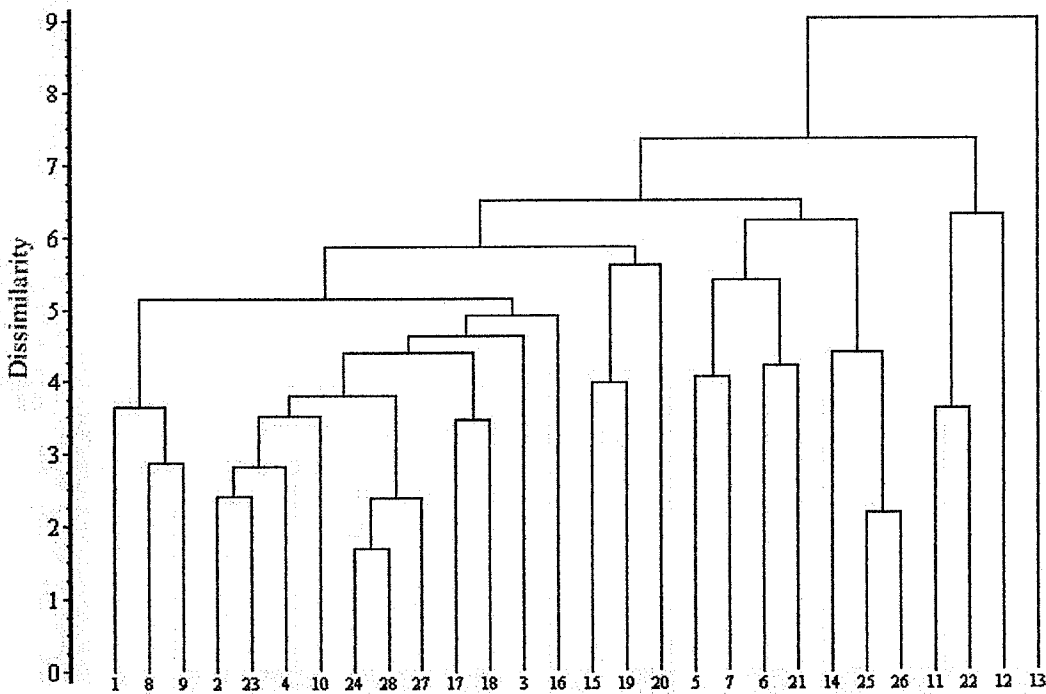


Figure 2.3 also reveals a pattern where mostly unarmoured sites occur on the left side of the dendrogram, while the majority of armoured sites and natural exposed bedrock sites

(sites 13-16) are positioned on the right side of the diagram (for site status, see Appendix B). Specifically, when the dendrogram is divided vertically between sites 15-16, the left side contains four armoured sites, one bedrock site and nine unarmoured sites. This contrasts with the right side of the figure, where eight sites are armoured, three are bedrock and three are unarmoured. This pattern broadly suggests that armouring status contributes to forming a distinctive shoreline type.

2.3 THE SAMPLE UNITS

In this section I: 1) review the principles of sample site selection methodology; 2) describe the specific method by which sample units were selected along the shoreline; 3) discuss the sample site allocation among “treatment” blocks, along with sample unit size, number and habitat-characteristics similarity; and 4) evaluate the sample units for spatial autocorrelation.

2.3.1 PRINCIPLES OF SAMPLE UNIT PLACEMENT

According to Gauch (1982a), sample units are selected on a landscape according to one of four methods: (1) completely at random; (2) through regularised placement according to map coordinates; (3) via preferential selection of a site considered typical and homogeneous; or (4) by stratified random sampling (where the sample area is subdivided according to important criteria and then each sub-section is sampled using random selection). These methods are discussed briefly here.

Random selection is the most suitable method for meeting the assumptions required by most statistical methods (Jager and Looman 1987)¹¹. However, Dutilleul (1993) argues that except in large scale studies with spatially homogeneous sample sites, random sampling should not be used. Thus, generally in ecological studies, stratified random sampling is preferred to random sampling (Jager and Looman 1987). Although the regularised placement method presumably produces unbiased samples, it can be tedious and time consuming. More importantly, this approach has no means of ensuring that sample sites contain the target treatment(s). The preferential selection method is typically a European approach and contravenes the basic assumption of random selection inherent in most statistical methods. Where non-inferential statistical analytical methods are utilised, (such as ordination - see Legendre and Legendre 1998), and therefore sample site independence is not a necessary (although still preferred) requirement, this method offers the knowledgeable researcher a high degree of efficiency in sample site selection. On the other hand, preferential selection leaves room for researcher bias. In contrast stratified random sampling combines the advantages of both random sampling (i.e. statistical validity) and systematic placement which distributes samples equitably among community and/ or treatment types. Although stratified random sampling is the preferred method, I contend that it can quickly become a *de facto* preferential selection method as the number of strata used to qualify sample sites, increases. Therefore, some analysis of the inherent bias associated with sample site selection must be made to qualify the

11

Norman and Streiner (1999), bemoan that real randomised sampling is rarely done, but rather random allocation is used instead.

experimental results and conclusions which may be derived largely upon choice of sample site selection method.

2.3.2 Sample Site Selection - General Method

In the Pinawa study area, primary sample sites were selected at the outset of the 2002 field season and were retained through the 2004 field season. To minimise potential researcher bias, an assistant unfamiliar with the goals of the project was directed to select 12 sample sites from within the study area (Table 2.5; Fig 2.4) using stratified random

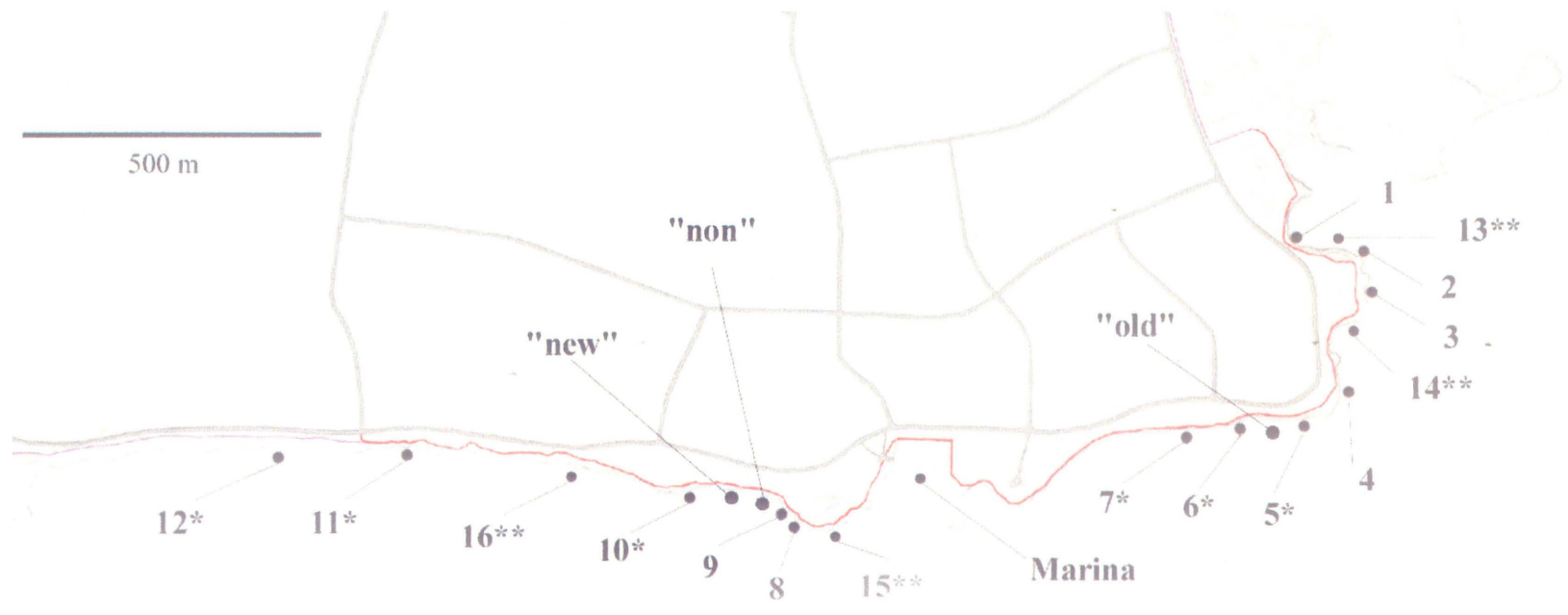
Table 2.5 Sample unit numbers 2002-4. Top row indicates sample site number. Bottom row indicates site type, where A= rip rap armoured; U = unarmoured; N= natural exposed bedrock. Sites are numbered sequentially from upstream to downstream, showing interspersed. Sites 1-12 selected in 2002; 13-16 were added in 2004.

Pinawa Sample Sites															
1	13	2	3	4	14	5	6	7	15	8	9	10	16	11	12
U	N	U	U	U	N	A	A	A	N	U	U	A	N	A	A

sampling. A “next most accessible location” criterion was applied, constrained by four strata: (1) sites were chosen from within groups of armoured and unarmoured stretches of shoreline. Dutilleul (1993), claims that for the assessment of environmental gradients under a treatment assignment, blocking “similar experimental units” is appropriate.

Thus, sample unit interspersed to within treatment blocks was somewhat predetermined according to the irregular placement of rip rap along this stretch of shoreline (Table 2.5 and the discussion below, sec. 2.3.3.4.1.2). (2) To prevent inconsistent impacts between

Fig 2.4. Map showing placement of sampling sites within the town site of Pinawa, MB. Armoured sites are indicated by a single asterisk (*); natural bedrock shoreline sites are indicated by a double asterisk (**). Unarmoured sites have site numbers with no asterisks. The red line traces the path of the Ironwood trail - part of the TransCanada Trail system. "New" "non" and "old" designations refer to additional sample sites from 2002 - see sec. 3.1.1.1.1.



sites on the results using boat access, shoreline accessibility to sample sites was required for sampling. Therefore, sites could not be shielded from the shore by dense, shrubby riparian vegetation. (3) Sample units were to be free of obvious snags and other underwater structure. It was felt that these objects would hamper sampling and confound the investigation at hand. (4) All sites were to exhibit similar substrate type (i.e. clay), notwithstanding cobble associated with the application of rip rap. By locating sample sites within this relatively uniform land-use type, concerns that adjacent land use may affect fish habitat more significantly than in-stream variables were minimised. All sites were located on the shoreline adjacent to the Ironwood Trail, a recreational riverside footpath which runs the length of the town of Pinawa (Fig 2.4).

2.3.3 SAMPLE SITE PARAMETERS

2.3.3.1 Sample shape and size

Gauch (1982a) argues that in ecology overall, rectangular sample plots (where the longest side is 2 to 4 times the length of the short side) “is ordinarily most accurate”. Following this model, all of the shoreline sites I laid out were nominally 3 x 5 m and measurement of all environmental variables were based on these parameters. However, each site was effectively 2.4m wide x 7.4m deep during fish capture; dimensions which still yielded an appropriate length to width ratio of approximately three ($7.4/2.4 = 3.1$). The reason site size was different from nominal during fish capture is that fish sampling gear width measured 2.4m. The sampling gear was extended to the back of the sample unit (i.e. 5m mark), perpendicular to the direction of sampling travel and then “swung” to

sample the plot, thus effectively adding the gear width to the sample unit length (5m + 2.4m = 7.4m). For a detailed description of fish sampling procedure, see below, (sec 3.1.1).

While the areal dimension of each sample site was homogeneous, sites varied volumetrically. I hypothesised that armouring might affect the slope of the littoral gradient by preventing eroding soils entering the water column and depositing on the littoral profundal, causing “flattening” of the littoral shelf. Indeed, the difference in slopes between armoured and unarmoured sites from the Pinawa study area (Table 2.6 and Appendix B) was significant ($P = 0.003$; $\alpha = 0.050$; $\beta = 0.927$) and supports this hypothesis. Consequently, unarmoured sites had a different shape than armoured sites with respect to the vertical gradient. The role of this environmental variable is discussed below (sec 6.2.2.3).

Table 2.6 Comparison of in-stream bank slope (0-5m from shore) for primary Pinawa sample sites.

Unarmoured		Armoured	
Site	Slope (°)	Site	Slope (°)
1	8.3	5	10.3
2	2.8	6	13.2
3	5.5	7	10.2
4	5.1	10	22.7
8	4.9	11	9.1
9	2.4	12	14.6

2.3.3.2 Sample site (Dis)similarity

2.3.3.2.1 Methods

In May of 2003, I compared the physical characteristics of the 12 main Pinawa sample sites using a habitat assessment protocol developed by Plafkin *et al.*, (1989). This comparison was intended to be an instrument for determining site (dis)similarity independent from using fish species composition information. Measured features include three strata of characteristics (Table 2.7). Individual metric scores (Table 2.7) were determined by a qualitative ranking of site specific observations against a numeric range of scores (eg. strata 1 metrics were scored as: 0-5 = "poor"; 6-10 = "fair"; 11-15 = "good" 16-20 = "excellent"). Total site scores (Table 2.7) were derived by summing individual site metric scores. A *t*-test was performed on site scores.

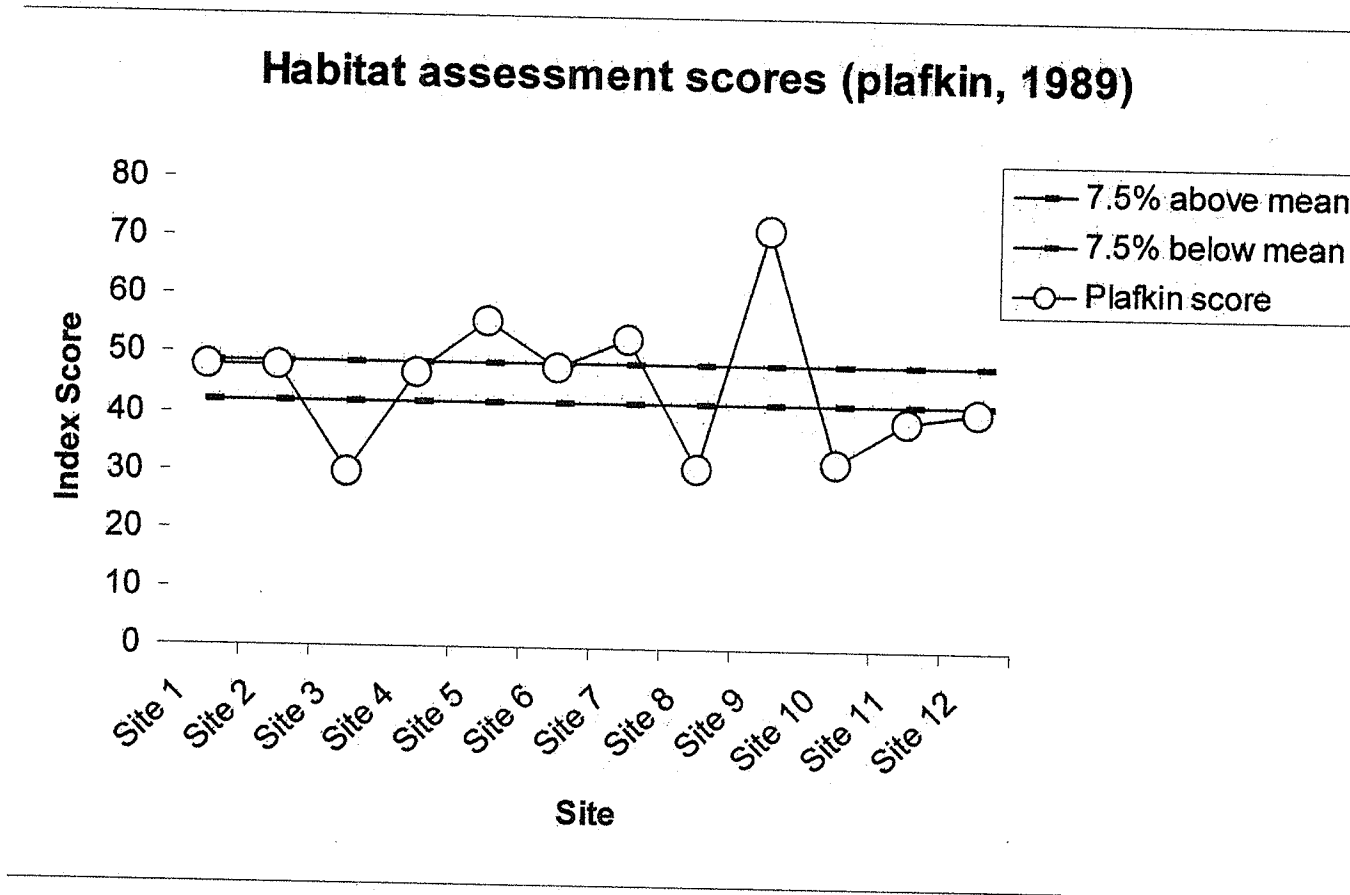
2.3.3.2.2 Results and Discussion

Scores for unarmoured sites (as a group) were not significantly different from scores from armoured sites ($p = 0.876$; $\alpha = 0.05$; $\beta = 0.05$). However, the power of the test was extremely low, meaning this result may be a false negative. Nevertheless, although unarmoured sites showed greater variability (range unarmoured sites in scores than did armoured sites (Fig 2.5), both site types showed roughly equal number of sites above and below the stated acceptable range of variance (15%), Plafkin *et al.*, (1989), suggesting that both site types were as likely as each other to provide high/low quality fish habitat. Furthermore, all site scores fell on or outside the accepted range of score variance suggesting these scores have little capacity to describe this habitat. In general, there is

Table 2.7. Physical habitat assessment scores for 12 main Pinawa sample sites using methods by Plafkin *et al.*, 1989.

Strata	Metric	Sites											
		U 1	U 2	U 3	U 4	A 5	A 6	A 7	U 8	U 9	A 10	A 11	A 12
1	substrate and available cover	10	11	5	0	11	11	6	6	16	0	5	0
1	embeddedness	11	11	11	16	16	16	16	11	11	16	16	11
1	flow	0	0	0	0	0	0	0	0	0	0	0	0
2	channel alteration	3	0	0	3	3	0	3	3	4	3	3	3
2	bottom scouring & deposition	9	11	9	12	11	14	11	11	15	0	13	12
2	pool / riffle bend ratio	0	0	0	0	0	0	0	0	0	0	0	0
3	bank stability	0	6	0	3	5	5	9	0	9	5	0	5
3	bank vegetation	9	0	2	10	2	2	2	0	9	2	2	2
3	stream side vegetation	6	9	3	3	8	0	6	0	8	6	0	8
	Total Score	48	48	30	47	56	48	53	31	72	32	39	41
	Variance from mean	2.6	2.6	-15.4	1.6	10.6	2.6	7.6	-14.4	26.6	-13.4	-6.4	-4.4
	Range of variance - unarmoured - 42												
	Range of variance - armoured - 24												

Fig 2.5 Range of habitat assessment scores for each of the 12 main Pinawa sample units. Scores generated in 2003 following Plafkin *et al.*, (1989). Horizontal lines 7.5% above & below site average shows range of acceptable scores -see text for discussion.



little agreement between site scores using this method and the results generated in chapters four, five and six, suggesting that this method of habitat assessment misses important information in larger fluvial systems and should be used cautiously in larger systems.

2.3.3.3 Sample Number

Gauch (1982a) contends that 10-50 samples per community type yield the most accurate data, after which the law of diminishing returns becomes evident. In 2002, six rip rap armoured and six unarmoured sample units were selected and retained through 2004 (Table 2.5). In 2002, I relied on 12 samples (a value within the guidelines suggested by Gauch 1982a, but clearly at the lower end of the range) with 13 repeated measures, where each repeated measure occurred on a weekly basis, in this case from June through September. Preliminary fish capture sampling occurred in May 2002, but these data were not retained for final analysis due to problems identifying some species of small-bodied fishes. Therefore, the 2002 data set was comprised of 156 samples (12 sites x 13 repeated measures) of fish species composition information. However, 1) to reduce reliance on repeated measures by increasing the base number of sample sites and; 2) to assess the impact of sampling period on overall results (sec 3.2), 24 additional sample units were added in 2003. These units, labelled "A" and "B" (Table 2.8) were situated two sample widths away, either side of each main sample unit (except at site 2), where both A and B sites were located on one side of the main unit due to spatial constraints. Therefore, a total of 36 sample sites (with 17 repeated measures on weekly sites, 7 repeated measures

Table 2.8 Pinawa study area sample unit type, arrangement and sampling frequency, 2002-3. W= sampled weekly; B= sampled bi-weekly; M = sampled monthly. Sites are numbered sequentially from upstream to downstream.

Unarmoured												Armoured					
1a	1	1b	2	2a	2b	3a	3	3b	4a	4	4b	5a	5	5b	6a	6	6b
M	W	B	W	M	B	B	W	M	M	W	B	B	W	M	B	W	M

Armoured			Unarmoured						Armoured								
7a	7	7b	8a	8	8b	9a	9	9b	10a	10	10b	11a	11	11b	12a	12	12b
M	W	B	M	W	B	M	W	B	B	W	M	B	W	M	M	W	B

on bi-weekly sites and 3 repeated measures on monthly sites, producing a 312 sample data set) were used to produce fish assemblage data for 2003. The addition of “A” and “B” sites furnished information potentially necessary to differentiate between site-type assuming that fish assemblage information might have a high noise-to-signal ratio (Gauch 1982a; Gauch 1982b). Put another way, the twelve 2002 sample sites may not have generated a clear enough signal to differentiate between site type. Adding additional sites in 2003 increased my sample number to the middle of the range as suggested by Gauch (1982a), thus strengthening the clarity of my results.

In 2004, an additional four sites were added to the original 12 main sites as a method for assessing how closely rip rap armoured sites mimicked naturally occurring bedrock shorelines. Sites were selected using the same criteria as in 2002, except natural exposed bedrock was used for shoreline strata rather than armour presence/absence. Since these

sites were inserted *a posteriori*, their numbers are interspersed with the main sites (Table 2.5). In addition, since sites were added at Whiteshell (3 armoured; 3 unarmoured) and St Georges (3 armoured; 3 unarmoured), “A” and “B” bi-weekly and monthly sites from the 2003 season were dropped for 2004. Consequently, the 2004 fish species composition data set was comprised of 280 samples (28 sites x 10 repeated measures).

It should be noted that sample numbers for each year assume site independence. This issue is discussed next (sec. 2.3.3.4).

2.3.3.4 Sample site analysis for autocorrelation

The impact on fish habitat of various environmental variables associated with each sample site is examined in chapter six. However, it is important to examine here the spatial relationships between the sample sites. Specifically, since sample sites could not be well interspersed, (sec 2.3.2), and existed in close proximity to one another, there may be concerns that the data were autocorrelated. Sites that are positively autocorrelated violate sample independence, thus affecting the number of sample sites, discussed above (sec 2.3.3.2).

2.3.3.4.1 Autocorrelation

Original sample sites ($M_1 \dots M_{12}$) were selected, among other reasons, to minimise the possible confounding effects varying land use might have on the resulting data. However, one drawback of proximate sample sites is the possibility that ecological information

associated with each site may be spatially autocorrelated. Positive spatial autocorrelation occurs when “samples that are close together have a tendency to become more similar than those randomly placed in the study area” (Dale *et al.*, 2002). Since the presence or absence of species in an adjacent area can influence the species composition within any given sample site (Legendre and Legendre 1998), autocorrelated sites are not truly independent. Thus, autocorrelated sites present a difficulty when using parametric statistics, since, for one, assessing appropriate degrees of freedom is problematic (Legendre and Legendre 1998). In circumstances where sites show positive spatial autocorrelation, results of the use of classical statistical tests, if distorted, will distort to show a significance (perhaps when no significant difference is really there). Conversely, where negative spatial autocorrelation occurs, genuinely significant results may be overlooked (Legendre and Legendre 1998).

2.3.3.4.1.1 Method

Distances between each site were paced out using snowshoes on the winter snow and ice. Distances paced follow the shoreline and represent the real distances fishes would be required to travel between sites should they do so. Repeated measures were taken for the eleven distance measures between the twelve sample sites and the mean distance between each site was used (Table 2.9). Between-site distances (not normally distributed) and 2002-3 fish species richness data (normal distribution - Kolmogorov-Smirnov normality test) were analysed using a Mantel test.

Table 2.9 Distance matrix between main sample sites (*units* = “*paces*”; where 1 pace ~0.45m). Batches of sites, sites 1-7 and 8-12, are indicated by solid lines in the table.

Distance between 2002-3 sample sites (paces)												
	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7	Site 8	Site 9	Site 10	Site 11	Site 12
Site 1	0	230	309	657	754	964	1108	2528	2568	2817	3042	3211
Site 2	230	0	79	427	524	734	878	2298	2338	2587	2812	2981
Site 3	309	79	0	348	445	655	800	2220	2260	2508	2734	2903
Site 4	657	427	348	0	97	307	452	1872	1912	2160	2386	2555
Site 5	754	524	445	97	0	210	354	1774	1814	2063	2288	2457
Site 6	964	734	655	307	210	0	144	1564	1604	1853	2078	2247
Site 7	1108	878	800	452	354	144	0	1420	1460	1709	1934	2103
Site 8	2528	2298	2220	1872	1774	1564	1420	0	40	289	514	683
Site 9	2568	2338	2260	1912	1814	1604	1460	40	0	249	474	643
Site 10	2817	2587	2508	2160	2063	1853	1709	289	249	0	226	395
Site 11	3042	2812	2734	2386	2288	2078	1934	514	474	226	0	169
Site 12	3211	2981	2903	2555	2457	2247	2103	683	643	395	169	0

2.3.3.4.1.2 Results and Discussion

The initial Mantel test revealed strong (positive) autocorrelation between sites. However, this test was conducted on the full distance matrix, where distances between sites 1-7 (1108 paces) and between sites 8-12 (683 paces) were 78% and 48% respectively of the non-sample area distance which separated these two sub-groups (i.e. the distance between sites 7 and 8 - 1420 paces). Furthermore, a straight line distance was measured from site 7 to site 8, rather than the much longer distance than if the entire shoreline through the Pinawa marina (Fig 2.4) was followed here. This large gap between the two sub-groups of sample units in essence made the distances between the sample sites themselves comparatively smaller than they otherwise would have been and using the longer shoreline distance through the marina would have exacerbated this issue further still. Therefore, a second Mantel test was run, where the associated species composition

data for sites 1-7 were examined for autocorrelation with the distances from these sites. Similarly, a third test was run using distance and fish species composition information from sites 8-12. In these subsequent tests, autocorrelation was not found to be relevant.

Nevertheless, while there is reasonable confidence that the sites do not exhibit signs of autocorrelation, there remains some level of doubt, based upon the first (full matrix) test. Fortunately, cluster analysis and ordination techniques which describe ecological structures do not rely on tests of statistical significance and are not as much affected by spatial and temporal autocorrelation. (Legendre and Legendre 1998) as are univariate statistical methods. Broadly speaking therefore, my conclusions are based on the more general patterns extracted using multivariate methods and tests of statistical significance were conducted on eigenvalues generated from multivariate methods. Taking this approach, generates confidence that sample sites are independent of one another and that the tests of statistical significant performed on the data collected from these sites meets the assumptions inherent in univariate statistical methods.

CHAPTER 3
ASSESSING SAMPLING METHOD AND PERIOD FOR SMALL AND JUVENILE FISHES
OF THE LITTORAL ZONE OF A LARGE BOREAL RIVER

GENERAL INTRODUCTION

In chapter two the physical arrangement of sample sites was established (Table 2.5), including introductory remarks about sampling period that necessarily accompanied that description. This chapter describes how sampling was conducted and compares the effect of sampling frequency and sampling time (i.e. time of season and time of day) on sampling results.

3.1 DESCRIPTION AND EVALUATION OF SAMPLING METHOD

3.1.1 LITTORAL ZONE FISH CAPTURE METHODS

Selection of appropriate sampling gear remains an important topic within the literature. Various sampling gears have been assessed and both electrofishing and beach seining figure among the leading methods for sampling littoral communities. Among those who have adopted beach seine capture methods exclusively are Boisclair and Legget (1985), Benson and Magnuson (1992), Pierce *et al.*, (1994), West and King (1996), Nash and Santos (1998), Griffiths and West (1999), Lee *et al.*, (2000) and Pierce *et al.*, (2001).

Arthaud (1992) assessed several gear types including a comparison of beach seines with

electrofishing. He found that beach seines outperformed electrofishing in three important areas. First, seines pulled toward shore rather than parallel to it as is one traditional practice – I discuss this point further, below (sec 3.1.1.1.1) – showed higher selectivity for smaller fishes than did electrofishing. Gears that capture smaller fishes is an important consideration for understanding fish habitat. Indeed, Angermeier and Karr (1986) state that exclusion of young-of-the-year (Y-O-Y) catch data reduces Index of Biotic Integrity scores by 2-10 points which is a significant portion of that 60 point scale (IBI is discussed at length in chapter four). Most important however, is that Y-O-Y information provides important information about sample site ecological function (in this case nursery habitat) and therefore should be included on that basis alone. Beach seining methodology selects better for these and other small fishes over large boat mounted electrofishing units. Second, electrofishing biased catch results in favour of those fishes which showed high galvanotaxis (i.e. attraction to electrical current, Bond 1996), especially when sampling in turbid water (also Pusey *et al.*, 1998). This is an important factor for this study. Assuming that different water turbidity exists between armoured and unarmoured sites, it seemed likely that sample types could be differentially represented if using electrofishing methods. Preliminary data from the 2001 field season showed significantly different values in turbidity readings between armoured and unarmoured shorelines. This trend was supported through-out the 2002-3 study period, see chapter six (sec 6.2.3.1). Third, beach seines are associated with essentially a non-existent fish mortality rate – an important consideration since I used repeated sampling (i.e. temporal replicates). Other researchers have also discussed the merits of using seines

for capturing fish to measure the representative ecological structure of the sample site. For example, Pusey *et al.*, (1998) found that supplementary beach seining discerned fish assemblage structure better than did additional electrofishing, following initial electrofishing sampling, while Gammon and Simon (2000) found that seining outperformed electrofishing in both species richness and abundance. Nevertheless, beach seining as a preferred method of littoral and shallow water sampling is not incontrovertible. Weaver *et al.*, (1993) compared various gear types including fyke, gill and seine nets within the littoral community and found that fyke nets best captured representative samples. On the other hand, Jennings *et al.*, (1996) remarked that fyke nets did not demonstrate any considerable improvement in discerning fish assemblages. Indeed, they argued that active gears such as electric seines were the preferred method. Fago (1998) compared mini fyke nets with seines and found that both methods were prone to missing some species. Still other researchers have found a combination of gears necessary. For example, Kempinger (1996) used a combination of methods, including seining, to capture young lake sturgeon and Shields *et al.*, (2000) used electrofishing, but relied on beach seines to block off either end of the sample reach.

Plafkin *et al.*, (1989) reviewed the major advantages and disadvantages associated with both beach seining and electrofishing (Table 3.1). Although some of these points have already been discussed above, two items require specific response. The first comment is with respect to electrofishing sensitivity to turbidity and conductivity. Recent developments in electrofishing technology means that newer units have built-in

Table 3.1 Evaluation of beach seining and electrofishing methods. Modified from Plafkin *et al.*, (1989).

Electrofishing		Seining	
Advantages	Disadvantages	Advantages	Disadvantages
standardised CPUE	sampling efficiency affected by turbidity and conductivity	not restricted by water quality parameters	no standardised CPUE
less selective than seining	selective to size and species - declining effectiveness as fish size increases. (Hendricks <i>et al.</i> , 1980)	seines are lightweight and easily transported	sample effort and results are the most variable among all methods
minimal adverse effects on fishes	species-specific behaviour and anatomy determine vulnerability to sampling (Reynolds 1983)	seine repairs are quick and can be done in the field	usually restricted to slower water with smooth bottoms and "minimal cover"
appropriate in a variety of habitat	potential hazards to field personnel	seines are relatively inexpensive	previous experience and knowledge of fish habitat are required for this method
more efficient use of manpower than some other methods		minimal effects on fish populations	

governors to adjust amperage output according to local water chemistry. However, this technology tends to be associated with larger, boat mounted units; the use of which may have presented capture problems – especially for very small fishes at my sample sites.

Personal and anecdotal evidence of backpack electrofishing units suggests that only boat mounted units have successfully overcome the sensitivity to local water chemistry parameters. The second comment is in response to the concern that no standardised catch per unit effort (CPUE) is available with seines. This point was valid at the time of Plafkin *et al.*, (1989) and therefore results from beach seine hauls were difficult to

compare with other studies; or indeed even within the same study. However, the modifications used in this study offer a means of ensuring within-study standardisation of catch results. The fixed-frame method described below allows researchers to analyse their catch; either as per unit time or alternatively, on a per-unit area sampled.

Ultimately there is no consensus on a single best sampling gear, although there is strong support for use of both beach seining and electrofishing methods within the littoral zone. However, given the fine resolution of sample sites and sample unit size, sampling gears which are small and nimble are most appropriate for this project.

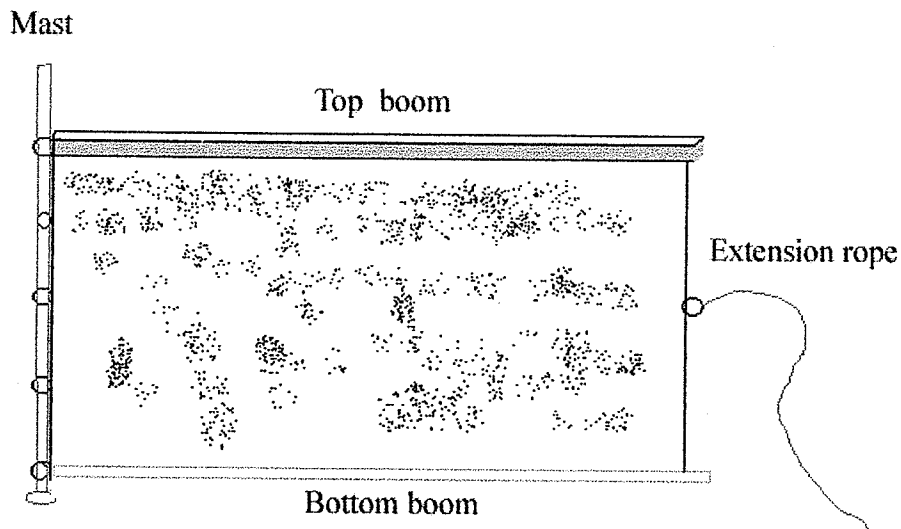
3.1.1.1 APPARATUS - MODIFIED BEACH SEINE

Resident fishes of study area sample sites were captured using a modified beach seine. These fishes included mostly small and juvenile specimens, but some adult large-bodied species were captured by this method as well. Work by Arthaud (1992), showed that a beach seine, pre-set and later pulled toward the shore using extension ropes, rather than parallel to shore, was an effective method for capturing littoral zone fishes. A modified beach seine constructed for use in this study was designed to follow Arthaud's directional use of the seine. At the same time, the modified seine incorporated the ability to sample immediately, without pre-setting. These modifications allowed repeated sampling without the need to pre-set nets. A description of the apparatus follows.

The modified beach seine (Fig 3.1) was constructed of a 2.4m long, 3-ply wooden boom with a 55mm (inside diameter) metal ring attached to one end. Embedded between the

layers of the boom is the top edge of a 2.4 x 1.2 m piece of 4 mm gauge beach seine mesh. A 42mm (outside diameter) galvanised pipe forms a second boom which was sewn into the bottom edge of the netting. These two booms form the two long sides of a rectangle. Into one side of the netting, three 55mm (inside diameter) metal rings were sewn, roughly equidistant apart. On the other side, a light gauge tensioning cable and 55mm ring assembly were woven. A second component of the apparatus is a simple 42mm galvanised mast with a small pad or “foot” attached to its bottom, designed to

Fig 3.1 Diagram of modified beach seine designed and constructed by the author.



keep the mast from sinking into the substrate. To prepare for sampling, the mast is fed

through the three net and upper and lower boom rings. A 6.5m extension rope with handle is hooked onto the cable-ring assembly. During sampling, one member of the crew holds the extension rope handle, while the second member of the crew collapses the apparatus so both booms are almost touching. The mesh and extension rope are stuffed between the booms. The crew member holding the booms wades out, paying out the extension rope, until it becomes slightly taut, thus limiting any further travel out into the river. Thus the distance out into the river is controlled, in part contributing to a fixed unit sampling area and thus to a standardised CPUE. The booms are then placed in the water, perpendicular to shore. The metal boom sinks to the bottom while the top boom floats. The shore-based crew member then slowly pulls the extension rope. The net “hinges” on the mast, cutting an arc of travel from its initial position to the back margin of the sample site. Once the apparatus is parallel to shore, the in-water crew member uses the vertical post to move that end of the apparatus towards shore, matching the speed of the end pulled by the extension rope (Fig 3.2).

Fig 3.2 Sampling site 7 using modified beach seine. Photo shows research assistant and DRSA tutor, Cristy Smith with DRSA students Amy Kowalski and Laura Melanson (L to R), July 2003.



The metal boom kicks along the substrate and forces benthic fishes to move up into the netting where they are captured. Observations made during sampling reveal that pelagic fishes tend to shoal away from the direction of disturbance, and thus swim toward the shore. At the shoreline, both crew members scoop the net thus changing its orientation from vertical to horizontal. During this scooping process, the pelagic fishes are captured within the mesh. While fish are being removed from the apparatus, the two booms act as handles and the apparatus has the appearance of a medical stretcher. By holding both booms slightly above water, this device simultaneously prevents the fish from escaping, and retains them in river water during transfer to collection containers (Fig 3.3).

Fig 3.3. Checking modified beach seine for fish at site 7. Photo shows DRSA students Amy Kowalski, Laura Melanson and Cristy Smith.



Over 43 000 fishes, belonging to 28 species (Appendix A) were captured using this method. In both the 2002 and 2003 field seasons, some effort was put into assessing the relative efficacy of the modified seine compared to traditional beach seining and to the use of passive gears - i.e. baited minnow traps.

3.1.1.1.1 Comparing modified seining to traditional seining and minnow traps

To compare gear species-selectivity of the modified seine with that of a traditional beach seine, two tests were conducted one during each of the 2002 and 2003 field season. I also examined the comparative effectiveness of baited minnow traps as a potential supplemental gear type.

3.1.1.1.1 Methods

2002 beach seine test

In 2002, three sampling sites were interspersed among the twelve main sites (Fig 2.4) and sampled weekly with a traditional seine (4 mm mesh size). Sites were sampled by a two-person crew wading parallel to the shoreline, pulling the seine behind us. Sites were classified as “non” (unarmoured); “new” (armoured with rip rap, in place ~2 years) and “old” (armoured with rip rap, in place ~30 years). Traditional seining sites were ~35m x 2 m. These parameters were established to make the total sampled area of each seine type (i.e. modified and traditional) roughly the same. Specifically, the weekly area sampled by modified seine sites was 2.4m x 7.4m (~17.7m²) x 12 sites, totalling

~213m², (ref sec 2.3.3.1).

Through preliminary trials at these sites, I established that we could not extend the outermost edge of the traditional seine sites out further than 2m from shore, since at this distance, average water depths neared over-topping my chest waders. Taking this value as fixed (during seining, my assistant and I clipped on a 2m rope that we kept taut to preserve our 2m sample unit width), I determined the site length, which, when multiplied by three "traditional seining sites", would approximate ~213m². All captured fishes were identified to species and released. Since group sizes were not at all similar, statistical analyses were not performed on the data.

2003 beach seining test

A single one-day test was conducted in 2003 to compare the effectiveness of my modified seine with a traditional one, when both sampling gears were used along the littoral gradient (i.e. perpendicular to the shore). During this test, two pairs of sample sites were selected along 50 m of sand beach in Sylvania lake (adjacent to the Pinawa study area). Each pair of sites was sampled four times in rapid succession, alternating gears between passes (Table 3.2) and starting each of the halves of each pair with a different gear type (Table 3.2). All captured fishes were identified to species and released. A *t*-test was used to compare total species richness by gear type.

Table 3.2 Sampling pattern to compare modified and traditional beach seine gears along the littoral gradient Sylvia Lake, MB, July, 2003.

	Pair 1		Pair 2	
	A	B	A	B
Pass 1	trad	mod	trad	mod
Pass 2	mod	trad	mod	trad
Pass 3	mod	trad	mod	trad
Pass 4	trad	mod	trad	mod

2003 minnow trap test

During Aug 2003, four baited (10 kernels each - dry dog food) cylindrical minnow traps (36 cm x 40 cm) were set at sites 1,2, 3 and 6 for approximately 36 hours. Traps were set over the course of a weekend, to avoid confounding effects of beach seine sampling on the test. All captured specimens were identified to taxon and released. Lack of data meant that statistical analyses were not performed.

3.1.1.1.1.2 Results

The 2002 beach seine test indicated that the modified seine was more effective capturing littoral zone fishes than was the traditional seine. While 21 species were captured during the 2002 field season using the modified seine, only 12 species were captured using the traditional seine. On the other hand, results of the 2003 test were not so obviously disparate as suggested by results of the 2002 test. Results from the 2003 test showed no significant difference ($p = 0.486$; $\alpha = 0.05$; $\beta = 0.05$) between total species richness

(Table 3.3) captured by gear type (although low test power may mean this is a false negative).

Table 3.3 Species richness results comparing modified and traditional beach seine gears along the littoral gradient Sylvia Lake, MB, July, 2003. Number of species captured listed in parentheses.

	Pair 1		Pair 2	
	A	B	A	B
Pass 1	trad (4)	mod (3)	trad (3)	mod (4)
Pass 2	mod (3)	trad (5)	mod (3)	trad (2)
Pass 3	mod (3)	trad (2)	mod (3)	trad (2)
Pass 4	trad (4)	mod (5)	trad (4)	mod (5)
Sub-total (mod)	6	8	6	9
Sub-total (trad)	8	7	7	4
Total (mod)	29			
Total (trad)	26			

The minnow trap test yielded crayfish, but no fish whatsoever and no further work was conducted with minnow traps, since beach seines had demonstrated success capturing fishes at each of these sites during normal sampling. Results (species richness) of all years and gears are provided (Table 3.4)

Table 3.4 Species richness summary table by sampling gear, 2002-2003.

	Traditional Seine	Modified seine	Baited Minnow trap
2002	12	21	---
2003	26	29	0

3.1.1.1.1.3 Discussion

Although the area sampled between gear types was essentially equal, using the modified beach seine to sample along the littoral gradient rather than across it as I did in 2002, meant that the test of gear type had an inherent bias. Indeed, the results from 2003, illustrate that either of the modified method or the traditional seine, when sampled along the littoral gradient, was an equally effective capture method as the other - at least based on species richness. Since the modified seine outperformed the traditional seine in the 2002 test, showed numerically (but not statistically) better results in the 2003 test and made standardisation of area sampled easier, I retained the use of the modified seine as my sampling gear. Minnow traps were not considered time effective for adding as a supplemental gear.

Overall, the modified beach seine method allowed small sample areas to be utilised while still capturing representative samples of the fish community along the littoral gradient. Together, the frame design and the extension rope ensured that each sample site was exposed to the same two-dimensional area of net (see chapter 5) – a key issue when comparing haul results and catch per unit effort (CPUE).

3.1.2. SAMPLES AND SAMPLING

3.1.2.1 Sampling method

Four seine hauls were used at each site to produce one sample. Each main site (Table 2.8) was sampled once per week, May through August /September. Bi-weekly and monthly sites were sampled accordingly, through 2003. Sampling order of the main sites was random throughout 2002, but designated to a specific day of the week throughout 2003-4 to improve logistics. However, regardless of year, all sites were visited on a 1 week rotation. Throughout the 2002-2004 field seasons, site order per sampling day was selected at random among the main sites. However, main sites were always sampled prior to bi-weekly sites and those sites prior to monthly sites. To avoid pseudo-replication, fish were not released back into the sample site until all hauls were completed. Since "A" and "B" sites were close to the main sites, all specimens were held in captivity (small containers stocked with river water between hauls, large pails stocked with river water between main and supplemental sites) until bi-weekly and monthly sites were sampled. To facilitate rapid processing of large numbers of fish and to keep fish mortality low, a mobile fish identification laboratory was designed and constructed prior to the 2003 field season (Figs 3.4a-d). Fish were identified to species, massed and measured for total length (TL) before being released back into the sample site from which they were captured. In general, species identification was not a problem. However, some young-of-the-year cyprinids could not be identified beyond genus and were listed as "unknown"shiners", although they were probably *Notropis volucellus* (mimic shiner).

Cottus cognatus (slimy sculpin) were generally distinguishable from one another, but were conservatively grouped and listed as sculpin *spp* deferring to Dr. K. Stewart's concerns (pers. comm.) about accurate field identifications of this genus.

Figs 3.4a. Mobile fish lab, showing fish processing / water chemistry work space and sample gear storage locker (front). Cristy Smith setting up water chemistry test kit, July 2003.



Fig 3.4b Mobile lab "dark room". This space was used to consult high resolution photos of fish for specific identification purposes (eg. scale counts along lateral line). Storage below.



Fig. 3.4c Cristy Smith sorting fish for processing on mobile lab, July 003.



Fig 3.4d Cristy Smith measuring fish on mobile lab, July 2003.



3.1.2.2 *Quality control procedures for sampling fishes*

Irrespective of the analytical methods utilised, study results rely extensively on correct fish identification. To that end, guidelines were used to ensure accurate data were extracted during sampling. To reduce sampling error, five quality control guidelines were identified by Hughes *et al.*, (1992) and used in this study. Each point is reproduced verbatim (in italics) followed by the standard practice of compliance over the course of this study:

1. *Establish sampling effort criteria (duration, distance, locations, gear types and effectiveness).* Sampling criteria were established (above) and strictly followed.
2. *Repeat sampling by different crew (10%) of sites to calculate precision and data comparability and reduce sampling error.* Sampling crews changed during the course of the season and over the three field seasons, using different DRSA¹² students and outside help. While the changing crew over the duration of my project did not provide a basis for a comparative analysis as was intended by the

¹²

DRSA - Deep River Science Academy, (Whiteshell Campus) is an NSERC award winning programme that provides an opportunity for high school students interested in science to participate in on-going research and earn academic credit while doing so. Each project is allocated a maximum of two students and one university undergraduate who acts as a tutor to the students. This project was used as a DRSA project from 2001-2004 inclusively. Programme now known as Manitoba Science Academy.

directive, it did mean that I refined my skills (and presumably increased precision) by teaching and thus reviewing all components of the project regularly.

3. *Use standard keys and nomenclature.* Identification keys used were:
 - a. Stewart and Watkinson (2004) - a pre-publication draft was used 2002-3; text in 2004
 - b. Scott and Crossman (1998).

4. *Preserve, label and curate voucher specimens.* Voucher specimens were preserved (70% ETOH) only for those few fishes which died during sampling or from the subsequent identification process. However, high resolution photographs were taken of many captured species, and both these photos and preserved specimens were available, along with a photo and textual description data bank stored in the mobile laboratory.

5. *Re-identify all voucher specimens to determine and improve taxonomic accuracy.* Voucher specimens (including photos) were re-examined on an ongoing basis. Periodically adjustments in identification were made. This process was simplified by referring uncertain species' identities to numbered and dated photographs cross-referenced with the field log.

3.2 ASSESSMENT OF SAMPLING PERIOD

Small and juvenile fishes use the littoral zone of aquatic systems as important spawning (Geiling *et al.*, 1996, Hamel *et al.*, 1997) and nursery (Floyd *et al.*, 1984, Casselman and Lewis 1996, Chiasson *et al.*, 1997) habitat. Ecosystem based approaches to management of this space, rather than of particular species, has resulted in the burgeoning interest in the entire fish assemblage of the littoral zone (Boisclair and Leggett 1985, Benson and Magnuson 1992, Weaver *et al.*, 1993, Minns *et al.*, 1994, Pierce *et al.*, 1994, Beja 1995, Jurajda 1995, Santos and Nash 1995, West and King 1996, Jones *et al.*, 1996, Jennings *et al.*, 1996, Fischer and Eckmann (1997a; b), Fago 1998, Lee *et al.*, 2000, McInerney and Cross 2000, Pierce *et al.*, 2001). Since the littoral zone is critical habitat, it is important to maximise our understanding of this habitat.

Determining spatial organization of sampling units to meet the assumptions of most statistical methods and to generate meaningful information is a key element in any experimental design (Gauch 1982a; Hurlburt 1984; Dutilleul 1993; Dale *et al.*, 2002). The spatial arrangement of sample units is an appropriate endpoint in the experimental design where there is no use of replicates. However, in designs where the effects of time or of repeated sampling on a community may influence results, understanding the impact of sampling periodicity is important. Ecological data sets are comprised of information generated by observations of biotic and abiotic circumstances operating in accordance with each other and impacts due to excessive repetition may negatively influence results. On the other hand, intensive investigation has the potential advantage of producing

greater completeness in species composition information than might be found in large scale, broader studies (Gauch 1982a). For instance, Angermeier and Karr (1986) found Index of Biotic Integrity (IBI) metrics sensitive to sampling effort, and increased effort yielded decreased sampling error. Balancing these competing perspectives requires quantified assessment of the impact of sampling on sampling results. Within this study, it is important to understand the point at which increased sampling effort results in a better grasp of the habitat and associated fish assemblage on the one hand, and reduced information as a result of excessive sampling and potential habitat damage on the other hand.

I conducted weekly sampling to allow me to examine some of the energy dynamics inherent within the system. I developed this sampling period based on the following rationale. First, I assumed that most of the fishes inhabiting the littoral zone had direct linkages to primary production, either as primary or secondary consumers. For example, Bond (1996) generally classifies the *Cyprinidae* family as herbivorous. *Cyprinidae* are the largest family of vertebrate animals (Stewart and Watkinson 2004) and account for at least 10-12 fish species within the Winnipeg River system (Stewart and Watkinson 2004). Goldstein and Simon (1999) developed a refined trophic level classification and the majority of species found within the Winnipeg River littoral zone, irrespective of feeding method, exhibit direct linkages to primary and secondary production, especially in the form of algae, periphyton and invertebrates. Similar trophic classifications were found in Coker *et al.*, (2001). Second, I assumed that the foraging behaviour of these

fishes or their animal prey would respond to changes in rates and amounts of primary production; this response would be evident as fish presence or absence. I expected a similar response from predator species such as *Esox lucius* (northern pike) and *Ambloplites rupestris* (rock bass) as they follow their prey. The generation times of the larger algae ranges from 4-8 days under natural conditions and shorter than that for smaller species (Wetzel 2001). As a result, I selected a sampling period which encompassed one complete generation of algal turnover but was still logistically viable. By this logic, I assumed that within a one week period, littoral fishes had time to engage in any of their various biological processes, including foraging, dispersal, growth, spawning or becoming prey; any of which has the potential to change the relative population structure.

3.2.1 METHODS

Fish composition data were collected according to weekly, bi-weekly (every fortnight) and monthly sites. All catch data were pooled by sampling period to build a species composition matrix for each site. The creation of composite samples from replicates is a recognised approach, particularly as a strategy for reducing noise (Gauch 1982a).

Additive descriptors are those measures that can be added together to form a composite; number of organisms in a unit is a good example (Legendre and Legendre 1998). Each matrix was natural log transformed ($\ln+1$) to approximate normal distribution patterns. Matrices with rows of zero variance (i.e. identical abundance information, especially including "0" (or no capture) across all 12 sample sites for those species measured in at

least one other matrix) were adjusted by removing those species. Thus, each of the three matrices compared was comprised of varying numbers of variables (fish species). The weekly site matrix was built on the relative abundances of 24 species, the bi-weekly site matrix on the relative abundances of 18 species and the monthly site matrix was based on relative abundances of 20 species. Each matrix was analysed using standardised PCA. Axis one scores from all three ordinations were used to compare site type (*t*-test: armoured vs unarmoured) and site type against sampling period (weekly; bi-weekly; monthly) using the most conservative (Tukey test) one way analysis of variance. ANOVA on ranks was performed on raw abundance values.

3.2.2 RESULTS

Total species abundance, when pooled across all twelve sites, (Table 3.5) was not significantly different among sampling periods ($p = 0.245$; but with low test power $\alpha = 0.05$; $\beta = 0.16$). Notwithstanding, armoured sites were always responsible for greater abundance and generally greater richness than unarmoured sites (Table 3.6). Similarly, regardless of sampling period, all three ordinations (Figs 3.5 - 3.7) showed that armoured sites exhibited a greater proportion of the trended information than did unarmoured sites. While sampling period does not seem to affect this specific conclusion about site type, it does play a role in the amount of plotted separation between site types, the arrangement of individual sites compared to one another and the percent of total variance explained among the three different analyses.

Table 3.5. Total species abundance by site sampling period, for 2003 field season.

Species	Total abundance (weekly sites)	Total abundance (bi-weekly sites)	Total abundance (monthly sites)
Black crappie	1	0	0
Blackchin shiner	2	1	1
Blacknose shiner	0	0	2
Brook stickleback	48	32	20
Burbot	0	1	0
Emerald Shiner	2119	40	16
Fathead minnow	3	0	2
Golden shiner	2	0	0
Iowa darter	14	8	2
Johnny darter	1644	640	192
Logperch	4	2	1
Longnose dace	3	1	1
Mimic shiner	1032	1402	1261
Mooneye	3	0	2
Ninespine stickleback	1	0	0
Northern pike	34	24	26
River Darter	1	0	0
Rock bass	635	521	240
Sculpin spp	15	17	6
Smallmouth bass	242	50	15
Spottail shiner	5845	2337	689
Unknown fry	345	56	31
Walleye	1	0	0
Weed shiner	175	42	36
White sucker	150	58	29
Yellow perch	822	786	267

Table 3.6 Summary of 2003 species composition by sampling period.

	Total richness	% total richness	Total abundance	% total abundance
Weekly	24	100	13141	60
Bi-weekly	18	75	6018	27
Monthly	20	83	2839	13
Weekly (unarmoured)	19	79	2507	19
Weekly (armoured)	21	87.5	10634	81
Bi-weekly (unarmoured)	15	88	1093	18
Bi-weekly (armoured)	17	100	4925	82
Monthly (unarmoured)	17	89	456	16
Monthly (armoured)	16	84	2383	84

More specifically, ordination of weekly sites information (Fig 3.5) shows site one (unarmoured) carrying important information, albeit on axis two. However, this information was only acquired by intensive sampling. Analysis from bi-weekly and monthly sampling (Figs 3.6 - 3.7) position this site in a much different object-space, thereby suggesting it is in some way different habitat than is conveyed from analysis of the weekly sampling data.

Analysis of first axis PCA scores (Table 3.7) revealed that overall, site types (armoured vs unarmoured) were found to be significantly different ($\alpha = 0.05$; $\beta = 1.00$) from one another. This result is intuitively consistent both with the amount of separation between site-types within the plotted information shown in figures 3.5 - 3.7, as well as with the

Fig 3.5 PCA scatterplot of 2003 weekly site species composition data (ln +1 transformed). Plotted objects are sample sites. Unarmoured sites indicated by ■, armoured sites by ●. Explained variance listed on axes.

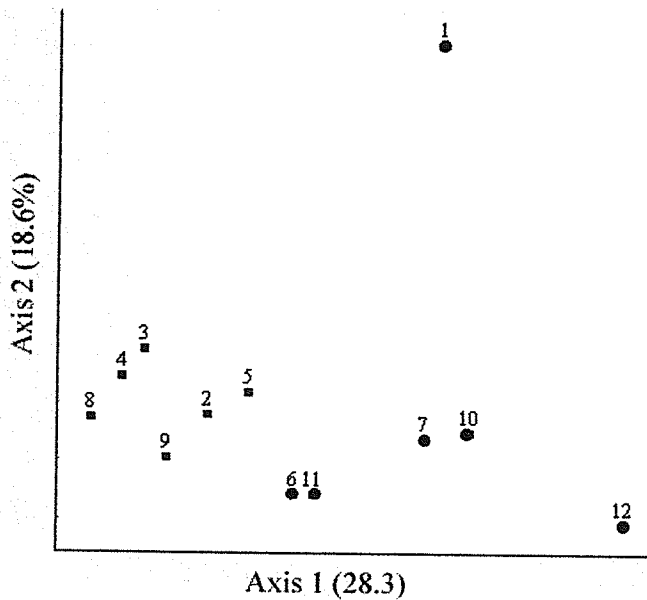


Fig 3.6 PCA scatterplot of 2003 bi-weekly site species composition data (ln +1 transformed). Plotted objects are sample sites. Unarmoured sites indicated by ■, armoured sites by ●. Explained variance listed on axes.

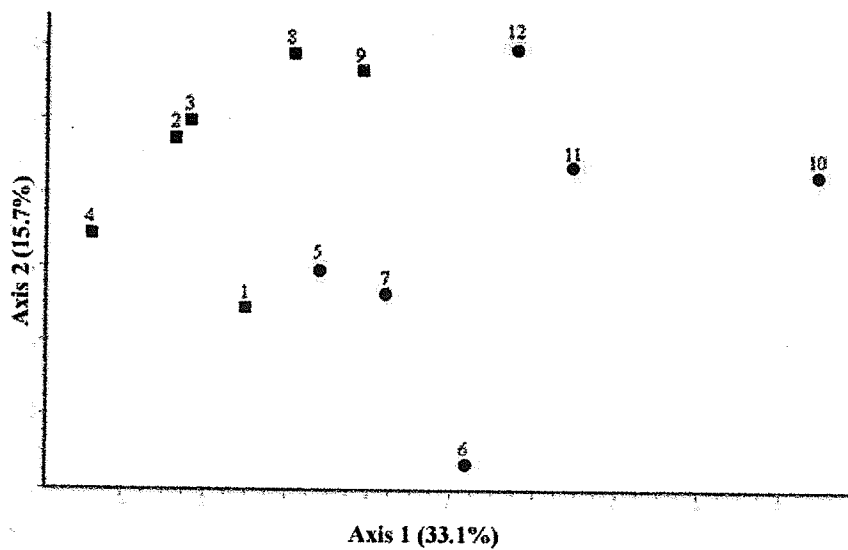
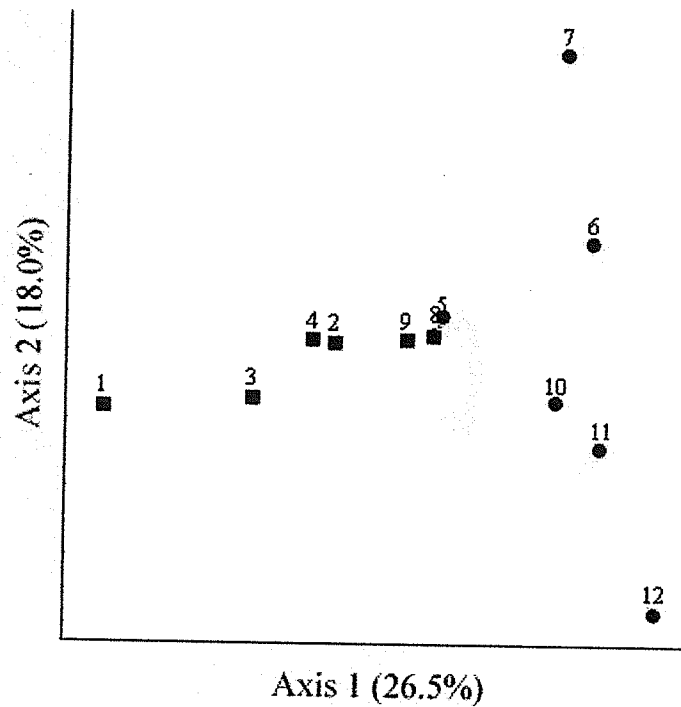


Fig 3.7 PCA scatterplot of 2003 monthly site species composition data (ln +1 transformed). Plotted objects are sample sites. Unarmoured sites indicated by ■, armoured sites by ●. Explained variance listed on axes.



raw abundance and richness data (Table 3.6). ANOVA on PCA 1 scores from all pairwise combinations showed significant difference ($\alpha = 0.05$; $\beta = 0.986$) between site type, but not between sample period. In contrast, PCA axis 2 scores (t -test) were inconclusive, showing no difference between site types ($\alpha = 0.05$; $\beta = 0.224$). ANOVA on site types and sampling period using PCA axis 2 scores did not show any significant difference, ($\alpha = 0.05$; $\beta = 0.05$). However, the test power for both these tests of PCA axis 2 scores is well below the desired power of 0.8 and therefore this result *may* be a

false negative.

Table 3.7 PCA axis 1 sites scores by shoreline type and sampling period. Site number listed in parenthesis following axis score.

PCA Axis 1 scores					
Weekly Site scores		Bi-weekly Site Scores		Monthly Site scores	
Armoured	Unarmoured	Armoured	Unarmoured	Armoured	Unarmoured
-0.747 (5)	2.318 (1)	-0.617 (5)	-1.489 (1)	0.049 (5)	-4.646 (1)
-0.024 (6)	-1.391 (2)	1.117 (6)	-2.343 (2)	2.132 (6)	-1.441 (2)
2.039 (7)	-2.402 (3)	0.216 (7)	-2.178 (3)	1.75 (7)	-2.591 (3)
2.764 (10)	-2.76 (4)	5.443 (10)	-3.386 (4)	1.6949 (10)	-1.7546 (4)
0.285 (11)	-3.239 (8)	2.478 (11)	-0.955 (8)	2.29 (11)	-0.035 (8)
5.199 (12)	-2.043 (9)	1.776 (12)	-0.121 (9)	3.032 (12)	-0.445 (9)

3.2.3 DISCUSSION

Each of the ordination diagrams (Figs. 3.5 - 3.7) illustrate the same overall result. That as a group, armoured sites scored higher on the first axis than unarmoured sites did, and as a result, account for more of the total trended information. Following general ecological principles (Wetzel, 2001), I interpreted sites with higher measured values of species composition to be equated with comparatively better habitat than their lower scoring counterparts. This approach is generally consistent with work in various related fields. For example, Rothrock *et al.*, (*in press*) report a positive correlation between littoral zone plant species richness and habitat quality, while Kuussaari *et al.*, (*in press*) describe a positive correlation between species richness among *Lepidoptera* and boreal habitat

quality. On the other hand, while Bower *et al.*, (*in press*) acknowledges that excessive species richness may indicate signs of habitat degradation (at least among stream bank plant communities), they argue that such an assessment might be due to anthropogenically derived disturbance and the influence of subsequent edge effect.

Presumably however, degrading habitats such as these would not support simultaneously both high species richness and abundance, due to limited resources. Since sites with higher axis scores (Table 3.7) contained both higher species abundance and richness values (Table 3.6), I interpreted these higher axis scores to be an analogue for good habitat quality; where habitat quality degrades toward the origin of the scatterplot.

Since these three ordination diagrams (Figs 3.5-3.7) consistently deliver the same message about site type difference, it might appear that sampling monthly is sufficiently frequent to reveal general ecological trends in the littoral gradient of a boreal river and that sampling more often is unnecessary. Such frequency is in line with Fore *et al.*, (1994), who suggest that fully independent replicates probably occur at 2-3 week intervals. Indeed, other workers too have sampled shallow water systems on a monthly basis. For instance, Duffy and Baltz (1998) used monthly sampling to compare fish assemblage and macrophyte associations.

However, while sampling every other week (i.e. twice as often as *appears* to be necessary) does not seem to provide any additional information about the comparative merit of fish habitat by site type, sampling weekly *does* contribute important additional

information. This phenomenon is especially evident with respect to site one, (Fig 3.5) which otherwise has the appearance of being comparatively poor habitat when assessed using either the monthly or bi-weekly analysis (Figs. 3.6 - 3.7). However, the weekly sampling data indicates that site one ranks among the highest quality sites (based on axis one scores). To explain this apparent dichotomy, I interpret this axis being driven, but not represented exclusively, by species abundance. Indeed site one accounted for the fourth highest abundance (n = 954) of the twelve main sample sites during weekly sampling in 2003 and the third highest axis two score (Table 3.7). A second distinction that emerged as a result of weekly sampling occurs on axis two; again with respect to site one. In this case, site one has the highest of all axis two scores, thus reinforcing the conclusion that this site is comparatively good habitat. I interpret this axis being driven, but once again not represented exclusively, by species richness. Interestingly, sites one and twelve accounted for the highest species richness (n = 18) during 2003 weekly sampling. At site one, two species of multiple individuals were captured that were not found at any other site (*Hiodon tergisus* (mooneye), n = 3 and *Percina caprodes* (logperch) n = 4) while site twelve accounted for three site singletons. Among this comparative richness at site one were found species which represented different trophic levels – or at least more so – than those trophic levels represented by the species assemblages of other sites. A specific example is the capture of juvenile mooneye at site one. This species is highly transient, and represents a different feeding level (crustaceans; insects and fishes - Croker *et al.*, 2001) than the *Cyprinidae* species that dominate this study area.

While I interpreted axis one being driven primarily by species abundance and axis two driven by species richness, I do not assume that abundance and richness are exclusively represented by these two axes, since I expect that the *interaction* of abundance and richness (say where high numbers of one species affect the presence / abundance of another) play into both these axes, and / or on subsequent ones. This is most clearly represented by the dichotomous positions of sites one and twelve on axis two (Fig 3.5). On the one hand, site one accounted for lower total abundance than site twelve, but was balanced more evenly (i.e. no site singletons) with equally high species richness, together resulting in high axis two scores. In contrast, while site twelve accounted for the third highest abundance ($n = 1134$), and was tied with site one for the highest richness, ($n=18$), it demonstrated less evenness than that exhibited at site one, having three site singletons.

My interpretation of high axis scores as a proxy for habitat quality can be reinforced by an examination of site seven. This site also exhibits this same basic phenomenon as discussed above, although in this case the additional information on the second axis occurs in the monthly data scatterplot (Fig 3.7). Monthly information suggests that site seven is comparatively better habitat (on both axes 1 & 2 - highest richness for monthly samples, $n = 12$) than would be concluded using more frequent sampling. In this case, the additional information derived from weekly sampling tempers the conclusion that site seven is rated among higher quality habitat sites and brings in into line with other similar (armoured) sites. The examination of these two most prominent examples suggests that

the most accurate picture of each site is derived from weekly sampling, and that infrequent sampling may lead to exaggerated or outright incorrect assessments of relative habitat quality. Sampling at this finer temporal grain suggests there is important latent ecological information which is uncovered only after intensive sampling and which may be related to other properties of the temporal sampling regime. Indeed, during the relatively short field seasons in the boreal biome, it is possible that 2-3 week sampling intervals may miss the movement of some species through given habitats, suggesting that fishes may be using various habitats differentially through-out the course of the open water season. It is also possible that time-of-day at which sampling was conducted may influence these results, accounting for differences in site position on axis two. Each of these ideas must be pursued before any final conclusions are made about sampling period.

3.3 EFFECT OF SEASONALITY

Although it is often assumed that fish samples collected at night will yield more complete information about the residents of specific habitats than samples collected during daylight hours (Reash 1999) – especially with the use of passive versus active capture gears – there is evidence to indicate that this accepted position is not universally true (McInerny and Cross 2000). Nor is it always the case that increases in species composition are positively correlated with temporal progression through the sampling season (McInerny and Cross 2000). Therefore sampling without sufficient replicates or

under a limited sampling regime where temporal peaks are either 1) ignored or alternatively, 2) form the only basis for analysis, may dramatically skew results and affect final conclusions about habitat quality. Given the potential uncertainty of the relationships between the sampling results associated with any particular habitat and time of day or time of season from which results were derived, there is an understandable concern that many authors who study large rivers do not report temporal variability (Reash 1999). Indeed, this concern is a general criticism covering the lack of reporting about use of replicates, time of year or time of day at which sampling occurs. To address these concerns, I examine the role time of season and time of day play in the use of habitat as part of the overall assessment of sampling period.

3.3.1 METHODS

A the start of each of the 2002-2004 field seasons, a sampling schedule was established. For purposes of analysis, all sample weeks in which all twelve main site were samples were retroactively allocated a number beginning in 2002 and designated as SW₁.....SW₃₉ (Table 3.8). Fish were captured according to methods described above (sec. 3.1.1 *et seq.*). Counts of species richness (Fig 3.8) and total fish abundance (Fig 3.9) were made for each site, for each sampling week. Using individual site data produced an image of the seasonal variation in general, but became too convoluted to trace each site over the 39 week sampling period. Therefore, mean values for armoured sites and unarmoured sites were used to compare site type difference over the course of the 2002-2004 field seasons.

Table 3.8. Sampling week number matched to month and date of first day of sampling week (2002-2003).

June 2002		July 2002					Aug 2002		Sept 2002				May 2003	
11 th	19 th	2 nd	8 th	16 th	23 rd	30 th	7 th	13 th	5 th	13 th	19 th	26 th	7 th	14 th
SW	SW	SW	SW	SW	SW	SW	SW	SW	SW	SW	SW	SW	SW	SW
1	2	3	4	5	6	7	8	9	10	11	12	13	14	15
May 2003		June 2003					July 2003				Aug 2003			
21 st	27 th	4 th	10 th	16 th	23 rd	30 th	7 th	14 th	21 st	28 th	4 th	11 th	18 th	
SW	SW	SW	SW	SW	SW	SW	SW	SW	SW	SW	SW	SW	SW	
16	17	18	19	20	21	22	23	24	25	26	27	28	29	
May 2004		June 2004				July 2004				Aug 2004				
	31 st	7 th	14 th	21 st	28 th	5 th	12 th	19 th	26 th	9 th				
	SW	SW	SW	SW	SW	SW	SW	SW	SW	SW				
	30	31	32	33	34	35	36	37	38	39				

Fig 3.8 Species richness time-series for all weekly sites by site type (armoured and unarmoured), 2002-2004

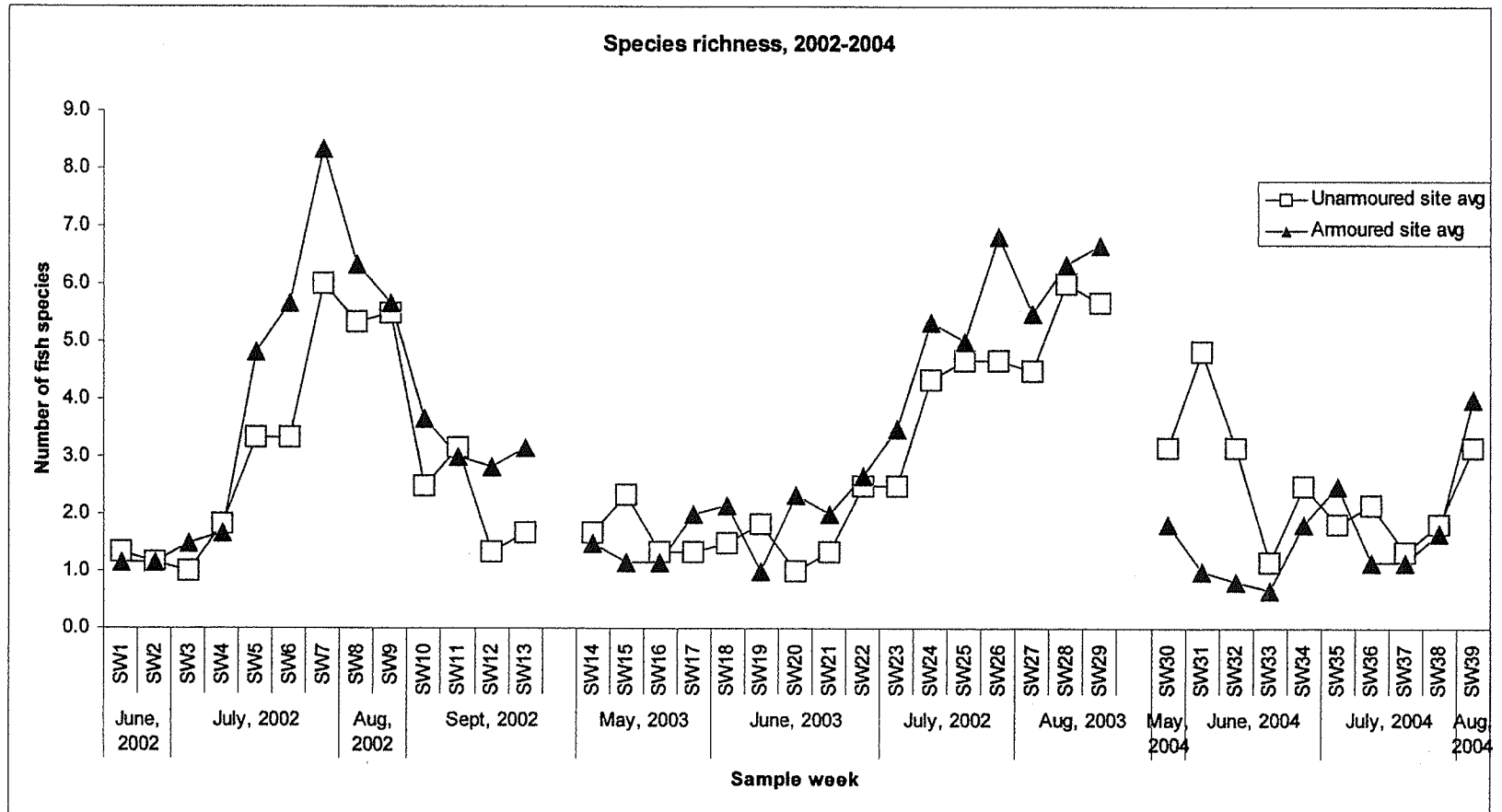
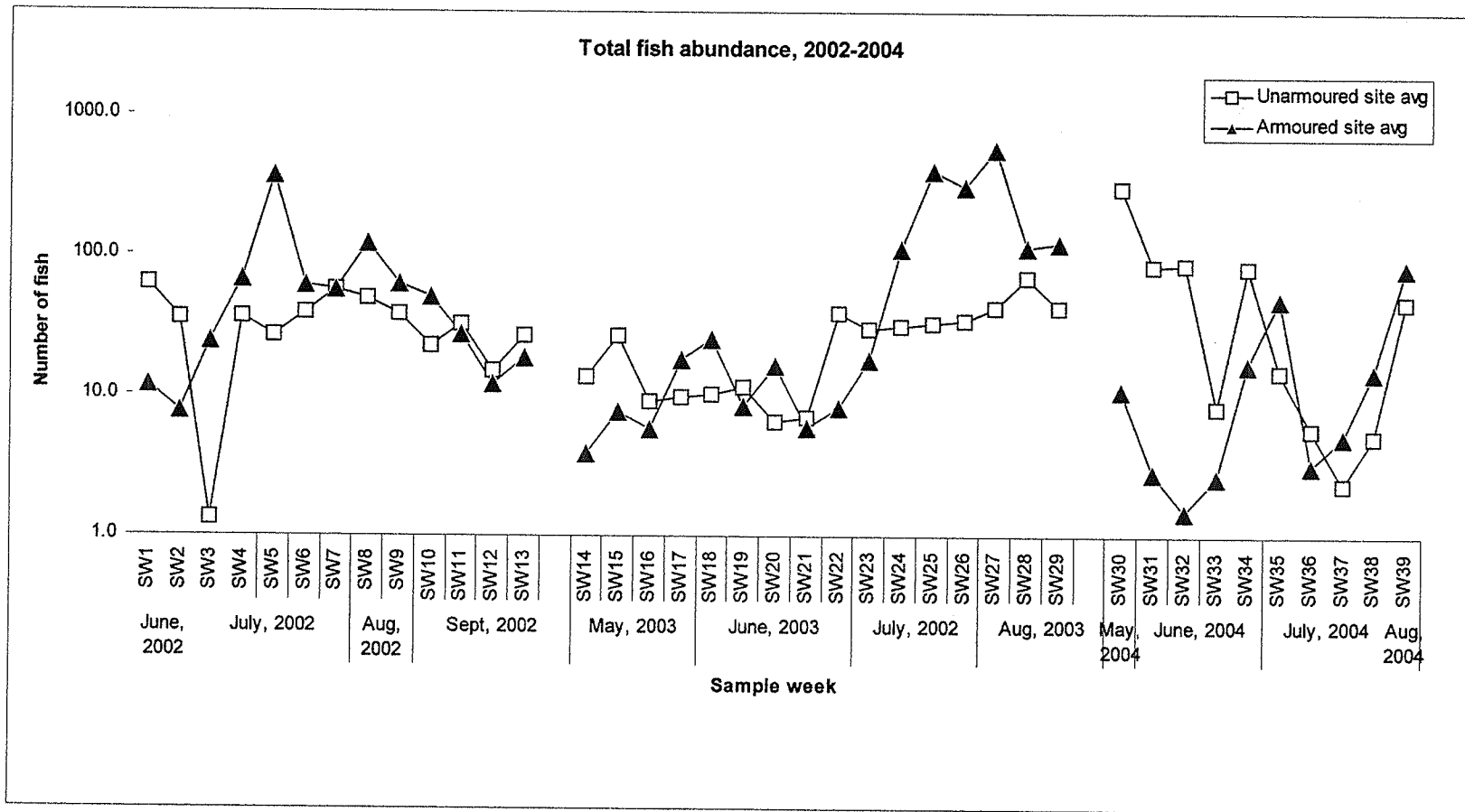


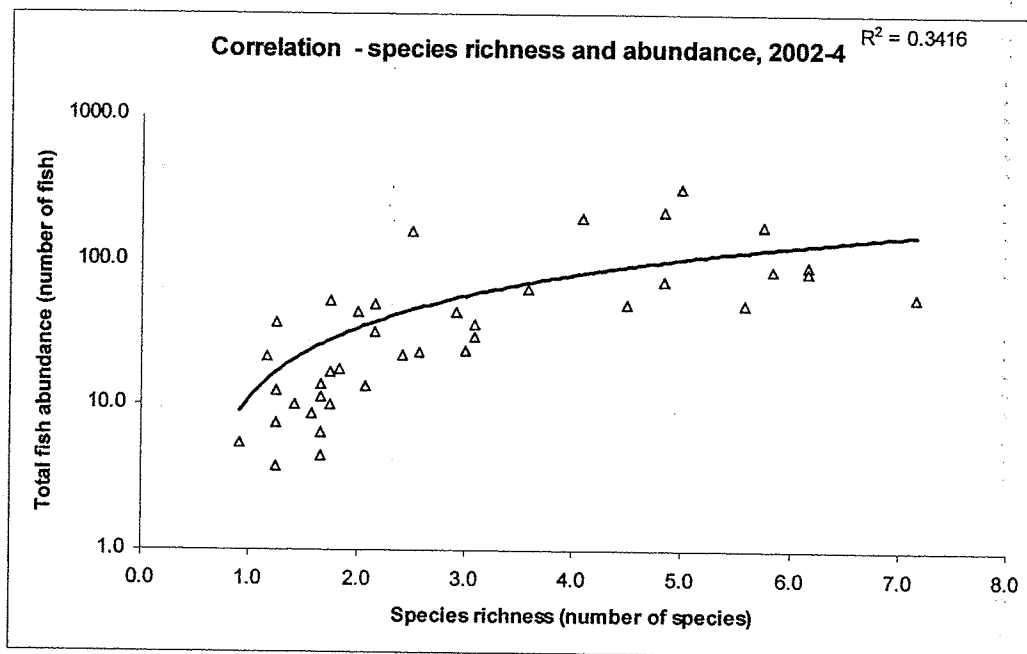
Fig 3.9 Total fish abundance time-series for all weekly sites by site type (armoured and unarmoured), 2002-2004.



3.3.2 Results and Discussion

A visual comparison of both Figures 3.8 and 3.9 reveal similar information about this study area; that in general terms, total fish abundance increases with species richness. Although there is a positive correlation between the two terms (Fig 3.10), the presence of one or two shoaling species (eg. *Notropis spp.*) can alter abundance values much more dramatically than they do with richness. As a result, correlation between richness and abundance is relatively low ($r^2 = 0.3416$). Therefore, any assessment of habitat based upon values derived partially or entirely from abundance data (see chapter 4) will contain this artifact. However, while such an artifact does not necessarily point either towards or away from ecologically valuable habitat, it may mask other ecological factors in the overall analysis (see also chapter 5).

Fig 3.10 Correlation between mean species richness and mean fish abundance for 39 sample weeks (2002-2004).



Notwithstanding the disproportionate weights of species richness and abundance in this data, both terms indicate that during the early and late parts of the season (associated with cooler water temperatures; higher water levels and only nascent or senescent macrophyte development) unarmoured sites show equitable species richness compared to armoured sites. As the season peaks through July and early August, species richness increases more at armoured sites than at unarmoured sites, though not by a statistically significant margin (unequal variance, thus Mann-Whitney rank sum test ($P = 0.909$); t -test $\alpha = 0.050$; $\beta = 0.050$ but low test power associated with the related t -test suggests this result may be a false negative and that the difference between samples is a result of more than sampling error). The only exception to this observation occurs in early June, 2004 which was by all anecdotal accounts, an exceptionally cool, wet year. Indeed, the inter-annual pattern shows marked increases in littoral zone species composition associated with the warmest water temperatures of July. By way of emphasising this point, the mean 2004 water temperature never exceeded 20°C , (personal work related observation and / or records of water temperature on the Winnipeg River between 1999-2007 have consistently (but 2004) shown summer peaks $>20^{\circ}\text{C}$) creating *de facto* circumstances for aquatic organisms of the summer that never was - where water temperatures reflected an extended spring season which then moved effectively into autumnal conditions (ref Figs 6.2a-c). On the other hand, early June 2002 shows a similar, but not so exaggerated pattern. This result suggests that unarmoured sites may provide better habitat early or late in the season for those species that can withstand higher turbidity and relatively lower water depths (see Appendix B and chapter six). Indeed, it may be argued that higher

turbidity levels at unarmoured sites provides an alternative to vegetative cover for those turbidity tolerant species. On the other hand, as macrophyte development progresses through the season, this phenomenon is reversed and species richness increases more at armoured sites than at unarmoured sites, rendering greater total species richness at armoured sites than at unarmoured ones. Similar results are evident with total abundance, (unequal variance thus Mann-Whitney rank sum test ($P = 0.901$); t -test $\alpha = 0.050$; $\beta = 0.212$). However, as with richness, given the low power of the test, this result too may be a false negative.

Clearly, the time of season at which sampling occurs can affect habitat assessment and therefore decisions about habitat use (Cudmore-Vokey and Minns 2002). The benefits of smoothing sample results by sampling across the variation associated with the open water season reduces the possible over-weighting of temporal peaks or troughs in the data generated by insufficient sampling frequency and yields a better total picture of habitat quality than would sporadic, infrequent sampling. At the same time, intensive frequent sampling throughout the open water season reveals potentially important short term habitat uses.

3.4 EFFECT OF TIME-OF-DAY

3.4.1 METHOD

Sampling during the 2003 field season was arranged to test specifically effect of

sampling frequency (above) as well as time-of-day effect. The physical sample site arrangement (Table 2.8), the time required to sample bi-weekly and monthly sites and a sampling rotation that fixed monthly sampling every fourth week, meant that time-of-day testing could only be carried out during every third week, sampling at weekly sites, in the evening (18:00 - 21:30). Species richness and total fish abundance were counted separately for each site and for each sample week for the 2003 field season. The 2003 data were separated into four equal sized groups so that comparisons within the four week period were fair and comparable given the seasonal variation illustrated in Figs 3.9 and 3.10. However, difficulty sampling during the evening meant that only data from three of these groups could be utilised for analysis (Table 3.9). Analysis of variance (ANOVA) was performed for each site on the four weekly subsets within each group to see if there was any difference between results generated from twilight sampling and the results from the weekly sites from any of the other three weeks within that group. The process was repeated using total abundance.

Table 3.9 Sample rotation for the 2003 field season.

	Week 1	Week 2	Week 3	Week 4
	Main	Main & Bi-weekly	Main	Main, Bi-weekly &
	(weekly)		(weekly - evening)	Monthly
	SW ₁₄	SW ₁₅	SW ₁₆ ¹³	SW ₁₇
Group I	SW ₁₈	SW ₁₉	SW ₂₀	SW ₂₁
Group II	SW ₂₂	SW ₂₃	SW ₂₄	SW ₂₅
Group III	SW ₂₆	SW ₂₇	SW ₂₈	SW ₂₉

3.4.2 RESULTS AND DISCUSSION

Results of species richness derived from evening sampling (week 3) was not significantly different from species richness during any other portion of the daylight sampling periods (weeks 1,2 & 4 in the rotation) in either group I (Kruskal-Wallis one way ANOVA on ranks - $P = 0.661$) or group III (one way ANOVA - $P = 0.322$; $\alpha = 0.050$; $\beta = 0.082$). The same results were found for abundance data, where there was no difference in results based on time of day in either group I (Kruskal-Wallis one way ANOVA on ranks - $P = 0.887$) or Group III. (Kruskal-Wallis one way ANOVA on ranks - $P = 0.979$). In contrast, a significant difference was found for both richness (one way ANOVA $P = <0.001$); $\alpha = 0.050$; $\beta = 0.960$) and abundance (Kruskal-Wallis one way ANOVA on ranks $P = <0.001$) in group II. A Tukey test was used to isolate which term(s) accounted for the difference. No statistical difference was found in richness or abundance either between

13

No evening sampling was conducted during SW₁₆. That week, sampling occurred during the normal work day schedule.

SW₂₂ and SW₂₃, or between SW₂₄ and SW₂₅. However, statistical differences were found between both SW₂₅ and SW₂₂ and SW₂₄ and SW₂₂. Together these results indicate the data falls into two sub-groups comprising the first two weeks and the last two weeks of the four week period respectively. While the weeks within each sub-group were similar, the two sub-groups were different from one another – a difference which can be attributed to the upward spike in both species richness and total abundance found during this four-week period (refer to Fig 3.8 & 3.9). While there was also a difference found in richness data between SW₂₅ and SW₂₃, there was no difference between these sample weeks found in analysis of the abundance data. Similarly, while richness data reveal a difference between SW₂₄ and SW₂₃, abundance data make no distinction. This ambiguity indicates that species richness distinctions exist at a finer resolution than does abundance dissimilarity. Nevertheless, ANOVA of richness (and abundance) does not separate SW₂₄ (evening sampling) from the all the rest of the sample weeks in Group II, but rather places it comparable to SW₂₅ with which it shares a seasonal similarity. Nor is the third week significantly different from any other week in any of the other two groups. Therefore, in this river system while time of season plays a role in sample results – apparently even down to differences between one week and the next, time of day does not appear to influence significantly sample results. Overall then, the weekly sampling regime used in this study appears to provide, rather than obscure, ecological information.

CHAPTER 4
SMALL SCALE APPLICATION AND ASSESSMENT OF AN INDEX OF BIOTIC INTEGRITY
(IBI) FOR A LARGE BOREAL RIVER

GENERAL INTRODUCTION

Although the accumulation of human disturbances over time undoubtedly have serious consequences for both the health and integrity of natural ecosystems, the management of environmental policy still tends to operate in discrete temporal or spatial units resulting in a project-scale focus (for instance, see DFO 1998). Therefore, project-scale assessment techniques are an important contribution to environmental assessment. However, this real need does not mean that such tools are necessarily available. In this chapter and the next, I examine several assessment methodologies designed to determine the comparative health of fish habitat. Specifically, I examine these methods as they apply to the littoral zone of a large boreal river. In this chapter, 1) I offer a modification to Karr's (1981) Index of Biotic Integrity (IBI) for use in boreal rivers; 2) I evaluate this modification and the use of IBI in a project-scale application and; 3) I pursue a line of inquiry derived from this assessment with respect to rip rap armouring and natural rock shorelines. Readers should note that the sections of this chapter which describe the modified IBI and its assessment were originally published as substantive portions of Long and Walker (2005) and are reproduced here with some minor changes. The comparison of rip rap armouring to exposed bedrock shorelines is new work which grew out of that article.

4.1 SMALL SCALE ASSESSMENT OF A BOREAL RIVER IBI

4.1.1 INTRODUCTION

The human capacity and tendency to alter the natural world usually outpaces our ability to assess the implications of these alterations on the biological inhabitants within the ecosystem. In partial response, Karr's Index of Biotic Integrity (IBI) was specifically designed to measure human impact on aquatic environments (Karr 1981). Going beyond measures of water quality, the IBI quantifies local biological condition (Mebane *et al.*, 2003) in relation to biological integrity through the direct uses of habitat by organisms at the individual, population and assemblage levels (Fore *et al.*, 1994). Fish hold several advantages over other biota as the taxa of choice for IBI (Karr 1981; Ganasan and Hughes 1998). In general, since fish rely on well functioning primary and secondary production, ecologically significant problems are revealed in fish (Yoder and Smith 1999). More particularly, since this study seeks understanding relationships about fish habitat, the presence of a resident fish assemblage available for sampling leads to the firm conclusion that this taxon is better suited for providing data for making decisions about fish habitat than can any other proxy or surrogate species, taxa or index. As such, the IBI has a substantial literature regarding both its development as an intellectual concept and as a working tool (Fausch *et al.*, 1984; Angermeier and Karr 1986; Karr *et al.*, 1986; Leonard and Orth 1986; Steedman 1988; Fausch *et al.*, 1990; Dionne and Karr 1992; Minns *et al.*, 1994; Moyle and Randall 1998; Hughes and Oberdorff 1999). In

particular, the IBI has been used to assess human impacts on physical habitats resulting from mechanised army training (Bramblett and Fausch 1991); substrate modification through river bed clogging (Belliard *et al.*, 1999); industrial deforestation at the watershed level (Toham and Teugels 1999); and concrete encased sewer line crossings (Koryak *et al.*, 2001). On the other hand, IBI does not work in all cases (Massicote *et al.*, 1990; Shields *et al.*, 1995b; Selong and Helfrich 1998; Schulz *et al.*, 1999). Therefore, it is important to validate any new version of IBI prior to relying on it to make environmental decisions.

The original IBI was conceptualised and developed using observations from small warm-water wadeable streams (Karr 1981). IBI works by taking observations from a specific area of interest and ranking them against pristine sites based on twelve individual metrics indicative of specific ecological relationships. Where the observations for any metric of the study area are roughly equivalent to the pristine site, it receives a score of “5”. Where the metric value is moderately less than the unperturbed analogue, it receives a score of “3”, and for metrics where the observations are substantially less than the benchmark, the metric is scored as “1”. The total site IBI score is the simple summation of all metrics scores. However, Simon and Emery (1995) argue that large rivers have few natural analogues, and of those, perhaps no pristine ones. This claim suggests a serious problem for scoring the IBI in a large river setting. Initially, IBI assessments of large rivers had been slow to keep pace of those in smaller systems. Indeed, the biological assessment of large rivers has lagged behind smaller rivers and streams in general (Simon and Sanders

1999). Nevertheless the index has been modified and applied to different aquatic systems (Steedman 1988; Minns *et al.*, 1994), including large rivers (Hughes and Gammon 1987; Simon and Emery 1995; Ganasan and Hughes 1998; Gammon and Simon 2000; Lyons *et al.*, 2001; Mebane *et al.*, 2003). Thus, conceptually, the IBI is appropriate for understanding anthropogenic impacts to large lotic systems, or their components.

Typically, IBI assessments occur at the watershed level scale (Karr 1981; Karr *et al.*, 1986; Karr *et al.*, 1987). However, many non-point sources of habitat degradation occur at small scales, or in discrete blocks. One such example is the use of rip rap shoreline armouring. In Canada, the Federal regulator which oversees fish habitat is the Department of Fisheries and Oceans (DFO). According to a DFO regulatory decision framework document (DFO 1998), rip rap use may present a harmful alteration, disruption or destruction (HADD) of fish habitat (sec 1.2.6). However, to date, there is little agreement and no comparison among possible methods suitable for making a HADD determination (but see chapter 5), particularly in small scale impact scenarios. Such a practical problem raises the question: can a watershed sized tool be scaled down to evaluate project-sized disturbances? In this section I assess the application of IBI in a small scale setting by 1) examining species composition using a multivariate approach; 2) comparing results from the modified IBI analysis to those generated by multivariate methods, and 3) assessing the relative contribution of the IBI metrics to the total IBI score.

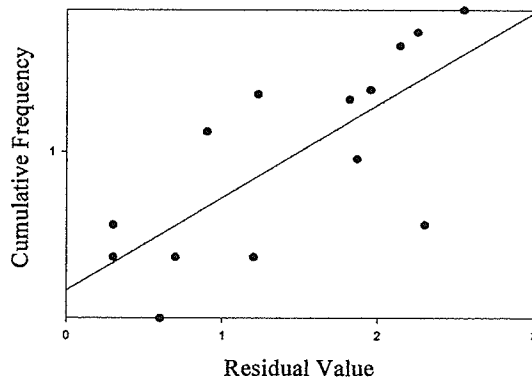
4.1.2 METHODS

Each site was sampled to determine its resident fish assemblage with particular emphasis on juveniles and small fish species. These species and year classes are associated with the comparatively small spatial ranges of the littoral community (for example *E. lucius* - see Casselman and Lewis 1996) and therefore are those fishes expected to be most likely affected by physical alteration of shoreline habitat at the project scale. Samples were collected from the Pinawa study area (described in chapter 2) using a modified beach seine (sec 3.1.2.1 *et seq.*). Samples were pooled (Ganasan and Hughes 1998) using data from the 2002 weekly sites and the 2003 weekly, bi-weekly and monthly sites (n = 39 samples pooled/site) to minimise seasonal and inter-annual variability and to reduce the incidence of site singletons. The data were natural log transformed ($\ln+1$) rendering species composition distributions (Fig 4.1) for 11 of the 12 sites log-normal¹⁴ (Kolmogorov-Smirnov normality test). A high degree of linearly normal data is generally assumed to be multivariate normal (Manly 1994) which is considered to be the best quality data for multivariate procedures.

¹⁴

Site 2 data were not found to be log-normal.

Fig 4.1 Normality plot for 2002-2003 composite samples (ln +1 transformed).



4.1.2.1 Modification of the IBI

I retained eleven of the original twelve metrics and modified four of these to suit better the local ichthyofauna of the Winnipeg River (Table 4.1). The metrics left unaltered were: (1) number of species; (4) number of sucker species¹⁵; (6) number of individuals; (8) proportion of omnivores; (9) proportion insectivorous cyprinids; (10) proportion top

15

Among the *Catostomidae* family, four species are reasonably expected to be present in the mid-reaches of the Winnipeg River, including: *Catostomus catostomus*, (longnose sucker) *C. commersoni*; (white sucker) *Moxostoma anisurum* (silver redhorse) and *M. macrolepidotum* (shorthead redhorse) - see Stewart and Watkinson (2004). Together, these species act as first to third level consumers, eating a variety of benthic insects and invertebrates and acting as prey for piscivorous fishes (Stewart and Watkinson 2004). As a result, this family can represent, to a limited extent, proper functioning ecological health and thus it warrants inclusion in the index. Although only *C. commersoni* was captured during this study (thus rendering the metric largely redundant), it makes sense to retain the metric for the application of this IBI to boreal rivers in general.

Table 4.1 Karr's (1981) metrics paired with modifications for the large boreal river littoral IBI. For all metrics except ¹, higher raw data values produced higher IBI scores. Exceptions work in reverse, where lower values yield higher IBI scores. Y-O-Y smallmouth bass were not assigned to metric 10 ² – see text. Where appropriate, source citations for modifications are provided. Species allocated to metric seven (insectivorous cyprinids) are according to Scott and Crossman (1998); Goldstein and Simon (1999) and to metric 10 (top carnivores) according to Scott and Crossman (1998).

	Metric	Karr's 1981 metric	Modified metric	Source	Study Context
Number of	1	Fish species	Unmodified		
	2	Darter species	Benthic species (darters, sculpins & Longnose dace)	Karr <i>et al.</i> , 1986 Steedman 1988 Ganasan and Hughes 1998 Schleiger 2000	Theoretical model Urban drainage basin Chemical contamination Land use effects
	3	Sunfish species	Centrarchidae species	Simon and Emery 1995	Water quality great rivers
	4	Sucker species	Unmodified		
	5	Intolerant species	Stickleback species	<i>sensu</i> Goldstein and Simon 1999	Guild structure assessment
	6	Individuals	Unmodified		
Proportion of	7	Green sunfish ¹	Johnny darters ¹	Karr 1981	Warm water streams
	8	Omnivores ¹	unmodified ¹		
	9	Insectivorous cyprinids	Unmodified		
	10	Top carnivores ²	Unmodified ²		
	11	Hybrids	--Removed--		
	12	DELTS ¹	Unmodified ¹	Yoder and Smith 1999	Ohio USEPA

carnivores¹⁶ and (12) proportion of individuals with DELTS (diseased, eroded fins, lesions or tumours), although only few examples of these were found. I modified four of the original metrics as follows. I designated (2) number of darter species as (2) number of benthic species. This new metric includes not only the resident darter species – *Etheostoma nigrum* (johnny darter); *E. exile* (Iowa darter); *Percina shumardi* (river darter) and *P. maculata* (blackside darter)¹⁷ – but also includes (*P. caprodes*), two species within the *Cottidae*¹⁸ family (following Karr *et al.*, 1986; Steedman 1988; Hughes and Oberdorf 1999; Schleiger 2000) and *Rhinichthys cataractae* (longnose dace). I replaced (3) number of sunfish species with (3) number of *Centrarchidae* species, excluding the genus *Micropterus* (following Simon and Emery 1995)¹⁹. I changed (5) number of intolerant species to reflect the *Gasterosteidae* family as an indicator of

16

Y-O-Y *Micropterus dolomieu* (smallmouth bass) were not assigned to metric 10, since most specimens captured were < 50mm TL, suggesting that these individuals had not yet undertaken an ontogenetic shift and therefore had not moved away from their juvenile role as insectivores to their year class 2 role as carnivores – see Scott and Crossman (1998).

17

No specimens of this species were captured. However, distribution of this darter does occur in the Winnipeg river drainage basin (Stewart and Watkinson 2004).

18

Although there is reason to believe that both *C. bairdi* (mottled sculpin) and *C. cognatus* (slimy sculpin) were captured during the 2002-2003 period, the IBI was calculated on sculpin spp. as a single species due to difficulty determining differences between the two species in the field.

19

Both *Micropterus dolomieu* (smallmouth bass - resident) and *M. salmoides* (largemouth bass - possible resident) are introduced species (*circa* early 1900's) and therefore their role regarding ecological health vs ecological integrity is probably conflicting – see discussion, below (sec 4.1.4.5)

intolerance to turbid water for feeding (Scott and Crossman 1998; *sensu* Goldstein and Simon 1999)²⁰ and the metric was renamed (5) number of stickleback species. This alteration is predicated on the assumption that boreal aquatic systems are generally clear and that levels of littoral zone turbidity experienced at some unarmoured sites may create systemic disturbance notwithstanding remarks made in chapter three. I replaced (7) proportion of green sunfish, with (7) proportion of johnny darters following the hitherto untested suggestion by Karr (1981). Lastly, I removed the metric, (11) proportion of hybrids, because of the difficulty determining hybridisation within the *Cyprinidae* and among some members of the *Percidae* families in a field setting.

4.1.2.2 Scoring the IBI

In chapter two (section 2) I indicated that the study area had been subject to a primary, macro scale disturbance in the form of damming for hydroelectricity production, the consequent flooding of the river and creation of a new littoral zone that resulted. As such, the study area itself could not provide a pristine site against which to compare the subsequent secondary disturbances of either: 1) application of rip rap armouring, or 2) systematic erosion as a result of the original disturbance. To determine if rip rap armouring offsets the effects of erosion and acts beneficially on fish habitat, or if it plays some other (positive or negative) role, demands that the two sites types (armoured / unarmoured) be compared against one another within the area of primary impact.

²⁰

For differences in turbidity values by shoreline type, see chapter 6.

However, this process meant that the scoring of IBI in its original form could not be used. So, rather than comparing site scores against a “pristine” benchmark (Karr 1981), I used a novel approach, where IBI scores were generated by comparing individual site information to the range of observed variability from within the study area. For each metric, raw data were sorted into quartiles (Table 4.2). A site was scored “1” where the observed value for the metric fell below the first quartile threshold. Similarly, a score of “5” was assigned to those sites for which the observation occurred above the fourth quartile mark. Sites whose observation values fell within the second and third quartiles scored “3”. One exception to this method was required for the metric “number of sucker species” where ancillary abundance information was used in assigning quartiles to avoid a zero variance²¹. Site scores for each metric were then summed to produce a total IBI site score according to the original index method (Table 4.2).

4.1.2.3 Statistical analysis

Principal component analysis (PCA, Legendre & Legendre 1998) was performed on natural log transformed ($\ln + 1$), pooled (2002-3) species composition data to generate object (site) and variable (fish species) scores using a correlation association matrix on the full species set ($p = 26$). PCA is the obvious ordination method to use, especially if neither a distance nor similarity matrix is used, (Manly 1994). Although Legendre and

²¹

Zero variance does not present a problem for use in the IBI itself, but is problematic for PCA (or any other of the multivariate methods) which was used to analyse the index information.

Table 4.2 IBI raw metric data (2002-2003 pooled sample) and ensuing IBI metric scores in parentheses. For metrics 7,8 & 11, lower values received higher scores. * indicates sites armoured with rip rap.

IBI Metric Data & Site Scores

	1	2	3	4	5	6	7	8	9	10	11	
	Number of						Proportion of					
	Species	Benthic species	Centrarchid species	Sucker species	Stickleback species	Individuals in sample	Johnny darter	Omnivores	Insectivorous Cyprinids	Top Carnivores	DELTS	Total IBI Site Score
Site 1	19(5)	6(5)	1(3)	1(3)	1(3)	1515(3)	0.326(1)	5.94(1)	17.0(3)	14.719(3)	0.660(5)	35
Site 2	17(3)	5(3)	1(3)	1(3)	1(3)	978(1)	0.308(3)	20.04(1)	16.2(1)	12.986(3)	7.157(1)	25
Site 3	18(3)	4(3)	1(3)	1(3)	1(3)	1071(3)	0.565(1)	2.33(3)	6.5(1)	8.217(1)	2.801(3)	27
Site 4	14(1)	4(3)	1(3)	1(3)	0(1)	953(1)	0.773(1)	0.84(3)	4.7(1)	5.352(1)	6.296(1)	19
Site 5*	16(3)	5(3)	1(3)	1(3)	1(3)	1305(3)	0.255(3)	3.83(1)	19.4(3)	18.238(5)	1.533(3)	33
Site 6*	19(5)	5(3)	1(3)	1(3)	1(3)	2027(3)	0.038(5)	0.20(5)	39.6(3)	18.106(3)	1.480(3)	39
Site 7*	18(3)	6(5)	1(3)	1(3)	2(5)	1293(3)	0.087(3)	1.24(3)	40.6(3)	27.920(5)	5.414(1)	37
Site 8	18(3)	3(1)	1(3)	1(3)	1(3)	1423(3)	0.203(3)	0.91(3)	23.3(3)	8.152(1)	4.919(3)	29
Site 9	14(1)	3(1)	1(3)	1(3)	0(1)	750(1)	0.240(3)	0.27(3)	35.1(3)	18.133(5)	0.000(5)	29
Site 10*	19(5)	5(5)	1(3)	1(3)	1(3)	10500(5)	0.005(5)	0.01(5)	64.0(5)	10.610(3)	0.667(3)	45
Site 11*	15(1)	3(1)	1(3)	1(3)	1(3)	4707(5)	0.031(5)	0.23(5)	56.6(5)	12.195(3)	0.000(5)	39
Site 12*	20(5)	4(3)	2(5)	1(3)	2(5)	3465(5)	0.051(3)	0.40(3)	57.6(5)	14.315(3)	1.154(3)	43
1st quartile	15.75	3.75	1	1	1	1048	0.047	0.26	16.8	10.011	0.665	
3 rd quartile	18	5	1	1	1	2387	0.312	2.71	44.6	18.113	5.043	

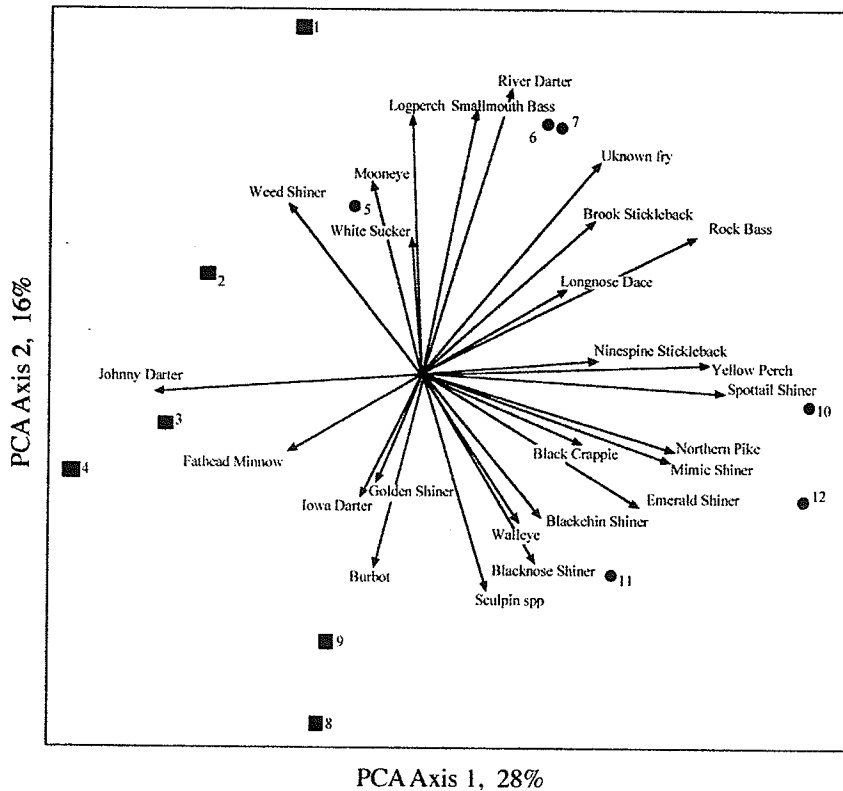
Legendre (1998) state that PCA does not always require multinormal distributions of the data, the optimal use of PCA occurs when the data are multivariate normal (see Fig 4.1). PCA 1 component scores were tested for correlation with derived IBI site scores (Table 4.2) using Spearman Rank correlation. A *t*-test was performed on PCA 1 scores and a Mann-Whitney U test was performed on IBI scores to determine if differences between site types were statistically significant. Lastly, PCA was conducted on the IBI metrics to examine the contribution of each metric to the total IBI score.

4.1.3 Results

4.1.3.1 Species composition

The first two PCA axes for the fish abundance data ($p = 26$ species) account for 44% of the trended information (28% and 16% respectively). The locations of the sites on the biplot (Fig 4.2) form an object constellation, indicating the main variation in species composition occurs across site types (i.e. armoured vs unarmoured), even though PCA is not *senso stricto* a discriminant method. All armoured sites (except site 5) had positive object scores on PCA 1 whereas object scores for all unarmoured sites on PCA 1 were negative. This interpretation of PCA 1 was verified by analysis of PCA 1 scores where armoured sites were found to be significantly different (*t*-test, $P < 0.001$; $\alpha = 0.05$; $\beta = 0.988$) from unarmoured sites. In contrast, none of the subsequent axes can be interpreted to separate site types.

Fig 4.2 2002-2003 species composition PCA biplot showing location of all sites (objects) in species-space ($p=26$), structure correlations indicated by vectors. Sites 1-4 & 8-9 unarmoured designated by ■; sites 5-7 & 10-12 armoured with rip rap, designated by ●. Percentage of linearly trended information provided on axis labels. Lower (negative) object scores occur closest to the origin.



4.1.3.2 Modified IBI

The modified IBI scored armoured sites higher than unarmoured sites in all but one case (Table 4.2). The modified IBI scores were weighted (i.e 12/11* score) so the 11 metric boreal IBI total scores were commensurable with Karr's (1981) 12 metric IBI habitat quality rankings. These scores indicate the range of habitat quality in the study area varies from "very poor" to "good" (Table 4.3). A Mann-Whitney U test on IBI scores also

Table 4.3 Karr's (1981) nine classes of fish habitat quality based on total (60 point) IBI score. VP - very poor (≤ 23); P-VP - poor / very poor (24-27); P - poor (28-35); F-P - fair / poor (36-38); F - fair (39-44); G-F - good / fair (45-47); G - good (48-52); E-G - excellent - good (53-56); E - excellent (57-60). * indicates armoured sites.

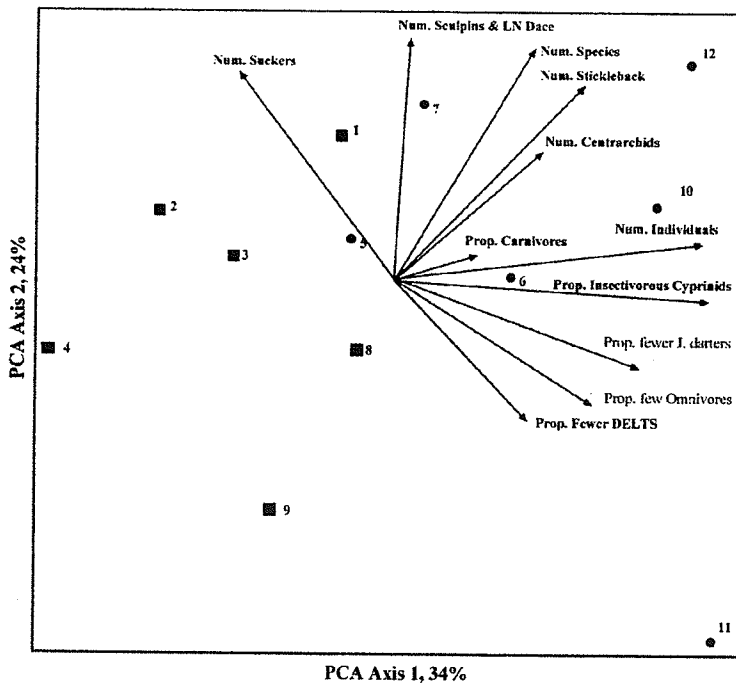
Site scores and habitat classification												
	Site 1	Site 2	Site 3	Site 4	Site 5*	Site 6*	Site 7*	Site 8	Site 9	Site 10*	Site 11*	Site 12*
Total IBI Score	35	25	27	19	33	39	37	29	29	45	39	43
Adjusted IBI score	38	27	29	21	36	43	40	32	32	49	43	47
Habitat Rating	F-P	P-VP	P	VP	F-P	F	F	P	P	G	F	G-F

showed a significant difference between armoured and unarmoured site types ($P < 0.001$; $\alpha = 0.05$; $\beta = 0.974$).

4.1.3.3 Metric redundancy

The 11 metrics of the modified IBI had high redundancy (i.e. most metrics produced similar results) and all but one (metric 4 - number of sucker species) had a positive structure correlation with the first axis. Concomitantly, site 11 occurs as an outlier, (Fig 4.3). However, the pre-analysis adjustment of the site 11 metric score for “number of

Fig 4.3 IBI metrics analysis PCA biplot showing location of all sites ($n = 12$) in IBI metric ($n = 11$) space, structure correlations indicated by vectors. For metrics scored in reverse (Table 4.1) the term “fewer” has been added within the metric label. Sites 1- 4; 8 & 9 are unarmoured sites and are designated by ■; sites 5 - 7; 10-12 are armoured with rip rap and designated by ●. Percentage of linearly trended information provided on the axis labels.



sucker species” to reduce zero variance (see above) seems to account both for the position of site 11 in IBI space, as well as for the conflicting contribution of this metric. Notwithstanding, overall, the first two PCA axes accounted for 58% of the total variation (34% and 24 % respectively). As with species composition, site type was separated on PCA 1, again with the exception of site 5.

4.1.3.4 Correlation of PCA to IBI

The first axis from the species composition analysis was highly correlated with the modified IBI scores (Spearman Rank correlation = 0.937; $p < 0.001$). This indicates a strong relationship between species composition and the modified IBI developed for the boreal river study site. The correlation between the first axis of the fish assemblage PCA on the IBI metrics and the modified IBI was also high 0.98 ($p < .001$).

4.1.4 DISCUSSION

4.1.4.1 Modified IBI

Among the four modifications made to Karr’s (1981) original IBI, one of the most substantial alterations was to replace the metric “number of intolerant species” with “number of Stickleback species” (Table 4.1). Among those allocated to the original metric, Karr (1981) listed *C. bairdi* (mottled sculpin), *Notropis heterolepis* (blacknose shiner) and *Ambloplites rupestris* (rock bass) as species least tolerant of suspended solids. However, within my data, these species were already assigned to metrics two;

nine; and three and ten respectively, apart from their obvious inclusion in metrics one and six. Therefore, to provide a more balanced species-to-metric allocation I followed the ecological assessments of both Scott and Crossman (1998) and Goldstein and Simon (1999) and selected the *Gasterosteidae* as a representative family intolerant of abioseston which I considered to be the chief cause of *in situ* turbidity, based on extensive observation of the eroding clay shoreline. Ecological justification for this decision was confirmed in my data where both *Culaea inconstans* (brook stickleback) and *Pungitius pungitius* (ninespine stickleback), had strong positive structural correlations on PCA 1 as did rock bass (Fig 4.2). (PCA 1 and 2 variable scores for brook stickleback were 0.544, 0.401; ninespine stickleback - 0.538, 0.038; rock bass - 0.866, 0.365). In contrast, both sculpin species and blacknose shiner generally had low abundance and a weak correlation with PCA 1 (PCA 1 and 2 variable scores for blacknose shiner were 0.370, -0.488; the sculpins - 0.222, -0.563). Therefore, use of these latter species for this metric likely would have contributed to ambiguous results. However, as it stands, this modification helped to differentiate between sites as is demonstrated by strong positive metric scores on both PCA 1 and 2 (Fig 4.3), (PCA 1 and 2 variable scores for this metric: -0.563; 0.630). The ecological information contained within this metric offers insight into habitat beyond the strong signal driven by metric six (number of individuals) and metric nine (proportion insectivorous cyprinids) – especially given the numerical dominance of cyprinids within the samples. Specifically, only sites seven and twelve provided instances of both stickleback species whereas all other sites produced only brook stickleback. The presence of ninespine stickleback at these two sites alone helped

to account for the positions of these sites in IBI space relative to the other ten sites (Fig 4.3). At the same time, the difference in site position of these two sites in IBI space relative to one another is a result of higher PCA 1 and PCA 2 site scores for site 12 based on higher IBI scores for both metrics six and nine. Therefore, this altered metric provided additional subtle information about habitats, but information which remained consistent with results produced by other key metrics. Overall, these results indicate that the modified metric “number of sticklebacks species” is a valid replacement for the metric “number of intolerant species” in the boreal IBI.

A second important modification was to find a suitable replacement for the original metric “proportion of green sunfish” since this species does not form a part of the Winnipeg River fish assemblage (Stewart and Watkinson 2004). Karr (1981) suggested that Johnny darter in the absence of other darter species may be a sign of habitat degradation. To date, there is little evidence that this line of inquiry has been pursued. However, the modified metric based on this suggestion was well supported within my data. In particular, sites 3 and 4, which ranked among the lowest IBI scores, were dominated by johnny darters. This species had the strongest negative structural correlation with PCA 1 and little correlation with PCA 2 (Fig 4.2)²². The influence of this metric is also evident in the analysis of IBI metrics, where unarmoured sites 3 & 4 were separated from armoured sites 6, 7, 10, 11 & 12 (Fig 4.3). These results support Karr’s

²²

PCA 1 & 2 variable scores for johnny darter -0.845, -0.048.

suggestion that a high proportion of johnny darters may be indicative of degraded habitats; although what constitutes a “high proportion” quantitatively is by no means certain. What does seem clear is that on the one hand, the presence of johnny darters – at least among other benthic species - should be interpreted as a sign of ecological health. On the other hand, the presence of these darters in the absence of other benthic species likely illustrates a sign of habitat ill health. This finding creates a conundrum for officials who must decide how to treat these habitats. The issue facing the manager becomes how to interpret a preponderance of johnny darters since this information suggests the habitat is either trending towards decay or conversely, at the early stages of recovery. Nevertheless, the empirical evidence that *E. nigrum* has this potential bi-polar indicator role is an important contribution coming out of this research.

4.1.4.2 IBI scoring

Apart from the specific modifications to the IBI metrics for the boreal river study area, I also developed a novel approach to scoring and applying the model itself. Given both the paucity of data regarding the historical range of variability in this system, and the research design which focussed on the role of secondary impacts, I was unable to define a “pristine” reference condition. Rather than score against an “outside” standard, I developed a quartile scoring technique. I found that after changes to the metrics to reflect better the local ichthyofauna, the quartile method of scoring generated a normal distribution of metrics scores [“1” (n=3); “3” (n=6); “5” (n=3)] for seven of the eleven metrics. These results suggest that my modified scoring technique provides a practical

adaptation of IBI in and of itself. Furthermore, this modification of the original basis for scoring IBI raises what is perhaps a semantic, but otherwise legitimate issue with respect to biotic integrity. This issue is discussed below.

Karr (1981) used total index scores to qualify habitat. I adjusted my 11 metric IBI to a 60 point total score equivalent in order to apply Karr's classifications to the Pinawa sites (Table 4.3). The value in these ratings is twofold: 1) to demonstrate the qualitative range of variability which can occur across even a small study area; and 2) to provide a standardised score useful for the comparison of these IBI scores to those generated in other studies representing various locales over time. For instance, Karr (1981) listed one excellent and several good sites. In contrast, no sites in the Pinawa study area ranked so well. Although these two examples are in no way sufficient basis for a conclusion, scores provided in this manner might form the basis for inquiry into the possible precipitous decline in the health of aquatic systems over the last quarter century.

4.1.4.3 Metric redundancy and correlation

The modification of any IBI requires that the new metrics be evaluated to understand their role in contributing to the overall assessment of habitat which they are designed to measure. On the one hand, using PCA as the multivariate method for determining objects in species space is a valuable method, since it seeks to reduce the number of variables which need to be considered (Manly 1994). Thus the correlation between IBI scores and PCA scores become meaningful as a comparison of two "summaries" while

simultaneously illustrating which metric(s) drive the index and which of the metrics provide subtleties. In an additive index such as IBI, metric redundancy is “built-in” to avoid high values in one metric being muted by lower values in others. In an early analysis of the original IBI, Angermeier and Karr (1986) found high metric redundancy. The PCA biplot for this boreal IBI shows that the modified metrics were also highly redundant (Fig. 4.3). The amount of shared linear redundancy on the first axis was 34% and is consistent with the findings of Angermeier and Karr (1986). Since the correlation of this axis to the IBI was high (0.98), essentially only the structure correlations between the metrics and PCA1 are interpretable in the context of the IBI. In this regard, all metrics trended positively with the first axis with the exception of metric 4, “number of sucker species”. Interpretation of this metric is discussed above. Metrics most strongly trended with PCA 1 include “number of individuals” and “proportion insectivorous cyprinids” – a metric reflecting schooling species and therefore commanding a strong correspondence to number of individuals. In contrast, “number of benthic species” is only slightly trended with PCA 1, but does trend highly along PCA 2. In the sampled species assemblage analysis I found that benthic species did not, as a group, share similar structure correlations and were generally independent of each other (Fig 4.2). For example, site 11 had the highest abundance of sculpins but few other benthic species and was ‘pulled’ to an outlier position in IBI metrics-space. However, some caution needs to be exercised with this statement, since the reduced score “forced” onto site 11 for metric 4 (number of sucker species) will have undoubtedly contributed to this outlier position as well (see above).

Substantial pairwise correlations were found between the modified IBI and the first axes derived from the species composition analysis and the IBI metrics analysis. Positive correlations were anticipated because the index is based on metrics that are recombined subsets of the species composition data. However, given that the index is additive and does not consider variable interaction terms and, that species are combined into ecological groupings (Fore *et al.*, 1994), the strength of these relationships needed to be determined. Indeed, the explained variance (using the r^2 equivalent, see Legendre and Legendre 1998 for definitions) between the IBI scores and first axis scores was 88% and 96% for the fish assemblage and metrics analyses respectively. Given the strength of these relationships, the second and subsequent axes provide little information with respect to interpreting the IBI. Nevertheless, there remains a large percentage of total variance unaccounted for by the first two axes of both analyses (66% for the fish assemblage data, and 42% for the metrics data). Since PCA axes are orthogonal, and the IBI is essentially summarized on PCA 1, (but see my discussion of species richness on PCA in chapter 3), this unaccounted variance represents information independent of the index as it is currently formulated. Based on these findings, there is opportunity to investigate whether the second and/or subsequent axes have a structural interpretation beyond that discussed earlier.

4.1.4.4 Site type differences

PCA situates ordinated data in such a way that sites which are similar are plotted close together while those that are different are positioned far apart (ter Braak 1987; Legendre

and Legendre 1998). Site constellations for armoured and unarmoured sites were significantly separated in both species-space and IBI metrics-space. All armoured sites (except site 5) scored positively on PCA 1, whereas all unarmoured sites scored negative values on this axis. Site type does not separate out on PCA 2 in either species-space or IBI metrics-space. Instead, scores on PCA 2 appear to be related to other physical properties of the habitat such as water depth, riverbank slope or other environmental variables - see chapter six. However, it should be noted that physical properties such as these are directly linked to species specific habitat requirements (eg. Casselman and Lewis, 1996), thus making it difficult to unbundle species richness from habitat properties. Still, the greatest amount of variation, in both species-space and metrics-space, is captured between site types on axis 1. Indeed, correlation was strong between IBI metrics and species assemblage PCA 1 scores ($r^2 = 0.94$) although some site positions were different on this axis. For example, sites 6 and 7 moved relative to one another on the first axis, because of differences introduced during scoring (Table 4.2). Scoring differences are most influential when they occur between metrics that trend strongly on PCA 1. This is consistent with the intention of IBI which attempts to maximise difference based on important ecological properties.

4.1.4.5 Biotic health & integrity

Both IBI and PCA 1 scores were consistently higher for rip rap armoured sites than for unarmoured sites. These results suggest that rip rap armoured sites may be associated with fish assemblages that are similar to those found through-out the predominantly

rocky geological formations of the Winnipeg River. Therefore, at the small scale it appears that rip rap armouring pushes the local system in a direction closer to the norm of the larger river system, suggesting higher “biotic integrity”. I examine this idea further below (sec. 4.2). Karr and Dudley (1981), define biotic integrity “as the ability to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitat of the region.” However, Karr’s seminal paper in which IBI was first posited (Karr 1981) makes no distinction between biotic health and biotic integrity: “...my contention is that by carefully monitoring fishes, one can rapidly assess the health (“biotic integrity” - *sic*) of a local water resource.” The scoring method of applying sample data against pristine site information indicates clearly that ecological integrity is being measured – that is the degree to which the environment has been altered from its natural “pristine” condition (by which we should read undue human influence). On the other hand, sound ecological health, while less appealing to those concerned with the ideals associated with deep ecology, is likely the highest standard to which we can aspire nowadays given the large scale nature of many environmental problems. Therefore, those practices which belong, either wittingly or unwittingly, to ecological restoration programmes may indeed deliver strong ecological health, but may never achieve any measure of ecological integrity. From the results found within this study, it may be fair to say that some uses of rip rap armouring in boreal ecosystems can contribute to increased ecological health, but not to ecological integrity - since it occurs by necessity as a human induced impact. As such, shoreline armouring shifts the environment away from its

natural state, irrespective of the results it may achieve. On the other hand, Karr's (apparently deliberate) blurring of the distinction between biotic (implied ecological) health and biotic (implied ecological) integrity suggests that ecological health can be measured using IBI. As such, modifications to the scoring method (that is where pristine sites are *not* used as a benchmark) are perfectly acceptable.

4.2 RIP RAP AS AN ANALOGUE FOR NATURAL ROCK SHORELINES

The 2004 field season was dedicated to determining the representativeness of the Pinawa study area (chapter two) and to pursuing the idea that rip rap armoured shorelines may represent natural exposed rock shorelines (NERS) better than do their unarmoured counterparts – or at least those unarmoured shorelines which are susceptible to systemic erosion as a result of hydro electric development. This investigation poses the question: are rip rap armoured shorelines a human-made analogue to natural rock exposed shorelines?

4.2.1 METHODS

The proliferation of rip rap and / or clay based shoreline in the study area presented few opportunities to find natural rock exposed shorelines, especially with anything more than nominal interspersions between sites and site types. Nevertheless, four additional NERS sample sites were added to the Pinawa study area sites as outlined above (Table 2.5).

Fishes from the littoral zone of each of the sixteen 2004 Pinawa sites were sampled using

the modified beach seine method with a one week sampling period as discussed in chapter three.

The 2004 Pinawa fish species composition information was analysed using both non-hierarchical cluster analysis and ordination techniques. Firstly, transformed data ($\ln + 1$) were forced into one of three clusters (using a global optimisation clustering procedure) based on the logic that all sites would form clusters according to site type (armoured, unarmoured and NERS). An alternative clustering analysis was also employed, based on four clusters - one cluster for each of the NERS sites. The logic utilised here sought to uncover any potential pattern between armoured and unarmoured sites that combined with each of the individual NERS sites. However, no such patterns emerged. Indeed, those sites which did coalesce around NERS sites did so despite relatively large distance between the NERS sites and the other sample sites which formed the cluster. Conversely, sites with physical proximity to a NERS site did not cluster with that NERS site. For example, although NERS site 14 was located between sites 3-4, it formed a cluster with sites 7 and 12; while sites 3 and 4 (among others) joined the cluster which included NERS site 16. These results further minimise concerns of spatial autocorrelation within the data, discussed in chapter two. The data set was also run through three ordination techniques - PCA, CA and PCoA (principal coordinates analysis) in search of strong interpretable results that matched the cluster analysis.

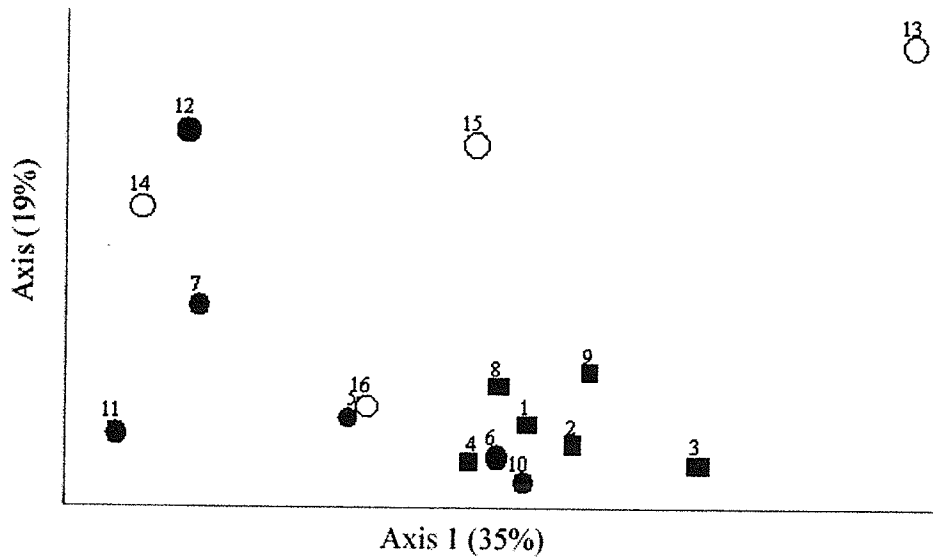
4.2.2 RESULTS

The three cluster method (Table 4.4) and PCoA analysis (Euclidean distance association matrix) (Fig 4.4) provided the best set of analyses whose results were comparable and were therefore mutually reinforcing. That is, PCoA sites formed an object constellation which was interpretable with respect to the cluster analysis. In contrast, both PCA and CA ordinations of this data set accounted for much less trended information on Axis 1 (22% and 25% respectively) than did PCoA (35%) and considerably less cumulative trended information on the first three axes (PCA = 56%; CA= 59%; PCoA = 71%).

Table 4.4. Pinawa study area sites 2004 species composition cluster membership.

	Cluster		
	1	2	3
Membership	7, 11, 12, (armoured)	13, 15 (NERS)	1,2,3,4,8,9 (unarmoured)
	14 (NERS)		5,6, 10 (armoured)
			16 (NERS)

Fig 4.4. 2004 PCoA species composition object scatterplot of Pinawa sample sites. Unarmoured site designated by ■; armoured sites by ●; and NERS sites by ○.



4.2.3 DISCUSSION

I account for the difference in CA, PCA and PCoA results of the 2004 data due to unusual ecological phenomena both generally and for the Pinawa study area associated with this field season. Firstly, some sites exhibited few or no fishes despite their relative profusion in previous years. Secondly, the distribution of site ranks (by abundance and by species richness) for 2004 did not match well with the distribution of ranks shown during both 2002 and 2003. Consequently, some sites which had high abundances did so with low species richness and other sites had high richness, but comparatively low abundance - conditions in which PCA does not perform well (*sensu* Legendre and Legendre 1998). A comparison of site species richness and abundance rankings across 2002, 2003 and 2004 showed that 2004 was an unusual year. Although 2004 sites which ranked in both the top and third quartiles (by abundance) showed similar rank distributions (by richness) as was the case in 2002-2003, this similarity did not extend to the distributions for the other two quartiles. For example:

- a. 2004 sites which ranked in the second quartile (by abundance) ranked only in the top and bottom quartiles (by richness) for 2004 data. However, both 2002 and 2003 sites which ranked in the second quartile (by abundance) always (but one) ranked in either of the top two quartiles (by richness).
- b. 2002-3, sites which scored in the bottom quartile (by abundance), also

generally scored in the bottom quartile (by richness). However, 2004 sites which scored in the bottom quartile (by abundance) scored in all but the top quartile (by species richness).

Overall, approximately half the site ranks (by abundance) in 2004 followed the ranks (by richness) shown in 2002-2003. On the other hand, the other half of the site ranks (by abundance) did not follow the pattern of distribution (by richness) as was seen in 2002-2003. This suggests that richness and abundance were not linked in 2004 like they were in 2002 and 2003. A one-way ANOVA which compared sample site species richness across years, showed that 2004 was significantly different (lower) than both 2002 and 2003 while no difference was found between those two years (Tukey test - $P = <0.001$; $\alpha = 0.050$; $\beta = 0.994$), causing greater difficulty for PCA than for PCoA. Therefore, although field results from 2004 should be treated with some caution - at least insofar as they are expected to represent normal circumstances²³ three comments can still be made, if with some reservation.

During the cold water portion of the open water season (or for the entire 2004 field season which, I contend, acted as an extended spring equivalent), NERS sites were as likely to cluster with armoured sites as with unarmoured sites (Table 4.4 - see clusters 1 & 3) even though the NERS site in each of these two groups was not in close physical

²³

Clearly the 2004 field season represents one extremity of environmental conditions but it seems safe to say that did not represent normal years.

proximity to the majority of sites which otherwise formed the cluster. More importantly, under the circumstances found in 2004, NERS sites were somewhat more likely to form unique clusters (Table 4.4 - cluster 2).

Armoured sites analysed during this period consistently appeared to be relatively poorer quality habitat than unarmoured sites. This finding supports the hypothesis posited in chapter three; that unarmoured sites are associated with shallower, warmer, more turbid water and form comparatively better habitat during spring (or spring-like conditions) than do armoured sites. Presumably, the relative value of each site type changes over the course of a normal season which accounts for the higher habitat rankings assigned to armoured sites using both PCA and IBI methods above (eg. sec. 4.1.4.4).

Earlier in this chapter (sec. 4.1.4.5), I suggested that rip rap armouring may drive those portions of the adjacent aquatic system towards the natural rock exposed shoreline type - that is armoured shorelines may occur at some midpoint on a gradient between eroding shorelines and enduring shorelines. However, the results from the analysis of this data suggest that this is not necessarily the case. Rather, it appears that armoured and unarmoured sites may exist along one gradient, while natural shorelines show habitat characteristics along an orthogonal (or at least different) gradient (Randall and Minns 2002 reported a similar phenomenon). What this might suggest is that while rip rap may push the aquatic system towards better aquatic health (although during the cold water portion of the season or in cool weather years such as 2004, the reverse appears to be

true) natural shorelines possess still other features which are better symbols of ecological integrity. This distinction would thus account for the unique position of sites 13 and 15 (natural exposed rock shorelines) both as a unique cluster and on the PCOA scattergram (Fig 4.4). From the 2004 data, it appears that rip rap armoured shorelines are not good analogues for naturally occurring exposed bedrock shorelines. This is not entirely surprising. Indeed, NERS sites have bedrock not only at the littoral / terrestrial interface, but also solid smooth rock as a continuous substrate through out the littoral zone. Rip rap sites by necessity were once clay bank shore and therefore are a hybrid of rock bank and muck substrate. This suggests that while ecological health may be enhanced through the use of rip rap armouring, ecological integrity cannot be. Furthermore, the hybrid site type may be better suited to a different species assemblage than the NERS sites.

The notion of the separation of ecological health and ecological integrity, at least with respect to some sites, is an interesting issue. If armoured shorelines exhibit properties associated with better ecological health, but not with better ecological integrity, then there is a second conundrum on which I ruminate in the form of a double question: what if metric(s) of an index of biotic integrity no longer represent metrics of biotic health? What then should be the goal for habitat managers? Consider for a moment an unrelated example: There is evidence of an increased Mid-Continental Lesser Snow Goose (*Anser caerulescens caerulescens*) population in Canada's Arctic, attributable in part to improved foraging conditions along the bird's main migratory routes (Jefferies, 2000). There is also evidence that global climate change has meant a general warming of the

Arctic, thereby exacerbating threats to this the sensitive landscape already posed by mounting foraging pressure from the growing snow goose population. A hypothetical Arctic index of biotic integrity would be expected to generate high index scores based upon the high abundances of naturally occurring migratory fowl. However, these high scores may not represent ecosystem health *because the circumstances surrounding species presumed to be reflective of ecological integrity are likely changing more quickly than species adaptations can reflect these environmental changes*. Returning to the modifications made for the boreal river IBI: it is clear that testing the inclusion of various metrics and recognising the potential for metrics to behave in a bi-polar manner must be part of the development and application of any new version of IBI. It is also clear that managers who use IBI to make decisions about habitat must be clear about their goals with respect to ecological health and integrity.

CHAPTER 5
SITE TYPE PRODUCTIVE CAPACITY OF THE LITTORAL ZONE
IN A LARGE BOREAL RIVER

INTRODUCTION

The analysis of species composition information (chapter 4) can play an important role in demonstrating structural differences in a resident fish assemblage. Implications about habitat can then be derived from this information. Similarly, the ecological principle of production applied specifically to fish communities can provide information about the underlying performance of habitats and may contribute to our understanding of the resident species composition. Furthermore, the analysis of ecosystem production may reveal trends about habitat response to disturbance that are different from those trends demonstrated by analysis of species composition since we understand that fish production generally has less natural variability than does fish species composition (Minns *et al.*, 1996). Specifically, physical alterations to localised habitat may change fish species composition and distribution within the ecosystem without changing its fish production (Jones *et al.*, 1996; Minns *et al.*, 1996). Therefore, differences established between sites and site-type based on species composition may not be distinguished when measured on the basis of production. The reason being, is that although production and species composition are usually correlated, the relationship is not perfect. For example, Randall and Minns (2002) assessed nearshore communities in the Great Lakes using an IBI and Habitat Productivity Index (HPI), and they found that while species richness correlated well with IBI, fish community biomass correlated more closely with the HPI. Therefore, before drawing any conclusions about habitat quality associated with specific

sites or site type, it is important to examine the impacts of habitat alterations both from the perspective of species composition and the perspective of production (Randall and Minns 2002).

Apart from the ecological rationale, there is also an operational basis for pursuing an assessment of ecosystem production. The 1986 DFO no-net-loss policy (NNL) was developed to protect and ensure the net gain of fish habitat in Canada. An essential component of the policy was to reduce the loss of productive capacity of these environments (Jones *et al.*, 1996). However, the challenge for scientists and habitat managers alike has been to determine: 1) how to quantify productive capacity; and 2) how to link changes in ecological production to human induced habitat alterations (Jones *et al.*, 1996). These points are particularly salient, since we operate fish habitat management on at least two assumptions: 1) that physical habitat and productive capacity have a direct (and understood) relationship (Pratt and Smokorowski 2003); and 2) that we know or can determine the relationships between species and their habitats (Minns 1997a). However, these conditions cannot always be met and thus we use surrogate measures for determining human impact on productivity and thence onto habitat (Jones *et al.*, 2003). Among these surrogates, integrative indicators such as production and community indices may be the most useful measures of detecting habitat responses to human impact (Minns *et al.*, 1996).

If we accept that human disturbance is more likely to result in changes to species

composition than in changes to production, it would seem reasonable to protect species assemblage (and therefore, simultaneously protect production) as the proper approach to managing ecosystems. However, since unbundling the role of human impact on species composition from other mechanisms of change is often quite difficult, and since changes in production and species composition are not perfectly linked, management of the ecological underpinnings of species composition – that is ecological production (“productivity capacity”) – sidesteps the difficulty of determining causally the effects of human impact through assessment at the level of species composition, and instead places its focus on the protection of production as the most fundamental method of protecting the general function of habitat.

Operationally therefore, projects proposals that trigger a HADD decision framework and any associated compensation are evaluated according to changes in “productive capacity.” However, since the productive competence of any given habitat has no necessary, prescriptive allocation of resource partitioning among the range of rare to abundant species within an assemblage, the protection of species composition is integrated into “productive capacity” and thus both production and composition are necessary components of habitat protection.

Overall, this project examines physical alteration of habitat from a stratified view of disturbance and therefore takes a perspective which is somewhat atypical. The primary disturbance occurred in the 1950s with hydro electric development that resulted in

isolated flooding and the creation of a new littoral zone along some portions of the Winnipeg River shoreline. Immediate and direct changes to species composition and productive capacity arising from this initial disturbance are not the subject of this investigation. Instead I measured impacts of two contrasting management strategies (i.e. continued systematic erosion as a result of flooding or rip rap armouring) as a choice of secondary disturbance; either of which is layered upon this backdrop of primary disturbance. As such, this work is set apart from other project assessment protocols which generally have as their environmental context a long temporal gradient of human induced ecological change rather than the acute change associated with the creation of reservoirs²⁴. This chapter examines the productive capacity of the Pinawa study area to determine if production information either corroborates or rejects my conclusions about site type, derived from the analysis of species composition data in chapter four. In this chapter, I assess two methods for determining the productive capacity of riverine littoral communities. I also provide a preliminary examination of the productive capacity of bedrock substrate which has been for too long considered poor quality habitat (eg. Frezza and Minns 2002a.)

5.1.1 What is productive capacity?

Productive capacity has often been referenced in the literature (Jones *et al.*, 1996; Minns *et al.*, 1996; Randall and Minns 2002) with respect to the 1986 DFO policy. There, it is

²⁴

This is especially the case in lacustrine environments such as the Great Lakes where considerable work on productive capacity has been carried out.

defined as “the maximum natural capacity of habitats to produce healthy fish, safe for human consumption, or to support or produce aquatic organisms upon which fish depend.” (DFO 1981; DFO 1998). In their appraisals of this definition Minns (1997a) and Randall and Minns (2002) both noted three aspects in particular: “production”; “maximum” and “natural”. Each of these aspects requires further comment.

Although generally speaking production refers to the total energy flux in a community (Wetzel 2001) it can also refer to specific populations or groups of organisms (Wetzel 2001)²⁵. For purposes of this study, production is the sum of all coexisting fish in an area per unit time (Minns 1995). Since production is rooted in, and dependent upon, photosynthetic activity, (Minns 1995) not only is production an acceptable measure of productive capacity (Jones *et al.*, 1996; Minns *et al.*, 1996), but it has also been gauged to be “the best indicator of the quantitative performance of a fish population in any type of habitat” (Randall and Minns 2000). Accordingly, change in production is a primary indication of habitat loss, (Jones *et al.*, 1996) and in such manner production can be linked to habitat alteration. The descriptor “maximum” implies the ecological concept of carrying capacity (Jones *et al.*, 1996). Operationally, habitats which are not functioning at their natural carrying capacity must therefore be considered somehow compromised. However, the measurement of carrying capacity is an elusive task. More simply, Minns (1997a) defined maximum production as “the sum of all production accrued by all stocks

²⁵

Production and productivity refer here exclusively to fishes and not to primary production or other forms of secondary production.

during the time they spend any part of their life history in that area” (Minns 1997a). The aggregation of many instantaneous measures over time and across a given spatial unit preempts the potential criticism that some habitats may attract organisms temporarily (and thus form only a portion of the species’ habitat) but do not possess the capacity to produce or sustain these populations over the long term. Finally, maximum production for DFO’s purposes, must exist in the context of naturally functioning ecosystems, which implies the preservation and integrity of naturally occurring species composition (Minns 1997a) and productivity. For instance, human induced eutrophication which increased production beyond the upper limits of natural systems would not enhance productive capacity (Minns and Nairn 1999) – at least from the legalistic or regulatory perspective. So, although the 1986 definition technically embodies all trophic levels of the aquatic ecosystem, productive capacity is chiefly concerned with the conversion of energy into the maximum production of naturally occurring species composition, according to natural rates and processes.

5.1.2 Measuring productive capacity

The quantification of productive capacity has received considerable attention and application (Baird and Assoc. 1996; Minns and Nairn 1999; Minns *et al.*, 2001; Frezza and Minns 2002a; Frezza and Minns 2002b), especially since Minns (1995) and Minns *et al.*, (1995) first posed a “Defensible Methods” approach. This technique served to bridge the gap which existed between the normative terms of the 1986 No-Net-Loss policy and the scientific ability to measure productive capacity directly. Indeed, despite the strong

position delivered by the NNL policy, proceedings of a 1996 DFO workshop indicated that a decade after the policy was instituted the measurement and quantification of productive capacity “remains the most important and most difficult aspect of fish habitat management (Levings *et al.*, 1997). As a result, the lack of a standardised method for predicting the habitat loss associated with development projects left the *a posteriori* measurement of biological, physical and chemical parameters as surrogates for assessing changes to productive capacity (Levings *et al.*, 1997). Therefore, in many cases production, and thus implicitly productive capacity, is often measured simply as fish biomass per unit area or time (Minns 1997b).

Although some surrogates for production beyond biomass were developed, such as Habitat Evaluation Procedures (HEP) and the Habitat Suitability Index (HSI) (Terrell *et al.*, 1982), these large scale approaches have tended to focus on the habitat requirements of a single target species (Minns and Nairn 1999). Therefore, these techniques are not necessarily well suited to smaller project-sized analysis or for the broader view of fish habitat implicit in the NNL policy, which encompasses all species of the fish assemblage. In contrast, the main strengths of the Defensible Methods approach (Minns *et al.*, 1995) is that it is predictive and is built on determining net change to habitat (not species), based on the product of the productivity and the physical area under impact. However, the Defensible Methods approach has several facets which make it unusable for river systems. Indeed, Minns *et al.*, (2001) state specifically that the approach was developed for shoreline development projects in the Great Lakes, and should not be used for lotic

systems. The technique has also been tested on small (<25 ha) lakes in the Precambrian Shield where it was shown to be “poorly suited” (Frezza and Minns 2002a). Furthermore, adaptation of the method for lotic systems and small water bodies seems unlikely since much of the method is built upon wind and wave energy driven by fetch length. In contrast, in rivers, boat traffic creates a much larger source of wave energy than does wind (Lubinski and Seagle 1981; Schnick *et al.*, 1982). Therefore, as it stands, there is no good direct measure for the quantification of productive capacity in the littoral zone of lotic systems (Minns and Nairn 1999).

5.1.3 Surrogate measures of productive capacity

5.1.3.1 Biomass

The absence of a “Defensible Methods” companion approach for lotic systems does not mean that the use of surrogate measures of production are without value or without precedent. Indeed, Minns (1997a) admits that “direct measures of productivity will always be a luxury” and in their stead we can successfully use substitutes such as species richness and biomass (Minns 1997b). The viability of biomass as a measure of production was synthesised by Randall and Minns (2000), stating that “for any defined time period, fish production is the product of average biomass and specific growth rate, and therefore, fish biomass and fish production are correlated.” Broadly speaking, the measurement of biomass is a common replacement for production. Examples include Knight and Cooper (1991), Minns *et al.*, (1996), Fischer and Eckmann (1997a), Shields *et al.*, (2000) and Pratt and Smokorowski (2003). Indeed, so prevalent is its use that some

researchers implement biomass as a signal of habitat quality and include it among Index of Biotic Integrity metrics (see Hughes and Oberdorff 1999).

Since production is a measure of some subset of photosynthetic activity, which in itself is primarily a function of the “areal interception of sunlight energy” (Minns 1995), the “common basis for comparison and assessment” of fish production is usually based on area (Minns 1995). I suggest that this two-dimensional assessment illustrates a habitat production nominal efficiency; that is, how well solar energy is converted to fish biomass. However, since fishes must inhabit three-dimensional habitat, it makes sense to consider a measurement of production in terms of water volume. Differences in production values between those measurements based on area and on volume are acknowledged by Mann (1975) and is in keeping with general limnological thinking, where biomass is measured either by area or by volume (Wetzel 2001). Consider this rationale: Generally, there is a negative correlation between water depth and light penetration (due to the colour, opacity and refraction properties of water), notwithstanding the offsetting and potentially confounding effect of the negative correlation between depth and turbidity (due to the increased potential for the physical disturbance of sediments in shallow water), at least within the littoral zone. Therefore, productivity (primary or subsequent) should be negatively correlated (although not linearly) with depth, within the littoral zone. For fixed-area sample sites such as those used in this study, volume is simply a function of bottom contour. Therefore, measures of production per unit volume *may* incorporate the linked physical habitat features which

affect the amount of water (i.e. depth), the quality of water (levels of abioseston; temperature) and their variation *and interaction* with water depth. Measures of production per unit volume *may* represent habitat production effectiveness; that is, how effectively solar energy is converted to fish biomass adjusted for the effects associated with varying depth. Therefore, measures of production per unit volume may account better for both species composition (i.e. species selectivity based on water quality and quantity parameters) and total production differences between sites. For instance, in clear, warm, shallow waters which can otherwise easily be made turbid by wind events or boat traffic, habitat nominal efficiency may be high. However, the conversion of energy to fish in these sites may be rendered relatively ineffective due to the high energy output costs associated with hunting, the repair of damaged tissue or other physiological energy demands after the impact of chronic disturbance is factored in. I hypothesise that volume might be an important variable in the calculation of littoral habitat production given the variability in riverbank slope and the consequent turbidity which can influence light penetration and thus all measures of primary and subsequent production.

5.1.3.2 Habitat Productivity Index (HPI)

Although biomass has a strong positive correlation to production, Randall and Minns (2000) argued that a better estimate of habitat production (and by extension, productive capacity) was possible. Since growth and mortality are related to size, population production varies with fish size. Therefore, a more sensitive method of calculating fish production is available if both biomass and fish size are used as variables (Randall and

Minns 2000). The Habitat Productivity Index (HPI) is a measure of habitat productive capacity which incorporates both these variables. The HPI is calculated by taking the product of unit biomass (P/B ratio) and average species biomass. These products are summed to produce a site score.

Since the HPI is still a relatively recent development, its usefulness as a more sensitive surrogate for production than biomass alone requires further testing. Randall and Minns (2000) suggested specifically that testing in littoral areas was necessary. Randall and Minns (2002) tested the HPI in the nearshore areas of the Great lakes, and they too suggested that further testing was necessary - particularly in "localised habitats."

Following these suggestions, I examine the relationship between HPI scores and biomass values using localised data from the littoral zone of a boreal river. These values are then used either to challenge or to corroborate the site type habitat assessments generated in chapter four.

5.2 METHODS

5.2.1 Fish sampling

Fish were captured from each site as described in chapter three. Capture information recorded for each site included species, abundance, TL (mm) and mass (g) (wet weight). Where multiple fishes of a species were captured, individuals were grouped into batches of similar lengths. A sample TL (mm) was recorded for each batch along with the total

number individuals per batch. The mass for each batch was determined by taking the mass of a subset of the group and recording the number of individuals within the subset. The subset mass was divided by the number of individuals in the subset to produce a derived individual mass. The batch mass was extrapolated by taking the product of the derived individual mass and the total number of individuals in the batch.

5.2.2 Biomass

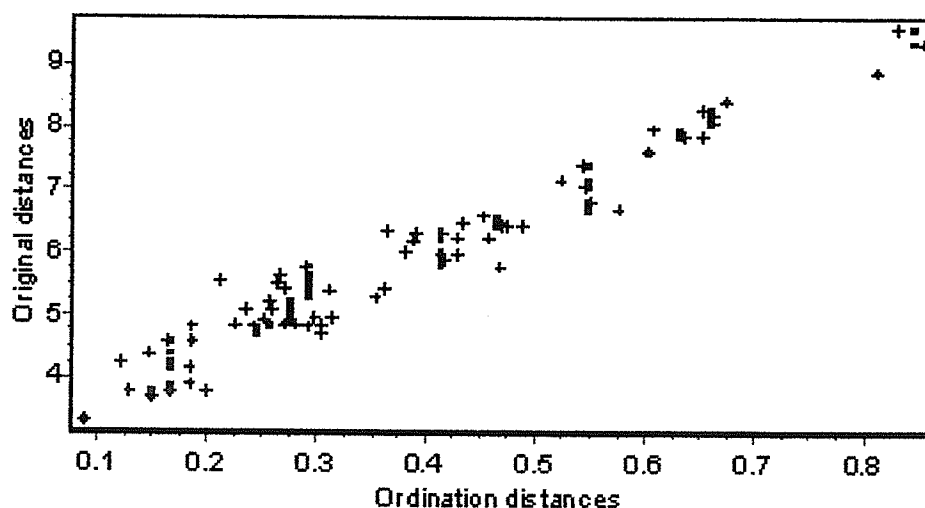
Raw biomass_{fish} data for each of the twelve 2002-2003²⁶ Pinawa sample sites were pooled to compare total production by site-type. Since the raw data did not follow a normal distribution, a Mann-Whitney rank sum parametric test was used to compare site-types. Following the hypothesis that volume-based measures of production might change the relative scores of individual site production, site biomass_{fish} was adjusted for production per unit of water volume sampled. Riverbank and river bottom slope was mapped for each of the twelve main Pinawa sample sites in 2002 and again in 2003. Water depth measurements were taken every 0.5m² against a floating grid (for site contours, see Appendix B, Table B1 for methods, sec. 6.2.2.3) and plotted. The integral of the absolute value of the slope function was calculated to determine the area of water above the curve. Since all sites had the same area, adjusted site biomass_{fish} per unit water sampled was the product of site biomass_{fish} and the volume site adjustment coefficient. Adjusted biomass_{fish} for all sites were sorted by site type (armoured; unarmoured) and compared

²⁶

Analyses were based on 2002-3 data, to match the data set used for analysis in chapter 4.

using a *t*-test. Total site biomass alone reveals little about habitat and multiple species interactions. Since fishes are dependent descriptors it makes sense to analyse these data using multivariate methods (*sensu* Legendre and Legendre 1998) and therefore, all individual species data were examined. Biomass_{fish} data for 2002-2003 were natural log transformed ($\ln+1$) and tested for normality. Although this transformation yielded mostly normal distributions with composition data (chapter 4) it did not render the mass site data normal. Although this condition can provide challenges for choice of analytical method, PCA is reasonably robust even with non-normal data, although strictly speaking it performs best using normal data (Legendre and Legendre 1998). Nevertheless, a Shepard diagram shows the data to be suitable for a meaningful ordination (Fig. 5.1).

Fig 5.1. Shepard diagram on 2002-3 transformed ($\ln + 1$) fish mass data produced using non-metric multidimensional scaling. The constellation suggests a meaningful ordination can be performed (See Legendre and Legendre 1998).



Transformed site biomass_{fish} per species was analysed using PCA to determine if species composition data correlated with species per site production information. Object scores (correlation association matrix) from PCA 1 (species composition - chapter 4) were examined for correlation to object scores (correlation association matrix) from PCA 1 (production). PCA was also performed on a reduced data set (n=25; no *Micropterus dolomieu* - smallmouth bass) to examine the role non-indigenous species had on objects in keeping with the idea of natural production, discussed above. Finally, two additional PCA ordinations were performed on volume-adjusted species biomass_{fish}. PCA on the full data set facilitated comparison between volume adjusted and unadjusted object constellations. PCA was also performed on a reduced data set, removing site singletons – *Pomoxis nigromaculatus* (black crappie); *Lota lota* (burbot); and *Hiodon tergisus* – to examine the influence species richness has on the positioning of site type within various object constellations.

5.2.3 Habitat Productivity Index (HPI) and HPI**

P/B ratios for all species (pooled across sites) within the boreal river littoral zone data set were calculated and compared to species values listed for Eastern Canada (Randall and Minns 2000) (Table 5.1). P/B ratios were calculated using the formula $P/B = 2.64W^{-0.35}$, (Randall and Minns 2000), where W = average weight of fish rather than weight at maturity (Randall and Minns 2002). Boreal littoral fish P/B ratios were used to calculate the HPI for each species. HPI is calculated as fish biomass (per site) * species P/B coefficient. Site HPI scores are the sum of these derived values (Randall and Minns

2002). HPI values were log transformed ($\ln + 1$) to make the data more normal for analysis. However, even after transformation, only data from four sites were normal (Kolmogorov-Smirnov test). Nevertheless, PCA was performed on these values to determine if the greater sensitivity attributed to HPI over biomass alone (Randall and Minns 2000) rendered any different object constellation of the Pinawa sample sites. HPI PCA 1 and total site scores were grouped by site type and compared for differences using *t*-tests. Since HPI measures site efficiency based on the pooled biomass values, across sites, there is little opportunity to use HPI to understand species efficiency within sites. Therefore, a derivative of HPI (i.e. HPI**) was calculated from site specific P/B ratios. Site specific ratios were calculated in the same manner as P/B ratios, using mean weight per site rather than mean weight per 12 sites. PCA was also performed on HPI**.

Table 5.1 Calculated P/B ratios from littoral zone of the Winnipeg River and from Randall and Minns (2000).

Species	Number of fish	Ave species wt (g)	P/B ratio - Wpg River littoral zone	P/B ratio (Randall & Minns 2000)
<i>Hiodon tergisus</i>	5	0.90	2.74	n/a
<i>Notemigonus crysoleucas</i>	5	0.46	3.46	1.59
<i>Notropis atherinoides</i>	5857	0.13	5.42	2.76
<i>Notropis heterodon</i>	11	0.20	4.60	2.73
<i>Notropis heterolepis</i>	10	0.22	4.52	4.00
<i>Notropis hudsonius</i>	10361	0.18	4.84	1.89
<i>Notropis texanus</i>	418	0.40	3.63	n/a
<i>Notropis volucellus</i>	3700	0.12	5.51	2.73
Unknown (shiner) fry	432	0.001	13.25	n/a
<i>Pimephales promelas</i>	12	0.28	4.10	1.51
<i>Rhinichthys cataractae</i>	5	0.39	3.67	1.28
<i>Catostomus commersoni</i>	851	0.32	3.92	0.37
<i>Esox lucius</i>	90	17.40	0.97	0.21
<i>Lota lota</i>	2	29.55	0.81	n/a
<i>Culaea inconstans</i>	126	0.21	4.55	1.99
<i>Pungitius pungitius</i>	3	0.15	5.13	2.73
<i>Cottus bairdi</i> inc <i>C. cognatus</i>	120	0.38	3.71	1.55
<i>Ambloplites rupestris</i>	1687	0.29	4.05	1.51
<i>Micropterus dolomieu</i>	550	0.12	5.50	0.33
<i>Pomoxis nigromaculatus</i>	1	0.70	2.99	0.46
<i>Etheostoma exile</i>	30	0.42	3.59	2.22
<i>Etheostoma nigrum</i>	3502	0.29	4.06	2.58
<i>Perca flavescens</i>	2102	1.28	2.42	0.53
<i>Percina caprodes</i>	57	0.29	4.07	1.03
<i>Percina shumardi</i>	46	0.12	5.58	n/a
<i>Sander vitreus</i>	4	2.60	1.89	0.29

5.2.4 Bedrock substrate

PCA and PCoA (Euclidean distance association matrix) were performed on log transformed ($\ln+1$) 2004 mass data to initiate discussion about the role of “productive capacity” associated with bedrock substrate site type. Although both methods generated similar object constellations, PCoA showed the objects (sites) with greater clarity (i.e. separation) and thus it was chosen as the selected method of ordination.

5.3 RESULTS

5.3.1 Biomass

Total biomass_{fish} at armoured sites (mean 977.3 g) was significantly different from unarmoured sites (mean 413.7 g) ($P = 0.002$), indicating that in absolute terms, armoured sites were associated with more fish biomass than were unarmoured sites. However, site-type distinction was less clear after the adjustment for site water volume was taken into account. Since all of the unarmoured sites were shallower (Appendix B, Table B.1) than the mean slope (9.1°) and therefore had less volume than the 12 site average, adjusted site biomass_{fish} (and therefore the extrapolated production values) at these sites increased. In contrast only one armoured site did not contain more (i.e. slope equal to mean) volume than the 12 site average (Table B.1) and fish production values per unit volume for the remaining armoured sites were adjusted downward. Although the mean adjusted production levels were still higher at armoured sites than at unarmoured sites (armoured - 807.4 g; unarmoured 761.7 g;) the differences were not significant ($\alpha = .05$; $\beta = .05$). On the other hand, low test power suggests that a false negative is possible.

PCA on the full species data set ($n = 26$ species) produced an object constellation which separated sites by site type (Fig 5.2) and is consistent with the analysis of species composition information (Fig 4.2). Indeed, PCA axis 1 species composition scores were highly correlated with PCA axis 1 $\text{biomass}_{\text{fish}}$ scores (Pearson Product Moment correlation $r^2 = 0.972$). The removal of non-indigenous species (Fig 5.3) did not affect the object constellation much from the full data set (Fig 5.2) except that sites 5 and 8 were bound more closely together in Fig 5.3 than in Fig 5.2. Nevertheless, this analysis also separated sites by site type.

Fig 5.2 PCA scatterplot for 2002-3 transformed $(\ln+1)$ $\text{biomass}_{\text{fish}}$ data. Percent explained variation listed on diagram axes. Unarmoured sites designated by ■, armoured sites by ●.

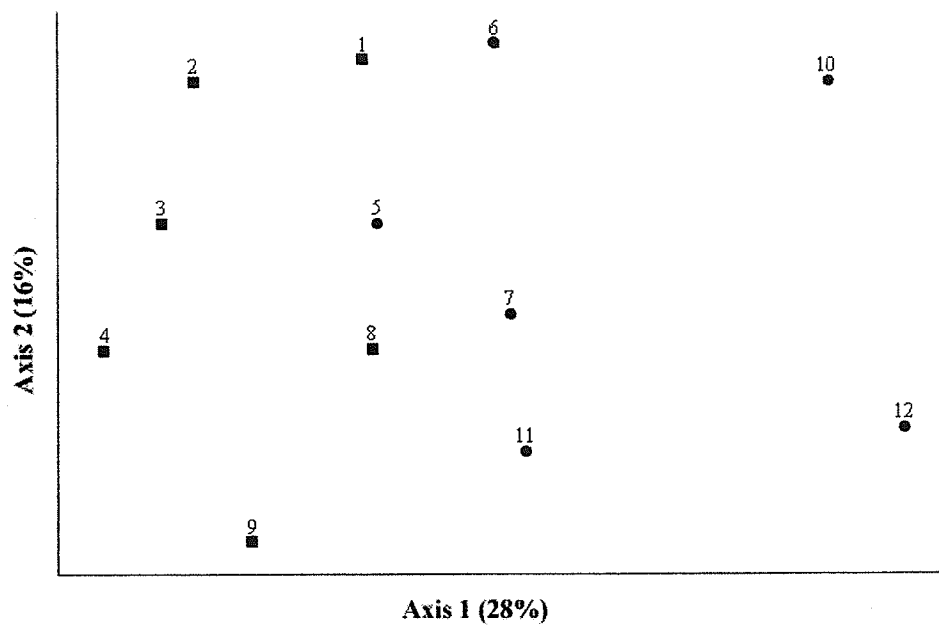
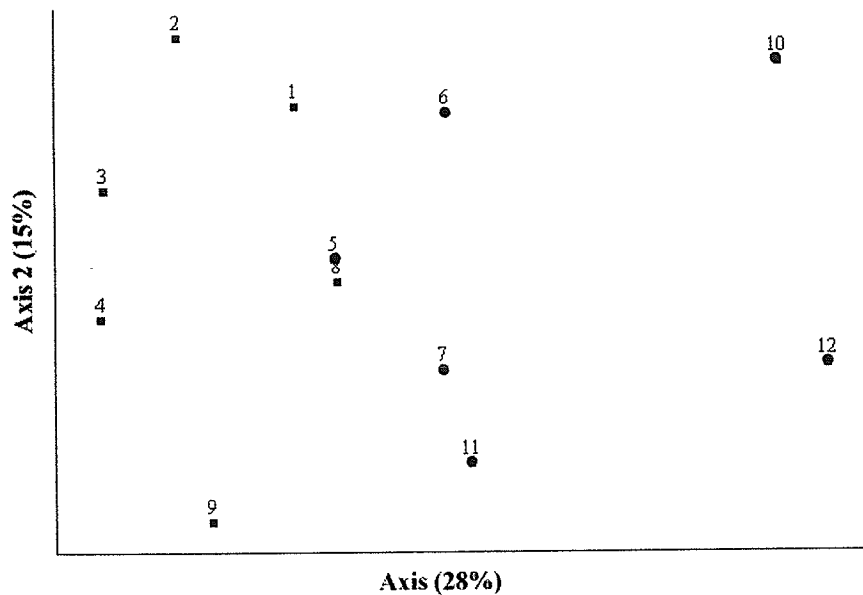


Fig 5.3 PCA scatterplot for 2002-3 transformed (ln+1) fish mass data - indigenous species only data set (n = 25). Percent explained variation listed on diagram axes. Unarmoured sites designated by ■, armoured sites by ●.



PCA on the volume-adjusted full biomass_{fish} data set (Fig 5.4) showed some similarities to the object constellation for the unadjusted data. In particular, the deepest-water armoured sites (sites 10-12) retained their position along PCA 1 with the highest scores. In contrast, the deepest-water unarmoured site (site 1) was re-positioned with a lower component score along PCA 1. The greatest difference between the volume-adjusted and unadjusted ordinations was the realignment of the very shallow, unarmoured sites 8 and 9 (although the shallow unarmoured sites 2 - 4 moved comparatively little) on PCA 1. This realignment of sites 8 and 9 was even more pronounced when site singletons were removed from the volume-adjusted data set (Fig 5.5).

Fig 5.4. PCA scatterplot for 2002-3 transformed (ln +1) volume-adjusted fish biomass, full species matrix (n=26). Percent explained variation listed on diagram axes. Unarmoured sites designated by ■, armoured sites by ●.

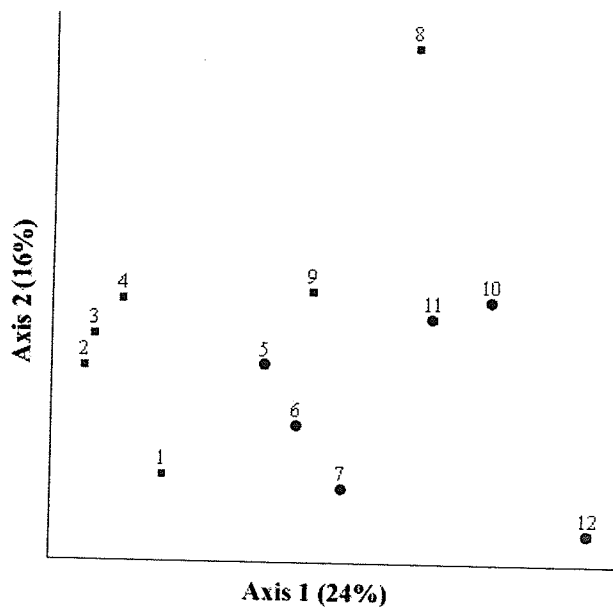
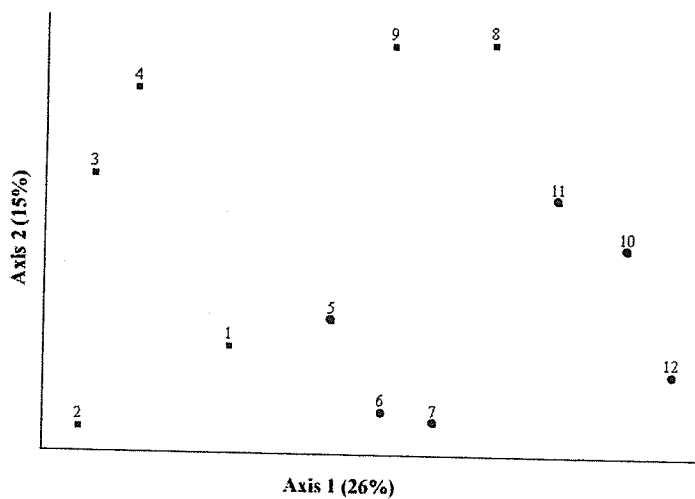


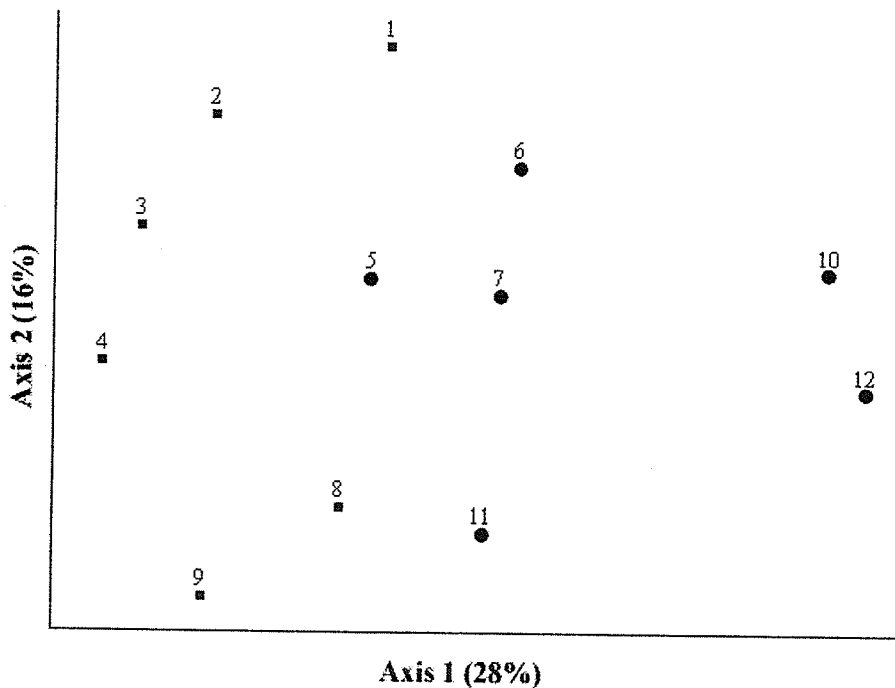
Fig 5.5 PCA scatterplot for 2002-3 transformed (ln +1) volume-adjusted fish biomass, reduced species matrix (n=23). Percent explained variation listed on diagram axes. Unarmoured sites designated by ■, armoured sites by ●.



5.3.2 Habitat Productivity Index (HPI) and HPI**

P/B ratios for Eastern Canada (Randall and Minns 2000) were significantly different ($P = <0.001$) from those calculated for the littoral assemblage of the Winnipeg River (Table 5.1). PCA on HPI values showed an object constellation (Fig. 5.6) which clearly separated sites by site type; where armoured sites showed significantly ($\alpha = 0.050$: $\beta = 0.928$; $P = 0.003$) higher PCA 1 scores than unarmoured sites. Armoured sites were also shown to be significantly different ($\alpha = 0.050$: $\beta = 0.302$) from unarmoured sites using HPI total site scores. While this result may be a false positive as can be interpreted from the

Fig 5.6. 2002-3 PCA scatterplot of HPI scores. Percent explained variation listed on diagram axes. Unarmoured sites designated by ■, armoured sites by ●.



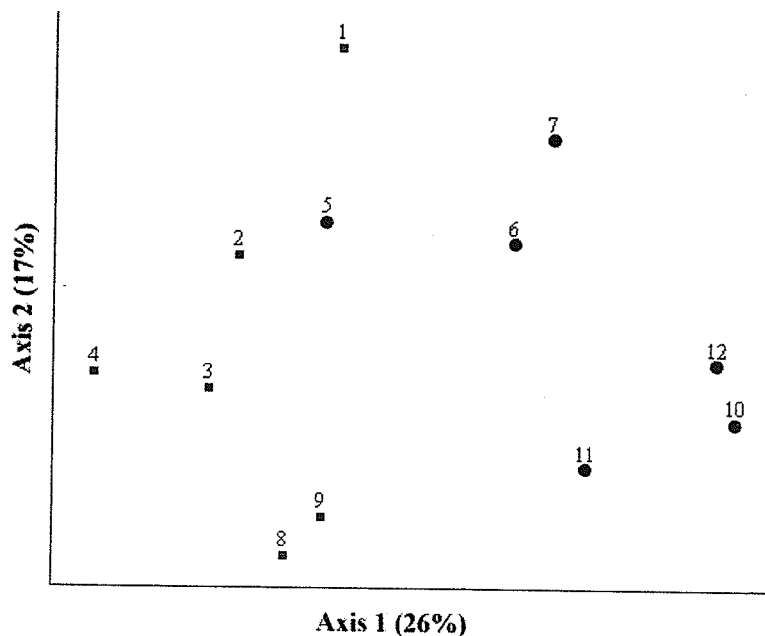
low test power, it is also possible that the result is true, and simply the limited sample size ($n = 12$) produced low test power. Strong positive correlations on PCA 1 between HPI and biomass ordinations were found with the exception of the volume-adjusted analysis (Table 5.2)

Table 5.2. Correlation between HPI and other production surrogates. All analysis (Pearson Product Moment Correlations) on PCA 1 scores. Applicable figure references included in heading.

	Total fish biomass (Fig 5.2)	Native species only (Fig 5.3)	Adjusted volume (Fig 5.4)	Adjusted volume no site singletons (Fig 5.5)
r^2	0.718	0.741	0.574	0.595
p value	0.009	0.006	0.051	0.041
result	strong positive correlation	strong positive correlation	no correlation	positive correlation

HPI** differentiated sites by site type (Fig 5.7) even more clearly than did either HPI (Fig 5.6) or biomass (Fig 5.2). As with other surrogates, the HPI** ordination also indicates armoured sites are better fish habitat than unarmoured sites, based on higher PCA 1 (and to a lesser extent PCA 2; eg. sites 5 -7) scores. Although PCA is not in a strict sense a discriminant method, HPI** situated objects in a constellation which most clearly resembles the physical habitat feature similarities between sites.

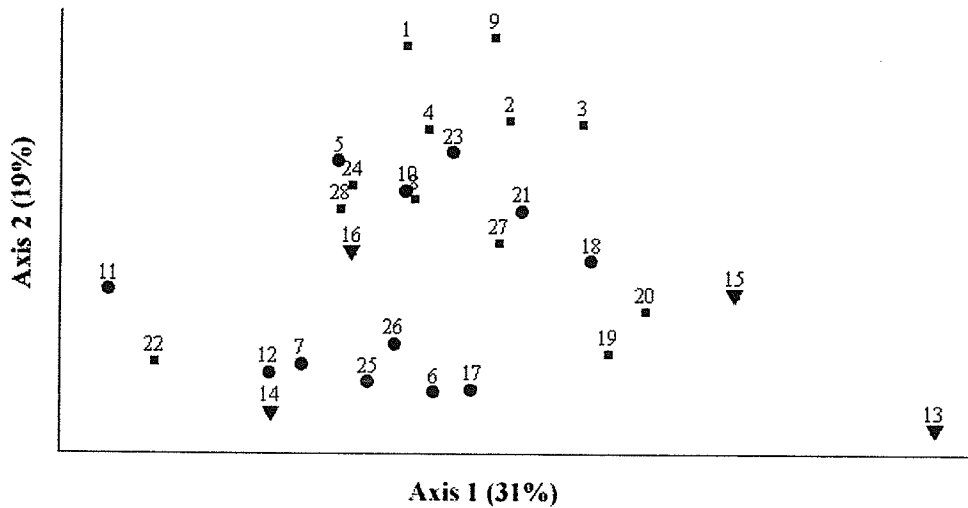
Fig 5.7 PCA scatterplot of 2002-3 (site specific PB based) HPI** data. Percent explained variation listed on diagram axes. Percent explained variation listed on diagram axes. Unarmoured sites designated by ■, armoured sites by ●.



5.3.3 Bedrock substrate

PCoA on 2004 fish biomass data (using all 28 sites) produced an object constellation (Fig 5.8) with somewhat ambiguous results. On the one hand, two of the NERS sites (sites 13 and 15) have the highest PCoA axis one scores indicating that these sites generated the highest biomass data for that year. On the other hand, the other two NERS sites (sites 14 and 16) are positioned generally in the object swarm at the centre of the plot with most other sites and indeed these two sites have lower PCoA 1 & PCoA 2 scores than many other sites.

Fig 5.8 PCoA on 2004 fish mass transformed ($\ln + 1$) data. Percent explained variation listed on diagram axes. Unarmoured sites designated by ■, armoured sites by ●, NERS sites by ▼.



5.4 DISCUSSION

5.4.1 Biomass, HPI and HPI**

In general, fish biomass and HPI surrogates for productive capacity differentiated sites based on site type. Overall, sites with the highest production measures were associated with rip rap armoured, while unarmoured sites were associated with the lowest production values, usually by significant margins. Whether these values actually measure fish production in the sense that a discrete habitat provides all the resources and mechanisms necessary for facilitating complete and important life-cycle stages, or whether they simply measure occurrences in time, where fishes are associated with these

sites, cannot be determined from these data. From the regulatory perspective however, this distinction is moot since it seeks protection of productive capacity of fish habitat which itself is defined as “spawning grounds and nursery, rearing, food supply and migration areas on which fish depend directly or indirectly in order to carry out their life processes” (*The Fisheries Act Sec. 34(1)*). Migration, or at least some form of dispersal, is part of the life stage of fishes – even if only to move from one habitat type (eg. nursery) to another (eg. adult) through a temporary corridor. However, this process makes the corridor vital habitat and therefore, it is irresponsible to exclude the role of short term inhabitation of a location from overall productive capacity / production. I contend that this perspective is also necessary at the systems level as well as the regulatory level and therefore argue that measured fish biomass must be considered a representation of site type production.

That issue aside, the similarities between ordinations using biomass and HPI scores (Figs 5.2 and 5.6) are to be expected since HPI is derived from biomass. The results produced from this data set show a strong positive correlation between biomass and HPI on both PCA 1 ($r^2 = 0.995$) and PCA 2 ($r^2 = 0.908$). This finding is consistent with Randall and Minns (2000) who also found a high correlation between these two surrogates and it suggests that either surrogate is about an equally suitable *a posteriori* method of measuring productive capacity. While the boreal river littoral P/B values were utilised rather than those calculated for Eastern Canada, further investigation of these values should be undertaken especially among large-bodied fishes, of which typically juveniles

were caught; an issue which tends to skew P/B ratios. Randall and Minns (2000) stated that HPI was a more sensitive surrogate of productive capacity than was biomass. Indeed, HPI** appears to be more sensitive than either biomass or HPI . P/B ratios are a comparable measure of efficiency among species. As such, HPI appears to be restricted to only a measure of comparison among species. However, site specific P/B ratios used to generate HPI** are measures of differences between species *and sites* that can indicate the ecological function (i.e. spawning, nursery, rearing, juvenile) and quality of sites, thus providing a better description of site productive capacity than does biomass or HPI. For instance, the calculated P/B ratio of *P. shumardi* (river darter) from pooled data was 5.58; a value which can be differentiated from other darters such as *Etheostoma exile* (Iowa darter) (pooled P/B = 3.59) but which says little about site. Therefore, application of this P/B for HPI masks important information. In contrast, the strong positive correlation ($r^2 = .967$) of site specific P/B ratios (river darter) to PCA 1 scores reveal some of this hidden information (Table 5.3). Specifically, it appears that sites with

Table 5.3 Site specific P/B ratios for *Percina shumardi* (river darter); 2002-3 data.

Site	1	2	5	6	7	10
P/B ratio	4.64	4.64	4.91	5.56	6.3	7.53
PCA 1 object score	-0.872	-2.163	-1.056	1.301	1.777	4.083

lower turbidity (sec. 6.2.2.1) convert available solar energy more effectively than turbid or semi-turbid sites, resulting in richer productive capacity and ultimately species

abundance (chapter 4). Although site 5 is armoured, it is physically proximate (downstream) to site 4 (unarmoured) and general observation shows episodic turbidity plumes originating near site 4 influencing water clarity at site 5. These observations may account for site 5 generating the lowest river darter P/B ratio for armoured sites.

Nevertheless, neither biomass nor HPI / HPI** are predictive methods for quantifying the habitat loss associated with proposed development schemes. That work remains to be done for lotic systems.

The broad conclusion that increased fish production is associated with armour stone sites is not fully supported in the literature. For instance, Knight and Cooper (1991) reported that fish biomass was lower at armoured sites than at unarmoured ones. Indeed, there is a theoretical basis for assuming that unarmoured sites should be associated with higher levels of productivity than armoured sites. Koetsier *et al.*, (1996) noted that streams draining catchments dominated by granite and basalt bedrock had low alkalinity and specific conductance. These measures can act as surrogates for levels of dissolved inorganic carbon (DIC), which is an essential form of elemental carbon for aquatic plants and consequently, there are positive correlations between alkalinity and primary production (Wetzel 2001). (Although carbon is rarely limiting, in systems adjacent to urban landscapes such as this one, the typically limiting nutrients (P & N), may exist in sufficient supply due to anthropogenic sources to make C the limiting nutrient.) There is also a positive correlation between alkalinity and invertebrate production (Koetsier *et al.*, 1996). Since fish rely to varying degrees on primary and invertebrate production, habitats

with higher levels of surrogates for DIC may be associated with increased fish production (Koetsier *et al.*, 1996). By this reasoning, it may be anticipated that sites which are not armoured with granite rip rap may have higher alkalinity (measured by specific conductance) than armoured sites, resulting in greater fish production. Koetsier *et al.*, (1996) concluded that “landscape scale variations in geology may strongly influence stream and riparian productive capacity among catchments, even within a single river basin, and thereby directly affect structural and functional properties of these riverine ecosystems.” It is not clear whether these variations in geology at a small scale would be expected to have the same effects on fish production. On the other hand, specific conductance was measured during a preliminary investigation into study area water quality parameters in 2001 and although there was no significant²⁷ difference between armoured and unarmoured site type, unarmoured sites still had higher conductivity values than armoured sites and therefore might still be expected to yield higher fish production as a result.

On the other hand, the association between higher quality habitat and armoured sites is consistent with findings from Jennings *et al.*, (1996) and Shields *et al.*, (2000) who found that fish biomass increased at stone spur sites. Similarly Fischer and Eckmann (1997a) found maximal biomass values over cobble substrate; a habitat configuration similar to

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The 2001 preliminary field season provided only a cursory examination of site type parameters. A small sample size translated into low test power and therefore there should be some caution about the possibility of a false negative ($\alpha = 0.050$; β value ranged 0.050 - 0.231, depending on the *t*-test).

rip rap armouring. As with the counter argument, there are also ecological reasons to suggest fish production should be higher at armoured sites than at unarmoured ones. Overall, turbidity levels at armoured sites are lower than at unarmoured sites (sec. 6.2.2.1). Apart from the reduced macrophyte development associated with increased turbidity (Jones *et al.*, 1996), there are various responses from organisms to excessive levels of turbidity which may affect site production. In chapter one I outlined some of the physiological impacts to fishes and their prey as the result of high levels of inorganic turbidity. Typically, the first reaction is avoidance behaviour (Wootton 1992; Anderson *et al.*, 1996) resulting in fewer fishes. However, this behaviour is not exhibited by all species (eg. *Sander vitreus*; *S. canadensis*; *Hiodon tergisus*), and therefore avoidance behaviour does not necessarily account for reduced production. Nevertheless, all resident species are susceptible to physiological changes which are considered to be the next level of reaction to abnormal levels of abioseston (Anderson *et al.*, 1996). These changes may become manifest as physical damage, or by impaired growth, usually the result of a chronic exposure to low doses (Anderson *et al.*, 1996). Impaired growth can occur due to the increased energy expended in the search for food²⁸ (for example, see Jones *et al.*,

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Gerking (1994) explained optimal foraging theory (OFT) as an inherent selection capacity used by fishes to minimise energy output (on search; pursuit; capture; ingestion and digestion) while maximising energy input. For instance, a predator will select the larger of two different sized (equally palatable) prey, equal distances away. This process will only break down when prey size is different and so too is distance from the predator. In this instance, the predator will take the prey that appears to be the largest, a process known as Apparent Size Modelling (ASM). When unnatural conditions (chronic conditions which have not been in place for sufficient time to allow behavioural or physiological adaption to occur) persist which alter OFT and ASM processes, we might expect altered physiological results. For instance, in waters that are unusually turbid due

2003) or as a result of the ingestion and processing of lower quality food (for example, see Kirk 1992; Shortreed and Morton 2000). All these factors acting in concert or individually might account for reduced fish production.

5.4.2 Natural productive capacity

The Pinawa study area is not highly influenced by non-native species (cf Figs. 5.2 and 5.3). As a result, the object constellation which shows productive capacity by site and by site type (Fig 5.3) remained essentially unchanged from Fig 5.2 (but see sites 5 and 8). However, the removal of a less ubiquitous non-native species (smallmouth bass occurred at every site but one) or in habitats with highly altered species composition, would likely render natural productive capacity substantially different than total productive capacity (Fig 5.2). Therefore, habitats with many non-native species should be analysed for their impact on productive capacity, using the simple comparison utilised here. Furthermore, since smallmouth bass are lithophilic (Scott and Crossman 1998; Stewart and Watkinson 2004), I suggest the Pinawa study area be monitored to determine any change this species may have on productive capacity over the long term as a result of rip rap placement.

5.4.3 Adjustments for volume

The hypothesis that littoral zone productive capacity should be measured by a surrogate based on site volume alone seems to be without foundation. Keast and Harker (1977)

to sediment loading, fishes accustomed to higher clarity may be unable to feed as successfully as would members of those species which reside in a more turbid state.

reported a relationship²⁹ between fish biomass and water depth which is difficult to reconcile with my volume-based results since the deepest site always showed the greatest biomass, volume adjustment notwithstanding. On the other hand, the influence of hydrological regimes associated with varying water depths affects fishes and their habitat (Bond 1996). Apart from roughly calculated surface vectors to represent site specific hydrology during 2002 (data not reported), there has been no detailed study of the hydrology of the water column within this study. Nevertheless, the ordinations of volume-adjusted data (Figs 5.4 and 5.5) did not present results consistent with the analyses of species composition (Fig 4.2), unadjusted biomass (Fig 5.2) or with HPI scores (Figs 5.6 and Fig 5.7). Additionally, PCA 1 scores for volume-adjusted biomass (full species data set) showed no correlation to site water depth. Furthermore, both PCA ordinations of volume-adjusted biomass (Figs 5.4 and Fig 5.5) positioned sites 8 and 9 (low total biomass) proximate to sites 10-12 (high total biomass), while leaving essentially unchanged the positions of sites 2-7. The reduction in differences between sites (especially evident in the ordination of the reduced species data set) is a common occurrence when sites are compared by volume (Mann 1975), although as such, we might expect all sites to be grouped more closely than either Figures 5.4 or 5.5 indicate. This suggests that some other phenomenon is present. Operationally, the shift of sites 8 and 9 towards sites 10-12 (Figs 5.4 - 5.5) suggests that the development of *very* shallow sites such as these would create good habitat, while the creation of shallow sites (sites 2- 4)

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$\log B = a + b Z$ where $B =$ biomass and $Z =$ depth.

would not. Indeed, such a suggestion is found within the literature. On the one hand, armour stone is associated with lost habitat because of its in-fill properties creating an area that is lost to fishes³⁰ (Minns *et al.*, 1995). On the other hand these same authors acknowledge “the potential gain of fish habitat with low slopes of rock could be substantial in water bodies that experience a wide annual range in water levels”. However, we know that some species are depth specific. For instance, Casselman and Lewis (1996) reported a specific fish length - habitat depth ratio for *E. lucius*. Furthermore, smaller fishes, even within the same species, tend to convert energy better than large fishes (Mann 1975), resulting in higher P/B ratios for smaller fish. Consequently, analysis of volume-adjusted biomass tends to emphasise the situation where shallow water species and age-classes tend to be small and therefore generate a relatively high P/B ratio. However, as some of these fishes mature and require greater depth, the P/B ratio per site and species declines. For habitat managers this means the protection / development of maximum productive capacity habitat at one point in time may institute habitat constraints for those species which require habitat of greater depth and seemingly lower productive capacity – at least as measured by lower P/B ratios – at a subsequent point in time. Since the growth of young fish contributes to population levels, the productive capacity of the ecosystem is influenced by habitat constraints (Jones *et al.*, 2003) Complicating matters further, recent work on allometry shows that growth spurts among fishes are seasonal (Bacon *et al.*, 2005) and therefore the measurement of

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Especially among the grey literature - examples include DFO (1999); Environment Canada (1999).

productive capacity through P/B ratios may be a function of time of year and water temperature variables. Overall, this problem suggests that production over depth must be considered into assessments of habitat loss/gain. Consequently, this result suggests that a quantified measurement of productivity constrained by depth, not averaged in as part of the efficiency ratio must be considered. This work is continued in chapter six.

5.4.4 Bedrock substrate

Recent evaluations have generally considered rock substrate to be inferior habitat (Frezza and Minns 2002a), especially when offered as new or compensatory habitat (Jones *et al.*, 2003). The grouping of most sites into a collage object constellation (Fig 5.8) makes it difficult to form any opinion about site type from the 2004 Pinawa study area fish mass data. Within this arrangement, four sites are set apart from the remaining 24 which form a tight cluster at the centre of the scatterplot. Sites 11 (armoured) and 22 (unarmoured) are positioned nearest the origin indicating low PCoA axis 1 scores; the opposite of sites 13 and 15 (exposed bedrock substrate). From these last two sites alone, there is some evidence that rock substrate provides good habitat – even if only during the cool water temperature conditions of 2004. However, sites 14 and 16 are also bedrock substrate and they are positioned closer to the origin and have lower PCoA scores than many of the other 26 sites in the plot. These equivocal results are not entirely surprising. Indeed, the habitat quality of rock substrates have received mixed commentary by other authors (see for instance Pratt and Smokorowski 2003). Further work on exposed rock substrate is warranted.

5.5 CONCLUSIONS

Biomass and HPI surrogate measures of productive capacity were highly correlated as was expected. Nevertheless, both measures generated results about site type that were consistent with results generated using species composition data. Species specific P/B ratios used to produce HPI** provided additional insight into site type difference, habitat quality and productive capacity. Agreement about habitat quality across composition and production measures lends credibility to: 1) the validity of each of the individual measures used to assess habitat, and; 2) to the general conclusion that rip rap armouring is associated with better quality fish habitat than unarmoured sites within the Pinawa study area. The role of non-native species can be assessed using multivariate methods. There is a recommendation for continued examination of the study area to assess the long term effects of rip rap on the relative contribution of non-native species to site productive capacity. Simple adjustments to biomass surrogates based on site water volume appear to confuse rather than add clarity to the understanding of site productive capacity. On the other hand, site water depth appears to be an important habitat feature. Further work involving examination of specific details of water column dynamics is suggested. A preliminary examination of exposed bedrock substrate suggests that the quality of this habitat may be difficult to assess due large variability in the performance of this site type.

CHAPTER 6

MEASURING FISH HABITAT BY DIRECT GRADIENT ANALYSIS - LINKING ENVIRONMENTAL VARIABLES AND FISH ASSEMBLAGES

6.1 INTRODUCTION

In order to refine environmental management priorities, human induced changes to the physical environment should be differentiated between those that have ecological significance and those that do not. In order to measure ecological significance, we need to understand, at least conceptually, how alterations are expected to influence ecosystems. From there, we must be able to match ecosystem response to anthropogenic activity by tractable descriptive measures. Physical alterations to fish habitat can lead to any of four community responses, including changes to: 1) fish production 2) species composition; 3) distribution over time / space; and 4) the non-fish biota (Minns *et al.*, 1996). Alterations to fish habitat which affect the chemical, physical or non-fish biological properties of the environment can be integrated into fish responses (Karr 1981) and thus the latter two responses can be embodied within the first two. This study assumes that the manifestation of any of these responses can be considered ecologically relevant.

Indirect gradient analysis (PCA) was used to examine the impact of shoreline armouring on species composition (chapter four) and surrogate measures for production (chapter five). In both cases, the results were unequivocal; rip rap shoreline armouring was

associated with sites with better fish habitat than were unarmoured sites. Although indirect gradient analysis is useful in establishing general trends in ecological data, it falls short of determining causation. Indeed, several strong positive correlations were found between armoured sites and indicators of higher quality fish habitat (esp chapters four and five). However, in no sense could the work of those chapters claim to provide a predictive model (such as regression, where for every x there is a predictable y response) for shoreline armouring. With particular respect to this study, it would be irresponsible to say that rip rap *improves* fish habitat based on the associations revealed in chapters four and five. For instance, from those analyses, it is impossible to know whether armoured sites have other attributes which are associated with better habitat, notwithstanding armouring; or indeed if armoured sites were *much* better *a priori* and were made *less* effective as a result of armouring, yet still happen to remain absolutely better than unarmoured sites, thereby confounding our interpretation.

Since the results from chapters four and five explain the structural composition (i.e. fish community) of each site (and, as it turns out, each site type), they provide some implicit understanding about species interactions, at least among fishes, within these sites.

However, since ecology and environmental science both are concerned with the interactions between biological communities and their environment – especially a human altered environment – this chapter investigates the relationship(s) between the structural composition revealed in those chapters and the specifics of the local environment.

6.1.1 Analytical approach

If the effect of only a single environmental variable on a community is sought, analysis of an array (environment data) and a matrix (species composition data) can be handled using multiple linear regression. However, this method of direct comparison allows only one environmental variable to be measured against a matrix of species variables (or vice versa) at any one time (Manly 1994). Therefore, the decision as to which single variable to use, while crucial, ultimately becomes arbitrary. Furthermore, it assumes that the behaviour of that variable does not act in concert with other environmental variables. Therefore, a poor decision over choice of variable can lead to results that are not necessarily ecologically meaningful. Furthermore, the aggregation of several separate multiple linear regressions, one for each environmental variable, is not a tenable solution to this problem, since this process would miss the effect of interactions among the environmental variables upon the biological data set. Indeed, environmental variables may be orthogonal (i.e. unrelated), collinear (redundant), or at some point between these two opposites. Therefore, using any single variable to represent a suite of variables without evidence that such a process has validity, is meaningless. Instead, it is the direct comparison of two data sets, one including environmental information and the other containing species information, which is the most likely method of producing a clear understanding about species - environment interactions which in turn lead to ecologically meaningful results.

Canonical methods (direct gradient analysis -see chapter 2) are those techniques which compare directly the constituents of a matrix comprised of mixed variables (eg. species and environmental) (Legendre and Legendre 1998). Since these methods sacrifice some of the broad structural view in order to make strong determinations between species assemblages and environment (see ter Braak 1987) it was important to have utilised indirect gradient analysis previously to ensure undue structural information was not overlooked prior to the utilisation of direct methods. However, assuming that the basic structure of site type has been established in chapters four and five, it is now important to evaluate site type with direct gradient methods. Nevertheless, the causal relationship between rip rap armouring and fish habitat will remain a point of some interpretation.

6.1.1.1 Canonical techniques

Several canonical forms of analyses exist and while it is not the purpose of this study to assess these methods critically, a brief comment on the appropriate application is in order. Using our understanding about the differences between regression (a mathematical model that predicts the response of the dependent variable on the independent variable) and correlation (a descriptive model measuring the relationship of co-occurrence of two random (i.e. either dependent or independent) variables), so too can we understand the differences between the various canonical methods. The development of a matrix containing both environmental and species data sets can be analysed in one of two manners. While Canonical Correlation Analysis (CCorA) maximises the linear correlation between the two sets of variables (i.e the multivariate extension of

correlation, Legendre and Legendre 1998), it does not necessarily produce a result which *explains* the variables on one side of the matrix by the variables on the other side. In contrast, when one data set within a matrix is to be explained by another data set (i.e. the multivariate extension of regression) with a matrix (such as is created when environment and species variables are melded into a single matrix for purposes of using the former to explain the latter) either Redundancy Analysis (RDA) or Canonical Correspondence Analysis (CCA) are best suited. Of these two, CCA has been described as the most appropriate canonical method for analysing data from a hypothetical model which examined the relationship between fish presence, measured along a gradient perpendicular to shore and substrate type (Legendre and Legendre 1998). CCA has also been used in applied research, where littoral zone fish assemblages were assessed by environmental variables (Weaver *et al.*, 1997). Thus, Canonical Correspondence Analysis (CCA) was selected as the canonical method for assessing data in this chapter. In preparation for CCA analysis, environmental variables were parsed through an initial assessment, followed by a preliminary analysis prior to their use in the final analysis. A description of these analytical methods along with the field methods used for measuring these variables are provided below (sec. 6.2)

6.2 METHODS

6.2.1 ENVIRONMENTAL VARIABLES - OVERVIEW AND PRELIMINARY ANALYSIS

During the course of the project (2001-2004) an initial suite of 12 environmental

variables were variously recorded (Table 6.1) some of which are described here briefly.

Detailed field methods for preliminary and final-analysis variables are provided below

(sec. 6.2.2).

Table 6.1 Summary table of measured environmental variables by field season. Table includes variable status ranking (i.e tier). For explanation of ranking, see text.

	Environmental variable	2001	2002	2003	2004
First tier variables	Turbidity	✓	✓	✓	--
	Macrophyte cover	--	✓	✓	--
	Shoreline profile	--	✓	✓	✓
	Armouring status / age	✓	✓	✓	✓
Second tier variables	Dissolved oxygen	✓	--	✓	--
	Water temperature	✓	✓	✓	✓
	Substrate heterogeneity	--	✓	✓	--
Third tier variables	River water level	--	✓	✓	✓
	pH	✓	--	✓	--
	Boat traffic	--	✓	✓	--
	Conductivity	✓	--	--	--
	Wind speed and direction	--	✓	✓	--

In 2001, study area water quality parameters were measured, including water temperature, turbidity, dissolved oxygen (DO), pH and conductivity (these last three variables with Multiline F/Set multimeter - WTW Wissenschaftlich Co.). Of these five

variables, pH and conductivity were not significantly different between site types to pursue any further measurement. In contrast, differences in site type turbidity values and water temperature suggested that further investigation was warranted and measures of these variables were continued in both 2002 and 2003. Equipment and sampling problems during 2001 resulted in the collection of limited DO information. While these limited results suggested there may be site type differences, further investigation of this variable was held off the 2003 field season when equipment difficulties could be resolved. Starting in 2002, site specific profiles (i.e. river bank and bottom contour) were measured and site specific macrophyte surveys were initiated. Measurements of both these variables were repeated in 2003. In 2004, site profiles were continued for sites added prior to the 2004 field season (sites 13-28). However, the onset of macrophyte development in 2004 was so delayed so as to be non-evident at most sites and therefore, no vegetation survey was conducted that year. Although equipment malfunctions prevented meaningful turbidity results during 2004, water temperature data were recorded that year.

6.2.2 FIELD METHODS

6.2.2.1 Turbidity

I hypothesised that armoured and unarmoured sites would display significantly different levels of suspended solids (measured as turbidity) in the water column, particularly adjacent to the river bank, where bank erosion and the resuspension of shallow water site substrate would show the greatest effect. In 2002, initial observations at unarmoured sites

indicated that turbidity clouds could extend well out into the river. At the downstream side of each of these sites, tall 3.85² cm wooden stakes were driven into the river bottom to indicate 2, 4 and 6 metre distances from shore as a means of estimating the areal extent of turbidity plumes. In contrast, armoured sites rarely showed a distinguishable turbidity cloud, and were denoted simply with shoreline axis site markers. At armoured sites, water samples were consistently collected from the water / shoreline interface at the midpoint of the site. In contrast, water samples were collected from unarmoured sites, according to observable signs of turbidity within the site. In all cases, where turbidity clouds were detected, the extent and pattern of distinguishable clouds were sketched as field notes.

In 2003, a revised sampling protocol was initiated. The downstream side of each site was marked with tall 3.85²cm stakes at the 2.5m and 5m positions out into the river. Water samples were collected along this gradient at 0m (i.e. water / shoreline interface), 2.5m and 5m for each site. Additionally, whereas in 2002, turbidity readings were collected at random, both in terms of time of day and day of week, 2003 turbidity readings followed a prescribed schedule. 2003 turbidity readings were conducted on Tuesday, Thursday and approximately every second Saturday (non fish capture days) three times per day (early morning, mid afternoon and early evening). Additionally, turbidity readings were taken on Monday Wednesday and Friday (fish capture days), prior to netting. In all cases, water samples were collected using a graduated telescoping pole to which was affixed a 284ml metal can. Samples were lifted from the water column and a sub-sample was

immediately pipetted into a cuvette. Turbidity values were determined in the field using a Palintest Micro 900 nephelometric turbidimeter.

During 2002, turbidity values were collected on an *ad hoc* basis and generally these values represented the visually assessed maximal values at the time of sampling. Despite this apparent bias, mean sites values for 2002 were consistent with 2003 results (Table 6.2). Nevertheless, to remove any bias associated with 2002 data, only 2003 turbidity values were utilised in the final analyses.

Although 2003 turbidity values were recorded on six days of the week, turbidity analysis was restricted to the sub-sample of data collected from three-per-day samples, resulting in 2484 individual turbidity readings (23 sample days * 3 samples per day per site (n = 12) at each of the 0, 2.5m and 5m positions). Site and site-type averages for each of the three sampling positions (0m, 2.5m and 5m) were calculated (Table 6.2) to produce a site type turbidity model (Fig 6.1). Decay curves were produced for each site type (Fig 6.1). From this model, turbidity values per given distance from shore were calculated for use in the preliminary analysis. Turbidity values at armoured sites were best represented by the exponential decay curve:

$$y = 10.062e^{-.2348x}; r^2 = .9997 \quad [\text{equ 3.1}]$$

Similarly, turbidity values at unarmoured sites were best represented by the exponential

expression:

$$y = 42.553e^{-.3589x}; r^2 = .9969; \quad [\text{equ 3.2}]$$

where y = the derived turbidity value and x = distance from shore (m).

Mean turbidity values at the near shore (0m position) were statistically different (t -test) between armoured and unarmoured sites ($p = 0.002$; $\alpha = 0.05$; $\beta = 0.952$). More

Table 6.2 Summary of site specific and site type mean turbidity measurements 2002 & 2003 field seasons. All values expressed in NTUs.

	2003			2002
	0.0m	2.5m	5m	--
Site 1	46.1	7.0	4.0	33.0
Site 2	27.3	13.4	4.8	25.6
Site 3	60.8	20.5	6.9	30.5
Site 4	53.1	15.8	5.2	32.6
Site 8	61.9	31.7	14.9	122.5
Site 9	13.7	9.8	7.8	41.2
Total unarmoured sites	43.8	16.4	7.3	47.6
Site 5	5.0	4.8	3.2	11.6
Site 6	4.6	3.4	2.7	10.3
Site 7	9.0	4.1	2.9	8.8
Site 10	6.7	2.5	1.7	9.9
Site 11	19.1	6.3	3.6	14.4
Site 12	16.4	12.0	4.6	13.3
Total armoured sites	10.1	5.5	3.1	11.4

importantly, however, is the ecological significance represented by the difference in observed values. By solving for x (equation [3.3]) it is apparent that across all unarmoured sample sites, the first 1.47m of the water column (Fig 6.1) is prone to turbidity levels in excess of the benchmark for impaired water of 25 NTUs (Anderson *et al.*, 1996).³¹

$$y = ae^{bx} \quad [\text{equ 3.3.1}]$$

$$y/a = e^{bx} \quad [\text{equ 3.3.2}]$$

$$\ln(y/a) = bx \quad [\text{equ 3.3.3}]$$

$$\ln(y/a)/b = x \quad [\text{equ 3.3.4}]$$

In contrast, all armoured sites showed mean turbidity values < 25 NTUs. (Fig 6.1; Table 6.3).

Similarly, decay curves were produced for each sample site. From each curve, a turbidity value for the 1m from shore position was derived to be used as site specific turbidity values in the preliminary and final statistical analyses (Table 6.3). I did not use the higher turbidity values from the 0m mark, given that at most sample sites, the water depth at the 0m mark was too shallow to provide meaningful habitat. During statistical analysis, negative turbidity values were used to ensure similarity of environmental variable directionality.

³¹

Anderson *et al.*, 1996 cite the MB DOE maximum acceptable level of 25 mg/L (~ 25 NTUs).

Fig 6.1 Pinawa study area site type turbidity model. Unarmoured sites represented by \square ; armoured sites represented by \bullet . Data based on pooled 2003 site turbidity readings taken at 0m; 2.5m; 5m from shore. Area above horizontal bar (at 25 NTU mark) represents fish habitat lost to excessive turbidity.

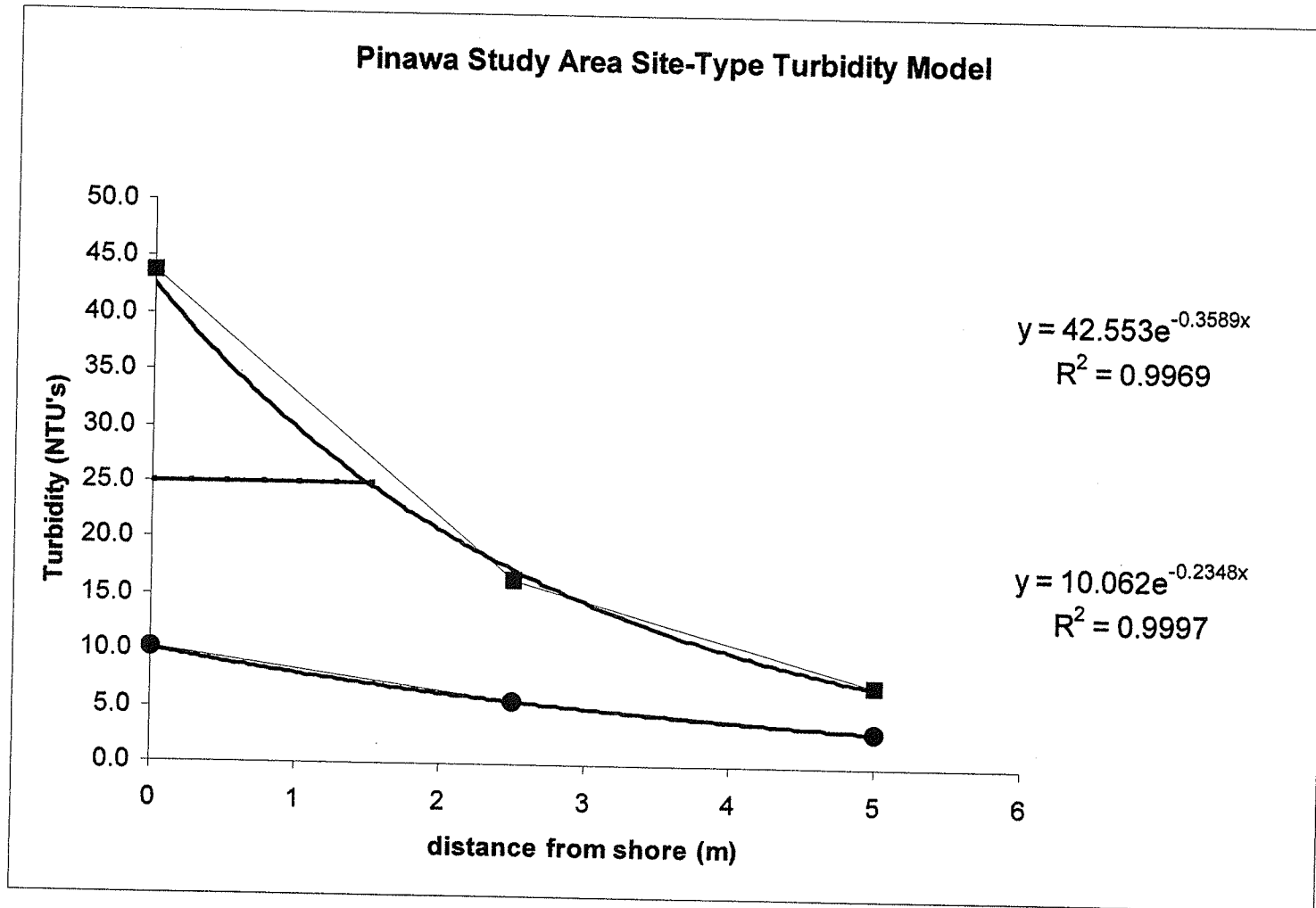


Table 6.3 Derived turbidity values from 12 Pinawa sites based on 2003 data. U = unarmoured site; A = armoured sites.

Site	Site type	Site turbidity value (NTUs)	Distance from shore (m)
		(1m from shore)	where turbidity \geq 25 NTUs
1	U	22.5	0.78
2	U	20.3	0.14
3	U	39.4	2.04
4	U	32.8	1.58
5	A	4.9	–
6	A	4.1	–
7	A	6.7	–
8	U	47.3	3.23
9	U	12	--
10	A	4.6	–
11	A	12.5	–
12	A	14.2	–

6.2.2.2 Macrophyte Cover

In 2002, a vegetation survey was conducted on each site. The survey was limited to an examination of percent cover. In 2003, each site was mapped again, between July 28 - August 1; a period estimated to be of maximum macrophyte development. Any of 14 species of macrophytes were identified using Newmaster *et al.*, (1997) to determine site specific species richness (Table 6.4) and a stem counting methodology (Minns *et al.*, 1993) was used to determine species abundance within each site. Percent cover was also estimated during the same survey. To conduct the survey, a 5 x 3m, 15 cell, floating rope

grid was made (each cell = 1 m²). The grid was attached temporarily to site stakes to prevent it from floating away during the survey. The survey was conducted by a 4 person crew. In turn, each site was flanked by three of the crew who conferred and jointly reported cell macrophyte composition (richness, abundance and percent cover) to a recorder who remained on shore. Macrophyte dominance was hypothesised to be an important factor in site differences and indeed, percent macrophyte cover (Table 6.5) was different between site types ($P = <0.001$; $\alpha=0.050$ $\beta = 0.996$). Accordingly, the macrophyte variable was utilised in the preliminary analyses in three different forms:

Table 6.4. List of aquatic plants found in the littoral zone of the Pinawa study area, July 28- Aug 1, 2003. Plants keyed using Newmaster, *et al.*, (1997).

Abbrev	Common Name	Scientific Name
CBJ	Canada Bluejoint	<i>Calamagrostis canadensis</i>
T	Common Cattail	<i>Typha latifolia</i> ; <i>Typha angustifolia</i> ; or <i>Typha x glauca</i>
QW	Spiny-spored Quillwort	<i>Isoetes echinospora</i>
SA	Stiff Arrowhead	<i>Sagittaria rigida</i>
FA	Floating Arrowhead	<i>Sagittaria cuneata</i>
RP	Richardson's Pondweed	<i>Potamogeton richardsonii</i>
FP	Flat-stemmed Pondweed	<i>Potamogeton zosteriformis</i>
WSW	Submerged Water Starwort	<i>Callitriche hermaphroditica</i>
WM	Northern Water Milfoil	<i>Myriophyllum sibiricum</i>
WC	Wild Celery	<i>Vallisneria americana</i>
CB	Common Bladderwort	<i>Utricularia vulgaris</i>
CF	Curly White Water Crowfoot	<i>Ranunculus longirostris</i>
CW	Common Waterweed	<i>Elodea canadensis</i>

1) percent vegetation cover per site; 2) species richness per site; 3) site patchiness.

Patchiness was measured as the ratio between barren and vegetated cells within the sampling grid (*sensu* Weaver *et al.*, 1997). Maximum patchiness would have a ratio of 0.5 (i.e. for every two cells, one would be vegetated and one barren). All site ratios were divided by 0.5, then multiplied by 100 to generate percent patchiness.

Table 6.5 Summary of 2003 Pinawa site macrophyte survey. U = unarmoured sites; A = armoured sites.

Site	Site type	% total cover	Species richness (# spp)	% patchiness
1	U	16.7	5	32.8
2	U	11.3	5	33.9
3	U	11.7	6	22.6
4	U	15.7	4	34.9
5	A	20.7	3	27.7
6	A	39.9	3	45.1
7	A	23.7	3	28.7
8	U	10.0	4	22.6
9	U	16.7	5	32.8
10	A	29.3	3	29.7
11	A	29.7	6	39.0
12	A	36.3	5	35.9

6.2.2.3 Water Depth and Bank Profile

Sample sites were mapped using a floating graduated telescoping pole against which depth measurements could be made out from shore at 50cm intervals, to the back of each

site. Four repeated measurements were made across the width of each sample site and an average slope was determined (Appendix B, Table B.1). The profile variable was applied to the preliminary analysis set in two forms: 1) maximum depth; 2) substrate / shoreline interface angle. A *t*-test on the data revealed that shoreline angles were significantly different ($P = 0.003$; $\alpha = 0.050$; $\beta = 0.927$) between site types, with armoured sites having a steeper angles than unarmoured sites. Maximum site depth was also significantly different between site types (Mann-Whitney rank-sum test; $P = 0.015$).

6.2.2.4 Armouring: Status and Residence Time

Since all sample sites were either armoured or unarmoured, a simple binary method was used to code armouring status; where one site type was allocated a “1” and the opposite type was allocated a “0”. On the face of it, the allocation of the zero value could have been arbitrary. However, the issue of variable directionality (where higher numbers indicate “more”) suggests that inappropriate allocation of value might pose some concern. Intuitively, more armour implies these sites must be as allocated “1”. On the other hand, unarmoured sites may imply more “pre-secondary disturbance condition” and thus equally entitle unarmoured sites a value of “1”. Therefore, to balance both perspectives, in the preliminary set of analyses, all variable combinations were used, where armoured sites were allocated both “ones” and “zeros”. Although these values necessarily produced significant differences between site types, (Mann-Whitney Rank sum test; $P = 0.002$) it was not clear *a priori* that armouring or its lack has any ecological explanatory power. However, since this matter is indeed the subject of the overall

investigation, armouring status was assumed to have some ecologically significant role and was included in the preliminary analysis.

Allocating residence times for armoured sites was achieved by sorting armoured sites into youngest, middle and oldest categories. I converted these terms to semi-quantitative data by allocating 0.1; 0.5; 1.0 values to each of these respective categories. An alternative to scoring residence time was to use estimated number of years in place (~2 yrs, ~15 yrs and ~30 yrs), but these values were no more accurate than the ratio I used. More importantly, since all unarmoured sites by definition had zero residence time, any non-zero value attributed to armoured sites tended to make this variable collinear with the armouring status variable and therefore would tend to separate site type more than it would separate individual armoured sites from one another. As a result, armour age was excluded from the preliminary analysis. Nevertheless, armour age was included in a version of the final analysis to generate some initial discussion about its potential ecological role.

6.2.2.5 Dissolved Oxygen

Dissolved oxygen values were measured in the field using a MultiLine F/SET -3 multimeter (Clean Earth Scientific WTW Wissenschaftlich w/ DO probe). Surface water samples were collected and DO was measured immediately following the removal of a sub-sample used to assess turbidity (described above). Site results were pooled and averaged by position (distance from shore) from all months, yielding a per position per

site matrix. Checks were made during pooling for outliers which may have explained daily fish capture vagaries, however none were found. Although DO is an important explanatory variable (see Brown 1975) DO levels were homogeneous (Table 6.6) and consequently, the three positions from each site were pooled to produce a single per site value. A *t*-test revealed no significant difference in DO values by site type. Moreover, since the range of values measured at any of these sites did not exclude any species from one site at which it might occur at another, this variable was excluded from the preliminary analysis.

Table 6.6 Summary of mean dissolved oxygen values for all 2003 Pinawa sites. Values in ml/ L.

	0m	2.5m	5m	Site
Site 1	6.7	6.8	6.8	6.8
Site 2	7.0	6.9	6.9	6.9
Site 3	6.8	6.9	6.8	6.8
Site 4	6.8	6.9	6.8	6.8
Site 5	6.9	6.9	6.6	6.8
Site 6	6.9	6.8	6.6	6.8
Site 7	6.8	6.8	6.6	6.7
Site 8	6.4	6.5	6.7	6.7
Site 9	6.4	6.5	6.7	6.5
Site 10	6.5	6.3	6.2	6.3
Site 11	6.8	6.9	6.8	6.8
Site 12	6.9	6.8	6.8	6.8

6.2.2.6 Water Temperature

Surface water samples were collected and water temperature was measured at the same time as dissolved oxygen, using a digital thermometer (traceable thermometer - Control company). Local water temperature is linked to general habitat, (Coker *et al.*, 2001; Jones *et al.*, 2003), physiological functions (*sensu* Fortin *et al.*, 1996; Hayes *et al.*, 1996; Hamel *et al.*, 1997; Bacon *et al.*, 2005) and to the general distribution of fishes (Brown 1975). As a general explanatory variable, water temperature has few parallels and seasonal variation in water temperature (Figs 6.2a-c) can influence fish distribution between years. The most numerous temperature measurements were compiled during 2003 (Figs 6.2a-c) and these data were used to infer site temperature gradients for all years. However, since there was no significant difference (*t*-test) in water temperature between site type during 2003, this variable was excluded from the preliminary analysis.

Fig 6.2a Pinawa study area water temperature, May- Sept 2002. Horizontal line shows 20°C mark.

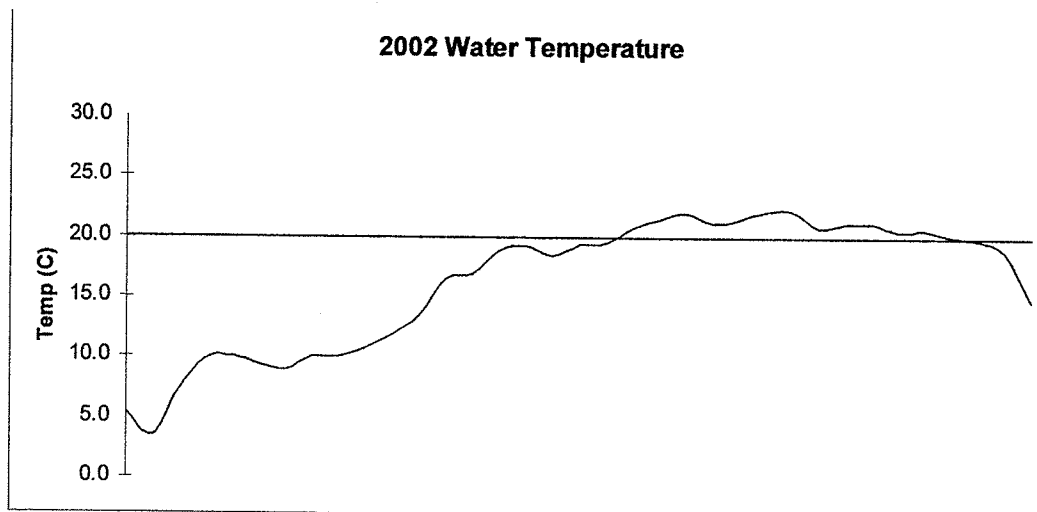


Fig 6.2b Pinawa study area water temperature, May - Aug 2003. Horizontal line shows 20°(C) mark.

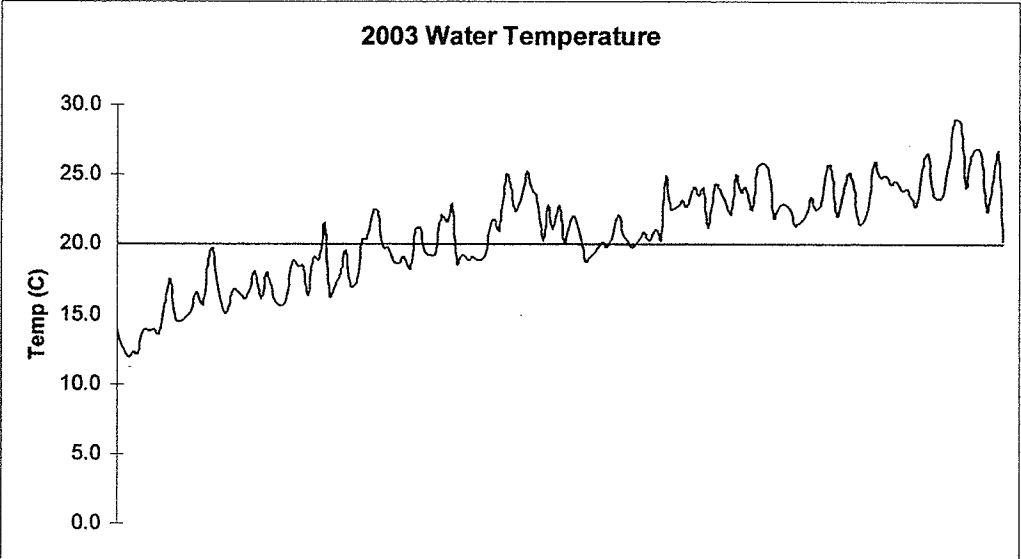
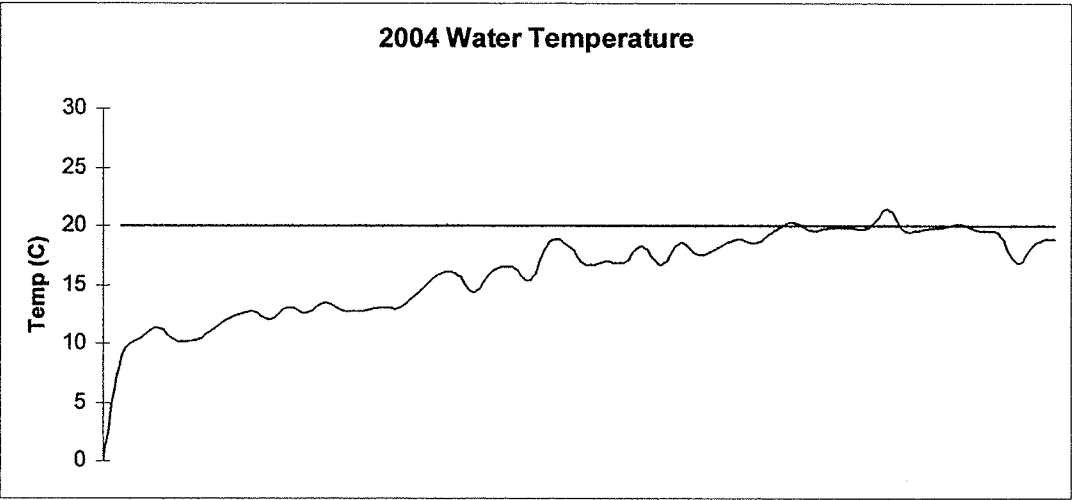


Fig 6.2c Pinawa study area water temperature, May - Aug 2004. Horizontal line shows 20°(C) mark.



6.2.2.7 Substrate Heterogeneity

Thigmotaxis (substrate affinity) forms an important component of fish habitat, and benthic species such as sculpins and darters may be especially attracted to the increased surface area associated with substrate heterogeneity (Bond 1996). In 2003, a single grab sample of substrate was scooped by hand through the water column from the mid point of each site using a 907g plastic sampling vessel. Each sample was spread on drying pans, air dried (168 hours) then sieved (Cummins 1962) through 2mm diameter mesh to separate “large” from “fine” particles. The variable was converted to a “% maximum heterogeneity” by taking the product of the percent dry mass of sample (Cummins 1962) <2mm diameter and percent > 2 mm diameter and multiplied by 25. This procedure assumes that equal amounts of “large” and “small” substrate particulate material (by mass) yields maximum heterogeneity. A *t*-test revealed that particulate was not different between site types and this variable was not included in the preliminary analysis.

6.2.3 ANALYTICAL METHODS

6.2.3.1 Environmental variables - Preliminary Analysis

Since multicollinear variables need to be minimised when canonical methods are used for purposes of inference (Legendre and Legendre 1998), it was necessary to establish a technique for determining environmental variable inclusion for the preliminary analysis. I used a hierarchical approach to sort the full suite of environmental variables into three sub-groups in preparation for the preliminary set of CCA analyses. For this analysis, all combinations of first and second tier variables (in all forms) were utilised to produce a

preliminary CCA analysis set (Table 6.7). Third tier variables were excluded from all analyses.

Table 6.7 Matrix of first and second tier Pinawa study area environmental variables used for preliminary set of CCA analysis searching for meaningful variables.

Environmental Variables							
Analysis Name	First tier variables				Second tier variables		
	Turbidity	Macrophytes	Profile	Armouring status	DO	Water temp (°C)	Substrate heterogeneity
CCA1	1m value	% cover	angle	rip rap =1	site avg	site avg	% max het
CCA2	1m value	% cover	depth	rip rap =1	site avg	site avg	% max het
CCA3	1m value	% cover	angle	rip rap =0	site avg	site avg	% max het
CCA4	1m value	% cover	depth	rip rap =0	site avg	site avg	% max het
CCA5	1m value	richness	angle	rip rap =1	site avg	site avg	% max het
CCA6	1m value	richness	depth	rip rap =1	site avg	site avg	% max het
CCA7	1m value	richness	angle	rip rap =0	site avg	site avg	% max het
CCA8	1m value	richness	depth	rip rap =0	site avg	site avg	% max het
CCA9	1m value	patchiness	angle	rip rap =1	site avg	site avg	% max het
CCA10	1m value	patchiness	depth	rip rap =1	site avg	site avg	% max het
CCA11	1m value	patchiness	angle	rip rap =0	site avg	site avg	% max het
CCA12	1m value	patchiness	depth	rip rap =0	site avg	site avg	% max het

Environmental variables were awarded first tier status if they 1) had general ecological explanatory power; 2) were quantifiable; 3) showed significant differences between sites / types; 4) integrated other measured and/or unmeasured environmental conditions; 5) were considered not to be in violation of the principle of Ockham's razor³². For instance, turbidity met these criteria and was included in the matrix rationalised as follows: 1) excessive turbidity has been shown (chapter one) to affect the species composition and health of biota; 2) it can be measured in a variety of standard units, (in this case NTUs); 3) it was shown in all years to be significantly different between sites; 4) it captured the effect of "lesser" variables such as wind and boat traffic; 5) and while likely somewhat responsible for the pattern of some other first tier variables, (eg macrophyte composition; profile) it is not perfectly collinear with them.

Second tier rankings were assigned to those variables which 1) had general ecological explanatory power; 2) were quantifiable; and 3) were considered not to be in violation of Ockham's razor. In this study, DO qualifies as a good example. Although immediately important to many biota, and therefore a selection mechanism for some species, DO levels were not significantly different between sites. Arguably, in this study and other non-chemically polluted conditions, DO may have surrogates, (such as macrophyte presence (Westlake 1975; Jeffries and Mills 1990) and temperature) which together

³²

The principle of Ockham's razor is to avoid unnecessary multiplicity. Thus, of the non collinear variables which could be made available for inclusion in the matrix, highly redundant environmental variables (i.e. that did not add significantly to the R^2) were removed. (Legendre and Legendre 1998).

capture some of its ecological contribution.

Third tier variables had general ecological explanatory power but the measures of which were recorded in qualitative and semi-quantitative fashion (eg. wind speed and direction – field estimates were used in conjunction with local weather station information) or which were otherwise difficult to specify (eg. boat traffic). Although these variables can play an important role in understanding the local environment, better and more direct measures were captured using first or second tier variables. For example, the first tier variable turbidity captured better the impact of the third tier variable boat traffic.

The $(\ln+1)$ transformed 2002-2003 pooled species data set utilised in the majority of analyses in chapters four and five formed the species matrix for all CCA analyses.

6.2.3.2 Environmental variables - Final analysis

All combinations of first and second tier variables were analysed using CCA (LC method - this method finds the best fit between environment and species data). The cumulative percent of explained information on the first four axes (80-84 %) did not deviate substantially between analyses (Table 6.8), suggesting that the first four environmental variables were sufficient for the subsequent analysis. To determine which variables (and in what form) should be included in the second analysis, a matrix comprised of normalised axis 1 and axis 2 object scores (Table 6.9) was generated. The highest axis 1

Table 6.8 Percent explained variance per environmental variable, preliminary CCA data set.

% Explained Variance (rounded) -preliminary CCA analysis set							
	Axis 1	Axis 2	Axis 3	Axis 4	Axis 5	Axis 6	Axis 7
CCA1	30	21	19	13	7	6	3
CCA2	30	21	19	13	7	6	4
CCA3	30	21	19	13	7	6	4
CCA4	30	21	19	13	7	6	4
CCA5	29	21	18	13	10	7	3
CCA6	29	20	18	14	9	7	3
CCA7	29	21	18	13	10	7	3
CCA8	29	20	18	14	9	7	3
CCA9	31	21	19	12	8	5	3
CCA10	31	21	20	12	8	5	3
CCA11	31	21	20	12	8	5	3
CCA12	31	21	20	12	8	5	3

score for each variable was highlighted. The arrangement and form of variables associated with the greatest number of these high scores was selected as the environmental variable model to use in the final analysis. CCA2 (Table 6.9) scored the greatest number of axis 1 high scores ($n = 3$) and therefore it was selected as the variable arrangement and form (based on the first four variables only - ie turbidity; macrophyte % cover; profile (depth); armouring status - rip rap = 1) for the final CCA analyses. As should be expected, CCA 4, which contains the same variable arrangement as CCA 2 – except where armoured site scoring = “0” rather than “1” - produced the same site type separation on axis 1 as did CCA 2 (Fig 6.3). However, the CCA4 constellation was

essentially the mirror image of CCA2 on axis 2 (figure not shown).

Table 6.9 Normalised environmental variable scores for Axis 1 & Axis 2. Var1= turbidity; var 2= macrophytes; var3 = profile; var 4= armouring status; var5 = DO; var6 = water temperature; var 7 = substrate heterogeneity. Bold values represent axis 1 variable high scores (includes ties).

		normalised environmental variable axis scores						
Variable #		var1	var2	var3	var4	var5	var6	var7
CCA1	Axis 1	0.734	0.527	0.444	0.484	0.322	-0.203	0.438
	Axis 2	0.48	0.769	0.733	0.766	-0.504	0.018	-0.382
CCA2	Axis 1	0.728	0.538	0.568	0.488	0.311	-0.186	0.43
	Axis 2	0.387	0.736	0.66	0.684	-0.516	0.151	-0.371
CCA3	Axis 1	0.734	0.527	0.444	-0.484	0.322	-0.203	0.438
	Axis 2	0.48	0.769	0.733	-0.766	-0.504	0.018	-0.382
CCA4	Axis 1	0.728	0.538	0.568	-0.488	0.311	-0.186	0.43
	Axis 2	-0.387	-0.736	-0.66	0.684	0.516	-0.151	0.371
CCA5	Axis 1	0.72	-0.316	0.422	0.462	0.337	-0.194	0.456
	Axis 2	0.404	0.007	0.678	0.631	-0.446	0.255	-0.29
CCA6	Axis 1	0.698	-0.321	0.541	0.454	0.329	-0.186	0.451
	Axis 2	-0.398	-0.005	-0.646	-0.629	0.448	-0.256	0.293
CCA7	Axis 1	0.72	-0.316	0.422	-0.462	0.337	-0.194	0.456
	Axis 2	0.404	0.007	0.678	-0.631	-0.446	0.255	-0.29
CCA8	Axis 1	0.698	-0.321	0.541	-0.454	0.329	-0.186	0.451
	Axis 2	-0.398	-0.005	-0.646	0.629	0.448	-0.256	0.293
CCA9	Axis 1	0.724	0.491	0.428	0.467	0.333	-0.185	0.453
	Axis 2	-0.464	-0.235	-0.741	-0.739	0.496	-0.123	0.339
CCA10	Axis 1	0.719	0.49	0.561	0.474	0.319	-0.168	0.442
	Axis 2	-0.342	-0.195	-0.662	-0.63	0.499	-0.24	0.334
CCA11	Axis 1	0.724	0.491	0.428	-0.467	0.333	-0.185	0.453
	Axis 2	-0.464	-0.235	-0.741	0.739	0.496	-0.123	0.339
CCA12	Axis 1	0.719	0.49	0.561	-0.474	0.319	-0.168	0.442
	Axis 2	-0.342	-0.195	-0.662	0.63	0.499	-0.24	0.334

6.3 RESULTS

CCA analysis of the final environmental variable set ($\text{var} = 4$) produced an object constellation that separated sites by site type (Fig 6.3). Site type differences were significant ($P = <0.001$; $\alpha = 0.050$; $\beta = 0.997$) based on axis one LC object scores (Table 6.10). A second CCA ($\text{var} = 4$) analysis (WA method -- weighted average for objects scores) also produced an object constellation that differentiated objects by site type (Fig 6.4). Site type differences were significant ($P = 0.001$; $\alpha = 0.050$; $\beta = 0.983$) based on axis 1 WA scores. Both LC and WA methods produced high (and identical) between-set correlations for both axes ($P \ll .0001$; $r^2 = 0.9579$; $P \ll .0001$; $r^2 = 0.9383$).

Fig 6.3 CCA (LC method) object constellation of Pinawa study area sample sites. ■ indicates unarmoured sites, ● indicates armoured sites. Species composition data constrained by turbidity, % macrophyte coverage, maximum site depth and armouring status (armouring = 1). Percent explained variation listed on diagram axes.

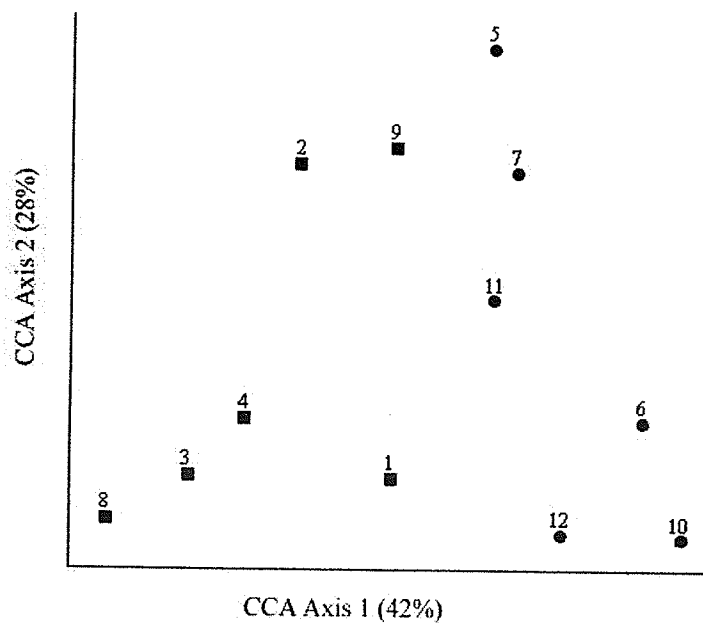


Fig 6.4 CCA (WA method) object constellation of Pinawa study area sample sites. ■ indicates unarmoured sites, ● indicates armoured sites. Species composition data constrained by turbidity, % macrophyte coverage, maximum site depth and armouring status (armouring = 1). Percent explained variation listed on diagram axes.

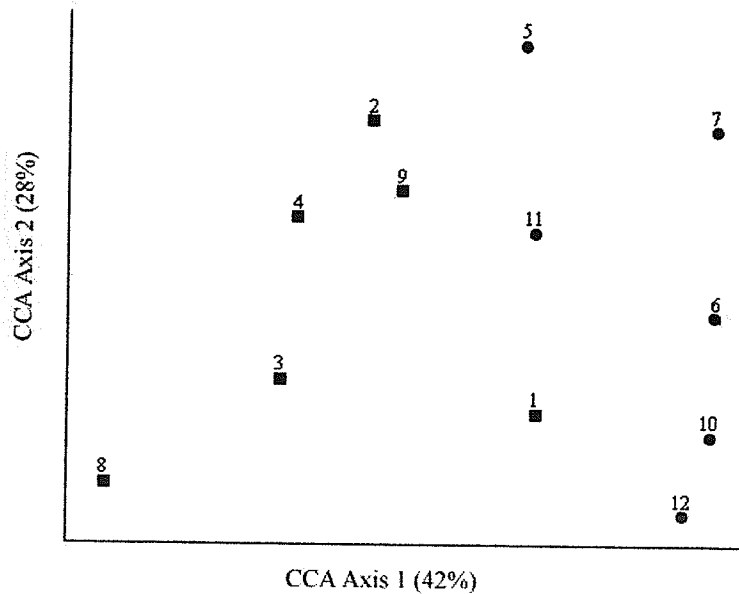


Table 6.10 Object scores for CCA (LC) and CCA (WA) methods. Unarmoured sites include sites 1-4; 8-9; armoured sites include sites 5-7; 10-12.

LC scores				WA scores			
Axis 1		Axis 2		Axis 1		Axis 2	
Unarmoured	Armour	Unarmoured	Armour	Unarmoured	Armour	Unarmoured	Armour
d		d		d		d	
-0.011	0.012	-0.028	0.074	0.011	0.006	-0.058	0.133
-0.032	0.047	0.047	-0.014	-0.069	0.1	0.094	-0.007
-0.058	0.018	-0.027	0.045	-0.115	0.099	-0.04	0.088
-0.045	0.055	-0.013	-0.042	-0.107	0.097	0.044	-0.069
-0.076	0.013	-0.037	0.014	-0.202	0.01	-0.094	0.036
-0.01	0.028	0.051	-0.041	-0.055	0.084	0.057	-0.11

Manipulation of the armour variable did not produce any appreciable change to the results. Neither removal of the armouring status variable (presence / absence) (Fig 6.5), nor allocation by armour age (Fig 6.6) resulted in any substantial change to the object constellations shown in Figs 6.3 and 6.4. High redundancy among the environmental variables is indicated by the strong common directionality of all variables (Fig 6.7).

Fig 6.5 CCA object constellation of Pinawa study area sample sites. ■ indicates unarmoured sites, ● indicates armoured sites. 8-9). Species composition data constrained by turbidity, macrophyte coverage and maximum site depth (armouring status variable removed). Percent explained variation listed on diagram axes.

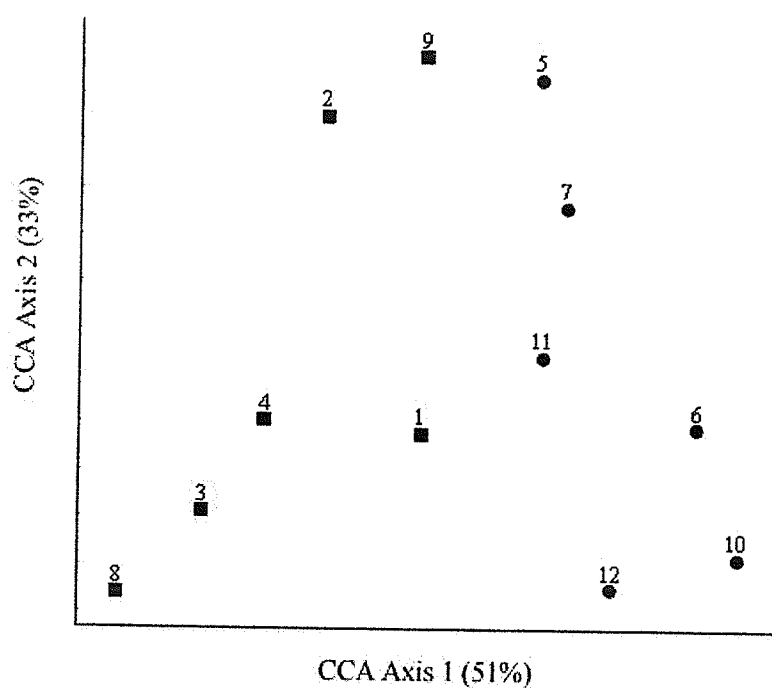


Fig 6.6 CCA object constellation of Pinawa study area sample sites. ■ indicates unarmoured sites, ● indicates armoured sites. 8-9). Species composition data constrained by turbidity, macrophyte coverage, maximum site depth and armouring status (including armour presence / absence & age). Percent explained variation listed on diagram axes.

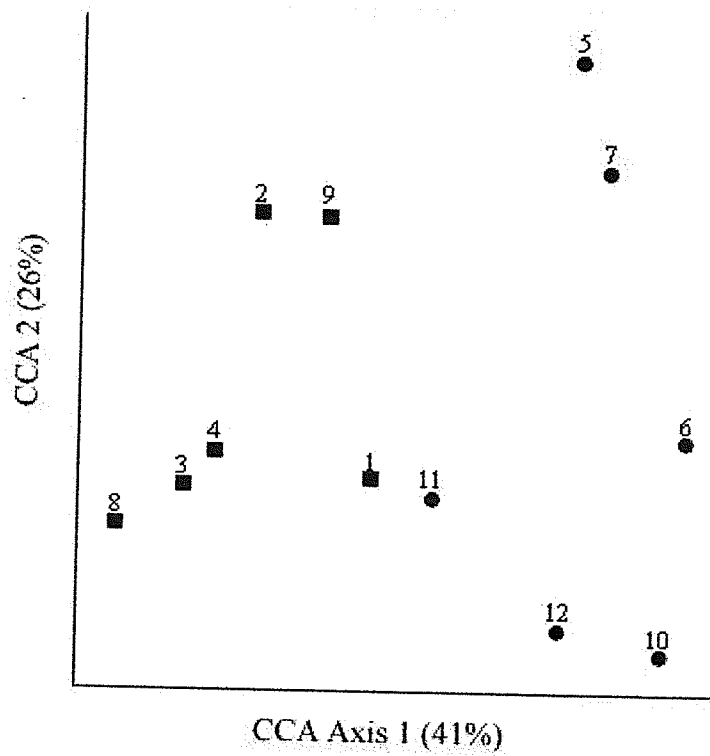
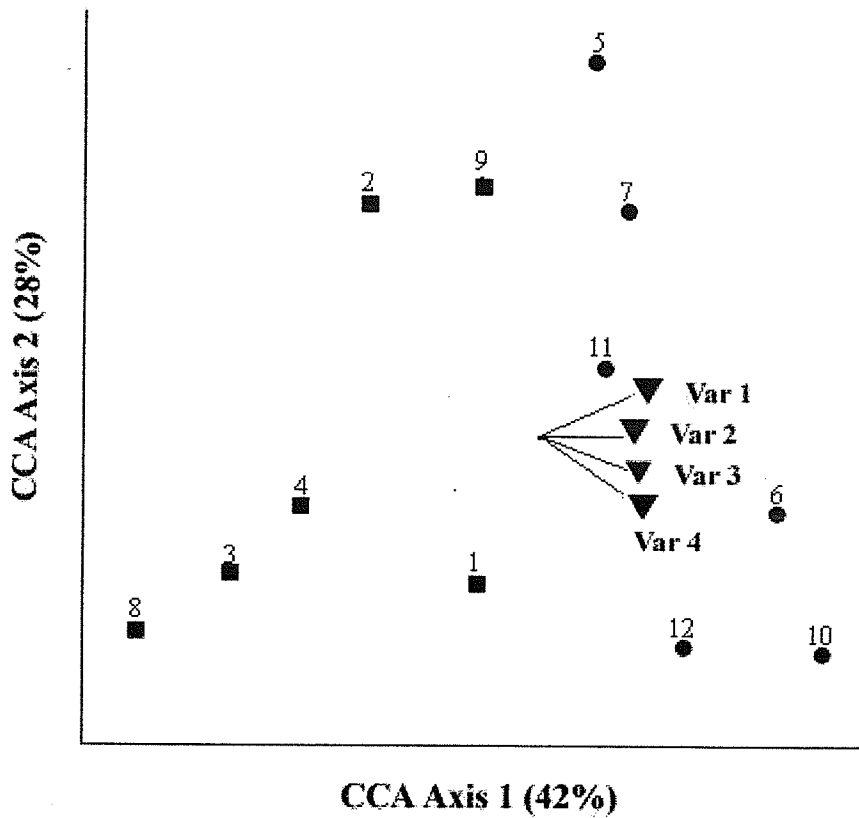


Fig 6.7 CCA triplot (objects and environmental variables shown; species not shown) of Pinawa study area sample sites. ■ indicates unarmoured sites, ● indicates armoured sites. 8-9). Species composition data constrained by environmental variables (var 1 = turbidity; var 2 = macrophyte coverage; var 3 = maximum site depth; var 4 = armouring status) designated by ▲. Percent explained variation listed on diagram axes.



6.4 DISCUSSION

6.4.1 Appropriateness of environmental variables

The choice of environmental variables as measures of habitat quality require validation if these variables are to have any meaningful ecological interpretation attached to them.

Since only significantly different variables were retained in the final CCA analyses ($n = 4$), one might merely conclude that different things are different, and seek no further conclusions about the relationship(s) between these variables or the habitats these variables help describe. However, quite apart from their measured differences, these variables do have important ecological interpretation.

Firstly, the strong positive between-set (species variables and environmental variables) correlations indicate that the littoral species assemblage is responsive to the disparate environmental conditions associated with each site type. Secondly, the suite of environmental variables utilised in the final set of CCA analyses showed a high level of redundancy among the variables (Fig 6.7), but without collinearity (but see var 4 - "armouring status" discussed below). Overall, these data reveal a negative correlation between sedimentation (var 1 - turbidity) and fish habitat. This broad conclusion is supported extensively within the literature (Berkman and Rabeni 1987; Kerr 1995). These results also indicate that these variables work in concert with one another. This phenomenon seems to be particularly valid with respect to var1 (turbidity) and var 2 (macrophyte cover). This finding is in line with Hanson and Butler (1994), who found a

strong inverse correlation between turbidity and macrophyte development. Apart from the its direct impacts on fishes, water-borne turbidity reduces light availability for aquatic plants which themselves also provide direct benefits to fishes and to fish habitat. Indeed, the general ecological importance of macrophytes to fish habitat is well accepted, since among other things, they provide structure and protection for young of the year and juveniles (Jeffries and Mills 1990; Hayes *et al.*, 1996; Langhorne *et al.*, 2001). Apart from influencing the development of macrophytes for important structure, turbidity has been negatively correlated to plant photosynthesis Blanch *et al.*, (1998) thus reducing primary energy inputs at the local level (also see sec 1.2.2.3). The strong positive correlation between (reduced) turbidity and maximum site depth / bank slope is also evident. Together, these three ecologically important variables show strong positive structural correlations with the armouring variable (Fig 6.7). Thirdly, the strong positive correlation between the LC method (which seeks the best fit between species and environment data) and the WA method (weighted average for objects scores) indicates that the variables are robust and the data are not susceptible to alternative interpretations based simply on the choice of analytical method. If these values had low correlation or if the object constellations generated by LC and WA methods values were substantially different, it would suggest that inappropriate environmental variables had been included in the analysis. However, such is not the case. Together, these results suggest that the method for selecting environmental variables was appropriate for showing 1) differences between site types and 2) for finding variables with correlated (eg. macrophyte cover) and casual (eg. turbidity) links to the properties of the variable under investigation (i.e.

shoreline armouring).

6.4.1.1 Role of armouring status variable

Comparison of Figs 6.3 and 6.5 shows the removal of the armouring status variable had minimal effect on the overall object constellation. Although some objects did shift (eg. see objects 1, 5, 9 & 10), this movement occurred ostensibly on axis 2, implying this variable is of lesser importance than other variables in the data set. However, this result does not necessarily mean the impact of armouring is not ecologically relevant. Instead, it may indicate that the measurement of armouring is not sufficient simply by the designation “armoured” or “unarmoured” and its binary manifestation. It seems clear from this work that some quantification of the various qualitative differences between types of armour (eg. felsic vs mafic rock) are necessary if we are to understand the impact of armour at a finer resolution. This suggestion is moderately supported by the movement of site 11 (armoured) on axis 1, after armour age was assigned, albeit as a semi-quantitative variable. Nevertheless, in general, the “armouring status” variable (var 4) was found to be highly (but not perfectly) collinear with the other environmental variables. This relationship is not surprising. The role shoreline armouring plays in the preservation of water clarity and in the preservation of bottom contour seem clear. Indeed, all the environmental variables within the final analysis trend in the same direction as those sites characterised by higher species composition values (Fig 6.7). These results indicate that these environmental variables are associated with comparatively better fish habitat than those sites which exhibit less of these qualities.

Specifically, sites with reduced turbidity, increased macrophyte cover and greater site depth (rather than exclusively shallow water) might be considered to be predictors of higher quality fish habitat in boreal river littoral ecosystems. It appears that rip rap shoreline armouring is associated with preserving this type of habitat, at least when the alternative management approach is leaving erodible shorelines to crumble into the watercourse and create clay-based littoral in-fill.

6.4.2 General conclusions

Since the object constellations from all CCA analyses are similar to those in chapters four and five, it is appropriate to suggest that species assemblage acts as a surrogate for habitat quality. This general result confirms one of the assumptions raised in chapter two, where I stated that the best mechanism for measuring fish habitat is through analysis of the information contained in the resident fish assemblage.

In some cases, the environmental data contained little variance and was ignored as a source of explanation for differences found in fish assemblages between site types.

However, it was hypothesised at the outset of the study that excessive turbidity arising from 1) shoreline erosion and / or; 2) resuspension of sediments at low slope sites typical of unarmoured shorelines was an important factor, in the differences found in fish assemblage information between site types.

Water-borne turbidity reduces light availability for aquatic plants which provide

important vertical structure for fishes. Additionally, excessive turbidity causes physiological damage to fishes. Whether the mitigation of these turbidity generated problems is a property exclusive to riprap shoreline armouring or not is impossible to tell within the framework of this study. Nor is it possible to determine from this work whether riprap armouring carries unique properties that affect fish habitat in other manners. However, from these data, it is clear that there is a strong correlation (and in the case of turbidity, causality) between the role of rip rap armouring, the environmental variables associated with its use and comparatively high quality fish habitat. However, there are intuitive conditions that limit the contributions rip rap can make to fish habitat. Although these limitations are not directly quantifiable at this point, they are indicated from the CCA analyses. Specifically, site depth plays an important explanatory role in all CCA analyses. Although the deepest site (site 10) has the highest CCA axis 1 scores, it is obvious that “excessive depth” is not tenable for littoral fishes. Therefore, from a management perspective, a guarded approach to rip rap use seems appropriate. Specifically, in order to retain a positive impact on fish habitat, the amount of rip rap in-fill must be limited to maximise littoral zone depth variation.

Chapter 7

Conclusions, Discussion and Recommendations

7.1 SUMMARY OF CONCLUSIONS

Through the assessment of the impact of rip rap armouring on fish habitat, this project pursued six general goals: 1) to select and evaluate an appropriate study area in which to conduct a mensurative study; 2) to evaluate sampling methodology; 3) to assess Karr's Index of Biotic Integrity as a method for comparing habitat types in a small scale application; 4) to use species composition and surrogate measures for productive capacity to assess habitat type; 5) to evaluate a suite of appropriate environmental variables which most substantially influence species composition; and 6) to compare assemblage structures in accordance with these environmental variables. In the pursuit of these objectives, I have reached several specific conclusions which I encapsulate here.

1) I used a project-based scale to designate the spatial scale of this research which occurred over four field seasons (2001-2004), providing diurnal, seasonal and yearly variation. Analysis of a larger data set showed that the Pinawa study area was a satisfactory analogue for a larger reach of the Winnipeg River. I also found that proximate units laid out on the landscape were not spatially autocorrelated (at least among small fishes). This finding provides insight into future sampling designs for small-bodied fishes. The fine grain approach used in this study demonstrated that within the

littoral zone substantial variability can occur, even between proximate sites. Therefore, it behoves resource managers to ensure their goals are consistent with such variation.

2) Various fish capture methodologies have been assessed by many researchers. I found a modified beach seine captured fishes sufficiently well to have confidence in the method for determining the littoral fish assemblage. Analysis of sampling frequency showed that while all sampling frequency regimes tended to yield similar conclusions about site type, increased sampling effort overall contributed to a more complete understanding of differences between sites and site types. Information which best characterised habitat arose from sampling weekly, rather than bi-weekly or monthly, and results suggested that infrequent sampling might lead to erroneous characterisations of habitat by missing important information associated with short term peaks or troughs in the fish assemblage. While time-of-day sampling (daylight hours) had little impact on the assessment of habitat, seasonality did affect results. For example, during cool weather seasons and spring-like conditions, fishes tended to deselect armoured sites in favour of shallow, warmer sites. This makes sense intuitively, as fishes will adapt their choice of otherwise preferred local habitat, based on the need to optimise the seasonal growth spurt which is seasonally and temperature driven (see Bacon *et al.*, 2005).

3) Although this project was an examination of a specific alteration to physical habitat, it is clear that physical habitat characteristics alone (such as those measured by Plafkin *et al.*, (1989)) may not sufficiently characterise fish habitat in a large river context – at least

not without a concomitant assessment of physical habitat and species interactions. This conclusion supports the seminal work by Karr (1981) who claimed that appropriate biological measures incorporate biological, chemical and physical components of ecosystems. With appropriate modifications for local ichthyofauna, Karr's (1981) Index of Biotic integrity (IBI) can be utilised at small scales to show differences between various habitats where these differences exist.

4) PCA on species composition and fish production surrogates generated consistent results. Although composition and production might be assumed to be co-linear (and thus self validating), this assumption is not necessarily true. Nevertheless, the similarity of results based on these two related but different ecological measures (composition and production) indicate that the distinction found between site types is well-founded.

5) Environmental variables with general explanatory power, which encapsulated other "lesser" variables and which were statistically different between site types were found to be the most appropriate variables for use in the constrained analyses of species composition data. This suite of variables include relatively straightforward measures such as turbidity, as well as more complicated measures such as depth. For instance, while the assessment of habitat using a volume-based measure was not appropriate (chapter five), its important ecological role insofar as it is related to site depth was selected as a significant environmental variable in a CCA analysis (chapter six). This apparent conflict is explicable given the non-linear primary and secondary production

curves associated with depth. Most pelagic species containing greater biomass, species abundance and richness are associated with the metalimnion and epilimnion, within the littoral zone while benthic species which generally occur fewer in number and with less species richness, inhabit the littoral profundal.

6) Shoreline armouring has several important co-variables, including water clarity; macrophyte development and maximum site depth. How closely these properties belong to rip rap as opposed to other forms of armouring (eg. geo-textiles) is not clear at this point and is work for another study. Notwithstanding, after constraining species composition information by a suite of select environmental variables, it became clear that species composition acts as a complex and sensitive indicator of habitat quality and change.

7.2 Final Discussion - Application

The general conclusion emanating from chapters four through six, is that rip rap armoured shorelines are associated with better quality habitat than are unarmoured shorelines. This association appears to be causal, since rip rap sites were shown consistently to have lower turbidity values than unarmoured sites. Increased water clarity improves fish habitat in at least three ways. Firstly, the reduction of abioseston in the water column directly improves the systemic living conditions of fishes and cognate biota by reducing the direct and indirect deleterious effects of suspended solids upon

these organisms. Secondly, improved water clarity reduces light attenuation through the water column, thereby enhancing macrophyte development. Habitats with macrophyte development are generally associated with better fish habitat than are non-vegetated sites. Thirdly, over longer time frames, the reduction in suspended solids contributes to the maintenance of shoreline contour by reducing the gradual in-fill of the littoral zone by sediment deposition. Although very shallow sites may experience emergent macrophyte development (eg. *Typha spp*), typically these sites do not have sufficient water depth to facilitate submerged macrophyte development which are the species most closely associated with higher quality fish habitat.

7.2.1 Quantifying a HADD

The in-fill properties of rip rap are obvious and cannot be denied. Generally, it is this characteristic which subjugates proposed rip rap installation to the HADD framework. However, the effective loss of shoreline (i.e. linear distance out from shore) in the study area due to “excessive” turbidity (i.e. ≥ 25 NTUs) was calculated to be 1.47m. Since mean turbidity values adjacent to armoured sites were always < 25 NTUs, it might reasonably be argued that any rip rap in-fill that occupied less than 1.47m linear distance from shore would constitute a net *gain* of fish habitat in the study area, by eliminating the excessive turbidity zone. Extending this thinking beyond the Pinawa study area, the quantification of rip rap as a potential HADD, or alternatively, as a mitigating procedure, can be based on calculating the difference between the proposed amount of rip rap in-fill and the loss of otherwise effective shoreline where site-specific ambient turbidity is >25

NTUs. Although this simple application clearly overlooks other subtle interactions between rip rap and the resident fish assemblage; it is in keeping with a precautionary approach and it can be suggested as an effective method for quantifying NNL and HADD concerns until further work on armoured shorelines is able to quantify further these other, likely more subtle, ecological relationships.

7.2.2 Ecological health and integrity

While Lange *et al.*, (2001) suggested that rip rap armoured shorelines may add fish habitat in certain circumstances, they classified its application (in the Great Lakes region) as “unlike habitat”. Long and Walker (2005) suggested that armouring may push sites towards greater biotic integrity in boreal riverine ecosystems. However, mixed results from further work here suggests that armouring may only push armoured sites towards better ecological health, though not to higher ecological integrity. Jones *et al.*, (1996) reminded habitat managers to be clear about management objectives. Indeed, if the improvement of ecological health is sought, then shoreline armouring should be considered a best management practice available for site-specific evaluation. On the other hand, if the management objective is the improvement of ecological integrity, then managers must determine if the processes associated with confronting new sources of erodible surfaces (either human or naturally induced) has higher integrity (though probably depressed ecological health in the medium term) by letting nature run its course than does the application of ameliorative methods (i.e. shoreline protection). This thought is neither idle speculation nor hyperbole, since the 1986 DFO NNL policy seeks

the maximum productive capacity (i.e. ecological health) via natural processes (ecological integrity).

7.2.3 *The Conundrum of species abundance*

During the winters of 2002-2005, I have noticed several ice fishing huts located on the ice between sites three / fourteen and site four³³. This stretch of the river is also active during the summer fishing season (extending down to site five). This activity suggests that the area is good for fishing and by extension, it is good fish habitat. However, sites three and four were shown to be the worst sites according to IBI, PCA and CCA analyses, using small fish information. Similarly, site five was shown to be the worst among armoured sites (but which still possessed better scores than unarmoured sites). This observation leads to a perplexing question: perhaps, the “good” sport fishing enjoyed by many anglers in this area throughout the year causes a reduction in species composition of small forage and bait fishes. Therefore, *these low-score sites actually possess good fish habitat, but it is not evident at the small fish level since the low abundances of these species is attributable to high sport fish predation of these species*. Conversely, poor large sport fish habitat (although this point is merely speculative) associated with other sites therefore, would facilitate the development of the other higher scored sites.

Therefore, this concern raises the need to add a measurement of large fish (that must

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It should be noted however, that these sites are closest to a boat /launch and this access point may be the only reason ice-fishing huts are positioned here, all other things being equal.

include a site-weighting for angling pressure and large bodied species distribution) or argues for the need to examine the issue experimentally; where predation and angling pressures cannot confound the results.

7.3 Final Discussion - Recommendations

Jones *et al.*, (1996) divided fish habitat management issues into two broad categories. Type A issues are those matters where management objectives have been specified *a priori* and from which arises the need to determine whether specific habitat alterations can achieve these stated goals. Type A issues are habitat driven, and carry a perspective that seeks analysis of methodologies such as might be utilised in ecological restoration programmes (for example, see Cairns 1988). In contrast, type B issues are those where specific habitat alterations are proposed and there is a need to determine what the impact of these manipulations might be. As such, type B issues require some form of predictive model (either through the determination of causal relationships or through synthesis based upon the aggregation of empirical evidence across independent *a posteriori* assessments) to facilitate management decisions regarding proposed management practices. Typically, Type B issues are primarily economically, or at least non-habitat driven, concerns. Traditionally, rip rap armouring has been installed as a best management practice to protect shorelines and other erodible surfaces from impacts associated with aquatic interaction upon terrestrial environments, usually as a result of socio-economic development projects (eg. dam, road or bridge abutment construction).

Under this paradigm, the examination of rip rap armouring impacts to fish habitat is a type B management issue and as such requires comment about its *expected* role in future applications. Although simple experiments can produce predictive relationships between clear variables, ecological interactions are complex and a simple experiment designed to demonstrate the impact of rip rap armouring on fish habitat may be methodologically sound and yield results that are statistically significant, but be ecologically meaningless. On the other hand, mensurative examinations of extant circumstances that are used to assess ecological interactions can have high explanatory and therefore, to some extent, predictive power, (Hurlburt 1984; see discussion chapter 2).

This project examined interactions between a littoral zone fish assemblage and its local environment to establish the general structure of this relationship. Through this process, a preliminary understanding of ecological relationships with respect to habitat and a specific physical alteration was developed. The general conclusion from this study consistently found a clear association between armoured sample sites and comparatively higher quality fish habitat. This broad conclusion suggests that the rip rap armouring in boreal rivers should now be treated as a type A management issue. Indeed, the 1986 DFO no-net-loss policy has been well established and to meet the goals of this policy, shoreline protection schemes should be considered and evaluated as part of the suite of metrics suitable for attaining those policy goals.

Although various researchers have examined the role shorelines play on fish habitat

(Knight and Cooper 1991; Jennings *et al.*, 1996; Shields *et al.*, 2000) which together begin to built an aggregate perspective on the interactions between various environmental conditions and biotic assemblages, no experimental work has been undertaken. The lack of experimental work in this area is likely as a result of two reasons: 1) the financial commitment required to developing a long term experimental facility is not inconsequential and to date, there has been little fiscal incentive to pay for this research. Secondly, the results of experimental work will not be universal, since they will necessarily be constrained by the conditions associated with the biogeography of the study area (i.e. regional biotic assemblage; water chemistry). Nevertheless, it is recommended that experimental approaches be developed to test the conclusions developed through the mensurative process for the conditions that do exist in the large Precambrian Shield biome.

Appendix A - Summary of Fish Species

Table A.1. Summary of fish abundance (by year) from all study areas and all sample sites

Species		Number of fish			
Scientific name	Common name	2002	2003	2004	Total
<i>Hiodon tergisus</i>	Mooneye	0	5	0	5
<i>Notemigonus crysoleucas</i>	Golden shiner	3	2	6	11
<i>Notropis atherinoides</i>	Emerald shiner	3682	2175	1228	7085
<i>Notropis heterodon</i>	Blackchin shiner	7	4	3	14
<i>Notropis heterolepis</i>	Blacknose shiner	8	2	16	26
<i>Notropis hudsonius</i>	Spottail shiner	1490	8871	1057	11418
<i>Notropis texanus</i>	Weed shiner	165	253	307	725
<i>Notropis volucellus</i>	Mimic shiner	5	3695	8428	12128
	Unknown (shiner) fry	0	432	469	901
<i>Pimephales promelas</i>	Fathead minnow	7	5	23	35
<i>Rhinichthys cataractae</i>	Longnose dace	0	5	0	5
<i>Catostomus commersoni</i>	White sucker	614	237	1063	1914
<i>Esox lucius</i>	Northern pike	6	84	28	118
<i>Oncorhynchus mykiss</i>	Rainbow trout	0	0	16	16
<i>Percopsis omiscomaycus</i>	Trout-perch	0	0	2	2
<i>Lota lota</i>	Burbot	1	1	0	2
<i>Culaea inconstans</i>	Brook stickleback	26	100	9	135
<i>Pungitius pungitius</i>	Ninespine stickleback	2	1	0	3
<i>Cottus bairdi</i> inc <i>C. cognatus</i>	Sculpin spp.	82	38	4	124
<i>Ambloplites rupestris</i>	Rock bass	291	1396	8	1695
<i>Micropterus dolomieu</i> *	Smallmouth bass	243	307	207	757
<i>Pomoxis nigromaculatus</i>	Black crappie	0	1	0	1
<i>Etheostoma exile</i>	Iowa darter	6	24	22	52
<i>Etheostoma nigrum</i>	Johnny darter	1026	2476	841	4343
<i>Perca flavescens</i>	Yellow Perch	227	1875	118	2220

<i>Percina caprodes</i>	Logperch	50	7	3	60
<i>Percina shumardi</i>	River darter	45	1	2	48
<i>Sander vitreus</i>	Walleye	3	1	1	5
	TOTAL	7989	21998	13861	43848

Note: I follow Stewart & Watkinson (2004) for the scientific spelling of smallmouth bass

Appendix B - Summary of Site Descriptions

Table B.1. Summary of site descriptors. Note 1 A= armoured site; U = Unarmoured site; N= natural exposed rock shorelines; Note 2 (Macrophyte % cover) - total site percent cover, including emergent and submerged species. Sites 1-16 (Pinawa study area); 17-22 (Whiteshell study area); 23-28 (St Georges study area).

Site	GPS Coordinates UTM 15	Compass Bearing	Site Context (position on the shoreline)	Shoreline & proximate vegetation	% < 2mm	Substrate other features	Status code (note 1)	Mac. % cover (note 2)	Contour	
									Site slope (°)	2004 z max (cm)
1	O295120 5559590	NE	base of large inlet / bay	grass; Mature birch upslope	64	birch leaves; soft clay	U	16.7	8.3	133
2	O295264 5559595	NE	small bay - 3x size of sampling unit	shrubs; conifers	98	sand & periodic woody debris	U	11.3	2.8	71
3	O295291 5559549	NE	base of gentle indentation in shoreline	grass; black willow; 1 small dead conifer at edge of site	93	soft clay	U	11.7	5.5	80
4	O295337 5559353	ENE	adjacent to straight shoreline	Ash at site edge - removed by beaver 2003; oak upslope	68	soft clay	U	15.7	5.1	73

Site	GPS Coordinates UTM 15	Compass Bearing	Site Context (position on the shoreline)	Shoreline & proximate vegetation	% < 2mm	Substrate other features	Status code (note 1)	Mac. % cover (note 2)	Contour	
									Site slope (°)	2004 z max (cm)
5	O295314 5559308	SE	adjacent to straight shoreline; influenced by site 4 (upstream) turbidity	Mature oak at shoreline	86	soft clay	A	20.7	10.3	109
6	O295184 5559231	SE	adjacent to straight shoreline w/ small protrusion into river	Shrubs near shore; mature oak upslope	96	soft clay	A	39.9	13.2	138
7	O295100 5559169	SE	adjacent to straight shoreline	Shrubs; mature oak upslope; one mature oak overhanging near site	48	soft clay	A	23.7	10.2	125
8	O294525 5558679	S	upstream edge of a long shallow bay (cf site 10)	<i>Typha</i> & 1 shrub; grass, & mature oak upslope	95	soft clay	U	10	4.9	66

Site	GPS Coordinates UTM 15	Compass Bearing	Site Context (position on the shoreline)	Shoreline & proximate vegetation	% < 2mm	Substrate other features	Status code (note 1)	Mac. % cover (note 2)	Contour	
									Site slope (°)	2004 z max (cm)
9	O294498 5558687	S	slightly downstream from site 8	<i>Typha</i> ; grass, ash sapling - removed by beaver in 2003; mature oak upslope	98	<i>Typha</i> roots; clay	U	16.7	2.4	78
10	O294324 5558638	SE	downstream edge of a long shallow bay (cf site 8)	none. Mature oak upslope	95	soft clay	A	29.3	22.7	221
11	O294181 5558593	S	adjacent to straight shoreline	none. Mature oak upslope	90	soft clay	A	29.7	9.1	111
12	O294061 5558554	S	adjacent to straight shoreline	<i>Typha</i> ; Mature oak & ironwood upslope	95	soft clay	A	36.3	14.6	144
13	O295236 5559607	N	proximate to outside bend	Conifers upslope	—	smooth granite	N	—	10.3	85
14	295294 5558554	ESE	proximate to outside bend	none; some grass tufts upslope	—	smooth & granite	N	—	14.4	120

Site	GPS Coordinates UTM 15	Compass Bearing	Site Context (position on the shoreline)	Shoreline & proximate vegetation	% < 2mm	Substrate	Status code (note 1)	Mac. % cover (note 2)	Contour	
						other features			Site slope (°)	2004 z max (cm)
15	O294589 5558674	E	small promontory with strong current	none; shrubs & mature oak upslope	—	smooth granite	N	—	19.4	115
16	O294589 5558605	S	small rock bay	none; shrubs & mature oak upslope	—	smooth granite shelf	N	—	28.7	85
17	O297157 5555244	NW	adjacent to straight shoreline along roadside	marginal <i>Typha</i> ; some grass upslope	—	clay / sand	A	—	17	76
18	O297287 5555421	NW	adjacent to straight shoreline along roadside	none; some grass upslope	—	clay / sand	A	—	10.2	82
19	O298287 5556275	NNW	site 50 m off road, separated by heavy bush	grassed bank w/ heavy ash cover to water's edge	—	soft clay	U	—	15.2	63
20	O298868 5558017	W	at outflow of small creek	heavy grass & small willow to water' edge	—	soft clay	U	—	21.4	74

Site	GPS Coordinates UTM 15	Compass Bearing	Site Context (position on the shoreline)	Shoreline & proximate vegetation	% < 2mm	Substrate other features	Status code (note 1)	Mac. % cover (note 2)	Contour	
									Site slope (°)	2004 z max (cm)
21	O303119 5559785	NNW	Beside boat launch; heavy armouring	none	—	smooth granite, gravel & cobble	A	—	8.4	49
22	O303187 5559182	NNW	Site is 25 m off road, separated by heavy bush	grassed bank; heavy mixed forest to water's; large black willow in water	—	soft clay & leaf /needle debris	U	—	7.2	59
23	O702566 5601764	SSE	site is armoured shore of old ferry landing	grassed bank; a few golden willow saplings upslope	—	clay	A	—	10.9	68
24	O702607 5601905	W	adjacent to oilseed fields (2m buffer)	grassed buffer strip	—	soft clay	U	—	11.3	39
25	0702028 5602018	E	adjacent to straight shoreline 25m from highway	some golden willow saplings adjacent to site	—	clay	A	—	10.8	57
26	702050 5601971	E	adjacent to straight shoreline 35 m from highway	some golden willow saplings adjacent to site	—	clay	A	—	9.6	61

Site	GPS Coordinates UTM 15	Compass Bearing	Site Context (position on the shoreline)	Shoreline & proximate vegetation	% < 2mm	Substrate other features	Status code (note 1)	Mac. % cover (note 2)	Contour	
									Site slope (°)	2004 z max (cm)
27	O703635 5599922	ENE	sits atop small inlet, 100 m from highway	grass down to water's edge	—	soft clay	U	—	6.9	49
28	O703681 5599910	SE	sits at the base of a steep cliff, 100 m from highway	none	—	clay; some sand	U	—	19.5	66

Appendix C - Approval for research
Office of Research Services

Approval for research - Animal Care Utilization Protocol, reference FO2-015

“The Effects of Rip Rap Shoreline Armouring on Fish Habitat”

Copy of approval Attached

Approval to capture live fishes - Manitoba Department of Conservation

Copy of approval Attached



UNIVERSITY
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27 June 2002

TO: Dr. R. Baydack, Natural Resources Institute
303-70 Dysart Road

FROM: L.D. Campbell, Chair, Fort Garry Campus Protocol Management
and Review Committee

RE: "The effects of rip rap shoreline armouring on fish habitat"

Please be advised that your Animal Care Utilization Protocol, reference **F02-015**, has received **approval** by the Fort Garry Campus Protocol Management & Review Committee and is valid until **June 30, 2003**. The procedures described by you in the protocol have placed this research in the Category "C" of invasiveness.

This approval has been granted with the understanding that:

- 1) No animals will be euthanised. If euthanasia is deemed necessary upon renewal of this protocol, discussion regarding an appropriate method of euthanasia will need to be addressed.
- 2) The predator/prey component of this study has been removed.

It is understood that these animals will be used only as described in your protocol. The protocol must be kept current. Should changes become necessary, very minor alterations can be made with the prior written approval of a university Veterinarian and written notification of the Chair of the Fort Garry Campus Protocol Management and Review Committee. More substantive changes will require resubmission to and reassessment by the Fort Garry Protocol Management and Review Committee. If approved, this will result in the assignment of a new protocol reference number.

F02-015/1 - June 03 - June 04

F02-015/2 - June 04 - June 05

...2

not emailed to
Shelley
4/24/02

MANITOBA CONSERVATION
FISH HABITAT MANAGEMENT SECTION

SCIENTIFIC COLLECTION PERMIT

sued under the authority of the Fisheries Act (Manitoba) and the Fishing
Licence Regulation and Fishing Licence Fee Regulation made thereunder.

sued to: Jeff & Michelle Long

hereby authorized to collect, transport and possess fish within the
Province of Manitoba subject to the following conditions:

Release live fish only in the water from which they were taken.

Fish may not be sold, traded or bartered.

The use of chemicals and explosives as aids in collecting fish is
prohibited.

This permit expires on 31-Oct-02 following date of issue.

A report must be submitted to Fisheries Branch, Box 40, 200 Saulteaux
Crescent, Winnipeg, MB R3J 3W3 upon expiration of this permit indicating;
location, species, number and disposition of the collected specimens.

Special Conditions: Capture fish by means of beach seine in the Winnipeg
River at Pinawa. Fish may be transported live to
nearby laboratory facilities and subsequently
returned to the Winnipeg River near Pinawa, or
preserved for later identification.

Shelley Matkowski

Issueu by

Fish Enhancement Biologist

Title

Signature of Permittee

5-Apr-02

Date of Issue

Permit Number: 11-02

MANITOBA CONSERVATION
FISH HABITAT MANAGEMENT SECTION

SCIENTIFIC COLLECTION PERMIT

Issued under the authority of the Fisheries Act (Manitoba) and the Fishing Licence Regulation and Fishing Licence Fee Regulation made thereunder.

Issued to: Jeff Long

Address: Box 534, Pinawa MB ROE 1LO

Is hereby authorized to collect, transport and possess fish within the Province of Manitoba subject to the following conditions:

- . Release live fish only in the water from which they were taken.
- . Fish may not be sold, traded or bartered.
- . The use of chemicals and explosives as aids in collecting fish is prohibited.
- . This permit expires on 31-Oct-03 following date of issue.
- . A report must be submitted to Fisheries Branch, Box 40, 200 Saulteaux Crescent, Winnipeg, MB R3J 3W3 upon expiration of this permit indicating; location, species, number and disposition of the collected specimens.
- . Special Conditions: See Attached.

Shelley Matkowski

Issued By

Fish Enhancement Biologist

Title

25-Mar-03

Date of Issue

Signature of Permittee

Permit Number: 03-03

MANITOBA WATER STEWARDSHIP
AQUATIC ECOSYSTEM SECTION

SCIENTIFIC COLLECTION PERMIT

Issued under the authority of the Fisheries Act (Manitoba) and the Fishing Licence Regulation and Fishing Licence Fee Regulation made thereunder.

Issued to: Mr. Jeff Long

of: _____

is hereby authorized to collect, transport and possess fish within the Province of Manitoba subject to the following conditions:

1. Release live fish only in the water from which they were taken.
2. Fish may not be sold, traded or bartered.
3. The use of chemicals and explosives as aids in collecting fish is prohibited.
4. This permit expires on 31-Oct-04 following date of issue.
5. A report must be submitted to Fisheries Branch, Box 40, 200 Saulteaux Crescent, Winnipeg, MB R5J 3W3 upon expiration of this permit indicating: location, species, number and disposition of the collected specimens.
6. Special Conditions: NOTP: Permittee must notify regional fisheries staff and the appropriate district office prior to commencing sampling.

Laureen Janusz

Issued By

Fish Enhancement Biologist

Title

Signature of Permittee

21-May-04

Date of Issue

Permit Number: 04-32

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