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Comité scientifique consultatif des péches canadiennes dans l'Atlantique

## Collected Papers on Fish Habitat with Emphasis on Salmonids

Anadromous, Catadromous and Freshwater Fishes Subcommittee<br>of CAFSAC<br>Department of Fisheries and Oceans<br>P.O. Box 1006<br>Dartmouth, N.S.<br>Canada<br>B2Y 4A2

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FOREWARD

The following compilation of papers was prepared in response to a request for scientific advice on the usefulness of various attributes for measuring productive capacity of fish habitats for evaluating the Department of Fisheries and Oceans's Policy for the Management of Fish Habitat. Specifically, these papers document information presented by invited participants at a special meeting of the Anadromous, Catadromous and Freshwater Fishes Subcommittee of CAFSAC, held April 30-May 5, 1990, in Moncton, New Brunswick. For the Atlantic Zone, Atlantic salmon was chosen as the first species for which changes in the productive capacity of their habitats were to be assessed. The habitat attributes initially identified included: substrate, stream width, cover, velocity, ice scour, stream order, turbidity, total dissolved solids, pH , winter and summer temperatures, discharge and gradient. While much information has been accumulated on these factors, other attributes were also identified, questions raised, and research proposed in order to enhance our current understanding of this topic.

The meeting was divided into four sessions; the first dealt with specific factors affecting productive capacity of fish habitat while the second session addressed various models used in estimating changes in productivity of fish habitat. Session three focused on current and proposed methods to measure factors used to estimate productivity or changes in productivity and session four summarized various techniques used to measure standing stock. Finally, syntheses of these sessions were prepared and have been included in this compilation of papers. Syntheses were prepared on the basis of material presented at the meeting and the ensuing discussions which took place.

Papers are presented in the format submitted by the author(s) and were not altered or reprinted to achieve a consistent presentation for this series.

## Abstract

This publication contains 21 papers dealing with various aspects related to the provision of advice on the usefulness of certain attributes for measuring the productive capacity of fish habitat. In most but not all cases, Atlantic salmon was the primary fish species in question. The first paper summarizes the overall objective of the national Policy for the Management of Fish Habitat and outlines the biological framework for the 'Tier $1^{\prime}$ process to provide a measurement of changes in productivity of fish habitats in Canada. Papers in the first session deal with various factors affecting productive capacity of fish habitat including effects of pH and influence of stream physio-chemical parameters on species associations, the importance of lacustrine habitat to Atlantic salmon production in some areas of Atlantic Canada, and preliminary models examining variation and utilization of fluvial habitat by juvenile Atlantic salmon. The second session deals with habitat suitability curves and other models to estimate changes in productivity of fish habitat. One paper focuses on the application of the instream flow incremental methodology while another introduces the use of nonparametric probability density estimation methods as a tool to provide advice on net gain or loss of productive capacity of fish habitat. Session three concentrates on methods to measure various factors used to estimate productivity or changes in productivity while the last session summarizes techniques to measure standing stock for various types of habitat. Finally, syntheses of the four sessions are provided on the basis of discussions that ensued following the presentation of individual papers.

## Résumé

La présente publication contient vingt et un textes portant sur divers aspects de la formulation de conseils scientifiques sur l'utilité de certaines caractéristiques dans la mesure de la capacité de production de l'habitat du poisson. Dans la plupart mais non dans la totalité - des cas, il est question du saumon de l'Atlantique. Le premier texte résume l'objectif général de la Politique de gestion de l'habitat du poisson et décrit le cadre biologique dans lequel s'inscrit le processus de premier palier destiné à mesurer les changements survenus dans la productivité des habitats du poisson au Canada. Les textes de la première partie du document traitent de divers facteurs qui affectent la capacité de production de l'habitat du poisson, notamment des effets du pHet de l'influence des parametres physio-chimiques des cours d'eau sur les associations d'espèces, de l'importance des habitats lacustres pour la production du saumon de l'Atlantique dans certaines régions du Canada atlantique et de modèles préliminaires illustrant les variations existant dans les habitats fluviaux et l'utilisation de ces derniers par les saumons de l'Atlantique juvéniles. Dans la deuxième partie, il est question des courbes d'appropriation de l'habitat et d'autres modèles permettant d'estimer les changements survenus dans la productivité des habitats. Un des textes porte sur l'application de la Instream Flow Incremental Methodology, tandis qu'un autre présente des méthodes d'estimation non paramétriques de densité de probabilité comme moyen de déterminer les gains nets ou les pertes nettes de capacité de production dans l'habitat du poisson. La troisième partie est axée sur les méthodes de mesure de divers facteurs servant à estimer la productivité ou les changements de productivité de l'habitat du poisson et la dernière partie résume les techniques de mesure des stocks actuels de divers types d'habitat. Finalement, les quatres parties sont résumés en tenant compte des discussions qui ont suivi chaque présentation.

# Atlantic salmon habitat evaluation in the context of the Department of Fisheries and Oceans (DFO) policy for the management of fish habitat 

by

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In October 1986, the Minister of Fisheries and Oceans released a new national Policy for the Management of Fish Habitat. The overall objective of the policy is to increase the level of economic and social benefits that fisheries provide to Canadians. The policy provides a framework and a series of implementation strategies whereby the conservation, restoration and development of fish habitat can contribute towards the achievement of this overall objective.

The question has been raised as to how effective the policy will be in achieving its objectives. The Program Evaluation Branch of DFO was requested to devise an evaluation framework which would detail indicators, methods, and data requirements for measuring the effectiveness of the policy. The Program Evaluation Branch ascertained that a biological framework, which has become the 'Tier 1' process, would provide measurement of changes in productive capacity of fish habitats in Canada. A framework of surrogate indicators, which has become 'Tier 2", would measure the performance of DFO in implementing the policy and managing fish habitat.

The Habitat Management Branch, Department of Fisheries and Oceans has examined the practicability of the Tier 1 approach in two workshops, one on Atlantic salmon and one on Pacific salmon. Both workshops concluded that Tier 1 type evaluations are possible, subject to modifications to make the evaluation suit the biology of the target species. An outline of the Tier 1 approach is given in Figures 1 and 2.

Figure 1 presents a generalized model of the relationship of productive habitat to a fishery. It shows that one or several habitats may be required during the life cycle of a target species. One particular habitat, termed a critical habitat, may function as an ecological bottleneck limiting the production of the species. Other habitats may be unlimited for particular stages of the life history.

Figure 2 is a flow chart describing the steps in executing a Tier 1 evaluation. The advice requested from CAFSAC pertains to Step 3 of the process, on the measurement of physical habitat used by the freshwater phase of the Atlantic salmon.

Step 1 requires that a target species be selected and the critical, or limiting, habitats be defined. A single species approach was adopted because biological species tend to be habitat specific. The life stage limited by the critical habitat should be defined. The estimate of productive capacity should be made for that life stage at the time that it is limited.

Step 2 accounts for geographic variation in productivity of the species, which is usually influenced by climate and the fertility of the ecosystem.

Step 3 is the quantification of the habitat deemed critical to the species.

Step 4 is a means of accounting for environmental impacts on the habitat of the target species.

Step 5 is a bookkeeping exercise to tally the productive capacity of the habitat. This is done at Time 1 to provide a baseline for subsequent comparisons

Step 6 tallies changes in habitat productive capacity at Time 2 and estimates the net effectiveness of application of the policy since the Time 1 estimate.

Step 7 attempts to relate changes in productive capacity to effects on the fishery, which would enable an economic analysis to take place.


Fig. 1. Bio-economic model linking habitat to economic return from a fishery.


Fig. 2. Flow chart illustrating steps in executing a Tier 1 evaluation.

## Session I

Factors Affecting Productive Capacity of Fish Habitat

Factors Affecting the Winter Survival of Juvenile Atlantic Salmon

by<br>R.A. Cunjak<br>Science Branch, Gulf Region<br>Department of Fisheries and Oceans<br>P.O. Box 5030<br>Moncton, New Brunswick<br>E1C 9B6


#### Abstract

An accurate assessment of the factors influencing winter survival is precluded by gaps in our knowledge about the winter biology of juvenile Atlantic salmon, and fishes in general. This paper begins by providing a brief overview of our current understanding of the winter habitat requirements and behaviour of this species. In general, juvenile salmon become photonegative as water temperatures decline in the autumn, move beneath substrate materials (or emigrate to ponds) while continuing to feed throughout the winter (although growth is negligible). Based on information from winter studies of other freshwater fishes, the importance of thermal regime, climatic conditions, habitat availability, ice, fish size, parasite load, and anthropogenic influences are discussed as they may impact on the winter survival of juverile Atlantic salmon.


RESUME
Les lacunes dans nos connaissances sur la biologie du saumon de l'Atlantique juvénile et des poissons en général durant d'hiver empêchent l'évaluation précise des facteurs qui influent sur la survie en hiver. La présente étude porte, en premier, sur un bref aperçu de nos connaissances actuelles sur les exigences en matière d'habitat d'hiver et le comportement de l'espèce. En général, les saumons juvéniles montrent un phototropisme négatif quand la température de l'eau baisse à l'automne, puis s'enfouissent sous les débris du fonds (ou migrent vers des étangs) tout en continuant de se nourrir pendant l'hiver (taux de croissance négligeable). D'après des données recueillies en hiver sur d'autres espèces de poisson dulçaquicoles, l'importance du régime thermique, des conditions climatiques, de la disponibilité d'habitats, de la glace, de la taille des poissons, de la charge en parasites et des facteurs antropogènes est examinée car ces paramètres peuvent influer sur la survie du saumon de l'Atlantique juvénile en hiver.

## INTRODUCTION

A reliable estimation of winter survival of freshwater fishes is precluded by an incomplete understanding of their basic biology during the winter. Those factors identified as affecting winter survival are largely derived from laboratory (or hatchery) experiments (Lindroth 1965; Oliver et al. 1979; Toneys and Coble 1980), inferences derived from environmental models (Chadwick 1982; Gibson and Myers 1988; Myers et al. 1986), or from field data collected in late autumn and or the early spring rather than throughout the winter season per se (Rimmer et al. 1983; Cunjak and Power 1986; Kennedy and Strange 1986; Hillman et al. 1987). Despite the early realization of the importance of many of these. factors (Hubbs and Trautman 1935), their quantification remains imprecise. The present paper attempts to remedy this deficiency by first reviewing the state of our knowledge of the winter behaviour of juvenile Atlantic salmon. This is followed by a listing of those abiotic, biotic, and anthropogenic factors deemed to be most critical to overwinter survival, and their relative importance. This information is based on pertinent data gleaned from the salmonid literature as well as unpublished data from the author's own research on the topic.

## WINTER BEHAVIOUR AND HABITAT:

In running water environments throughout most of the distributional range of Atlantic salmon, juveniles display an autumnal shift in microhabitat choice and behaviour. Coincident with a decline in mean daily water temperatures below $8^{\circ}-10^{\circ} \mathrm{C}$, young salmon move beneath suitably-sized stones in riffle-run habitats where they overwinter (Gibson 1966; Smirnov et al. 1976; Rimmer et al. 1984). "Home stone" diameters range between $8-40 \mathrm{~cm}$ and are positively correlated with salmon age (Rimmer et al. 1984; Cunjak 1988a). Unlike the situation during summer when they are strongly photopositive, young salmon in winter are photonegative, leaving their shelters only during the night (or at dawn/dusk) to feed (Cunjak 1988a). Limited evidence suggests that this generalized pattern of winter behaviour continues until mean water temperatures increase above $5^{\circ} \mathrm{C}-7^{\circ} \mathrm{C}$ in the spring (Cunjak 1988a).

There are, of course, exceptions to these generalizations. At the northern end of their distributional range, such as in subarctic rivers (e.g. Ungava), young Atlantic salmon remain active above the substrate in the autumn until water temperatures decrease below approximately $5^{\circ} \mathrm{C}$ (Cunjak, unpubl. data). It is also likely that many northern salmon fast during winter (e.g. Smirnov et al. 1976) such as is the case with other salmonids in the North (Dutil 1982; Boivin 1987). Emergence from winter shelters in the spring is probably also synchronized with a lower thermal regime than is found in southern rivers. In those river systems where lacustrine habitats are common (e.g. Newfoundland) and/or suitable substrates for overwintering are locally unavailable, some young salmon move to deep pools or ponds (Saunders and Gee 1964; Pepper 1976; Hutchings 1986).

FACTORS AFFECTING SURVIVAL:

1. Thermal and Hydrologic Regime of the Aquatic Environment.

Streams which receive much of their flow from groundwater discharge (as
compared with surface flows) tend to have more stable annual thermal and discharge regimes. Also, ice formation (with its negative effects - see below) is precluded, or at least minimized because of the relatively warmer winter water temperatures. This homeothermic characteristic was attributed as being the main reason for better increasing winter survival in populations of brown trout (Edwards et al. 1979) and brook trout (Hunt 1969). Winter research with stream salmonids has shown that groundwater discharge areas are favoured for overwintering (Cunjak and Power 1986), spawning (Benson 1953; Hansen 1975; Cunjak et al. 1986), and incubation of developing alevins (Sowden and Power 1985). Physiological studies on stream-dwelling brook trout, Salvelinus fontinalis, brown trout, Salmo trutta (Cunjak 1988b), and Atlantic salmon, Salmo salar (Gardiner and Geddes 1980) have suggested that it is the acclimatization to rapidly changing water temperatures in early (and late) winter which is metabolically more costly than the duration of freezing temperatures over the winter. Additional factors which stress fish at this time of year may further deplete critical energy reserves and, thereby, affect winter survival.

## 2. Local Climatic Conditions.

Alternating freeze-thaw events during the winter may necessitate a relatively high degree of activity by stream fishes (Reimers 1963) in order to avoid displacement by high discharges and freezing from ice while, at the same time, securing suitable food and habitat. Those salmon which have not accumulated sufficient energy reserves to meet the metabolic demands of overwintering in such unstable streams will not survive. Rain-on-snow floods have been associated with high mortalities of juvenile stream salmonids, especially where the flows are constrained by snow banks (Erman et al. 1988). In boreal and subarctic streams, conditions are more stable in that, after initial freeze-up, water discharge fluctuations are minimal and a continuous ice cover is maintained until spring break-up (e.g. Power 1969).

Extremely low flows during the winter are potential problems for winter survival. The redds of salmonids can become dewatered, resulting in freezing of the developing eggs and alevins (Reiser and Wesche 1979; Butler 1982) especially where in-substrate humidity is reduced (Neitzel and Becker 1988). Research at Indian River in Newfoundland found annual egg-to-fry survival to range between $22 \%$ and $67 \%$ and was significantly correlated with winter temperature and discharge (Chadwick 1982). Based on these data, Chadwick (1982) suggested that freezing of redds (during a particularly cold winter with low discharge) was the reason for low egg to smolt survival at Western Arm Brook, Nfld. Power (1969) similarly suggested that egg mortality might occur from redd freezing in the salmon rivers of Ungava Bay.

## 3. Habitat Availability.

In winter, habitat is often considered the critical limiting factor to salmonid production in streams, more important than food (Chapman 1966). In streams with low winter streamflows and freezing conditions, habitat availability can be markedly reduced. Calkins and Brockett (1988) noted that $100 \%$ of the bed of an Atlantic salmon river in New Hampshire was blanketed with anchor ice prior to the formation of a continuous surface ice sheet. Subsequent research by Calkins (1989) in New England salmon rivers found that ice could occupy $60-80 \%$ of the cross-sectional area of a stream. Under these circumstances, young salmon would be required to move to more suitable areas
and in so doing, expend critical energy reserves. In arctic latitudes, entire lengths of streams can freeze with the result that resident fishes (including salmonids) move to overwinter in relatively restricted sections of groundwater upwelling (Craig and Poulin 1975) or to estuaries and lakes (Bendock 1981; Craig 1984). Preliminary data from Maritime rivers, indicate that winter movements by juvenile salmon are common. The extent of emigration (and possibly winter mortality) appears to be related to the availability of suitable winter habitat. In relatively stable, cobble-bottomed streams, 16\%21\% of the salmon captured in December were found again in April compared with only $2 \%$ in a stream where suitable winter habitat was scarce (R.A. Cunjak, unpubl. data). These values are subject to further modification because microhabitat requirements for overwintering vary with fish size and with lifestage.

It should be noted that alternative winter habitats may exist for young salmon, the biological significance of which is still not fully appreciated. For example, recent evidence from Newfoundland indicates that some Atlantic salmon parr overwinter in estuaries (J.B. Dempson, D.F.O., St. John's, pers. comm.). Winter research in British Columbia has shown that various salmonid species successfully overwinter in ponds (Peterson 1982) and inundated valley bottoms (Brown and Hartman 1988).
4. Ice.

The impacts of ice (surface, anchor, and frazil forms) on stream fishes is probably the least understood aspect of winter biology. Substrate scouring, de-watering of stream sections, and the freezing of redds are the most common causes of mortality and are largely attributable to subsurface ice (Maciolek and Needham 1952; Needham and Jones 1959; Reiser and Wesche 1979). Anoxic (or very low dissolved oxygen) conditions in rivers and ponds beneath ice cover have been documented from geographically distant regions (e.g. Hynes 1970; Ranjie and Huimin 1987) but appear to be most prevalent in higher latitudes (Shreier et al. 1980; Bendock 1981) where ice cover persists for longer periods of time. Late winter depressions in dissolved oxygen can be exacerbated due to anoxic groundwater intrusions and can reach concentrations which are lethal to young salmonids (Adams and Cannon 1987; Schmidt et al. 1989).

The most serious impact to stream fishes from surface ice occurs during ice break-up. Calkins (1989) and Calkins et al. (1989) have measured transient flow velocities in salmon rivers which are $>5 \mathrm{~m} / \mathrm{s}$ and capable of moving 70 kg boulders. These strong current velocities could, therefore, dislodge shelters and injure stream fishes such as dace, cottids, and salmonids which overwinter within the substrate. Further, flooding and ice-dams can result in displacement and mass strandings of riverine species which eventually suffocate as the water levels recede.
5. Biological Factors.

Large body size (with its concomitant large store of energy reserves) by the end of the fish' first autumn, is considered an important factor influencing winter survival in many species (Lindroth 1965; Mason 1976; Oliver et al. 1979; Toneys and Coble 1979; Johnson et al. 1987; Post and Evans 1989). A hatchery study by Lindroth (1965) of underyearling Atlantic salmon showed that the smallest individuals suffered the greatest mortality rate (2\%-6\%).

Lindroth believed that size position within a year-class was more important than absolute size per se, a contention which is supported by the work of Reimers (1963) and Toneys and Coble (1980).

Spawning depletes energy stores which could otherwise be used for overwintering. A study of Arctic char in the N.W.T. (Dutil 1982) indicated that post-spawnwers had $35-46 \%$ less available energy than immatures by the end of the winter. The acknowledged high rate of mortality among precocious male Atlantic salmon parr over the winter (Mitans 1973) has been attributed to insufficient energy reserves having been accumulated after spawning and particularly if attempting to smoltify the following spring (Dalley et al. 1983). This phenomenon led Myers et al. (1986) to suggest a size threshold of $70-72 \mathrm{~mm}$ FL below which Atlantic salmon cannot mature and subsequently overwinter. Gardiner and Geddes (1980), in studying changes in body composition during winter in a Scottish stream, showed that nutritional insufficiency was an important factor in winter survival of underyearling Atlantic salmon. They suggested that a body water content of $>82.5 \%$ was probably an indicator of insufficient energy reserves. The fact that body weight remained relatively constant over winter (despite a large decrease in energy content) suggests that determinations of weight alone may be misleading as indicators of condition.

Food is unlikely to be a limiting factor to salmon survival in winter for two reasons. First, benthic biomass is often high at this time of year (Hynes 1970, especially for insects); and second, low water temperatures greatly reduce metabolic rates and hence, the demand for food. However, competition for space (intra- or inter-specific) has been suggested as a factor in winter survival where habitats overlap and are in short supply (Chapman 1966), in both lakes and in running waters. Cunjak and Power (1987) and Cunjak et al. (1987) have demonstrated, for wild brook and brown trout, that the reduced assimilation efficiency and slow gut evacuation rates at low winter temperatures combine to make food resources somewhat ineffective in meeting high metabolic demands of cold acclimatization, such as occurs in the early winter. This situation can severely deplete energy reserves (Cunjak 1988b) with still a great part of the winter season remaining for these salmonids to survive.

Nutritional insufficiencies and parasite loads can act as stressors which debilitate young salmon and ultimately result in winter mortality. A recent study (Cunjak and McGladdery 1990) on the effect of glochidial parasitism on the gills of overwintering juvenile Atlantic salmon in a Nova Scotian river showed that heavy parasite loads were contributing to the poorer condition and smaller size in a localized population of salmon parr. Both these factors have been identified as causes of winter mortality in juvenile salmon (see above). Winter temperatures are not lethal to juvenile salmonids because their plasma freezing points are less than the freezing point of freshwater. Potentially lethal situations develop only if fish come in contact with ice or saltwater (with its lower freezing point).

## 6. Anthropogenic Factors.

The activities of man in and around river systems can have profound impacts on the winter survival of Atlantic salmon although specific studies on the phenomenon are unknown. Agriculture and forestry practices can increase sediment loads to streams such that the substrate becomes compacted with fines. Such a situation reduces access beneath stones by those fishes which
usually overwinter in the substrate (see Hillman et al. 1987; Cunjak 1988a) and also restricts water flow (and oxygen) into redds (Sowden and Power 1985). Recent research has shown that clear-cut sections of streams are subject to more severe winter flooding (Erman et al. 1988) and loss of instream cover and pools (Tschaplinski and Hartman 1983; Heifetz et al. 1986; Brown and McMahon 1988) which are important habitat requirements for stream salmonids. Hydroelectric projects and water withdrawal practices (e.g. snow-making for alpine ski operations) can change water levels and discharge regimes in a manner which is detrimental to overwintering fishes. Pollution, where it leads to eutrophication, can accelerate deoxygenation, particularly beneath ice cover (e.g. Lasko 1987).

## SUMMARY:

Based on a rather general appreciation of winter behaviour and the limited available scientific evidence, the following list of factors having the greatest effect on winter survival includes: body size, the amount of energy reserves (which is often a function of body size), availability of suitable microhabitats for overwintering, and the severity of the physical stream environment (e.g. ice, river discharge, duration of freezing water temperatures).

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# Spatial and Temporal Distributions of Salmonids in Junction Pond, Northeast River, Placentia and Conne Pond, Conne River, Newfoundland 

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#### Abstract

The spatial and temporal distributions of salmonids were examined in Junction Pond, Northeast River, Placentia, and in Conne Pond, Conne River, Newfoundland, using Lundgren multiple mesh experimental gillnets. Daytime and overnight sets of benthic and pelagic nets were conducted once per month from June to October for Conne Pond and from June to November for Junction Pond, in 1987. In addition, sampling was carried out in the winter of 1989 for Junction Pond. Atlantic salmon and brook trout were found in both ponds while Arctic charr and brown trout were encountered only in Junction Pond. Most captures of Atlantic salmon parr occurred in the benthic zone in both ponds and of these most were taken in the shoreline littoral areas. Most brook trout and brown trout were also taken in the benthic zone but there were fewer occurrences in littoral areas relative to Atlantic salmon parr. Arctic charr were more pelagic than the other species and when present in the benthic zone they mainly occurred at depths beyond the littoral zone. Atlantic salmon parr captured in the pelagic zone and taken from benthic depths beyond the littoral zone were significantly greater in length than those from the littoral zone in both ponds. There was no significant difference in size between benthic and pelagic zones for brook trout in Junction Pond, but in Conne Pond, pelagic specimens were larger than benthic specimens; in the benthic zone, deeper benthic specimens were larger than littoral specimens in both ponds. Older Atlantic salmon parr tended to occupy the pelagic and deeper benthic zones than found in the littoral zone in junction Pond while in Conne Pond there were no differences. Brook trout in the pelagic zone were significantly older than in the benthic zone in Conne Pond but not in Junction Pond; in the benthic zone, deeper benthic trout were significantly older than littoral trout in both ponds.


## Résumé

A l'aide de filets expérimentaux Lundgren à maillages multiples, on a déterminé la distribution spatiale et temporelle des salmonidés dans l'étang Junction de la rivière Northeast, Placentia et dans l'étang Conne de la rivière Conne (Terre-Neuve). Les filets ont été mouillés pour la durée du jour et la durée de la nuit, près du fond et entre deux eaux, une fois par mois de juin à octobre 1987 dans le cas l'étang Conne, et de juin à novembre dans le cas de l'étang Junction. De plus, on a effectué un échantillonnage de 1'étang Junction au cours de l'hiver 1989. Des saumons de l'Atlantique et des ombles de fontaine étaient présents dans les deux étangs, mais seuls des ombles chevaliers et des truites brunes peuplaient l'étang Junction. La plupart des tacons de saumon de l'Atlantique ont été capturés près du fond dans les deux étangs, surtout dans la zone littorale. La plupart des truites brunes et des ombles de fontaine ont aussi été capturés près du fond, mais ces espèces étaient moins communes dans la zone littorale par rapport aux tacons du saumon de l'Atlantique. L'omble chevalier montrait une plus grande préférence pour la pleine eau que les autres espèces et lorsqu'il était présent près du fond, il fréquentait surtout des profondeurs au delà de la zone littorale. Les tacons de saumon de l'Atlantique capturés dans la zone pélagique et dans les eaux benthiques au delà de la zone littorale étaient nettement plus longs que ceux capturés dans la zone littorale des deux étangs. Dans le cas de l'omble de fontaine de l'étang Junction, aucune différence significative de la longueur n'a été notée entre les zones benthique et pélagique; par contre, dans l'étang Conne, les ombles pélagiques étaient nettement plus longues que les ombles benthiques. De plus, dans la zone benthique, les individus qui fréquentaient les eaux profondes étaient plus longs que les individus qui fréquentaient la zone littorale des deux étangs. Dans l'étang Junction, les tacons de saumon de l'Atlantique qui avaient tendance à fréquenter les zones pélagiques et les zones benthiques profondes étaient plus gros que ceux de la zone littorale. Par contre, aucune différence n'a été notée dans l'étang Conne. Les ombles de fontaine de la zone pélagique étaient nettement plus âgés que ceux de la zone benthique dans l'étang Conne mais non dans l'étang Junction; dans la zone benthique, les ombles fréquentant les eaux profondes étaient nettement plus âgés que ceux de la zone littorale dans le cas des deux étangs.

## Introduction

While there have been several studies estimating population numbers and biomass of juvenile anadromous Atlantic salmon, Salmo salar, (Pepper 1976; Chadwick and Green 1985; Pepper et al. 1984, 1985; Ryan 1986; $0^{\prime}$ Connell and Ash 1989), Brook trout, Salvelinus fontinalis, (Wiseman 1970, 1971, 1972; Ryan 1984, 1986), and Brown trout, Salmo trutta, (Wiseman 1972) in lacustrine habitat in insular Newfoundland, there have been no definitive studies attempting to partition standing stock according to the relative contribution of different lentic areas.

A first step in such a determination is to examine the spatial and temporal usage of different lacustrine habitats by salmonids for rearing purposes. In this paper we present the results of such a study that was undertaken in two ponds with different salmonid species composition, located on two different river systems, in insular Newfoundland. The study utilized Lundgren multiple mesh experimental gillnets.

## Description of Study Areas

The study was conducted in Junction Pond, Northeast River, Placentia (Fig. 1) and Conne Pond, Conne River (Fig. 2), Newfoundland. Northeast River is located on the Avalon Peninsula and flows into Placentia Bay. Conne River flows into Fortune Bay on the south coast of Newfoundland.

Northeast River, with a drainage area of $93.8 \mathrm{~km}^{2}$, is underlain almost entirely by Precambrian rock. The drainage basin of Conne River ( $602.4 \mathrm{~km}^{2}$ ) consists of about equal amounts of Ordovician sedimentary and acidic intrusive rocks. Soils are mainly acidic in both areas (Roberts 1983). Vegetation in both areas consists of Kalmia angustifolia barrens with the dominant tree being Abies balsamea; ombrogenous slope bogs, shallow oligotrophic bogs and fens, are the dominant peatlands (Damman 1983).

Lakes form a substantial proportion of the total amount of salmonid habitat available in both systems. Northeast River possesses 1,072 ha of lacustrine habitat and $1.35 \times 10^{5} \mathrm{~m}^{2}$ of fluyial habitat. Corresponding values for Conne River are 4,620 ha and $1.32 \times 10^{6} \mathrm{~m}^{2}$. The ratio of lacustrine habitat to fluvial habitat (both converted to $\mathrm{m}^{2}$ ) for Northeast River is 79.3 compared to 35.1 for Conne River. Shoreline substrate is predominantly boulder and rubble in both ponds.

The morphometry of Junction Pond and Conne Pond is presented in Table 1. Bathymetric maps are provided in Figs. 3 and 4 and percentage hypsographic curves in Figs. 5. Water chemistry (determined from a single sample taken in July 1986 for each pond) is presented in Table 2. Water colour information is not available for Junction Pond; however, apparent colour values previously recorded for a pond located downstream ranged from 20 to 50 units with a mean of 34 units ( $0^{\prime}$ Connell and Andrews 1987). This compares with a true colour value of 70 units for Conne Pond presented in Table 2.

Fish species present in both Northeast River and Conne River include Atlantic salmon, brook trout, American smelt, Osmerus mordax, American eel, Anguilla rostrata, and threespine stickleback, Gasterosteus aculeatus. Species found in Northeast River but not Conne River include brown trout, and Arctic charr, Salvelinus alpinus, while alewife, Alosa pseudoharengus, is also found in Conne River but not in Northeast River.

## Materials and Methods

The location of the sampling area in each pond is shown in Figs. 3 and 4. Samples were taken with Lundgren multiple mesh experimental gillnets. These nets and the manner in which they were set were the same as described for Newfoundland applications by Hammar and Filipsson (1985). In Junction Pond, benthic nets were set along the $1,2,3,5,7,9$, and 11 m contours; in Conne Pond, benthic nets (each 1.5 m in height) were set along the $1,2,3$, 5 , and 7 m contours. Although the maximum depth of Conne Pond is 10 m , the shape and slope of the basin at depths beyond 7 m is not conducive to setting benthic nets. A pelagic net was set over the deepest area of each pond. This net extended from the surface to a depth of 6 m .

Samples were collected diurnally once per month in both ponds. The sampling period extended from June to November 1987 in Junction Pond and from June to October 1987 in Conne Pond. Sampling was conducted only during the night for Junction Pond in November. Actual sampling dates for each month will be shown in the presentation of results. For Junction Pond, nets for night sampling were set as close to darkness as possible and hauled just after dawn. Because of the large number of specimens encountered in night sets and the time required for their removal from the nets, day sets were not conducted until the following day. For day sampling, nets were set just after dawn and hauled just prior to darkness. For Conne Pond, the numbers of specimens caught allowed day sampling to proceed on the same day that nets set during the previous night were hauled. From August to September, low catches necessitated two consecutive night and day sets in Conne Pond.

During the period February 28 - March 1, 1989, sampling was conducted under the ice in Junction Pond. Benthic nets were set along the 2, 5, and 10 m contours. A pelagic net was also set in the same manner as described above for open water sampling.

The benthic zone in each pond was divided into littoral and nonlittoral (sublittoral plus profundal) zones on the basis of average Secchi disc depth. Canfield et al. (1985) and Chambers and Kalff (1985) reported significant positive predictive relationships between Secchi disc depth (in the range of $0-5 \mathrm{~m}$ ) and the extent of macrophytic vegetation within lakes. The average Secchi disc depth recorded during sampling in Junction Pond was 3 m . Hence, specimens captured at depths of $1-3 \mathrm{~m}$ were allocated into the littoral zone and those captured at greater depths were assigned to the nonlittoral zone. In Conne Pond, the average Secchi disc depth was 2 m and allocations into the littoral and nonlittoral zones were made accordingly. In the pelagic zone, fish were assigned to the trophogenic and tropholytic zones according to the same average Secchi disc depths.

Tests of independence of temporal versus spatial occurrences of each species were carried out utilizing contingency table analysis and the loglikelihood ratio test (G-test) incorporating Williams correction as recommended by Sokal and Rohlf (1981). Only test values conforming to criteria respecting expected frequencies less than 5 as outlined by Sokal and Rohlf (1981) were accepted. Spatial categories were those presented above (i.e. benthic, pelagic, littoral, etc.). Temporal variables were night and day and season (summer, June-August; fall, September-November). The Wilcoxon two-sample test (Z) was used for statistical comparisons involving fork length and age. The level of significance for all statistical tests was 0.05 .

## Results

FREQUENCY OF OCCURRENCE BY LENTIC ZONE

Benthic Zone versus Pelagic Zone

## Junction Pond

The percentage frequency of capture of Atlantic salmon parr in the benthic and pelagic zones (all depths combined in each category) during night and day sets for each month are shown in Fig. 6. For both night and day sets, parr were found in benthic nets in all months. Pelagic captures of parr during the night occurred only in June and July, with the highest percentage being taken in July. For day sets, pelagic captures occurred in June-August. The distribution of parr into the benthic zone versus the pelagic zone in summer was not independent of diurnal sampling period ( $G=12.532 ; \mathrm{df}=1 ; \mathrm{P}=0.000$ ). The distribution of parr into benthic and pelagic zones was also not independent of season for both night ( $G=62.305$; $\mathrm{df}=1 ; \mathrm{P}=0.000$ ) and day $(\mathrm{G}=6.554 ; \mathrm{df}=1$; $\mathrm{P}=0.010$ ).

Overall numbers of brook trout taken during night and day in both zones was lower than observed for Atlantic salmon (Fig. 6). Nighttime benthic captures occurred in all months except August and September. Daytime benthic captures occurred in all months with only a single specimen being taken in August and September. Pelagically, brook trout were caught during the night in July only and during the day in June and July. Lentic distribution during the night was not independent of season ( $G=5.496 ; \mathrm{df}=1$; $\mathrm{P}=0.019$ ).

Pelagic captures of Arctic Charr during the summer were higher in number than benthic captures, which was the reverse of what was observed for the other species (Fig. 6). In the fall, however, more specimens were caught in the benthic than in the pelagic zone. The distribution of Arctic charr by lentic zone was independent of diurnal sampling period in summer ( $G=1.767$; $\mathrm{df}=1$; P $=0.184$ ) but not independent of season during night ( $G=16.852$; $\mathrm{df}=1$; $P=$ 0.000 ).

Brown trout was the less frequently encountered species with a total of only 15 specimens being taken throughout the entire sampling period. of these, 12 were caught in benthic nets with the remainder in the pelagic net. Daytime
benthic captures ( $\mathrm{N}=8$ ) occurred in June, July, September, and October. The remaining benthic fish ( $N=4$ ) were taken at night in July and October. The 3 pelagic specimens were captured in July ( 2 during the night and 1 during the day).

## Conne Pond

The percentage frequency of capture of Atlantic salmon parr in benthic and pelagic zones (all depths combined in each category) during night and day sets for each month are illustrated in Fig. 7. As observed in Junction Pond, salmon parr were found in benthic nets in all months. Similarly, pelagic captures of parr occurred only during the summer (June-August) and was higher during the night sets than in the day sets. For Conne Pond, the distribution of salmon parr into the lentic zones in summer (benthic vs. pelagic) also was not independent of diurnal sampling period ( $G=8.194$; $\mathrm{df}=1$; $\mathrm{P}=0.004$ ). The distribution of parr into benthic and pelagic zones was not independent of season for night in Conne Pond ( $G=9.850$; $\mathrm{df}=1 ; \mathrm{P}=0.002$ ) but was independent of season for day sets $(G=1.893 ; \mathrm{df}=1$; $\mathrm{P}=0.169$ ).

In general, total numbers of brook trout caught in Conne Pond were similar to Atlantic salmon parr (Fig. 7). Brook trout were caught in benthic nets during all months. In contrast with Junction Pond, brook trout were also found in pelagic nets in all months except for the September day set sample (Fig. 7), but were generally less frequent in the pelagic nets in comparison with benthic nets. Results of contingency analyses indicated that the distribution of brook trout into benthic and pelagic zones in summer was independent of diurnal sampling period ( $G=1.853$; $\mathrm{df}=1 ; \mathrm{P}=0.173$ ). As well, the distribution of brook trout into benthic and pelagic zones was independent of season for night sets ( $G=0.093$; $\mathrm{df}=1 ; \mathrm{P}=0.760$ ) but was not independent for day sets ( $G=$ 4.320; $\mathrm{df}=1$; $\mathrm{P}=0.038$ ).

Benthic Zone - Littoral versus Nonlittoral

## Junction Pond

The percentage distribution of Atlantic salmon parr captured during night and day in benthic nets by individual contour depth each month is shown in Fig. 8. Individual contour depths are summarized in terms of littoral and nonlittoral zones in Fig. 9. For the night period it is evident that with exception of July and October, most parr were taken in the littoral zone; during the day, there was a tendency for more parr to be taken in the nonlittoral zone involving three months (July, August, and October) out of five. The distribution of parr into the littoral versus the nonlittoral zone was not independent of diurnal sampling period in both summer ( $G=5.404$; $\mathrm{df}=1$; $\mathrm{P}=$ 0.020 ) and fall ( $G=6.354 ; \mathrm{df}=1 ; \mathrm{P}=0.012$ ). However, distribution into these lentic areas was independent of season during both night ( $G=1.742 ; \mathrm{df}=1 ; \mathrm{P}=0.187$ ) and day $(\mathrm{G}=2.829 ; \mathrm{df}=1 ; \mathrm{P}=0.093$ ).

The percentage diurnal distribution of brook trout for each individual contour depth is shown for each month in Fig. 10 and summarized in terms of littoral and nonlittoral zones in Fig. 9. During the night there was a tendency for most brook trout to be taken in the nonlittoral zone which was in contrast to Atlantic salmon parr. During the day except in each of August and September when only a single specimen was taken in the littoral zone, the same tendency more or less applied. The distribution of brook trout by lentic zone was independent of diurnal sampling period in both summer ( $G=3.228$; $\mathrm{df}=1$; $\mathrm{P}=$ 0.072 ) and fall ( $\mathrm{G}=0.045$; $\mathrm{df}=1 ; \mathrm{P}=0.831$ ). Lentic zone distribution was not independent of season for the night period ( $G=7.881$; $\mathrm{df}=1$; $\mathrm{P}=0.005$ ) but was independent for the day period ( $G=0.005$; $\mathrm{df}=1 \mathrm{P}=0.945$ ).

Table 3 shows the diurnal distribution (numbers of fish) of Arctic charr captured in benthic nets by month. All captures except one occurred in water with a depth equal to or greater than 3 m . In terms of littoral and nonlittoral zones (Fig. 9), as expected most specimens were found in the nonlittoral zone. The lentic distribution of Arctic charr was independent of season during the day ( $\mathrm{G}=0.151$; $\mathrm{df}=1$; $\mathrm{P}=0.698$ ).

Of the 12 brown trout captured in benthic nets, 8 were taken in the nonlittoral zone and were spread over all depths. The remaining 4 were spread over all depths in the littoral zone.

## Conne Pond

The percentage distribution by individual contour depth of Atlantic salmon parr captured during night and day in benthic nets each month is illustrated in Fig. 11. Individual contour depths are summarized in terms of littoral and nonlittoral zones in Fig. 12. As observed in the Junction Pond night period samples, most of the Conne Pond salmon parr were also caught in the littoral zone. This was also the case for day period in Conne Pond which is in contrast to Junction Pond. Also in contrast to Junction Pond, the distribution of parr into the littoral versus the nonlittoral zone was independent of diurnal sampling period in summer ( $G=0.770 ; \mathrm{df}=1 ; \mathrm{P}=0.038$ ). Similar to Junction Pond, distribution into these lentic areas was independent of season during the day period ( $G=0.055$; $\mathrm{df}=1$; $\mathrm{P}=0.815$ ).

The percentage diurnal distribution of brook trout for each individual contour depth is shown for each month in Fig. 13 and summarized in terms of littoral and nonlittoral zones in Fig. 12. During the night there was a tendency for most brook trout to be taken in the littoral zone. this is in contrast to that observed in Junction Pond (Fig. 9). However, during the day period the opposite was true for Conne fond when most of the trout were caught in the nonlittoral zone (fig. 12). the distribution of brook trout by lentic zone was not independent of diurnal sampling period during the summer ( $g=$ 14.030; $\mathrm{df}=1 ; \mathrm{p}=0.000$ ). lentic zone distribution was independent of season for the night period ( $g=2.963$; $\mathrm{df}=1$; $\mathrm{p}=0.085$ ) and the day period ( $\mathrm{g}=$ 2.501; df = 1; $p=0.114$ ).

## Pelagic - Trophogenic versus Tropholytic

## Junction Pond

The diurnal distribution of Atlantic salmon parr captured in pelagic net sets (numbers of fish) by depth interval and by trophogenic and tropholytic zones each month is presented in Table 4. Most parr were taken in the tropholytic zone in both night and day. More captures occurred during the night than during day. The summer distribution of parr into the trophogenic zone versus the tropholytic zone was not independent of diurnal sampling period ( $\mathrm{G}=$ 4.283; $\mathrm{df}=1$; $\mathrm{P}=0.038$ ).

In June, most captures of brook trout occurred in the trophogenic zone (Table 4). In July the reverse occurred. The overall number of specimens involved however was low.

For Arctic charr (Table 4), all specimens in June came from the trophogenic zone. In July, equal numbers were found in each zone during night but during day specimens were confined to the tropholytic zone.

Of the 3 brown trout captured in July, 1 was found in each zone during the night and the remaining specimen was found in the tropholytic zone in the day.

## Conne Pond

The diurnal distribution of Atlantic salmon parr captured in pelagic net sets (numbers of fish) by depth interval and by trophogenic and tropholytic zones each month is presented in Table 5. With respect to the night period, virtually all parr were caught in the trophogenic zone while during the day period the opposite occurred where most of the parr were taken in the tropholytic zone. As also found in Junction Pond, the summer distribution of parr into the trophogenic zone versus the tropholytic zone in Conne Pond was not independent of diurnal sampling period ( $G=16.622$; , $\mathrm{df}=1$; $\mathrm{P}=0.000$ ).

Over all months the majority of the brook trout were caught in the tropholytic zone regardless of the period of day (Table 5). The summer distribution of brook trout into the trophogenic and tropholytic zones was independent of diurnal sampling period ( $G=0.627$; $\mathrm{df}=1$; $\mathrm{P}=0.429$ ).

LENGTH DISTRIBUTION BY LENTIC ZONE

Benthic Zone versus Pelagic Zone

## Junction Pond

Table 6 shows length frequency distributions (percent) of Atlantic salmon parr captured in benthic and pelagic nets (all depths combined in each category) in summer and fall and for seasons combined. Data for night and day were
combined. In summer the range in size of parr captured in benthic nets was greater than in pelagic nets. The upper limit of the range was the same for both lentic areas; the smallest fish were taken in the benthic net. The range in length encountered in benthic nets in the fall was similar to that of summer. Overall length of parr taken in pelagic nets in summer was significantly higher than in benthic nets (Table 6). In terms of seasons combined, mean length of pelagic fish was also significantly higher than mean length of benthic fish.

The range in size of brook trout captured in pelagic nets in summer was much smaller than observed for those taken in benthic nets (Table 7). However, sample size for the pelagic zone was much smaller than for the benthic zone. The range in size of benthic specimens in the fall was not as great as in summer. There was no significant difference in length between benthic and pelagic zones in summer nor was there a significant difference for comparisons with seasons combined.

The range in size of Arctic charr in benthic nets in summer was similar to that of pelagic nets (Table 8). In fall the range for benthic nets was greater than in summer. Length was not significantly different between lentic zones for seasons separately and combined.

Brown trout captured in benthic nets ranged in size from 12 to 25 cm in summer ( $\mathrm{N}=3$ ) and from 21 to 31 cm in fall ( $\mathrm{N}=6$ ). Two fish in the pelagic zone were 18 and 20 cm in length.

## Conne Pond

Table 9 summarizes the length frequency distributions (percent) of Atlantic salmon parr captured in benthic and pelagic experimental gill nets (depths combined in each category) during summer and fall periods and seasons combined. Data for night and day were combined. There was a greater range of sizes caught during the summer in the benthic nets in comparison with the pelagic nets. During the fall, the range in length was also less than during the summer for benthic nets. As observed in Junction Pond, overall length of parr caught during the summer in pelagic nets was significantly higher than in the benthic nets (Table 9). This was consistent when the fall benthic samples were combined for the total analysis. Conne Pond parr were smaller overall for both benthic and pelagic samples in comparison with Junction Pond.

Similar patterns in the size distribution observed for salmon were found in brook trout (Table 10). Benthic net samples were characterized with a greater range in sizes than pelagic nets during the summer (only 3 trout were available for length analysis from fall samples). In contrast with Junction Pond, trout caught in the pelagic nets were significantly larger than those captured in the benthic nets for summer samples and when seasons were combined (Table 10).

## Benthic Zone - Littoral versus Nonlittoral

## Junction Pond

The range in size of Atlantic salmon parr in the littoral zone in summer and fall was similar and greater than that observed for parr in the nonlittoral zone for each season (Table 11). In summer, compared to the littoral zone, the upper limit of the range for the nonlittoral zone was similar but the smallest sizes were not represented. In the fall, the range for the nonlittoral zone was narrower than that of the littoral zone and sizes at the extremities of the distribution were not present for the former zone. The length of parr captured in the nonlittoral zone was significantly higher than for the littoral zone for seasons separately and combined (Table 11).

The overall range in size of brook trout in the nonlittoral zone was somewhat less than for the lentic zone in summer (Table 12). The range for both lentic zones was similar in the fall but narrower than observed for the summer. Length for specimens taken in the fall was lower than in summer in both lentic zones. There was no significant difference in length between zones in either summer or fall; however for seasons combined, length for the nonlittoral zone was significantly higher.

The greatest range in size of Arctic charr was encountered in both seasons in the nonlittoral zone (Table 13). The few specimens taken in the littoral zone fitted into the upper end of the distribution exhibited by nonlittoral zone Arctic charr. Length for specimens in the littoral zone was significantly higher than in the nonlittoral zone in the fall. For seasons combined, there was no significant difference in length between zones.

Two specimens of brown trout in the littoral zone in the fall were 17 and 31 cm in length. Specimens captured in the nonlittoral zone in summer were $12-25 \mathrm{~cm}$ in length ( $\mathrm{N}=3$ ); those captured in the fall were $17-27 \mathrm{~cm}$ in length ( $\mathrm{N}=4$ ).

## Conne Pond

Atlantic salmon parr caught in the nonlittoral zone were significantly larger than those parr captured in the littoral zone for both the summer sampling period, and when seasons were combined (Table 14). Salmon parr caught in both zones, however, consisted of small and large individuals. Few fish were available in the fall in the nonlittoral zone but these were characterized by larger sized parr. As expected, fall samples tended to be larger than summer samples considering the growth that would have occurred over the summer.

Again, brook trout followed the same general pattern with fish caught in the nonlittoral zone significantly larger than trout caught in the littoral zone for both the summer sampling period and when seasons were combined (Table 15). Ranges in size of trout caught were similar for both zones although the smallest trout was caught in the littoral zone. In Junction Pond the smallest trout were caught in the nonlittoral zone (Table 12).

## Pelagic Zone - Trophogenic versus Tropholytic

## Junction Pond

Size distribution ranges of Atlantic salmon parr in both the trophogenic and tropholytic zones were similar (Table 16). There was no significant difference in length between zones.

Of the brook trout encountered in the pelagic zone in summer, those found in the trophogenic zone were $19-22 \mathrm{~cm}$ in length ( $N=3$ ) and those in the in the tropholytic zone were $19-26 \mathrm{~cm}(\mathrm{~N}=3)$.

A single specimen of Arctic charr in the trophogenic zone in fall was the smallest encountered in the pelagic zone (Table 17). The range in size was greatest in the tropholytic zone in summer. There was no significant difference in length between zones.

Length values were available for 2 brown trout and they were 18 (trophogenic zone) and 20 (tropholytic zone) cm.

## Conne Pond

Few salmon parr were available for comparison of size distribution between pelagic fishing zones. However from the information available, a greater range in size occurred in the tropholytic zone although the size of parr was not significantly different (Table 18).

With respect to brook trout caught in the pelagic zone, ranges of fish caught in both zones were similar and there were no significant differences in length for the both summer period or when seasons were combined (Taple 19).

AGE DISTRIBUTION BY LENTIC ZONE

## Benthic Zone versus Pelagic Zone

## Junction Pond

Age composition (percent) of Atlantic salmon parr taken in the benthic and pelagic zones in summer and fall (data for all depths combined in each category) separately and for seasons combined is presented in Table 20. Data for night and day were combined. Five age-groups ( $0+-5_{+}$) were present in benthic nets in the summer and 4 age-groups in the fall ( $0+-4+$ ), Underyearling ( $0+$ ) parr were first encountered in benthic nets in August, presumably after they had reached a size at which they became susceptible to the gear. Age-groups $1+$ to $4+$ were encountered in pelagic gear in summer. The $2+$ age-group was predominant
throughout. There was no significant difference in age of parr between zones in summer (Table 20). However, for summer and fall combined, benthic parr were significantly younger than pelagic parr.

Brook trout ranged in age from $1+$ to $8+$ in the benthic zone in summer (agegroups $5+-7+$ were not represented) and from $1+$ to $3+$ in fall (Table 21). Ages $2+-3+$ were encountered in the pelagic net in summer. The $3+$ age-group predominated in summer and $2+$ in fall. There was no significant difference in age between zones.

Few specimens of Arctic charr were available for age analysis. Those that were available from fish caught during the summer ranged in age from $4+$ to $9+$ years. Pelagic specimens $(N=28)$ had a mean age of 6.5 years while those caught in benthic nets ( $N=14$ ) had a mean age of 6.8 years. There was no significant difference in age between lentic zones ( $Z=0.603$; $P=0.547$ ).

Brown trout in the benthic zone ranged in age from $2+$ to $5+$ years in the summer ( $N=4$ ) and from $2+$ to $4+$ in the fall ( $N=6$ ); pelagic fish (summer) ranged from $2+$ to $4+(N=3)$.

## Conne Pond

Age composition (percent) of Atlantic salmon parr taken in the benthic and pelagic zones in summer and fall (data for all depths combined in each category) separately and for seasons combined is summarized in Table 22. Data for night and day periods were combined. Six age-groups (ages $1+-6+$ ) were present in benthic nets during the summer but only three age-groups ( $1+-3+$ ) present during the fall. Similarly, only three age-groups ( $2+-4+$ ) were caught in the pelagic nets during the summer. No underyearling parr were caught in Conne Pond in contrast with that observed in Junction Pond. Age-groups $2+$ and $3+$ were the most predominant in Conne Pond in both lentic zones. There were no significant differences in ages of parr between zones for either the summer period or when seasons were combined (Table 22).

Brook trout ranged in age from $2+$ to $5+$ in the benthic zone in summer or fall and from $3+$ to $6+$ in the pelagic summer sample (Table 23). Age-groups $3+$ and $4+$ were the most predominant in both lentic zones. In contrast with Junction Pond, trout in the pelagic zone were significantly older than in the benthic zone for both the summer season and when the seasons were combined (Table 23).

## Benthic Zone - Littoral versus Nonlittoral

## Junction Pond

Underyearling ( $0_{+}$) Atlantic salmon parr were caught in the littoral zone but not in the nonlittoral zone for both summer and fall (Table 24). Littoral age-groups ranged from $0_{+}$to $4+$. Ages in the nonlittoral zone ranged from $1+$ to $5+$ in the summer and from $1+$ to $4+$ in the fall. Nonlittoral zone parr were significantly older than those of the littoral zone for seasons separately and combined.

The single $8+$ year old specimen of brook trout was taken in the nonlittoral zone in summer (Table 25). For age-groups $1+-4+$ there was no consistent distributional pattern by zone or season. Age was not significantly different between zones in summer and fall separately but there was a significant difference when seasons were combined (the nonlittoral zone had the oldest parr). For brown trout, age-groups $2+-5+$ were present in the nonlittoral zone in summer ( $\mathrm{N}=3$ ) and $2+-4+$ in the fall ( $\mathrm{N}=4$ ). A single $3+$ specimen was found in the littoral zone in summer and two specimens aged $3+$ and $4+$ in the fall.

## Conne Pond

Age $1+$ salmon parr were caught in both littoral and nonlittoral zones from the benthic net samples (Table 26). The oldest parr ( $6+$ ) were caught only in the nonlittoral zone. Age distributions were generally similar between summer and fall sampling periods and there were no significant differences between any categories.

The age-groups of brook trout caught in littoral and nonlittoral zones in summer were similar and ranged from $2+$ to $5+$ years (Table 27). During the fall, the same age-groups were encountered in the littoral samples but only 3+ and 4+ trout were caught in the nonlittoral zone. Brook trout in the nonlittoral zone were significantly older than in the littoral zone for both the summer season and when seasons were combined.

## Pelagic Zone - Trophogenic versus Tropholytic

## Junction Pond

Atlantic salmon parr aged $1+-3+$ were present in the trophogenic zone and age-groups $1+-4+$ were found in the tropholytic zone (Table 28). There was no significant difference in age between zones.

For brook trout found in the trophogenic zone in summer, age -groups $2+$ and $3+$ were represented ( $\mathrm{N}=3$ ) ; only the $3+$ age-group was found in the tropholytic zone.

For Arctic charr captured in the trophogenic zone in summer ( $N=6$ ), ages ranged from $4+$ to $8+$ years with a mean of 5.8 years. Specimens caught in the tropholytic zone ( $N=22$ ) ranged in age from $4+$ to $9+$ years with a mean of 6.7 years. Ages were not significantly different between zones ( $Z=1.474$; $P=$ 0.141).

Brown trout were represented by the $4+$ age group ( $\mathrm{N}=1$ ) in the trophogenic zone and by ages $2+$ and $3+$ in the tropholytic zone ( $N=2$ ).

## Conne Pond

Atlantic salmon parr aged $2+$ and $3+$ were present in the trophogenic zone while ages $2+$ to $4+$ were present in the tropholytic zone during the summer
season (Table 29). Few specimens, however, were available and ages were not significantly different between zones.

Age-groups 3+ and 4+ were the predominant groups for brook trout captured in either the trophogenic or tropholytic zones during the summer (Table 30). During the fall, only age $4+$ trout were sampled and these were caught in the tropholytic zone. There were no significant differences in ages between any categories.

## WINTER SAMPLING

Atlantic salmon parr, brook trout, and Arctic charr were encountered in benthic nets under the ice in Junction Pond. No specimens of either species were taken in pelagic net sets. Two salmon parr were taken at night at 2 and 5 m depths and 1 was taken during the day at 5 m . Seven brook trout were caught in the night set at 2 and 5 m and 6 during day at the same depths. A single Arctic charr was caught at 10 m during the night.

## Discussion

While there were many similarities in temporal and spatial distributions of Atlantic salmon parr between Junction Pond and Conne Pond, there were also differences. For both ponds, most parr were captured in the benthic zone as opposed to the pelagic zone. The distribution into these zones was not independent of diurnal sampling period in either pond. However, there was a difference between ponds with respect to season in this regard. With respect to the partitioning of the benthic zone, most parr were found in the littoral zone in Conne Pond both night and day while in Junction Pond night captures were mainly littoral and day captures mainly nonlittoral. Distribution was not independent of diurnal sampling period in Junction Pond while in Conne Pond it was; there was no difference with respect to season for either pond. For the pelagic zone, most parr occurred in the tropholytic zone during night and day in Junction Pond but in Conne Pond, most were trophogenic at night but tropholytic during day. Occurrences were not independent of diurnal sampling period in both ponds.

Combining all data temporally, in Junction Pond $88 \%$ of all Atlantic salmon parr were taken in the benthic zone; this compares to $85 \%$ for Conne Pond. Overall in the benthic zone, $67 \%$ were littoral occurrences in Junction Pond compared to $85 \%$ in Conne Pond. Out of the total number of parr taken in all zones, the benthic littoral zone comprised $59 \%$ of captures in Junction Pond and 72\% in Conne Pond making it by far the single most frequented zone. Catches in each zone were not adjusted for fishing effort or the relative proportion of each habitat type. In terms of fishing effort, however, the single pelagic net had a total surface area greater than all benthic nets combined in Conne Pond, but similar to the area of benthic nets in Junction Pond.

Similar to Atlantic salmon parr, most brook trout were taken in benthic nets in both ponds. Overall there were more pelagic occurrences in Conne Pond than in Junction Pond and there were differences between ponds with respect
distribution into the benthic versus the pelagic zone both diurnally and seasonally. In the benthic zone, brook trout were mainly nonlittoral at night which is opposite of what was observed for salmon parr. During day both salmon parr and brook trout were mainly nonlittoral. In Conne Pond, in contrast to Junction Pond most brook trout were littoral at night but had a similar distribution during the day. The ponds showed differences in distribution into littoral and nonlittoral zones in terms of diurnal sampling period and season. The total number of benthic nets fished in the nonlittoral zone exceeded that of the littoral zone in both ponds, which emphasizes the relative importance of the littoral zone. In the pelagic zone, brook trout were mainly trophogenic in June but tropholytic in July in Junction Pond compared to mainly tropholytic throughout for Conne Pond. The spectrum of tests of independence of occurrences with respect to diurnal sampling period and season for brook trout differed between ponds.

In Junction Pond, $96 \%$ of all brook trout were captured in the benthic zone compared to $79 \%$ for Conne Pond. Within the benthic zone, overall $63 \%$ were nonlittoral in Junction Pond and $47 \%$ were nonlittoral in Conne Pond. In contrast to Atlantic salmon parr, the littoral zone was less important overall in terms of occurrences for brook trout in both ponds ( $35 \%$ for Junction Pond; 42\% for Conne Pond).

Of the few brown trout encountered in Junction Pond, most were benthic and of these most were nonlittoral. Arctic charr were more pelagic than the other species and when present in the benthic zone occurred mainly in the nonlittoral zone.

Atlantic salmon parr captured in the pelagic zone overall and in the nonlittoral portion of the benthic zone were significantly larger than those from the littoral zone in both ponds. There was no significant difference between benthic and pelagic zones for brook trout in Junction Pond, but in Conne Pond, pelagic specimens were significantly larger than benthic specimens. In the benthic zone, nonlittoral brook trout were significantly longer than littoral trout in both ponds.

With respect to age composition, older Atlantic salmon parr tended to occupy the pelagic and nonlittoral zones than found in the littoral zone in Junction Pond while in Conne Pond there were no differences. Pepper et al. (1985) studied the spatial distribution of parr in three ponds in northeastern Newfoundland using beach seines, fyke nets, and gillnets (not the Lundgren type). They found that $1+$ and $2+$ parr were captured most often in shallow water ( $<2 \mathrm{~m}$ deep), whereas $3+$ and $4+$ parr were rarely found inshore and were captured in deeper offshore areas. The degree of overlap in age groups between littoral and nonlittoral areas in Junction Pond, however, was greater than reported by Pepper et al. (1985). Underyearling (0+) parr were not taken in Conne Pond; their presence in Junction Pond is likely related to the fact that this pond is immediately downstream from a major spawning area. Pepper et al. (1985) did not encounter $0+$ parr in their ponds.

Brook trout in the pelagic zone were significantly older than in the benthic zone in Conne Pond but not in Junction Pond. In the benthic zone, trout in the nonlittoral zone tended to be older than in the littoral zone in both ponds.

The differences in the temporal and spatial usage of various lacustrine habitats may be the result of an overall tendency of certain groups to occupy a particular lentic area. In order to meaningfully delineate biomass or define production in terms of different lentic areas, more research is needed to determine the extent of movements of fish of different sizes among zones within a given lake, among lakes, and between lacustrine and fluvial habitats. Analyses are currently underway attempting to explain the observed differences in terms of physical and chemical variables and feeding relationships.

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Table 1. Morphometry of Junction Pond, Northeast River, Placentia and Conne Pond, Conne River, Newfoundland.

| Parameter | Junction Pond | Conne River Pond |
| :---: | :---: | :---: |
| Surface area (ha) | 61.9 | 75.0 |
| Volume (m) | $1.93 \times 10^{6}$ | $2.03 \times 10^{6}$ |
| Maximum length (km) | 1.72 | 1.78 |
| Maximum width (km) | 0.66 | 0.74 |
| Mean width (km) | 0.36 | 0.42 |
| Maximum depth (m) | 12.0 | 10.0 |
| Mean depth (m) | 3.12 | 2.71 |
| Length of shoreline (km) | 5.6 | 4.8 |
| Mean depth - maximum |  |  |
| depth relation | 0.26 | 0.27 |
| Mean depth - surface relation | 0.015 | 0.012 |
| Shore development | 2.01 | 1.58 |
| Volume development | 0.78 | 0.81 |

Table 2. Water chemistry for Junction Pond, Northeast River, placentia and Conne Pond, Conne River, Newfoundland. Both ponds were sampled in July 1986.

| Parameter | Junction Pond | Conne River Pond |
| :---: | :---: | :---: |
| pH | 6.68 | 6.14 |
| Alkalinity (mg/l $\mathrm{CaCO}_{3}$ ) | 3.6 | 4.2 |
| Specific conductance ( $\mu \mathrm{S} / \mathrm{cm}$ ) | 32 | 34 |
| Total hardness (mg/l $\mathrm{CaCO}_{3}$ ) | 7.3 | 8.5 |
| Calcium (mg/l) | 1.92 | 2.53 |
| Magnesium (mg/2) | 0.60 | 0.54 |
| Chloride (mg/l Cl) | 5.9 | 3.5 |
| Sulfate ( $\mathrm{mg} / \mathrm{l} \mathrm{SO}_{4}$ ) | 2.6 | 3.4 |
| Turbidity (NTU) | 0.31 | 0.36 |
| True colour | - | 70 |

Table 3. Distribution (no. of fish) of Arctic charr captured in benthic nets by contour depth, night and day, each month. $N=$ night, $D=$ day.

| Contour depth (m) | Month |  |  |  |  |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | June |  | July |  | Aug. |  | Sept. |  | Oct. |  | $\frac{\text { Nov. }}{\mathrm{N}}$ |
|  | N | D | N | D | N | D | N | D | N | D |  |
| 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 2 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1 | 1 |
| 5 | 1 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 7 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 3 | 1 | 1 |
| 9 | 0 | 0 | 9 | 0 | 0 | 0 | 0 | 1 | 0 | 1 | 2 |
| 11 | 0 | 0 | 3 | 0 | 0 | 1 | 0 | 0 | 0 | 2 | 2 |
| Total | 2 | 0 | 15 | 0 | 0 | 1 | 0 | 1 | 4 | 5 | 6 |

Table 4. Distribution (no. of fish) of Atlantic salmon pary, Brook trout and Arctic charr captured in pelagic experimental gill nets by depth interval and by trophogenic and tropholytic zones, Junction Pond, Newfoundland. $N=$ night, $D=$ day.

| ```Depth interval (cm)``` | Sampling Period |  |  |  |  |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | June |  | July |  | Aug. |  | Sept. |  | Oct. |  | $\frac{\text { Nov. }}{\mathrm{N}}$ |
|  | N | D | N | D | N | D | N | D | N | D |  |
| ATLANTIC SALMON Trophogenic |  |  |  |  |  |  |  |  |  |  |  |
| 0-1 | 3 | 0 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 1-2 | 0 | 1 | 4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 2-3 | 0 | 0 | 9 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Total | 3 | 1 | 16 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Tropholytic |  |  |  |  |  |  |  |  |  |  |  |
| 3-4 | 2 | 1 | 9 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 4-5 | 1 | 2 | 4 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 5-6 | 7 | 3 | 9 | 2 | 0 | 1 | 0 | 0 | 0 | 0 | 0 |
| Total | 10 | 6 | 22 | 4 | 0 | 1 | 0 | 0 | 0 | 0 | 0 |
| Grand Total | 13 | 7 | 38 | 4 | 0 | 1 | 0 | 0 | 0 | 0 | 0 |
| BROOK TROUT Trophogenic |  |  |  |  |  |  |  |  |  |  |  |
| 0-1 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 1-2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 2-3 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Total | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Tropholytic |  |  |  |  |  |  |  |  |  |  |  |
| 3-4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 4-5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 5-6 | 1 | 1 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Total | 1 | 1 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Grand Total | 4 | 1 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| ARCTIC CHARR <br> Trophogenic |  |  |  |  |  |  |  |  |  |  |  |
| 0-1 | 7 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 |  | 0 |
| 1-2 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |  | 0 |
| 2-3 | 1 | 0 | 10 | 0 | 0 | 0 | 0 | 0 | 0 |  | 0 |
| Total | 9 | 0 | 11 | 0 | 0 | 0 | 0 | 0 | 0 |  | 0 |
| Tropholytic |  |  |  |  |  |  |  |  |  |  |  |
| 3-4 | 0 | 0 | 6 | 3 | 0 | 0 | 0 | 0 | 0 |  | 0 |
| 4-5 | 0 | 0 | 5 | 0 | 0 | 0 | 0 | 0 | 0 |  | 0 |
| 5-6 | 0 | 0 | 0 | 4 | 0 | 0 | 0 | 0 | 0 |  | 0 |
| Total | 0 | 0 | 11 | 7 | 0 | 0 | 0 | 0 | 0 |  | 0 |
| Grand Total | 9 | 0 | 22 | 7 | 0 | 0 | 0 | 0 | 0 | 3* | 0 |

* Depth interval information not available.

Table 5. Distribution (no. of fish) of Atlantic salmon parr and Brook trout captured in pelagic experimental gill nets by depth interval and by trophogenic and tropholytic zones, Conne Pond, Newfoundland. $\mathrm{N}=$ night, $D=$ day.

| $\begin{aligned} & \text { Depth } \\ & \text { interval (cm) } \end{aligned}$ | Sampling Period |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: |
|  | June | July | Aug. | Sept. | 0ct. |
|  | N D | N D | $\mathrm{N} \quad \mathrm{D}$ | N D | N D |

ATLANTIC SALMON
Trophogenic

| $0-1$ | 2 | 0 | 7 | 0 | 0 | 2 | 0 | 0 | 0 | 0 |
| :---: | ---: | ---: | ---: | ---: | :--- | :--- | :--- | :--- | :--- | :--- |
| $1-2$ | 0 | 0 | 3 | 0 | 3 | 0 | 0 | 0 | 0 | 0 |
| Total | 2 | 0 | 10 | 0 | 3 | 2 | 0 | 0 | 0 | 0 |

Tropholytic

| $2-3$ | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| $3-4$ | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| $4-5$ | 0 | 0 | 0 | 1 | 0 | 1 | 0 | 0 | 0 | 0 |
| $5-6$ | 0 | 1 | 0 | 0 | 0 | 6 | 0 | 0 | 0 | 0 |
|  | 0 | 1 | 0 | 1 | 1 | 7 | 0 | 0 | 0 | 0 |
| Total |  |  |  |  |  |  |  |  |  |  |
|  | 2 | 1 | 10 | 1 | 4 | 9 | 0 | 0 | 0 | 0 |

## BROOR TROUT

Trophogenic

| $0-1$ | 0 | 0 | 3 | 1 | 0 | 0 | 0 | 0 | 0 | 0 |
| :---: | :--- | :--- | :--- | :--- | :--- | :--- | :--- | :--- | :--- | :--- |
| $1-2$ | 0 | 2 | 0 | 3 | 1 | 0 | 0 | 0 | 0 | 0 |
| Total | 0 | 2 | 3 | 4 | 1 | 0 | 0 | 0 | 0 | 0 |

Tropholytic

| $2-3$ | 3 | 2 | 1 | 5 | 0 | 0 | 1 | 0 | 1 | 0 |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| $3-4$ | 0 | 0 | 0 | 1 | 1 | 1 | 0 | 0 | 0 | 0 |
| $4-5$ | 0 | 0 | 0 | 1 | 1 | 1 | 0 | 0 | 0 | 1 |
| $5-6$ | 0 | 7 | 0 | 0 | 1 | 2 | 0 | 0 | 0 | 0 |
| Total | 3 | 9 | 1 | 7 | 3 | 4 | 1 | 0 | 1 | 1 |
|  |  |  |  |  |  |  |  |  |  |  |
| Grand Total | 3 | 11 | 4 | 11 | 4 | 4 | 1 | 0 | 1 | 1 |

Table 6. Percentage distribution of Atlantic salmon parr by length caught in benthic (B) and pelagic (P) experimental gill nets in Junction Pond, summarized by seasons separately and combined. Also included are statistical comparisons in the benthic versus pelagic categories (Wilcoxon two-sample test, Z).

| $\begin{aligned} & \text { Fork } \\ & \text { Length (cm) } \end{aligned}$ | Season |  |  |  | Total |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Summer |  | Fall |  |  |  |
|  | B | P | B | P | B | P |
| 5 | 1.9 | 0.0 | 0.6 |  | 1.3 | 0.0 |
| 6 | 0.5 | 0.0 | 6.3 |  | 3.1 | 0.0 |
| 7 | 3.3 | 0.0 | 5.7 |  | 4.4 | 0.0 |
| 8 | 10.8 | 2.0 | 0.6 |  | 6.2 | 2.0 |
| 9 | 8.0 | 0.0 | 1.1 |  | 4.9 | 0.0 |
| 10 | 8.5 | 2.0 | 6.9 |  | 7.7 | 2.0 |
| 11 | 8.5 | 9.8 | 5.7 |  | 7.2 | 9.8 |
| 12 | 7.0 | 21.6 | 4.6 |  | 5.9 | 21.6 |
| 13 | 7.0 | 7.8 | 8.6 |  | 7.7 | 7.8 |
| 14 | 6.1 | 7.8 | 10.9 |  | 8.2 | 7.8 |
| 15 | 9.9 | 5.9 | 21.1 |  | 14.9 | 5.9 |
| 16 | 10.8 | 11.8 | 18.9 |  | 14.4 | 11.8 |
| 17 | 5.2 | 7.8 | 6.3 |  | 5.7 | 7.8 |
| 18 | 6.1 | 13.7 | 1.7 |  | 4.1 | 13.7 |
| 19 | 4.7 | 3.9 | 0.6 |  | 2.8 | 3.9 |
| 20 | 0.9 | 2.0 | 0.6 |  | 0.8 | 2.0 |
| 21 | 0.9 | 3.9 | 0.0 |  | 0.5 | 3.9 |
| No. of fish | 213 | 51 | 175 | 0 | 388 | 51 |
| Mean |  | 14.7 | $13.2$ |  | 13.0 | 14.7 |
| SD | 3.78 | 3.10 | 3.36 |  | 3.60 | 3.10 |
| 2 |  |  |  |  |  | 82 |
| P |  |  |  |  |  | 005 |

Table 7. Percentage distribution of Brook trout by length caught in benthic (B) and pelagic (P) experimental gill nets in Junction Pond, summarized by seasons separately and combined. Also included are statistical comparisons in the benthic versus pelagic categories (Wilcoxon two-sample test, Z).

| $\begin{aligned} & \text { Fork } \\ & \text { Length (cm) } \end{aligned}$ | Season |  |  |  | Total |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Summer |  | Fall |  |  |  |
|  | B | P | B | P | B | P |
| 11 | 1.3 | 0.0 | 0.0 |  | 0.6 | 0.0 |
| 12 | 2.6 | 0.0 | 0.0 |  | 1.3 | 0.0 |
| 13 | 0.0 | 0.0 | 2.6 |  | 1.3 | 0.0 |
| 14 | 1.3 | 0.0 | 14.3 |  | 7.7 | 0.0 |
| 15 | 1.3 | 0.0 | 7.8 |  | 4.5 | 0.0 |
| 16 | 3.9 | 0.0 | 19.5 |  | 11.6 | 0.0 |
| 17 | 1.3 | 0.0 | 10.4 |  | 5.8 | 0.0 |
| 18 | 2.6 | 0.0 | 5.2 |  | 3.9 | 0.0 |
| 19 | 5.1 | 50.0 | 3.9 |  | 4.5 | 50.0 |
| 20 | 11.5 | 16.7 | 3.9 |  | 7.7 | 16.7 |
| 21 | 12.9 | 0.0 | 10.4 |  | 11.6 | 0.0 |
| 22 | 2.6 | 16.7 | 9.1 |  | 5.8 | 16.7 |
| 23 | 6.4 | 0.0 | 6.5 |  | 6.5 | 0.0 |
| 24 | 5.1 | 0.0 | 3.9 |  | 4.5 | 0.0 |
| 25 | 10.3 | 0.0 | 2.6 |  | 6.5 | 0.0 |
| 26 | 6.4 | 16.7 | 0.0 |  | 3.2 | 16.7 |
| 27 | 6.4 | 0.0 | 0.0 |  | 3.2 | 0.0 |
| 28 | 7.7 | 0.0 | 0.0 |  | 3.9 | 0.0 |
| 29 | 5.1 | 0.0 | 0.0 |  | 2.6 | 0.0 |
| 30 | 2.6 | 0.0 | 0.0 |  | 1.3 | 0.0 |
| 31 | 2.6 | 0.0 | 0.0 |  | 1.3 | 0.0 |
| 32 | 1.3 | 0.0 | 0.0 |  | 0.6 | 0.0 |
| No. of fish | 78 | 6 | 77 | 0 | 155 | 6 |
| Mean | 22.9 |  | 18.1 |  | 20.5 | 20.8 |
| SD | 4.72 | 2.79 | 3.40 |  | 4.76 | 2.79 |
| Z |  |  |  |  |  | 0.15 |
| P |  |  |  |  |  | 0.879 |

Table 8. Percentage distribution of Arctic Charr by length caught in benthic ( $B$ ) and pelagic ( $P$ ) experimental gill nets in Junction Pond, summarized by seasons separately and combined. Also included are statistical comparisons in the benthic versus pelagic categories (Wilcoxon two-sample test, $Z$ ).

| $\underset{\text { Forkgth }}{\text { Lom }}$ ( | Season |  |  |  | Total |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Summer |  | Fall |  |  |  |
|  | B | $P$ | B | P | B | P |
| 7 | 0.0 | 0.0 | 6.7 | 0.0 | 3.1 | 0.0 |
| 8 | 0.0 | 0.0 | 26.7 | 0.0 | 12.5 | 0.0 |
| 9 | 0.0 | 0.0 | 13.3 | 100.0 | 6.3 | 9.1 |
| 10 | 5.9 | 0.0 | 0.0 | 0.0 | 3.1 | 0.0 |
| 11 | 5.9 | 3.3 | 0.0 | 0.0 | 3.1 | 3.0 |
| 12 | 0.0 | 0.0 | 13.3 | 0.0 | 6.3 | 0.0 |
| 13 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| 14 | 0.0 | 0.0 | 6.7 | 0.0 | 3.1 | 0.0 |
| 15 | 5.9 | 16.7 | 6.7 | 0.0 | 6.3 | 15.1 |
| 16 | 52.9 | 50.0 | 0.0 | 0.0 | 28.1 | 45.5 |
| 17 | 23.5 | 26.7 | 26.7 | 0.0 | 25.0 | 24.2 |
| 18 | 5.9 | 3.3 | 0.0 | 0.0 | 3.1 | 3.0 |
| No. of fish | 17 | 30 | 15 | 3 | 32 | 33 |
| Mean | 15.6 | 16.0 | 11.9 | 9.0 | 13.9 | 15.4 |
| SD | 2.06 | 1.20 | 3.96 | 0 | 3.60 | 2.34 |
| 2 |  |  |  | 484 |  |  |
| P |  |  |  | 628 |  | 32 |

Table 9. Percentage distribution of Atlantic salmon parr by length caught in benthic (B) and pelagic (P) experimental gill nets, in Conne Pond, summarized by seasons separately and combined. Also included are statistical comparisons in the benthic versus pelagic categories (Wilcoxon two-sample test, $z$ ).

| ```Fork Length (cm)``` | Season |  |  |  | Total |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Summer |  | Fali |  |  |  |
|  | B | P | B | P | $\bar{B}$ | P |
| 7 | 11.2 | 0.0 | 3.6 |  | 9.6 | 0.0 |
| 8 | 3.7 | 0.0 | 0.0 |  | 3.0 | 0.0 |
| 9 | 16.8 | 0.0 | 10.7 |  | 15.6 | 0.0 |
| 10 | 15.9 | 10.0 | 14.3 |  | 15.6 | 10.0 |
| 11 | 22.4 | 20.0 | 14.3 |  | 20.7 | 20.0 |
| 12 | 14.0 | 40.0 | 28.6 |  | 17.0 | 40.0 |
| 13 | 8.4 | 20.0 | 17.9 |  | 10.4 | 20.0 |
| 14 | 2.8 | 0.0 | 10.7 |  | 4.4 | 0.0 |
| 15 | 2.8 | 10.0 | 0.0 |  | 2.2 | 10.0 |
| 19 | 0.9 | 0.0 | 0.0 |  | 0.7 | 0.0 |
| 24 | 0.9 | 0.0 | 0.0 |  | 0.7 | 0.0 |
| No. of fish | 107 | 10 | 28 | 0 | 135 | 10 |
| Mean | 10.6 | 12.1 | 11.5 |  | 10.8 | 12.1 |
| SD | 2.50 | 1.37 | 1.73 |  | 2.38 | 1.37 |
| z |  |  |  |  |  |  |
| P |  | 012 |  |  |  | 028 |

Table 10. Percentage distribution of Brook trout by length caught in benthic (B) and pelagic ( $P$ ) experimental gill nets, in Conne Pond, summarized by seasons separately and combined. Also included are statistical comparisons in the benthic versus pelagic categories (Wilcoxon two-sample test, Z).

| $\begin{aligned} & \text { Fork } \\ & \text { Length (cm) } \end{aligned}$ | Season |  |  |  | Total |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Summer |  | Fall |  |  |  |
|  | B | P | B | P | B | P |
| 10 | 0.8 | 0.0 | 0.0 | 0.0 | 0.7 | 0.0 |
| 11 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| 12 | 0.0 | 0.0 | 3.7 | 0.0 | 0.7 | 0.0 |
| 13 | 0.8 | 0.0 | 0.0 | 0.0 | 0.7 | 0.0 |
| 14 | 2.5 | 0.0 | 0.0 | 0.0 | 2.0 | 0.0 |
| 15 | 6.7 | 0.0 | 0.0 | 0.0 | 5.4 | 0.0 |
| 16 | 6.7 | 0.0 | 0.0 | 0.0 | 5.4 | 0.0 |
| 17 | 6.7 | 0.0 | 3.7 | 0.0 | 6.1 | 0.0 |
| 18 | 5.0 | 9.1 | 3.7 | 0.0 | 4.7 | 8.3 |
| 19 | 8.3 | 9.1 | 7.4 | 0.0 | 8.2 | 8.3 |
| 20 | 11.7 | 0.0 | 14.8 | 0.0 | 12.2 | 0.0 |
| 21 | 10.8 | 21.2 | 25.9 | 66.7 | 13.6 | 25.0 |
| 22 | 7.5 | 18.1 | 11.1 | 0.0 | 8.2 | 16.7 |
| 23 | 9.2 | 6.1 | 18.5 | 0.0 | 10.9 | 5.6 |
| 24 | 8.3 | 3.0 | 7.4 | 0.0 | 8.2 | 2.8 |
| 25 | 5.0 | 3.0 | 0.0 | 0.0 | 4.1 | 2.8 |
| 26 | 1.7 | 9.1 | 3.7 | 33.1 | 2.0 | 11.1 |
| 27 | 3.3 | 6.1 | 0.0 | 0.0 | 2.7 | 5.6 |
| 28 | 1.7 | 9.1 | 0.0 | 0.0 | 1.4 | 8.3 |
| 29 | 0.8 | 0.0 | 0.0 | 0.0 | 0.7 | 0.0 |
| 30 | 0.8 | 0.0 | 0.0 | 0.0 | 0.7 | 0.0 |
| 31 | 0.8 | 0.0 | 0.0 | 0.0 | 0.7 | 0.0 |
| 32 | 0.8 | 6.1 | 0.0 | 0.0 | 0.7 | 5.6 |
| No. of fish | 120 | 33 | 27 | 3 | 147 | 36 |
| Mean | 20.7 | 23.2 | 21.0 | 22.7 | 20.7 | 23.1 |
| SD | 4.04 | 3.78 | 2.66 | 2.89 | 3.82 | 3.68 |
| Z |  |  |  | 0.74 |  | . 04 |
| P |  | . 03 |  | 0.460 |  | . 002 |

Table 11. Percentage distribution of Atlantic salmon parr by length caught in benthic experimental gill nets in littoral (L) and nonlittoral (N) zones in Junction Pond, summarized by seasons separately and combined. Also included are statistical comparisons in the littoral versus nonlittoral categories (Wilcoxon two-sample test, Z).

| $\begin{aligned} & \text { Fork } \\ & \text { Length (cm) } \end{aligned}$ | Season |  |  |  | Total |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Summer |  | Fall |  |  |  |
|  | L | N | L | N | L | N |
| 5 | 2.7 | 0.0 | 0.9 | 0.0 | 1.9 | 0.0 |
| 6 | 0.7 | 0.0 | 10.1 | 0.0 | 4.7 | 0.0 |
| 7 | 4.7 | 0.0 | 9.2 | 0.0 | 6.6 | 0.0 |
| 8 | 14.1 | 3.1 | 0.9 | 0.0 | 8.5 | 1.5 |
| 9 | 10.7 | 1.6 | 1.8 | 0.0 | 7.0 | 0.8 |
| 10 | 9.4 | 6.3 | 10.1 | 1.5 | 9.7 | 3.9 |
| 11 | 9.4 | 6.3 | 8.3 | 1.5 | 8.9 | 3.9 |
| 12 | 8.1 | 4.7 | 6.4 | 1.5 | 7.4 | 3.1 |
| 13 | 7.4 | 6.3 | 10.1 | 6.1 | 8.5 | 6.2 |
| 14 | 6.0 | 6.3 | 11.0 | 10.6 | 8.1 | 8.5 |
| 15 | 8.1 | 14.1 | 17.4 | 27.3 | 12.0 | 20.8 |
| 16 | 9.4 | 14.1 | 9.2 | 34.9 | 9.3 | 24.6 |
| 17 | 2.7 | 10.9 | 1.8 | 13.6 | 2.3 | 12.3 |
| 18 | 4.0 | 10.9 | 0.9 | 3.0 | 2.7 | 6.9 |
| 19 | 1.3 | 12.5 | 0.9 | 0.0 | 1.2 | 6.2 |
| 20 | 0.7 | 1.6 | 0.9 | 0.0 | 0.8 | 0.8 |
| 21 | 0.7 | 1.6 | 0.0 | 0.0 | 0.4 | 0.8 |
| No. of fish | 149 | 64 | 109 | 66 | 258 |  |
| Mean | 11.8 | 15.2 | 12.0 | 15.3 | 11.9 | 15.2 |
| SD | 3.57 | 3.17 | 3.56 | 1.49 | 3.56 | 2.45 |
| Z | 6.01 |  | 6.81 |  | 8.98 |  |
| P | 0.000 |  | 0.000 |  | 0.000 |  |

Table 12. Percentage distribution of Brook trout by length caught in benthic experimental gill nets in littoral (L) and nonlittoral (N) zones in Junction Pond, summarized by seasons separately and combined. Also included are statistical comparisons in the littoral versus nonlittoral categories (Wilcoxon two-sample test, Z).

| $\begin{aligned} & \text { Fork } \\ & \text { Length (cm) } \end{aligned}$ | Season |  |  |  | Total |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Summer |  | Fall |  |  |  |
|  | L | N | L | N | L | N |
| 11 | 0.0 | 1.8 | 0.0 | 0.0 | 0.0 | 1.0 |
| 12 | 4.6 | 1.8 | 0.0 | 0.0 | 1.8 | 1.0 |
| 13 | 0.0 | 0.0 | 2.9 | 2.4 | 1.8 | 1.0 |
| 14 | 4.6 | 0.0 | 17.1 | 11.9 | 12.3 | 5.1 |
| 15 | 4.6 | 0.0 | 8.6 | 7.1 | 7.0 | 3.1 |
| 16 | 4.6 | 3.6 | 20.0 | 19.1 | 14.0 | 10.2 |
| 17 | 0.0 | 1.8 | 11.4 | 9.5 | 7.0 | 5.1 |
| 18 | 4.6 | 1.8 | 5.7 | 4.8 | 5.3 | 3.1 |
| 19 | 4.6 | 5.4 | 2.9 | 4.8 | 3.5 | 5.1 |
| 20 | 4.6 | 14.3 | 5.7 | 2.4 | 5.3 | 9.2 |
| 21 | 13.6 | 12.5 | 5.7 | 14.3 | 8.8 | 13.3 |
| 22 | 4.6 | 1.8 | 8.6 | 9.5 | 7.0 | 5.1 |
| 23 | 9.1 | 5.4 | 5.7 | 7.1 | 7.0 | 6.1 |
| 24 | 9.1 | 3.6 | 2.9 | 4.8 | 5.3 | 4.1 |
| 25 | 0.0 | 14.3 | 2.9 | 2.4 | 1.8 | 9.2 |
| 26 | 4.6 | 7.1 | 0.0 | 0.0 | 1.8 | 4.1 |
| 27 | 9.1 | 5.4 | 0.0 | 0.0 | 3.5 | 3.1 |
| 28 | 9.1 | 7.1 | 0.0 | 0.0 | 3.5 | 4.1 |
| 29 | 9.1 | 3.6 | 0.0 | 0.0 | 3.5 | 2.0 |
| 30 | 0.0 | 3.6 | 0.0 | 0.0 | 0.0 | 2.0 |
| 31 | 0.0 | 3.6 | 0.0 | 0.0 | 0.0 | 2.0 |
| 32 | 0.0 | 1.8 | 0.0 | 0.0 | 0.0 | 1.0 |
| No. of fish | 22 | 56 | 35 | 42 | 57 | 98 |
| Mean | 22.1 | 23.3 | 17.7 | 18.4 | 19.4 | 21.2 |
| SD | 5.01 | 4.61 | 3.37 | 3.42 | 4.59 | 4.8 |
| Z |  |  |  |  |  |  |
| P |  |  |  |  |  |  |

Table 13. Percentage distribution of Arctic charr by length caught in benthic experimental gill nets in littoral (L) and nonlittoral (N) zones in Junction Pond, summarized by seasons separately and combined. Also included are statistical comparisons in the littoral versus nonlittoral categories (Wilcoxon two-sample test, $Z$ ).

| $\begin{aligned} & \text { Fork } \\ & \text { Length (cm) } \end{aligned}$ | Season |  |  |  | Total |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Summer |  | Fail |  |  |  |
|  | $\bar{L}$ | N | $L$ | N | L | N |
| 7 | 0.0 | 0.0 | 0.0 | 8.3 | 0.0 | 3.6 |
| 8 | 0.0 | 0.0 | 0.0 | 33.3 | 0.0 | 14.3 |
| 9 | 0.0 | 0.0 | 0.0 | 16.7 | 0.0 | 7.1 |
| 10 | 0.0 | 6.3 | 0.0 | 0.0 | 0.0 | 3.6 |
| 11 | 0.0 | 6.3 | 0.0 | 0.0 | 0.0 | 3.6 |
| 12 | 0.0 | 0.0 | 0.0 | 16.7 | 0.0 | 7.1 |
| 13 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| 14 | 0.0 | 0.0 | 0.0 | 8.3 | 0.0 | 3.6 |
| 15 | 100.0 | 0.0 | 33.3 | 0.0 | 50.0 | 0.0 |
| 16 | 0.0 | 56.3 | 0.0 | 0.0 | 0.0 | 32.1 |
| 17 | 0.0 | 25.0 | 66.7 | 16.7 | 50.0 | 21.4 |
| 18 | 0.0 | 6.3 | 0.0 | 0.0 | 0.0 | 3.6 |
| No. of fish | 1 | 16 | 3 | 12 | 4 | 28 |
| Mean | 15.0 | 15.7 | 16.3 | 10.8 | 16.0 | 13.6 |
| SD | - | 2.12 | 1.54 | 3.60 | 1.15 | 3.74 |
| z | 1.22 |  | 1.99 |  | 0.90 |  |
| P | 0.221 |  | 0.047 |  | 0.367 |  |

Table 14. Percentage distribution of Atlantic salmon parr by length caught in benthic experimental gill nets in littoral (L) and nonlittoral (N) zones, in Conne Pond, summarized by seasons separately and combined. Also included are statistical comparisons in the littoral versus nonlittoral categories (Wilcoxon two-sample test, $Z$ ).

| $\begin{aligned} & \text { Fork } \\ & \text { Length ( } \mathrm{cm} \text { ) } \end{aligned}$ | Season |  |  |  | Total |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Summer |  | Fall |  |  |  |
|  | L | N | L | N | L | N |
| 7 | 12.0 | 6.7 | 4.2 | 0.0 | 10.3 | 5.3 |
| 8 | 4.4 | 0.0 | 0.0 | 0.0 | 3.5 | 0.0 |
| 9 | 18.5 | 6.7 | 12.5 | 0.0 | 17.2 | 5.3 |
| 10 | 16.3 | 13.3 | 16.7 | 0.0 | 16.4 | 10.5 |
| 11 | 21.7 | 26.7 | 16.7 | 0.0 | 20.7 | 21.1 |
| 12 | 14.1 | 13.3 | 20.8 | 75.0 | 15.5 | 26.3 |
| 13 | 7.6 | 13.3 | 16.7 | 25.0 | 9.5 | 15.8 |
| 14 | 1.1 | 13.3 | 12.5 | 0.0 | 3.5 | 10.5 |
| 15 | 3.3 | 0.0 | 0.0 | 0.0 | 2.6 | 0.0 |
| 19 | 1.1 | 0.0 | 0.0 | 0.0 | 0.9 | 0.0 |
| 24 | 0.0 | 6.7 | 0.0 | 0.0 | 0.0 | 5.3 |
| No. of fish | 92 | 15 | 24 | 4 | 116 | 19 |
| Mean | 10.4 | 12.1 | 11.3 | 12.3 | 10.6 | 12.2 |
| SD | 2.16 | 3.78 | 1.83 | 0.50 | 2.12 | 3.34 |
| z | 2.06 |  | 0.94 |  | 2.38 |  |
| P | 0.040 |  | 0.349 |  | 0.017 |  |

Table 15. Percentage distribution of Brook trout by length caught in benthic experimental gill nets in littoral (L) and nonlittoral (N) zones, in Conne Pond, summarized by seasons separately and combined. Also included are statistical comparisons in the littoral versus nonlittoral categories (Wilcoxon two-sample test, Z).

| $\begin{gathered} \text { Fork } \\ \text { Length (cm) } \end{gathered}$ | Season |  |  |  | Total |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Summer |  | Fall |  |  |  |
|  | L | N | L | N | L | N |
| 10 | 1.7 | 0.0 | 0.0 | 0.0 | 1.3 | 0.0 |
| 12 | 0.0 | 0.0 | 5.0 | 0.0 | 1.3 | 0.0 |
| 13 | 0.0 | 1.6 | 0.0 | 0.0 | 0.0 | 1.5 |
| 14 | 3.4 | 1.6 | 0.0 | 0.0 | 2.5 | 1.5 |
| 15 | 11.9 | 1.6 | 0.0 | 0.0 | 8.9 | 1.5 |
| 16 | 8.5 | 4.9 | 0.0 | 0.0 | 6.3 | 4.4 |
| 17 | 10.2 | 3.3 | 5.0 | 0.0 | 8.9 | 2.9 |
| 18 | 6.8 | 3.3 | 5.0 | 0.0 | 6.3 | 2.9 |
| 19 | 10.2 | 6.6 | 0.0 | 28.6 | 7.6 | 8.8 |
| 20 | 15.3 | 8.2 | 10.0 | 28.6 | 13.9 | 10.3 |
| 21 | 8.5 | 13.1 | 30.0 | 14.3 | 13.9 | 13.2 |
| 22 | 3.4 | 11.5 | 15.0 | 0.0 | 6.3 | 10.3 |
| 23 | 6.8 | 11.5 | 20.0 | 14.3 | 10.1 | 11.8 |
| 24 | 0.0 | 16.4 | 10.0 | 0.0 | 2.5 | 14.7 |
| 25 | 6.8 | 3.3 | 0.0 | 0.0 | 5.1 | 2.9 |
| 26 | 1.7 | 1.6 | 0.0 | 14.3 | 1.3 | 2.9 |
| 27 | 1.7 | 4.9 | 0.0 | 0.0 | 1.3 | 4.4 |
| 28 | 1.7 | 1.6 | 0.0 | 0.0 | 1.3 | 1.5 |
| 29 | 0.0 | 1.6 | 0.0 | 0.0 | 0.0 | 1.5 |
| 30 | 0.0 | 1.6 | 0.0 | 0.0 | 0.0 | 1.5 |
| 31 | 1.7 | 0.0 | 0.0 | 0.0 | 1.3 | 0.0 |
| 32 | 0.0 | 1.6 | 0.0 | 0.0 | 0.0 | 1.5 |
| No. of fish | 59 | 61 | 20 | 7 | 79 | 68 |
| Mean | 19.4 | 21.9 | 21.0 | 21.1 | 19.8 | 21.8 |
| SD | 3.94 | 3.77 | 2.76 | 2.54 | 3.72 | 3.65 |
| Z |  |  |  |  | 3. |  |
| P |  |  |  | 519 |  |  |

Table 16. Percentage distribution of Atlantic salmon parr by length caught in pelagic experimental gill nets in trophogenic (TG) and tropholytic (TL) zones in Junction Pond, summarized by seasons separately and combined. Also included are statistical comparisons in the trophogenic versus tropholytic categories (Wilcoxon two-sample test, Z).

| $\begin{aligned} & \text { Fork } \\ & \text { Length (cm) } \end{aligned}$ | Season |  |  |  | Total |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Summer |  | Fall |  |  |  |
|  | TG | TL | TG | TL | TG | TL |
| 8 | 0.0 | 2.6 |  |  | 0.0 | 2.6 |
| 9 | 0.0 | 0.0 |  |  | 0.0 | 0.0 |
| 10 | 7.7 | 0.0 |  |  | 7.7 | 0.0 |
| 11 | 7.7 | 10.5 |  |  | 7.7 | 10.5 |
| 12 | 30.8 | 18.4 |  |  | 30.8 | 18.4 |
| 13 | 7.7 | 7.9 |  |  | 7.7 | 7.9 |
| 14 | 15.4 | 5.3 |  |  | 15.4 | 5.3 |
| 15 | 7.7 | 5.3 |  |  | 7.7 | 5.3 |
| 16 | 0.0 | 15.8 |  |  | 0.0 | 15.8 |
| 17 | 7.7 | 7.9 |  |  | 7.7 | 7.9 |
| 18 | 7.7 | 15.8 |  |  | 7.7 | 15.8 |
| 19 | 0.0 | 5.3 |  |  | 0.0 | 5.3 |
| 20 | 0.0 | 2.6 |  |  | 0.0 | 2.6 |
| 21 | 7.7 | 2.6 |  |  | 7.7 | 2.6 |
| No. of fish | 13 | 38 | 0 | 0 | 13 | 38 |
| Mean | 13.9 | 14.9 |  |  | 13.9 | 14.9 |
| SD | 3.12 | 3.10 |  |  | 3.12 | 3.10 |
| Z |  |  |  |  |  |  |
| P |  |  |  |  |  |  |

Table 17. Percentage distribution of Arctic charr by length caught in pelagic experimental gill nets in trophogenic (TG) and tropholytic (TL) zones in Junction pond, summarized by seasons separately and combined. Also included are statistical comparisons in the trophogenic versus tropholytic categories (Wilcoxon two-sample test, $Z$ ).

| Fork <br> Length (cm) | Season |  |  |  | Total |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Summer |  | Fall |  |  |  |
|  | TG | TL | TG | TL | TG | TL |
| 9 | 0.0 | 0.0 | 100.00 |  | 15.6 | 0.0 |
| 11 | 0.0 | 7.1 | 0.0 |  | 0.0 | 7.1 |
| 15 | 25.0 | 7.1 | 0.0 |  | 21.1 | 7.1 |
| 16 | 43.8 | 57.1 | 0.0 |  | 36.8 | 57.1 |
| 17 | 25.0 | 28.6 | 0.0 |  | 21.1 | 28.6 |
| 18 | 6.3 | 0.0 | 0.0 |  | 5.3 | 0.0 |
| No. of fish | 16 | 14 | 3 | 0 | 19 | 14 |
| Mean | 16.1 | 15.9 | 9.0 |  | 15.0 | 15.9 |
| SD | 0.89 | 1.51 | 0 |  | 2.79 | 1.51 |
| 2 |  |  |  |  |  |  |
| P |  |  |  |  |  | 385 |

Table 18. Percentage distribution of Atlantic salmon parr by length caught in pelagic experimental gill nets in trophogenic (TG) and tropholytic (TL) zones, in Conne Pond, summarized by seasons separately and combined. Also included are statistical comparisons in the trophogenic versus tropholytic categories (Wilcoxon two-sample test, $Z$ ).

| Fork <br> Length (cm) | Season |  |  |  | Total |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Summer |  | Fall |  |  |  |
|  | TG | TL | TG | TL | TG | TL |
| 10 | 0.0 | 14.3 |  |  |  | 14.3 |
| 11 | 0.0 | 28.6 |  |  |  | 28.6 |
| 12 | 33.3 | 42.9 |  |  | 33.3 | 42.9 |
| 13 | 66.7 | 0.0 |  |  | 66.7 | 0.0 |
| 14 | 0.0 | 0.0 |  |  |  | 0.0 |
| 15 | 0.0 | 14.3 |  |  |  | 14.3 |
| No. of fish | 3 | 7 | 0 | 0 | 3 | 7 |
| Mean | 12.7 | 11.9 |  |  | 12.7 | 11.9 |
| SD | 0.57 | 1.57 |  |  | 0.58 | 1.57 |
| Z |  |  |  |  |  |  |
| P |  |  |  |  |  |  |

Table 19. Percentage distribution of Brook trout by length caught in pelagic experimental gill nets in trophogenic (TG) and tropholytic (TL) zones, in Conne Pond, summarized by seasons separately and combined. Also included are statistical comparisons in the trophogenic versus tropholytic categories (Wilcoxon two-sample test, $Z$ ).

| Fork <br> Length (cm) | Season |  |  |  | Total |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Summer |  | Fall |  |  |  |
|  | TG | TL | TG | TL | TG | TL |
| 18 | 0.0 | 14.3 |  | 0.0 | 0.0 | 12.5 |
| 19 | 8.3 | 9.5 |  | 0.0 | 8.3 | 8.3 |
| 20 | 0.0 | 0.0 |  | 0.0 | 0.0 | 0.0 |
| 21 | 16.7 | 23.8 |  | 66.7 | 16.7 | 29.2 |
| 22 | 16.7 | 19.1 |  | 0.0 | 16.7 | 16.7 |
| 23 | 0.0 | 9.5 |  | 0.0 | 0.0 | 8.3 |
| 24 | 8.3 | 0.0 |  | 0.0 | 8.3 | 0.0 |
| 25 | 8.3 | 0.0 |  | 0.0 | 8.3 | 0.0 |
| 26 | 16.7 | 4.8 |  | 33.3 | 16.7 | 8.3 |
| 27 | 8.3 | 4.8 |  | 0.0 | 8.3 | 4.2 |
| 28 | 8.3 | 9.5 |  | 0.0 | 8.3 | 8.3 |
| 32 | 8.3 | 4.8 |  | 0.0 | 8.3 | 4.2 |
| No. of fish | 12 | 21 | 0 | 3 | 12 | 24 |
| Mean | 24.4 | 22.5 |  | 22.6 | 24.4 | 22.5 |
| SD | 3.65 | 3.75 |  | 2.89 | 3.65 | 3.60 |
| 2 |  |  |  |  |  |  |
| P |  | 46 |  |  |  |  |

Table 20. Percentage distribution of Atlantic salmon pary by age caught in benthic (B) and pelagic (P) experimental gill nets in Junction Pond, summarized by seasons separately and combined. Also included are statistical comparisons in the benthic versus pelagic categories (Wilcoxon two-sample test, $Z$ ).

| Age (yr) | Season |  |  |  | Total |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Summer |  | Fall |  |  |  |
|  | B | $\mathbf{P}$ | B | P | B | P |
| 0+ | 1.8 | 0 | 10.0 |  | 5.6 | 0 |
| 1+ | 28.2 | 13.2 | 22.2 |  | 25.5 | 13.2 |
| 2+ | 30.6 | 39.6 | 44.4 |  | 36.8 | 39.6 |
| $3+$ | 24.1 | 35.9 | 22.2 |  | 23.2 | 35.9 |
| 4+ | 14.4 | 11.3 | 1.1 |  | 8.3 | 11.3 |
| $5+$ | 0.9 | 0 | 0 |  | 0.5 | 0 |
| No. of fish | 216 | 53 | 180 | 0 | 396 | 53 |
| Mean | 2.2 | 2.5 | 1.82 |  | 2.1 | 2.5 |
| SD | 1.08 | 0.87 | 0.93 |  | 1.04 | 0.87 |
| Z |  |  |  |  |  |  |
| P |  | 126 |  |  |  | . 05 |


| Age (Yr) | Season |  |  |  | Total |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Summer |  | Fali |  |  |  |
|  | B | P | B | P | B P |  |
| 1+ | 1.3 | 0 | 16.9 |  | 9.0 | 0 |
| 2+ | 20.5 | 33.3 | 48.1 |  | 34.2 | 33.3 |
| 3+ | 48.7 | 66.7 | 35.1 |  | 41.9 | 66.7 |
| 4+ | 28.2 | 0 | 0 |  | 14.2 | 0 |
| $8+$ | 1.3 | 0 | 0 |  | 0.7 | 0 |
| No. of fish | 78 | 6 | 77 | 0 | 155 | 6 |
| Mean | 3.1 | 2.7 | 2.2 |  | 2.7 | 2.7 |
| SD | 0.93 | 0.52 | 0.70 |  | 0.94 | 0.52 |
| Z | 1.35 |  |  |  |  |  |
| P | 0.177 |  |  |  |  | 98 |

Table 22. Percentage distribution of Atlantic salmon parr by age caught in benthic ( $B$ ) and pelagic ( $P$ ) experimental gill nets, in Conne Pond, summarized by seasons separately and combined. Also included are statistical comparisons in the benthic versus pelagic categories (Wilcoxon two-sample test, $z$ ).

| Age (y) | Season |  |  |  | Total |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Summer |  | Fall |  |  |  |
|  | B | P | B | $\mathbf{P}$ | B | $P$ |
| 1+ | 8.5 | 0.0 | 3.7 |  | 7.5 | 0.0 |
| 2+ | 45.3 | 40.0 | 51.9 |  | 46.6 | 40.0 |
| 3+ | 41.5 | 50.0 | 44.4 |  | 42.1 | 50.0 |
| 4+ | 2.8 | 10.0 | 0.0 |  | 2.3 | 10.0 |
| 5+ | 0.9 | 0.0 | 0.0 |  | 0.8 | 0.0 |
| 6+ | 0.9 | 0.0 | 0.0 |  | 0.8 | 0.0 |
| No. of fish | 106 | 10 | 27 | 0 | 133 | 10 |
| Mean | 2.5 | 2.7 | 2.4 |  | 2.4 | 2.7 |
| SD | 0.81 | 0.67 | 0.57 |  | 0.76 | 0.67 |
| Z |  |  |  |  |  |  |
| P |  |  |  |  |  | 264 |


| Age (y) | Season |  |  |  | Total |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Summer |  | Fali |  |  |  |
|  | B | $\mathbf{p}$ | B | P | B | P |
| $2+$ | 5.0 | 0.0 | 7.4 | 0.0 | 5.4 | 0.0 |
| 3+ | 42.5 | 25.0 | 44.4 | 0.0 | 42.9 | 22.9 |
| 4+ | 40.0 | 62.5 | 44.4 | 100.0 | 40.8 | 65.7 |
| 5+ | 12.5 | 6.3 | 3.7 | 0.0 | 10.9 | 5.7 |
| 6+ | 0.0 | 6.3 | 0.0 | 0.0 | 0.0 | 5.7 |
| No. of fish | 120 | 32 | 27 | 3 | 147 | 35 |
| Mean | 3.6 | 3.9 | 3.4 | 4.0 | 3.6 | 3.9 |
| SD | 0.77 | 0.76 | 0.70 | 0.00 | 0.76 | 0.73 |
| 2 | 2.04 |  | 1.46 |  | 2.48 |  |
| P | 0.041 |  | 0.145 |  | 0.013 |  |

Table 24. Percentage distribution of Atlantic salmon parr by age caught in benthic experimental gill nets in littoral (L) and nonlittoral (N) zones in Junction Pond, summarized by seasons separately and combined. Also included are statistical comparisons in the littoral versus nonlittoral categories (Wilcoxon two-sample test, $Z$ ).

| Age (yr) | Season |  |  |  | Total |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Summer |  | Fail |  |  |  |
|  | L | N | L | N | L | N |
| 0+ | 2.6 | 0 | 15.8 | 0 | 8.2 | 0 |
| $1+$ | 33.1 | 16.1 | 31.6 | 6.1 | 32.5 | 10.9 |
| 2+ | 33.1 | 24.2 | 37.7 | 56.1 | 35.1 | 40.6 |
| 3+ | 21.4 | 30.7 | 14.0 | 36.4 | 18.3 | 33.4 |
| 4+ | 9.7 | 25.8 | 0.9 | 1.5 | 6.0 | 13.3 |
| $5+$ | 0 | 3.2 | 0 | 0 | 0 | 1.6 |
| No. of fish | 154 | 62 | 114 | 66 | 268 | 128 |
| Mean | 2.0 | 2.8 | 1.5 | 2.3 | 1.8 | 2.5 |
| SD | 1.02 | 1.11 | 0.95 | 0.61 | 1.02 | 0.91 |
| Z |  |  |  |  |  | 47 |
| P |  | 000 |  | 000 |  | 000 |

Table 25. Percentage distribution of Brook trout by age caught in benthic experimental gill nets in littoral (L) and nonlittoral (N) zones in Junction Pond, summarized by seasons separately and combined. Also included are statistical comparisons in the littoral versus nonlittoral categories (Wilcoxon two-sample test, $z$ ).

| Age (yr) | Season |  |  |  | Total |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Summer |  | Fal1 |  |  |  |
|  | L | N | L | N | L | N |
| 1+ | 0 | 1.8 | 17.1 | 16.7 | 10.5 | 8.2 |
| 2+ | 31.8 | 16.1 | 51.4 | 45.2 | 43.9 | 28.6 |
| 3+ | 40.9 | 51.8 | 31.4 | 38.1 | 35.1 | 45.9 |
| 4+ | 27.3 | 28.6 | 0 | 0 | 10.5 | 16.3 |
| 8+ | 0 | 1.8 | 0 | 0 | 0 | 1.0 |
| No. of fish | 22 | 56 | 35 | 42 | 57 | 98 |
| Mean | 3.0 | 3.2 | 2.1 | 2.2 | 2.5 | 2.8 |
| SD | 0.79 | 0.97 | 0.69 | 0.72 | 0.83 | 0.99 |
| Z | 0.90 |  | 0.47 |  | 2.03 |  |
| P | 0.371 |  | 0.636 |  | 0.042 |  |

Table 26. Percentage distribution of Atlantic salmon parr by age caught in benthic experimental gill nets in littoral ( L ) and nonlittoral (N) zones, in Conne Pond, summarized by seasons separately and combined. Also included are statistical comparisons in the littoral versus nonlittoral categories (Wilcoxon two-sample test, $\mathbf{Z}$ ).

| Age (y) | Season |  |  |  | Total |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Summer |  | Fall |  |  |  |
|  | $\bar{L}$ | N | L | N | L | N |
| 1+ | 8.8 | 6.7 | 4.4 | 0.0 | 7.9 | 5.3 |
| 2+ | 44.0 | 53.3 | 52.2 | 50.0 | 45.6 | 52.6 |
| $3+$ | 42.9 | 33.3 | 43.5 | 50.0 | 43.0 | 36.8 |
| 4+ | 3.3 | 0.0 | 0.0 | 0.0 | 2.6 | 0.0 |
| 5+ | 1.1 | 0.0 | 0.0 | 0.0 | 0.9 | 0.0 |
| 6+ | 0.0 | 6.7 | 0.0 | 0.0 | 0.0 | 5.3 |
| No. of fish | 91 | 15 | 23 | 4 | 114 | 19 |
| Mean | 2.4 | 2.5 | 2.4 | 2.5 | 2.4 | 2.5 |
| SD | 0.75 | 1.13 | 0.58 | 0.58 | 0.72 | 1.02 |
| z |  |  |  |  |  |  |
| $\mathbf{P}$ |  |  |  |  |  | 13 |

Table 27. Percentage distribution of Brook trout by age caught in benthic experimental gill nets in littoral (L) and nonlittoral (N) zones, in Conne Pond, summarized by seasons separately and combined. Also included are statistical comparisons in the littoral versus nonlittoral categories (Wilcoxon two-sample test, $z$ ).

| Age (y) | Season |  |  |  | Total |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Summer |  | Fall |  |  |  |
|  | L | N | $\bar{L}$ | N | L | N |
| 2+ | 5.1 | 4.9 | 10.0 | 0.0 | 6.3 | 4.4 |
| 3+ | 52.5 | 32.8 | 50.0 | 28.6 | 51.9 | 32.4 |
| 4+ | 33.9 | 45.9 | 35.0 | 71.4 | 34.2 | 48.5 |
| 5+ | 8.5 | 16.4 | 5.0 | 0.0 | 7.6 | 14.7 |
| No. of fish | 59 | 61 | 20 | 7 | 79 | 68 |
| Mean | 3.5 | 3.7 | 3.4 | 3.7 | 3.4 | 3.7 |
| SD | 0.73 | 0.79 | 0.75 | 0.49 | 0.73 | 0.77 |
| z |  |  |  | 1.28 |  |  |
| P |  |  |  | 0.201 |  | 11 |

Table 28. Percentage distribution of Atlantic salmon parr by age caught in pelagic experimental gill nets in trophogenic (TG) and tropholytic (TL) zones in Junction pond, summarized by seasons separately and combined. Also included are statistical comparisons in the trophogenic versus tropholytic categories (Wilcoxon two-sample test, Z).

| Age (yr) | Season |  |  |  | Total |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Summer |  | Fall |  |  |  |
|  | TG | TL | TG | TL | TG | TL |
| 1+ | 25.0 | 8.1 |  |  | 25.0 | 8.1 |
| 2+ | 25.0 | 45.9 |  |  | 25.0 | 45.9 |
| 3+ | 50.0 | 29.7 |  |  | 50.0 | 29.7 |
| 4+ | 0 | 16.2 |  |  | 0 | 16.2 |
| No. of fish | 16 | 37 | 0 | 0 | 16 | 37 |
| Mean | 2.3 | 2.5 |  |  | 2.3 | 2.5 |
| SD | 0.86 | 0.87 |  |  | 0.86 | 0.87 |
| $z$ |  |  |  |  |  |  |
| P |  |  |  |  |  | 17 |

Table 29. Percentage distribution of Atlantic salmon parr by age caught in pelagic experimental gill nets in trophogenic (TG) and tropholytic (TL) zones, in Conne Pond, summarized by seasons separately and combined. Also included are statistical comparisons in the trophogenic versus tropholytic categories (Wilcoxon two-sample test, Z).

| Age (y) | Season |  |  |  | Total |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Summer |  | Fall |  |  |  |
|  | TG | TL | TG | TL | TG | T'L |
| 2+ | 33.3 | 42.9 |  |  | 33.3 | 42.9 |
| 3+ | 66.7 | 42.9 |  |  | 66.7 | 42.9 |
| 4+ | 0.0 | 14.3 |  |  | 0.0 | 14.3 |
| No. of fish | 3 | 7 | 0 | 0 | 3 | 7 |
| Mean | 2.7 | 2.7 |  |  | 2.7 | 2.7 |
| SD | 0.58 | 0.76 |  |  | 0.58 | 0.76 |
| 2 |  |  |  |  |  |  |
| P |  |  |  |  |  | 000 |

Table 30. Percentage distribution of Brook trout by age caught in pelagic experimental gill nets in trophogenic (TG) and tropholytic (TL) zones, in Conne Pond, summarized by seasons separately and combined. Also included are statistical comparisons in the trophogenic versus tropholytic categories (Wilcoxon two-sample test, Z).

| Age (Y) | Season |  |  |  | Total |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Summer |  | Fall |  |  |  |
|  | TG | TL | TG | TL | TG | TL |
| 3+ | 16.7 | 30.0 |  | 0.0 | 16.7 | 26.1 |
| 4+ | 66.7 | 60.0 |  | 100.0 | 66.7 | 65.2 |
| 5+ | 8.3 | 5.0 |  | 0.0 | 8.3 | 4.4 |
| 6+ | 8.3 | 5.0 |  | 0.0 | 8.3 | 4.4 |
| No. of fish | 12 | 20 | 0 | 3 | 12 | 23 |
| Mean | 4.1 | 3.9 |  | 4.0 | 4.1 | 3.9 |
| SD | 0.79 | 0.75 |  | 0.00 | 0.79 | 0.69 |
| $\mathbf{z}$ |  |  |  |  |  |  |
| P |  |  |  |  |  |  |



Fig. 1. Map of Northeast River, Placentia, Newfoundland showing the location of Junction Pond.


Fig. 2. Map of Conne River, Newfoundland showing the location of Conne' Pond.


Fig. 3. Bathymetric map of Junction Pond, Northeast River, Newfoundland. Contours are in m.



Fig. 5. Percentage hypsographic curves for Junction Pond and Conne Pond, Newfoundland.


Fig. 6. Percent distribution of Atlantic salmon parr, Brook trout, and Arctic charr captured in night and day sets of benthic and pelagic experimental gill nets in Junction Pond, Newfoundiland. Number above each bar represents the number of fish.


Fig. 7. Percent distribution of Atlantic salmon parr and Brook trout captured in night and day sets of benthic and pelagic experimental gill nets in Conne Pond, Newfoundland. Number above each bar represents the number of fish.


Fic. 8. Percent distribution of Atlantic salmon parr captured in night and day sets of benthic experimental gill nets in Junction Pond, Hewfoundland, by inaividual contour depth. Number above each bar represents the number of fish.


Fig. 9. Percent distribution of Atlantic salmon parr, Brook trout, and Arctic charr captured in benthic experimental gill nets in Junction Pond, Newfoundland, partitioned into littoral and nonlittoral zones. Number above each bar represents the number of fish.


Fig. 10. Percent distribution of Erook trout captured in night and day sets of benthic experimental gill nets in lunction Pond, Newfounciland, by individual contour depth. l'umber above each bar represents the number of fish.


Fig. 11. Percent distribution of Atlantic slamon parr captured in night and day sets of benthic experimental gill nets in Conne Pond, Hewfoundland, by individual contour depths. Number above each bar represents the number of fish.


Fig. 12. Percent distribution of Atlantic salmon parr and Erook trout captured in benthic experimental gill nets in Conne Pond, liewfoundland, partitioned into littoral and nonlittoral zones. Number above each bar represents the number of fish.


Fig. 13. Percent distribution of Brook trout captured in night and day sets of benthic experimental gill nets in Conne Pond, liewfoundland, by individual contour depths. liumber above each bar represents the number of fish.

# Variation in Atlantic Salmon Juvenile Densities with Stream Gradient 

## by

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#### Abstract

Densities of age-1+ and age-2+ Atlantic salmon were consistently maximal in moderate gradient sections of streams and nil or minimal in very low or high gradient sections at varying annual indices of escapement. In the Stewiacke River, moderate gradient sections in close proximity to the head of tide had consistently higher densities whereas densities more proximate to the head of tide in the St. Mary's River were not higher. Density estimation models based on gradient and distance may be used to more precisely and accurately estimate standing populations of parr. Models adjusted to optimium densities for each stream gradient have the potential to be used for river specific target spawning requirements, site specific assessments and less precisely for environmental impact analysis on larger drainages.


## RÉsumé

A des indices annuels variables d'échappées, la densité de saumons de l'Atlantique d'âges $1+$ et $2+$ était uniformément maximale dans les tronçons de cours d'eau à pente d'écoulement modérée, et nulle ou minimale dans les tronçons à très faible pente d'écoulement ou à pente d'écoulement élevée. Dans la rivière Stewiacke, les tronçons à pente d'écoulement modérée a proximité de la limite de la marée abritaient régulièrement des densités plus élevées de saumon tandis que les densités dans les eaux d'aval de la rivière St. Mary's n'étaient pas régulièrement élevées. Des modèles d'estimation de la densité basés sur la pente d'écoulement et la distance peuvent être utilisés pour déterminer avec plus de précision la biomasse de tacons. Des modèles ajustés en fonction des densités optimales pour chaque pente d'écoulement peuvent servir à determiner les exigences en matière du nombre cible de géniteurs particulières à une rivière, les évaluations de sites et, de façon moins prècise, l'analyse d'un impact environnemental dans de plus grands bassins versant.

## INTRODUCTION

Distribution of juvenile Atlantic salmon within streams has been associated with a variety of physical attributes of streams such as water depth (Egglishaw and Shackley 1982), water velocity (Symons and Heland 1978) and substrate size (Rimmer et al. 1984). These attributes are inter-dependant and functionally related in hydrological models such as the Chezy or Manning formulae which include slope or gradient as an important additional variable (Dunne and Leopold 1978). The possibility that stream gradient could account for a considerable amount of the variance in juvenile densities and distribution was postulated by Symons and Heland (1978).

Salmonid population distributions have been described as contagious (Bohlin et al. 1981) and stratified sampling with respect to biotope was suggested as one way to reduce the rather large (0.70) coefficient of variation among locations. Multi-stage sampling designs have also been suggested as one way of reducing sampling variance when attempting to estimate the standing population of fish in small streams (Hankin, 1984). Stream gradient, a physical attribute potentially differentiating habitat preference for juvenile salmonids, is measurable both proximately (in-stream surveys) and by remote sensing and therefore suitable for use in widely applicable distribution models. Attempts to relate juvenile salmon densities to gradient were unsuccessful (Symons and Heland 1978; Kennedy and Strange 1982). However, area-weighted gradient of continuous ecologically similar units explained 79 \% of the variation at eight locations on the Stewiacke River, N.S. (Amiro 1984).

As a requisite to the collection and use of gradient as a basic parameter for the evaluation of habitat suitability and perhaps capacity to produce juvenile Atlantic salmon, this paper examines the hypothesis that juvenile Atlantic salmon, collected in the Stewiacke, 1984-88 (Fig. 1) and St. Mary's rivers, 1985-86, (Fig. 2) were systematically distributed with respect to stream gradient.

## DATA AND METHODS

## Proximate Surveys

Proximate stream surveys for electrofishing sites were conducted between contour limits (remote reaches), determined from orthophotographic maps at locations where the 5.0 m contour lines cross streams, and located in the feild using ortho-photo maps and color aerial photographs.

Locations for proximate surveys were randomly selected from remote reaches stratified by ortho-grade into intervals summarized in a matrix of 0.0-0.12, 0.121-0.249, 0.25 steps to 3.49, 3.5-5.0, and
$>5.0 \%$ ortho-grade and 10 km distance intervals from the head of tidal influence. Selection of proximate locations was weighted by the proportion of the total water surface area represented by a distance and gradient cell.

Surveying proceeded with regard to ecological unit types, termed proximate reaches in the manner similiar to Amiro (1984). Proximate reaches had similar surface, bottom and width characteristics. Widths were measured at the beginning, end and every 30 m or midpoint of the reach length. Widths and lengths were measured (to $10^{-2} \mathrm{~m}$ ) with a fiber measuring tape. Depths (to $10^{-3} \mathrm{~m}$ ) were measured with a survey rod at one quarter intervals across width transects. Total area for remote reaches was the sum of all proximate reaches calculated as the product of the average width for the beginning, every 30 m and ending point of each proximate reach and the proximate reach length.

The water surface grade of each proximate reach was determined from distance between points at the mid-stream depth locations using an engineering auto-level and standard levelling techniques. Gradients of each remote reach and potential electrofishing site (a combination of consecutive proximate reaches) were weighted according to the area of each contributing proximate reach and termed area-weighted-percent-surface-grade (AWSG) . Areas and AWSG's for use in population estimates were calculated by omitting proximate reaches with grades greater than $5 \%$, i.e. white water chutes and falls.

Surveys were conducted once at each site during 1984, 1985 or 1986. New sites were added and others dropped in an attempt to distribute the sampling with respect to the area by distance-gradient matrix for the entire river system.

## Electrofishing

An electrofishing site consisted of single or consecutive proximate reaches totaling aproximately 100 m of stream length, the exact limits of which were suggested by the ecological unit types. The cumulative length of consecutive electrofishing sites usually equalled the remote reach length (a location). However, larger remote reaches were subdivided into sections and subsampled to make smaller areas allowing one day for marking and one day for capture at a location.

Electrofishing equipment consisted of a shore-mounted, generator-driven transformer, single anode, dip nets ( 2 max) and lipseine (Elson 1962). The four-person crew fished in a streambank to streambank pattern from bottom to top markers of the section. Locations in the main river where wading was not possible were fished with a generator-driven double-boom electrofishing boat. ${ }^{1}$

[^0]Fish captured in a site were removed and held in an instream flow-through holding box for wadable sites and in an on-board live box for electro-boat sites, until the site was fished over once. Fish were anaesthetized (MS222) and those larger than 5.0 cm were marked by squaring the tip of either the upper or lower caudal, or right or left ventral, or anal fin. Fish were allowed to recover from the effects of the anaesthetic in a separate compartment of the holding box and were then dispersed throughout the site. Population estimates for specific sections (electrofishing sites) were made possible by differential fin-clipping. One to three days generally passed between marking and sampling runs.

All fish larger than 5.0 cm were measured and recorded to a 0.5 cm fork length interval. Scale sampling proportionate to length frequency classes enabled ageing by scale reading and therefore population estimates by age-classes.

Adjusted Petersen population estimates (Ricker 1975) for each site were calculated from mark-recapture data; mortalities were added after estimates were made. Minimum population estimates for sites with zero recaptures of marked fish were conservatively derived from the total catch. A Schnabel multiple-mark-recapture estimate was calculated for one electro-boat site where two separate days of recapture effort occurred.

Juvenile data were collected in the Stewiacke River (Fig 1) during 1984 to 1988 and from the Saint Mary's River (Fig 2) during 1985 and 1986 and are reported by Amiro et al. (1989).

Gradient terms were surface grades weighted by contributing areas and sine ${ }^{-1}$ converted to normalize their distribution. Distance values were divided by 100 for use in the analysis.

Data were analyzed a priori by least squares regression analysis (SYSTAT ${ }^{2}$ ). Smoothing plots were distance weighted least squares fits.

## RESULTS

## Stewiacke Parr Distribution

A total of 174 density estimates for age-1+ and age-2+ parr with associated area-weighted-percent-surface-grades (AWSG) was determined in the Stewiacke River from the years 1984 to 1988 (Tables
${ }^{2}$ SYSTAT:The system for Statistics. Evanston, IL:

4 to 8, in Amiro et al. 1989). Densities were distributed in a domed shaped distribution with respect to grade as indicated by a smoothing curve fitted to the data (Fig. 3).

A hypothesis that total (age-1+ and $2+$ ) parr densities for all years were distributed in a quadratic fashion with respect to sine ${ }^{-}$ ${ }^{1}$ converted area-weighted-percent-surface-grade (ASAWSG) was tested by least squares regression analysis for the model;

Total parr density $=$ Constant + ASAWSG + ASAWSG $^{2} \quad \mid$ Model 1
and the null hypothesis rejected ( $\mathrm{F}=5.253$, $\mathrm{p}<0.006$ ). The regression coefficients were significant ( $p=0.003$ and 0.008 ) and the constant not significant ( $p=0.211$ ). Examination of individual years indicated that only 1986 was significant ( $p=0.053$ ). Distribution of residuals and probability plots for Model 1 were unacceptable indicating the data did not meet the assumptions of the analysis.

The residuals plotted against the estimates of parr densities for Model 1 revealed that variance increased with the size of the estimate, and that log conversion of the density variable was appropriate. Densities +1 ( to accmodate zero density values) were Ln transformed and the model ;

$$
\text { Ln (density+1) }=\text { Constant }+ \text { ASAWSG }+ \text { ASAWSG }^{2} \quad \mid \text { Model } 2
$$

for combined years was highly significant ( $\mathrm{F}=25.024$; $\mathrm{p}=0.000$ ) and all coefficients were significant. The function was significant each year except 1988 and accounted for $3 \%$ (1988) to $54 \%$ (1985) of the variance as indicated by adjusted multiple $\mathrm{R}^{2}$ (Table 1). Residual and probability plots indicated acceptable compliance to the assumptions of the analysis but variance still increased with grade. Field observations suggested that similar habitats at increasing distance from the mouth of the river did not have similar fish densities. This concept was first explored to no avail through correlation analysis and
 grade and distance from tidal influence for all years (Fig. 4). This plot revealed the form that the density, grade and distance function might take.

Interaction with distance in quadratic models for dependant variables; age-1+, $2+$ and combined parr, was examined by computing the interaction terms of distance with grade and conducting multiple regression analysis. Interactions between distance and grade and/or grade ${ }^{2}$ provided best fits to the data. Models which included the distance and/or distance ${ }^{2}$ interaction were significant ( $p<0.001$ ) for each year and for age-1+ and age-2+ separately and explained up to 70\% of the variance (Table 2). Combining all years of total parr densities the equation;

```
LnDensity+1 = 1.05*Constant + 0.54*ASAWSG + 0.39*ASAWSG*DIST
    - 0.118*ASAWSG **DIST
    | Model 3
```

was highly significant ( $p<0.0001$ ), had all significant coefficients and accounted for $49 \%$ of the variance. Residuals and probability plots for this equation indicated excellent compliance with the assumptions of the analysis. A plot of Model 3 (Fig. 6) compared to the smoothing surface plot (Fig. 5) visually indicates the agreement of the model to the data.

## Saint Mary's Parr Distribution

A total of 64 density estimates for age-1+ parr and 58 estimates for age-2+ parr with associated-area-weighted-percent--surface-grades (AWSG) was determined in the St. Mary's River during July and August of 1985 (28 sites) and 1986 ( 36 sites) (Tables 35 and 36, in Amiro et al. 1989). With site 17 deleted, because of a zero density above a barrier, densities were again distributed in a domed shaped distribution with respect to sine ${ }^{-1}$ converted area-weighted-percent-surface-grade (ASAWSG) when fitted with a smoothing curve (Fig. 6).

Model 1 regressions using the St. Mary's data were significant for $1985(p=0.006)$ and $1986(p=0.002)$ and combined years ( $\mathrm{p}<0.0001$ ). Parameter fits were generally not significant with only the grade (ASAWSG) parameter significant in 1986 ( $\mathrm{p}=0.019$ ) and in the combined data ( $p=0.009$ ). The grade ${ }^{2}$ term was neither significant in 1985 ( $p=0.883$ ) nor in 1986 ( $p=0.096$ ). The adjusted multiple R2 indicated $28 \%$ of the variance was accounted for by the regression in 1985, 29\% in 1986 and 29\% in combined years.

Model 2 (ln total density) regression analysis (Fig. 7, Table 3) was significant in 1985 ( $p=0.003$ ) , 1986 ( $p<0.0001$ ) and for combined years ( $p=0.000$ ). The parameter estimate for grade (ASAWSG) was not significant ( $\mathrm{p}=0.073$ ) in 1985, but was significant ( $\mathrm{p}=0.001$ ) in 1986 and for the combined data ( $p<0.0001$ ). The grade ${ }^{2}$ term was significant in 1986 ( $p=0.014$ ) and in years combined ( $p=0.006$ ) but not in 1985. Adjusted multiple $R^{2}$ indicated $33 \%$ of the variance was accounted for in 1985, 47\% in 1986 and $42 \%$ in combined years.

The Model 3 distance*grade ${ }^{2}$ interaction term was not significant in any age-class, year or combination thereof. Additional models utilizing distance and/or distance interaction terms were tested and found significant in some instances. However none improved the $\mathrm{R}^{2}$ values substantially.

## DISCUSSION

Juvenile Atlantic salmon were spatially distributed with respect to water surface grade in both the Stewiacke and St. Mary's Rivers. Parr (age-1+ and $2+$ ) were found in higher densities in moderate to higher gradient sections of streams and were absent or at very low densities in lower gradient sections. The absence of juveniles in lower gradient sections could be both a function of
preference for moderate gradient sections or avoidance of predators in lower and slower flowing sections of streams which provide fewer sheltered locations because of smaller substrate size. However additional data collected at the time and reported by Amiro et al. (1989) indicated that parr were not present in lower gradient sections of streams unoccupied by Brook trout (Salvelinus fontinalis) or Brown trout (Salmo trutta). Occupation of pool habitat by juvenile Atlantic salmon is associated with low water conditions (Saunders and Gee 1964) and preferred velocities found in smaller pools (Morantz et al. 1987). Limited numbers of juveniles found in low gradient sections of the main Stewiacke River were located in short runs connecting longer stillwater sections.

It is arguable that a preference of parr for moderate gradient sections of streams with coarser substrate optimizes both foraging, by way of increased frequency of drift items in faster flowing water and protection from predators, by way of broken water surfaces and interstitial hiding opportunity. Locations of optimum micro habitat are more frequent in moderate gradient sections where substrate is coarse and flows varied among substrate. Locations of optimal nose velocity, a primary selection criteria for micro habitat, are more numerous in moderate gradient sections. Also, better visual isolation between cohorts may be afforded by the coarser substrate which leads to reduced intra-specific aggression and better energy budgeting.

Extremely high parr densities ( 257 parr $10^{-2} \mathrm{~m}^{2}$ )
were noted in one tributary (where gradient was moderate) in two years of data collection. Densities were high at these locations in spite of ample opportunity for parr to migrate to less densely occupied areas of the same stream but with lower gradient. This is evidence of preferred habitat selection.

Distance from the head of tide was a factor in the distribution of parr in the Stewiacke River but not in the St.Mary's River. This apparent contradiction raises an interesting question concerning the application of complete habitat surveys and parr distribution models that may attempt to estimate production potential of rivers. Accuracy of standing parr populations may be improved by factoring distance into the estimate for the Stewiacke but not for the St.Mary's. However, if a population were to be estimated for another river system, would distance be a factor? What additional attributes of rivers influence parr distribution? River specific differences such as run-timing and the distribution of suitable holding areas for adults at greater distances up-river in the St. Mary's than in the Stewiacke, may influence the distribution of egg depositions and therefore the distribution of juvenile salmon.

These models may be used to assess the present production of the Stewiacke or St. Mary's rivers. If a theoretical optimum curve were set then river specific populations and by way of survival values from egg to parr, river specific target spawning escapements may also be derived for any river where an optimum model was inferred. Target parr population values for individual reaches are available depending
on the proximate measured gradient structure for each reach. Thus the basis for a systematic site-specific monitoring procedure is possible for all sites. Proximate surveys conducted at separate time intervals would document and indicate the physical limitations for site-specific production of parr. Remote surveys in conjunction with optimum model parameters could be used to estimate the implications of degradation or mitigation on vast stretches of rivers without the costs associated with intensive proximate surveying. Losses, measured as recruits*spawner ${ }^{-1}$, due to environmental degradation are factored as reduced survival values from a theoretical egg deposition to the presmolt stage.

The step from proximate to remote modelling is not without a substantial loss in resolution associated with error in the measurement of area and gradient. A direct relationship between remote reaches and parr densities could not be derived without the use of the correction formula to adjust remote grades to proximate grades. However the loss of precision must be weighed against the required level of precision, finances, time available and the potential impact of the results.

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Table 1. Parameter estimates, probabilties, adjusted $R^{\wedge} 2$, variance ( $S$ ), $F$ value and df of MODEL 2 quadratic fits of combined Ln age-1+ and -2+ Atlantic salmon parr on arcsine area weighted surface grades for Stewiacke River electrofishing sites 1984 to 1988.

| Year | Dep.var. | Adj.mult. $\mathrm{R}^{\wedge} 2$ |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  | Indep.vars. | Coeff. | P(coeff.) | $\begin{aligned} & \mathrm{S} \\ & \mathrm{~F} \end{aligned}$ | df | P (reg.) |
| 1984 | LDENTOT | constant | 0.847 | 0.334 | 0.172 | 2,41 | 0.008 |
|  |  | ASAWSG | 0.871 | 0.007 | 0.646 |  |  |
|  |  | ASAWSG^2 | -0.079 | 0.003 | 5.456 |  |  |
| 1985 | LDENTOT | constant | 0.313 | 0.575 | 0.536 | 2,24 | 0.000 |
|  |  | ASAWSG | 1.098 | 0.000 | 0.595 |  |  |
|  |  | ASAWSG^2 | -0.091 | 0.000 | 15.904 |  |  |
| 1986 | LDENTOT | constant | 0.065 | 0.942 | 0.225 | 2,35 | 0.004 |
|  |  | ASAWSG | 1.107 | 0.001 | 0.662 |  |  |
|  |  | ASAWSG^2 | -0.094 | 0.001 | 6.386 |  |  |
| 1987 | LDENTOT | constant | 1.240 | 0.201 | 0.139 | 2,33 | 0.032 |
|  |  | ASAWSG | 0.618 | 0.094 | 0.769 |  |  |
|  |  | ASAWSG^2 | -0.040 | 0.229 | 3.815 |  |  |
| 1988 | LDENTOT | constant | 2.114 | 0.003 | 0.034 | 2,26 | 0.242 |
|  |  | ASAWSG | 0.356 | 0.157 | 0.500 |  |  |
|  |  | ASAWSG^2 | -0.027 | 0.233 | 1.498 |  |  |
| all | LDENTOT | constant | 0.724 | 0.040 | 0.217 | 2,171 | 0.000 |
|  |  | ASAWSG | 0.893 | 0.000 | 0.664 |  |  |
|  |  | ASAWSG^2 | -0.075 | 0.000 | 25.024 |  |  |

Table 2. Parameter estimates, probabilities, adjusted R^2, standard error (S) of the estimate, $F$ value, degrees of freedom and probability of $F$ of MODEL-3 quadratic fits of $\ln$ age-1+, $2+$ and total age-1 and $2+$ Atlantic salmon parr on arcsine converted area weighted surface grades for Stewiacke River electrofishing 1984 to 1988 and combined years.

| Year | Dep.var. | Indep.vars: | Coeff. | P(coef.) | $\begin{aligned} & \hline \mathrm{R}^{\wedge} 2 \\ & \mathrm{~S} \\ & \mathrm{~F} \end{aligned}$ | df | P(reg.) |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| 1984 | LDEN1 | constant | 2.154 | 0.000 | 0.414 | 2,41 | 0.000 |
|  |  | ASAWSG | 0.333 | 0.006 | 0.606 |  |  |
|  |  | ASAWSG^2*DIST. | -0.071 | 0.000 | 16.169 |  |  |
|  | LDEN2 | constant | -0.806 | 0.202 | 0.243 | 2,41 | 0.001 |
|  |  | ASAWSG | 0.636 | 0.000 | 0.873 |  |  |
|  |  | ASAWSG^2*DIST. | -0.064 | 0.004 | 7.894 |  |  |
|  | IDENTOT | constant | 1.962 | 0.000 | 0.364 | 2,41 | 0.000 |
|  |  | ASAWSG | 0.417 | 0.000 | 0.567 |  |  |
|  |  | ASAWSG^2*DIST | -0.069 | 0.000 | 13.283 |  |  |
| 1985 | LDEN1 | constant | 0.376 | 0.346 | 0.704 | 3,23 | 0.000 |
|  |  | ASAWSG | 0.644 | 0.000 | 0.476 |  |  |
|  |  | ASAWSG*DIST | 0.451 | 0.020 | 21.570 |  |  |
|  |  | ASAWSG^2*DIST | -0.121 | 0.000 |  |  |  |
|  | LDEN2 | constant | 0.287 | 0.571 | 0.442 | 2,24 | 0.001 |
|  |  | ASAWSG | 0.469 | 0.001 | 0.697 |  |  |
|  |  | ASAWSG^2*DIST | -0.071 | 0.000 | 11.311 |  |  |
|  | LDENTOT | constant | 0.622 | 0.102 | 0.740 | 3,23 | 0.000 |
|  |  | ASAWSG | 0.650 | 0.000 | 0.444 |  |  |
|  |  | ASAWSG*DIST | 0.446 | 0.014 | 25.635 |  |  |
|  |  | ASAWSG^2*DIST | -0.124 | 0.000 |  |  |  |
| 1986 | LDEN1 | constant | 0.844 | 0.098 | 0.631 | 3,34 | 0.000 |
|  |  | ASAWSG | 0.488 | 0.000 | 0.554 |  |  |
|  |  | ASAWSG*DIST | 0.616 | 0.004 | 22.083 |  |  |
|  |  | ASAWSG^2*DIST | -0.158 | 0.000 |  |  |  |
|  | LDEN2 | constant | -1.734 | 0.005 | 0.506 | 3,34 | 0.000 |
|  |  | A. :SG | 0.764 | 0.000 | 0.669 |  |  |
|  |  | A. SG*DIST | 0.609 | 0.017 | 13.643 |  |  |
|  |  | ASAWSG^2*DIST | -0.137 | 0.000 |  |  |  |
|  | LDEntot | constant | 0.591 | 0.144 | 0.656 | 3,34 | 0.000 |
|  |  | ASAWSG | 0.606 | 0.000 | 0.441 |  |  |
|  |  | ASAWSG*DIST | 0.545 | 0.002 | 24.558 |  |  |
|  |  | ASAWSG^2*DIST | -0.144 | 0.000 |  |  |  |

Table 2.Continued.

| Year | Dep.var. | Indep.vars. | Coeff. | P(coef.) | $\begin{aligned} & R^{\wedge} 2 \\ & S \\ & F \end{aligned}$ | df | P (reg.) |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| 1987 | LDEN1 | constant | 2.996 | 0.000 | 0.000 |  |  |
|  |  |  |  |  | 0.187 |  |  |
|  | LDEN2 | constant | -1.532 | 0.026 | 0.554 | 3,32 | 0.000 |
|  |  | ASAWSG | 0.718 | 0.000 | 0.500 |  |  |
|  |  | DIST | 1.725 | 0.041 | 15.464 |  |  |
|  |  | ASAWSG^2*DIST | -0.104 | 0.000 |  |  |  |
|  | LDENTOT | constant | 1.671 | 0.001 | 0.282 | 3,32 | 0.001 |
|  |  | ASAWSG | 0.445 | 0.000 | 0.702 |  |  |
|  |  | ASAWSG^2*DIST | -0.054 | 0.007 | 7.874 |  |  |
| 1988 | LDEN1 | constant | 2.231 | 0.000 | 0.383 | 2,26 | 0.001 |
|  |  | ASAWSG | 0.258 | 0.002 | 0.411 |  |  |
|  |  | ASAWSG^2*DIST | -0.054 | 0.000 | 9.685 |  |  |
|  | LDEN2 | constant | 0.035 | 0.925 | 0.465 | 2,26 | 0.000 |
|  |  | ASAWSG | 0.446 | 0.000 | 0.526 |  |  |
|  |  | ASAWSG^2*DIST | -0.042 | 0.013 | 13.155 |  |  |
|  | LDENTOT | constant | 2.174 | 0.000 | 0.478 | 2,26 | 0.000 |
|  |  | ASAWSG | 0.329 | 0.000 | 0.368 |  |  |
|  |  | ASAWSG^2*DIST | -0.055 | 0.000 | 13.818 |  |  |
| all | LDEN1 | constant | 1.148 | 0.000 | 0.320 | 3,170 | 0.000 |
|  |  | ASAWSG | 0.449 | 0.000 | 0.734 |  |  |
|  |  | ASAWSG*DIST | 0.422 | 0.001 | 28.151 |  |  |
|  |  | ASAWSG^2*DIST | . -0.121 | 0.000 |  |  |  |
|  | LDEN2 | constant | -0.921 | 0.001 | 0.404 | 3,170 | 0.000 |
|  |  | ASAWSG | 0.615 | 0.000 | 0.683 |  |  |
|  |  | ASAWSG*DIST | 0.362 | 0.003 | 40.051 |  |  |
|  |  | ASAWSG^2*DIST | -0.106 | 0.000 |  |  |  |
|  | LDENTOT | conexant | 1.050 | 0.000 | 0.494 | 3,170 | 0.000 |
|  |  | ASAWSG | 0.539 | 0.000 | 0.533 |  |  |
|  |  | ASAWSG*DIST | 0.399 | 0.000 | 57.390 |  |  |
|  |  | ASAWSG^2*DIST | -0.118 | 0.000 |  |  | - |

Table 3. Parameter estimates, probabilties, adjusted R^2, standard error of the estimate (S F-value and df of MODEL-2 quadratic fits of $L n$ total age-1 and $-2+$ Atlantic salmon parr den arcsine converted area weighted surface grades for St. Mary's River electrofishing for 1985, 1986 and combined years.

| Year | Dep.var. | Indep.vars. | R^2 |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  |  |  |  | S |  |  |
|  |  |  | Coeff. | P(coeff.) | F | df | P (reg.) |
| 1985 | LDENTOT | constant | 0.237 | 0.740 | 0.327 | 2,25 | 0.003 |
|  |  | ASAWSG | 0.513 | 0.073 | 0.551 |  |  |
|  |  | ASAWSG^2 | -0.028 | 0.279 | 7.565 |  |  |
| 1986 | LDENTOT | constant | 0.065 | 0.863 | 0.466 | 2,32 | 0.000 |
|  |  | ASAWSG | 0.582 | 0.001 | 0.494 |  |  |
|  |  | ASAWSG^2 | -0.038 | 0.014 | 15.845 |  |  |
| all | LDENTOT | constant | 0.119 | 0.718 | 0.420 | 2,60 | 0.000 |
|  |  | ASAWSG | 0.579 | 0.000 | 0.499 |  |  |
|  |  | ASAWSG^2 | -0.037 | 0.004 | 23.800 |  |  |



Fig. 1.Map of the Stewiacke River, Nova Scotia showing the 34 electrofishing locations.


Fig. 2. St. Mary's River with electrofishing site locations.


Figure 3. Smooth fit by distance weighted least squares of the log. total densities $\left(m^{-2} \star 100\right)$ of age-1+ and age-2+ Atlantic salmon parr and arcsine converted area-weighted percent surface grades determined in 174 sites in the Stewiacke River 1984 to 1988.


Figure 4. Smooth fit by distance weighted least squares of the log. total age-1+ and age-2+ Atlantic salmon parr densities $\left(10^{-}\right.$ ${ }^{2} \mathrm{~m}^{2}$ ), arcsine converted area-weighted percent surface grades and distances $\left(\mathrm{km} * 10^{-2}\right)$ determined in 174 sites in the Stewiacke River 1984 to 1988.


Figure 5. Plot of the Model 3 Atlantic salmon parr density distribution equation for the Stewiacke River.


Figure 6. Smooth fit by distance weighted least squares of log. total age-1+ and age-2+ Atlantic salmon parr densities ( $10^{-2} \mathrm{~m}^{2}$ ) and arcsine converted area-weighted percent surface grades determined in 63 sites in the St. Mary's River 1985 to 1986.


Figure 7. Scatter plot of log. total age-1+ and age-2+ Atlantic salmon parr densities ( $10^{-2} \mathrm{~m}^{2}$ ) and sine ${ }^{1}$ converted area-weighted percent surface grades determined by electrofishing at 28 sites in 1985 (O) and 35 sites in 1986 ( $\square$ ) in the St. Mary's River. Plots of Model 2 regression equations are shown for 1985 (upper), and 1986 (lower) and combined (middle).

# Influence of Stream Physico-Chemical Parameters on Fish Species Associations 

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#### Abstract

The objective of the paper is to show how stream variables influence distributions of fish species within a watershed.

Fitty-seven riffle sites on three watersheds, the Saint Croix, Medway and Gold, were investigated. Relative numbers of fish species inhabiting the study sites were estimated by electrofishing. Stream discharge, width, canopy, mean temperature, slope, and major chemical parameters were measured. All measurements and sampling were done at summer low flow.

The four major fish species inhabiting the three watersheds were creek chub (Semotilus atromaculatus), brook trout (Salvelinus fontinalis), Atlantic salmon (Salmo salar), and eel (Anguilla rostrata). Blacknose dace (Rhinichthys atratulus) and fallfish (Semotilus corporalis) were major species in the Saint Croix watershed, but were absent from the Gold and Medway.

From smallest tributaries to largest main stem rivers, creek chub, brook trout, blacknose dace, salmon, and eel - when present - exhibited sequential peaks in relative abundance. Temperature is considered an important variable influencing the distribution patterns.


Résumé

La présente étude porte sur l'influence que les variables d'un cours d'eau exercent sur la distribution des espèces de poisson à l'intérieur de bassins hydrographiques. A cette fin, on a étudié 57 seuils des bassins de la Sainte-Croix, de la Medway et de la Gold. La pêche à l'électricité a permis de déterminer l'abondance relative des espèces de poisson fréquentant les sites expérimentaux. On a quantifié le débit, la largeur du cours d'eau, le couvert forestier, la température moyenne, la pente et les principaux paramères chimiques. Toutes ces variables et l'échantillonnage ont été effectués pendant l'étiage estival.

Le mulet à cornes (Semotilus atromaculatus), l'omble de fontaine (Salvelinus fontinalis), le saumon de l'Atlantique (Salmo salar) et l'anguille d'Amérique (Anguilla rostrata) sont les quatre principales espèces de poisson qui fréquentent ces trois bassins hydrographiques. Le naseux noir (Rhinichthys atratulus) et la ouitouche (Semotilus corporalis), principales espèces fréquentant le bassin de la Sainte-Croix, sont absentes des rivières Gold et Medway.

Des plus petits tributaires aux grandes rivières principales, le mulet à cornes, l'omble de fontaine, le naseux noir, le saumon de l'Atlantique et l'anguille d'Amérique, lorsque présents, montraient des pointes séquentielles de l'abondance relative. On considère la température comme une variable importante qui influe sur les régimes de répartition.

## Introduction

Field studies have demonstrated reduced recruitment and production of Atlantic salmon in Nova Scotian streams of low pH (Lacroix et al., 1985; Lacroix, 1989). The Atlantic salmon, however, is not the sole inhabitant of these streams, but must coexist with other fish species with which there may be varying degrees of competition. In this paper, we compare the fish species assemblages inhabiting riffle habitat of two acidic watersheds with the assemblages present in a circumneutral watershed in southwestern New Brunswick.

Species richness of stream fish assemblages is related to stream size, and many studies have demonstrated greater species richness with increasing stream size within a watershed (e.g. Platts, 1979; Harrel et al., 1967; McNeely, 1986; Naiman et al., 1987). Several authors have concluded that increasing species richness with increasing stream size is primarily by species additions (Harrel et al., 1967; McNeely, 1986). Fewer species inhabit headwaters due to uncertainties in flow regime and possibly less habitat diversification, but these species, for the most part, persist throughout the watershed and there is little turnover of species throughout the length of the watershed. In contrast to the concept described above, ecological community analysis frequently assumes that differing ecological "optima" (e.g. temperature) exist for various species, and that abundance of each species rises and falls in a Gaussian-type curve on either side of its optimum (reviewed in Gauch, 1982). We examine the applicability of these concepts to distributions of fish species inhabiting riffles within three watersheds under mid-summer conditions.

## Methods

Three watersheds were surveyed. The Saint Croix is the major watershed in southwestern New Brunswick, with the main stem forming the Canada-U.S. boundary over much of its length. The Gold and Medway watersheds form parallel drainage systems in southwestern Nova Scotia - separated by the LaHave watershed (Fig. 1). Twenty-three sites were sampled on the Saint Croix watershed, 21 on the Medway, and 13 on the Gold. Detailed descriptions of the sites and geographic coordinates are available in Peterson and van Eeckhaute (1990). Practically all suitable, accessible sites on all three watersheds were sampled.

Measured physical characteristics of study sites were: discharge (integrated depth-velocity profiles), width of stream channel, percent canopy (five transects averaged, use of mirror in opaque cylinder), gradient (hand level), median midsummer temperature (max-min thermometers read bi-weekly, late June to midAugust) and bi-weekly temperature fluctuation. Measured chemical parameters were pH , Gran alkalinity, humic acids (absorption at 250 nm ), DOC, total dissolved $\mathrm{Al}, \mathrm{Ca}$, $\mathrm{Na}, \mathrm{Mg}, \mathrm{K}, \mathrm{SO}_{4}$ and Cl .

Fish assemblages in riffle habitats were censussed by back-pack electrofisher (Smith-Root® VIII-A). At most sites, a riffle was fished between the upstream and downstream pools bounding it - typically 30 to 50 m of stream length. In small brooks, the riffle lengths were often shorter, so two to three riffles were fished to make up an equivalent distance. For some large rivers, riffle areas fished did not extend the full width of the stream. The time required to fish a site depended upon the densities of fish present - sites with greater number of fish requiring more time. Usually approximately $1 / 2 \mathrm{~h}$ was required to fish a riffle. Each site was fished by a three-person team starting at the downstream limit and fishing upstream against the current, with the individual bearing the cathode in the centre and a person with dipnets on either side. Fish were identified and enumerated as collected, with fish of doubtful identity preserved for later verification. Sites were fished during summer base flows in late July and early August. Intermittent sites were fished when surface flow was present. Each site was fished 2 yr, between 1987 and 1989, and the data pooled for the 2 yr .

Certain species which were occasionally sampled were not included in the analyses because they were not deemed primarily riffle species. It was thus considered that the data did not reflect their actual distributions. Juvenile alewife (Alosa pseudoharengus) were occasionally sampled on their seaward migration, and were omitted from analyses. Chain pickerel (Esex nigen), yellow perch (Perca flavescens), sticklebacks (primarily Pungitius pungitius), white sucker (Catostomus commersoni) and banded killifish (Fundulus diaphanus) were all collected occasionally, but excluded from analyses as primarily pool species. The banded killifish was collected only at two large river sites on the Medway watershed where it may be invading riffle habitat occupied by cyprinids, excluded from these sites by acidity or distributional limits (Livingstone, 1953). Although widespread in lakes of the Saint Croix watershed, it was not sampled in any riffle habitat.

Sites were classified on the basis of species assemblages by a polythetic, divisive classification algorithm (TWINSPAN). A minimum eigen-value of 0.3 was used as the criterion for separating classes. Canonical discriminant function analysis was used to relate classes of sites to stream physico-chemical variables. Relative abundance curves were constructed for the principal species in relation to stream size, using the proportion of the total catch represented by a given species as a measure of relative abundance. This analysis does not require quantitative data, but does assume that the proportions of species caught reflect true proportions in the sampled habitat. Plots of relative abundance vs stream size were smoothed by using a three-point running average.

Results

The results presented here represent work performed over the past 5 yr on three watersheds in Scotia-Fundy Region - the Saint Croix, Gold and Medway watersheds (Fig. 1). The work demonstrates the manner in which stream sizerelated variables, the most important of which may be temperature, determine midsummer distributions of various fish species. It also demonstrates how acidity may modify the influence of stream size and/or temperature. I emphasize that the findings probably apply only to mid-summer conditions (late June to mid-August) when stream temperatures are maximal and stable.

The Saint Croix watershed (Fig. 1), although considered a soft-water system, is well enough buffered that pH is not a significant variable in the system. All tributaries measured to date have mid-summer pH levels ranging from 6.6 to 7.4. The value of 7.8 listed on the map is actually a tributary of the Saint John watershed. Even in spring freshet, I have never recorded a pH less than 6.0.

The Gold watershed features a much wider range of pH levels (Fig. 2). The highest pH's are found in tributaries in the northwestern part of the watershed, so that the upper main stem is about 5.8. With the addition of acidic waters from tributaries lower in the watershed, the main stem pH falls to 5.4 at the mouth.

The Medway watershed is the most generally acidic of the three (Fig 3), with tributaries ranging from 4.6 to 6.1. The main stem tends to be most acidic (ca. 5.1) in the upper part of the watershed, and rises to about 5.6 near the mouth.

As is expected, stream discharge increases with stream width (Fig. 4) - the relationship being linear on a full logarithmic transformation. The slope (ca. 1.7) of the line indicates that the river basins closely approximate shallow triangles. A perfect triangular basin cross-section would result in a slope of exactly 2 . The Gold and Medway were at slightly greater volumes when calibrated than was the Saint Croix.

The amount of canopy covering the stream is also a function of stream size, with most sites of less than $0.03 \mathrm{~m}^{3} / \mathrm{s}$ (width ca. 2 m ) being totally canopied (Fig. 5). Exceptions were a couple of sites where road construction had resulted in loss of canopy. Canopy opens progressively for streams of $0.03-1.0 \mathrm{~m}^{3} / \mathrm{s}$ (= widths of $>2-20 \mathrm{~m}$ ) and is practically nil for larger rivers.

Stream size also accounts for a fair amount of variance in median midsummer temperature of these stream systems (Fig. 6). The encircled Gold site is an intermittent rivulet that had ceased to flow with a concomitant rise in median temperature. The variance is greatest for median-sized streams ( $0.01-1.0 \mathrm{~m}^{3} / \mathrm{s}$ ) where canopy is variable, and where influences such as aquifers and shallow, unshaded deadwaters can influence temperature. Largest rivers tend to be more uniform. In general, temperature rises about $2.0^{\circ}$ per decade increase in log
discharge. The Nova Scotian streams (particularly the Medway) tend to be warmer than the Saint Croix tributaries, size for size. This may be due to many more lakes in the Medway watershed serving as heat sinks. Temperatures were also recorded from the Nova Scotian sites in a different summer which may have had higher mean air temperatures.

Vannote et al. (1980), in the river continuum hypothesis, postulated that diel temperature fluctuations would be maximal in middle order streams, corresponding to an opening of the canopy in these watersheds. Perhaps the variance tends to obliterate any such tendency (Fig. 7). Small streams drying to intermittence or with no canopy due to streamside clearing have very great diel fluctuations (encircled points). Also, lake outfalls tend to fluctuate less - the two lower points are such instances. If these two points are omitted, the R2 rises to 0.2.

If we lump the three watersheds together, $88 \%$ of the variance in pH can be accounted for by calcium and DOC levels (Fig. 8). However, inspection of the figure indicates that DOC (thus organic acidity) is the main influence for water with calcium levels less than $2-3 \mathrm{mg} / \mathrm{L}$. At higher calcium levels (or higher alkalinity), DOC does not influence pH . In other words, bicarbonate buffering overrides organic acidity. Thus for the Saint Croix, pH is controlled by bicarbonate alkalinity, while for the Nova Scotian streams it is largely controlled by organic acidity - at least under mid-summer conditions.

We began our analysis of fish communities in these three watersheds by classifying the sites with the use of the TWINSPAN program based on an iterative, reciprocal averaging procedure. This program clusters groups of sites on the basis of their fish assemblages, and identifies "key" species involved at each clustering step (Fig. 9). The first classification step separated the 52 sites supporting fish into two groups of 45 and 7 . The seven sites were larger brooks to large rivers which contained one or more of a complex of smallmouth bass, fallfish or common shiner. These sites are all warm Saint Croix sites - the Nova Scotian systems lacked these species.

The group of 45 sites contained eels or salmon or both. This group of 45 broke down into 11 small, cool stream sites and 34 larger, warmer sites, with brook trout and salmon the key species.

Five small, intermittent or very acidic sites contained no fish, and this could not be utilized in the classification program. They are shown connected by a dotted line at the top of Fig. 9. If we look at the final groupings to the left of Fig. 9, we see that they form a progression from smallest stream sites to large river sites from top to bottom of the dendrogram. Stream size factors obviously were influential in determining the fish assemblages. Acidic sites did not separate particularly well because the major abundant species were found at most appropriate sites in all three systems, and the TWINSPAN program uses relative abundance-weighted data. The influence of acidity was much more obvious when analyzing the invertebrate data from these sites - using close to 200 taxa rather than a dozen or so.

Since the species and sites were classified mainly in relation to stream size, we constructed "dominance" curves for each major species in the three watersheds (Fig. 10). These curves were generated by calculating the relative abundances ( $p$, the proportion of total fish caught belonging to the particular species). These relative abundances were plotted according to stream size, using estimated discharge. A 3-point running average was used to smooth the data. For the Saint Croix, the creek chub, brook trout, blacknose dace, Atlantic salmon, eel and fallfish curves form successive maxima as we progress from the smallest to largest stream sites. Atlantic salmon had highest relative abundances for streams with 0.1-1.0 $\mathrm{m}^{3} / \mathrm{s}$ summer base flow. The proportion of salmon might have been higher had not the main stem of the Saint Croix lost its salmon run several decades ago.
Nevertheless, we did collect small numbers of salmon at the most downstream site on the main Saint Croix (also above all the dams). We made no distinction between landlocked and sea-run juveniles in calculating these curves.

The curves for brook trout and salmon in the Gold Watershed show similar maxima to those obtained for the Saint Croix. Eels did not show a maximum, but continued to increase in relative abundance with increasing stream size. Blacknose dace and fallfish do not occur in the Gold and Medway watersheds. The dominance of creek chub in intermittent sites did not occur in the Gold system because these sites were very acidic.

In the Medway watershed, the eel dominated over a wide range of stream sites, probably due to decreased competition from other species in the acidic watershed. The relative abundance of salmon showed no distinct maximum because sites too acidic for the species survival were scattered throughout the watershed. Brook trout exhibited a curve similar to that for the other two systems.

For our final analysis, we plotted the relative abundances of the four main species on a temperature- pH grid (Fig. 11). The estimated low limiting pH is shown by the vertical dotted line. Atlantic salmon were most abundant for sites with median mid-summer temperatures of $18-22^{\circ} \mathrm{C}$, and were absent from sites of pH less than 4.9-5.0. Approximately six of the otherwise suitable sites for salmon on the Gold and Medway were too acidic for their occurrence (about 25\% of the suitable sites). The low limiting pH shown for eels is probably not correct and should be lower. The two very acidic sites with no eels were small cold brooks, thus were not good eel habitat on the basis of temperature.

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Fig. 1. (Inset) Outline map of New Brunswick and Nova Scotia with the Iocation of the three stream systems surrounded by dashed IInes. 1: Saint Croix Rlver; 2: Kentville, N.S.; 3: Wolfville, N.S.; 4: Gold River; 5: Medway River; 6: St. Andrews, N.B. (Main figure) Locations of various study sites on the Saint Crolx River system Indicated by arrows. Typical mid-summer pH levels are shown for each site.


Fig. 2. Locations of the study slies on the Gold River system are indicated by arrows. Typlcal mid-summer pH values are shown for each site.


Fig. 3. Locations of the study sites on the Medway River system are indicated by arrows. Typical mid-summer pH values are shown for each site.


Fig. 5. Percentage of stream width with leaf canopy is related to stream discharge. Closed circles represent Saint Croix sites, open clrcles are Medway sites, and open triangles are Gold sites. The two encircled Saint Crolx sltes had less canopy due to road construction modifications within 10 yr prior to the study. Line fitted by eye.


Fig. 4. Stream discharge is related to stream width for the three stream systems studied. The Medway and Gold regresslons did not differ significantly, so the discharges for these two stream systems were pooled for a common regression equation correlation.


Fig. 6. Mean mid-summer temperatures Increased with stream discharge. The regression equation is for all three systems pooled. The encircled Gold River data point was omitted from the regression as discharge had ceased part way through the summer.


Fig. 7. Bi-weekly ranges of mid-summer temperature fluctuations for the various sites were fitted to a parabolic curve (given In figure). Encircled points were omitted from the regression as these represent intermittent streams whose stagnant pools exhibited extreme mid-summer temperature fluctuations.


Fig. 8. Isopleths of pH for various combinations of DOC and $\mathrm{Ca}^{2+}$ are shown as derived from the multiple regression equation at top left of figure.


Fig. 9. Dendrogram of TWINSPAN classification of study sites. Numbers in parentheses are eigenvalues associated with corresponding blfurcation. Numbers whthout parentheses Indicate number of sites associated with each branch. Key species are Identified throughout the dendrogram. Dotted line connects five sites where no fish were found, hence could not be used in the analysis. Descriptors of each class of sites are to the left of the dendrogram.


Fig. 10. Relative abundance curves of the four major fish species are shown for the three watersheds in relation to discharge rate of the various study sites. p: proportion of total numbers of fish collected belonging to a given species. Top two panels: Saint Crolx watershed; third panel: Gold; bottom panel: Medway.


Fig. 11. Relative abundances of the four major fish species are shown for each study site in relation to site pH and median mid-summer temperature. Vertical line indicates estimated low limiting pH. A: brook trout; B: Atlantic salmon; C: creek chub; D : eel.

# Quantification of Productive Capacity of Stream Habitat for Atlantic Salmon 

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#### Abstract

Spawning habitat of Atlantic salmon (Salmo salar) is usually at the tail of pools on the upstream edge of a gravel bar, ideally with depths about 25 cm , surface water velocities of about $30-45 \mathrm{~cm} \cdot \mathrm{~s}^{-1}$, and with a substrate of irregularly shaped stones of cobble, pebble and gravel. Fine sediments are deleterious for survival of eggs and alevins. Underyearling salmon (<7. cm TL) are most common in shallow ( $<15 \mathrm{~cm}$ ) pebbly riffles. Older and larger parr are usually in riffles deeper than 20 cm , with surface water velocities between 50 and $65 \mathrm{~cm} \cdot \mathrm{~s}^{-1}$, but occupy pools and lentic waters where inter-specific competition and fish predation is low. Multiple linear regression models have identified coarse substrate, lower order rivers, and stable discharge as important variables regulating productive capacity of habitat. Water chemistry also has important effects.


## Résumé

Les frayères du saumon de l'Atlantique (Salmo salar) sont généralement situées dans les eaux d'aval des fosses, du côté amont de bancs de gravier. Les conditions idéales sont des profondeurs d'environ 25 cm , un débit de surface variant de 30 à $45 \mathrm{~cm} . \mathrm{s}^{-1}$, et un substrat composé de pierre, de galet et de gravier de formes irrégulières. Les sédiments fins sont nuisibles à la survie des oeufs et des alevins. Les tacons $0^{+}$( $<7 \mathrm{~cm}$ LT) sont plus abondants dans les seuils peu profonds ( $<15 \mathrm{~cm}$ ) à substrat de galet. Les tacons plus gros et plus âgés fréquentent surtout les seuils de plus de 20 cm de profondeur ou le débit de surface varie de 50 à $65 \mathrm{~cm} . \mathrm{s}^{-1}$, mais fréquentent aussi des fosses et des eaux stagnantes où la compétition inter-spécifique et le niveau de prédation sont faibles. Selon des équations de régression linéaire multiple, un substrat grossier, des rivières d'ordre inférieur et un débit stable constituent les variables importantes qui règlent la capacité de production d'un habitat. La chimie de l'eau est également importante.

## Introduction

The Department of Fisheries and Oceans' Policy for the Management of Fish Habitat (1986) has the objective of an achievement of an overall net gain of the productive capacity of fish habitat, and states that there should be no net loss of fish habitat. Problems arise in quantifying the productive capacity of habitat for individual species in different biogeoclimatic regions. Atlantic salmon has been chosen as a "test case" for describing required habitat of a species. As a consequence of two workshops organized by the Rawson Academy of Aquatic Sciences, CAFSAC has been asked to provide advice on the usefulness of measuring the following attributes in assessing suitable habitat: substrate, stream width, cover, velocity, ice scour, stream order, turbidity, total dissolved solids, pH , winter temperature, summer temperature, discharge and gradient; and to provide advice on the possible value of other attributes. Aspects of some of these variables are reviewed in the present paper.

General characteristics of salmon rivers are well recognized. Elson (1975) described these as having gradients moderately low ( $2 \mathrm{~m} \cdot \mathrm{~km}^{-1}$ ) to moderately steep ( $11.5 \mathrm{~m} \cdot \mathrm{~km}^{-1}$ ), with a substrate composed of assorted gravel, cobble and boulder. Adult salmon may migrate through the deep, slow stretches of larger, higher order, (sensu Horton, 1945) rivers, but spawning areas and habitat for production of juvenile salmon are in relatively shallow fast water areas, characterized by riffles and pools, more commonly found in second, third and fourth order rivers. Major differences among rivers in productivity of young salmon can be attributed to climate, species composition, and water chemistry (Egglishaw and Shackley, 1985; Gibson and Myers, 1986; Gibson and Haedrich, 1988). For maximum metabolic activity salmonids require fully oxygen saturated water (Fry, 1971). However, suitable habitat for the various riverine life stages also regulates production. Required freshwater habitat of the various life history stages can be classified generally as that suitable for, (i) spawning, (ii) for feeding during the growing period, and (iii) for overwintering. Habitat suitable as holding areas for adult salmon has not been covered in the following review, as it is assumed pools with suitable water velocities for holding adult migratory salmon are not normally limiting.

## Review and Discussion

## Spawning Habitat

White (1942) observed that generally salmon redds were built at the tail of pools on the upstream edge of a gravel bar, where the water depth was decreasing and the water current was accelerating.

Jones (1959) found in experimental stream tank studies that spawning took place in a depth of rarely more than 30 cm , and in a preferred surface water velocity of $31-46 \mathrm{~cm} \cdot \mathrm{~s}^{-1}$. Spawning ceased if surface water velocities were reduced to $5-8 \mathrm{~cm} \cdot \mathrm{~s}^{-1}$. He described spawning beds as ideally areas of gravel in riffles, sloping gently downstream with large pools at either end. Subsequent findings have been similar, related to more refined measurements, and size of the fish. Crisp and Carling (1989) observed a lower water velocity limit to spawning of $15-20 \mathrm{~cm} \cdot \mathrm{~s}^{-1}$, measured at 0.6 depth, and an upper limit of <c. 2.0
body lengths $s^{-1}$. Beland et al. (1982) found that redds were in a mean depth of 38 cm , (range $17-76 \mathrm{~cm}$ ), with mean water velocity, measured 12 cm above the substrate, of $53 \mathrm{~cm} \cdot \mathrm{~s}^{-1}$, (range $25-90 \mathrm{~cm} \cdot \mathrm{~s}^{-1}$ ). Heggberget et al. (1988) observed in the River Eira, Norway, with mean weight of salmon 4.8 kg , that mean depth of redds was 49.3 cm (SD 16.6 ), and mean mid-water velocity $38.6 \mathrm{~cm} \cdot \mathrm{~s}^{-1}$ (SD 18.2). In three other rivers (Alta, Gaula, and Driva) where mean weights of salmon were $9.0,4.0$, and 4.0 kg respectively, mean depth was 45.1 cm (SD 16.1) and mean mid-water velocity $55.5 \mathrm{~cm} \cdot \mathrm{~s}^{-1}$ (SD 22.4). Mean diameter of substrate in the surface layer of the redds was 9.2 cm (SD 6.8) in the Eira, and 11.5 cm (SD 5.5) in the Alta, Gaula, and Driva. Pratt (1968) found in Newfoundland with smaller salmon that preferred depths and water velocities for spawning in an experimental channel were about $21-25 \mathrm{~cm}$ and $31-43 \mathrm{~cm} \cdot \mathrm{~s}^{-1}$ (measured 12 cm off the bottom, although depths of $9-70 \mathrm{~cm}$ and velocities of $15-91 \mathrm{~cm} \cdot \mathrm{~s}^{-1}$ were used. Average depth of egg pits dug by grilse in the experimental channel was 14.2 cm . Larger females (two sea-winter salmon or older) bury their eggs deeper than grilse, the egg burial depth being related to length of the female (Crisp and Carling, 1989). A 5 kg female may make a depression up to 30 cm below the normal gravel bed, with the mound of gravel at the lower end of the redd covering batches of eggs to a depth of 60 cm (Jones, 1959). Egg survival was increased in a spawning channel by replacing the substrate with coarser gravel (Pratt et al., 1974). Sufficient porosity in the substrate is needed not only to provide flow for oxygen requirements and to remove metabolites of the developing eggs, but also to prevent entombment of emerging fry (Scrivener and Brownlee, 1989). Substrate also should be heterogeneous enough to provide stability. Peterson (1978) in New Brunswick streams found typical particle size distribution of spawning gravel (for a 15 cm deep core sample) was: cobble (22.2-256 mm), 40-60\%; pebble (2.2-22.2 mm), 40-50\%; coarse sand ( $0.5-2.2 \mathrm{~mm}$ ), $10-15 \%$; and fine sand ( $0.06-0.5 \mathrm{~mm}$ ), $0-3 \%$. Permeabilities were usually $<2000 \mathrm{~cm} \cdot \mathrm{hr}^{-1}$, with some gravels averaging $>5000 \mathrm{~cm} \cdot \mathrm{hr}^{-1}$. A permeability of about $1500 \mathrm{~cm} \cdot \mathrm{hr}^{-1}$ would result in a mean \% emergence of $2-3 \%$. Fry survival would likely be very low at permeabilities less than $1000 \mathrm{~cm} \cdot \mathrm{hr}^{-1}$, which corresponded to a sand content of $12-15 \%$ in the study areas. No emergence of fry was obtained in gravel with mean permeability as low as $600 \mathrm{~cm} \cdot \mathrm{hr}^{-1}$. Sand was found to reduce fry emergence in a gravel and sand composition, with a lower threshold of $8 \%$ for fine sand and $16 \%$ for coarse sand where effects were noticeable (Peterson \& Metcalfe, 1981). Also, time to first and median emergence, duration of emergence, stage of development at emergence, and within-substrate behaviour have been found to be each correlated positively with mean particle size and negatively with sediment loading rates (MacCrimmon and Gots, 1986). It has also been suggested that fertilization of eggs may be reduced by fine sediments clogging the micropyle (Billard, 1982).

## Habitat and Production of Young Salmon

Keenleyside (1962), by underwater observations in the Miramichi River in New Brunswick, noted that salmon fry, or underyearlings, were more abundant in the upper reaches of the river ( $5-30 \mathrm{~m}$ wide) than in the lower downstream sections ( $50-100 \mathrm{~m}$ wide), were in shallow water ( $\langle 50 \mathrm{~cm}$ ), and were commonly found in rapids where the bottom was made up of small gravel and stones. Parr were most common in the same sections of the river where fry were most abundant, except that parr generally stayed in deeper, faster water and over bottom that varied in size from small gravel to cobble and rubble, with occasional large
boulders. Elson (1967) observed similar distributions from collections made by electro-fishing. Other studies have corroborated these observations with further quantitative delimitations.

Symons and Heland (1978) by electro-fishing and by observations in laboratory streams found that underyearlings occurred in shallow ( $10-15 \mathrm{~cm}$ ) pebbly riffles, whereas yearling or older parr ( $>7 \mathrm{~cm} \mathrm{TL}$ ) did not occur in riffles shallower than 20 cm where there were no boulders. Similarly, in a Scottish stream, densities of underyearling salmon were positively correlated with depths between 0 and 19 cm , and inversely correlated with deeper water, whereas yearling salmon were in water $>15 \mathrm{~cm}$ deep (Egglishaw and Shackley, 1982). Highest densities of underyearling and yearling salmon were found in stream sections where the proportion of shallow water was highest. Also, in a Northern Ireland stream, the River Bush, Kennedy (1981), Kennedy and Strange (1982, 1986) captured more than $75 \%$ of the salmon fry in sites with mean depth $<20 \mathrm{~cm}$. The yearling salmon were in all water depths, but with a trend for higher numbers as water depths increased. They did not find the relative abundance of salmon parr to have any significant correlation with gradient, and concluded that salmon apparently will live as readily in sluggish low gradient areas. However, in a Nova Scotian river Amiro (1984) related juvenile salmon densities to gradient, illustrating the interactions of factors in different regions.

Many physical habitat variables are auto-correlated (Leopold et al., 1964), and Kozel et al., (1989) found features of channel morphology and of habitat, including substrates, to be related to gradient, which they correlated with trout standing stocks.

Power (1973) considered substrate to be the most important factor affecting production of salmonids. In a tributary of the River Bush, where some sections had been damaged by channelization, Kennedy and Johnston (1986) found a highly significant correlation between density of young salmon and the proportion of the substrate comprised of stones $>10 \mathrm{~cm}$ in diameter.

Young salmon are generally associated with fast water velocity (Wankowski and Thorpe, 1979; Degraaf and Bain, 1986; Morantz et al., 1987), but regulate selected water velocities ('nose' velocity) by their height above the substrate, according to mean water velocity (Tables 1 and 2), temperature and social status (Gibson, 1988). Although riffles are considered to be preferred habitat of young salmon, a wide range of types of habitat (Table 3) can be occupied (Table 4) depending partially on densities and intra-specific competition. Distribution and production of young salmon is also affected by inter-specific competition and predation and the availability of food (Gibson, 1973). Also, holding stations and behaviour, change with types of habitat and with season (Gibson, 1978 ${ }^{\text {a }}$; Huntingford et al., 1988).

Selected feeding stations are related to a favourable cost-benefit ratio of energy expended versus energy gained (Fausch, 1984), although are modified by factors as reviewed above. Habitat choice therefore results from a complex of factors, which may interact synergistically. For example, although young salmon are generally regarded as associated with a coarse substrate, the importance of this habitat variable is dependent on interactions with water velocity, depth,
light and cover. Interactions of these factors was demonstrated in the following experiments. In aquarium experiments using artificial illumination with depths of 30.5 cm , Gibson and Keenleyside (1966) found that over a smooth sand substrate at surface illumination greater than $300 \mathrm{ft}-\mathrm{c}$ and up to $1220 \mathrm{ft}-\mathrm{c}$, salmon preferred a shaded half of the tank. When the bottom was changed to a broken type of substrate, parr were attracted into the illuminated area, which they attributed to photopositive behaviour in the presence of available cover from potential predators. However, in stream tank experiments with water velocities of $2-34 \mathrm{~cm} \cdot \mathrm{~s}^{-1}$ under natural light (higher intensity of illumination than in the aquarium experiments), Gibson and Power (1975) found that types of substrate were not important in affecting the distribution of salmon, but that overhanging shade cover preferentially attracted parr in shallow water ( $24-29 \mathrm{~cm}$ ), although was not attractive in deeper water ( $43-50 \mathrm{~cm}$ ). In further stream tank experiments (Gibson, 1978a) and in a field experiment (Gibson, 1966) a turbulent broken water surface was found to be more attractive to parr in shallow water than shade cover. Therefore salmon parr may not always select type of substrate per se, for feeding stations or as protection from predators, and the preferred cover changes with environmental conditions. The apparent preference for coarse substrate in streams may be due to other related conditions, such as a broken water surface providing cover from avian predators, preferred habitat of prey, pockets of reduced water velocity in fast water, inducement of smaller territories, etc. In running water young salmon frequently hold station on the tops of rocks in a characteristic way, by inclining the anterior margins of their pectoral and pelvic fins against the rock to result in downward pressure from the current (Kalleberg, 1958; Keenleyside, 1962), so may also choose the type of substrate for this reason.

Despite the adaptability of young salmon to types of habitat, certain areas of a river system provide more productive habitat than other parts of the system. A number of studies have shown poor correlations between estimated densities of salmon parr and subsequent yield of smolts, mainly because a mean density of parr per area for the whole area of the system was calculated from random surveys, and the wide range of production from types of habitat and reaches within a river system had not been appreciated (e.g. Elson and Tuomi, 1975). During the past two decades it has become apparent to stream ecologists that biotic dynamics and interactions are intimately and inextricably linked to variation in abiotic factors (Power et al., 1988). We have attempted to relate salmon density, biomass and production to variables of the habitat (Table 5) in a number of rivers in Newfoundland. In the Highlands River, a fourth order river in southwest Newfoundland, biomass of salmon ranged from 1.50 to $3.53 \mathrm{~g} \cdot \mathrm{~m}^{-2}$ at six sites in 1980 , and $0.69-2.90 \mathrm{~g} \cdot \mathrm{~m}^{-2}$ at 10 sites in 1981 (Gibson et al., 1987). Production to mean biomass ratios over the growing season in 1980 ranged from 0.37 to 1.29 . Salmonid production was highest in second order streams and at a station below a lake. Lowest production was at sites in the main river. Biomass of salmon parr in a lake was about $0.1 \mathrm{~g} \cdot \mathrm{~m}^{-2}$. Biomass and densities were correlated with various variables by a stepwise regression model (Neter \& Wasserman, 1974). Variables were entered in the stepwise regression only if the variable was significant at $\mathrm{P}<0.15$. There was a sample size of 24 observations. With total density of parr ( $\mathrm{D}_{\mathrm{ST}}$ ) two variables were retained in
the model, substrate rating $\left(X_{1}\right)$ and mean width ( $X_{2}$ ), to give the following relationship:

$$
\begin{gathered}
\mathrm{D}_{\mathrm{ST}}=-12.88+12.54\left(X_{1}\right)-0.48\left(X_{2}\right) ; \text { model } r^{2}=0.48 . \\
\quad\left(\text { partial } \mathrm{r}^{2}: \mathrm{X}_{1}=0.26 ; \mathrm{X}_{2}=0.22\right)
\end{gathered}
$$

By using a logarithmic transformation salmon biomass ( $\mathrm{B}_{\mathrm{S}}$ ) had a positive relationship with substrate rating ( $X_{1}$ ), and negative relationships with the variables mean width $\left(X_{2}\right)$, and \% overhanging cover ( $X_{3}$ ) as follows:

$$
\begin{aligned}
\log \left(B_{S}^{+1}\right) & =0.42+1.17\left(\log X_{1}+1\right)-0.51\left(\log X_{2}+1\right)-0.20\left(\log X_{3}+1\right) ; \\
\text { modeI } r^{2} & =0.68 .\left(\text { partial } r^{2}: X_{1}=0.37 ; X_{2}=0.20 ; X_{3}=0.11\right)
\end{aligned}
$$

The stepwise regression analyses in some of the models also retained the variables depth and ice scour height, with negative relationships. These conclusions are consistent with qualitative observations that juvenile salmon are most abundant in association with open riffle areas (Keenleyside, 1962) and that generally there is higher production in stable streams than in ones with less stable flows, and in lower order streams than in bigger rivers (Hynes, 1970).

The approach of estimating juvenile salmon production by partitioning a river system into segment systems, reach systems and types of habitat within reaches, differing in relative production (e.g. Frissell et al., 1986) was given some validity by the Highlands River study, since estimates of smolt yield, by using estimated large parr densities and probable overwinter survival rates (Symons, 1979) gave in two years estimates differing in counts at a downstream trap by $-21 \%$ (in 1981) and $-24 \%$ (in 1982), although only about $1 \%$ of the habitat was sampled. The similar underestimate each year suggests a similar error, perhaps an underestimate of survival or, since accurate habitat mapping was not undertaken, of available habitat.

Analyses of relationships for Northeast Brook, Trepassey, a third order river in southeast Newfoundland in 1984 are given in Table 6 (Gibson et al., submitted). Variable selection was done by a forward stepwise procedure with deletion. Variable inflation factors (VIF) and condition numbers were less than 10 , indicating no colinearity problems. Both model and adjusted $r^{2}$ values are presented. The adjusted $r^{2}$ provides a modified measure which recognizes the number of independent variables in the model (Neter \& Wasserman, 1974). Twenty stations were sampled. Results are similar in that they indicate that juvenile salmon were most abundant over a coarse substrate and in smaller order streams. $2+$ parr had a negative relationship with mean water velocity, related probably to their more frequent occurrence in medium water depths (Gibson, 1988). The negative relationship of salmon biomass with water conductivity is unexpected, but there may have been confounding factors related to stream width or intra- or inter-specific interactions and distance from the sea, with the latter influencing deposition of sodium chloride, which has little effect on production.

## Competition

Distribution and production of juvenile salmon may be affected by inter-specific competition (Gibson 1988), changing the definitions of salmon habitat. It may be possible to derive quantitative relationships between close competitors and types of habitat, so that natural fluctuations in the abundance of one species could affect abundance of the other, related to the habitat, providing a measure of the status of one of the species. For example, small brook trout may be abundant in riffles with low abundance or absence of salmon parr, but trout may be sparse in such habitat when salmon were abundant. Probably in some waters these relationships could be quantified. Also, growth rates in young salmon are controlled by both productive capacity of the habitat, and by density, or intra-specific competition, so that is should be possible to derive a model from which one of the variables could be predicted as a function of the other (Gibson, 1978b, Gibson and Dickson, 1984).

## Invertebrates

Aquatic invertebrates are the principal prey of young salmon, which generally show opportunistic feeding (Thonney and Gibson, 1989). A positive relationship has been shown between biomass of salmonids and amounts of invertebrates suitable as food (Egglishaw, 1967; Gibson and Galbraith, 1975; Gibson et al., 1984). Lindroth (1965) hypothesized that food supply, in conjunction with available space, would limit growth and survival of young salmon, and ultimately smolt production. However, further research is needed on methods of quantifying the available food.

## Water chemistry

Where limiting, in lower order streams additions of nutrients have positive effects on salmonid survival and production, probably through enhanced production of food organisms (Peterson et al., 1985). For example, in an enriched third order stream in St. John's, Newfoundland, juvenile salmon showed exceptional production (Gibson and Haedrich, 1988). In late summer, and in low flows, at the experimental riffle station, concentrations of nitrate and nitrite were 0.190 to $0.890 \mathrm{mg} \cdot \mathrm{l}^{-1}$, and total phosphorous 0.067 to $0.198 \mathrm{mg} \cdot \mathrm{l}^{-1}$. Large salmon parr ( $>10 \mathrm{~cm}$ ) had densities in two consecutive years of 0.30 and 0.28 $\mathrm{m}^{-2}$, three to five times greater than would be found in local salmon rivers. Also, despite high densities of $0+$, of 1.02 and $1.53 \mathrm{~m}^{-2}$, daily specific growth rates (GW, \% $\mathrm{d}^{-1}$ ) of the $0+$ over the summer were, in two years, 2.98 and 3.59 . Mean fork lengths in September over two years were 6.7 and 6.6 cm for the $0_{+}$, and 11.7 and 12.0 cm for $1+$. Comparative sizes at the same age in an unenriched local river were 4.7 cm for $0_{+}$, and 7.7 cm for $1+$.

Similarly, enhanced production of juvenile salmon has been shown in an enriched river in southern Norway (Bergheim and Hesthagen 1990), where in good habitat, and with nitrate concentrations of $1.10-1.40 \mathrm{mg} \cdot \mathrm{l}^{-1}$, and total phosphorous of 0.031 to $0.084 \mathrm{mg} \cdot \mathrm{l}^{1}$, density of large parr was $0.51 \mathrm{~m}^{-2}$.

Johnston et al. (1990) in British Columbia increased juvenile salmonid growth by experimental fertilization of a stream poor in nutrients. They showed that enrichment by inorganic nitrogen and phosphorous, to $0.01-0.015 \mathrm{mg} P .1^{-1}$ and $0.03-0.10 \mathrm{mg} \mathrm{N} \cdot \mathrm{l}^{-1}$, increased periphyton and standing crops of benthos.

Hynes (1969) pointed out that nutrients may have little effect in larger order and deeper rivers, and that smaller order rivers, where there is generally canopy cover, normally are net consumers of organic matter, but that where canopy cover is lacking, autotrophic production can be important in shallow streams, and that enrichment in moderation can increase the production of game fish. Generally juvenile salmon rearing habitat is in open shallow rivers with clear water, where there is suficient gradient to curtail growth of emergent vegetation. There appears to be great potential for increased productive capacity of such habitat for salmon by artificial enrichment, especially where nutrients are low, and fish species diversity sparse, as in typical Newfoundland rivers.

Binns (1979) suggests as a rating for salmonid habitat models, that nitrate nitrogen ( $\mathrm{mg} / \mathrm{l}$ ) had the poorest rating at $<0.01$ or $>2.0$, and best at $0.15-0.25$, and that total phosphorus ( $\mathrm{mg} / \mathrm{l}$ ) should have the poorest rating at $<0.002$ or $>0.112$ and best rating for $0.051-0.096$. In the rivers of St. John's, Newfoundland, there is little industrial pollution, but there is increasing enrichment downstream from agricultural fertilizers and sewage overflows. Further research on young salmon production in different parts of the systems varying in water chemistry could identify optimum levels of nutrients for maximum salmon production.

Water conductivity, an indicator of total dissolved solids, may be an indicator of stream fertility. Calcium and magnesium ions help neutralize acid waters and enhance bacterial and fungal colonization of organic detritus. Other ions are needed for primary production and for bacterial and fungal activity (Hynes, 1970). Several authors have observed that water chemistry affects salmonid production (e.g. Huet, 1964; Cuinat, 1971; Egglishaw and Shackley, 1985). However, within the influence of the sea, as are most salmon rivers, conductivity may be principally related to sodium chloride, which has little effect on production.

## Turbidity

Turbidity has been shown to change behaviour of stream salmonids and reduce production (Crouse et al. 1981; Bisson and Bilby 1982; Sigler et al. 1984; Berg and Northcote 1985). High turbidities interfere with visual feeding and at very high suspension ( $>90 \mathrm{mg} / \mathrm{l}$ ) suspended sediments may damage the gills and kill salmonids (Hynes, 1960, 1973; Sorenson et al 1977). Turbidity reduces primary production, and siltation adversely affects benthic animals (Hynes, 1970;Lemly, 1982). Waters that contain more than $25-80 \mathrm{mg} \cdot \mathrm{l}^{-1}$ of suspended solids are unlikely to support good fisheries, and ideally streams should be clear.

## Winter habitat

Stream salmonids towards winter typically seek more sheltered habitat (Chapman \& Bjorn, 1969). In running water at temperatures less than about $9^{\circ} \mathrm{C}$ young salmon seek shelter under coarse substrate or move to pools (e.g. Allen, 1940; Smirnov et al., 1976; Karlström, 1977; Gibson, 1978a; Gardiner \& Geddes, 1980; Gardiner, 1984; Rimmer et al., 1984;), although they continue feeding overwinter (Cunjak, 1988). Overwinter survival is poor where suitable habitat is lacking (Hutchings, 1986), emphasizing that suitable wintering habitat of coarse substrate, deep pools, or lakes when predatory or competitive fish species are few, is essential for high survival to the smolt stage.

## Winter and summer temperatures and stream discharge

Although low temperatures reduce metabolism, lethally low temperatures $\left(<0.7^{\circ} \mathrm{C}\right.$ ) are not reached in freshwater, although mortality of developing eggs increases below $4^{\circ} \mathrm{C}$ (Peterson et al. 1977). However, mortalities due to high water temperatures have been recorded (Huntsman, 1942, 1946). The lethal limits for prolonged exposure to high water temperatures are about $28.0^{\circ} \mathrm{C}$ for salmon parr (Fry, 1947). Optimum growth of salmon fry occurs at $16.6^{\circ} \mathrm{C}$ (Siginevich, 1967) similar to the final preferendum of $17^{\circ} \mathrm{C}$ (Javaid and Anderson, 1967). Dwyer and Piper, (1987) reported optimal growth for juvenile salmon at $13^{\circ}-16^{\circ} \mathrm{C}$. Salmon parr will move into cooler water, if locally available, at temperatures above about $22^{\circ} \mathrm{C}$ (Gibson, 1966). Power (1969) suggested that 100 growing days above a water temperature of $6^{\circ} \mathrm{C}$ were a minimum requirement for existence of salmon. However, in a cold Norwegian salmon river Jensen and Johnsen (1986) noted that growing days were less and that water temperatures were at or above $7^{\circ} \mathrm{C}$ for only about 67 days a year, suggesting that there may be genetic adaptation to rigorous conditions. Nevertheless, the longer photoperiod and therefore available feeding time during the summer at the relatively higher latitudes of northern European rivers should probably also be taken into consideration.

Streams with wide ranges in discharge are less productive than more stable systems (Hynes 1970; Binns and Eiserman 1979) and positive relationships have been shown between river discharge and survival of underyearling salmon for summer flows (Lishev 1959; Havey and Davis 1970; Frenette et al. 1984) and for winter flows (Chadwick 1982; Frenette et al., 1984; Gibson and Myers 1988). Gibson and Myers (1988) examined stock-recruitment data from six rivers of Newfoundland and New Brunswick and related survival of underyearling salmon to river discharges. For all rivers combined, survival and winter discharge were related ( $P$ <0.05). A reduced subsurface flow in the gravel, and therefore oxygen supply, has been shown to be a consequence of lowered streams depth and velocity (Wickett 1958). The hypothesis that summer discharge was unrelated to survival could not be rejected. However, there was evidence that summer discharge enhances survival in the Miramichi and Northwest Miramichi rivers of New Brunswick. The authors concluded that in many boreal rivers such as those in Newfoundland, discharges in summer generally did not limit survival. However, exceptionally low water levels such as those experienced in 1987 are likely to negatively affect survival of underyearlings.

Survival of parr has been shown to be positively related to summer river discharge in some rivers (Huntsman 1937, 1973; Amiro and McNeill 1986) and may have more general application. Effects of relative survival between year-classes in different rivers are probably dependent on the amounts of types of habitat suitable for each year-class in the various rivers. In rivers with relatively low densities of salmon abiotic factors have greater influence on survival than where densities are high, where density-dependent factors have greater influence (Elliott 1984).

## Summary

## Introduction

Juvenile Atlantic salmon are generally regarded as riffle dwellers. Underyearlings are found predominantly in shallow riffles over pebbles, whereas parr are found in deeper, faster water associated with cobble, rubble and boulders, with large parr being common in pools. However, young salmon are flexible in their habitat distribution and occur also in lentic habitats, (Table 4), depending to a large extent on the fish community.

With a view to quantifying productive capacity of stream habitat for Atlantic salmon, CAFSAC has been asked to provide advice on the usefulness of measuring the following attributes in assessing suitable habitat: substrate; stream width; cover; velocity; ice scour; stream order; turbidity; total dissolved solids; pH ; winter temperature; summer temperature; discharge; and gradient, and to provide advice on the possible value of other attributes.

In the present paper a brief review is given of the freshwater habitat of Atlantic salmon and factors affecting production. Preliminary models are presented from two rivers in Newfoundland, Highlands River in the southwest, and Northeast Trepassey Brook in the southeast part of the Avalon Peninsula. A stratified random sampling method was used by estimating densities and biomass of salmonids from different types of habitat in similar reaches of the river system, in which standard depletion or mark and recapture techniques were employed, using electro-fisher, seine net, or fyke nets. Habitat variables of stations as shown in Table 5, were measured at the same time. Rigorous predictive models are lacking at present, and further data are needed, to include a wider range of values in the variables being measured.

## Results and Discussion

Multiple linear regression models from both the Highlands River and Northeast Trepassey Brook selected substrate as an important variable, with a positive relationship to an index of coarseness. Also, there was a negative relationship with mean width, indicating that smaller order streams in these systems were more productive than the wider reaches. Some of the models indicated a negative relationship with range of discharge, and that overhanging cover was unimportant.

A stepwise multiple linear regression with total biomass ( $\mathrm{B}_{\mathrm{s}}$ ) of salmon for the Highlands River gave:
$\log B_{s}+1=0.42+1.17\left(\log X_{1}\right)-0.51\left(\log X_{2}+1\right)-0.20\left(\log X_{3}+1\right) ;$ model $r^{2}=$ 0.68
and for Northeast Trepassey Brook gave:

$$
B_{S}=6.10+1 \cdot 13\left(X_{1}\right)-0.12\left(X_{2}\right)-0.16\left(X_{4}\right) ; \text { model } r^{2}=0.69
$$

where,
$X_{1}=$ substrate rating; $X_{2}=$ mean width;
$X_{3}=\%$ overhanging cover; $X_{4}=$ specific conductivity

Coarse substrate also is required for survival of eggs and alevins in redds, and fine sediments are deleterious.

It can also be concluded that changes to lengthen the growing season by raising water temperatures would be beneficial, but temperatures ideally should remain below about $22^{\circ} \mathrm{C}$.

Although length of the growing season is a major factor in annual production of salmonids, water chemistry (excluding factors having direct deleterious physiological effects) is of equal or greater importance, probably mediated through abundance of food organisms but depending also on habitat and the fish community. This is illustrated by research in a city river of St. John's, Newfoundland, where in an enriched stream salmon production was at least five times greater than similar local rivers with less nutrients. This would suggest that research on effects of water chemistry on productive capacity of stream habitat for Atlantic salmon is of major importance.

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Table 1. Diving observations of juvenile Atlantic salmon in rivers near Cape Race, southeast Newfoundland, at temperatures between $10^{\circ} \mathrm{C}-19^{\circ} \mathrm{C}$. Mean depth, water velocity (nose), and distance above the bottom were measured (standard deviations are in parentheses).

| Habit Fish size: | Riffle |  |  | Flat |  |  | Pool |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | L | S | 0 | L | S | 0 | L | S | 0 |
| $\begin{aligned} & \text { Depth (cm) } \\ & \text { (S.D.) } \end{aligned}$ | $\begin{aligned} & 24.1 \\ & (5.59) \end{aligned}$ | $\begin{aligned} & 22.5 \\ & (6.20) \end{aligned}$ | $\begin{aligned} & 17.4 \\ & (5.83) \end{aligned}$ | $\begin{aligned} & 32.3 \\ & (8.02) \end{aligned}$ | $\begin{aligned} & 28.1 \\ & (8.20) \end{aligned}$ | $\begin{aligned} & 23.0 \\ & (5.00) \end{aligned}$ | $\begin{aligned} & 56.8 \\ & (15.16) \end{aligned}$ | $\begin{aligned} & 41.6 \\ & (11.21) \end{aligned}$ | $\begin{aligned} & 31.8 \\ & (14.22) \end{aligned}$ |
| $\begin{aligned} & \text { Velocity }\left(\mathrm{cm} \cdot \mathrm{~s}^{-1}\right) \\ & \text { (S.D.) } \end{aligned}$ | $\begin{aligned} & 20.3 \\ & (8.66) \end{aligned}$ | $\begin{gathered} 20.0 \\ (10.87) \end{gathered}$ | $\begin{aligned} & 12.8 \\ & (15.18) \end{aligned}$ | $\begin{aligned} & 12.7 \\ & (7.04) \end{aligned}$ | $\begin{aligned} & 10.7 \\ & (5.62) \end{aligned}$ | $\begin{aligned} & 12.7 \\ & (8.56) \end{aligned}$ | $\begin{aligned} & 12.8 \\ & (7.47) \end{aligned}$ | $\begin{aligned} & 6.2 \\ & (4.15) \end{aligned}$ | $\begin{aligned} & 6.1 \\ & (2.34) \end{aligned}$ |
| $\begin{aligned} & \text { aeight above bottom (cm) } \\ & \text { (S.D.) } \end{aligned}$ | $\begin{aligned} & 0.3 \\ & (0.72) \end{aligned}$ | $\begin{aligned} & 0.6 \\ & (1.55) \end{aligned}$ | $\begin{gathered} 0.20 \\ (0.53) \end{gathered}$ | $\begin{aligned} & 2.9 \\ & (4.11) \end{aligned}$ | $\begin{gathered} 2.2 \\ (3.70) \end{gathered}$ | $\begin{aligned} & 0.3 \\ & (0.65) \end{aligned}$ | $\begin{gathered} 7.3 \\ (8.84) \end{gathered}$ | $\begin{gathered} 9.6 \\ (11.26) \end{gathered}$ | $\begin{gathered} 0.3 \\ (0.82) \end{gathered}$ |
| Number of observations | 22 | 16 | 17 | 27 | 25 | 23 | 25 | 33 | 6 |

```
L = large salmon parr (>10 cm)
S = small salmon parr (>6 cm <10 cm)
0=0+ salmon parr (<6 cm}
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Table 2. Mean width, depth, and mid-depth water velocity characteristics of the types of habitat at diving stations. (Standard deviations are given in parentheses.) $n=$ number of observations.

| $(\mathrm{n})$ | Riffle <br> $(19)$ | Flat <br> $(25)$ | Run <br> $(4)$ | Pool <br> $(19)$ |
| :--- | :---: | :---: | :---: | :---: |
| Mean width (m) <br> (S.D.) | 5.9 | 5.3 | 2.7 | 6.6 |
| Mean depth (cm) <br> (S.D.) | $(3.26)$ | $(1.64)$ | $(1.33)$ | $(2.99)$ |
| Mean velocity (cm•s <br> (S.D.) | 18.8 | 22.9 | 38.3 | 41.4 |

Table 3.

The major types of habitat that are recorded. Habitat types were taken from Allen (1951).

Pools: Of two groups: pools, with current of less than $38 \mathrm{~cm} \cdot \mathrm{~s}^{-1}$, and depth 46 cm to 68 cm ; and, deep pools, with current less than 38 $\mathrm{cm} \cdot \mathrm{s}^{-1}$, and depth over 68 cm .

The flow is smooth apart from a small turbulent area at the head of some pools.

Flats: Current under $38 \mathrm{~cm} \cdot \mathrm{~s}^{-1}$, mean depth under 46 cm . Flats are sections of relatively shallow water, but with a smooth surface.

Runs: Current over $38 \mathrm{~cm} \cdot \mathrm{~s}^{-1}$, mean. depth over 23 cm . The flow is usually turbulent. In such places the stream is usually of less than average width.

Riffles: Current over $38 \mathrm{~cm} \cdot \mathrm{~s}^{-1}$, mean depth under 23 cm . These are shallow water with a rapid current and usually a broken flow.

Cascades: These are rapids in which a steep gradient, combined with a bed of stones or rocks large in proportion to the size of the stream, produces a very irregular rapid flow, of ten with some white water.

Table 4. Mean biomass $\left(g \cdot \mathrm{~m}^{-2}\right)$ of juvenile salmon (standard deviations in brackets or 95\% C.L. where one station) at stations in Northeast Trepassey Brook in 1984 ( $n=$ no. of stations). The designation of fluvial habitat types are given in Table 3.

| Riffles | 4.06 |
| :---: | :---: |
| $\mathrm{n}=11$ | $(1.42)$ |
| Run | 2.36 |
| $\mathrm{n}=1$ | $(1.87)$ |
| Flats | 2.78 |
| $\mathrm{n}=4$ | $(1.24)$ |
| Pools |  |
| $\mathrm{n}=3$ | 3.25 |
| Lake | $(0.38)$ |
| $\mathrm{n}=1$ | 0.47 |
|  | $(0.20)$ |
| Mean area of stations other |  |
| than lake (m2) |  |
| (Range, $65-463$ ) | 198 |
| Area of lake (ha) |  |
| Mean depth of lake (m) | 7.27 |

Table 5. The habitat variables measured for derivation of the stepwise multiple regression equations (Adapted from Platts et al. 1983 and Bain et al. 1985).

Mean stream width ( $m$ ) - usually at three locations in the station $\times \mathrm{n}^{-1}$. (Both wet, and bank to bank)

Mean depth (cm) - usually five_equidistant locations at the same transects as the width measurements $x(n+1)^{-1}$.

Mean water velocity $\left(m \cdot s^{-1}\right)$ - measured at 0.6 of the depth at a quarter and half distance locations at the same transects as the depth measurements.

Maximum flood height (cm) - experimental rivers, or ice scour height (m), Highlands River - an indicator of range of discharge.

Maximum depth (cm).

Each proportion of substrate type is multiplied by the rating, and the results summed for a general substrate rating.

Instream cover (\%) - undercut banks, tree debris, aquatic plants, etc.
Overhanging cover (\%) - structures up to about 1 m above the surface and providing shade, such as alder bushes, etc.

Canopy cover (\%) - shade over the stream provided by trees.
Specific conductivity ( $\mu \mathrm{si} / \mathrm{cm}$ )

Table 6. Stepwise multiple regression equations for salmon in Northeast Brook.


# Invertebrate Biomass in Predicting the Productive Capacity of Stream Habitat for Brook Trout (Salvelinus fontinalis). 

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## Abstract

Colonization baskets were placed within the Freshwater River, Drook River and Northeast Trepassey Brook systems on the south eastern Avalon Peninsula, Nfld. The baskets were removed after approximately four weeks, and the invertebrates were identified to the family level and counted. The study was repeated over four field seasons. During the last season, invertebrates were identified to the species level and the assemblages were defined using Two-Way Indicator Species Analysis (TWINSPAN). The species, abundances and displacement volumes of invertebrates differed between rivers. Invertebrate abundances and volumes were found to be important in predicting trout abundances.

Résumé
Des paniers de colonisation ont été placés dans les rivières suivantes Freshwater, Drook et Northeast Trepassey, sur la côte est de la péninsule d'Avalon, Terre Neuve. Les paniers ont été retirés après environ quatre semaines et les invertébrés ont été identifiés d'après leur famille et comptés. Pendant la dernière saison les invertébrés ont été identifiés au niveau des espèces. L'étude a été répétée sur une periode de quatre saisons experimentales et la communauté a été identifée par "Two-Way Indicator Species Analysis (TWINSPAN)". Les espèces, la quantité et le volume des invertébrés différaient parmi les rivières. La quantité des invertébrés et le volume s'est révélés être important pour prédire l'abondance des truites.

## Introduction

Since 1984, Dr. R.J. Gibson of the Department of Fisheries and Oceans (DFO), St. John's, Nfld. has been studying the ecology of juvenile salmonids within Freshwater River (46010'N 53006'W), Drook River (46041'N 53012'W) and Northeast Trepassey Brook (46046'N 53021'W), Cape Race, Nfld. All were relatively small, third order rivers and will be referred to as the experimental rivers (Figures l-3). His work entailed the collection of data concerning juvenile Atlantic salmon (Salmo salar) and brook trout (Salvelinus fontinalis), in conjunction with chemical and physical variables, and the collection and identification of invertebrates. This paper will describe the methodology used in collecting the invertebrates as well as the results of these collections. The focus will be upon the importance of invertebrates in determining abundances of brook trout, as this was the only salmonid present at all times in the experimental rivers.

## Materials and methods

Benthic population estimation began during the fall of 1984 (Thonney, Gibson and Hillier 1987). Six cylindrical colonization baskets were placed within riffles at each of the selected stations (Table 1 and Figures 1-3) during September. The baskets (diameter $=20.0 \mathrm{~cm} ;$ height $=10.0 \mathrm{~cm}$ ) were made of Vexar (mesh diameter $=1.9 \mathrm{~cm}$ ) and were filled with smooth, rounded beach stones that were uniform in size [In 1989, the mean count of stones per basket was 53 (S.E. $\pm 12$ ). The number of stones within the baskets did not vary significantly (Kruskal-Wallis one-way analysis of variance, $P>0.05$ ) among stations. The mean displacement volume of all of the stones per basket was 2.129 L (S.E. $\pm .222$ ) and did not significantly vary among stations (Kruskal-Wallis one-way analysis of variance, $\mathrm{P}>0.05$ )]. After approximately 30 days, the baskets were removed by carefully lifting the baskets while holding a small mesh net under each basket. The net captured any falling invertebrates. The baskets and individual stones were brushed; all invertebrates and debris were sieved and stored in Kahle's solution (Martin 1977). The invertebrates were identified to the family level, counted and where possible their displacement volumes were determined.

During subsequent years, the colonization baskets were set late in April and were removed late in May. Only four replicates at each station instead of the previous six replicates were sampled from the 1989 collections. The 1989 animals were identified to the species level wherever possible and Two-Way Indicator Species Analysis (TWINSPAN) was used to define species assemblages. This analysis made use of the programs SPPLST, CONDAT and TWIN (Carleton 1985). Among year, among river and among station variability of invertebrate abundances and
displacement volumes were determined using Wilcoxon 2-sample and Kruskal-Wallis one-way analyses of variance tests. KruskalWallis tests were also used to describe among year and among river variability for the following edaphic variables: watershed area for each station, greatest station depth, mean station depth, mean station width, mean water velocity, water temperature, in stream cover, overhanging cover, specific conductivity of the water, magnesium concentration, hardness, sulphates, calcium, turbidity and color. Volumes and abundances of invertebrates, as well as, the various edaphic variables were related to numbers of brook trout by way of backward stepwise regression and correlation coefficient analyses (SPPS Inc. 1986). Among river comparisons of edaphic data were limited to 1984 because of data limitations.

## Results

Qualitative (Table 2) and significant quantitative (Table 3) differences existed among invertebrates sampled from Drook River, Freshwater River and Northeast Trepassey Brook. Hyalella azteca, members of the genus Hydropsyche, Drunella cornuta, Leptophlebia cupida, Lepidostoma sp. were present within Freshwater River during 1989 but were not found within Drook River, during the same period. Alloperla concolor, Epeorus pleuralis, Paraleptophlebia adoptiva and Rhyacophila ignorata were present within Drook River but were not within Freshwater River. An intermediate assemblage was present within Northeast Trepassey Brook. Fewer than 1 in 10 within river comparisons indicated significant abundance or volume of invertebrate differences (Table 4).

There were no significant among river differences in physical environmental variables (Table 5). However, calcium carbonate and sulphate concentrations differed significantly between rivers (Table 6).

The fall 1984 abundances of brook trout were significantly correlated with field conductivity, mean station width and mean water velocity. Stepwise elimination of variables produced a model that included invertebrate displacement volumes, watershed area, mean station width, greatest depth, stream order and turbidity (Table 7). This listing of variables changed when the spring data for the years 1985, 1986 and 1987 were combined. The spring abundances of trout were related to the number and displacement volumes of invertebrates, watershed area, mean station depth, mean station width and water temperature (Table 8). The fall model accounted for $86 \%$ of the variation in brook trout numbers; whereas, the spring model accounted for $0.96 \%$ of the variation.

Turbidity was the only water chemical variable that was significantly correlated with the fall 1984 brook trout abundances. Watershed area, field conductivity, water hardness, magnesium and turbidity were included in the multiple regression model to predict trout abundances during the fall of 1984 (Table 9). When the $1985-87$ spring water chemistry data were combined, the model was expanded to include volumes of invertebrates, Sulphates and colour (Table lo). The 1984 fall water chemistry and invertebrate model accounted for 54\% of the variation in trout abundances, whereas the 1985-87 spring model accounted for $87 \%$ of the variation in number of trout.

Discussion
Drook river runs through a steep valley with numerous cold water springs and is typically a cold fast moving river; Freshwater River is a slower warmer river with a number of shallow lakes throughout its length whereas Northeast Trepassey Brook is intermediate. These physical differences may have been primarily responsible for the differences in assemblages. TWINSPAN was able to identify assemblages only at the species level since the same family might occur in all systems, although species within a family might have a restricted distribution within one of the systems (Table 2). For example, Leptophlebia cupida and Paraleptoblebia adoptiva are both members of the family Leptophlebiidae; however the former species was found in Freshwater River while the latter was found only in Drook River and Northeast Trepassey Brook. The 1984 to 1987 data masked any differences in assemblages as a result of identifications made at only the family level.

Similarly, the Hydropsychidae and Emphemerellidae are other examples of families that held several species of animals in which individual species were restricted to one of the systems.

Species assemblage identification is interesting from a biogeographic standpoint and could be important when categorizing ecosystems; however, in terms of salmonid production it may be academic if the salmonids are unable to identify the differences in species of invertebrates. In other words, do different species within each family differ in taste or catchability? It is therefore important to determine whether salmonid diets change between the rivers. It is also important to determine whether the fish are feeding opportunistically. The latter may be assessed by comparing the potential food items with the quantities of ingested food items. If a potential food item is rare but is often chosen it is said to be selected for. In a previous study, Thonney and Gibson (1989) suggest that feeding by young brook trout and salmon was primarily opportunistic, and that availability within a habitat was an important factor in determining the
diet. However, feeding studies in the experimental rivers would provide more insight into preferred prey and energetics and mechanisms of competition between the two species in different habitats.

Three out of four multiple regression models include either displacement volumes of invertebrates, abundances of invertebrates, or both. Further research is necessary to determine the importance of this relationship between invertebrates and salmonid abundances. Qualities and quantities of invertebrates may be factors that control the abundances and distribution of trout.

Trout abundances were also related to watershed area, mean water depth, mean station width, water temperature, stream order, turbidity, field conductivity, water hardness and Magnesium content within the water. The models indicate that smaller watersheds are more productive than large watersheds. Previous manipulations with the data indicated that a similar negative relationship exists between abundances of invertebrates and watershed area (D. Orr unpub. data).

It must be noted that brook trout are the only salmonid within Drook River, while brook trout and Atlantic salmon are present within Northeast Trepassey Brook. Since 1986, Atlantic salmon adults have been introduced to Freshwater River on an annual basis. The regression models and correlation coefficients may therefore be biased by inter-specific competition for resources.

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Table 2 (continued)

| Drook River |  |  |  |
| :---: | :---: | :---: | :---: |
| Phylum | Order | Family | Genus species |
| Annelida (Class Oligochaeta) |  |  |  |
| Arthropoda | Aranaea |  | Oulimnius sp. |
|  |  | Elmidae | promoresia tardella |
|  | Diptera | Chironomidae |  |
|  |  | Empididae | Chelifera 5p. |
|  |  | Nymphomyiidae |  |
|  |  | Simuliidae | Prosimulium spp. <br> Simuliidae spp. |
|  |  | Tipulidae | Dicranota sp. |
|  | Ephemeroptera | Baetidae | Baetis pyomeaus Baetis tricaudatus |
|  |  | Ephemerellidae | Drunella cornuta <br> Ephemerella aurivilli |
|  |  | Heptageniidae | Epeorus pleuralis |
|  |  | Leptophlebidae | Paraleptophlebia adoptiva |
|  | Ostracoda <br> Plecoptera |  |  |
|  |  | Chloroperlidae <br> Leuctridae | Leuctra ferruqinea |
|  | Trichoptera | Glossosomatidae | e Glossosoma sp. |
|  |  | Hydroptilidae | Hydroptila metoeca |
|  |  |  | Oxyethira sp. |
|  |  | Philopotamidae | Dolophiloides sp. |
|  |  | Rhyacophilidae | Rhyacophila ignorata |
| Nematoda |  |  |  |
| Platyhelminthes Tricladida Planariidae |  |  |  |

Table 2 (continued)

|  |  | theast Trepass | y Brook |
| :---: | :---: | :---: | :---: |
| Phylum | Order | Family | Genus species |
| Annelida (Class Oligochaeta) |  |  |  |
| Arthropoda | Amphipoda | Talitridae | Hyalella azteca |
|  | Coeloptera | Elmidae | Oulimnius sp. |
|  | Diptera | Chironomidae |  |
|  |  | Empididae | Chelifera sp. |
|  |  | Nymphomyiidae |  |
|  |  | Simuliidae | Simulium spp. |
|  |  | Tipulidae |  |
|  |  |  | Dicranota sp. |
|  | Ephemeroptera | Baetidae | Baetis pygmeaus |
|  |  |  | Baetis tricaudatus |
|  |  | Ephemerellidae | Drunella cornuta |
|  |  |  | Ephemerella aurivilli |
|  |  |  | Ephemerella subvaria |
|  |  | Leptophlebiida | Paraleptophlebia adoptiva |
|  |  |  |  |
|  | Plecoptera | Chloroperlidae | Alloperla concolor |
|  |  | Leuctridae | Leuctra ferruginea |
|  |  | Nemouridae | Nemoura macdunnaughi |
|  |  | Perlodidae | Isoperla transmarina |
|  | Trichoptera | Glossosomatidae Glossosoma sp. |  |
|  |  |  |  |
|  |  |  | Hydrospyche morosa |
|  |  |  | Hydrospyche slossonea |
|  |  |  | Hydrospyche sparna |
|  |  | Hydroptilidae | Hydroptila metoeca |
|  |  |  | oxyethira sp. |
|  |  | Lepidostomatid Philopotamidae | ae Lepidostoma sp. |
|  |  |  | Dolophiloides sp. |
|  |  | Rhyacophilidae | Rhyacophila ignorata |
| Mollusca (Class Gastropoda)Nematoda |  |  |  |
|  |  |  |  |  |  |  |

Table 3 Annual between river comparisons of invertebrate abundances and displacement volumes. The data were from the Drook, Freshwater and Northeast Trepassey Rivers.

## Season: Fall

Year

84
Season: Spring
85
86
87
89
** $0.01<\mathrm{P}<0.05$

Kruskal-Wallis $\boldsymbol{X}^{2}$ approximation

| number | volume |
| :--- | :--- |
| 0.58 | $9.03 * *$ |

0.58
10.86(sig.)
$21.05($ sig. $)$
0.82
0.89
$16.60(\mathrm{sig}$.
2.09
0.42
14.77(sig.)

Table 4 Annual among station/ within river comparisons of invertebrate abundances and displacement volumes.

| Year | River | Degrees of freedom | Kruskal-Wallis $\boldsymbol{X}^{2}$ approximation number volume |  |
| :---: | :---: | :---: | :---: | :---: |
| Season: Fall |  |  |  |  |
| 84 | Drook | 6 | 15.99** | 17.53(sig.) |
| 84 | Freshwater | 7 | 7.28 | 9.11 |
| 84 | Northeast | 8 | 14.40 | 14.59 |
| Season: Spring |  |  |  |  |
| 85 | Drook | 1 | 0.07 | 0.27 |
| 85 | Freshwater | 2 | 10.19(si | .) 8.10 ** |
| 85 | Northeast | 2 | 2.16 | 1.38 |
|  | Trepassey |  |  |  |
| 86 | Freshwater | 1 | 0.23 | 0.92 |
| 86 | Northeast | 1 | 5.03** | 4.39** |
|  | Trepassey |  |  |  |
| 87 | Northeast | 1 | 0.00 | 4.43** |
|  | Trepassey |  |  |  |
| 89 | Drook | 1 | 0.00 | 1.71 |
| 89 | Freshwater | 3 | 7.43 | 9.25** |
| 89 | Northeast | 2 | 8.23** | 6.75** |
|  | Trepassey |  |  |  |



Season: Fall

Variable
Wilcoxon 2-sample test |Z|

```
greatest depth in cm
    1.491
mean depth in cm
mean width in m
    1.685
    0.646
mean velocity in cm/sec
    0.150
water temperature in *}\mp@subsup{}{}{\circ}\textrm{C
    0.389
instream cover 1.953
overhanging cover
substrate code
    0.530
    1.420
```

Table 6 Comparisons between Drook, Freshwater and Northeast Trepassey Rivers using 1984 water chemistry data.

Season: Fall
Variable
Kruskal-Wallis $X^{2}$ approximation

Calcium carbonate (mg/l)
11.36(sig.)

Sulphates (mg/l)
12.90(sig.)

Magnesium (mg/l)
0.30
total Phosphates (mg/l)
0.27

Total Alkalinity (mg/l
5.41

Calcium carbonate)
Colour
6.59**

Turbidity (JTUs)
2.58**
** $0.01<\mathrm{P}<0.05$

```
Table 7 Regression analysis of number of brook trout versus abundances and displacement volumes of invertebrates and physical environmental variables. The data were collected from the Experimental Rivers, Cape Race, Nfld. during the fall of 1984.
```

| Variables | N | Correlation coefficients |
| :--- | :--- | :--- |
|  |  |  |
| no. of invert. | 73 | 0.219 |
| vol. of invert. (mhos at | 73 | 0.050 |
| field conduct. (mh | 0.329 (sig.) |  |
| 250C) |  |  |
| watershed area (ha) | 73 | 0.062 |
| mean station width (m) | 73 | $0.453(\mathrm{sig})$. |
| greatest depth (cm) | 73 | 0.065 |
| mean velocity (cm/sec) | 73 | 0.406 (sig.) |
| stream order | 73 | -0.091 |
| Turbidity (JTUs) | 73 | $0.242 * *$ |

Multiple regression model derived through backward stepwise regression with an acceptance criteria of $\mathrm{P} \geq 0.100$.
$r^{2} 0.863$ mse 162.617 Prob>F 0.0001
number of observations 197

| variable | B value | STD error | F |
| :--- | ---: | :--- | ---: |
| intercept | 18.328 |  |  |
| vol. invert. | 0.151 | 0.053 | $7.98($ sig.) |
| watershed area | -0.194 | 0.081 | $5.85 * *$ |
| mean width | 2.224 | 0.119 | $349.69(\mathrm{sig})$. |
| greatest depth | 0.464 | 0.174 | $7.09(\mathrm{sig})$. |
| stream order | -63.283 | 7.667 | $68.12(\mathrm{sig})$. |
| turbidity | 0.798 | 0.057 | $198.31(\mathrm{sig})$. |

mse refers to the mean sum of the square error
** $0.01<\mathrm{P}<0.05$

Table 8 Regression analysis of number of brook trout versus abundances and displacement volumes of invertebrates and physical environmental variables. The data were collected from the Experimental Rivers, Cape Race, Nfld. each spring between 1985 and 1987.

Multiple regression model derived through backward stepwise regression with an acceptance criteria of $\mathrm{P} \geq 0.100$. $r^{2} 0.965$ mse 59.476 Prob>F 0.0001
number of observations 44

| variable | B value | STD error | F |
| :--- | ---: | :--- | :--- |
| intercept | -20.408 |  |  |
| no. invert. | 0.010 | 0.002 | 16.61 (sig.) |
| vol. invert. | -0.078 | 0.037 | $4.38 * *$ |
| watershed area | -8.939 | 0.919 | $94.52(\mathrm{sig})$. |
| mean depth | -24.800 | 3.412 | $52.84(\mathrm{sig})$. |
| mean width | 27.724 | 2.804 | $97.75(\mathrm{sig})$. |
| water temp. | -1.045 | 0.133 | $61.30(\mathrm{sig})$. |

mse refers to the mean sum of the square error
** $0.01<\mathrm{P}<0.05$

Table 9 Regression analysis of number of brook trout versus abundances and displacement volumes of invertebrates and water chemistry. The data were collected from the Experimental Rivers, Cape Race, Nfld. during the fall of 1984.

| Variables | N | Correlation coefficients |
| :--- | :--- | :--- |
|  |  |  |
| no. of invert. | 34 | 0.233 |
| volof invert. | 34 | 0.293 |
| Chloride (mg/l) | 34 | $-0.339 * *$ |
| turbidity in JTUs | 34 | 0.490 (sig.) |
| Calcium carbonate (mg/l) | 34 | 0.138 |
| colour | 34 | 0.275 |
| Magnesium (mg/l) | 34 | 0.202 |

Multiple regression model derived through backward stepwise regression with an acceptance criteria of $\mathrm{P} \geq 0.100$.
$r^{2} 0.537$ mse $665.096 \quad$ Prob>F 0.0001
number of observations 108

| variable | B value | STD error | F |
| :--- | ---: | :--- | :--- |
| intercept | -139.969 |  |  |
| watershed area | -0.166 | 0.035 | $22.83($ sig.) |
| conductivity | 3.307 | 0.992 | 11.1 (sig.) |
| hardness | -0.014 | 0.002 | 41.27 (sig.) |
| Magnesium | 1.277 | 0.348 | 13.44 (sig.) |
| turbidity | 0.686 | 0.126 | 29.72 (sig.) |

** $0.01<\mathrm{P}<0.05$
mse refers to the mean sum of the square error

```
Table 10 Regression analysis of number of brook trout versus abundances and displacement volumes of invertebrates and water chemistry. The data were collected from the Experimental Rivers, Cape Race, Nfld. each spring between 1985 and 1987.
```

Multiple regression model derived through backward
stepwise regression with an acceptance criteria of $\mathrm{p} \geq 0.100$.
$r^{2} 0.870$
mse 183
Prob>F 0.0001
number of observations 56

| variable | B value | STD error | F |
| :--- | ---: | :--- | ---: |
| intercept | -214.520 |  |  |
| vol. invert. | 0.146 | 0.042 | $12.00(\mathrm{sig})$. |
| conductivity | 14.176 | 0.936 | $229.36(\mathrm{sig})$. |
| hardness | 0.008 | 0.003 | $6.33 * *$ |
| Sulphates | -1.990 | 0.154 | $164.21(\mathrm{sig})$. |
| Magnesium | -1.089 | 0.282 | $14.95(\mathrm{sig})$. |
| colour | 2.363 | 0.169 | $196.64(\mathrm{sig})$. |
| turbidity | 1.083 | 0.207 | $27.30(\mathrm{sig})$. |

mse refers to the mean sum of the square error.
** $0.01<\mathrm{P}<0.05$



Figure 3
A scale map of Northeast Trepassey River ( $46^{\circ} 47^{\prime} \mathrm{N}$ $53^{\circ} 21^{\prime}$ w). Cape Race, Nfld. and the 1984-1989 invertebrate sampling stations.

# Habitat Utilization by Juvenile Atiantic Salmon 

 in Newfoundland Rivers
## by

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#### Abstract

Variation in the local densities of juvenile Atlantic salmon (Salmo salar) in relation to habitat was analyzed within and between river systems for several rivers in Newfoundland, for data collected from 1980 to 1988. One river (Northeast at Trepassey) was used as a "standard" river. Biomass ( $\mathrm{g} \mathrm{m}^{-2}$ ) varied greatly among sampling stations, reaching $7.95 \mathrm{~g} \cdot \mathrm{~m}^{-2}$ in Northeast Brook, and usually varied more than 10 fold within a river and among stream habitats. The highest biomass were observed in riffles and pool habitats and the lowest in pond habitats. Several stream attributes were found to be of consistently high explanatory power. Up to $79 \%$ of the variance in juvenile distribution and abundance can be explained on the Northeast Brook at Trepassey. In particular, Nitrate concentration, substrate type, and dimensions of the basin were important. The general model incorporating all rivers explained $37 \%$ of the variance in biomass, with nitrates again being the dominant factor, and including 5 of the 8 variables used in the Northeast at Trepassey model.


The introduction of salmon to Freshwater River in the Avalon peninsula provides useful information on habitat selection of juvenile salmon. Suggestions for further improvement are given.


#### Abstract

RESUNE L'auteur a analysé la variation des densités locales du saumon de l'Atlantique (Salmo salar) juvénile par rapport à l'habitat intra fluvial et inter-fluvial de plusieurs rivières de Terre-Neuve, à partir de données receuillies de 1980 à 1988. La rivière Northeast à Trepassey a été utilisée comme rivière de référence. La biomasse (g.m. ${ }^{-2}$ ) a grandement varié d'une station d'échantillonnage à l'autre; elle atteint $7,95 \mathrm{~g} \cdot \mathrm{~m}^{-2}$ dans le ruisseau Northeast et varie généralement par plus de 10 fois au sein d'une rivière et parmi les habitants fluviaux. La plus importante biomasse a été observée dans les radiers et les fosses, et la plus faible dans les étangs. Plusieurs caractéristiques des cours d'eau montrent un pouvoir explicatif élevé constant. Jusqu'a $79 \%$ de la variance de la distribution et de l'abondance des juvéniles peut ètre expliquée dans le ruisseau Northeast, à Trepassey. En particulier, la teneur en nitrate, le type de substrat et l'étendue du bassin sont importants. Le modèle général regroupant toutes les rivières explique 37 \% de la variance de la biomasse; à nouveau, la teneur en nitrate est le facteur dominant. Ce modèle comprend 5 des 8 variables utilisées dans le modèle de la rivière Northeast à Trepassey.

L'ensemencement de saumons dans la rivière Freshwater, de la presqu'ile Avalon, fournit des données utiles sur le choix d'habitat des saumons juvéniles. $L^{\prime}$ auteur présente des suggestions pour $d^{\prime}$ autres améliorations.


## INTRODUCTIOA

A good understanding of the dynamics of distribution and abundance of juvenile salmon in relation to habitat use is thought to eventually provide improved smolt and adult salmon production estimates. This work on habitat use hopes to go beyond the "black box" treatment of the relationship between juvenile habitat and production, and eventually provide general models for predicting the smolt output from rivers based on the physical characteristics of the river in question.

This report represent a preliminary survey of the results obtained with the data collected under the guidance and direction of Dr . Gibson. Much of the work in this contract involved the manipulation of historical records from different sources, namely population estimation data, data from samples taken for sex, age and maturity determination, physical characteristics of sampling stations, and water quality analysis.

The present paper can be divided into several conceptual sections 1)
Population estimation of juvenile salmon. 2) reduction of the very large data set into a smaller set of potentially important variables 3) Development of the model based on a "standard" river. Data from Northeast Brook at Trepassey provides an extensive survey of salmon habitat over a 5 year period and can be used alone to produce a habitat model. 4) Verification of the model inputs by testing other sampled rivers in the island of Newfoundland. This work is of a preliminary nature and is a step towards building and testing habitat models (see Talbot final report to the Salmon Association of Eastern Newfoundland, August 1990). The techniques used in this report were used with the objective of identifying potentially important physical attributes of rivers for the prediction of the variation in productivity within river, and not to establish precise parameter coefficients. The variables identified by the processes presented here were used in building habitat models. These results are presented elsewhere.

## METHODS

Several experimental and non-experimental rivers on insular Newfoundland were surveyed systematically throughout their respective watershed. A number of stations were selected as representative of the major types of habitat in the river systems. These were usually sampled yearly from June to August for population estimation. Some stations were also sampled in May and Sept/Oct of the same year for growth documentation. The habitat characteristics measured are given in Table 1 and Appendix A. Gibson et al (1987) provides details and reasoning for the sampling design for the Highlands River, one of the rivers which is included in this study.

## Population Estimation

Population estimates were done by one of 3 methods, depending on the physical characteristics of the sampling site and capture method. Stations with shallow riffles, runs or flats were sampled by successive sweeps of an electrofishing apparatus. The sampling design was conceived for determination of population size by the Zippin depletion method. However, for reasons described below, we use a proxy of Zippin in this atudy ("pseudo-Zippin"), but one which is still based on Zippin estimates.

As envisaged in the original design, some habitats cannot be sampled by electrofishing apparatus, or consecutive sweeps do not result in decreasing global catches. In such cases, population estimates cannot be derived from the Zippin method. In these stations, the sampling design was adapted to the Peterson mark-recapture method. Similarly, in ponds, the multiple markrecapture (Schnabel) method was used. In this study, ponds are reported in the descriptive statistics, but were not used in correlation and regression analysis, because they represent a special case of river system hydrology.

Population estimates were calculated for each species and each year class separately, as well as for the pooled year classes. Biomass estimates were calculated from the population estimate and the weight of the fish. The weight of the fish sampled was derived from the condition factor, estimated separately for mature males and for the remainder of the population. This was necessary since only a fraction of the fish captured at a station were weighed, whereas all fish were measured for length.

Missing values.
Occasionally, physical station parameters were not taken on the same day as the sampling (the actual collection of fish), or was done over 2 days or more. In order to associate station data with biological data, dates were adjusted and combined (if collected over several days), and merged according to the new dates if within a few days ( $<5$ ) of the biological sampling day. On occasion, the station water quality or other non-biological data was not collected or coded for a particular biological sample. Also, a water quality station was occasionally designed to cover several biological sampling stations of similar configuration. In such cases, the coding of the station was replicated for the missing stations according to the sampling design, and merged accordingly. For parameters that are stable over time, such as bottom type, depth, width, etc., the mean station value for that station (for all years) was inserted. If the missing value was assumed to be from highly variable parameters such as nitrate concentration or organic load, no replacement value was substituted for the missing value.

Overall, the above data substitutions allowed us to eliminate some of the gaps in the data which, if not done and because of the list wise deletion of missing values in multivariate models, would result in the serious loss of degrees of freedom. The trade-off is that an unaccountable amount of precision is lost in the analysis because mean values may not represent adequately the yearly fluctuation in that variable, particularly if it is considerable.

## Zero counts.

Lack of capture of salmon or trout at a station, or the absence of a particular year class at a station during sampling event should not be treated as missing values. However, there is no coding of the absence of a species or year class at a station in the data sets, for the obvious reason. Therefore, a program for checking for zero counts was written, and these values, once detected, were inserted in the data set. It is still possible to have missed some counts if, for example, no fish were caught at a station on a particular date, and no entry into the computer file was made. It seems however that there were always fish of one species or another captured each and every time, but even though this is true historically, it is still worthwhile keeping in mind for future work. Records of sampling events should always be made, whether fish are captured or not. A significant number of zero counts in a data set (such as would be expected in a highly grouped species such as herring) increases the complexity of the analysis, and may result in biases in regression analysis.

## 3. Pseudo population estimates.

Zippin population estimates require that a precise set of assumptions are met, the most important being that the successive number of capture sweeps in a sampling event result in successively fewer fish captured per sweep. This is not necessarily the case in at least half the samples, particularly if year classes are treated separately, but is less likely to occur in the combined year class samples. Working with the Northeast Trepassey data, I was able to demonstrate that a linear relationship exists between the subset of precise Zippin population estimates (those with small standard errors) and the total number of fish caught at that station (Fig. 1). Assuming a linear relationship holds for all rivers, a simple correction factor can be applied to the total catch data to get reliable population estimates. To obtain a correction factor for the total catch at each station, we multiplied the total capture with the slope of the true original Zippins regreased against the total captures at a station. The regression was weighted by the following 2 methods: 1) by removing all points whose Zippin confidence intervals for the population estimate overlapped zero. This produced a slope of 1.22. 2) We also weighed the regression by the inverse of the confidence interval for the Zippin population estimated, which produced a slope of 1.10 . In this analysis, we use the slope of 1.22 derived by method 1) because of the excellent fit to the

Peterson population estimates we obtained when we tested our data (see below). This transformation was applied to all depletion data, including cases where Zippins are impossible. We call these pseudo-Zippins. The advantages of this technique are that 1) it normalized the distribution of population estimates, which were originally highly skewed and with population estimates of questionable quality, 2) it greatly increased the number of usable data points, and 3) it is an unbiased estimator of population size. Possible disadvantages are that the pseudo-Zippin do not take into account the rate of diminishing total capture per sweep, and, if this rate is small (ie. the number of fish caught at each sweep does not decrease quickly), there will be a large difference between the pseudo-Zippin and the actual Zippin. However, keep in mind that in such case, the confidence interval for the Zippin is quite large, very often overlaps zero and always overlaps the pseudo-Zippin.

## 4.Other population estimation methods.

In addition to [pseudo] Zippins, simple mark-recapture (Peterson) and repeated mark-recapture (Schnabel) were used at stations where the depletion technique did not work because of the physical characteristics of the area or because successive recaptures did not result in the reduction of the catches, indicating that depletion was not effective. The two techniques were used without much modification. However, the risk of combining different population estimation techniques is that they yield different values for identical stations or populations, over or underestimating the size of the actual population. We tested for this bias by using data from a series of control stations whose sampling was designed to allow for both Zippin and Peterson mark-recapture population estimates. These stations were sampled by electrofishing apparatus as for the depletion studies, but fish were marked and released at the point of capture after a series of sweeps. The stations were resampled on the following day. Unmarked fish were used as the final capture for the depletion estimate, whereas the proportion of marked to unmarked fish were used for the Peterson estimate. Population estimates were calculated on the combined year classes, making the test conservative (population estimates were also done on individual year classes elsewhere, which results in smaller sample sizes to work with and creates more frequent problems with data). The results of this test are presented herein.

Population estimates for the $\theta^{+}$year classes are unreliable because of the difficulty of capture of such small fish in natural waters. It is almost impossible to get an adequate representation of this size class. However, I have included the 0+ data in the biomass estimates because some data, such as presence or absence, average size, and initial distribution may be useful in some types of analyses.

## Statistical Analyses

Acronyms for variables used in the text are given in Appendix A. Further discussion of the variables themselves can be found in Gibson et al (1987). The stepwise regression method was used to identify possible important variables for further analysis. Statistical analyses were done using linear models exclusively, without the use of quadratic or higher regression terms. The limitation of using linear models is that the possible effect of nonlinear relationships will be either underestimated or missed completely, depending on the nature of the data. However, there are many variables, such as stream width and nitrates, that are expected to behave, within the restricted range observed, as linear covariates of biological factors related to production. Transformations were done where appropriate, either log transforms for skewed distributions or arcsin square root transform for proportions. Most univariate and bivariate distributions were inspected visually. Correlations used for input into models are Pearson's for 2 continuous variables, polyserial or biserial for a continuous and an ordinal variable, and polychoric for 2 ordinal variables. Stepwise regression was the technique used to reduce the number of original variables to fewer important ones, and arrive at preliminary predictive models. A probability of 0.15 was used for entry into the model. Overall probabilities of the models were corrected for bias in the stepwise procedure (Wilkinson 1979). Several methods of doing the stepwise analysis were used to identify important variables, and the model developed using Northeast Brook at Trepassey was compared to parameters and data from other rivers. For the complete data set (which includes Northeast Trepassey) tested by stepwise regression, the biomass estimates are standardized to mean=0 and variance $=1$ within rivers before parameters are estimated.

## RESULTS

## Population Estimation

We first compared the population estimates obtained from the samples which were designed to allow the 3 different methods to be applied concurrently to the same stations. These methods are Zippin, Peterson, and our Pseudo-Zippin. For these samples, we test the hypothesis that the mean population estimates are equal and the correlation between estimates is 1.0 . The mean estimate for Zippin was 212.98 parr ( $\pm 295.4 \mathrm{sd}, \mathrm{N}=67$ ), compared to 110.76 parr ( $\pm 123.4 \mathrm{sd}$, $\mathrm{N}=104$ ) for the Peterson estimate. The mean estimate for the Pseudo-Zippin was 105.4 parr ( $\pm 136.1 \mathrm{sd}, \mathrm{N}=109$ ), very close to the Peterson estimate. Furthermore, when both the Zippin and Pseudo-Zippin are regressed against the Peterson estimate, the latter fit is much better ( $R^{2}=0.56$ for the Zippin vs 0.85 for the Pseudo-Zippin, Fig. 2a,b), and the slope (coefficient of transformation) are 1.84 and 1.03 respectively (a coefficient of 1 is ideal). Our results indicate that, using combined year class data (which increases the individual sample sizes), the pseudo-Zippin method is an acceptable substitute. It substantially increased the number of usable population
estimates and gave comparable results to mark-recapture, unlike Zippin estimates. The Zippin population estimation method is often unreliable for small populations, because of the frequent overestimation and seriously skewed distribution (not shown) observed. We conclude that the Pseudo-Zippin population estimator is suitable for use in combination with Peterson. Although not tested, it is probably also close to the Schnabel estimates as well, since Schnabel is also a type of mark-recapture. Zippin population estimates may not be suitable for difficult-to-capture small populations such as juvenile salmon populations in shallow rocky streams. Further testing of this assumption is required and planned.

## B. Population Estimates

Habitat types were categorized into 5 distinct groups: riffles, runs, flats, pools and ponds. Biomass and density of Atlantic salmon parr varied significantly among habitat types in Northeast Brook at Trepassey (NET) (Table 2, Density: Fs=5.82, $\mathrm{df}=4,53, \mathrm{P}=0.0006$, Biomass: $\mathrm{Fs}=5.74$, $\mathrm{df}=4,53, \mathrm{P}=0.0007$ ). On average, the highest densities were observed in flats ( 0.85 fish $\cdot \mathrm{m}^{-2}$ ) and the lowest by far in Millers pond ( 0.005 fish $\cdot \mathrm{m}^{-2}$ ), then pools at $0.43 \mathrm{fish} \cdot \mathrm{m}^{-}$ 2. Riffles and runs had similar densities ( 0.71 and $0.65 \mathrm{fish} \cdot \mathrm{m}^{-2}$ respectively). Biomass was highest in pools ( $3.22 \mathrm{~g} \cdot \mathrm{~m}^{-2}$ ) and lowest in Millers pond ( $0.02 \mathrm{~g} \cdot \mathrm{~m}^{-2}$ ). Other than Millers pond, flats and runs had low biomass (1.46 and 1.48). Riffles had the second highest biomass. Excluding Millers pond, the densities in NET ranged from 0.2 to 1.5 fish $\cdot \mathrm{m}^{-2}$. Biomass ranged from 0.15 to $7.95 \mathrm{~g} \cdot \mathrm{~m}^{-2}$ (which is the highest biomass observed over all rivers).

When compared to Freshwater River, which had adult salmon introduced for the first time in 1985, a somewhat different pattern emerges. For 1988, densities were highest in riffles ( 0.48 fish $\cdot \mathrm{m}^{-2}$ ) although also lowest in ponds ( 0.01 fish $\cdot \mathrm{m}^{-2}$ ), and equally low in flat and pools ( 0.10 and 0.11 ). Runs were not sampled or were not present in this river. Densities were even more markedly higher in riffles in earlier years, but the average size of the fish is consistently lower than in other habitat types. Some juvenile salmon in Freshwater River had migrated over 4 km upstream from one summer to the next, and other studies have shown downstream migration, indicating that it is not unreasonable to assume that habitat selection, especially microhabitats (such as riffles, flats, etc) is occurring, and that juvenile salmon will select riffles when at low densities or biomass. Overall (all rivers), densities ranged from 0.0 fish $\cdot \mathrm{m}^{-2}$ to 2.44 fish $\cdot \mathrm{m}^{-2}$, and biomass ranged from 0.0 to 7.95 $\mathrm{g} \cdot \mathrm{m}^{-2}$.

## C.Correlations of stream variables to population estimates

Table 3 gives a list of the correlation of the main habitat variables with density and biomass overall. Several variables have large correlation coefficients. Significant correlations are printed in bold. Of the most important physical stream attributes (on a simple linear correlation basis), there is DEBRI, MNIDTH, NPOLS, SORD, SS1, SS2, SS4, TPOOLEN and PTR. Of the water quality attributes, those with highest correlations are: FCON, MAG, NITR, and SULP. Biological parameters with high correlations to density and
biomass include AVLEN, AVWT, and K_MM, but they have not been considered in the regression models presented here. However, they are of biological interest and require further investigation. Our approach in this paper is to identify the physical attributes which influence average size (AVLEN and AVWT) and to use these variables to predict production rather than to base the predictions on average size of the fish itself. Other variables may be important on a nonlinear theoretical basis, specifically those variables which are thought to have optima such as pH , but the multitude of non-linear models possible would greatly complicate the analysis. We decided to build a linear model first, and potential non-linear terms should only be included if the linear model does not succeed.

Many of the stream variables included in this study are highly correlated among themselves, such that high correlations to biological parameters may not be a result of the variable itself, but because of its association with another variable. Tables 4 and 5 represents a correlation matrix of water quality attributes and physical stream attributes respectively. It is evident that several highly correlated sets of variables exist in both cases.

## Habitat Models

The main technique used is stepwise regression. Tables $6,7,8$ and 9 show the results of the analysis using data from Northeast Brook at Trepassey. NITR consistently ranks as the most important parameter explaining biomass variation, explaining between 23 and $38 \%$ of the variance. TPOOLEN and MWIDTH also appear in most models. These are both measures of the dimension of the topographic feature of the stream. TPOOLEN loads positively, while a negative loading typifies MWIDTH. Since the latter measures the width of the stream and TPOOLEN is a measure of length, the sign of the loading would indicate that salmon abundance is positively associated to the banks of the stream and negatively associated to its width, as expected. Further work on predicting juvenile salmon abundance might concentrate on units of biomass as a function of the stream bank (either shoreline area or simply length), rather than the complete station area. There are also statistical reasons why this would be desirable. Substrate type, either as SS1, SS2, SS4, SS5, or SS6, is also consistently included in the habitat model, contributing a significant amount of variance explained. However, the actual substrate variable changes with different models. Other variables that appeared in the models are TPHO, MAG, PHL, COVOV, COND, and NPOLS.

The stepwise habitat models explain between 64 and $79 \%$ of the variance in biomass of juvenile salmon in Northeast Brook at Trepassey.

If we compare this result with the habitat model derived for the entire data set ( 16 distinct rivers including Northeast Trepassey, Table 10), we find that 5 of the 8 variables in that model have been represented in the Northeast at Trepassey model. Most noteworthy, NITR again classes as the most important variable, explaining $11 \%$ of the variance. The total model explains $37 \%$ of the variance, which is quite large considering the complexity and diversity of the various systems tested. However, many degrees of freedom are lost in the analysis because of the often slight differences in experimental design among rivers and among time periods, resulting in pockets of missing values.

## DISCUSSION

The stepwise regression method was able to identify several variables that are likely important determinant of parr distribution in rivers. Although it is limited to linear trends, it is not unreasonable to assume that a series of factor will behave with production estimates in an approximate linear manner in the range observed in Newfoundland rivers. This is particularly true for factors with controlling or limiting effects. The identification of these factors is an important element in the construction of habitat models, and will generate much interest and further field observations and tests. The variables that we have identified as important is certainly not exhaustive, and may also interact with other unmeasured variables that are themselves the important elements. Nevertheless, NITR, MWIDTH, TPOOLEN, MAG, NPOLS, TPHO, and several classes of substrate particle size (ss2, ss3, ss4, ss5 ,ss6) have all been selected as significant contributors of the predictive relationship.

One source of noise in this analysis relates to the dynamics of habitat use by juvenile salmon in relation to total population abundance (Talbot 1989). They found that habitats with the highest densities on any particular year was not necessarily the "best" habitat, indicating that density may not be the best indicator of habitat quality. Furthermore, Talbot (1989) found, while testing the hypothesis that primary habitats are always filled and secondary habitats serve as buffer zones, that some habitat were stable over time while others fluctuated in proportion to total abundance and others still were below the threshold of use for the current population levels. Therefore, further analysis should incorforate indicators of total population abundance, such as the size of the smolt run, the adult run, or egg deposition rates. Habitat quality then becomes a function of density and growth rate, but also one of stability over time (see also Van Horne, 1983).

The analysis of habitat use by juvenile salmon does not stop here. Stepwise regression models were a simple first step in the determination of important habitat variables, but the nature of the relationship and its the causal relationship, as well as the effect of scale (between river differences) all need to be tested empirically and by modelling exercises. One should consider a more sophisticated approach to this data. Firstly, regression analysis of raw data creates several problems, including listwise deletion of missing values. If a station is dropped because of a missing habitat value, then the regression model is based on a different data set than with the station included. This is a major limitation of this technique. A far better approach would be to input a covariance or correlation matrix into a regression model, based on pair-wise (rather than list wise) deletion of missing values, and adjusting the matrix for singularity and instability (negative eigenvalues). In this report, there are several variables which appear to be important but which have too few data points to be used (eg. SVEL). Using the matrix approach would allow us to use these variables. Combining highly correlated variables into factors or "indicator variables" would also help simplify the final analysis, but not the data collection or the sorting of variables and factors.

Secondly, many variables are clearly not normally distributed, and many of the
assumptions of multivariate normality have not been tested or met. There are statistical tools that can be used as substitutes or in addition to the Pearson correlation (for example, Biserial and Polychoric correlations, on continuous, ordinal or truncated distributions).

Thirdly, a structural modelling approach is warranted with this data. Structural modelling involves constructing a theoretical predictive model and testing it with data using a maximum-likelihood approach. Such models can be derived from our expectation of the relationship among stream and biological variables. Lisrel (Jöreskog and Sörbom 1988) is a programming environment that incorporates all the above facets, and would be an ideal testing environment for the habitat data. The applicability of structural models to habitat data has been discussed in Talbot (1990).

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Table 1: Station characteristics measured for this study.

| Physical | Chenical | Substrate | Diological |
| :---: | :---: | :---: | :---: |
| Area of sampling site | Witrate | Soulders | Riparian veg |
| Mean streas vidth | Total alkalinity | Rutble | Fish lensity |
| Mean Depth | Total phosphorus | Coblle | Fish Bionas |
| Mean vater velocity | Total dissolved solids | Pebble | Grouth rate |
| Type of substrate | Hydrogen ions | Gravel | Ser |
| later teaperature | Total hardness | Sand | haturity |
| Ice scour marks | Calciua | Silt |  |
| Ouephead cover | Chloride | Clay |  |
|  | Sulphate | Organic detritus Convoluted bedrock Snooth Redrock |  |
|  | Nater colour |  |  |
|  |  |  |  |

Table 2. Correlations among water quality variables for Northeast Brook at trepassey ( $\mathrm{N}=54$ ). Data are correlated under pairwise deletion of missing values (actual number of data points varies with correlation). Many stream parameters are highly correlated. See discussion on covariance of stream attributes.
alk calc chl colr cond foon hard ag nitr phf phl sulp teap thar tpho alk 1
alt . 58 I
(chl -. $09.46 \quad 1$
colr -. $57-.42-.24 \quad 1$
cond . 09.57 . 98 -. 451
fron . 16 -. 05 -. $83-.14$-. 061
hard . 72 . 92.68 -. 18 . $67-.131$

nitr . 88 -. 18 -. 18 -. $02-.13$. 18 -. 21 -. 02 I
phf . 59.36 . $28-.46$. $29-.23$. 39.41 . 041
phl . 86 . 48 -. 06 -.74 . 86 . 06 . 69 . 12 -. 05 . 661
sulp -. 34 . 17 . 54 . 16 . 46 . $24-.22$. 42.08 -. 58 -. 441


tpho -. 85 -. $84-.17 \quad .17-.13$. $09-.25-.13 \quad .26-.09-.12-.11 \quad .09-.031$


Table 3. Correlation among physical stream attributes for Northeast Brook at Trepassey, excluding Miller's pond ( $\mathrm{N}=54$ ). Data are correlated under pairwise deletion of missing values (actual number of data points varies with correlation). Many stream parameters are highly correlated. See discussion on covariance of stream attributes.
area couchn covif covoy mebri meepsp fcom icescar mep mlen nel mudith mpols

| AREA | 1.888 |  |  |  |  |  |  |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| COVCAM | -8.247 | 1.088 |  |  |  |  |  |  |  |  |  |  |  |
| covil | 8.859 | -1.058 | 1.888 |  |  |  |  |  |  |  |  |  |  |
| covo | -8.357 | 8.335 | 8.331 | 1.088 |  |  |  |  |  |  |  |  |  |
| DEBRI | 8.487 | -8.181 | 1.101 | 0.815 | 1.888 |  |  |  |  |  |  |  |  |
| deepsp | 1.672 | -8.467 | 8.831 | -8.468 | 8.496 | 1.888 |  |  |  |  |  |  |  |
| FCOHD | 0.168 | 8.118 | 8.121 | 8.881 | 8.262 | \$.1996 | 1.888 |  |  |  |  |  |  |
| Icescar | 8.245 | 8.455 | 8.246 | 8.838 | 1.939 | 8.123 | 8.366 | 1.088 |  |  |  |  |  |
| HDEP | 1.769 | -8,369 | 8.832 | -8.397 | 8.519 | 1.832 | -8.887 | 8.055 | 1.108 |  |  |  |  |
| MEE | 1.647 | 8.180 | -8.887 | -8.124 | 0.021 | 8.285 | 8.249 | 1.183 | 8.395 | 1.818 |  |  |  |
| nuel | 8.277 | 8.142 | -8.193 | -8.162 | -8.156 | -8.174 | -0.171 | -8.868 | 8.288 | 8.187 | 1.808 |  |  |
| Muidith | 0.763 | -8.284 | 8.817 | -0.415 | 0.785 | 1.618 | -0.823 | 8.216 | 1.668 | 8.861 | 8.266 | 1.888 |  |
| HPOLS | 1.753 | -8.361 | 8.891 | -0.367 | 0.191 | 0.769 | 8.115 | 1.881 | 1.629 | 2.533 | -8.138 | 8.465 | 1.808 |
| PTR | -8.396 | 8.813 | 8.866 | 8.263 | -0,858 | -8.239 | 8.149 | -8.417 | -8.447 | 1.887 | -8.244 | -8.621 | -8.861 |
| SORD | 0.511 | -8.183 | -8.229 | -8.312 | 1.858 | 8.328 | -8.149 | 1.338 | 1.511 | -1.023 | 0.336 | 1.796 | 8.128 |
| \$51 | 1.585 | -8.282 | -8.837 | -8.221 | 8.214 | 6.633 | 8.133 | 8.129 | 8.412 | 8.449 | -8.148 | 8.335 | 0.782 |
| 552 | 8.131 | 8.883 | 8.281 | 8.874 | 8.471 | 8.887 | 8.897 | 8.298 | -8.852 | 8.148 | -8.252 | 8.886 | 8.128 |
| 553 | -9.345 | 8.269 | 0.011 | 0.165 | 0.826 | -8.305 | -18.288 | 0.142 | -8.357 | -8.177 | -8.202 | -8.301 | -0.196 |
| 554 | -8.336 | 8.878 | -8.283 | 8.132 | -1.321 | -8.663 | -8.258 | -8.116 | -8.114 | -8.165 | 0.828 | -8.368 | -8.888 |
| S5S | 8.229 | 8.086 | 8.831 | -8.872 | 8.384 | 8.826 | B. 182 | 0.182 | 8.272 | 8.422 | 8.693 | 8. 058 | -8.885 |
| \$56 | 8.287 | -8,043 | 0.223 | 1.882 | -8.836 | -0.815 | 8.156 | -8.128 | 8.833 | -8.844 | 2. 158 | 8. 325 | -8.857 |
| SUlSTR | 0.134 | -8.893 | 0.189 | -8.184 | -8.156 | -9.029 | 8.234 | - 0.113 | 1.169 | -1.017 | 8.223 | 8.237 | -1.129 |
| SUEL | -8.495 | -8.892 | -8.488 | -8.115 | . | 1.614 | . | . | 8.879 | -8.337 | . | -4.871 |  |
| TEMP | 0.346 | -1.276 | -1.193 | -0.185 | 1.358 | 1.373 | -1.276 | 1.839 | 8.317 | - 1.178 | -8.035 | 8.528 | 8.263 |
| TPPOLEM | 1.788 | -9.352 | 8.859 | -8.386 | 8.191 | 1.798 | 0.154 | 8.881 | 1.681 | 0.571 | -8.171 | 8.461 | 1.898 |
|  | 8.246 | -8.265 | 8.854 | -0.135 | 8.181 | 8.212 | -8.183 | 0.185 | 8.133 | -9.186 | -8.088 | 8.429 | 8.138 |



| PIR | 1.868 |  |  |  |  |  |  |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| S0RD | -0.776 | 1.888 |  |  |  |  |  |  |  |  |  |  |  |
| \$51 | -0.048 | 8.113 | 1.888 |  |  |  |  |  |  |  |  |  |  |
| \$52 | -9.166 | 8.839 | 0.388 | 1.808 |  |  |  |  |  |  |  |  |  |
| 553 | 8.865 | -0.142 | 0.897 | 1.912 | 1.088 |  |  |  |  |  |  |  |  |
| 554 | 8.186 | -1.136 | -1.118 | -1.881 | 1.341 | 1.108 |  |  |  |  |  |  |  |
| SS5 | -9.119 | 8.056 | -8.248 | -1.256 | -1.885 | -8.413 | 1.018 |  |  |  |  |  |  |
| S56 | 8.167 | 8.868 | -1.198 | -1.255 | -1.511 | -8.768 | 0.166 | 1.081 |  |  |  |  |  |
| SUBSTR | 8.822 | 8.874 | -0.302 | -1.513 | -1.782 | -1.638 | 0.491 | 1.832 | 1.088 |  |  |  |  |
| SVEL | . | . | 8.153 | 1.44 | 0.351 | 8.338 | -8.419 | -0.288 | -1.417 | 1.809 |  |  |  |
| TENP | -8.865 | 0.414 | 1.368 | -1.028 | -1.863 | -8.035 | -8.284 | 0.138 | -8.018 | 1.796 | 1.888 |  |  |
| TPOOLEW | -8.859 | 8.117 | 1.785 | 0.899 | -1.227 | -8.821 | 1.119 | -0.447 | -1.188 | -1.364 | 8.264 | 1.018 |  |
| HTEMP | 8.813 | 0.362 | 0.183 | -8.113 | -8.188 | -1.222 | -1.135 | 1.222 | 1.138 | -1.754 | 1.873 | 8.136 | 1.888 |

Table 4: Correlation between density and biomass with physical stream attributes.

|  | Density (\# $\mathrm{m}^{-2}$ ) | Biomass ( $\mathrm{g} \mathrm{m}^{-2}$ ) |
| :---: | :---: | :---: |
| ALK | 0.09 (54, 0.5111) | -0.04 (54, 0.7916) |
| AVLEN | -0.36 (54, 0.0072) | 0.57 (54, 0.0001) |
| AVWT | -0.36 (54, 0.0071) | 0.58 (54, 0.0001) |
| CALC | 0.11 (54, 0.4216) | 0.09 (54, 0.5052) |
| CHL | -0.18 (54, 0.18) | -0.37 (54, 0.006) |
| COLR | 0.13 (46, 0.3837) | $0.04(46,0.7723)$ |
| COVCAN | 0.19 (54, 0.1633) | -0.10 (54, 0.4927) |
| COVIN | -0.03 (54, 0.8241) | -0.00 (54, 0.9960) |
| COVOV | 0.15 (54, 0.2766) | -0.06 (54, 0.6888) |
| DEBRI | -0.08 (24, 0.71) | -0.77 (24, 0.0001) |
| DEEPSP | -0.26 (52, 0.0582) | 0.06 (54, 0.6895) |
| FCON | -0.55 (18, 0.018) | 0.46 (18, 0.054) |
| HARD | 0.15 (37, 0.3688) | -0.19 (37, 0.2511) |
| K | -0.02 (54, 0.8894) | 0.11 (54, 0.4329) |
| K_MM | -0.32 (7, 0.4882) | -0.57 (7, 0.1761) |
| ICESCAR | 0.12 (37, 0.4847) | -0.25 (37, 0.1382) |
| MAG | -0.04 (54, 0.07891) | -0.27 (54, 0.0494) |
| MDEP | -0.21 (52, 0.1421) | -0.16 (52, 0.2656) |
| MVEL | 0.06 (48, 0.6615) | -0.17 (48, 0.2622) |
| MWIDTH | -0.32 (54, 0.0141) | -0.32 (54, 0.019) |
| NITR | 0.37 (54, 0.005) | 0.60 (54, 0.0001) |
| NPOLS | -0.38 (54, 0.0052) | 0.12 (54, 0.3689) |
| PHF | 0.05 (50, 0.7508) | -0.05 (50, 0.7072) |
| SORD | -0.14 (54, 0.3118) | -0.45 (54, 0.0006) |
| SS1 | -0.37 (54, 0.0063) | -0.05 (54, 0.7213) |
| SS2 | -0.29 (54, 0.036) | -0.33 (54, 0.0155) |
| SS3 | 0.18 (54, 0.1848) | -0.22 (54, 0.1106) |
| SS4 | $0.38(54,0.005)$ | 0.04 (54, 0.7569) |
| SS6 | -0.12 (54, 0.3795) | 0.17 (54, 0.2318) |
| SS5 | -0.09 (54, 0.5233) | 0.02 (54, 0.8970) |
| SULP | -0.28 (54, 0.0393) | -0.32 ( $54,0.017$ ) |
| SVEL | 0.54 (6, 0.27) | 0.36 (6, 0.49) |
| THAR | -0.10 (17, 0.6919) | 0.03 (17, 0.8941) |
| TPHO | 0.18 (54, 0.1929) | 0.25 (54, 0.0692) |
| TPOOLEN | -0.37 (54, 0.0056) | 0.14 ( $54,0.3087$ ) |
| TURB | -0.03 (54, 0.8064) | -0.01 (54, 0.9690) |
| PTR | 0.17 (54, 0.21) | 0.65 (54, 0.0001) |

Table 5. Mean biological characteristics of juvenile atlantic salmon in the main habitat types (HABTYPE) in the Northeast Brook at Trepassey.

| TYPE | N | DENSITY <br> $\left(\# \cdot \mathrm{~m}^{-2}\right)$ | BIOMASS <br> $\left(\mathrm{g} \cdot \mathrm{m}^{-2}\right)$ | LENGTH <br> $(\mathrm{mm})$ | WEIGHT <br> $(\mathrm{g})$ |  |
| :--- | ---: | :--- | :--- | :--- | :--- | :--- |
|  |  |  |  |  |  |  |
| RIFFLE | 36 | $0.71(.2-1.5)$ | $2.56(1.1-7.9)$ | 66.3 | 4.79 |  |
| FLAT | 1 | 0.85 | 1.46 | 57.4 | 3.32 |  |
| RUN | 10 | $0.65(.3-1.3)$ | $1.48(.2-2.5)$ | 63.3 | 3.96 |  |
| POOL | 7 | $0.43(.3-.9)$ | $3.22(2.3-6.8)$ | 93.0 | 9.20 |  |
| POND | 4 | $0.005(.003-.006)$ | $0.02(.01-.03)$ | 86.8 | 8.62 |  |

TABLE 6. Stepwise regression analysis for physical stream attributes, using total density and biomass and complete variables (no missing values). The entire model is significant ( $\mathrm{R}^{2}=0.67, \mathrm{f}_{5} 15.98, \mathrm{df}=6,47, \mathrm{P}=0.0001$ ).

| VARIABLE | PARAMETER <br> ESTIMATE | PARTIAL <br> $R^{2}$ | MODEL <br> R | F | P |
| :--- | :--- | :--- | :--- | :--- | :--- |
|  |  |  |  |  |  |
| INTERCEPT | 6.94 | 0.0 | 0.0 | 26.85 | 0.0001 |
| NITR | 0.19 | 0.36 | 0.36 | 29.40 | 0.0001 |
| SS2 | -3.64 | 0.10 | 0.46 | 9.46 | 0.0034 |
| TPOOLEN | 0.42 | 0.06 | 0.52 | 6.30 | 0.0154 |
| TPHO | 0.003 | 0.05 | 0.58 | 6.18 | 0.0164 |
| SS4 | -2.26 | 0.05 | 0.62 | 6.14 | 0.0168 |
| MWIDTH | -0.88 | 0.05 | 0.67 | 6.84 | 0.0119 |

TABLE 7. Stepwise regression analysis for physical stream attributes, using density and biomass for the $2^{\dagger}$ year class, and using complete variables (no missing values). The entire model is significant ( $\mathrm{R}^{2}=0.64, \mathrm{~F}=9.76, \mathrm{df}=8,44, \mathrm{P}=0.0001$ ).

| VARIABLE | PARAMETER <br> ESTIMATE | PARTIAL <br> $R^{2}$ | MODEL <br> $R^{2}$ | $F$ | P |
| :--- | :---: | :---: | :---: | :---: | :---: |
|  |  |  |  |  |  |
| INTERCEPT | -1.49 | 0.0 | 0.0 | 0.75 | 0.39 |
| NITR | 0.074 | 0.23 | 0.23 | 14.94 | 0.0003 |
| MAG | -0.045 | 0.12 | 0.34 | 8.80 | 0.0046 |
| PHL | 0.010 | 0.09 | 0.44 | 8.23 | 0.0061 |
| TPOOLEN | 0.26 | 0.06 | 0.50 | 5.94 | 0.0186 |
| SS5 | 0.91 | 0.05 | 0.54 | 4.67 | 0.0358 |
| MWIDTH | -0.72 | 0.03 | 0.57 | 2.94 | 0.0931 |
| SS6 | 0.91 | 0.05 | 0.62 | 5.34 | 0.0255 |
| COVOV | -0.84 | 0.02 | 0.64 | 2.77 | 0.1031 |

TABLE 8. Stepwise regression analysis for physical stream attributes, using total density and biomass, and allowing into the model some potentially important variables with missing values. The entire model is significant ( $\mathrm{R}^{2}=0.79, \mathrm{f}=21.61, \mathrm{df}=7,40, \mathrm{P}=0.0001$ ).

| VARIABLE | PARAMETER <br> ESTIMATE | PARTIAL <br> $R^{2}$ | MODEL <br> R | F | P |
| :--- | :---: | :---: | :---: | ---: | :---: |
|  |  |  |  |  |  |
| INTERCEPT | 6.11 | 0.0 | 0.0 | 27.84 | 0.0001 |
| NITR | 0.17 | 0.38 | 0.38 | 28.03 | 0.0001 |
| TPOOLEN | 0.33 | 0.09 | 0.47 | 7.74 | 0.0079 |
| SS1 | -0.49 | 0.11 | 0.58 | 11.27 | 0.0016 |
| COND | -0.09 | 0.07 | 0.65 | 8.37 | 0.0060 |
| NPOLS | -2.53 | 0.08 | 0.73 | 12.82 | 0.0009 |
| SS6 | 0.021 | 0.02 | 0.74 | 2.45 | 0.1255 |
| MWIDTH | -0.15 | 0.05 | 0.79 | 8.84 | 0.0050 |

TABLE 9. Stepwise regression analysis for physical stream attributes, using density and biomass for the $2^{4}$ year class, and allowing into the modes some potentially important variables with missing values. The entire model is significant ( $\mathrm{R}^{2}=0.75, \mathrm{~F}_{\mathbf{5}} 19.56$, $\mathrm{df}=6,40, \mathrm{P}=0.0001$ ).

| VARIABLE | PARAMETER <br> ESTIMATE | PARTJAL <br> $R^{2}$ | MODEL <br> $R$ | F | P |
| :--- | :---: | :---: | :---: | ---: | :--- |
|  |  |  |  |  |  |
| INTERCEPT | -0.68 | 0.0 | 0.0 | 0.19 | 0.66 |
| NITR | 0.07 | 0.24 | 0.24 | 13.88 | 0.0005 |
| TPOOLEN | 0.29 | 0.18 | 0.41 | 13.32 | 0.0007 |
| NPOLS | -2.73 | 0.13 | 0.55 | 12.80 | 0.0009 |
| MAG | -0.05 | 0.14 | 0.68 | 18.20 | 0.0001 |
| PHL | 0.007 | 0.03 | 0.71 | 4.06 | 0.0506 |
| SS1 | -0.37 | 0.03 | 0.75 | 5.17 | 0.0285 |

Table 10. Stepwise regression for all rivers combined, using biomass estimates standardized within rivers and raw or transformed stream attributes. The complete model is significant ( $\mathrm{R}^{2}=0.37, \mathrm{~F}_{5}-4.80, \mathrm{df}=8,66, \mathrm{P}=0.0001$ ).

| VARIABLE | PARAMETER <br> ESTIMATE | PARTIAL <br> $\mathrm{R}^{2}$ | MODEL <br> R | F | P |
| :--- | :--- | :--- | :--- | :--- | :--- |
|  |  |  |  |  |  |
| INTERCEPT | -0.37 | 0.0 | 0.0 | 0.30 | 0.58 |
| NITR | 0.053 | 0.11 | 0.11 | 8.61 | 0.0045 |
| TURB | 0.011 | 0.07 | 0.17 | 5.87 | 0.0179 |
| TPHO | 0.003 | 0.05 | 0.22 | 4.43 | 0.0389 |
| SS1 | 1.44 | 0.03 | 0.25 | 3.00 | 0.0876 |
| MWIDTH | -0.74 | 0.03 | 0.29 | 3.17 | 0.0794 |
| CALC | 0.012 | 0.03 | 0.32 | 2.91 | 0.0926 |
| COVCAN | -1.03 | 0.03 | 0.35 | 3.05 | 0.0854 |
| SS2 | -1.35 | 0.02 | 0.37 | 2.33 | 0.1314 |



Figure 1: Relationship between true Zippin population estimates and total catch for juvenile Atlantic salmon in Northeast River at Trepassey. The regression line represents the slope of the regression ( $b=1.22$ ) for the subset of Zippin estimates whose confidence interval did not overlap zero. All outliers above the regression lines had confidence limits overlaping zero.


Figure 2: Relationship between a) True Zippin population estimates and b) "Pseudo-2ippins", against Peterson mark-recapture population estimates for the test stations which allowed both methods to be calculated. It is evident from the correlation and comparison of the means (see text) that Pseudo-Zippins are much more tightly related to Peterson's than true Zippins. It seems that Zippin estimates are unsuitable for this type of data.

Appendix A. List of codes contained in the text. See Gibson et al (1987) for details.

| VARIABLE | TR DRFINITION |
| :---: | :---: |
| ALK | Total alkalinity |
| AREA | Station area |
| AVLEN | Average length of the fish sampled |
| AVWT | Average weight of the fish sampled |
| BIODENS | Biomass estimate/area of the station |
| CALC | Calcium |
| CHL | Chlorine |
| COLR | Colour |
| COND | Specific conductivity |
| COVCAN | Canopy cover |
| COVIN | Instream cover |
| COVOV | Overhanging cover (shade) |
| DEBRI | Debris |
| DEEPSP | Deepest spot at a station |
| FCOND | Field conductivity |
| HABTYPE | Habitat type |
| HARD | Water hardness |
| HGROUP | Habitat grouping |
| ICESCAR | Height of ice scar |
| K | Condition factor |
| K_MM | Condition factor of mature males |
| MAG | Magnesium |
| MANG | Manganese |
| MLEN | Stream station length |
| MWIDTH | Stream station width |
| NITR | Nitrates |
| NPOLS | Number of pools at a station |
| NUM_MM | Number of mature males |
| ORPH | Ortho-phosphates |
| PHF | Field pH |
| PHL | Lab pH |
| PTR | Primary Tributary number (starting from the mouth) |
| SORD | Stream order |
| SS1 | Substrate type 1 percentage (see Gibson et al 1987) |
| 552 | Substrate type 2 percentage |
| SS3 | Substrate type 3 percentage |
| SS4 | Substrate type 4 percentage |
| SS5 | Substrate type 5 percentage |
| SS6 | Substrate type 6 percentage |
| SUBSTR | Substrate rating |
| SULP | Sulphates |
| SVEL | Stream velocity at a station |
| TEMP | Temperature |
| THAR | Total hardness |
| TPHO | Total phosphates |
| TPOOLEN | Total pool length |
| TURB | Turbidity |
| WTEMP | Water temperature |
| YRCLASS | Year class |

## Session II

Use of Habitat Suitability Curves and Other
Models to Estimate Changes in Productivity of Fish Habitat

# Habitat Suitability Curves for Atlantic Salmon (Salmo salar L.) in Insular Newfoundland and the Potential Applicability of Suitability Indices to Habitat Management 

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#### Abstract

Habitat suitability curves (HSCs or HSIs) are relationships between a species and the habitat variables believed to be important to the well being of that species. In Newfoundland, HSCs and HSC based models are being considered for use to help standardize, where possible, habitat management decisions based on professional judgements. Data are presented from 18 rivers ( 242 stations) in insular newfoundland. The habitat variables explored are mean stream width, mean depth, station velocity, station discharge, proportion of instream and overhanging cover, ice scour height, and proportions of substrate types. Histograms with curve smoothing techniques were used to develop a curve to relate the indicator variable (number of fish per $100 \mathrm{~m}^{2}$ ) to the range in each habitat variable. HSCs for Atlantic salmon fry were consistent, suggesting a well defined set of habitat preferences. Parr curves demonstrated more variability with the most consistent curves for the variables stream width, mean depth, and discharge. Curves developed in this paper contain a number assumptions and limitations that may constrain their potential application. Problems of lack of independence among habitat variables and methods to aggregate HSCs to form more complex models are discussed. Alternative approaches to curve development are reviewed. Potential application of HSCs and HSC based models to habitat management practices is also discussed.


## RESUME

Les courbes de salubrité de 1'habitat (HSC ou HSI) sont des relations entre une espèce et les variables de l'habitat que l'on croit importantes à son bien-être. On étudie la possibilité d'utiliser des HSC et des modèles basés sur des HSC pour normaliser, si possible, les décisions relatives à la gestion des habitats de Terre-Neuve basées sur des opinions professionnelles. On présente des données concernant 18 cours d'eau ( 242 stations) de Terre-Neuve dont la largeur et la profondeur moyennes, le débit et la vitesse d'écoulement à la station échantillonnée, le pourcentage de couverture végétale, dans le cours d'eau et surplombant le cours d'eau, la profondeur de l'affouillement et le pourcentage des divers substrats. À l'aide d'histogrammes et de méthodes de lissage de courbes, on a obtenu une courbe de relation entre la variable indicatrice (le nombre de poissons par $100 \mathrm{~m}^{2}$ ) et l'écart de chaque variable de l'habitat. Les HSC des alevins de saumon de 1'Atlantique sont régulières, ce qui porte à croire qu'ils ont des préférences d'habitat marquées. Toutefois, les courbes des tacons sont plus variables; les courbes les plus régulières concernent les variables largeur moyenne du cours d'eau, profondeur moyenne du cours d'eau et débit. Les courbes présentées reposent sur un certain nombre d'hypothèses et comportent certaines limites qui peuvent restreindre leur application potentielle. On examine les problèmes du manque d'indépendance entre les variables de l'habitat et de la pénurie de méthodes d'intégration des HSC pour élaborer des modèles plus complexes. On passe aussi en revue d'autres techniques d'élaboration de courbes et on examine l'application potentielle des HSC et des modèles basés sur des HSC à la gestion de l'habitat.

## INTRODUCTION

Currently, freshwater fish habitat in Newfoundland is managed predominantly in a quantitative context (number of habitat units) with some general qualitative considerations (habitat types, general quality of the habitat, etc.). Qualitative considerations are subjective and tied to professional judgement which leads to lack of standardization in decision making between projects, individuals charged with making evaluations, and between geographical regions. There exists a need to develop habitat evaluation methodologies to effectively determine habitat losses/gains, and these methods must be such that the Department is confident that conservation of the productive capacity of fish habitat is being achieved. Any approach that is developed must offer flexibility and alternatives in finding solutions for habitat compensation, mitigation, etc.

Indices of the suitability of habitat or HSIs, based on individual or aggregated habitat parameters, have been developed and used extensively in evaluating potential impacts of habitat alterations for aquatic and terrestrial species (Bovee and Zuboy 1986). Habitat suitability is the relationship between a species and habitat variables believed important to the growth, survival, standing crop (population density), or other expression of the well being of that species. The mathematical expression of the relationship varies from 1.0 (optimum habitat condition for this variable) to 0.0 (unsuitable habitat with minimal/no potential for use by the species). HSIs for a variety of important variables can be aggregated into an overall suitability index (SI) through a habitat suitability index model. A virtue of the HSI approach, when integrated with the Habitat Evaluations Procedures (or HEP), is that habitat is managed on the basis of quality and quantity and not in relation to use of the habitat, which can be effected by many other influences (eg. exploitation, escapement, competition, etc.). Habitat suitability indices and HSI models are used to simplify habitat/species relationships and to develop tools to apply existing knowledge to problem solving. HSIs have inherent assumptions and limitations and these must be clearly understood and considered by developers and users of habitat suitability criteria.

In this paper habitat suitability indices (HSIs) or curves (HSCs) are developed for Atlantic salmon juveniles from fish population and habitat (macrohabitat) attribute data collected from sampling stations on insular Newfoundland rivers. Alternative approaches to developing suitability relationships are briefly discussed. The potential use of habitat suitability indices (and models) in habitat evaluation and management is also considered.

## MISTHODS

The data set employed in the development of habitat suitability relationships (curves) for juvenile Atlantic salmon includes fish abundance and habitat attribute data from 242 stations from 18 rivers in insular Newfoundland. These data was collected over the period from 1980 to 1988 by Dr. R. J. Gibson of the Freshwater and Anadramous Fish Division, Science Branch, Newfoundland Region. Table 1 contains a listing of rivers selected, number of stations included, and the period of sampling. The large data set was considered as 2 data sets; one that included all stations from the 18 diverse rivers (ALL, $n=242$ ) and a geographical subset of this larger data set for 5 rivers included in the Experimental Rivers program on the southwest Avalon Peninsula (EXP. RIV., $\mathrm{n}=156$ ).

The indicator variable used in curve development was numbers of fish of a life stage (fry, or all parr) per unit of habitat ( $100 \mathrm{~m}^{2}$ ). Parr were not partitioned into size and/or age groups. Fish were collected from stations ranging from 52 to $657 \mathrm{~m}^{2}$ ( 0.52 to 6.57 units) by electrofishing, seining, or combinations of the two methods. Numbers of fish per habitat unit were actual total catches and were not estimates of population size. In order to maximize the size of the data set employed in curve development, it was decided to use actual catch data, as population estimates were not available for a large part of the data set and, in some instances, estimates were not calculated (partitioned) by life stage (fry/parr).

The habitat attributes included in the data analysis included measurements of stream width, depth (mean and maximum), velocity (mean velocity and surface velocity), discharge, cover (instream, overhanging, and canopy), ice scar (scour) height, and proportions of substrate types. Chemical variables were not used to evaluate habitat preference for those parameters in this paper. Measurements of attributes of pool habitat were also available (number of pools, pool. length) however these could not be related quantitatively to area/proportion of pool type habitat in each station or to some pool rating approach (eg. Binns and Eiserman 1979) and consequently were not be employed in curve development. Details on how these variables were collected are available in Gibson et al. (1987). It should be recognized that these data were not collected for the purpose of developing habitat suitability criteria.

In all cases habitat attribute data were treated as collected with the exception of proportion of substrate types within each station. Unlike habitat suitability curves developed from data collected for individual fish
(individuals are associated with one substrate type), it was necessary to combine the proportions of the different substrate types within each station into one measure or index. An approach was adopted that employed a weighting scheme in relation to the coarseness of substrate. The rating for each substrate type was determined from $\log _{10}$ of the mean particle size of that type as follows:
(1) coarseness rating $=\log _{1 \circ}$ (mean particle size +1 )

The coarseness rating for each substrate type was multiplied by the proportion (\%) of each type in the station and the totals for gravels, pebble, cobble, rubble, and boulders were summed to give an overall rating for the station. Bedrock and fines (sands through organic detritus) were considered unsuitable substrates and were not included in determining the coarseness rating.

Normally, when curves are developed from individual data (observations of individuals in relation to a habitat variable), both habitat use and habitat preference (habitat use adjusted in relation to availability) need to be determined. In this paper, the indicator variable (numbers of fish/unit) is adjusted relative to available habitat (divided by the station area) consequently, the resulting curves are considered to reflect habitat suitability or preference.

Frequency analysis (histograms) relating the indicator parameter (\# fish/unit area) to the range in the habitat variable is used to express habitat preference/suitability. The bin (interval) size for each variable was established, as recommended by the U.S. Fish and Wildlife Service (Cheslak and Garcia 1988) in relation to the sample size and range in measured values of each habitat parameter, through use of the Sturge equation (Sommerville 1958) as follows:

$$
\text { (2) } \begin{aligned}
\mathrm{C}=\mathrm{R} /(1 & \left.+3.322 * \log _{10} \mathrm{~N}\right) \\
\text { where } \mathrm{C} & =\text { width of interval } \\
\mathrm{R} & =\text { range of measurements of the habitat variable } \\
\mathrm{N} & =\text { sample size }
\end{aligned}
$$

Use/preference curves were developed using the (common) average performance approach (Bovee and Cochnauer 1977). Curves are developed separately for two life stages of salmon; fry (YOY, juveniles during their first summer of life) and parr (PARR, juveniles during their second and subsequent, up to fourth, year of life with visible parr markings). The range (or interval) of each habitat variable most frequently used by each life stage (fry, parr), as indicated by the abundance data, is considered to be optimum habitat (Bovee and Cochnauer 1977) and is assigned a habitat suitability of 1.0. All other values were assigned a suitability index between 1.0 and 0.0 based on their frequency relative to the optimum range. Values with a suitability greater than 0.90 were, by convention, assigned a suitability of 1.0 . The suitability curve was constructed by connecting the corners of the bins expressing suitability of 1.0 (peak of the curve) and drawing a line to the mid-point of adjacent bins (Slauson 1988). On the tails of the distribution, the curve is extended to the end of the data range. Figure 1 displays how this approach was
used to develop the suitability curve for stream width. Curve smoothing using a 3-way running mean filter (no more than 2 passes) was applied, where appropriate, to assist in defining monotonic or unimodal curves (Bovee 1986). Figure 2 demonstrates how the suitability curve was developed for discharge, using 2 passes of a 3 -point running mean filter.

## RESULTS

A statistical summary of the habitat attribute data and fish abundance data for the 2 data sets (ALL data and the EXP. RIV. data) is listed in Tables 3 and 4. Histograms of the distributions of the habitat variables for the 2 data sets are presented in Figures 3 and 4. The distributions of the habitat variables, used to define habitat availability, were compared statistically using the Wilcoxon Rank Sum test (SAS Institute Inc., 1985). The test indicated (at $\mathrm{p}=0.10$ ) that the distribution of the habitat variables in the 2 data sets was not significantly different for most habitat attributes with the exception of the variables area, discharge, maximum depth, ice scar height, pool number, pool length, overhanging cover, canopy cover, \% gravel, \% pebble, and substrate rating. The fish abundance variables were not significantly different (at $\mathrm{p}=0.10$ ).

Suitability curves (for fry and parr) were developed for eight habitat variables, separately for both data sets. Curves for stream width, mean depth, mean velocity and discharge are contained in Figure 5 while curves for ice scar height, instream cover, overhanging cover, and substrate rating are contained in Figure 6. Five of the variables could be considered measures of morphometric/hydrological conditions (width, depth, velocity, discharge, ice scar height) while the other three attributes are considered cover variables (instream cover, overhanging cover, substrate rating).

## Width

The fry suitability curve for stream width demonstrates a preference of fry for small streams with optimum suitability in the range less than 2.5 m and a rapid decline in suitability after that interval, with stream widths of greater than 17.5 m demonstrating poor to no suitability (HSI of 0.0 ). The curve for parr is similar with a wider optimum range, from 0.0 to 5 m , and a more gradual decline in suitability to reach 0.0 at 22.5 m (ALL data) or 17.5 $m$ (EXP. RIV. data). Curves developed from the two data sets were extremely consistent with the extended tail of the parr suitability curve for ALL data being a result of having data points at these widths from which to develop the curve.

## Depth

The depth (mean depth for the station) suitability curves for fry demonstrates optimum suitability in the range of 15 to 20 cm with a steady decline in suitability greater than 20 cm to reach an HSI of 0 at 65 cm (ALL data) or 52.5 cm (EXP. RIV. data). The optimum depth range for parr is from 15 to 25 cm and the parr HSIs demonstrate a more gradual decline in suitability at depths greater than 25 cm to reach 0.4 (ALL) or 0.58 (EXP. RIV.) at 57.5 cm . Depth suitability for parr demonstrated a declining trend in the higher ranges,
however it would be difficult to extrapolate suitability beyond the available data. The curves for both life stages demonstrate very sharp declines in suitability in the lower ranges ( $<10 \mathrm{~cm}$ ). The curves developed from the 2 data sets demonstrate very similar trends with the EXP. RIV. data demonstrating higher parr suitability in the depth ranges greater than 25 cm .

## Velocity

Velocity (mean station velocity) suitability curves for both fry and parr demonstrated a wide range in optima reflecting a fairly flat response of the indicator variable (numbers of fish per unit habitat) to mean station velocity. Optimum suitability for fry ranged from $0.20 \mathrm{~m} / \mathrm{s}$ to $0.60 \mathrm{~m} / \mathrm{s}(0.70$ $\mathrm{m} / \mathrm{s}$ for EXP. RIV.) with sharp declines in suitability on either end of this range. Optimum parr suitability was in the range of 0.10 to $0.50 \mathrm{~m} / \mathrm{s}$ with declines in suitability at velocities greater than $0.50 \mathrm{~m} / \mathrm{s}$. The lower velocity range, 0 to $0.10 \mathrm{~m} / \mathrm{s}$, demonstrated higher suitability for parr ( 0.79 ) than for fry ( 0.25 to 0.33 ). Again, curves developed from the 2 data sets were consistent in shape and range of optima, with the exception of the sharp decline in suitability for parr at velocities greater tan $0.5 \mathrm{~m} / \mathrm{s}$ for the EXP. RIV. which was not as evident in the larger data set (ALL).

## Discharge

Suitability curves for discharge for both life stages demonstrated identical shape and range of optima. Preferred discharge was at the low end of the range, 0.0 to $0.25 \mathrm{~m}^{3} / \mathrm{s}$, with a steady decline in suitability above this range. Both life stages reflected HSIs less than 0.5 above a discharge of 1.25 $\mathrm{m}^{3} / \mathrm{s}$. Curves developed from both data sets were similar, however the tail of each curve diverged at discharges greater than $1.25 \mathrm{~m}^{3} / \mathrm{s}$. This is a result of the wider range of discharges used to develop the curves for the large data set (ALL).

## Ice Scar Height

Suitability curves for ice scar height, a proxy variable for range of discharge or 'flashiness', demonstrated similar range of optima and trends for both fry and parr in the large data set. Optimum suitability was evident in the lower ranges, from 0 to 150 cm , with declining suitability above 150 cm . Fry HSIs demonstrated a steep decline over the higher values while parr response was more gradual, and relatively flat over the range from 200 to 350 cm . Curves developed from the EXP. RIV. data set were markedly different from those developed from the large data set and reflect the narrow range values of ice scar height in the data set and possibly the small number of samples used in defining the curves ( $\mathrm{n}=65$ ). For the EXP. RIV. data, optimum suitability for parr was in the range 100 to 150 cm and for fry in the range from 50 to 100 cm , with declining suitability on either end of these optima.

## Instream Cover

Fry suitability curves for instream cover (as a percent of the station), demonstrated optimum suitability from 0 to $10 \%$ with a gradual decline in HSIs above this range to reach 0.42 ( 0.38 for EXP. RIV.) at $100 \%$. The parr curves
demonstrated a wide range in optimum, from 30 to $80 \%$ ( 20 to $80 \%$ for EXP. RIV.), which is evidence of flat response of parr densities to instream cover. The parr curves demonstrated less than optimum suitability in the lower ranges and a sharp decline in suitability above $80 \%$ instream cover. Curves developed from both data sets for both life stages were similar.

## Overhanging Cover

Suitability curves for fry in relation to overhanging cover (as a percent of the station) demonstrated optima in the lower ranges ( 0 to $30 \%$ for ALL data, 0 to $10 \%$ for EXP. RIV. data), with steady declines in suitability above the optimum. The shape and apparent trends in the fry curves for both data sets were similar. The parr curve from the large data set demonstrated optimum values over the range from 0 to $60 \%$ overhanging cover (flat response), and a sharp decline at higher percentages. Conversely, the parr curve for the EXP. RIV. data was markedly different from the parr suitability for the large data set (ALL) and had an optimum range from 60 to $70 \%$ with sub-optimum HSIs in the lower range to reach a minimum of 0.46 from 0 to $25 \%$ overhanging cover.

## Substrate Rating

Fry suitability curves for substrate rating for coarseness (calculated from percentages of substrate types in each station) demonstrated optima in the range 100 to 125 which would indicate preference for pebble/cobble dominated stations. Suitability indices in the range lower than the optima (finer substrates) declined rapidly while in the higher range HSI declined more slowly with a sharp decline above 225 (boulder dominated sites). Fry curves from both sets of data were consistent. The parr suitability curve (for ALL data) demonstrated increasing preference with increasing coarseness to reach optimum at the highest range (> 225), which would indicate highest suitability for boulder dominated sites. The parr curve for the EXP. RIV. demonstrated a flat response with optima across the range in substrate rating from 50 to 175. The shape and trend in the parr suitability from this data set is a reflection of the narrower range in rating values and the absence of stations with substrate coarseness ratings at either end of the distribution (relative to the larger data set)(Tables 3 and 4).

## DISCUSSION

Habitat suitability curves developed in this paper from diverse rivers/habitats resulted in reasonable curves for width, depth, discharge, ice scour height and substrate rating. The wide range in optima for velocity and cover variables (instream, overhanging) suggest these curves will require refinement or conversely, that these variables are of lesser importance in determining habitat selection by juvenile salmon. Fry suitability curves for the two data sets were remarkably consistent suggesting a well defined set of habitat preferences for salmon fry. Parr curves between data sets were not nearly as consistent and this could suggest that parr are more plastic in their habitat requirements than are fry or possibly that other influences have obscured habitat preference (eg. competition, available habitat, etc.). It
might also indicate that selection of habitat changes as parr grow and that separate suitability curves may be required for different size/age ranges of parr.

Of the eight habitat attributes for which curves were developed the most consistent criteria were developed for stream width, mean depth, and discharge. These variables are related to stream order with width, depth, velocity and discharge tending to increase as stream order increases. Hydrological variables (eg. velocity or discharge) have been assumed to be good correlates or indicators of other habitat characteristics and have been related to juvenile salmon production (eg. Chadwick 1982, Frennette et al. 1984). Production (per unit of habitat) of salmon has been demonstrated to be higher in smaller, headwater streams (second and third order). Talbot (pers. comm.) found stable, high parr densities to be associated with the upper watershed in the Little Codroy River, Newfoundland, and concluded that primary habitats were those that were hydrologically stable located in the headwaters. Keenleyside (1962) has noted fry to be more abundant in upper reaches of rivers, and parr were most abundant in the same reaches as fry. Gibson et al. (1990) have found that, in the Experimental Rivers, the most productive juvenile salmon habitat is the smaller streams with stable discharge, and coarse substrate. Parr densities on these rivers were most influenced by substrate (positive correlation with coarseness) and stream width (negative correlation). The higher suitability of wider streams for parr (Figure 5) might be reflective of the greater ability of parr to migrate and find a wider diversity of habitat than fry. The discharge, width, depth and substrate coarseness suitability curves presented in this paper are consistent with these considerations.

DeGraff and Bain (1986) found considerable overlap in utilization curves and optimum velocity for fry and parr in 2 Newfoundland rivers. Morantz et al. (1987) also found substantial overlap in velocities selected by fry and parr with preferenda for fry from 5 to $10 \mathrm{~cm} / \mathrm{s}$, for small parr ( $<100 \mathrm{~mm}$ ) from 7 to $15 \mathrm{~cm} / \mathrm{s}$, and large parr ( $>100 \mathrm{~mm}$ ) from 10 to $20 \mathrm{~cm} / \mathrm{s}$. The relatively flat response of juvenile salmon parr to velocity (wide range of optima), as developed in this paper, is likely not indicative that this variable is unimportant in habitat selection; conversely there is a wealth of literature demonstrating parr and velocity interactions. DeGraff and Bain (1986) found fairly wide preferendum for velocity in their study and suggested that lack of competition from other species in Newfoundland rivers has permitted more extensive use of habitat of lesser suitability.

Microhabitat researchers have cautioned against comparing velocity measurements of a different type (i.e. mean station velocity, water column velocity, nose velocity) as the values are frequently not well correlated. Our results might indicate that the measure of velocity employed, mean station velocity, is a poor measure of available holding sites within a station with suitable/preferred nose velocities. Other habitat attributes (eg. cover, depth, etc.), in addition to the relevance of the type of measurement, may have obscured preference for a given velocity range. Further refinement and evaluation of the velocity curves presented in this paper would be required before considering application.

Depth suitability curves presented in this paper are supported by those reported in other microhabitat studies. Morantz et al. (1987) found parr to select deeper waters than fry (optima from 30 to 60 cm as opposed to 20 to 40 for fry) and depths selected by both life stages were rarely less than 15 cm and did not exceed 100 cm . The consideration that, generally, salmon occupy faster, deeper water over coarser substrates as they grow has been well established (as reviewed by Gibson et al. 1990) and substantiated in microhabitat studies (eg. Trail and Stanley 1984, deGraff and Bain 1986, Morantz et al. 1987). Morantz et al. (1987) concluded that juvenile salmon tolerate a wide range in depth and substrates and that water velocity, at the holding position (nose velocity), is the dominant factor influencing habitat selection. The high suitability associated with the greater depths for parr in this paper, as opposed to that reported in Morantz et al. (1987), may be related to the ability of juveniles in Newfoundland to use deeper, slower water and pool habitats in the absence of competition from other species, similar to that reported by deGraff and Bain (1986).

The suitability curves for fry and parr (ALL data) in relation to ice scar height, as a surrogate for range in discharge or 'flashiness', is consistent with the previous mentioned relationships between stream order and unit fish production. While small streams respond more quickly to hydrological events than mainstems, the effect of peak flows, and associated ice conditions, is more extreme in the larger river habitat. Rivers (sites) with a high range in discharge may be hydrologically unstable and may have low egg to fry survival due to movement of substrates and washing out of redds. These hydrologically unstable systems may not have well developed riparian vegetation and may also experience erosion of stream banks and siltation at certain periods of an annual cycle. The parr curve shows greater HSIs at the higher ice scar heights than did the fry curve, and this could be reflective of coarser substrates associated with 'flashy' sites.

Preference of juvenile salmon for substrate has been determined from an index that was developed weighted to increasing coarseness. While this index cannot be related directly to dominance any one substrate type, the substrate rating curves (Figure 6) suggest preference of fry for pebble/cobble dominated sites and parr for boulder dominated stations. Substrate particle size is related to water velocity as greater flows are capable of transporting larger materials. Talbot (pers. comm.), in evaluating parr densities and habitat attributes for the Little Codroy River in Newfoundland, found bottom type to most influence parr densities with highest numbers associated with rough cobbled bottoms with large boulders and coarse rock. Densities were also related to flow, width, distance from salt water and altitude but not to depth. Morantz et al. (1987) found salmon to occupy faster water over larger substrates as they grew. Gibson et al. (1990) has cautioned that apparent preference of parr for coarse substrates may be determined by other conditions (eg. water velocity, territoriality, turbulence, etc.). Fry substrate preference may also be related to the association of fry with spawning habitats (gravel/pebble dominated reaches) and the fact that fry migration from spawning reaches is predominantly downstream (Gibson et al. 1990).

The suitability curves for the two cover variables (instream and overhanging) provide results that are not as readily interpretable. For fry, highest
suitability was in association with low values for the cover variables with declining suitability as the percentages of these cover types increased (Figure 6). Gibson et al. (1990) determined salmon biomass on the Experimental Rivers to have a negative relationship with overhanging cover. Gibson and Keenleyside (1966) have reported that overhanging cover attracted parr in shallow water but not in deeper water. Increasing amounts of overhanging cover may result in lesser illumination (inducing low levels of primary production) and cooler temperatures providing habitat conditions that may be better suited to trout, or, alternatively may be associated with headwater streams that migrating salmon may have difficulty accessing. Trout in the Experimental Rivers were most abundant in small streams in association with pool habitat and overhanging cover (Gibson et al. 1989). The parr curve for instream cover is extremely flat and possibly is indicative that this variable has little influence, perhaps relative to other variables, in selection of habitat. The influence of other variables on selection of cover, and the possible role of brook trout in this selection, should be examined in evaluating and refining the cover suitability curves in this paper.

The distribution of the habitat variable used in suitability curve development can have a profound influence on the shape of the curve and the range of applicability of the curve (what values of the habitat attribute that the curve can be applied to). This is most apparent when comparing curves developed for ice scar height and substrate rating (parr) for the EXP. RIV. data and the large (ALL) data set. The relative distributions of these variables (Figure 4) have influenced the shape of the suitability functions. An important consideration when comparing the distribution of attributes between the 2 data sets is the small sample size, or absence of data, at the tails of the distributions, and the resulting effect on the curves. The small number of samples at the tails of the distributions can give a disproportionate weighting to the values of the indicator variable at these ranges. Developers of suitability relationships have to be particularly sensitive to these concerns and some subjective judgement must be applied when extreme values, and possible outliers, are encountered. Curves developed from a narrow range of data will be limited in applicability to a comparative range in habitat attributes.

Habitat suitability models have been criticised mostly as to how they are aggregated from individual suitability relationships into a one model yielding an overall suitability index. These criticisms are largely based on the absence of mathematical sophistication, through the inability to realistically consider compensating mechanisms between variables, limiting factors, and the effects of competition from other species. In some instances combining or aggregating SIs into an HSI model may not be required or even appropriate. The use of individual HSIs for key variables may be more suited to addressing a particular habitat management decision. Specialized models developed from data from a narrow range for the species (eg. one watershed) often do not have general applicability and consequently their expanded use, without modification or testing, is limited. Development of a single HSI model or a suite of suitability indices, for a species, to cover the entire geographical range, is of course unrealistic. Suitability criteria and models may need to be modified, altered, redefined, or in certain circumstances abandoned depending on each application. No single model/index will have universal
applicability. Accuracy of application of a particular model to a given geographical situation is often traded off against models with wider, more generalized, habitat suitability functions.

Ideally habitat suitability curves, to be useful in management of habitat, should have wide applicability. In this paper, curves have been developed from a diverse data set and suitability criteria could be considered to be representative of the range in habitat use across this data set. Suitability curves assembled from pooled data represent a range in habitat that juvenile salmon will use in relation to the variety of conditions to which they are exposed throughout their distribution. Conversly, these curves contain a host of assumptions and limitations, in addition to those developed for sitespecific microhabitat variables, that may constrain their potential application or increase the liabilities associated with their use.

In insular Newfoundland, due to the low numbers of freshwater fish species, juvenile salmon are able to utilize a wider range of habitats than they do in other parts of their distribution where habitat availability and use is influenced by interspecific competition. Consequently the curves developed in this paper would need to be very cautiously applied to other geographical regions where interspecific competition will have more influence on habitat selection by salmon. Caution should be expressed when applying these criteria to any management decision and any user should have an understanding of available habitat in the system they are evaluating. DeGraff and Bain (1986) have demonstrated that, in insular Newfoundland, salmon use atypical habitats and criteria developed from literature reference or from other regions would result in underestimation of available habitat. They suggest it may be necessary to develop separate suitability criteria for different types of habitats. Gibson et al. (1990) has suggested that, as estimates of carrying capacity (habitat suitability) and production cannot be made on every system, rivers need to be classified on the basis of climatic, geological, hydrological, chemical, and other factors and that methods of estimating carrying capacity and production be worked out for river classes. This same approach could be applied to suitability/preference relationships.

## Alternative Approaches to Developing Suitability Criteria/Curves

Habitat suitability indices, and criteria to represent these relationships, can be developed and expressed in a variety of ways. Suitability criteria were developed in this paper through frequency analysis using histograms to relate the indicator variable (fish abundance), expressed as an interval or ratio, to the selected habitat variables. Smoothing techniques were used to define a monotonic or unimodal suitability curve, where deemed appropriate. Suitability curves can be used to express both use of that habitat or, when adjusted for availability of habitat as in this paper, habitat preference.

There have been a variety of other approaches used in developing and displaying suitability relationships for a single habitat variable (Terrell 1985, Bovee and Zuboy 1988). In some instances a simple X, Y plot is used to display the relationship and a line is fitted to the data. Regression analysis has commonly been used to develop suitability criteria and, since most species/habitat response suggests a curve rather than a straight line,
polynomial regression, primarily cubic and quadratic functions, have been most frequently employed (eg. Orth and Maughan 1982). One advantage to this approach is that it employs widely used standardized techniques for which computer packages are available. Residual analysis will also provide statistical measure of goodness of fit which can permit comparison of equations.

Another approach that has been employed in habitat preference work, primarily in association with the Instream Flow Incremental Methodology, has been the use of nonparametric tolerance (not tolerance in the ecological sense) intervals (Bovee 1986). This method is used in describing frequency data and basically places an 'umbrella' suitability curve over the frequency distributions. In this approach a suitability index of 1.0 is assigned to the central $50 \%$ of the population, 0.5 to the central $75 \%, 0.2$ to $90 \%$ and 0.1 to 95\% (Sommerville 1958). This approach works well for unimodal distributions, however for monotonic curves, a suitability index of 1.0 would need to be assigned to the left (or right) most $50 \%$ of the distribution, in a one-tailed sense. One major disadvantage is that this approach will generate a curve for random data or a very flat species response.

The major assumption in developing suitability criteria for an individual habitat attribute is that of independence between the effects of variables. One of the principal difficulties in developing habitat suitability relationships, and the primary criticism of this approach, is the erroneous assumption of independence between the effects of various habitat variables on fish distribution. For Atlantic salmon, the interdependence between factors, some in the context of compensatory mechanisms and some in relation to limitations imposed by key variables, has been well established and habitat selection is considered to be in relation to a complex interaction of several variables (as reviewed in Gibson et al. 1990, deGraff and Bain 1986). Users of the HEP/HSI approach have tried to consider these interactions when aggregating HSIs for individual variables into habitat suitability index models, with varying degrees of acceptance. Others have pursued developing bivariate and multivariate suitability criteria (eg. Voos and Lifton 1988, Hansen 1988). These have included bivariate polynomial models and have been employed primarily in describing the interaction between depth and velocity. Lambert and Hansen (1989) have investigated the use of two-dimensional curve smoothing techniques as an alternative approach to display of bivariate suitability. They found the approach did not have any obvious advantages to the use of univariate curves, when the interaction of variables was considered in aggregation into a HSI model.

Another criteria (approach) commonly employed is based on professional judgement from appropriate 'experts' and life history information in the literature, and relies on little, if any, empirical data (Crance 1987). This approach, defined by the USFWS as Category 1 criteria, involves using professional judgement to define suitability curves from relevant literature. In some instances the Delphi technique has been employed to get an anonymous polling of expert opinions on habitat suitability through a well defined series of questions. The responses are analyzed and fed back to the experts who reanalyse the questions in light of the compendium of advice. This can be repeated until a consensus is reached. An alternative to this is for one
'expert' to develop and propose suitability criteria and have others critique the result. For some variables (eg. temperature influences) this may be the most effective way to define suitability criteria. Habitat suitability criteria for Atlantic salmon have previously been developed from literature reference, for use with the IFIM methodology (Bietz and Kiell 1982) and in development of a HSI model (Trail and Stanley 1984).

## Application of HSIs to Habitat Management

It is important in developing and using HSI models/indices to clearly state and clearly understand assumptions associated with the development and application. HSI models are most frequently used in habitat impact analyses/management decisions which frequently require short term decision response (weeks to months for EIA type evaluations to days to hours for referral type decisions) and the 'best decision' possible. Consequently, the use of any model/index cannot require extensive data collection because assessments often have time and monetary constraints. Suitability index/model developers must keep this consideration paramount to ensure potential applicability. Habitat suitability curves and/or models for freshwater species in Atlantic Canada should be pursued and developed with consideration that these approaches to habitat evaluation, if they become operational, will be applied by habitat managers (Fish Habitat Management Division, Area Habitat Coordinators, Fisheries officers) and not habitat scientists. In this situation ease of use, costs, time, reproducibility will become important considerations.

It is also important that indices be developed, and models built, with the potential application in mind. As HSI models are most frequently used to evaluate positive and negative impacts associated with some development, variables used in HSI models must be those that are potentially altered by some habitat perturbation, or are reflective of change in habitat quality. Consequently, while a variable such as gradient may be important and convenient in describing habitat suitability (Amiro 1984) for a given species, it is one that would rarely, if ever, be affected (perturbated) by development. These types of variables consequently have limited value to HSI models developed for use in habitat impact analyses and management. HSI models need to predict how changes in physical and chemical features of specific habitat will be changed by a development activity, and how this change will affect carrying capacity, and by inference, populations.

It could be argued that developing habitat suitability (preference) relationships from population level data for a stream reach or station is of little relevance because the averaging of information to describe a station will obscure the relationship between the microhabitat preferences and available habitat. It also follows that habitat suitability developed from individual data (eg. velocity, depth, substrate data for use in IFIM) will have limited value when applied to habitat assessments of stream reaches/stations. For example, velocity suitability using nose velocity data, while being a useful variable for the IFIM approach and in hydrological simulation models, will have very limited applicability to habitat evaluations of lesser sophistication. In a habitat management context, in most instances, potential perturbations will be evaluated in a general sense (eg. change in
cover, depth, discharge, etc.) and not in a microhabitat context, largely because of time and money constraints. These gross level predictions of change/impact may be better addressed from suitability indices developed from population and station level data. Further, habitat assessments from day-to-day operational level activities will permit only minimal data collection and input to decision making. On large projects with considerable potential for habitat destruction, greater resourcing and time allocated to assessments may permit a detailed evaluation to fine tune the potential habitat losses and gains. Clearly, there exists a need for both levels of habitat suitability criteria, those that can be applied at an operation level as well as detailed (possibly site specific) suitability curves, for intensive habitat assessments that require a greater degree of precision and confidence.

## Recommendations

Variables evaluated in this paper were those that have been collected to elucidate relationships between production of juvenile Atlantic salmon and habitat attributes, and were not collected with the intention of defining or developing habitat suitability curves. If suitability criteria are considered for application to habitat management, a more comprehensive set of habitat evaluation criteria would need to be developed. Suitability criteria developed in this paper will require further refinement, and testing over a wide range of habitat types. Additional parameters that could be beneficial to evaluating potential habitat perturbations would include some measure of the availability and contribution of pool habitat (either as \% pool type habitat in a station or through some pool rating approach) and \% fines and degree of imbeddedness of substrate (Terrell et al. 1982). Water chemistry and other environmental parameters, and there relationship to habitat selection, will need to be evaluated. Variables could include water temperature (including maxima/minima in a limiting factor sense, the range in temperature over the growing season in a growth/production sense), dissolved oxygen (at temperature maxima, possibly as a limiting factor), some measure of nutrient content (measure of nutrients, or some proxy variable that is more readily and frequently measured, eg. conductivity, hardness, total dissolved solids, etc.), and turbidity.

There is a need to consider the effects of interspecific competition, primarily from brook trout (Salvelinus fontinalis L.) in Newfoundland, on habitat selection and the shape and trend of suitability curves. Habitat selection by salmon and brook trout has been demonstrated to be influenced by the cohabitation of the two species (Gibson 1978, Gibson and Power 1975, Gibson et al. 1990). The influence that anthropogenic acidification has had on salmon rivers in Canada and Scandinavia would need to be considered in a suite of suitability criteria for application to acid sensitive, eastern Canadian rivers. The seasonal availability of key attributes in relation to key life processes may be as important, or more so, than a mean or average condition. The most influential of these tend to be related to water levels and flows, frequently during egg incubation and during summer low flows in relation to rearing of fry, and minima/maxima of key water quality variables. Standing waters contribute significantly to salmon parr rearing and potential production in insular Newfoundland (and possibly elsewhere) (Pepper et al. 1985) and this fact would need to be recognized in the development of habitat
suitability indices/models to be applied in this geographical area. Separate HSIs would need to be developed for different habitats (i.e. for lacustrine and riverine habitats) or conversely a model would need to consider habitat requirements/use of the various life stages in both lentic and lotic habitats.

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Table 1. Rivers (name and Waldron's river code) and dates sampled for fish abundance and habitat attribute data used for development of habitat suitability relationships for Atlantic salmon. Highlighted rivers are those in the Experimental Rivers (EXP. RIV.) subset.

| Name | Waldron's Code | N | Sampling Period |
| :--- | ---: | ---: | :---: |
| Gander R, | 040861 | 2 | $06-1980$ |
| Anchor Bk. | 090894 | 3 | 07 to 08-1983 |
| Deadmans Bk. | 090895 | 3 | 07 to 08-1983 |
| Northwest Bk. (BB) | 111013 | 6 | $08-1982$ |
| Southwest Bk. | 111027 | 10 | $08-1981$ to 08-1982 |
| Wings Bk. | 111028 | 24 | $08-1981$ to 08-1983 |
| Salmon Bk. | 121089 | 1 | $08-1982$ |
| Southwest R. | 121093 | 1 | $09-1982$ |
| North Arm R. | 221622 | 7 | $06-1980$ to 07-1983 |
| Freshwater R. | 270002 | 71 | $05-1984$ to 08-1988 |
| Long Beach R. | 270006 | 5 | $08-1985$ to 08-1988 |
| Drook Bk. | 270015 | 24 | $05-1984$ to 08-1988 |
| N. E. Trepassy R. | 270036 | 57 | $05-1984$ to 08-1988 |
| St. Shotts R. | 280065 | 1 | $08-1985$ |
| Tides Bk. | 320641 | 5 | 07 to 08-1983 |
| Big Salmonier Bk. | 320695 | 1 | $08-1983$ |
| Highlands River | 400083 | 16 | $05-1980$ to 07-1981 |
| Western Arm Bk. | 490519 | 6 | $07-1980$ |

Table 2. A summary of sample size and range in habitat variables used in determination of the appropriate bin size (interval) for presentation of frequency distributions of the habitat variable for development of habitat suitability index (HSI) curves. Bin size was estimated by use of the Sturge (1926) equation. In several instances, a practical interval was preferable to that calculated by the Sturge equation (eg. data collected as percentages).

| Variable | N | Range | Bin(Sturge) | Bin(Practical) |
| :--- | :---: | :---: | :---: | :---: |
| Width | 242 | 23.4 |  |  |
| Mean Depth | 239 | 70.0 | 7.62 | 2.5 |
| Mean Velocity | 200 | 0.89 | 0.10 | 5.0 |
| Discharge | 199 | 9.05 | 1.04 | 0.1 |
| Instream Cover | 181 | 99.0 | 11.6 | 10.0 |
| Overhang. Cover | 196 | 75.0 | 8.7 | 10.0 |
| Ice Scar Height | 65 | 335.0 | 47.7 | 50.0 |
| Substrate Rating | 233 | .0 | 26.3 | 25.0 |

Table 3. Summary of habitat attributes and fish abundance data used in developing suitability curves from the large data set (ALL).

| Variable | N | Min. | Max. | Mean | S.D. |
| :---: | :---: | :---: | :---: | :---: | :---: |
| Discharge ( $\mathrm{m}^{3} / \mathrm{s}$ ) | 224 | 0.00 | 2.47 | 0.52 | 0.41 |
| Area ( $\mathrm{m}^{2}$ ) | 230 | 38.08 | 657.90 | 219.17 | 134.22 |
| Width (m) | 230 | 1.70 | 21.30 | 7.70 | 3.41 |
| Depth (cm) | 227 | 8.00 | 65.00 | 22.54 | 10.17 |
| Max. Depth (cm) | 212 | 20.00 | 194.00 | 52.43 | 25.71 |
| Mean Velocity (m/s) | 226 | 0.00 | 0.87 | 0.31 | 0.17 |
| Sur. Velocity (m/s) | 45 | 0.00 | 0.76 | 0.25 | 0.19 |
| Pool Length (m) | 124 | 0.00 | 43.10 | 7.23 | 12.75 |
| No. of Pools | 133 | 0.00 | 3.00 | 0.39 | 0.64 |
| Ice Scar Hgt. (cm) | 66 | 0.00 | 335.00 | 83.08 | 67.87 |
| Instream Cover (\%) | 177 | 0.00 | 99.00 | 18.89 | 21.91 |
| Overhang. Cover (\%) | 192 | 0.00 | 75.00 | 7.84 | 13.09 |
| Canopy Cover (\%) | 156 | 0.00 | 50.00 | 3.21 | 9.01 |
| Fines (\%) | 222 | 0.00 | 100.00 | 4.53 | 13.30 |
| Gravel (\%) | 222 | 0.00 | 90.00 | 6.30 | 12.97 |
| Pebble (\%) | 222 | 0.00 | 90.00 | 12.00 | 14.33 |
| Cobble (\%) | 222 | 0.00 | 90.00 | 32.26 | 20.11 |
| Rubble (\%) | 222 | 0.00 | 90.00 | 27.62 | 17.70 |
| Boulders (\%) | 222 | 0.00 | 100.00 | 17.18 | . 54 |
| Bedrock (\%) | 222 | 0.00 | 25.00 | 0.11 | 1.68 |
| Substrate Rating | 233 | 0.00 | 233.00 | 121.61 | 34.91 |
| Water Temp. ( ${ }^{\circ} \mathrm{C}$ ) | 210 | 1.70 | 27.00 | 17.29 | 3.98 |
| Conductivity (uS/cm) | 109 | 18.00 | 248.00 | 72.25 | 62.09 |
| NYOY | 181 | 0.18 | 153.69 | 21.92 | 31.98 |
| NPARR1 | 210 | 0.19 | 90.94 | 21.44 | 19.04 |
| NPARR2 | 205 | 0.36 | 64.56 | 9.88 | 10.95 |
| NPARR3 | 139 | 0.15 | 26.25 | 3.30 | 3.70 |
| NPARR4 | 31 | 0.19 | 4.42 | 1.25 | 1.10 |
| NPARRT | 230 | 0.00 | 122.35 | 30.55 | 24.85 |
| NTOTAL | 230 | 0.58 | 202.59 | 47.80 | 41.17 |

Note: NYOY=number of young of the year, NPARR1=number of $1+$ parr, NPARR2=number of $2+$ parr, NPARR3=number of $3+$ parr, NPARR4=number of $4+$ parr, NPARRT=total number of parr.

Table 4. Summary of habitat attributes and fish abundance data used in developing suitability curves from the Experimental Rivers data set (EXP. RIV.).

| Variable | N | Min. | Max. | Mean | S.D. |
| :---: | :---: | :---: | :---: | :---: | :---: |
| Discharge ( $\mathrm{m}^{3} / \mathrm{s}$ ) | 154 | 0.00 | 1.51 | 0.49 | 0.33 |
| Area ( $\mathrm{m}^{2}$ ) | 156 | 38.08 | 657.90 | 204.13 | 131.23 |
| Width (m) | 156 | 1.70 | 15.40 | 7.92 | 3.30 |
| Depth (cm) | 154 | 8.00 | 59.00 | 21.84 | 9.80 |
| Max. Depth (cm) | 150 | 20.00 | 194.00 | 51.93 | 26.94 |
| Mean Velocity (m/s) | 156 | 0.00 | 0.87 | 0.31 | 0.18 |
| Sur. Velocity (m/s) | 28 | 0.00 | 0.76 | 0.20 | 0.18 |
| Pool Length (m) | 106 | 0.00 | 43.10 | 5.93 | 12.84 |
| No. of Pools | 115 | 0.00 | 1.00 | 0.22 | 0.41 |
| Ice Scar Hgt. (cm) | 43 | 0.00 | 162.00 | 61.63 | 48.18 |
| Instream Cover (\%) | 141 | 0.00 | 99.00 | 20.62 | 23.34 |
| Overhang. Cover (\%) | 137 | 0.00 | 60.00 | 6.01 | 10.01 |
| Canopy Cover (\%) | 131 | 0.00 | 50.00 | 2.63 | 8.30 |
| Fines (\%) | 152 | 0.00 | 50.00 | 3.28 | 8.13 |
| Gravel (\%) | 152 | 0.00 | 20.00 | 4.09 | 4.71 |
| Pebble (\%) | 152 | 0.00 | 60.00 | 12.73 | 11.46 |
| Cobble (\%) | 152 | 0.00 | 75.00 | 32.55 | 16.78 |
| Rubble (\%) | 152 | 0.00 | 75.00 | 29.07 | 15.56 |
| Boulder (\%) | 152 | 0.00 | 75.00 | 18.28 | 16.95 |
| Bedrock (\%) | 152 | 0.00 | 0.00 | 0.00 | 0.00 |
| Substrate Rating | 152 | 59.45 | 202.67 | 125.92 | 28.45 |
| Water Temp. ( ${ }^{\circ} \mathrm{C}$ ) | 145 | 1.70 | 23.50 | 17.17 | 3.70 |
| Conductivity ( $u \mathrm{~S} / \mathrm{cm}$ ) | 66 | 18.00 | 232.00 | 54.24 | 49.78 |
| NYOY | 121 | 0.20 | 153.69 | 28.48 | 36.14 |
| NPARR1 | 136 | 0.19 | 90.94 | 23.59 | 20.60 |
| NPARR2 | 132 | 0.38 | 64.56 | 11.41 | 12.71 |
| NPARR3 | 89 | 0.15 | 26.25 | 4.06 | 4.32 |
| NPARR4 | 18 | 0.38 | 4.42 | 1.44 | 1.22 |
| NPARRT | 156 | 0.00 | 122.35 | 32.70 | 26.96 |
| NTOTAL | 156 | 0.58 | 202.59 | 54.79 | 44.82 |

Note: NYOY=number of young of the year, NPARR1=number of $1+$ parr, NPARR2=number of $2+$ parr, NPARR3=number of $3+$ parr, NPARR4=number of $4+$ parr, NPARRT=total number of parr.


Figure 1. Frequency histograms and suitability curves for Atlantic salmon fry and parr in relation to stream width for the large data set (ALL).


Figure 2. Frequency histograms and suitability curves for Atlantic salmon fry and parr in relation to stream discharge for the large data set (ALL). Curve smoothing, using 2 passes of a 3-point running mean filter, was applied.


Figure 3. Frequency distributions of habitat availability for the variables width, mean depth, maximum depth, velocity, and discharge for the 2 data sets.


Figure 4. Frequency distributions of habitat availability for the variables instream cover, overhanging cover, canopy cover, ice scar height and substrate rating for the 2 data sets.


Figure 5. Habitat suitability curves for Atlantic salmon fry and parr in relation to stream width, mean depth, mean velocity, and discharge. (Solid line is the curve for ALL data, dashed line is the curve for the EXP. RIV. data).


Figure 6. Habitat suitability curves for Atlantic salmon fry and parr in relation to ice scar height, instream cover, overhanging cover, and substrate rating. (Solid line is the curve for ALL data, dashed line is the curve for the EXP. RIV. data).

# Instream Flow Incremental Methodology: an Efficient Tool for the Application of the <br> "no net loss" Principle to Salmon Habitat 

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#### Abstract

The Instream Flow Incremental Methodology is presented as a promising and reliable approach to determine the optimum flow for the conservation of fish species in river systems located downstream from hydroelectric works. The impact study of the diversion into the Sainte-Marguerite River of a Moisie River upstream tributary on Atlantic Salmon, is based on a physical and hydrodynamic simulation using the classical approach of bidimensional numerical modeling by the finite element approach. The model involved outputs velocity and depth and is of a drying-wetting type which permits the determination of the flow boundary dynamically. Simulations covered a wide range of discharges taking into account the critical periods for the growth of young salmons. The recurrence analysis of flow events was done using standard hydrology methods.

Special attention was payed to simultaneous biological and physical surveys and to the establishment of reliable habitat preference curves (HSI: Habitat Suitability Indexes) for salmon fry and parr specifically which are some of the strategic life stages that can be considered by the methodology. The velocity, the riverbed substrate and the depth were taken into account. The proposed methodology takes care of obtaining fish specific preferences instead of river specific preferences that are commonly evidenced by classic methods based on simple histograms.

A bidimensional spatial analysis of the usable areas for salmon enabled us to relate numerically the Weighted Usable Area (WUA) and the discharge for six sample sites. As the discharge variable is a time series, one can also build some WUA time series that may be used in the definition of guaranteed optimum or minimum flow regimes for salmon.

This procedure based on the "no net loss" principle was applied to the most sensitive and rich sample site of the study and gave results that can be considered not only as impact mitigation measures for the project but also as new opportunities to improve some of the worst natural flow events in the river. A complementary approach for the spawning period was also developed to determine winter flow regimes.

KEY-WORDS: Instream Flow Incremental Methodology (IFIM), simulation, Atlantic Salmon, Moisie River, flow allocation, guaranteed minimum flow, fish conservation, hydrodynamic numerical modeling, Habitat Suitability Index (HSI)


## RESUME

La modélisation des microhabitats (Instream Flow Incremental Methodology: IFIM) est présentée comme une approche prometteuse et crédible pour déterminer les régimes d'écoulement appropriés pour la conservation de certaines espèces piscicoles de rivière en aval d'ouvrages hydroélectriques.

L'étude d'impact de la dérivation d'un tributaire d'amont de la rivière Moisie (rivière Aux Pékans) sur la population de saumon atlantique de ce cours d'eau a utilisé avantageusement une démarche classique de modélisation numérique bidimensionnelle des courants. Un modèle de type couvrant-découvrant a été utilisé et une gamme biologiquement pertinente de débits a fait l'objet des simulations. L'analyse des débits a été réalisée avec les méthodes classiques de l'hydrologie statistique.

Un attention particulière a été accordée à la détermination de relations numériques rigoureuses entre les caractéristiques physiques (abiotiques) du milieu et les préférences d'habitat du saumon à différents stades de son cycle de vie en rivière. Il en est résulté des courbes de préférence d'habitat (Habitat Suitability Index: HSI) qui ont servi à cartographier l'habitat potentiellement disponible sur six sites-échantillons ayant servi à l'étude.

L'analyse spatiale des aires utilisables par les saumons juvéniles a conduit à l'élaboration de relations entre les Aires Pondérées Utiles (APU ou WUA: Weighted Usable Areas) et la gamme des débits simulés aux six sites. La mise en relation de ces courbes avec les séries chronologiques de débits naturels ou modifiés (résultant de l'aménagement Aux Pékans) a permis de définir un concept de débit minimum réservé permettant d'harmoniser, voire mettre à profit le régime d'écoulement futur de la Moisie pour les exigences de conservation du saumon. Ayant nécessité l'élaboration de séries temporelles de APU, cet exercice a été mené sur les sites les plus riches et sensibles du cours d'eau pour l'élevage des juvéniles. Cette procédure a été érigée en principe de la méthode.

Mots-clés: modélisation des microhabitats, débit minimum réservé, simulation, saumon atlantique, rivière Moisie, allocation du débit, conservation du poisson, modélisation hydrodynamique, préférences d'habitat.

## CONTEXT

Hydro-Quebec has undertaken studies of the energy potential of the drainage basin of the Sainte-Marguerite River, neighbor to the Moisie River (Fig. 1 ). The preliminary planning scheme includes the catchment and the diversion into the Sainte-Marguerite River of a significant part of the discharge of the Aux Pekans, an upstream tributary of the best Atlantic Salmon river in Canada, the Moisie River in the Québec North Shore region of the St.-Lawrence River.

The drainage basin of the Moisie River extends 330 km northward on the Canadian shield and occupies a surface of $19000 \mathrm{~km}^{2}$. The average annual runoff ( $\mathrm{N}=33$ years) is about $425 \mathrm{~m}^{3} / \mathrm{s}$ at the mouth. The Atlantic salmon is the principal fish species encountered in the riverine system. However, a greater biological variety characterizes the estuarine lower part of the river.

The Aux Pekans River diversion ( $74 \%$ of the discharge) represents a $42 \%$ reduction of the total Moisie River discharge at the confluence of the two rivers. However, at the mouth of the Moisie, the reduction would be $13,4 \%$ of the total discharge as a result of the additional flow contributed from other downstream tributaries.

This flow reduction raised many questions related to the conservation of the salmon and necessitated an impact assessment study. Numerous methods are available to enable the relationship between the river discharge and the amount of habitat usable by the biological species to be quantified. The expression used for this kind of method is "Instream Flow Incremental Methodology" or IFIM. For specific examples of these methods, consult Morhardt (1986). The IFIM is a promising and reliable approach to determine the optimum flow for the conservation of fish species in river systems located downstream from hydroelectric works.

The present study (Boudreault et al, 1988; 1989) was undertaken with an improved approach of IFIM. In order to maximize the precision of the analysis, the physical and hydrodynamic data were obtained using the classical approach of bidimensional numerical modeling by the finite element approach. The determination of the habitat suitability indexes that enables the relationship between the flow characteristics and the fish preferences were also improved as we will see further. The general method and the main results of the study will be presented in this paper.

## METHOD

## Introduction

The general method used (displayed in detail in Leclerc et al, 1990a) is diagrammed in Fig. 2 . Multidisciplinary by nature, it involves the following themes: the selection of sample sites and the surveys, the hydrological analysis and hydrodynamic modeling, the characterization of the salmon habitat preferences (HSI) and the sensitivity analysis of the habitat (WUA) to the discharge. Finally, the approach is completed by determining the optimum flow regime that can be adopted to maintain or improve the habitat availability on the most sensitive sample site(s) of the river.

The present method is based on a physical and hydrodynamic simulation using the classical approach of bidimensional numerical modeling by the finite element approach. This particular aspect of the method is the main factor that distinguishes our approach from the classic ones. Moreover, the model involved is of a drying-wetting type which permits to determine the flow boundary dynamically. Simulations covered a wide range of discharges taking into account the critical periods for the growth of young salmons.


Figure 1. Drainage basin of the Moisie River and location of the sample sites (Taoti site is numbered 1)


Figure 2. Methodology of the study
Six representative sample sites on the Moisie River (Fig. 1) were selected for the simulation purposes. Three main factors played a role in the selection of the sites: the salmon habitat quality in the Moisie River, the relative importance of the projected reduction in the discharge and the type of morphological profile of each potential reach.

Among the six sample sites, one (hereafter called Taoti) is likely to be the most affected by the Aux Pekans river diversion as it is located further upstream than the others and consequently, closer to the planned diversion. Furthermore, its bed profile is of a braided reach type which is likely to be the most favorable configuration for a salmon habitat. As a matter of

Among the six sample sites, one (hereafter called Taoti) is likely to be the most affected by the Aux Pekans River diversion as it is located further upstream than the others and consequently, closer to the planned diversion. Furthermore, its bed profile is of a braided reach type which is likely to be the most favorable configuration for a salmon habitat. As a matter of fact, a decrease of the discharge in this type of reach is normally accompanied by a more rapid loss of habitat due to the reduction of the wetted perimeter. As this site was retained as a reference site for the guaranteed minimum flow determination, we chose to illustrate our approach with the results obtained on it.

## Surveys

Physical data. For the purpose of hydrodynamic modeling, the physical data measured at various sampling points are a vertical averaged velocity, the bottom and the water surface levels and the corresponding riverbed substrate. Our first aim is to draw up a bathymetric and a dominant riverbed material map of each considered sample section of the river. We also wish to obtain a good quantitative description of the hydrodynamic behavior of the river sections on which subsequent calibration and validation of the hydrodynamic model is based. For this purpose, we also need reliable stage-discharge relationships at the upstream and downstream boundaries of each site under study.

Biological data. The main objective of the biological field survey is to determine the Atlantic salmon habitat preferences that will further be translated in suitability curves (habitat suitability indexes: HSI). In classic approaches, these curves are often established in a way that reflects more the frequency of fish observed in a given reach of river (histograms) than the intrinsic habitat preferences of the fish species. As it takes into account the local availability of physical factors in the river system, the method used reveals the relative preferences of habitat of the species in a given river.

As for the measured data, the presence or absence of individuals juveniles of the species (smaller: fry and larger: parr) in the immediate neighborhood of sampling points is visually observed during dives. Simultaneously, at each sampling point, the main physical characteristics mentioned earlier is measured in addition to some behavioral variables or observations like the "fish's snout velocity", territoriality (home rock) and feeding of each individual observed. See Tab. 1 for the entire content of the database and Fig. 3 for an example of the distribution of the sampling points on our example sample site (Taoti).

## Hydrodynamic modeling

The model. The two-dimensional numerical modeling of free surface flows by the shallow water equations has become a scientifically recognized tool in engineering studies. Canals, rivers and estuaries are usually the favorite cases for the application of this type of model. Probably due to inter-disciplinary barriers and/or gaps, there has not yet been much use of these numerical models for the description of biological habitats. However, this approach is more and more used to analyze possible environmental impacts from engineering projects. The actual model was presented in detail in Leclerc et al. (1990b,c). It is based on the classic mass and momentum conservation equations for two-dimensional depth averaged model of shallow waters.

The present model is of a drying-wetting type, that is to say that the location of the lateral flow boundary depends on the discharge. As this moving limit is an "a priori" unknown, it is therefore determined during resolution. This non-linear capability is very precious in discharge (or tide) sensitivity studies since the selected algorithm (Leclerc et al, 1990c) avoids adaptative meshing in the region touching the lateral closed boundaries for every state simulated and it furnishes reliable hydrodynamic data in this zone as well.

Table 1. Content of the database ( $\mathrm{N}=467$ ) on the Atlantic salmon juveniles of the Moisie River measured during dives.

| DATA \# | DESCRIPTION |
| :---: | :---: |
|  | Identification parameters |
| 1 | Sample site \# |
| 2 | Longitudinal section \# |
| 3 | Sampling point \# |
|  | Biological data |
| 4 | Presence of fry (0+): 1 or more |
| 5 | Presence of parr (1+): 1 or more |
| 6 | Presence of parr (2+): 1 or more |
| 7 | Absence of juvenile: 0 |
|  | Physical data |
| 8 | Local depth (H) |
| 9 | Velocity at 0,2H |
| 10 | Velocity at 0,8H |
| 11 | Velocity at 2 cm above the substrate |
| 12 | Fish's snout velocity |
| 13 | The three main constitutive elements of the substrate in the vicinity of the point |
|  | Juveniles behavior (descriptive data) |
| 14 | Position of the juvenile with respect to |
| 15 | the riverbed level <br> Orientation of the juvenile with respect to the main |
| 15 | direction of the flow current |
| 16 | Presence or absence of territorial behavior |
| 17 | Presence or absence of feeding behavior |
| 18 | Mobility |



Figure 3. Distribution of the biological sampling points on sample site Taoti.

Discretization. The discretization (partitionning, meshing) of the flow domain and the nodal interpolation of the variables within each geometrical element of the model is performed with the finite element method using a six nodes triangular element. The main advantage of this discretization method is that it provides an excellent representation of the geometry of the physical domain in terms of morphology, substrate variability and the desired density of information. See Fig. 4 c for an example of a discretized sample site (Taoti site).

Calibration and validation. As mentioned previously, the application of the model involves first an adjustment of the flow resistance parameters (calibration), then a validation using a different set of verification data. Let us mention that the size of the dominant riverbed material is used to frame the value of the Manning's roughness coefficient, the main parameter of the hydrodynamic model, as well as to ultimately enable the habitat to be classified, as we shall see. This delicate procedure also involves a good knowledge of the upstream and downstream stage-discharge relationships. One must pay a special attention to the representation of the lateral boundary location. However, this particular aspect depends more on the reliability of the bathymetric data than on the accuracy of the hydrodynamic model itself.

We consider that the implementation of the hydrodynamic model at the six chosen sites gave very satisfactory results on an appropriate scale for the needs of the study. A precision of the order of $10 \%$ for the velocities can be considered as an excellent result for this kind of model. Some results on Taoti site are presented on Fig. 5. Are represented on this Fig. the flow regime through current vectors for discharges of 188,135 and $72 \mathrm{~m}^{3} / \mathrm{s}$ and the stage-discharge relationships of the site. The displayed results correspond to quite a pertinent statistical flow range considering the purpose of the study.

Hydrological analysis. The hydrodynamic analysis of the samples sites for the purpose of habitat evaluation is based on one hand, on a reliable hydrodynamic model, and on the other hand, on a "species specific" hydrological analysis. The purpose of the hydrological analysis is therefore to estimate the low range discharges at the sample sites for a given duration and recurrence period and this, for strategic periods in the life cycle of the species (Fig. 6). The present hydrological regime is considered, as well as the effect of the projected diversion of $74 \%$ of the Aux Pekans River.

The method used (Bobée and Robitaille, 1977) has become classic in statistical hydrology. It is based on the use of the log-Pearson type III law adjusted to the observed (or modified) time series. The discharge values corresponding to the sample sites are interpolated proportionately to the area of their drainage basin.

The statistical analysis of the discharge as modified by the diversion of a tributary (the time series of the main river from which is subtracted the time series of the tributary to be diverted) is performed the same way as that used for the primitive series.

## Salmon habitat modeling

Habitat suitability indexes $(H S I)$. The habitat suitability indexes are the result of the development of preference criteria for various physical factors for certain biological functions of the species studied. The procedure used is illustrated in Fig. 7.


Figure 4. Bathymetry (a), riverbed substrate (b) and finite element discretization of Taoti site (c)




Figure 5. Simulated flow regime for discharges of 188,135 and $72 \mathrm{~m}^{3} / \mathrm{s}$ on Taoti site and corresponding stage-dicharge relationships.


Figure 6. Characteristic chronology of the biological functions of the Atlantic salmon - vs - hydrological regime at the mouth of the Moisie River


Figure 7. Standard algorithm for the representation
of the fish habitat preferences with HSI

Let's consider the principles of this approach. Following the identification of sensitive vital functions, for example, the growth of the juveniles, abiotic (usually physical) variables that are determinant in the constitution of a preferable habitat are selected. So far, the considered factors are more often classified independently of each other according to their own degree of preferability for the specific function considered. The result is curves representing basic indexes varying between 0 and 1 , depending on the degree of adequacy of the value of the variables with respect to the biological functions.

The basic indexes are then combined algebraically by a geometric mean:

$$
H S I \equiv I_{G}=I_{1}^{a_{1}} \times I_{2}^{a_{2}} \times \ldots \times I_{j}^{a_{j}} ; \quad j=1, N
$$

with

$$
\sum_{1}^{N} a_{j}=1
$$

where
$I_{G} \quad$ a global index for a given species and a particular function;
$I_{j} \quad$ a basic index (variable j specific) for the function;
$a_{j}$ geometric weights;
$N$ the number of abiotic variables j considered.
The weighting factors may be chosen equal, but in order to translate the relative importance of the various factors in the species behavior, it is suitable to use variable weights. These may be determined by a procedure taking into account a multivariate statistical analysis of the results of the biological surveys and/or the advice of a panel of species experts.

Atlantic salmon's HSI. Fig. $8 \mathrm{a}, \mathrm{b}, \mathrm{c}$ and $9 \mathrm{a}, \mathrm{b}, \mathrm{c}$ display the basic HSI's for the growth of salmon parr and fry for the velocity, the substrate and the depth respectively. Some alterations were carried out in the low suitability range of the curves following species expert's opinion.

Obtained by satistical means and experts panel, the relations used to represent the global parr's and fry's habitat suitability indexes are the following:

$$
\begin{aligned}
H S I(\text { parr }) & \equiv I_{P G}=I_{D}^{0,40} \times I_{V}^{0,39} \times I_{S}^{0,21} \\
H S I(\text { fry }) & \equiv I_{F G}=I_{D}^{0,26} \times I_{V}^{0,44} \times I_{S}^{0,3}
\end{aligned}
$$

A four levels scale of interpretation for the global habitat suitability may be defined as follows:

| HSI VALUE | INTERPRETATION |
| :---: | :--- |
| $0,0-0,1$ | dry or unacceptable to mediocre |
| $0,1-0,4$ | mediocre to medium |
| $0,4-0,7$ | medium to very good |
| $0,7-1,0$ | very good to ideal. |




Figure 8. Basic habitat suitability indexes for Atlantic Salmon parr (N sampling points = 467); a: velocity; b: substrate; c: depth


Figure 9. Basic habitat suitability indexes for Atlantic Salmon fry ( $\mathrm{N}=467$ ) a: velocity; b: substrate; c: depth

Habitat sensitivity to flow discharge. As the abiotic (physical) variables are themselves known as a function of the space coordinates ( $\mathrm{x}, \mathrm{y}$ ), the value of the resulting indexes $I_{j}$ and $I_{G}$ can be mapped for an entire sample site. Furthermore, this can be done for every simulated discharge. For this purpose, several GIS (geographical information system) already exist (like SPANS from Tydac Technologies Inc.) that can be used to display the results in significant colors and perform complete spatial analysis.

It is then relatively easy to calculate the areas of a river reach corresponding to different classes of biological suitability and therefore, quantify the total habitat availability by a procedure of weighted sums.

$$
W U A=\sum_{i=1}^{m} H S I_{i} x A_{i}
$$

where

| WUA | the weighted usable area; |
| :--- | :--- |
| HSI $_{i}$ | the mean suitability value of the class $i ;$ |
| $\mathrm{A}_{\mathrm{i}}$ | the area of the class i ; |
| m | the number of classes defined |

As the areas are calculated for a certain range of flow discharge, the resulting WUA is defined as a function of the discharge as well. The result of this procedure is curves (for parr and fry) representing the habitat availability with respect to the flow discharge and the life stage of the species. From a careful analysis of these curves, we can determine some single value of a guaranteed minimum flow that may be introduced as a constitutive part of the hydrological management of the hydropower works.

Habitat Time series. A time dependent expression of the WUA which could be called DWUA (Dynamic Weighted Usable Areas) is essential to determine an adequate flow regime for biological purpose. Let's define this concept. As, on one hand, the discharge is by itself, a dynamical process (often documented with long and reliable time series, sometime, 30 years of duration or more) and, on the other hand, there can exist a valuable numerical relationship between the flow, the related instream variables and the habitat availability (WUA), we can easily build some time series of the latest variable.

This fairly new type of time series may represent either the current natural "habitat regime" (reference state) or the eventual modified ones as several water management scenarios are usually considered along with the planning process.

## RESULTS

## Mapping of the parr habitat

In order to illustrate the location and mobility of usable areas with respect to flow and to demonstrate the global relative effect of flow modification proposed on the Moisie River, we chose to display on Fig. $10 \mathrm{a}, \mathrm{b}$ the value of the global parr habitat suitability index (HSI) of Taoti site for discharges of 135 and $91 \mathrm{~m}^{3} / \mathrm{s}$. These usual values correspond to the actual and projected average summer low flow discharge respectively. Mention that the latter scenario assume no water release from Aux Pekans River which is presumably the worst case possible.


Figure 10. Mapping of the global HSI value for parr on Taoti site Flow discharges of 135 (a) and 91 (b) $\mathrm{m}^{3} / \mathrm{s}$

## Habitat - vs - discharge curves

Fig. 11 shows percent usable areas classified according to their suitability level with respect to the discharge for the growth of parr on Taoti site. Note that this figure incorporates the evolution of dry areas with respect to the flow value.

Fig. 12 demonstrates the percent usable area (PUA\%) for fry and parr of this site as a result of a weighted summation of the previous percent curves with respect to their own suitability level. When multiplied by the total absolute area, this first result becomes the classic Weighted Usable Area in $\mathrm{m}^{2}$. The PUA\% type of result was preferred to the latter in order the keep a relative point of view on the habitat sensitivity on flow discharge.

Note that the usable areas (UA) tend obviously to disappear for a zero discharge. Since the simulations could not be carried past trough a certain reliable minimum extrapolated from the stage-discharge relationship, the UA curves were extended arbitrarily by a dashed line connected to the zero origin.

These figures also indicate the typical actual discharges pertinent to the growth of juveniles. The summer average flow and 15 consecutive days minimum flows with two and ten years recurrence periods are represented. The first event represents the most common hydrological situation encountered while the latter indicate respectively a high and a low recurrent flow discharge. An hypothetical future situation not taking into account any water release from the Aux Pekans River is also included on these curves. This particular assumption constitutes only one possibility among others but it corresponds in fact to the worst case possible.

We see at first glance that the total area of usable habitat is not very sensitive to even relatively large variations of the discharge in the usual observed range. This corresponds to a fairly optimum situation. However, a noticeable decrease in usable areas appears at a threshold flow of $135 \mathrm{~m}^{3} / \mathrm{s}$ and a dramatic drop comes out at a flow value of $50 \mathrm{~m}^{3} / \mathrm{s}$.

Fig. 13 presents the natural WUA time series for the period 1957-1989. As the mean duration of the rearing period of the juvenile salmon in the river is three years, we built a times series with a mobile average corresponding to this time increment. As there exists an hydrological module for the flow regime in rivers, there also may exist a "biological module" that reveals the average condition for the rearing salmon. On Taoti site, this biological module corresponds to a flow discharge of $137 \mathrm{~m}^{3} / \mathrm{s}$.

## Guaranteed minimum flow

The determination of the "guaranteed minimum flow" for biological purposes is based on the following principles:

- "no net loss" of habitat;
- take the opportunity of the water storage potential of the upstream dams to improve some severe natural conditions through appropriate water releases;
- consider the most sensitive site as reference site for the determination of the reserved flow values.
Following these principles, Taoti site was chosen to evaluate the minimum flow and a flow discharge ranging from 135 to $140 \mathrm{~m}^{3} / \mathrm{s}$ was adopted for the summer period (rearing phase).


Figure 11. Discharge sensitivity of percent usable areas (PUA\%) with respect to the suitability level for parr on Taoti site


Figure 12. Discharge sensitivity of percent usable areas (PUA) for fry and parr on Taoti site


Figure 13. Natural time series of WUA (1957-1989)
Simulation were made with a minimum flow sustaining to a $135 \mathrm{~m}^{3} / \mathrm{s}$ level applied to the entire series of historical hydrological data ( $\mathrm{N}=33$ years). We produced pre- and post-diversion summer hydrographs. As an example (Fig. 14 ), in 1989 which was a year of relatively low flow, the proposed methodology would have lead to water releases from the Aux Pekans installations that would have permitted not only to mitigate the impact of the diversion but to improve the flow regime for biological purpose. In 1984 (Fig. 15 ), the water release for salmon rearing would have been lower than in 1989 in summer but the severe conditions in fall would have necessitated additional flow contribution from Aux Pekans installations for maintaining spawning conditions. Note that the guaranteed minimum flow for spawning was established distinctively but in a similar way as for the juvenile habitat.

## CONCLUSION

Classic statistical and complementary studies can be undertaken on and with this type of data with either cognitive, planning and/or management objectives. For example, very useful information can be extracted on the best and the worst possible biological years from a species point of view. The dynamic aspect of these data can also contribute to the definition of the duration and the temporal variability of some characteristic habitat availability levels.

Moreover, this type of analysis, which is very close to the hydrological approach used to evaluate the energetic potential of a river as well as to design the works, is well understood by the specialists (usually, engineers) involved in the planning and the management of physical hydro-powerinstallations. This approach, as it facilitates the multidisciplinary communication between different specialists, can serve as a very powerful tool to assess different scenarios during the technical negotiations about the minimum flow to maintain in the river network.


Figure 14. Flow regimes of Taoti site in 1989 - Natural (a), modified without water release from Aux pekans River (b), and modified with proposed release (c)


Figure 15. Flow regimes of Taoti site in 1984 - Natural (a), modified without water release from Aux pekans River (b), and modified with proposed release (c)

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# Limitations of Habitat Evaluation Procedure (H.E.P.) Models for Fish Habitat Management 

## by

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#### Abstract

Habitat Evaluation Procedures (HEP) are methods which have been developed in the United States to integrate the different environmental factors which affect fish habitat into models. The outputs of these models must be dimensionless Habitat Suitability Indices (HSI's) which express the relative quality of habitat. HSI's range from 0 (unsuitable) to 1 (optimum). They are multiplied by the surface area of available habitat to estimate the amount of available habitat in "Habitat Units". HEP models consider only physical and chemical factors. Interactions with other species, including man are ignored so that the procedures can not estimate actual production or harvest of a species. They purport to estimate 'carrying capacity' of habitat. As carrying capacity is not a measurable feature of habitat, the validity of these estimates can not be tested. Models can be tailored to fit existing data but HSI's are almost never correlated with population density of fish or, presumably, with quality of their habitat when applied to new data. Habitat suitability models take several years to develop and models developed for one area usually cannot be transferred to another. This will limit the usefulness of models to environmental assesments of large projects with long lead times.


## RÉSUMÉ

Les procédures d'évaluation de l'habitat (HEP), méthodes élaborées par des chercheurs américains, intègrent en modeles les divers facteurs environnementaux qui influent sur l'habitat. Les sorties de ces modèles doivent être des indices de la salubrité de l'habitat (HSI), sans dimension, qui expriment la qualité relative de l'habitat. Ces indices peuvent varier de 0 (non adéquat) à 1 (optimum). Ils sont multipliés par la superficie de l'habitat disponible pour obtenir le nombre d'habitats disponibles en "unités d'habitat". Les modeles HEP ne tiennent compte que des facteurs physiques et chimiques. Étant donné que les interactions avec d'autres espèces, y compris l'homme, sont ignorées, les procédures ne peuvent pas servir à déterminer la production actuelle ou le niveau d'exploitation dont peut faire l'objet une espèce particulière. Elles prétendent plutôt servir à déterminer la capacité de support de l'habitat. Etant donné que la capacité de support n'est pas une caractéristique mesurable de l'habitat, la validité de ces estimations ne peut être vérifée. Des modèles peuvent être ajustés en fonction des données disponibles, mais les HSI ne sont presque jamais en corrélation avec la densité des populations de poisson ou, probablement, avec la qualité de leur habitat quand ils sont appliqués à de nouvelles donnees. L'élaboration de modèles de la salubrité de l'habitat prend plusieurs années; ceux qui s'appliquent à une région ne sont généralement pas applicables à une autre région. Cette particuliarité limite l'utilité des modèles aux évaluations environnementales effectuées dans le cadre d'importants projets à long échéancier.

## INTRODUCTION

In 1986, the Department of Fisheries and Oceans adopted the National Habitat Policy whose objective is to increase the economic and social benefits of Canadian fisheries by conserving, restoring and developing fish habitat. The objective of this policy is "the achievement of an overall net gain in the productive capacity of fish habitats".

In future, no net loss of fish habitat will be acceptable. The objective of minimization of losses of habitat will be replaced by one of complete compensation for losses. If fish habitat is destroyed, an equivalent gain must be produced elsewhere. This will be done project by project. Where feasible, existing habitat will be improved and new habitat created. The performance of the Department in accomplishing the goals of the new policy must be monitored regularly.

The existence of a habitat policy implies that the amount and quality of fish habitat is a limiting factor which determines the production of fisheries. Although it may be hard to demonstrate a particular relationship between habitat and fisheries productivity most biologists feel intuitively that such a relationship exists. Other factors cause fluctuations which may obscure the effects of certain kinds of destruction of fish habitat. Nevertheless, it is believed that such destruction reduces productivity.

The achievement of the policy's goals entails the ability to identify and measure the productive capacity of fish habitat. To be useful in habitat management the ideal method of estimating fish habitat should allow the results of different decisions to be predicted and compared. It should be quick and should not require extensive data sets to provide output. Where lost habitat must be replaced, it should provide a way to determine equivalent value. The estimation of equivalent habitat is complicated by the fact that methods of estimating production from natural habitat may not apply to artificially made habitat. It is likely impossible to devise a model which meets all administrative and scientific requirements. That is to say cheap, fast and accurate over the geographical range of the species.

Habitat Evaluation Procedures (HEP) are already widely used in the Unites States to evaluate the impact of industrial and government projects on fish and wildife habitats and are being considered for use in the Canadian fish habitat management program. These procedures were meant to be used to evaluate particulur projects in environmental impact assessments but not to determine the success or failure of government policy. Even if these procedures are not adopted by the Department of Fisheries and Oceans it is possible that they will be used occasionally by consulants.

## DESCRIPTION OF HABITAT EVALUATION PROCEDURES

Habitat assessments in the HEP process are based on habitat units. These are computed by the equation:

$$
\text { Habitat Units }=(H S I) \times \text { (Area of available habitat) }
$$

Habitat units are expressed in terms of area, not biomass or productivity. The HSI or habitat suitability index is an expression of the quality of the available habitat. As its name implies, it must be in index form. An index is the ratio of some value to a standard of comparison. The standard of comparison in HSI models represents the maximum potential productivity of optimum habitat. The HSI is dimensionless and can range between 0 and 1.

The relationship between HSI and carrying capacity must be positive and the HSI of a habitat should be directly proportional to its productivity or standing crop. That is to say, a change in HSI from 0.2 to 0.3 indicates an increase in productivity equal to that which accompanies a change in HSI from 0.8 to 0.9. This requirement can be satisfied by mathematically transforming any known relationship between habitat and model output to a linear one.

Detailed instructions for the preparation of HSI models have been issued by the United States Fish and Wildlife Service (1981). HEP models include physical and chemical components of habitat but biological interactions between species are not considered. Consequently, some kinds of projects can not be assessed by this process. The HEP process is only applied to species of direct value to humans. For example invertebrates and forage species are generally ignored. A separate model is needed for each species which is considered.

It is not necessary to evaluate the effects of projects on every species which may be present. If many species are present they are divided into 'guilds' of species with similar environmental requirements and which presumably would react similarly to a given environmental change. A single species is then chosen from each guild and only that species need be evaluated. In Nova Scotia the number of freshwater species of fish is low and so it might be possible to evaluate all of the important ones. There are more freshwater species in other provinces and so it might be necessary to select certain species to act as proxies for their guilds.

Statistical models. Statistical methods can be used to determine which habitat variables have predictive value to estimate the performance of a species. Models in which multiple regression equations are used to estimate the population size, productivity or other value belong to this class. Other techniques can be used. For example, discriminant analysis models can be used to classify study areas into different categories with differing HSI's.

Aggregated Suitability Index Models. In this type of model habitat is divided into components and a separate Suitability Index (SI) assigned to each component. These Suitability indices are then combined together into an overall HSI. Each SI expresses the quality of the habitat with respect to the set of variables from which it was derived. As with the overall HSI the separate SI's associated with different components of habitat range from 0 to 1.

It is common to organize the components and environmental variables of an HSI model into an hierarchical arrangement as shown in Figure 1. The lowest tier in the hierarchy consists of measurable environmental variables. Intermediate tiers include components such as feeding habitat, spawning habitat and cover and the highest tier is the averall HSI. A Suitability Index is assigned to each variable in the lowest tier and these SI's are combined by functions to give combined SI's for higher tiers in the hierarchy.

The choice of functions used to join SI's into index values for the next higher level should be determined by the way in which the habitat variables are thought to control the species. For example, if the variable with the lowest suitability is a limiting factor, then the HSI could be equal to the lowest of its component's SI's. Under other circumstances, possible functions are arithmetic or geometric means, weighted means and others.

There is no one way in which the variables must be organised in HSI models. In Figure 1, the habitat is divided into main components according to the requirements of different life stages and each of these components further subdivided into cover, temperature or other environmental factors. Such a division might be convenient where the species' habitat changes greatly over the life of its members. Other arangements are possible. For example the habitat might be divided first into cover, feeding habitat and breeding habitat and then subdivided further.

Word Rankings. This is the simplest type of model. Descriptive or word models assign a suitability index to the habitat depending on whether given environmental variables or combinations of variables meet certain criteria. Word models are not in mathematical form. Habitat may be rated into categories and each category assigned a numerical rank. The best habitat should be given the rank with the highest number. The HSI value of a given habitat would then equal its rank in the rating scheme divided by the highest possible rank.

Other types of models are possible but seem to be rarely used. Existing models relating habitat variables to population characteristics may be suitable for application within HEP if their output can be converted to a 0 to 1 index form.

Development of Suitability Index Curves SI's are derived from habitat variables through suitability index curves which show the
relationship which is thought to exist between the variable and the suitability of the habitat. Figure 2 shows several kinds of suitability index curves. All of these curves were taken from a HSI model for chinook salmon (Raleigh et al. 1986). The values of the environmental variables used in the model are plotted on the horizontal axis and the appropriate Suitability Index can be read from the vertical axis. SI's may be based on physiological or population responses and many HSI models contain SI curves of both types. Terrell and Nickum (1984) warn against models of this mixed type.

The two methods of developing SI curves are the maximum performance and linear. The maximum performance method defines the SI for a given value of an environmental variable as the quotient of the maximum value of the response variable observed in conjunction with the selected value of an environmental variable divided by the over-all maximum value of the response variable.

$$
\text { SI }=R(i) / R m \text {-----------------------------(1) }
$$

where: SI $=$ Suitability Index.
$R(i)=$ maximum response observed when habitat variable $=i$.
$\mathrm{Rm}=$ maximum response observed for any value of variable.
An SI curve derived by the maximum performance method is shown in Figure 3a. This method is used because the population density, productivity or other response variable of a species may be well below the true carrying capacity of its habitat. It is assumed that the carrying capacity of the habitat is seldom exceeded.

The linear method of developing SI's is based on the average rather than the maximum level of the response variable. This method defines the SI as the quotient of the average value of the response variable observed for a given value of the environmental variable divided by the over-all maximum value of the response variable.

$$
\begin{equation*}
S I=\bar{R}(i) / R m \tag{2}
\end{equation*}
$$

where: $\overline{\mathrm{R}}(\mathrm{i})$ = average response observed when habitat variable $=\mathrm{i}$

Figure 3b shows an SI curve derived by the linear method.
Computer software has been developed which facilitates construction of HSI models.

FIELD TESTS OF HSI MODELS
Since populations are often below the carrying capacity of their habitats, they are not expected to be proportional to HSI's. However, on average, the more suitable habitats should support denser populations. Most field tests of HSI models actually are
attempts to demonstrate correlations between HSI's and estimates of standing crop.

Lister (1988) evaluated juvenile habitat portions of coho (Oncorhynchus kisutch) and chinook (ㅇ. tshawytscha) models (McMahon, 1983, Raleigh et al. 1986). He modified the models slightly to accommodate the existing data which were available to him. Lister showed that the existing models developed in the U.S. could not be transferred to British Columbia. The suitability curves were not accurate and caused the quality of some habitats to be under-estimated. Population densities of both species were correlated with distance upstream regardless of HSI scores. In order to evaluate the performance of the model, Lister transformed his final HSI scores because much of the variation in population densities was owing to this relationship which was not considered in the models. Differences in population densities between years were found although no corresponding changes in habitat were seen.

Densities of chinook were correlated with transformed but not raw HSI's by all aggregation methods. The population densities of coho were not highly correlated with HSI's whether transformed or not. Inclusion of most variables in the models did not appreciably improve correlations between HSI's and population densities.

McMahon (1987) used 15 years of data from Carnation Creek, B.C. to test the performance of the HSI models for coho (McMahon, 1983) and chum (ㅇ. keta) (Hale et al. 1985) salmon. HSI's for chum salmon reflected declines in the quality of streambed gravel and were positively correlated with adult recruitment. The HSI for coho salmon was not positively correlated with fall population numbers, fall biomass, fall density, smolt numbers or adult returns and was significantly negatively correlated with numbers of coho smolt which were counted in the following spring. Furthermore, model component SI's were not positively correlated with coho population abundance parameters.

The Carnation Creek watershed was heavily logged during most of the period when these data were collected. If data from the years of logging were omitted, positive correlations between HSI and population responses were found but only in the least impacted sections of the stream. The HSI declined least in areas which were most severely impacted by logging and was not correlated with population responses in sections of Carnation Creek which were clear-cut right to the banks. HSI was not linearly correlated with carrying capacity and thus an important assumption of HSI models was not satisfied in this case.

McMahon suggested that the coho model which was developed for use in Washington and Oregon placed too much weight on summer conditions and that a model emphasizing winter habitat would be more valid in British Columbia.

Trial, et al. (1984) tested HSI models for brook trout (Salvelinus fontinalis) (Raleigh, 1982), Atlantic salmon (Salmo salar) and three species of cyprinidae in Maine. Their habitat data were collected on only one site visit. The models for brook trout and common shiners accurately predicted the presence or absence of these species. Models for blacknose dace and fallfish did not predict whether those species would be present.

Layher and Maughan (1984) devised and tested HSI models for eight warmwater species of fish in Kansas. Although these authors found that the individual suitability index curves which had been developed elsewhere seemed to be valid for Kansas fish, none of the aggregated HSI models yielded indices which were correlated with standing crops of fish. Different habitat variables seemed most important with different sampling techniques. However, many methods of estimating standing crops had been used. The authors suggested that sampling technique may have been related to habitat type and that this may have biased their results. The authors designed new models based on discriminant analysis. These new models, not surprisingly, predicted the presence or absence of species when tested on the data from which they were derived but not when tested on new data.

Persons and Bulkley (1984) evaluated a draft model for cutthroat trout (ㅇ. clarki) in rivers and found that biomass was negatively correlated vith many of the SI's, significantly so in some cases. As expected, the HSI calculated from these SI's was not correlated with biomass. The model was also tested for rainbow trout ( $\underline{0}$. mykiss) with similar results. This failure of the cutthroat model to predict rainbow trout biomass is not surprising but similar tests of valid models on species other than those for which they were developed might be a way of testing the validity of combining species into 'guilds' for HEP evaluations.

Nelson and Miller (1984) compared standing crops of 3 species of centrarchidae in borrow pits in 5 southern states with standing crops predicted by HSI models. Correlation coefficients were less than 0.1. The models were modified by removing variables which seemed irrelevant at those sites and tested again. The correlation improved but these revised models were tested against the data which had been used to derive them.

Gilbert (1984) tested HSI models for 8 species of fish in both reservoirs and rivers. None of his calculated HSI's was significantly correlated with standing crop.

Shirvell (1989) reviewed the performance of six fish habitat models of various kinds. Although only one of these was an HSI model, his conclusions apply to models of that type. He found that no model performed well when tested on new data from habitats other than those from which they were derived and concluded that models should be recalibrated for application to new geographical regions.

## DISCUSSION

The environmental effects of habitat alterations can be predicted either by experts who are knowledgeable in the relevant fields or by habitat models. Models are written opinions of experts on the relative importance of different habitat variables in determining the success of species. HEP systematizes what biologists already do but specifies standard ways of collecting data and expressing results. Neither expert opinion nor habitat models can be considered reliable unless subjected to validation and testing.

The use of standardized models has some administrative advantages over other ways of assessing environmental impacts. The procedures used to collect data and to draw conclusions from them are documented and can be reviewed critically. Furthermore, it is possible to collect data for a HSI model in a consistent way and obtain consistent habitat assessments. However, although consistency in methodology is desirable in the administration of habitat policy, it does not necessarily imply that habitat assessments are accurate.

Over-reliance on habitat models can lead to the conclusion that fish habitat is being protected when it is not. If HSI models are used to evaluate fish habitat, then the policies followed will be calculated to maximize the Habitat Suitability Index which may or may not be related to real habitat quality. If used to evaluate the success of the habitat program, the same model might yield a high HSI, indicating that the program was successful whether or not fish habitat actually was protected. HSI models can not be applied without validation. The ultimate standard against which the fish habitat protection program is measured should be maintenance of fish populations or supply of fish to recreational and commercial fisheries. It will not be sufficient to demonstrate that the amount of habitat estimated by some model has not diminished.

The testing of habitat models can be focused either on comparison of the model's output with the some appropriate population response of a species or on testing of the many assumptions embodied in the model. The first approach may indicate how well a model performs but it will not show why it performs at that level and so may do little to advance knowledge of the species. The second approach may throw some light on the species' biology.

The validation of HSI models presents many problems. HSI's purport to measure the 'carrying capacities' of habitats. They do not estimate productivity, standing crop or other measurable characteristic of a population and so testing of models has usually consisted of measurements of correlation between the models, outputs and population densities. Correlation does not indicate that populations responses can be estimated from HSI's but only that the likely direction of response can be predicted. The
necessary assumption that HSI and habitat be linearly related has usually not been tested and probably is untestable. Presumably there are upper limits to the possible standing crop and productivity in any area but factors other than the quality of available habitat may reduce the actual crop and productivity below the limits imposed by carrying capacity.

A factor, not generally considered in the development and testing of HSI models is that populations may exceed the carrying capacity of their habitats. Fish habitat is seldom stable long enough for fish populations to adjust to current conditions and populations do not respond instantaneously to changes in environmental conditions. Most tests of HSI models assume that the biomass present in a given area actually was produced there and that the population is there because the habitat is suitable. The population density seen at a given site reflects the conditions which have existed up to that time and may often exceed the current carrying capacity of the site. In these overpopulated habitats, loss of biomass through deaths will exceed gains through growth and reproduction but the populations may remain high owing to immigration. Measurements of population densities or standing crops cannot detect these conditions and such habitats may be overrated.

Testing of HEP models is a lengthy process taking several years. It is unlikely that any short-term habitat study will sample the full range of variation of all of the important environmental variables. Habitat models based on data sets collected over short periods or small areas will be valid, if at all only for the limited range of conditions which occurred during the study. Development and validation should be carried out over several years so as to test the models under the full range of environmental conditions which is likely to occur. Therefore, unless there is a lead time of several years, each impact assessment would be performed with models which have never been tested for the relevant sites. This problem, combined with the difficulty of devising models of wide applicability confines their use to large environmental impact studies with long lead times.

The experience to date does not indicate that simple models suitable for routine project evaluations will give valid or reliable estimates of environmental effects. The majority of tested fish habitat models did not predict the level of any indicator of habitat quality.

General habitat models which are valid for wide geographical areas do not exist for any species of fish. HSI models which seem to give reliable indications of habitat quality in one habitat may not do so in others. The requirements of a species usually do not vary greatly over their geographical ranges but the relative importance of different variables in determining population processes evidently differs from place to place. Consequently, existing
models will have to be validated and tested or new models developed for each new application.

All new or modified models should always be developed with one set of environmental data but tested with totally different data collected at another time or place. Shirvell (1989) showed that models which appear to describe the data from which they were derived may not describe new data. Unless models give repeatible results they must be considered invalid. The widespread observation that models usually are invalid when transferred between different areas may also apply to models transferred between years within the same area. Such models would be invalid even in the areas to which they supposedly apply. Few tests of models have been performed on new data from the same area where the models were first developed.

Usually environmental impacts on a species are estimated by substituting future values of environmental variables for existing values in a habitat model. The model itself is not changed. Unfortunately, modification of a habitat may change the relative importance of different variables to a species. It is unlikely that the same model would be valid both before and after substantial environmental modifications have been made. This will be especially true of models which use surrogate measurements. The relationships between the surrogates and the environmental variables which actually control a species may be changed substantially by the project which is being evaluated.

The exclusion from some models of variables which do not account for variations in species' performances reduces the number of environmental measurements which must be made but also reduces the range of habitats to which the model can apply. The failure of an environmental factor to contribute to variation of standing crops or other characteristics of populations may not necessarily indicate that the factor is unimportant but only that it is relatively constant. On the other hand, if environmental factors are included which may be important but do not account for much variation the model may be applicable to many habitats but will entail more costly and perhaps unnecessary environmental measurements.

It is unlikely that HSI models can be used to assess the numerous small projects which are administered by habitat managers. At present, the Habitat Management Branch in the Scotia-Fundy Region processes roughly 1000 proposals for stream alterations annually. Large numbers of similar proposals are dealt with in other regions. These proposals may be received at any time of year and the Branch tries to deal with each proposal within one month of receiving it. Habitat models would need to use only those data which could be collected on a few site surveys carried out at any time of the year to satisfy these administrative restraints. Documented and verified models would be needed in advance to respond to each
application within the time allowed. Owing to the limited areas to which HSI models apply, many different models would be needed and the appropriate model would have to be determined for each project being assessed.

The assumptions that species can be grouped into 'guilds' of species which can be represented adequately by a single member of the group and that the effects of a project on all members of a guild will be similar is questionable. Correct division of species into guilds will be difficult.

HSI models are limited to the assessment of physical and chemical changes in habitat. The HSI process does not evaluate biological factors and therefore cannot be used to assess the effects of projects which might change the incidence of diseases, parasites or competitors. Likewise, introductions of exotic species and some fish enhancement projects cannot be evaluated by HSI models.

The method of preparing suitability curves is only approximate. Detailed studies of the response of a species to a particular variable are often lacking and in their absence it is necessary to interpolate between a few known points or even extrapolate beyond them. The very approximate nature of the known relationships can be seen by examining suitability curves published in HSI models. Many suitability curves are based on physiological rather than ecological studies and the physiological optimum of a species may be somewhat different from its ecological optimum. Where suitability curves are only approximate, the suitability indices derived from them will also be inaccurate. When the HSI is obtained by combining several SI's derived from suitability curves, the accumulated error obtained by performing arithmetic operations on many slightly inaccurate numbers may be very great.

When developing Suitability Index Curves, optimum conditions are assumed when population density is at the maximum. Therefore, the highest observed population density is taken as a standard against which other observations are measured. That is tantamount to assigning a Suitability Index of 1 to the habitat which seems to contain the largest population. Suitability indices are calculated by dividing estimates of environmental variables by estimates of the same variables under optimum conditions so that all SI's are based on the most extreme outliers in the population data.

HSI models may appear to provide assessments which are free of the subjective judgements and biases of individual experts but this appearance is an illusion. The following criteria for acceptance of habitat models are taken from Standards for the Development of Habitat Suitability Index Models (U.S. Fish and Wildlife Service, 1981) -

- Model outputs based on sample data appear reasonable to the evaluation team.
- Model outputs based on sample data appear reasonable to a species authority.
- The model outputs rank study sites in a manner similar to a species authority's rankings.
- The output of the model is correlated with carrying capacity as measured by populaton estimates.
- The model outputs predict carrying capacity as measured by populations, within $10 \%$ with a confidence level of $90 \%$.

It is obvious that models validated against the first three of these criteria would be no more than the subjective judgements of experts presented in an apparently objective format.

It is possible to manipulate the conclusions of HSI models by selecting the variables which they use and by appropriate selection of the procedures for deriving Habitat Suitability Indices. Proponents would cast their projects in the best possible light and their opponents will do the opposite. Therefore, the validity of each step in a habitat model would have to be examined before the model could be accepted for use in fish habitat protection. Alternatively, the 'No Net Loss' policy might be best served by using procedures which result in the most pessimistic predictions.

It should be accepted that there is at present no way to calculate the precise effects of complex habitat changes and that it is impossible to determine what mitigation measures would compensate for proposed habitat changes. Currently devised models are more expensive and no more reliable than expert opinions. On the other hand the habitat requirements of many species for particular environmental factors are or can be found and these can be protected or improved. Their interactions are mostly unknown and very complex. Habitat protection can only be ensured at present by defending habitat one factor at a time and not by trying to compensate for habitat damage by improving other habitat.

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Habitat variables
Minimum D.O. (V1)
Maximum Temperature (V2)
Percent pools (V3) $\qquad$

Minimum D.O. (V4)
Average velocity (V6)
$\qquad$

Juvenile $\square$ HS I

Average gravel size (V7)
Percent fines (V8)
Mean pH (V9) $\qquad$ Embryo $\qquad$
Mean temperature (V10)
Average peak flow (V11) $\qquad$

Figure 1. Diagram showing the relationship among environmental variables, components and Habitat Suitability Index (HSI) for hypothetical species

Average percentage of fines in spawning gravel in major spawning areas. Measure within 30 days after spawning is over and at the same sites as $V_{B}$.
A. Fines $\leq 0.8 \mathrm{~mm}$ in size (silt).
B. Fines $>0.8$ to 30 mm in size (sand).




Figure 3a. Suitability Index by maximum response method


Figure 3b. Suitability Index by linear method


# Contribution to the Problem of Quantifying Productive Capacity of Fish Habitat 

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## Abstract

This paper reviews the important properties of quantitative tools useful in developing advice on habitat-stock relationships. The properties include applicability with many different habitat attributes robustness to underlying distributions of variables, and ability to produce quantitative predictions. Kernel estimators are a group of nonparametric probability density estimation methods shown to have those properties.

The paper illustrates uses of kernel estimators with three different data sets of fish population and habitat attributes; trout and salmon biomasses vs. stream features, cod trap catches vs. depth of isotherms, and capelin abundance vs. bottom temperatures. In each example the kernel estimators were readily applied and gave quantitatively rigorous and biologically reasonable results.

The paper also discusses the types of management uses that can be made with the results of these analyses. Because the methods produce full probability density functions (pdf) rather than point estimates, much more information is provided to managers. Providing the full pdf of possible outcomes may require managers to be much more explicit in the types of objectives they achieve or avoid. Providing the full pdf's also gives managers a much more complete picture of the range of possible consequences of a management option and more realistic idea of the likely results of field tests of management programs. The results suggest kernel estimators may be a very useful tool in the kit of habitat scientists.

## Résumé

On passe en revue les importantes propriétés des outils quantitatifs utiles pour l'élaboration de conseils sur les relations habitat-stock. Les propriétés comprennent les possibilités d'application à diverses caractéristiques de 1'habitat, la tolérance des diverses distributions fondamentales des variables et la capacité de production de prévisions quantitatives. Les estimateurs Kernel sont un groupe de fonctions de distribution de probabilité non paramétriques qui possèdent ces propriétés.

On illustre l'utilisation des estimateurs Kernel à l'aide de trois séries de données sur des populations de poisson et leur habitat: les biomasses de truite et de saumon $v$. les caractéristiques du cours d'eau, les prises de morue à la trappe $v$. la profondeur des isothermes et l'abondance du capelan $v$. la température des eaux du fond. Dans chaque exemple, les estimateurs Kernel ont été facilement appliqués et ont donné des résultats précis au niveau quantitatif et raisonnables au niveau biologique.

On examine aussi l'application des résultats de ces analyses à la gestion. Étant donné que ces méthodes produisent des fonctions de distribution de la probabilité au lieu d'estimations ponctuelles, les gestionnaires disposent de plus amples informations. La prestation de ces pdf des résultats possibles peut signifier que les gestionnaires devront être beaucoup plus explicites en ce qui concerne les objectifs atteints ou évités. Ceux-ci disposeront aussi d'une meilleure vue d'ensemble des diverses conséquences possibles d'une option de gestion et d'une meilleure appréciation des résultats possibles des essais à grande échelle des programmes de gestion. Les résultats portent à croire que les estimateurs Kernel pourraient étre un outil utile dans la trousse des scientifiques qui étudient l'habitat.

## Introduction

Fisheries scientists face a new responsibility. In the near future it can be expected that the Department's Habitat Managers will, to a significant degree, depend on the provision of sound and timely scientific advice on the appropriate means to measure changes in the attributes of fish habitats, to evaluate the consequences of these changes which are often caused by anthropogenic impacts, and to translate these changes into losses and gains of productive capacity of fish habitats. Scientists must provide advice on what specific changes in habitat mean for productive capacity. Multi-year site-specific studies are simply impossible at the very large number of sites where habitat changes will occur.

Scientists also must be aware that although it may be possible to base our advice on a few selected variables, it will not be possible to constrain industry and society at large to have their activities impact always and only on those selected variables. Advice is required on the meaning of changes in diverse types of habitat attributes. For some attributes, fisheries scientists may have a wealth of background data, and may understand well the mechanisms linking the habitat attribute to fish production. For many, scientists will not be in such a comfortable position, yet advice will still be needed. Even without a knowledge of the underlying processes, scientists should be unwilling to advise that just because habitat attribute "X" is not one of the first 2 or 3 variables to enter a multiple regression model, there will be no impact on fish populations if an activity were to alter that characteristic of the environment substantially. Scientists need flexible and widely applicable quantitative tools which can use the best data available (even if the amount is scanty) to estimate impacts of the vast range of human activities which potentially are going to impact on fish habitat in the coming years.

To further complicate matters, scientists need to portray the uncertainty in advice realistically when the advice is provided. There are at least two reasons for this. First, just because it will not be possible to study every special case in depth, scientists will be requested to advise instances when their understanding is incomplete, but the proposed undertaking cannot be ignored. Moreover, even if science had the luxury of studying each site in detail, natural fisheries ecosystems are intrinsically variable. Scientific credibility is compromised if only point estimates are provided for situations which will vary from year to year, season to season, and even week to week.

In short, scientists need to develop quantitative tools which can give some estimate (with associated uncertainty) of the productive capacity of sites before and after some activity under consideration. The habitat measures used often will have to be ones readily quantified in at most a few hours by field staff with modest levels of training in habitat research. The scales of inquiry will vary between 10 's of meters to 10 's or even hundreds of kilometers.

## QUANTITATIVE METHODS

## A. METHODS PREVIOUSLY USED

Quantifying consequences of habitat change is not a new problem. Legislation in some other jurisdictions has required comparable advice for well over a decade. Two classes of approaches have been tried: habitat evaluation procedures (HEP) and related methods, and multiple regression and its relatives. Both have serious flaws.

HEP has the advantages of being flexible, and not demanding of data. Users need to make no distributional assumptions about their data, nor propose any specific equation for the functional relationship between measure of habitat and measure of fish productive capacity. It is seriously compromised, however, by the subjectivity inherent in drawing the initial suitability curves. Attempts to make these suitability indices more empirical use the convex hull approach; the smallest convex curve which includes all (or most) the observations. This throws away all information on variance, other than the extreme points. The second great flaw is that users get a separate suitability index for each habitat attribute. These indices are not interval measures on a common scale, despite falling on a continuum usually scaled arbitrarily from, zero to one. Therefore, they cannot be combined into an overall estimate of habitat quality (or productive capacity) for either the before or the after condition, unless arbitrary scaling assumptions are made.

Linear and polynomial regression models of habitat-stock relationships (and discriminant function models, which are just regression models with categorical dependent variables) are empirically determined rather than subjective. Also, predicted values are interval measures, so there are circumstances in which they can be combined. However, these models assume there are some simple and consistent functional relationships between stock and habitat over the entire range of the habitat variable. Moreover, not only must such relationships exist, but they must be reflected accurately by the models. There is almost no theoretical framework to guide in the development of parameterized models of habitat-stock relationships, so by default linear or simple polynomial regression models are used.

Despite decades of research, fisheries scientists are unable to agree on the shape of the functional relationship linking stock to recruitment, nor are they able to estimate parameters of candidate functions with reasonable confidence (Walters and Ludwig 1981). Scientists generally know even less about habitat-stock relationships than they do about stock-recruit relationships. However, the habitat-stock relationships are likely to be even less smooth (there will be thresholds, asymmetric curvilinearities, areas of no association at all) and more variable. In most cases scientists will have insufficient data to either identify the true relationship between habitat attribute and stock, or to estimate the parameters of the function if they do have ways to identify it.

## B. PROBABILITY DENSITY FUNCTIONS IN HABITAT RESEARCH

Recently, Evans and Rice have explored applications of nonparametric probability density estimation methods to stock-recruit (Evans and Rice 1988; Rice and Evans 1988) and abundance-habitat relationships. These methods make
no assumptions about the overall shape of the association between independent and dependent variable, nor about the shape of the underlying distribution of either variable. Rather, they assume that for a given value of any independent variable, there is a probability density function (pdf) for the dependent variable. This pdf could be flat (all values of the dependent variable are equally likely), sharply peaked (for any position on the independent variable, only a narrow range of values of the dependent variable are likely to be observed), or something in between. Consider what this approach says about data bases, as well as about any predictions to be made. The approach assumes observations are random samples from these pdfs; not a set of individually best point estimates. It is easier to picture the cumulative pdfs, or ogives. If the ogives have steep slopes, only a narrow range of abundances are likely, given a position on the habitat gradient. Each observation taken is likely to be close to the "best" value (the median of the pdf, which equals the mean and mode if the data fit regression assumptions). If the ogives have more gradual slopes, however, individual samples are likely to have a great deal of scatter. They will be individually not very reliable indicators of what to expect from a series of samples at similar sites. If one has replicate measures at a site, the replicates are used to estimate the full pdf, rather than just its first moment.

Applying the pdf approach to the task of evaluating the productive capacity of a site, given some data on a habitat attribute likely to be altered by an activity, is the task of estimating the pdf of abundance (or agreed upon measure of productive capacity), given the pre-activity value(s) for the attribute(s) and the pdf using post-activity value(s). Of course the activity need not take place; one can estimate the likely direction of change in the habitat attribute, and explore how the pdf changes in that direction on the independent variable. The net gain or loss in productive capacity is the integrated difference between the pre and post pdf's.

Several terminological points need to be kept in mind. When I use pdf, I mean the cumulative probability density function, or ogive. The "reference data base" are the data on productive capacity and habitat one has to work with, whatever their source was. (If one were doing a regression analysis, they are the data used to estimate the slopes and intercepts of the model(s). "Reference sites" are individual cases from the reference data set. When I refer to "distance" or "nearby" or "similar", this is distance along the axis of the habitat gradient of interest. The "site of interest" or the "test site" is the state of the habitat for which a prediction is required. It is unlikely that one has measured productive capacity at a site identical to the site of interest, but one has measures of the habitat. (Again using the regression analogy, this is the "test" point for which the model is used to make a prediction about productive capacity). Other sites are the reference data base, for which one does have measures of habitat and productive capacity. I use abundance for "productive capacity", but the methods work in identical ways for any other attribute which is accepted as reflecting productive capacity.

## C. ESTIMATING PDF'S USING KERNEL ESTIMATORS

To estimate the pdf of abundance at one habitat site, the kernel method gives more weight to abundance measures from reference sites nearby in habitat space, than to distant sites. The task is to define "nearby" in an empirical
and rigorous way. If the pdf's are steep, observations from quite similar reference sites get a large weighting in constructing the pdf and dissimilar reference sites receive little weight in the pdf. If the pdf's are shallow, the weight given to observations declines slowly as the reference sites become progressively more different from the site of interest.

Cross-validation, or the "leave-one-out" procedure, is a standard method for determining exactly how fast influence should fall off with distance from the point of interest (Silverman 1986, Hastie and Tibshirani 1986). To start, from the reference data set the median abundance of all the observations from the second site to the last site is calculated, and the squared difference between the observed abundance at the first site and this median is calculated. The procedure is repeated for the second through the last observation, each time leaving out the observed abundance at the site being contrasted to the median based on all other observations. The sum of the squared differences is the jackknifed variance, when all observations are given equal weight. At each step, all observations except the deleted one received equal weight in determining the median, so this is the same as having influence not change at all with distance, or having an infinitely wide smoothing window. It represents the null hypothesis that the pdf of abundance does not change with habitat. For any window sizes smaller than infinite one needs a rule for allocating influence as a function of distance. A second power Cauchy function has been applied successful in a variety of univariate cases:

```
    W = 1/(1+[X/D] ')
where W = weight
    X = distance between trial value and reference value
    D = tuning parameter (or "window size")
```

Note, when $D$ is very large relative to $X, W \sim \sim 1$, so all points receive equal weight, regardless of X . This is our Null Hypothesis.

When $D$ is somewhere within the range of $X$ 's for a set of observations, observations close to the test value ( $X<D$ so $X / D$ is a small fraction) get relatively large weight (say $1 /(1+.25)$ for $X$ one half of $D$ ). Observations further the test value ( $X>D$ so $X / D$ is a value greater than 1 ) get relatively little weight (say $1 /(1+4)$ for $X$ twice $D$.

For multivariate cases (several habitat attributes predicting abundance, as with multiple regression) higher power Cauchy weightings appear to work well. They have not been shown to be optimal, although they are better than normal weights for all cases explored to date. There is no reason to expect one weighting function to be optimal for all data sets. However, independent explorations of general spline weighting functions did not find any weighting functions which outperformed our Cauchy weightings. In many instances where the Cauchy algorithm could be applied routinely, the family of spline functions was mathematically intractable).

Crossvalidation consists of recalculating the sets of ogives with progressively smaller values of the smoothing window, "D", (hence, "tuning" it). For each trial value of $D$ the $n$ unique ogives, each based on $n-1$
observations, are calculated. For each ogive the weight for each observation is allocated according to X (distance of the observation from the left-out value) and the weighting rule. The median is taken from each ogive, the difference between the left-out abundance and the abundance of the median case is squared, and summed over all n repetitions. This gives a jackknifed variance for each trial value of " D ".

If the habitat attribute does not influence abundance, no value of "D" gives a variance noticeably smaller than variance with infinite window. If there is a relationship, the variances will decrease with $D$ to some minimum, then increase again as the window becomes too narrow. With too narrow a window, observations with information useful in specifying the pdf are underweighted, so the ogives reflect the sampling error of the very closest points too strongly. The $D$ which produces the minimum variance is the proper value for weighting the reference observations when estimating the pdf of abundance at the site for which a prediction is required.

Once an optimum value for "D" has been determined, for each case in the reference data set, residuals can be calculated. For each reference case, the residual is the deviation of the observed value of the dependent variable from the median of the ogive based on all other observations given the optimum value of "D" and the observed value of the independent variable for the case whose residual is being estimated. These residuals can then be used to investigate influences of other independent variables, just as in stepwise regression analyses.

Regression models can be graphed with the dependent variable as one axis and the independent variable as the other (conventionally ordinate and abscissa, respectively). From one curve predicted values of the dependent variable can be taken for several values of the independent variable, by referring to different positions along the axes. No information on uncertainty of the predicted values are presented in these graphs, however. For the ogives, the graphs have axes of dependent variable (x-axis) and probability ( $y$-axis). The entire curve refers to expectations at a single value of the independent variable. To show predictions at several values of an independent variable, several ogives are mapped on the graph. This additional apparent complication occurs because each prediction contains the full uncertainty of the reference data. This additional information allows users to contrast the full difference in expected values of the dependent variable for different values of the independent variable, rather than just seeing how the medians change. The next section shows that the uncertainty surrounding the median predicted value often changes much more than the median value itself changes.

## APPLICATIONS

## A. SALMON AND TROUT AT HIGHLAND RIVER

## 1. The data base

These data are part of the data set reported in Gibson et al. (1987). Of the data in that report, estimates of total biomass of salmon and total biomass of trout were used as dependent variables. Independent variables used
were ones which were practical for habitat evaluation purposes; ones which could be collected with realistic levels of technical effort: They included:

Mean width of stream<br>Percentage cover<br>Height of apparent ice scour mark<br>Stream velocity<br>Minimum stream depth<br>Grain of substrate

Preliminary analyses demonstrated these six habitat attributes were highly intercorrelated. Use of cover, depth, and substrate gain captures the large majority ( $>88 \%$ ) of all the information in the six variables. Data from 20 sites on Highland River are available.

The scatter plot of salmon biomass and substrate grain show substantial variability in biomass at moderate to large grain, with low salmon biomass at fine grains (Fig. 1). Polynomial regressions up to order 3 all fail to reflect important aspects of the data.

The biomass of trout is very low at open sites, highly variable on sites of low cover, and fairly high at all sites of moderate to high cover. Although the polynomial of order 3 reflects some of the pattern in the data (Fig. 2), this curve does not reflect the variance at $10-30 \%$ cover, and would be very dangerous if used to extrapolate to high cover levels.
2. Analysis results - salmon

With a very large value of " D ", the variance in biomass was 1.018. Of cover, substrate and depth, the largest reduction in variance came with substratę grain as the independent variable. The residual variance was 0.559 , for an $r$ of 0.451 . This reduction came with a "D" value of between . 365 to .384 , around $1 / 8$ the observed range of the independent variable. Using this value of "D", ogives are drawn for four hypothetical sites, with substrate grain covering the range of observed values. For substrate grain finer than 3.3 there is a high probability ( $\geq .6$ ) of a biomass below 1.2 , and a low probability (<.15) of a biomass above 1.7 (Fig. 3). For an intermediate grain size of 4.4 , a wide range of biomasses are possible, although there is a greater than $50 \%$ chance of a biomass $>2.5$. For even coarser substrates intermediate biomasses are most likely, with only around a $20 \%$ chance of biomasses less than 1.2 or greater than 2.7. Note the gradual slope of the ogive at grains of 4.4 and 5.6 reflecting the variability in the observations, and the large uncertainty in expected value for individual samples from such sites.

Once the independent variable accounting for most variance in abundance was determined, residuals were calculated, and their relationships with Depth and Cover were explored. Depth accounted for none of the residual variance (no value of "D" produced a variance lower than with the largest "D" value), but Cover accounted for $8.3 \%$ of the residuals in substrate. With large "D" variance in residuals $=0.609$; with a $D=11.0$, variance $=0.559$. For test sites all ogives are fairly shallow (Fig. 4). With very low cover (5\%) negative residuals have a probability of over .75. For cover levels of 70 to $90 \%$ the ogives change little. Positive residuals have a probability >.6. For sites with a given substrate grain, ones with very low cover will have slightly lower abundances than expected, whereas sites with moderate to high cover will have slightly higher abundances. From the shape of the ogives, predictions for open, fine grained sites have the least uncertainty (i.e. narrowest range of likely values). Predictions for sites of intermediate grain and high cover will have the widest range of likely values.

## 3. Analysis results - trout

With large values of " D ', the variance in trout biomass was 5.763. The crossvalidation reduction in variance wąs greatest for Cover; with "D" in the range 2.0-2.4 variance was 1.663, for $r=0.711$. For a series of hypothetical sites the influence of cover on trout abundances is apparent. The ogive is very steep for sites of low cover, with less than a 0.1 probability of a biomass greater than 1.0 (Fig. 5). For 35\% Cover, there is still a $20 \%$ chance of a very low biomass, but the very shallow slope between biomasses of 0.5 and 4 suggest values anywhere in this wide range are likely to be observed. For Cover of $70 \%$ there is a very low likelihood of biomass less than 4.0. For very high Cover, the ogive is still steep above 4.0, but there is nearly a $40 \%$ chance of observing biomasses anywhere in the wider range from 0.5 to 4.0 .

Residuals from the reference data were calculated, given the specific cover values and the crossvalidated optimum value of "D". These residuals were again unrelated to Depth, but $8.9 \%$ of the residual variance was explained by ogives based on substrate grain with "D" $=0.22$ (Fig. 6). At fine (2.1 and 3.3) or coarse (5.6) substrates the ogives are steep between -0.5 and +0.3 . This suggests that trout biomass is not strongly affected by extreme substrate levels, given a specific level of cover. For intermediate (4.4) grain, however, the ogive is somewhat less steep, with residuals likely to be from the range -0.4 to +1.5 . This suggests that for substrate of intermediate grain, trout biomass is likely to be somewhat higher than expected given the level of cover. Were an activity such as forest cutting to affect stream cover levels, or cause siltation, these ogives could be used to quantify the impacts.

## B. COD TRAP CATCHES VS. WATER TEMPERATURE

## 1. Data base

In 1986 NORDCO conducted a study of the inshore fishery in Conception Bay, under contract to DFO. A number of cod traps were instrumented to collect water temperature profiles, and daily catches in these traps were recorded. NORDCO reported depth of various isotherms as their indices of water temperature, and catches were recorded in kilograms. These analyses
used the depth of the $2^{\circ}$ isotherm as the independent variable, because this is suggested to be the lower end of the range of temperatures preferred by cod. All isotherm depths were highly intercorrelated.

The original NORDCO study noted the wide scatter and nonlinearity of the catch-temperature relationship (Aggett et al. 1987). Their analyses fit quadratic models to the data, curiously constraining all polynomials to pass through 0,0 . Although the constraint allowed parameters of quadratic regression models to be estimated, in all cases the top of the curve was just to the left of the first observation in the scatterplot (Fig. 7). This suggests that the 0,0 constraint, which lacked any biological justification (and is counterintuitive in that it was applied to all isotherms), is largely responsible for the parameter estimates.

Predictions of such models cannot be reliable. Moreover, the models do not represent the large uncertainty present in the observations, nor how the uncertainty varies along the habitat axis.

## 2. Analysis results

The crossvalidated variance in trap catch per day is $11.32 \times 10^{5} \mathrm{~kg}^{2}$, with a "D" value of 100 (much greater than the observed range of depths; $16-40 \mathrm{~m}$ ). With a "D" value of $3 \mathrm{~m}, 28 \%$ of the variance is explained. Although $28 \%$ appears modest, given the huge variability in catches with $2^{\circ}$ isotherms between 16 and 25 m , little more could be expected. It is slightly better than the $r=.268$ ), from a linear regression model giving a very large intercept ( 2165 kg with the $2^{\circ}$ isotherm at the surface) linear and an unacceptable pattern of residuals. Other covariates may be important, but are not available.

Using the kernel estimators for the hypothetical case of the $2^{\circ}$ isotherm at 36 m , the ogive is very steep (Fig. 8). Trap catches under these very cold conditions have a probability less than .25 of exceeding even 400 kg . For somewhat warmer conditions, with the $2^{\circ}$ isotherm at 28 m , the ogive suggests catches can be anywhere between 200 and 2000 kg , but are very unlikely to exceed 2000 kg . For the warmest test condition (18 m), catches below 500 kg are very unlikely, and there is a .5 probability catches exceed 1500 kg . The very good catches of $2000-4000 \mathrm{~kg}$ still have a probability of only about .15 but that is the reality of the fishery. There is no temperature regime which ensures very good catches. To have more than a negligible chance at such catches, however, the $2^{\circ}$ isotherm had to be above 20 m . Were an undertaking such as building a causeway to influence mixing regimes, these ogives could be used to quantify impacts on the resource, and the fishery.

## C. CAPELIN DENSITY VS. WATER TEMPERATURE

## 1. Data base

In several years, research surveys of 3 N O capelin have produced hydroacoustic estimates of capelin density and recorded bottom temperatures. These data have been contoured in various ways, but extensive analyses have not been conducted. It is believed, however, that variation in water temperature influences the distribution of capelin, and indirectly capelin reproductive success. Stepwise polynomial regression suggests there is little
relationship between density and temperature in 1981, and a modest curvilinear one in 1987. Note the quadratic function underestimates abundances at temperature above $2^{\circ}$, and suggests density is increasing at negative temperatures (Fig. 9).

## 2. Analysis results

The 1981 data have a crossvalidated variance of 423.06. No value of "D" reduces this variance more than $1.5 \%$. This suggests the 1981 data do not demonstrate any relationship between density and temperature. The very steep left limit of the ogives (Fig. 10) at a variety of temperatures reflects the low density which prevailed at most sites. Log transformations of density do not change these relationships.

In 1987, there were more cases of high density of capelin. The total variance is correspondingly greater; 744.57. At a temperature window of 0.4 the crossvalidated variance is 351.68 for an $r$ of .528. This is substantially better than the regression analyses of these data. The ogives at different test temperatures are markedly different as well (Fig. 11). For a very cold water ( -0.5 ) the ogive resembles the 1981 ogives, rising steeply with little likelihood of capelin densities above 10 . A $2^{\circ}$ increase to $+1.5^{\circ} \mathrm{C}$ changes the ogive very little. Another $2^{\circ}$ increase to $3.5^{\circ}$ changes the ogive dramatically. There is still a . 25 probability of very low density but a . 50 probability of capelin density greater than 70 units. Note the nearly flat ogive between 2 and 70 reflects the schooling nature of capelin. In years when capelin are abundant, if they are present in an area, they are likely to be in dense schools. Again, however, water temperatures which appear preferred do not ensure dense schools of capelin will be present.

This example is included although there are not direct parallels between the capelin-temperature relationships and habitat impact evaluations. The example shows that the kernel estimators perform in clear ways when there are no relationships (1981 data), and how the ogives display clearly the patterns of abundance characteristic of schooling species. Note the quadratic regression equation in 1987 produces a line falling between the low and high densities, predicting the average condition which is rarely observed. The kernel estimation methods do not create relationships where none exist, and handle data from aggregated species much more appropriately than regression models do. HEP models, of course, do not differentiate areas of dense aggregation from other distribution patterns, in applying convex hull analyses.

CONVERTING OGIVES TO IMPACTS
Once the ogives of the site are calculated for the pre- and post-activity conditions, the impacts of the project can be measured. On the surface this could be a very simple exercise, either picking off medians from the ogives, or contrasting the integrated abundances from the ogives. If one has ogives for a main predictor, and ogives of residuals, these can be combined readily, as the joint probability of abundance is the product of the independent probabilities. Either comparing medians or integrated abundances is comparable to what is possible with predictions based on regression on HEP models; point estimates are compared.

Given the full ogives, however, managers can really investigate consequences of activities. Ogives with similar medians can differ greatly on the limbs. An increase in the lower limb of an ogive indicates the likelihood of a population's disappearance from an area has increased, regardless of the shape of the upper end of the ogive. Such differences may have substantial implications for population dynamics of the species and for resource users. Managers may not consider a large increase in uncertainty of a population's status to be a trivial impact, even if the mean population size is not altered greatly. The ogives provide such information directly.

Ogives may be particularly useful in planning mitigation for unavoidable habitat losses. Ogives can be calculated for the existing conditions at series of possible restitution sites. Ogives can also be calculated for likely post-mitigation conditions at each site, and the gains from each alternative mitigation program, can be estimated directly. If a specific level of gain is required, the degree of habitat change needed to produce the gain can be estimated for each candidate mitigation site. In either context the ogives give managers enough information to conduct meaningful negotiations with those required to mitigate. For a variety of alternatives of comparable cost, managers could select the one with most desirable properties; lowest likelihood of low populations; lowest uncertainty; highest probability of a large population; or whatever.

The fact that there has been little attention to exactly what outcomes of mitigation are most desirable (or what outcomes of undertakings are most important to avoid) says something of how primitive our tools for measuring habitat-stock associations have been. The provision of these ogives may prompt a long-overdue discussion between habitat scientists and habitat managers about what characteristics of populations should be the target of habitat management.

There is a final important consideration of the provision of ogives as management advice. Follow-up field evaluations of some projects will certainly occur. In the past the results of follow-up studies frequently suggested the initial evaluations were flawed, because the field study estimated abundances which differed from those predicted. When the ogives are available, the full uncertainty of outcomes is presented clearly. The ogives often show that a 1 or 2 year follow-up study with modest levels of sampling is a poor test of the effectiveness of a habitat management initiative. Even if the initiative were successful, any of a wide range of population estimates could be expected from such a study. When consulted, habitat scientists commonly provide pessimistic opinions about the value of small, short-term impact evaluations. The ogives give the habitat scientists quantitative support for the positions they take on the basis of their educated intuition. The whole domaine of impact evaluation, mitigation and negotiation is moved to a more realistic foundation. Scientific credibility may be enhanced greatly.

## Discussion

The ogive estimation approach to quantifying stock-habitat relationship has been illustrated for three different types of independent and dependent variables. In all cases applications were straightforward, and results readily interpretable, although in none of the cases were the data collected
for the purpose of estimating the pdf's of abundance from habitat characteristics. The method has the required flexibility and generality. Without asserting any specific, or even consistent relationship between habitat and stock, predictions are possible anywhere along the observed range of the habitat attributes.

The ogives themselves contain a great deal more information than is provided by alternative ways of handling such data. Although in theory regression based models can predict confidence intervals as well as expected values, such predictions require stringent assumptions about underlying error distributions. Moreover, one or two apparent outliers in the reference data will strongly influence the parameters estimated for regression models, and hence predictions anywhere along the range of the habitat attribute. In the pdf approach an occasional aberrant value will appear as an extended limb of the ogive, but the influence of the outlier will diminish quickly outside the range of the habitat attribute where the exceptional abundance was recorded.

Providing ogives to managers, instead of just mean values demands more of the managers. Minimally, it will require managers to consider more thoroughly exactly what properties of abundance they wish to manage for. This is not a bad thing. It also gives all users a clear picture of the range of observations likely to be collected if the population at any specific site were actually surveyed. This could lead to much more realistic expectations from all clients.

Finally, the pdf estimation approach forces scientists to consider their observations as individual random samples from pdfs. This provides a natural and consistent framework for systematically treating variability among replicate samples at a site and time, as well as seasonal and annual variation. Again, any advice will be improved by such a framework.

All the illustrations presented in this paper are univariate or stepwise multivariate applications. Our recent work has generalized the method to multiple predictors, which may be intercorrelated to unknown extents. We have also applied them to a spatially large but finely grided site. Both extensions become more demanding of computer time, but present no new conceptual difficulties.

Summary
The provision of advice on net gain or loss of productive capacity of fish habitat requires new quantitative tools. These tools must be applicable with any habitat attributes potentially altered by human activity, and be realistic in their data requirements. Nonparametric probability density estimation methods have these properties.

This paper describes how one variant of these methods - kernel estimators - can be used. The kernel estimator used is a family of Cauchy functions. The parameter of these functions is the smoothing term; how influence falls off with difference between the value of the independent variable at a reference data point and at the test site. Crossvalidation can be used to estimate this parameter, given a reference data set.

The method is illustrated with three different problems. The first problem relates salmon and trout biomass to cover and substrate grain. The second problem relates daily catches of cod by traps to the depth of the $2^{\circ} \mathrm{C}$ isotherm. The third problem relates hydroacoustic estimates of capelin density to bottom temperatures. These problems were chosen because the statistical properties of the independent and dependent variables are quite different, yet the kernel estimators work equally readily in all cases.

The kernel estimators provide full cumulative probability density functions for each prediction. These pdf's contain much more information than predictions of regression or HEP models. Pre and post activity evaluations can be made by comparing differences in the full ogives, or in summary properties of them. Mitigation planning is also facilitated by the full ogive because activities can be designed to develop the properties of the ogive which are most desired. The ogives may demand more of habitat decision makers and show explicitly the range of outcomes likely under a given scenario. This should be considered an asset of the methods.

The pdf approach is general, flexible, easy to use, and provides information of direct relevance to decisions regarding habitat management. The information provided allows resource scientists, managers and users to make realistic and informed decisions about habitat use, planning, and mitigation. The method is a good candidate as the basic tool for provision of advice on fish habitat.

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Fig. l. Scatterplot of salmon biomass by substrate grain for Highland River data, with linear, quadratic, and third-power regression lines.

SALMON: Biomass on Substrate - Regress


Fig. 2. Scatterplot of trout biomass by cover density for Highland River data, with linear, quadratic, and third-power regression lines.

## TROUT : Biomass on Cover - Regress



Fig. 3. Ogives from Cauchy algorithm for salmon biomass, at four levels of substrate grain.

## SALMON: Biomass on Substrate



Fig. 4. Ogives from Cauchy algorithm for residuals of salmon biomass from ogives in Fig. 6, at four levels of cover density.

SALMON: Substrate residuals on cover


Fig. 5. Ogives from Cauchy algorithms for trout biomass, at four levels of cover density,
TROUT: Biomass on cover


Fig. 6. Ogives from Cauchy algorithms for residuals of trout biomass from ogives in Fig. 8, at four levels of cover residuals.

TROUT: Cover Residuals on Substrate


Fig. 7. Scatterplot of daily trap catch of cod by depth of the $2^{\circ}$ isotherm, for Conception Bay, in 1986.


Fig. 8. Ogives from Cauchy àlgorithm for cod trap catch, at three depths
of $2^{\circ}$ isotherm.


Fig. © Scatterplot of hydroacoustic estimates of capelin density by bottom temperature, for Southeast Shoal (NAFO Div. 3N) in 1981 and 1987. Linear and quadratic regression lines are plotted.


Fig. 10. Ogives, from Cauchy algorithm, for capelin density in 1981, at four different bottom temperatures.


Fig. 11. Ogives, from Cauchy algorithm, for capelin density in 1987, at four different bottom temperatures.


# Decision-Making Model for Assessment of Habitat Production Capacity 

## by

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#### Abstract

This paper presents a short review of a computerized dichotomous key which is being used to monitor the "ecosystem health" of the Great Lakes. The key is an expert system which requires yes or no answers to a variety of questions concerning biological, environmental, and fisheries data from which an overall measure can be calculated. The calculation is based on a large data base compiled from various sources, which are documented in the key. The resultant measures of ecosystem health for any particular time are saved for comparison with previous calculations and an assortment of summary statistics are provided. A similar key could be assembled for assessing and monitoring Atlantic salmon habitat productive capacity which would make existing information more readily available to managers and help to identify important areas where information is lacking.


## RÉSUMÉ

On présente court résumé d'une clé dichotomique informatisée qui est utilisée pour surveiller la "santé des écosystèmes" des Grands Lacs. Cette clé est un système expert qui requiert une réponse affirmative ou négative à diverses questions sur la biologie, l'environnement et la pêche desquelles un index global peut être calculé. Le calcul est basé sur une importante base de données recueillies dans diverses sources qui sont documentées dans la clé. Les indices ainsi obtenus sur la santé des écosystèmes à un momemt donné sont retenus à des fins de comparaison avec les indices déjà calculés. On obtient aussi divers résumés de statistiques. Une clé semblable pourrait être élaborée pour évaluer statistiques. Une clé semblable pourrait être élaborée pour évaluer et surveiller la capacité de production de l'habitat du saumon de l'Atlantique. Les gestionnaires disposeraient ainsi plus facilement des données disponibles et pourraient identifier les principaux domaines où il y a peu d'information.

## INTRODUCTION

Environmental degradation, loss of productive fish habitat in particular, has been observed in many situations over the past few decades: For various, reasons, the Laurentian Great Lakes became a serious concern some decades before aquatic ecosystems in Atlantic Canada were perceived to be at risk. It follows that the experience gained in coping with the various problems on the Great Lakes can provide useful guidance in suggesting means to deal with similar patterns of degradation in Atlantic Canada.

An important result of the complex of scientific, social and political problems associated with degradation of the Great Lakes was the Great Lakes Water Quality Agreement between Canada and the United States, signed in 1978 and subsequently extended. In essence, the agreement had as its purpose the restoration and maintenance of "the chemical, physical and biological integrity of the waters of the Great Lakes Basin Ecosystem". The implementation of this agreement gave rise to the need for methods to evaluate and monitor the health of the ecosystem. We wish to briefly review one such evaluation technique devised for that purpose, in the belief that the general approach used may be applicable to evaluating Atlantic salmon habitat.

The problem of evaluating Atlantic salmon habitat is different from the Great Lakes experience only in detail. Our concern is not the ecosystem "integrity" consideration inherent in the Water Quality Agreement, it is instead to find measures to satisfy the policy of the Department of Fisheries and Oceans of "no net loss" in the productive capacity of fisheries habitat. The emphasis may be different, but the analytical challenge is similar. In effect we need to learn how to quantify measures of environmental change that are as yet incompletely understood.

One such approach applied in the Great Lakes context was the compilation of a simplified form of "expert system" which aimed to provide scientists and managers, without extensive technical expertise, ready access to the essential consequences of the available scientific information on one key member of the Great Lakes biota. Based on the experience of a wide variety of scientists, extensively consulted by a committee of the Great Lakes Fishery Commission, a dichotomous key was chosen as a means of quantifying ecosystem health using lake trout, Salvelinus namaycush, as an indicator species (Marshall et al. 1987). This key takes yes and no responses to a set of questions on various aspects of lake trout biology, ecology and exploitation, and calculates a score which indicates the "health" of the associated ecosystem. The key may be run at regular intervals, or as data becomes available, to assess ecosystem health and compare the present score with past results. In this way improvements or degradation in ecosystem status can be monitored over time.

Our supposition is that lake trout exhibit sufficient similarities to Atlantic Salmon so as to suggest that a comparable system of analysis can be applied for measuring the habitat production capacity for Atlantic salmon. In this paper we briefly describe the
inputs and outputs of the key, present example questions from the key and suggest the types of modifications necessary to make it useful for Atlantic salmon.

## DESCRIPTION OF THE KEY

The key originally described in the Great Lakes Fishery Commission technical report number 49 (Marshall et al. 1987) was developed primarily for use in Lake Superior. Since then it has been expanded to include data for all of the Great Lakes and now readily allows its application to inland lakes and, in the present case, possibly rivers. The key is designed as a menu-driven software system which currently runs on MS DOS. The operator is presented with 47 questions in four general categories 1) Exploitation and Production, 2) Environmental (Biotic), 3) Environmental (Abiotic) and 4) Contaminants.

We have chosen several questions from the various categories to present as examples. Figure 1 shows question number 4 from the Exploitation and Production category. At the bottom of the screen a scale from yes (positive) to no (negative) responses is used to input the required answer for each question. The value used, from 1 to 5, indicates the level of certainty. Zeros indicate a total lack of information. In this way the model attempts to accommodate uncertainties and holes in the available data.

A help function is available to provide the operator with a "current status" of the ecosystem for each question along with a list of references to the scientific publications or scientific authority which provided the information. Similarly a rationale, along with references, can be selected which describes in detail the scientific basis for including the question in the key (Fig. 2). When all the questions have been answered (yes, no, maybe or don't know) the key generates summary statistics (Fig. 3) which determines the ecosystem score and its status, together with a variance estimate which indicates the degree of confidence in the estimate. The score ranges from 0 , which indicates extreme degradation of the ecosystem, to 100 which corresponds to a pristine habitat. The key displays results of past runs, allowing comparison of ecosystem scores over time, as a means to evaluate changes with time. There are additional data outputs which display the scores by category: Exploitation and Production, Biotic environmental, Abiotic environmental and Contaminants, so that particular sources of stress can be isolated. Graphic displays of summary statistics are also supported by the package.

## APPLICATION TO ATLLANTIC SALMON

In the present instance of assessing Atlantic salmon habitat attributes, the indicator species concept is probably appropriate only for the freshwater phase. A reasonable alternative is to develop a dichotomous key which is specific for evaluating changes in productive
capacity of Atlantic salmon habitat, using the general framework which has been laid down through the efforts of the Great Lakes experience.

Obviously, many of the questions in a key designed for lake trout would be inappropriate for direct application to Atlantic salmon. For example in question 3 of the environmental biotic section (Fig. 4), sculpins are unlikely to make up this percentage of food ingested by salmon in riverine environments. Appropriate questions for salmon concerning specific dietary components would have to be devised, based upon available expertise.

There are alternative questions in the key, concerning the biomass of forage species which support production at higher trophic levels, which can be adapted to the case of Atlantic salmon, e.g. question 10 of the exploitation and production section (Fig. 5). This question relates to the work carried out by the authors over the last several years (Dickie et al. 1987, Boudreau and Dickie 1989). Patterns in the ratios of biomass density at various trophic stages have been established, which are related to general production processes and organism body size. In the case of Atlantic salmon, one should be able to establish characteristic biomass densities and organism sizes for both salmon and its major prey components. We would expect relationships between these parameters to be predictive of habitat productive capacity in ways similar to observations in other predator/prey situations. Note, for this question, that where information is lacking for a particular ecosystem, relevant information from nearby ecosystems can be employed.

There are several questions in the key which deal with abiotic environmental effects which would be easily applied to the Atlantic salmon case. They deal with critical levels in temperature, pH , dissolved oxygen, photic zone depth, substrate characteristics and chemical concentrations. In figure 6 we present question number 6 in the environmental (abiotic) section. The necessary information on the range of ph required to maintain successful salmon populations is available (Watt 1987; Lacroix 1987, 1989) and suitable critical levels could readily be selected.

After a process of evaluating, selecting and modifying the existing queries in the key, it would be possible, using the general framework of the key, to incorporate new information on parameters which affect salmon habitat capacity. One example would be the use of information on preferred bottom gradients as indicators of salmon density (Amiro 1990). Estimates of threshold gradients for particular runs from remote or proximate surveys could be used by the key in determining potential salmon production.

## DISCUSSION

A dichotomous key for Atlantic salmon would be useful only to the extent that it can be made to reliably quantify net loss or gain of salmon habitat. There are several obvious considerations which would determine the usefulness of a computerized key in the case of Atlantic salmon habitat.

One concern is that the lake trout key has been designed for evaluating lake habitats and thus treats whole lakes as the spatial area of interest. In terms of habitat evaluation for riverine salmon populations, many different spatial scales could be seen as areas of interest. The most general spatial scale of interest would be broad geographic areas which could be distinguished on the basis of underlying geology, within which drainage basins would have some common habitat parameters such as pH buffering capacity (Watt 1987). More commonly, information on individual drainage basins, rivers or streams are used to characterize good or bad habitats and should be amenable to analysis using the key. Where data is available on finer space scales, such as for individual runs or riffles (Amiro 1990), it would be useful to utilize the key to evaluate critical spawning or feeding areas.

To facilitate development of a key suited to various spatial scales, it would be advantageous to incorporate geographic information system (GIS) techniques for data storage, handling and analysis (Minns 1989). GIS techniques provide the necessary methods for interpolating and contouring, so that areal estimates can be derived from point sampling. The ability to spatially model and overlay several parameters, and carry out an areal expansion of the results, allows ready identification of particular areas which will be most sensitive to changes in habitat criteria and the degree to which the productive capacity of different areas will be affected. In general, the use of GIS techniques in a key would aid habitat managers in determining the relevant spatial scales for assessing habitat loss and, as most requests for advice are site specific, the ability to query the data base by site location would be consistent with present practices.

A second consideration concerning the implementation of a key for Atlantic salmon is the availability of relevant information on the habitat parameters. It may be difficult at present to identify and quantify a set of appropriate parameters which would be sufficient to evaluate salmon production potential. The development of such a key for Atlantic salmon necessarily requires input from a great many researchers, using both existing data as well as commitments for future work. This was and is the experience with the lake trout key, which depends on extensive continuing consultation with a wide variety of experts.

There are several advantages to be gained from such a key. The greatest strength of this approach is that it assembles and organizes the existing scientific information and presents it in an interpretable form that is readily accessible to scientists, managers and other interested parties. It succeeds in this regard in direct proportion to the ability of the expert formulators of the questions to express their expert knowledge in the form of questions that are easily interpretable and not readily amenable to distortion or misunderstanding by the users of the system. In doing so, it makes use of the knowledge of scientists who have carried out the research to formulate questions which require only yes or no answers from operators who are not necessarily able or even interested in carrying out detailed analysis.

A second strength of the approach, in common with all attempts to model or formally codify a system of any sort, is that it identifies glaring lacks of knowledge about essential behaviours of the system being analyzed. The present considerations are no exception. It seems quite likely that much of the requisite information is available to construct such an expert system for the freshwater phases of Atlantic salmon, but our impression is that the necessary information is not readily available for the estuarine or marine phases. Although incomplete, information on the habitat requirements of salmon at sea seem relatively better understood for the Pacific salmons (T. Parsons per. comm.). Research on Canada's west coast may provide a basis for extending the application of a computerized Atlantic salmon key to estuarine and marine situations.

## CONCLUSION

We believe that a computerized dichotomous key incorporating available knowledge on Atlantic salmon, could be a useful approach for the evaluation of habitat productive capacity.

## ACKNOWLEDGMENTS

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LAKE TROUT DICHOTOMOUS KEY - LAKE GUPERIOR
SECTION: Exploitation and Production PAGE: 4 of 47
 Question 1

Do the lake trout use the majority of the historical spawning and feeding habitats?

DDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDD

DDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDD Current status:
Many of the historical spawning and feeding habitats are currently unused due to the apparent demise of a number of lake trout stocks.
$(35,37,73)$



Figure 1. Question number 4 from the dichotomous key in the Exploitation and Production section showing "Current Status" information.

## LAKE TROUT DICHOTOMOUS KEY - LAKE EUPERIOR

SECTION: Exploitation and Production
PAGE: 4 of 47
 Question 4

Do the lake trout use the majority of the historical spawning and feeding habitats?

DDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDD Rationale:
In the past, lake trout consisted of diversified stocks using many different spawning habltats and food resources. The demise of some lake trout stocks altered use patterns of these habitats.
$(26,35,73,74)$
DDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDD Current status:
Many of the historical spawning and feeding habitats are currently unused due to the apparent demase of a number of lake trout stocks.
(35,37, 73)


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## LAKE TROUT DICHOTOMOUE KEY

FOR 工凡KE BUEERIOR
assignment by：PRB Date：04／19／90

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CuIzent mOOBYEtEm status
```

Lake Trout exlbit extreme symptoms of＇stress＇，which represents a potentially critical concern and indicates the need for imediate and eassive rehabilitation measures for lake superior．

The overall level of uncertainty assoclated vith your responses to the questions is fairly lov，but inparts a silght concern for the reliability of your diagnosis．

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    上COBYBtem •ECOIE'
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Scores from this and previous runs through the Key provide an indication of the relative effectiveness of rehabilitation measures applied to Lake Superior．Although imprecise，they may be used to estimate the direction and rate of change in ecosystem＇health＇．scores represent an assessment of ecosystem well－being based on percentages，vith 100\％ indicating a near pristine state and 0 indicating extreme degradation．

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| TRM | $01 / 01 / 88$ | 69 | 10 |
| TRM | $01 / 01 / 70$ | 58 | 17 |
| TRM | $01 / 01 / 50$ | 83 | 19 |

Figure 3．Sample output from the lake trout dichotomous key showing ecosystem status and score from a particular running of the key along with date，score and uncertainty from three previous runs．

## LAKE TROUT DICHOTOMOUS KEY - LAKE GUPERIOR

SECTION: Environmental (Biotic) PAGE: 15 of 47
 Question 3

Your ansver 5
Do sculpins (MyOXOCEPHALUS sp., COTTUS sp.) constitute at least 10 by volume of all food ingested by adult lake trout?

DDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDD Rationale:
adult sculpins are extremely sensitive to cultural intervention and their absence in lake trout diet aight indicate a degraded habitat. sculpins
traditionally comprised about iof of the lake trout diet in Lake superior.
DDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDODDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDD Current status:
sculpins probably constitute $10 \%$ (by volume) or more of the diet of adult lake trout in offshore vaters, but do not contribute this much inshore.



Figure 4. Question number 3 from the Envirommental (Biotic) section showing a question which would require rephrasing for application in a potential Atlantic salmon key.

## LAKE TROUT DICHOTOMOUB KEY - LAKE BUPERIOR <br> 

SECTION: Exploitation and Production PAGE: 10 of 47
 Question 10

Do standing stocks (biomass) of lake trout of size range 1-10 kg constitute at least 15\% of the biomass of the principal forage species of size range 1-1000 g? DDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDD Rationale:
Assuming a flat, silghtly negative size spectrum characteristic of balanced system, lake trout biomass 3hould be $30 \%$ that of their forage vithin these size ranges. Reducing this ratio (through fishing) to $15 \%$ or less may be catastrophlc. DDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDD Current status:
While size structure relationships between lake trout and their forage have not been complied specifically for Lake superior, it has been predicted that these patterns do occur in Lake Ontario.

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Figure 5. Question number 10 from the Exploitation and Production section showing a question appropriate for application in a potential Atlantic salmon key.

## LAKE TROUT DICHOTOMOUS KEY - LAKE SUPERIOR <br> ~~N~~2



DDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDD Rationale:
A mean surface pH of 7.6-8.3 is within the traditional range of surface pH for Lake superior waters. Reduced growth and survival of lake trout has been associated with pH values less than 5.6.
$(128,48,40)$
DDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDDD current status:
Lake superior surface pH ranged from 7.8 to 8.3 in 1973 and averaged 8.0 for the entire year.

: Next Previous edit Review Citation
: Question Question answer Rationale quit Look-up


Figure 6. Question number 6 from the Environmental (Abiotic) section showing a question requiring slight modification for application in a potential Atlantic salmon key.

# Microhabitat Selection by Juvenile Atlantic Salmon in a Small, Unstable Stream 

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#### Abstract

Breacs Brook, Cape Breton Island, is a small (mean flow < 8 cubic metres per second) river flowing into the East Bay of the Bras d'Or Lakes. A kilometer long section has been disturbed by gravel mining. and has become very unstable, with considerable erosion and channel hopping. The study area consisted mostly of very broad, shallow riffles flowing between deep pools eroded around obstructions such as fallen trees. Juvenile Atlantic salmon were numerically dominant and were in sympatry with brook, brown and rainbow trout. The density of salmon was much greater in the disturbed area than in the control reach. Microhabitat selection by the salmon is described. The water depth above the fish's head was found to have a minimum of about 15 cm . The selection of focal point velocity and mean water velocity was directly related to river discharge. It is hypothesized that juvenile salmon utilize riffle habitat until forced to abandon it because of falling water levels. The fish resort to pool habitat in order to maintain suitable overhead water depth. This is contrary to findings from most published salmon habitat studies where water velocity appeared to be the dominant factor. The significance of marginal salmon habitats is discussed.


## RÉSUMÉ

Le ruisseau Breacs est un petit cours d'eau (débit moyen
inférieur à 8 m 3 par seconde) qui se déverse dans la baie Est du lac Bras d'or, au Cap-Breton. Un tronçon d'un kilomètre où s'est effectué l'exploitation du gravier est devenu très instable à cause de l'érosion et de la déviation des chenaux. La zone expérimentale couvrait surtout des seuils très larges et peu profonds bisectant des fosses profondes creusées autour d'obstacles comme des arbres morts. Le saumon de l'Atlantique juvénile était l'espèce la plus abondante et vivait en sympatrie avec la truite brune, la truite arc-en-ciel et l'omble de fontaine. La densité des saumons était beaucoup plus élevée dans la zone altérée que dans le tronçon témoin. On décrit aussi les microhabitats choisis par le saumon. Le saumon a besoin d'au moins 15 cm d'eau au-dessus de sa tête. Le choix de la vitesse d'écoulement recherchée et de la vitesse d'écoulement moyenne était directement lié au débit fluvial. On formule l'hypothèse que le saumon juvénile fréquente des seuils jusqu'a ce qu'il soit forcé de se déplacer à cause d'une baisse du niveau de l'eau. Il habite alors des fosses afin que la profondeur de l'eau au-dessus de sa tête soit adéquate. Cette hypothèse contredit la plupart des données publiées sur l'habitat du saumon selon lesquelles le débit semble être le facteur dominant. On examine aussi
l'importance des habitats marginaux du saumon.

## INTRODUCTION

The data presented in this paper were collected as a baseline for a study of the effects of stream stabilization on the production and habitat choice of juvenile Atlantic salmon. Breacs Brook, Cape Breton Island, was chosen as the study site because it had been perturbed by gravel mining and was subject to channel hopping and significant erosion of the stream banks. The 1986 reorganization at DFO removed the resources and manpower for the study and so it had to be cancelled.

It was obvious to observers that the habitat selection of the juvenile salmon differed from what is known from larger Maritime region rivers. The baseline data were adequate to allow estimates of density and the derivation of habitat suitability curves for the population. The Breacs Brook data present an opportunity to learn about salmon habitat use in marginal habitats and compare this to habitat use in apparently ideal habitats.

## METHODS

Field work for the study was conducted in Breacs Brook, Cape Breton Island. This stream originates near Loch Lomond, and flows into East Bay of the Bras d'Or Lakes at Big Pond. It has a gradient of about .6\%, good supplies of groundwater, a favourable pH , areas of well-sorted alluvial substrate and well-vegetated banks supplying overhanging arboreal cover. It is a small river with a mean flow less than 8 cubic metres per second. A kilometer long section has been disturbed by gravel mining and has become very unstable, with considerable erosion and channel hopping. The study area consisted mostly of very broad, shallow riffles flowing between deep pools eroded around obstructions such as fallen trees. Juvenile salmonids were found throughout this and the middle reaches of the stream.

Four species of salmonid were found in the river. They were Atlantic salmon (Salmo salar), brook trout (Salvelinus fontinalis), brown trout (Salmo trutta), and rainbow trout (Oncorhynchus mykiss).

Density of fish was estimated by electrofishing with a Smith-Root backpack unit. The area to be fished was blocked off with barrier nets. Fishing was done in a side to side and downstream direction with most of the subjects collected in a lip seine set just downstream of each in a mosaic of small fishing areas. Three passes were made over the barricaded area with a half hour rest period between sweeps. Density was estimated using the Zippin maximum likelihood method.

Microhabitat data were collected as in Morantz et al.(1987). Briefly, a snorkeller crept slowly upriver until a fish was sighted. The fish was observed at a distance from underwater until the snorkeller was satisfied it was holding in a position
of its choice. A suction gun or the Longard inverted hoopnet (Morantz et al. 1987) was used to capture the fish. The location of the fish over the streambed was marked with a fluorescent orange marker either immediately before or after capture.

Captured fish were measured in water in a small measuring trough to the nearest millimetre, and weighed in water to the nearest tenth gram on an Ohaus C501 pan balance. The use of anaesthetic was unnecessary because the fish remained relatively calm if kept immersed during handling. Microhabitat data were collected from each fish's holding position shortly after capture.

Variables measured were: fish species, fork length, weight, height of fish above substrate, water depth, focal point velocity (Fausch and White 1981), mean water velocity (at standard hydrologic 6/10 depth), substrate (percent of sand, gravel, cobble, boulder), and cover (overhanging or submerged branches, broken water, weeds, undercut banks, fish under stone, etc.). At the end of each snorkelling session (about three hours) or at the completion of fishing a stream reach, the fish were released from a live box as close as possible to the area of capture.

Habitat suitability curves were developed using the methods of Bovee and Cochnauer (1977) and Morantz et al. (1987). The curves are essentially the frequency distributions for values of each variable at the locations selected by the fish. Chi- square tests were used to determine the significance of each frequency distribution from an equal distribution of expected values. Most. computations used appropriate programs from Nie et al. (1975) or Wilkinson (1987).

## RESULTS

Density estimates for the four species of salmonid present in Breacs Brook are presented in Table 1. The data are for September 1985, which represents the season when all species would be well represented by the $0+$ year class, the rainbow trout being a spring spawning stock with the fry emerging in mid-July. Density estimates were made earlier in the year and the data showed similar patterns with the exception of fewer rainbow trout. Numerically, salmon fry were the dominant life history stage with salmon parr dominant over other parr. Brown trout fry were the next most plentiful group followed by rainbow trout fry. Rainbow trout parr were present at low density (2-5/100 sq. metres) earlier in the season but disappeared in the autumn.

Four variables were considered in defining the microhabitat used by juvenile Atlantic salmon. These were focal point velocity, column velocity, water depth, and distance of the fish from the river bank. Focal point velocity and water depth are discussed below. Rejected variables were column velocity, which was autocorrelated to focal point velocity, and distance from the river bank, which was found to be random.

Suitability curves for water depth favoured by juvenile Atlantic salmon are presented in Figure 1. The solid lines represent data from the current study while the dotted lines represent data from Morantz et al. 1987. Water depths of less than 10 cm were rarely chosen while depths between 10 and 20 cm were chosen more often by fry but seldom by parr. Inspection of the data showed that most of the depths chosen in this range were in fact deeper than $15 \mathrm{~cm}(\mathrm{n}=42,69 \%)$. Only 5\% ( 30 of 622 ) of all the salmon sampled were in depths of less than 15 cm . The most favoured depth for fry was from 20 to 30 cm while for parr it was 30 to 50 cm .

Suitability curves for focal point or nose velocity for juvenile Atlantic salmon are presented in Figure 2. The solid lines represent data from the current study while the dotted lines represent data from Morantz et al. 1987. The apparent preference in all three age classes in the present study is for slow velocities between 0 and $5 \mathrm{~cm} / \mathrm{sec}$. with a secondary preference for velocities between 10 and $15 \mathrm{~cm} / \mathrm{sec}$. This represents quite a departure from the results obtained by Morantz et al. (1987) which showed a progressive increase in velocity preference that was positively related to the size of the fish and ranged from 5 to $20 \mathrm{~cm} / \mathrm{sec}$.

The relationship of mean water depth and mean focal point velocity to river discharge over the sampling season is depicted in Figures 3 and 4 for 1985 and 1986 respectively. Anova and ttests showed that the mean values within an age group were heterogenous over time in each year. The figures show that the choice of focal point velocity was strongly related to discharge ( $\mathrm{r}=.69$, $\mathrm{p}<.01$ ) while the choice of water depth was independant of discharge ( $\mathrm{r}=.078$, $\mathrm{p}=.66$ ). The strong relationship of focal point velocity to discharge shows that the fish did not compensate for the effects of flow changes in their microhabitat. Conversely, the poor relationship of water depth chosen by the fish to discharge shows that the fish did compensate for falling water levels by moving into deeper water. This can be seen in Figure 3 where during the low flow period the mean water depth chosen by fry increased, indicating that the fish had moved into pools.

## DISCUSSION

Juvenile salmon remain numerically dominant in Breacs Brook even after spending the summer season in 'classically' unfavourable habitat. Also, the juvenile salmon remain numerically dominant even after an unknown number of generations sharing the habitat with three other salmonid species, two of which are exotics. At present, one can only speculate as to whether this is a result of genetic selection for existence in a small stream habitat. An alternative hypothesis is that it is the result of greater egg deposition by adult fish.

The juvenile salmon of Breacs Brook selected water velocity early
in the season but maintained a minimum overhead depth by abandoning riffles for the deeper but slower water in pools during the low water period. They returned to faster velocity when water flow increased in the autumn. Consequently, what could be termed a habitat suitability curve for focal point velocity is in fact no more than a statement of where the fish were located.

Morantz et al. 1987 (with references) advanced the concept of universality in habitat selection by a species and affirmed that this appeared to be the case for Atlantic salmon. The present data appear to contradict this assumption. A more likely explanation may be that the animal has a hierarchy of habitat variables that it will optimize within thresholds. When a critical variable reaches an unacceptible threshold, selection for another variable will be abandoned in order to satisfy the demands of the primary variable. In the present study, the need to maintain a satisfactory water depth during the summer low water period overrode the selection of focal point velocity and the fish moved into pool habitat.

The freshwater phase of the Atlantic salmon's life history is generally regarded to be a period of feeding and growth. However, juvenile salmon may spend a large proportion of their time in refuge habitats. Barbour et al. (1979) found that oananiche parr and adults had added most of the growth rings to their scales by early July and had a good condition (K) factor. Amiro (pers. comm.) has found that parr put on a greater proportion of their growth early in the season. It may be that some Atlantic region salmon, from less than ideal habitats, may actually devote relatively little of their year to actual growth. The rest of the year may be spent coping with the environment, probably by resorting to pools in low water in summer and burying themselves in the substrate to overwinter (Rutherford pers. comm.).

This study raises several points about habitat suitability curves. First, a suitability curve drawn from a small range of habitats or stream conditions may not be a suitability curve at all simply because the fish may have been forced into the observed habitat. Second, when deriving curves, it is important to account for temporal changes in behaviour and habitat use, be that on a diel, seasonal, or annual basis. Third, the study of an unusual circumstance may point out previously unacknowledged survival techniques in a species.

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Table 1: Density estimates of various salmonids in Breacs Erook.

## BREAC'S BROOK

SALMONID DENSITIES - SEPT. 1985

| SITE | TREATMENT | CONTROL | U. CONTROL |
| :---: | :---: | :---: | :---: |
| SALMON FRY | 74 | 114 | 5 |
| SALMON PARR | 6 | 7 | 7 |
| BROOK TROUT FRY | 7 | 7 | 8 |
| BROOK TROUT PARR | 1 | 4 | - |
| BROWN TROUT FRY | 22 | 13 | 26 |
| BROWN TROUT PARR | 2 | 4. | - |
| RAINBOW TROUT FRY | 26 | 9 | - |
| RAINBOW TROUT PARR | - | - | - |



Fig. 1: Suitability curves for water depth favoured by juvenile Atlantic salmon.


Focal Point Velocity (cm/s)
Fig. 2: . Suitability curves for focal point velocity for juvenile Atlantic salmon.

## USE OF VELOCITY AND DEPTH - 1985



Fig. 3: Relationship between mean water depth and mean focal point velocity to river discharge over the 1985 sampling season.

## USE OF VELOCITY AND DEPTH - 1986



Fig. 4: Relationship between mean water depth ana mean focal point veiocity to river discharge over the 1986 sampling season.

## Session III

Methods to Measure Factors used to Estimate Productivity or Changes in Productivity

# Methods to Measure pH 

by

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#### Abstract

The measurement of hydrogen ion activity in natural water systems is widely reported in pH units. A brief synopsis of the theory of pH measurement is presented, including potentiometric and colorimetric methods. The particular difficulties of pH measurements in water of low conductivity are outlined and various troubleshooting and maintenance points are reviewed. A procedure for collecting samples and measuring pH in brown water of low conductivity is recommended.


## Résume

La mesure de l'activité des ions hydrogène dans les systèmes aquatiques naturels est généralement signalée en unités de pH . On présente un résumé de la théorie sous-tendant la mesure du pH , y compris les méthodes potentiométrique et colorimétrique. On fait aussi un bref compte-rendu des problèmes particuliers à la quantification du pH dans les eaux à faible conductivité, ainsi que divers diagnostic d'anomalies. Une méthode de collecte d'échantillons et de mesure du pH des eaux brunes de faible conductivité est recommandée.

## Introduction

The pH unit is a commonly used parameter of water quality. Variations in measuring techniques, however, often result in inaccurate and irreproducible values. A standard approach to pH measurements in highly colored water of low conductivity typical of the salmon rivers of Nova Scotia would result in more reliable data and allow meaningful comparisons of data sets.

## Theory and Methodology

The measurement of hydrogen ion activity in solutions is reported in pH units, which range on a logarithmic scale of decreasing hydrogen ion concentration from 0 to 14. A pH measurement system consists of an indicating electrode, a reference electrode, the test solution and a potentiometer, typically a pH meter. The indicating, or glass electrode, is comprised of an internal reference element and related electrolyte of fixed ionic concentration (for example, silver element and silver chloride electrolyte) within a sensitive glass membrane. When immersed in test solution, a constant potential, or electromotive force, from reactions between the reference element and electrolyte develops on the inner surface of the glass membrane, while on the outer surface another potential is built up with strength dependent upon the hydrogen ion activity in the test solution. A reference electrode is required to measure the difference between these two potentials. A piece of wire immersed in the sample could complete the circuit and give a measurement, but a stable and constant potential is required. The reference electrode is constructed to maintain a constant potential, again with a reference element and electrolyte. The measuring circuit is completed with a "liquid junction" whereby small amounts of the reference electrode electrolyte flow into the test solution. Thus, the potential across the glass membrane affects the potential of the whole system because the other potentials are designed to remain constant. This potential difference is measured and translated into pH units by the pH meter. The glass and reference electrodes are often combined into a single probe body; this is a combination pH electrode.

The performance of the electrodes and meter system is calibrated with solutions or buffers of known pH values. At pH 7 , the potentials on the inside and outside of the glass membrane are equal; this is the electrode isopotential point. Calibration with two reference solutions spanning the expected pH values of the test solution allows the pH meter to accurately translate the measured electromotive forces of the particular electrode assembly into pH units. The potential generated is 59.1 mV per pH unit at $25^{\circ} \mathrm{C}$, but this value varies with temperature, as do ionization rates. Many pH meters have temperature adjustment controls to compensate for these factors, but it is generally recommended to have test and standard solutions at the same temperature.

In waters of low conductivity and weak acidity, readings of potentials are erratic and slow to stabilize because of low ionic activity in the test solution. This problem is alleviated by using special low resistance glass for the membrane of the indicating electrode, allowing stable potentials to be more readily attained. Buffer solutions, high in ionic activity and cause of possible contamination at the membrane surface, must be thoroughly rinsed from the electrodes when measuring waters of low conductivity.

The electromotive force of the liquid junction, located where the electrolyte of the reference electrode comes in contact with the test solution, is measured along with the potentials of the reference and glass electrodes, and can change with pH , temperature and ionic strength of solutions. Large differences in ionic strength between buffers and water samples such as those that are of concern to us, can produce different junction potentials and cause slow and erratic electrode response. This effect can be reduced by diluting the ionic strength of buffers and increasing ionic strength of samples by adding an inert salt, such as potassium chloride (Boyle et al. 1986) or by using standards of dilute solutions of strong acids rather than buffers; these acid standards must be frequently and carefully verified by titration (Galloway et al. 1979). The effect of temperature and pH on liquid junction potential can be mitigated by selecting standards close to expected pH values of samples and having test and standard solutions at the same temperature.

The type of liquid junction is important in dealing with samples of low.conductivity. The electrode assembly performs best with the smallest junction potential, and this occurs with a fast flowing junction ( $100 \mu \mathrm{~L} / \mathrm{h}$ ) such as provided by a sleeve junction. Other liquid junctions include porous ceramic plug ( $8 \mu \mathrm{~L} / \mathrm{h}$ ) and cracked bead ( $2 \mu \mathrm{~L} / \mathrm{h}$ ). Although the relatively large amount of electrolyte flowing into the test solution enhances conductivity and response time in samples of low ionic strength, it could cause significant contamination of very small samples.

Stirring in samples of low conductivity is necessary to maintain an equilibrium with atmospheric carbon dioxide. However, this produces a streaming potential at the indicating electrode which varies according to electrode type and ionic strength of solutions. Opinions about the necessity of stirring while measuring pH in samples of low ionic strength vary. Bates (1973) recommends constant vigorous stirring, Galloway et al. (1979) indicate that initial stirring followed by measurements on quiescent solutions should occur, whereas Boyle et al. (1986) found no effect of stirring.

When electrode response becomes slow or erratic, cleaning may be necessary . Commercial solutions are available for removing proteinaceous contaminants from the glass membrane and for clearing plugged liquid junctions, a common source of malfunction. Electrode membranes can also undergo leaching from the sensitive glass surface, resulting in noisy or slow response. These electrodes can generally be reconditioned with reactivating solutions of dilute hydrofluoric or hydrochloric acid. Proper storage is essential to maintain an accurately functioning electrode. Glass electrodes should be stored in buffer solutions, while reference electrodes should be stored in saturated potassium chloride to keep the liquid junction moist and free-flowing. Combination electrodes are best maintained in a mixture of buffer and potassium chloride. Most electrode companies sell prepared solutions for purposes of cleaning, reactivating, and storing their electrodes.

The colorimetric method is another way to determine pH that was used extensively in the past. Indicators are added to the sample and the resulting color is compared with colors of solutions of known pH containing the same amount of indicator. The indicator compounds exist in two ionic forms of different colors, the amount of each form, and therefore overall color, dependant upon hydrogen ion concentration. By first approximating pH with a wide-range indicator, the most appropriate narrow-range indicator can be selected for color matching. In waters with no alkalinity, colorimetric indicators vary widely in accuracy over their operating range; some are consistently lower than the electrometric results, some are consistently higher, and others are higher at the low end of their range and
lower at the high end of their range (Haines et al. 1983). The method is generally not very suitable for pH determination in waters of low alkalinity such as those of most salmon rivers of Nova Scotia with acidic pH levels. These waters are typically highly colored, a factor which can also interfere with comparisons to standard colors. Furthermore, the indicator solutions are weak acids and bases, so poorly buffered waters such as those found in the Nova Scotia river systems undergo significant pH change upon addition of the indicator thus producing unreliable results (Bates 1973).

In comparing historical water quality data with recent data, the following points should be considered. Comparison of colorimetric pH measurements with recent electrometric results generally requires an adjustment; some formulae to that effect are outlined in Kramer and Tessier (1982). They noted that most container material available before 1960 (e.g., soft glass and copper collectors) could have contributed alkalinity to the water sample and increased pH levels in past studies. In contrast, the inert plastics of high quality now used for sample collection and storage generally do not affect the pH of samples. However, the reuse of these sample bottles and the procedure of "acid washing" them can, even after thorough rinsing, leave traces of acid which then contaminate samples and can introduce a bias towards a greater acidity than that in situ. The use of disposable bottles to be used only once is therefore recommended.

The brown waters typical of salmon rivers in Nova Scotia are colored by high concentrations of humic and fulvic acids, and pH levels of 4.5 or less are regularly recorded. Conductivity is typically less than $35 \mu \mathrm{~S}$, reflecting the low ionic strength of these waters (Farmer et al. 1980). The method using dilute buffers for calibration and samples spiked with potassium chloride to increase conductivity as previously described has until recently been the preferred way of measuring pH in samples from these rivers. However, recent technological advances have produced electrodes designed for pH measurement in low conductivity waters, overcoming the problems of slow response and drift, and replacing the "dilute and spike" method in many labs.

## Conclusion

The following is a recomrnended laboratory/field procedure for pH measurements in low conductivity, highly colored waters of low alkalinity:

- Collect samples in unused 500 to 1000 mL polyethylene bottles; rinse bottle 3 to 5 times with the water sample to be collected, fill completely by immersing bottle below surface and facing upstream, and cap underwater; keep sample cool and analyze as soon as possible. - Use Ingold ("Acid Rain Electrode" Model \# 41-401-3000/G) or Ross (Ross-Orion Research Ltd. Model \# 810200) low conductivity combination glass pH electrode.
- Electrode should have been stored in electrode storage solution; make sure it is clean, the membrane not in need of reconditioning, and the electrolyte well full.
- Only use the electrode for pH measurement in samples of river water and not for other applications or in other solutions.
-Ensure that buffers, electrode, and samples are at room temperature if in the laboratory, or at similar ambient temperature if in the field.
- Use a high quality pH meter (e.g., Fisher, Metrohm, Orion, Radiometer) and calibrate electrode as recommended by the manufacturer with pH 7 and then pH 4 buffers using appropriate corrections for temperature.
- Always rinse electrode copiously with deionized water after use in the buffers.
- Rinse laboratory glassware with sample and use adequate sample volume for measurement (minimum recommended volume, 100 mL ).
- Use magnetic mixer with tetrafluoroethylene (TFE) fluorocarbon-coated stirring bar to provide continuous stirring while taking measurements, and place insulating material such as styrofoam between beaker and mixer to prevent sample warming.
- Record pH level when reading is stable; this should occur within 2 to 5 minutes.
- Both samples and buffers may be "spiked" with 1 mL 1 N KCl solution/ 100 mL sample to increase conductivity and improve response if necessary, and then following the above procedure.
- In cases where measurements are made in the field, the above procedure should be followed as closely as possible, and the use of a portable mixer/stirrer is recommended; one should definitely avoid placing the electrode directly into the stream and avoid large temperature differences between electrode, buffers, and samples.


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# Methods to Identify and Measure Winter Habitat 

by

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## ABSTRACT

The methods used to describe and quantify the winter habitat of salmonid fishes are dependent on the prevailing environmental conditions (e.g. pond vs. stream, presence of ice, etc.), time-scale of the collections (day vs. night, early vs. late winter), the life-stage of the fish, and the winter behaviour of the particular species. A variety of methods (including diving, traps, mark-recapture experiments, electro-fishing, freeze-cores, radar), together with their specific advantages and disadvantages, are discussed.

## RÉSUME

Les méthodes utilisées pour décrire et quantifier l'habitat d'hiver des salmonidés dépendent des conditions environnementales (p. ex. étang ou cours d'eau, présence de glace. etc.), du moment des prélèvements (jour ou nuit, début ou fin de l'hiver), du stade du cycle vital du poisson et du comportement des espèces particulières durant l'hiver. On discute des différentes méthodes d'étude (plongée, pièges, expériences de marquage-recapture, pêche électrique, carottage-congélation, radar), et on expose les avantages et les inconvénients de chacune d'entre elles.

## INTRODUCTION:

The identification and measurement of winter habitat presupposes an adequate understanding of the habitat preferences of the species in question. Unfortunately, this is not the case at present. For example, scientific studies of the use of off-channel habitats, estuaries, and ponds for overwintering by salmonid fishes are relatively recent phenomena (e.g. Peterson 1982; Brown and Hartman 1988; J.B. Dempson, D.F.O., St. John's, pers. comm.). Further, the particular winter habitat chosen varies with local availability, complexity of the physical environment, the time of sampling (diel and seasonal scales), life-stage, and the species of fish (Cunjak and Power 1986 a,b; Cunjak 1988a). The present paper reviews the most common techniques for sampling and quantifying juvenile salmonids in the winter with descriptions of their advantages and disadvantages. Finally, a list of the more important physical characteristics of winter habitats of juvenile Atlantic salmon is included.

## TECHNIQUES:

Snorkelling/SCUBA - This is an excellent method for quantifying microhabitat utilization of juvenile salmonids in 'open' or ice-covered streams and ponds. It is limited by depth of water and ice thickness, substrate heterogeneity, and by experience of the observers (Cunjak et al. 1988; Shardlow et al. 1987). Because many salmonids are photonegative in winter (e.g. Atlantic salmon, Rimmer 1980) and prefer dark microhabitat locations (Bustard and Narver 1975; Rimmer et al. 1983; Cunjak 1988a), handheld lights and searches beneath substrate materials and instream cover are requisites for assessing habitat utilization and behaviour. It is less useful as a means of accurately quantifying population abundance in winter (Rimmer et al. 1983; Emmett and McElderry 1986), particularly for species which overwinter beneath rock shelters. Where SCUBA is required in ice-covered rivers, it can be time-consuming and costly, especially if the intention is to survey large sections of river (e.g. Emmett and McElderry 1986).

Electrofishing - The use of pulsed DC electricity is an effective method for sampling salmonids in small or shallow (<lm mean depth) streams (see review in Reynolds 1983). In winter, however, the low water temperature and the habitat shift by many fish species to deeper water can markedly reduce electrofishing efficiency (Gardiner 1984) especially when attempting to compare catches between seasons (Johnson et al. 1987). Also, the cryptic behaviour and movement beneath rock shelters by species such as Atlantic salmon (Rimmer et al. 1983; Cunjak 1988a) may influence catches in consecutive "sweeps" (Figure 1) while performing standard population estimations (Zippin 1956; Seber 1982). It appears that, compared with summer sampling, the second sweep during winter surveys was occasionally as high or higher than the first sweep in some Atlantic salmon rivers (Figure 1). This may be attributable to the species' substrate-hiding behaviour in winter which would make them less vulnerable to sampling. This problem was never realized for brook trout (Valleyfield River) which do not overwinter beneath rock shelters (Cunjak and Power 1986a).

Where extensive ice cover occurs, sampling by back-pack, or boatmounted, electrofisher is not practical. A recent innovation, a diver-held narrow-field electrofishing device (Emmett and McElderry 1986) may prove to be a satisfactory alternative in ice-covered rivers and ponds.

Traps and Nets - Baited minnow-traps are used extensively on the West Coast of Canada during winter for mark-recapture experiments (Brown 1987; Brown and Hartman 1988) and habitat utilization studies of small salmonids in streams, ponds, and side-channels (Emmett and McElderry 1986; Swales et al. 1988). They are more versatile and less cumbersome than fyke nets, and unlike gill nets, permit live-sampling. Their usefulness for sampling Atlantic salmon needs to be tested but could prove to be an excellent means of assessing diel activity patterns and instream distribution in winter. Their ability to capture fish is largely dependent on the presumption that the target species is an active forager in winter which, in the case of Atlantic salmon in Maritime rivers, has recently been ascertained (Cunjak 1988a). The main problem with their use would be one of anchoring in deep, fast rivers or of installation in ice-covered stream sections too shallow to accommodate their dimensions (i.e. $<35 \mathrm{~cm}$ water depth).

Fish-counting Fences - The basic type described by Anderson and McDonald (1978) are common in Atlantic salmon rivers for monitoring runs of juveniles andlor adults (e.g. Chadwick et al. 1985). The usefulness of these devices is precluded after initial freeze-up and/or where winter flows are very high. However, in streams where autumnal flows are moderate, delaying the removal of fences (or maintaining downstream trapping efficiency) until first iceformation would provide useful information on movements of young salmon to overwintering areas.

Ultrasonic Transmitters - This technique has proven to be useful for monitoring fish movements (Diana et al. 1977), metabolic rates, and habitat selection in rivers, ponds and estuaries based on following signals from implanted (or attached) transmitters. Disadvantages of the technique include cost of labour and materials, experience of personnel, and telemetry errors by triangulation (Nams 1989).

Suction Pump/Dredge - Belzile et al. (1982) used such a device in subarctic rivers of Ungava Bay to sample substrate and overwintering juvenile Atlantic salmon beneath ice. The method yielded few fish and was very disruptive but may be necessary in extreme environments where salmon may not be active in winter but instead become torpid for most of the season. Diverheld 'slurp guns' and suction devices have been used with varying degrees of success for sampling juvenile salmonids from beneath rocks and within crevices in winter (Emmett and McElderry 1986).

Freeze-cores - This technique (described by Walkotten 1976) provides a depth profile of sediment structure. It has been generally used for sampling redds to measure sediment composition (Everest et al. 1982) and, more recently, in relation to the location of salmonid eggs in egg-pockets (Chapman 1988). It could prove very useful for estimating temporal survival in redds during winter in conjunction with measures of permeability change (see Young et al. 1989). The greatest disadvantage of the freeze-core method is the bulkiness of the device for use. in the field and the volume of heavy materials retrieved for subsequent analyses. Wesche et al. (1989) have suggested an alternative method of fine sediment measurement in streams which appears to be simpler to use and more efficient but which requires further field testing to establish reliability of data collected.

Fish Toxins and Anesthetics - Rotenone has been used in an ice-covered Quebec stream to sample salmonids (Coleman and Power 1967) with reasonable success. The difficulty in retrieving dead fish from beneath stones or ice, however, make this an inefficient and undesirable technique. Anesthetics
(MS222 and metomidate) were tested by divers sampling fish from rock shelters in an ice-covered river in British Columbia (Emmett and McElderry 1986). The technique was promising but inefficient in capturing most of the fish seen due to escape behaviour and problems with applicator design.

Short-pulse Radar - This method has been used for measuring the volume of water and ice in ice-covered rivers and lakes in the Arctic (Arcone et al. 1989). It was developed for use by helicopter which makes it a costly venture (Arcone and Delaney 1987). It is most effective in rivers with relatively small streambed materials (D. Calkins, U.S. Department of the Army, Hanover, N.H., pers. comm.). Despite these limitations, it could be very useful in quantifying available habitat space for juvenile salmon in ice-covered rivers and how the volume changes over the winter months. This is potentially important as space is a critical habitat factor in winter (Chapman 1966).

## VARIATION IN HABITAT MEASUREMENTS AND IDENTIFICATION

A variety of factors can influence the types of habitat utilized by young Atlantic salmon during the winter, and therefore, their relative importance and quantification. Some of these are listed below. For the purposes of this summary, anthropogenic factors are not considered here.
(1) Diel variation in behaviour, attributed to a photonegative response by Atlantic salmon at low water temperatures (Rimmer 1980; Cunjak 1988a), indicates that shallow, littoral zones are utilized during darkness (and crepuscular hours ?).
(2) It is possible that there are variations in the types of habitat utilized in late versus early winter which may be related to activity and metabolic patterns during acclimatization to the 'direction' of temperature and photoperiod change in the two periods (e.g. Cunjak 1988b; Cunjak et al. 1987).
(3) Although generally considered to be substrate-hiders during winter, young salmon which overwinter in ponds (and estuaries?) may utilize different habitats. This needs to be investigated and if found to be the case, the environmental characteristics typical of these habitats need to be measured as they influence fish habitat.
(4) At the extremes of their distribution, the microhabitat requirements and behaviour of salmon may vary from that found in temperate and boreal environments. In sub-Arctic streams, for example, Atlantic salmon parr remain active during the day at water temperatures below $6^{\circ} \mathrm{C}$ (personal observation) whereas in more southerly climes, they seek rock shelters for overwintering as daytime temperatures in the autumn decline below $9-10^{\circ} \mathrm{C}$ (Rimmer et al. 1983; Gibson 1978).
(5) Finally, research in Wyoming trout streams in winter has shown that physical habitat features (e.g. ice formation, snow depth, discharge) varied with elevation (Chisholm et al. 1987). The importance of this factor (and probably numerous others) in relation to winter habitat of salmon is yet to be determined.

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FIGURE 1. Total catches of salmonid fishes by successive electrofishing sweeps in streams during summer/autumn (top panel) and winter (bottom panel). Valleyfield River data are for brook trout only; Betts Mill and Catamaran Brooks (Atlantic salmon); River Philip (Atlantic salmon and brown trout).

# Remote Surveying for Derivation of Stream Habitat for Atlantic Salmon 

## by

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## ABSTRACT

Methodology to collect Atlantic salmon stream habitat from aerial photography and orthophotographic maps is presented. Remote measured data are compared to proximate measured data and formulae to convert remote measured area and gradient of streams to proximate measured gradient and area are presented. Water surface area is adjusted to the mean summer discharge and presented in a gradient and distance stratified matrix.

## résumé

On présente une méthodologie pour la collecte de données sur l'habitat lotique du samon de l'Atlantique au moyen de la photographie aérienne et des cartes orthophotographiques. On compare les données mesurées à distance aux données mesurées au sol et on présente des formules pour convertir la superficie et la pente des cours d'eau mesurées à distance en valeurs correspondantes mesurées au sol. La superficie d'eau est ajustée au débit estival moyen et présentée sous forme d'une matrice stratifiée de valeurs de pente et de distance.

## INTRODUCTION

Production of Atlantic salmon (Salmo salar) is dependant on the number of eggs deposited (Paloheimo and Elson 1974, Watt and Penny 1980, Chadwick 1982), the quantity of habitat (Elson 1957, 1975), and growth rate. Growth rate has been shown to vary with a range of environmental factors such as chemical water quality (LeCren 1972), temperature (Allen 1941), and photoperiod (Higgins and Talbot 1985) or the interaction of these variables e.g. photoperiod and temperature (Metcalfe and Thorpe 1990). Carrying capacity for any stream to produce smolts is a function of these variables. Rearing area (habitat) accounted for $62 \%$ of the variation in mean sport catch in rivers of Newfoundland and Labrador and $86 \%$ of the variation in catches for Maritime rivers (Chadwick 1985). Therefore, assessments of Atlantic salmon stocks based on habitat area and optimum escapement parameters are dependant to a considerable degree on accurate surveys of complete river systems.

Atlantic salmon are distributed over a wide geographical range in North America from Ungava Bay in the north to the Conneticut River in the south. Proportionate distribution of physical attributes varies considerably within river systems and among river systems. Surveys of salmon producing rivers must discriminate habitat quality in order to account for variation in river topography. Because the range of Atlantic salmon is extensive, complete quantitative surveys of habitat are only economical through some form of remote sensing.

Distribution of juvenile Atlantic salmon within streams has been associated with a variety of physical attributes of streams such as water depth (Egglishaw and Shackley 1982), water velocity (Symons and Heland 1978) and substrate size (Morantz et al. 1987). The possibility that a remotely measured parameter such as stream gradient could account for a considerable amount of the variance in juvenile densities and distribution was postulated by Symons and Heland (1978).

Salmonid population distributions have been described as contagious (Bohlin et al. 1981) and stratified sampling with respect to biotope was suggested as one way to reduce the rather large (0.70) coefficient of variation between locations. Multi-stage sampling designs have also been suggested as one way of reducing sampling variance when attempting to estimate the standing population of fish in small streams (Hankin, 1984). Attempts to relate juvenile salmon densities to gradient were unsuccessful (Symonds and Helland 1978; Kennedy and Strange 1982). However, area weighted gradient of continuous ecological units explained $79 \%$ of the variation at eight locations on the Stewiacke River, N.S. (Amiro 1984).

Description of an entire river system according to a vector such as gradient could allow the derivation of more precise estimates of standing populations of juvenile salmon.

Techniques for surveying juvenile rearing. habitat of salmonids can be categorized as :1) continuous proximate surveys e.g.

Dunfield and Sweeney, (pers. comm. ${ }^{1}$, Amiro, 1984 ; 2) stratified systematic proximate surveys e.g. Herrington and Dunham 1967 ; Gray et al. 1989 ;3) low-level aerial observation e.g. Riche, 1972 ; 3) remote sensing of high resolution (1:6000) aerial photography e.g. Greentree and Aldrich, 1976, and low resolution (1:20,000) aerial photography e.g. Amiro, 1983. While continuous proximate surveys are the most comprehensive, they are time consuming, expensive and require complete access to the river and are usually reserved for smaller areas requiring intensive scrutinizing. Systematic or random stratified proximate surveys reduce the amount of field work, but still require considerable access to the river system.

High resolution aerial photography can provide excellent interpretation of habitat but is costly and dependant on subjective interpretation. Lower resolution photography, including 1:10,000 color aerial photography is more generally available and adequate to provide reasonable estimates of water surface area, but inadequate for habitat interpretation (Amiro unpub. data).

The collection of systematic estimates of habitat area and gradient could be an important step in the rationalization of Atlantic salmon assessment methodology and perhaps contributory to the long-term large-scale monitering of total salmon production potential. This paper provides methodology for the derivation of fluvial water surface area and gradient of remote measured reaches delineated from orthophotographic maps. Lengths are measured from orthophotographic maps and widths from aerial photographs. Areas of streams, adjusted to a mean summer low width, are summerized for complete river systems.

## MATERIALS AND METHODS

## Remote Surveys

Numerical longitudinal profile descriptions of each stream of the Stewiacke River, Nova Scotia, (Fig. 1) were derived from digital measurement of stream lengths between 5 -meter contour intervals on 1:10,000 orthophotographic maps (L.R.I.S., 1978, from 1973 photography). ${ }^{2}$ Water surface area for stream lengths between contour lines crossing the streams (remote reaches) were calculated as the product of the stream length and mean width as measured from 1:10,000 color aerial photographs.

Sampling intervals for width measurments were generated on data forms according to an algorithm which required a minimum of two

[^1]widths within remote reaches less than 50 m (using the first measurement of the next reach for calculating an average), or width measurments approximately every 50 meters for remote reaches up to 1.6 km (maximum of 32 measurements). The interval for sampling reaches longer than 1.6 km was increased by 10 m until 32 or fewer samples were required. Exact distance between width samples was rounded according to the integer value of the length of the remote reach divided by the est.imated number of sampling intervals. Widths were measured using a $6 x$ monocular comparator equipped with a $10^{-1} \mathrm{~mm}$ divisional line reticle.

Remote reaches were located on air photos by comparing attributes on the orthophotographic map using stereo scopes when necessary. Remote reaches were sampled continuously upstream until the ortho-grade exceeded 15\% unless bridged by a fishway. Where aerial viewing of a stream was obstructed by overhanging cover or shadow, width measurements were attempted for the next four reaches and if no further measurements were possible the survey was terminated for that stream. Average widths were interpolated where widths could not be measured for three or less reaches.

When the first four remote reaches of a stream were not viewable on the photos and the stream consisted of many reaches and confluences, a search for a width window was made upstream on the photo set. If a window was observed and the stream width was resolvable then the survey was continued and widths for the non-viewable portion of the stream were extrapolated to downstream reaches.

Width measurements taken from photographs were converted to meters using the exact scale factor calculated for each photo according to the flying height, focal length and ground elevation (value of the lower remote reach contour).

Mean widths for reaches were adjusted to a standard summer low discharge. Daily water gauging data over as many years as available were used to calculate the Average Daily Flow (ADF, cubic meters second ${ }^{-1}$ ) and the mean summer low (MSL) for the months July, August and September. A discharge correction factor was based on a regression between log \% change in mean top width (based on $100 \%$ ADF) on $\log \% \mathrm{ADF}$ for two tributaries of the North Platte River, Wyoming (data in Wesche 1973). The resulting equation $(\log Y=8.6897 * \log X-$ 15.4158, $\mathrm{r} 2=.9035$; $\mathrm{p}=0.001$ ), which excluded values $<12.5 \%$ of ADF because these values are below most MSL flows found on unregulated streams, was used to calculate the percent change in top width for the estimated MSL as well as the percentage change in top width for the date of the photo. The ratio between the \% of ADF top width for MSL and \% of MSL top width for the photo day was the correction factor.

## Proximate Surveys

Locations for proximate surveys were randomly selected from remote reaches stratified by ortho-grade intervals summarized by 0.0$0.12 \%$ grade and by $0.25 \%$ stepped intervals from $0.12 \%$ to $5.0 \%$ orthograde and by 10 km distance intervals from the head of tidal influence.

Selection of proximate locations was weighted by the proportion of the total water surface area represented by a distance and gradient cell.

Proximate stream surveys for comparison to areas derived by remote survey, were conducted at remote reaches located in the feild using ortho-photo maps and color aerial photographs.. Surveying proceded with regard to ecological unit types, termed proximate reaches (Fig. 2) (Amiro 1984). Proximate reaches had similar surface, bottom and width characteristics, and were width sampled at the beginning, end and every 30 m or midpoint of the reach length. Widths and lengths were measured (to $10^{-2} \mathrm{~m}$ ) with a fiber measuring tape. Depths (to $10^{-3} \mathrm{~m}$ ) were measured with a survey rod at one quarter intervals across width transacts. Total area for remote reaches was the sum of all proximate reaches calculated as the product of the average width for the beginning, every 30 m and ending point of each proximate reach and the proximate reach length.

The water surface grade of each proximate reach was determined from distance between points at the mid-stream depth locations using an engineering auto-level and standard levelling techniques. Gradients of each remote reach and potential electrofishing site (a combination of consecutive proximate reaches) were weighted according to the area of each contributing proximate reach and termed area-weighted-percent-surface-grade (AWSG). Areas and AWSG's for use in population estimates were calculated by omitting proximate reaches with grades greater than $5 \%$, i.e. white water chutes and falls.

Remote reaches greater than about $1,000 \mathrm{~m}$ were sub-sampled to obtain a surface grade and AWSG for comparison to the remote orthophotographic grade. Samples were randomly selected from the set of proximate reaches making up the remote reach.

## RESULTS

## Stream Areas

Water surface area available for Atlantic salmon juvenile production was estimated at $26,762 * 10^{2} \mathrm{~m}^{2}$ for the Stewiacke River (Table 1). Areas adjusted to a standard summer low discharge were collated and reported by 10 km distance intervals (above tide head) and by 0.12 and $0.25 \%$ stream gradient intervals. Area by gradient interval for individual reaches, tributaries and branches, while appearing combined in tabular format, remained unique for each reach in the computer files, enabling calculation at the reach level, and summation by tributary or branch.

## Accuracy of Area Measurments

Aerial photographic measurement of ortho-photo defined remote reaches could not be directly measured for all reaches because
of shadows on the photos and/or overhanging cover. Widths for these remote reaches were inferred (where possible) from the closest fully visable reaches. When three or more consecutive non-measurable reaches were encountered, the air-photo survey for that stream was terminated. Hence only 12 remote reaches with complete aerial and ground surveys were available for comparison (Table 2). Regression analysis of these data (Fig 3) indicated a highly significant relationship accounting for $97 \%$ of the variance. The slope of the equation;

$$
\text { Photo area }=0.87 * \text { Ground area }-143.39
$$

together with the negative intercept indicates that areas are underestimated.

## Accuracy of Gradient Measurments

Proximate surveyed gradients (sine-1 converted square root of proportion i.e. ASAWSG) for 55 remote reaches on the Stewiacke River were compared to $\log _{\mathrm{e}}$ transformed (sine-1 converted square root of proportion i.e. LnASORT) values determined from the orthophotographic maps. Analysis indicated a statistically significant relationship with uniformly distributed residuals accounting for $72 \%$ of the observed variance (Fig.4, dottedline). However, 8 of these observations were the result of sub-sampling of larger remote reaches and when excluded reduced the coefficient of variation to $46 \%$. Regression of proximate grades for the 10 sub-sampled larger remote reaches and the corresponding orthogrades accounted for $76 \%$ of the observed variation. These reaches were also the flatter and therefore more accurately determined gradients, both remotely and proximately. Regression of ASAWSG on LnASORT weighted by proximate reach area (Fig.4, dotted line) resulted in the equation;

Area weighted surface grade (ASAWSG)=2.96 * Orthograde(LnASORT) $+0.238$
and accounted for $89 \%$ of the observed variation in area weighted proximate grades. Utilization of this equation to estimate proximate grades from remote measured orthogrades implies that X values (orthogrades) must be greater than $0.049 \%$ in order to derive rational $Y$ values (area-weighted surface-grades).

## DISCUSSION

## Remote Surveying

Comparison of the remote and proximate estimates of areas of reaches indicated a bias toward underestimation of remote measured reaches. This bias however is small, and is less for larger streams and greater for smaller tributaries. The impact of the exclusion of over-hang bank area is of minimal concern to estimates of juvenile Atlantic salmon habitat ir that juvenile Atlantic salmon are principally a photo-positive

Where overhanging canopy completely blocks aerial view of the stream, habitats are more favorable for the cooler water species such as brook trout (Salvelinus fontinalis). The exclusion of some reaches due to overhang or shadow conservatively estimates the production area for Atlantic salmon.

The biased nature and magnitude of the error associated with the remote estimation of area-weighted-percent-surface grade (AWSG), while adjustable through regression, is a limitation to the resolution and perhaps the accuracy of the technique. Correction of this bias and reduction of this error involves deriving estimates of water surface grades for each ecological unit. While photo-interpretive techniques are avaiable to do this, the benefits of increasing resolution would have to be weighed against the cost and the requirememt for higher resolution estimates. Proximate surveys, as conducted in this study, may be used for site-specific areas eliminating the remote graderesolution error at costs associated with $3-4$ person-days $\mathrm{km}^{-1}$ of stream.

The gradient conversion formula requires that orthophoto measured grade be greater than $0.105 \%$ in order to derive rational values for ASAWSG; used by Amiro (1984) to relate stream gradient to juvenile salmon density. Thus very low gradient areas would be defaulted to production estimates derived at zero grade. However, if significant production of juvenile Atlantic salmon is determined from gradients below this value, then modification of the conversion formula or the gradient to density relationship would require examination.

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Table 1. Area ( $m^{\wedge} 2^{\star 100)}$ by percent orthogradient and distance above the 10-m contour for the Stewiacke River.

| Orthogradient interval |  |  |  |  |  |  |  |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Dist. interval <br> (km) | 0-0.12 | 0.121-0.249 | 0.25-0.49 | 0.5-.99 | 1-1.49 | 1.5-1.99 | 2-2.49 | 2.5-2.9 | 3-3.49 | 3.5-5.0 | >5.0 | Row <br> totals | Row <br> percents <br> of total <br> area |
| 00-09.999 | 3,000 | 116 | 142 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3,258 | 12.2 |
| 10-19.999 | 3,238 | 668 | 63 | 29 | 6 | 0 | 0 | 1 | 0 | 0 | 0 | 4,007 | 15.0 |
| 20-29.999 | 2,824 | 86 | 244 | 419 | 21 | 16 | 2 | 3 | 0 | 0 | 0 | 3,616 | 13.5 |
| 30-39.999 | 2,865 | 0 | 264 | 732 | 157 | 29 | 0 | 0 | 0 | 0 | 0 | 4,047 | 15.1 |
| 40-49.999 | 1,972 | 1,557 | 411 | 175 | 33 | 8 | 6 | 0 | 5 | 2 | 0 | 4,170 | 15.6 |
| 50-59.999 | 28 | 1,480 | 1,390 | 702 | 288 | 55 | 33 | 7 | 4 | 6 | 0 | 3,994 | 14.9 |
| 60-69.999 | 0 | 0 | 1,046 | 1,092 | 523 | 166 | 88 | 21 | 20 | 7 | 5 | 2,968 | 11.1 |
| 70-79.999 | 0 | 0 | 80 | 267 | 265 | 91 | 0 | 0 | 0 | 0 | 0 | 703 | 2.6 |
| Column totals | 13,928 | 3,907 | 3,640 | 3,416 | 1,293 | 366 | 130 | 32 | 29 | 16 | 5 | 26,762 | 100.0 |
| Colum <br> percents of total area | 52.0 | 14.6 | 13.6 | 12.8 | 4.8 | 1.9 | 0.5 | 0.1 | 0.1 | 0.1 | $<0.1$ |  |  |

Table 2 . Description of sites within the 34 locations surveyed in the Stewiacke River, 1984-1989, by proximate-measured and orthophotographic map-measured attributes.

| Location .site | Total area ( $\mathrm{m}^{\wedge} 2$ ) | $\begin{gathered} \text { Area } \\ \text { < } 5.0 \% \\ \text { grade } \\ \left(m^{\wedge} 2\right) \end{gathered}$ | Surface grade (\%) | Area wt'd surf.grade (\%) 9 . | Area wt'd surf.grade (\%) $h$. | Ortho grade (\%) | Distance at start (km) | Site length (m) | Ortho area (m^2) |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| 1.1 | 320 | 304 | 1.02 | 0.94 | 0.94 | - | 71.258 | 107 | c. |
| 1.2 | 234 | 234 | 1.10 | 1.43 | 1.43 | - | 71.365 | 104 | c. |
| 1.1+2 | 554 | 538 | 1.06 | 1.15 | 1.15 | 0.62 |  | 211 | $c$. |
| 2.1 | 2,404 | 2,404 | 0.86 | 0.88 | 0.88 | - | 62.238 | 103 | c. |
| 2.2 | 2,833 | 2,833 | 0.33 | 0.34 | 0.34 | - | 62.341 | 97 | c. |
| 2.1+2 | 5,237 | 5,237 | 0.61 | 0.59 | 0.59 | 0.41 |  | 200 | c. |
| 3.1 | 1,181 | 1,181 | 1.46 | 1.35 | 1.35 | 4.07 | 67.153 | 120 | d. |
| 3.2 | 1,320 | 1,320 | 2.66 | 2.58 | 2.58 | 3.73 | 67.276 | 151 | d. |
| 3.3 | 1,206 | 1,206 | 3.35 | 3.18 | 3.18 | 6.67 | 67.410 | 100 | d. |
| 4.5 | 633 | 633 | 1.08 | 1.03 | 1.03 | - | 58.410 | 107 | c. |
| 4.6 | 878 | 878 | 0.26 | 0.28 | 0.28 | - | 58.517 | 111 | c |
| 4.7 | 878 | 878 | 0.63 | 0.62 | 0.62 | - | 58.628 | 105 | c. |
| 4. $5+6+7$ | 2,389 | 2,389 | 0.65 | 0.60 | 0.60 | 0.52 |  | 323 | c |
| 4.1-9 | 6,703 | 6,703 | 0.85 | 0.86 | 0.86 | 0.52 | 57.989 | 954 | 5,330 |
| 5.2 (above | 1,263 | 1,263 | 1. 60 | 1.61 | 1. 61 | 3.18 | 60.325 | 144 | 1,575 |
| 5.3 falls) | 2,674 | 2,365 | 1.74 | 1.46 | 2.20 | 1.82 | 60.482 | 284 | 2,551 |
| 6.2 | 1,027 | 1,027 | 0.60 | 0.59 | 0.59 | - | 53.526 | 95 | c. |
| 6.4 | 1,130 | 1,130 | 0.41 | 0.50 | 0.50 | - | 53.351 | 102 | $c$. |
| 6.8 | 1,260 | 1,260 | 0.30 | 0.32 | 0.32 | - | 52.932 | 98 | c. |
| $6.2+4+8$ | 3,417 | 3,417 | 0.19 | 0.46 | 0.46 | 0.41 |  | 295 | c. |
| 6.1-12 | 12,651 | 12,651 | 0.42 | 0.45 | 0.45 | 0.41 | 52.390 | 1,196 | 10,664 |
| 7.1 | 1,432 | 1,432 | 1.33 | 1.39 | 1.39 | 1.66 | 68.022 | 309 | d. |
| 7.2 | 1,692 | 1,692 | 1.69 | 1.71 | 1.71 | 1.67 | 68.331 | 301 | d. |
| 8.1 | 1,042 | 984 | 2.10 | 1.74 | 2.16 | 3.27 | 55.200 | 186 | 398 |
| 8.2 | 963 | 761 | 3.31 | 1.57 | 3.78 | 2.90 | 55.353 | 170 | 450 |
| 9.1 | 592 | 485 | 3.31 | 2.39 | 3.19 | 4.04 | 65.266 | 121 | 344 |
| 9.2 | 528 | 452 | 3.17 | 2.56 | 3.24 | 4.86 | 65.390 | 100 | 286 |
| 9.3 | 395 | 395 | 2.50 | 2.52 | 2.52 | 5.05 | 65.493 | 100 | 275 |
| 10.1 | 959 | 959 | 1.97 | 2.00 | 2.00 | 1.96 | 58.606 | 202 | 250 |
| 10.2 | 703 | 703 | 2.45 | 2.65 | 2.65 | 2.85 | 58.781 | 170 | d. |
| 10.3 | 393 | 393 | 2.59 | 2.66 | 2.66 | 3.00 | 58.948 | 123 | d. |
| 10.4 | 357 | 357 | 2.85 | 2.82 | 2.82 | 4.21 | 59.067 | 112 | d. |
| 10.5 | 348 | 348 | 2.53 | 2.72 | 2.72 | 4.53 | 59.177 | 103 | d. |
| 10.6 | 329 | 329 | 3.33 | 3.25 | 3.25 | 5.30 | 59.272 | 96 | d. |
| 10.7 | 648 | 648 | 3.43 | 3.28 | 3.28 | 5.34 | 59.365 | 152 | d. |
| 11.1 | 764 | 764 | 1.91 | 1.98 | 1.98 | 3.40 | 61.403 | 151 | d. |
| 11.2 | 1,013 | 1,013 | 1.97 | 1.98 | 1.98 | 2.20 | 61.550 | 219 | d. |
| 11.3 | 594 | 522 | 3.76 | 2.18 | 4.25 | 3.23 | 61.777 | 147 | d. |
| 12.1 | 336 | 336 | 1.47 | 1.47 | 1.47 | 5.26 | 30.130 | 93 | d. |
| 12.2 | 407 | 407 | 1.13 | 1.14 | 1.14 | - | 30.225 | 103 | c. |
| 12.3 | 327 | 327 | 1.64 | 1.60 | 1.60 | - | 30.312 | 89 | c. |
| 12.2+3 | 734 | 734 | 1.36 | 1.34 | 1.34 | 2.76 |  | 192 | d. |
| 13.1 | 286 | 286 | 1.53 | 1.70 | 1.70 | 7.75 | 71.077 | 63 | d. |
| 13.2 | 871 | 871 | 1.75 | 1.75 | 1.75 | 2.37 | 70.943 | 210 | d. |
| 13.3 | 749 | 745 | 1.69 | 1.46 | 1.52 | 2.67 | 70.732 | 182 | d. |
| 14.1 | 981 | 981 | 0.81 | 0.80 | 0.80 | - | 50.591 | 141 | c. |
| 14.2 | 918 | 918 | 0.71 | 0.72 | 0.72 | - | 50.732 | 138 | $c$. |
| 14.3 | 965 | 965 | 0.90 | 1.03 | 1.03 | - | 50.871 | 160 | c. |
| $14.1+2+3$ | 2,864 | 2,864 | 0.81 | 0.85 | 0.85 | 1.15 |  | 439 |  |
| 15.1 | 570 | 567 | 2.23 | 2.08 | 2.22 | - | 56.603 | 123 | C. |
| 15.2 | 493 | 388 | 2.79 | 0.85 | 2.84 | - | 56.726 | 122 | $c$. |
| 15.3 | 414 | 379 | 1.48 | 1.01 | 1.51 | - | 56.848 | 119 |  |
| 15.1+2+3 | 1,477 | 1,334 | 2.17 | 1.42 | 2.23 | 1.38 |  | 364 | 842 |
| 16 | 553 | 505 | 1.53 | 0.57 | 1.72 | 3.16 | 46.048 | 160 | d. |
| 17.1 | 283 | 283 | 1.51 | 1.53 | 1.53 | 7.85 | 56.178 | 61 | e. |
| 17.2 | 192 | 192 | 0.94 | 0.81 | 0.81 | 14.40 | 56.115 | 33 | e. |
| 17.3 | 457 | 457 | 1.29 | 1.28 | 1.28 | 7.17 | 56.080 | 71 | e. |
| 18.1 | 380 | 380 | 0.62 | 0.70 | 0.70 | - | 28.635 | 97 | c. |
| 18.2 | 368 | 368 | 0.98 | 1.08 | 1.08 | - | 28.732 | 90 | c. |
| 18.1+2 | 748 | 748 | 0.79 | 0.89 | 0.89 | 2.67 |  | 187 | 333 |

Table 2. Cont, d 337

| Location .site | Total area ( $\mathrm{m}^{\wedge} 2$ ) | $\begin{gathered} \text { Area } \\ <5.0 \% \\ \text { grade } \\ \left(\mathrm{m}^{\wedge} 2\right) \end{gathered}$ | Surface grade (\%) | Area wt'd surf.grade (\%) 9 . | Area wt'd surf.grade (\%) $h$. | Ortho grade (\%) | Distance at start (km) | Site length (m) | Ortho area ( $\mathrm{m}^{\wedge} 2$ ) |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| 19.1 | 327 | 273 | 3.85 | 1.25 | 3.78 | 5.51 | 37.436 | 89 | d. |
| 19.2 | 210 | 210 | 2.71 | 2.59 | 2.59 | - | 37.527 | 58 | c,e. |
| $19.1+2$ | 537 | 483 | 3.40 | 1.83 | 3.31 | - |  | 147 | c, e. |
| 20.2 | 1,583 | 1,583 | 0.15 | 0.21 | 0.21 | - | 27.925 | 115 | c. |
| 20.6 | 1,167 | 1,167 | 0.00 | 0.16 | 0.16 | - | 28.400 | 105 | c. |
| 20.9 | 1,160 | 1,160 | 0.61 | 0.63 | 0.63 | - | 28.751 | 109 | c. |
| 20.10 | 2,142 | 2,142 | 0.84 | 0.85 | 0.85 | - | 28.860 | 148 | c. |
| $20.2+6+9$ | 3,910 | 3,910 | 0.20 | 0.32 | 0.32 | 0.42 b . |  | 329 | $c$. |
| 20.1-10 | 11,152 | 11,152 | 0.33 | 0.40 | 0.40 | 0.42 | 27.808 | 1,200 | 11,056 |
| 21.1 | 401 | 398 | 1.46 | 1.31 | 1.44 | - | 71.925 | 128 | $c$. |
| 21.2 | 278 | 260 | 1.43 | 1.09 | 1.42 | - | 72.053 | 90 | c. |
| 21.3 | 214 | 209 | 1.56 | 1.42 | 1.57 | - 6 | 72.143 | 91 | c. |
| $21.1+2+3$ | 893 | 867 | 1.48 | 1.27 | 1.47 | 1.62 |  | 309 | 440 |
| 21.4 | 747 | 716 | 1.63 | 1.41 | 1.64 | 3.02 | 72.233 | 168 | d. |
| 22.1 | 455 | 439 | 1.07 | 0.84 | 1.12 | - | 59.624 | 120 | $c$. |
| 22.2 | 409 | 370 | 1.66 | 1.20 | 1.61 | - | 59.744 | 119 | c. |
| 22.1+2 | 865 | 809 | 1.36 | 1.01 | 1.35 | 2.16 |  | 239 | 667 |
| 23.0 | 1,799 | 1,799 | 0.30 | 0.26 | 0.26 | 0.82 b . | 69.306 | 128 | c. |
| 24.0 | 6,874 | 6,874 | 0.00 | 0.00 | 0.00 | 0.02 b . | 17.299 | 250 | c. |
| 25.0 | 19,900 | 19,900 | 0.00 | 0.00 | 0.00 | 0.02 b . | . 14.387 | 750 | C. |
| 26.1 | 282 | 181 | 3.08 | 1.65 | 3.34 | 6.19 | 60.982 | 81 | d. |
| 26.2 | 429 | 334 | 3.09 | 2.05 | 3.08 | 3.90 | 61.063 | 127 | d. |
| 27.1 | 1,531 | 1,531 | 0.21 | 0.22 | 0.22 | - | 7.934 | 113 | c. |
| 27.4 | 1,251 | 1,251 | 0.32 | 0.43 | 0.43 | - | 8.312 | 97 | c. |
| 27.1-9 | 13,406 | 13,406 | 0.29 | 0.33 | 0.33 | 0.49 |  | 1,385 | 10,812 |
| 28.1 | 412 | 408 | 0.56 | 0.48 | 0.52 | - | 13.440 | 106 | c. |
| 28.3 | 423 | 418 | 0.73 | 0.63 | 0.71 | - | 13.656 | 106 | c |
| 28.1+3 | 835 | 827 | 0.37 | 0.56 | 0.62 | - -76 |  | 212 | c. |
| 28.1-7 | 2,986 | 2,962 | 0.63 | 0.64 | 0.68 | 0.76 |  | 656 | 1,279 |
| 29.1 | 450 | 450 | 1.17 | 1.25 | 1.25 | - | 23.414 | 114 | d. |
| 29.2 | 447 | 447 | 1.46 | 1.46 | 1.46 | - | 23.527 | 111 | d. |
| 29.3 | 398 | 398 | 1.99 | 1.99 | 1.99 | - | 23.639 | 97 | d. |
| $29.1+2+3$ | 1,295 | 1,295 | 1.52 | 1.55 | 1.55 | 1.22 |  | 322 | d. |
| 30.1 | 904 | 904 | 0.71 | 0.72 | 0.72 | - | 31.317 | 105 | c. |
| 30.2 | 1,009 | 1,009 | 0.56 | 0.50 | 0.50 | - | 31.422 | 116 | c. |
| $30.1+2$ | 1,913 | 1,913 | 0.63 | 0.61 | 0.61 | - |  | 221 | c. |
| 30.3 | 562 | 562 | 0.96 | 0.97 | 0.97 | - 52 | 31.538 | 69 | $c$. |
| $30.1+2+3$ | 2,475 | 2,475 | 0.71 | 0.69 | 0.69 | 1.52 |  | 290 | 2,949 |
| 31.1 | 985 | 985 | 0.00 | 0.05 | 0.05 | - | 51.242 | 100 | c. |
| 31.2 | 858 | 858 | 0.38 | 0.39 | 0.39 | - 12 | 51.324 | 100 | c. |
| $31.1+2$ | 1,843 | 1,843 | 0.13 | 0.21 | 0.21 | 0.12 b . |  | 200 | c. |
| 32.0 | 415 | 415 | 1.04 | 0.98 | 0.98 | 4.29 | 47.615 | 131 | 229 a. |
| 33.1 | 831 | 791 | 0.96 | 0.68 | 0.95 | - | 67.245 | 121 | c. |
| 33.2 | 1,112 | 1,112 | 0.58 | 0.58 | 0.58 | -74 | 67.366 | 116 | c. |
| $33.1+2$ | 1,943 | 1,902 | 0.42 | 0.62 | 0.62 | 0.74 b . |  | 237 | c. |
| 34.1 | 690 | 690 | 1.16 | 1.08 | 1.08 | - | 69.987 | 139 | c. |
| 34.2 | 1,088 | 1,088 | 1.36 | 1.36 | 1.36 | - | 70.126 | 173 | c. |
| 34.3 | . 680 | 680 | 1.09 | 0.98 | 0.98 | - | 70.299 | 139 | $c$. |
| $34 \cdot 1+2+3$ | 2,459 | 2,459 | 1.22 | 1.19 | 1.19 | 1.29 |  | 451 | 1,928 a. |
| 34.4 | 674 | 643 | 1.43 | 1.07 | 1.07 | - | 69.987 | 113 | c. |
| 34.5 | 578 | 565 | 1.49 | 1.29 | 1.29 | - | 70.126 | 149 | c. |
| 34.6 | 695 | 695 | 0.96 | 1.29 | 1.29 | - | 70.299 | 134 | $c$. |
| $34 \cdot 4+5+6$ | 1,947 | 1,903 | 1.29 | 1.21 | 1.21 | 1.29 |  | 396 | 1,928 a. |

[^2]

Fig. 1.Map of the Stewiacke River, Nova Scotia showing the 34 electrofishing locations.


| SURFACE |  |  |  |  |  |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| SMOOTH | 90 | 10 | 100 |  | 40 | 90 | 40 | 10 | 10 | 90 |  |
| RIFFLE | 10 | 70 |  |  | 80 | 10 | 50 | 70 | 80 | 10 | 80 |
| ROUGH |  | 20 |  |  |  |  | 10 | 20 | 10 |  |  |
| BROKEN |  |  |  |  |  |  |  |  |  |  |  |
| BOTTOM |  |  |  | DAM |  |  |  |  |  |  |  |
| SAND | 70 | 35 | 40 |  | 50 |  |  |  |  |  |  |
| GRAVEL | 20 | 36 | 30 |  | 40 | 60 | 40 | 35 | 40 | 60 30 | 20 |
| COBBLE | 10 | 30 | 16 |  | 10 | 30 | 10 | 10 | 10 |  |  |
| BOULDER | 10 |  | 15 |  |  | 10 |  |  | 10 |  |  |
| LEDGE |  | 70 | 30 |  | 20 |  | 20 | 10 | 20 |  |  |

Fig. 2. Example of a proximate stream survey showing ecological unit areas, percent surface grades, lengths, surface and bottom composition of units. Survey technique is that of Amiro (1984) and Amiro et al. (1989).


Figure 3. Comparison of photo-estimated to proximately measured water surface areas for 12 locations in the Stewiacke River.


Figure 4. Relationship between proximately measured sine ${ }^{-1}$ converted area-weighted percent surface grade and log. sine ${ }^{-1}$ converted orthophotographic measured percent grades for 45 complete ( $)$ upper line and 10 incomplete ( ( ) lower line contour to contour reaches in the Stewiacke River. The dotted line was calculated using proximate reach area for case weighting and is the reported conversion fromula.

# Physico-Chemical Measurements in Stream Habitat Studies 

## by

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Almost all gravel or rocky-bed streams feature alternating erosional and depositional zones, generally expressed in alternating riffle-pool sequences. Downstream ends of pools (or crests of riffles) are depositional areas. Bed load is principally transported in saltatory fashion (usually at high water) from one depositional zone to another. Riffle areas are erosional zones, characterized by a surficial pavement or armour of larger sediments. Pools are generally stable areas. Salmon typically spawn in depositional zones.

Temperature is most influential as an influence on fish distributions in summer, and is directly correlated with stream size.

Stream major ions form four natural components (as determined by principal components analysis). One group is related to bicarbonate buffering chemistry, another to generation of organic acidity, and a third to sea-salt influence. The primary source of sulfate is the atmosphere, and it fits none of the three categories described above. If calcium concentrations exceed $2 \mathrm{mg} / \mathrm{L}$, the organic acids have little influence on pH , while at $\leq 1 \mathrm{mg} / \mathrm{L} \mathrm{Ca}$, they are the major influence on stream pH - at least under summer base flow conditions.

Dissolved trace metals (principally iron and aluminum) normally vary inversely with pH . If organic acids are high in concentration, they may complex the metals.

## Résumé

Presque tous les cours d'eau dont le lit est graveleux ou rocheux se caractérisent par une alternance de zones d'érosion et de sédimentation, qui se manifestent généralement par une alternance de fosses et de radiers. La partie aval des fosses, ou crête de radier, est une zone de sédimentation. Le charriage de fond s'effectue surtout par saltation, d'une zone de sédimentation à l'autre, en période de crue. Les radiers sont des zones d'érosion caractérisées par un pavé superficiel de dépóts de plus grande taille. En général, les fosses sont des zones stables où le saumon fraie habituellement dans la partie aval.

La température, directement corrélée à la taille du cours d'eau, joue un róle primordial dans la répartition du poisson en été.

Les principaux ions présents dans un cours d'eau forment quatre composantes principales telles que déterminées par l'analyse en composantes principales. Un groupe est lié à la capacité tampon du bicarbonate, un second, à la production de l'acidité organique et un troisième, à l'effet de l'eau salée. L'atmosphère est la principale source de sulfate; cette composante n'appartient à aucun des trois groupes précedents. Si la concentration de calcium dépasse $2 \mathrm{mg} / \mathrm{L}$, les acides organiques influent peu sur le pH , mais lorsqu'elle est égale ou inférieure à $1 \mathrm{mg} / \mathrm{L}$, les acides organiques influent considérablement sur le pH du cours d'eau, du moins dans les conditions de débit de base estival.

Habituellement, la concentration des oligoéléments dissous (surtout du fer et de l'aluminium) varie de façon inversément proportionnelle au pH. Si les acides organiques sont présents en concentrations élevées, ils peuvent complexer les métaux.

Most physical parameters of interest are related to either stream gradient or stream size. Streams are generally classified as "gravel"-bottomed, sand-bottomed or braided. Gravel in this case has a broad connotation and includes rocky substrates as well. Braided streams occur where bedload additions to the stream exceed the transport capacity, as is the case downstream of avalanches. They are characterized by channel instability, with new channels frequently being cut through the large gravel deposits. As an example, a section of the Digdeguash River is braided downstream of a stream bank gravel pit. Sand-bottom rivers do not exist for many of the smaller coastal watersheds. Such river segments are usually associated with meanders or ox-bow lakes. They are probably not of interest in the context of this workshop.

Rocky and gravel bottom streams form primary salmon habitat and are usually characterized by repeating riffle-pool sequences. The downstream side of a pool (or crest of a riffle) is a depositional zone with most bedload moving in saltatory fashion from one crest to another in periods of freshet. The main portion of the riffle is an erosional zone. In such areas, surface fines are absent for the most part, and an "armour" or "pavement" of coarser particles (clasts) forms on the surface. Pools are scoured by turbulent water currents, particularly during high water, and may be viewed as net erosional zones as well. In the absence of a bedload supply upstream, as when dams are constructed, depositional zones may be degraded with the pool-riffle sequence disappearing. In this case, a stable, impermeable armour layer will eventually form.

Particle size distributions are skewed so that a logarithmic transformation is used to define particle aggregates in streams. The "phi" scale is such a log scale, widely used to define characteristics of stream sediments. To define the sediment characteristics of a particular stream segment, one may superimpose a squared grid and measure the diameter of each particle under an intersection. This will work for particle sizes down to a millimetre or so. After several hundred such measurements, the geometric mean size may be converted to the phi scale.

Bottom roughness may be important in assessing cover for fish, and it is dependent upon sediment size and the extent to which particles project above the substrate "surface." Again, a grid of intersecting lines could be used with measurements of projection heights made at intersecting lines and converted to a phi basis. For large boulders, irregularities may be ecologically important. I have seen no publications dealing with this satisfactorily - perhaps fractal geometry could be adapted for the purpose.

Stream gradients, on a macro scale, may be determined from topographic maps. For more detailed measurements (several meters to several hundred meters), a hand level may be used over a measured length of stream.

The quality of stream sediments for spawning by salmon has also been given frequent consideration. The important criterion is the velocity of water flowing past the egg within the gravel interstices. This velocity is a function both of gravel permeability and of hydraulic head. While permeability is measured fairly readily and at most times of the year - with a permeability stand-pipe as designed by Terhune and Wickett in the 1950s, hydraulic head varies with stream discharge and local stream geometry. Thus, permeability measurement is usually more meaningful than measurement of sub-gravel water velocity. Permeabilities greater than 1000 $\mathrm{cm} / \mathrm{h}$ are usually required to ensure any survival of salmon eggs incubated in the substrate. Salmon typically spawn in stream depositional areas, such as at the downstream end of pools or on point bars. Sediments in such localities are in a state of dynamic equilibrium as described previously, without the formation of stable pavement surface layers. Enhanced permeability in these situations outweighs such hazards as sediment shifting during stream events or by ice scour.

Sediments (silts, clays) less than 100 microns are usually transported as suspended sediment, and are largely transported through the gravel-bed stream system during storm events as a result of runoff into the stream. In cases of extreme suspended sediment loads (e.g. "Yellow" River in China), such physical parameters as viscosity are influenced (i.e. the "fluid" can no longer be considered water).

Stream size-related physical parameters of importance include discharge, width, depth, velocity, temperature and shading.

Stream discharge can be measured by several methods, V-notched weir and depth-velocity integration being common ones. Stream discharge is a highly variable parameter, with episodic events resulting in a large percentage of the annual runoff occurring over a relative low number of days. These events are also responsible for much of the annual sediment transport. If knowledge of day-to-day fluctuations in stream flow is not required, then this may be obtained with reasonable accuracy by integrating the annual rainfall over the stream drainage area, subtracting about $15 \%$ to account for evapo-transpiration.

Width of the permanent stream channel is highly correlated with discharge the relation being linear for a log-log plot. Velocity and depth are extremely variable parameters, depending upon stream channel characteristics, discharge levels, and local variation in stream gradient. They are probably very important in studies of niche characteristics, but must be qualified with respect to the variations noted above. Velocity may be measured by various types of flowmeters - based on Pitot principles or on calibrated propeller meters (as with the "Ott"-type flowmeters). Mean velocities over distance may be measured by dye tracer or salt tracer methods.

Although shading usually is a function of stream-side vegetation, it is closely correlated with stream width for relatively unaltered systems in the Maritimes, at least for streams of 1 to 10 m width. For larger streams, shading is largely
confined to stream margins. A light meter is probably the standard method of measuring shading of streams. Since shading may be highly variable in middlesized streams, several transects should be averaged.

Temperature is another complex physical parameter. For intensive studies, in-stream recording thermometer systems should be employed. Thermal characteristics of possible importance, depending upon the nature of the study, include mean or median daily temperature, time-weighted annual mean temperature (or annual degree-days), diel temperature pulse, daily or annual thermal maxima or minima. At certain times of the year such as mid-summer, temperature may be a major influence on distributions of stream fishes. At other times of year (spring and fall), temperature differentials throughout a stream system may be less marked, and considerable within-stream movements may occur. In winter, discharge or depth may be more important than temperature. Under summer conditions, there is a net heat gain as water passes through a watershed, so that larger streams have higher mean temperatures than smaller ones. Local influences, such as in-stream aquifers, may introduce some localized temperature variability upon the general trend.
Streams which are in the process of drying up may have severe diurnal temperature fluctuations.

Although conductivity is a physical parameter, it is based upon aspects of the chemical composition of the water, i.e. the concentrations of dissolved ions. Its measurement is a rather standard procedure with many excellent conductivity meters on the market. Conductivity has been treated as a surrogate for productivity as, for example, in Ryder's morphoedaphic index. In Maritime waters, one has to be cautious of such usage because sea-salt influence - principally $\mathrm{Na}^{+}$and $\mathrm{Cl}^{-}$- may be the major determinant of conductivity. Hence, some Nova Scotian waters of fairly high conductivity are notably unproductive. pH may be a better surrogate than conductivity in such circumstances. Alkalinity is probably at least as good under all circumstances.

Dissolved gases represent another suite of physical variables. Oxygen and free carbon dioxide are usually the two of greatest importance in ecological work. Ammonia may be of importance in some special circumstances. All three can be measured with electrodes, using selectively permeable membranes. Calibration of these electrodes requires either equilibration of samples with known gas mixtures, or by chemical methods as with the Winkler method for oxygen. Water pH is an important factor to keep in mind when dealing with $\mathrm{CO}_{2}$ or $\mathrm{NH}_{3}$, as $\mathrm{CO}_{2}$ may be fixed (converted to $\mathrm{HCO}_{3}{ }^{-}$) at high $\mathrm{pH}\left(>5.5\right.$ ) or $\mathrm{NH}_{3}$ (converted to $\mathrm{NH}_{4}{ }^{+}$) at low pH (<6.0). Normally, $\mathrm{O}_{2}$ and $\mathrm{CO}_{2}$ are at near atmospheric saturation in natural waters, unless extremely high photosynthesis occurs, or there is a pollution problem. Subgravel $\mathrm{O}_{2}$ concentrations, however, may be somewhat lower than surficial concentrations, particularly if temperatures are fairly high.

Chemical parameters may be grouped into major ions, trace metals, organic molecules (mostly acids), and nutrients.

The major ions are present in most waters in $\mathrm{mg} / \mathrm{L}$ concentrations, and include $\mathrm{Na}^{+}, \mathrm{Ca}^{2+}, \mathrm{Mg}^{2+}$, and $\mathrm{K}^{+}$among the cations, with $\mathrm{Cl}^{-} \mathrm{SO}_{4}{ }^{2-}$, and $\mathrm{HCO}_{3}{ }^{-}$being the major anions (Table 1). Sodium and chloride are usually considered to be primarily derived from sea salt injected into the weather systems. Accordingly, their concentrations in fresh water is greater the nearer the coast the water is. This is probably more apparent for storage water bodies like lakes than for streams. High concentrations of $\mathrm{Na}^{+}$and $\mathrm{Cl}^{-}$in many well waters indicates some caution should be used in making sea salt corrections for waters derived largely from aquifers. Calcium, $\mathrm{Mg}^{2+}$ and $\mathrm{HCO}_{3}$ are thought to be derived mainly from soils and sediments, and contribute to the buffering capacity of natural waters. However, $\mathrm{Mg}^{2+}$, along with $\mathrm{SO}_{4}{ }^{2 \cdot}$ can have a major component derived from sea salt. Sulfate, by and large, is the ion of most constant concentration - at least in the southern Maritimes - with a large component derived from precipitation. All the major cations can be measured either by flame photometry, ion chromatography, or specific ion electrodes. One must keep in mind that the specific ion electrode measures only the free ion, while the other methods measure the total ion. In the case of $\mathrm{Ca}^{2+}$, $\mathrm{Mg}^{2+}$, and perhaps even $\mathrm{K}^{+}$, the difference between the two quantities can be significant if organic chelating agents are present in the water sample. Flame photometry is best for large numbers of measurements on a single ion, as it is faster. Ion chromatography will yield a spectrum of all ionic species present.

Among the anions, $\mathrm{HCO}_{3}$ is usually measured indirectly through alkalinity titration with a strong mineral acid ( HCl by choice). The results are plotted on a "Gran" plot and the equivalence point derived by extrapolation. Chloride may be measured either by chloridimetry or by ion chromatography, while ion chromatography is the method of choice for sulfate. The older, cobalt blue colorimetric method of sulfate method is adequate for water low in DOC ( $<5 \mathrm{mg} / \mathrm{L}$ ). In highly stained water, however, organic acids interfere, leading to overestimation of $\mathrm{SO}_{4}{ }^{2 .}$.

The most abundant trace metals in solution in natural waters are iron and aluminum, the concentrations normally being determined by pH and organic acids present. At low pH and high DOC levels, total dissolved concentrations of these metals can attain several hundred micrograms per litre. Much of this, however, may be complexed with organic acids if present. Total concentrations may be determined by the graphite furnace modification of atomic absorption spectrophotometry. Free ions are somewhat more difficult to determine - polarimetry or stripping electrode voltammetry may be applicable.

The major nutrients are phosphate ( $\mathrm{PO}_{4}{ }^{3-}$ ) and $\mathrm{NO}_{\mathrm{X}}$ (nitrate plus nitrite). These are usually present only in microgram amounts in unpolluted waters particularly in mid-summer when ecosystem metabolism is highest. At such times, these nutrients may be rate-limiting entities. Phosphate and nitrogen oxides are easily measured by ion chromatography, as well as older, colorimetric methods.

Dissolved organic molecules
carbon by IR oxidation and organic carbon analyated as total dissolved organic value can be converted to acectroscopy at a wavelength of 250 numces may also by 9 . $\quad$ actual equivalents with fair accuracy 250 nm . The DOC

Table 1. Varimax rotated correlations between principal components and chemical variables - Alk $=$ alkalinity, $\mathrm{H}=$ humic acids. Only significant correlations listed ( $\mathrm{p}<0.05$ ).

|  | 1 | Principal component 2 3 |  | 4 |
| :---: | :---: | :---: | :---: | :---: |
| Log Alk | 0.93 | - | - | - |
| Log K | 0.85 | - | - | - |
| Log Mg | 0.82 | - | - | - |
| Log Ca | 0.76 | - | - | - |
| pH | 0.71 | -0.57 | - | - |
| $\log \mathrm{H}$ | - | 0.91 | - | - |
| DOC | - | 0.83 | - | - |
| Al | -0.47 | 0.57 | - | - |
| $\log \mathrm{Na}$ | - | - | 0.94 | - |
| $\operatorname{Log~Cl}$ | - | - | 0.93 | - |
| $\underline{L o g ~ S O} 4$ | - | - |  | 0.88 |

## Session IV

Techniques to Measure Standing Stock

# Estimating the Abundance and Biomass of Juvenile Atlantic Salmon in Streams and Rivers of Eastern Canada 

by

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#### Abstract

2RSITRACT

In this paper, techniques used to estimate the abundance of juvenile Atlantic salmon in shallow streams and rivers using electrofishing gear are discussed. Within individual study sites, both the removal estimator and the petersen estimator (two of the most cammonly used procedures) provide reasonably precise estimates of salmon abundance. However, assumptions concerning the catchability of salmon must be tested. If catchability of salmon decreases with successive electrofishing samples, the removal estimator will be negatively biased. For the Petersen method, marking fish may affect their catchability, and the bias can be either negative or positive.


The spatial distribution of juvenile salmon within a stream is highly contagious. Therefore, to estimate the absolute abundance of salmon in a large area, large numbers of sites must be surveyed to provide estimates with reasonable precision. Efficiency of the field estimates may be inproved significantly if stratified randam survey procedures are used, if survey areas of unequal sizes are chosen, and if sampling is proportional to the survey areas. When the objective is to monitor changes in population abundance with time, an index of relative abundance, derived from sampling at fixed stations, may be useful.


#### Abstract

Rtsund Dans le present document, on traite des techniques d'estimation de l'abondance des saumons de l'Atlantique juvéniles par électropéche dans les rivieres et cours d'eau peu profonds. Dans chacun des sites étudiés, la méthode d'évaluation des retraits et celle de Petersen (deux des méthodes les plus courantes) fournissent des estimations assez précises de l'abondance du saumon. Toutefois, les postulats relatifs au potentiel de capture doivent étre vérifiés. S'il s'avère que le potentiel de capture de saumon diminue dans les prises successives d'échantillons à l'électropêche, l'estimation des retraits s'en trouve faussée à la négative. En ce qui concerne la méthode de Petersen, le marquage du poisson peut avoir des effets positifs ou négatifs sur le potentiel de capture.

La distribution spatiale des saumons juveniles dans un cours d'eau est très contagieuse. Par conséquent, pour obtenir une estimation de l'abondance absolue du saumon sur un vaste territoire, il faut étudier de nombreux sites afin d'obtenir des chiffres suffisamment précis. On peut améliorer considérablement les estimations réalisées sur place en utilisant un échantillonnage aléatoire stratifie, en sélectionnant des sites de grandeurs inégales et en veillant à ce que les échantillons soient proportionnels aux sites d'où ils proviennent. Dans les cas où on cherche à étudier les changements temporels dans l'abondance de population, un indice d'abondance relative, établi d'après un échantillonnage à des stations fixes, peut s'avérer utile.


## INIRODOCTION

Quantitative estimates of the abundance of juvenile Atlantic salmon in streams of eastern Canada began with P.F. Elson's work in the Pollett River in the 1940's (Elson 1962; Elson 1967). Parr densities were estimated with a fourperson team, where two people held a 15 foot seine, and two others 'assisted by splashing around and disturbing the bottom so as to scare and hold fish within reach of the seiners'. Electrofishing gear became available soon after (Smith and Elson 1950), and although the early generators were large and cumbersome to use, the efficiency of cencusing techniques improved appreciably. Since these early investigations, and with the development of more efficient and portable electrofishing gear, field estimates of juvenile salmon densities have become commomplace throughout Atlantic Canada.

Objectives for obtaining information on the abundance of juvenile salmon in fluvial habitats have been many. Field censuses have been used to identify the effect of pollutants from forest based industries (Elson 1967;1974), to evaluate enhancement techniques (Elson 1957a and b; 1962; 1975); to define predator-prey relationships (Elson 1962); to determine the suitability of different habitats (Symonds and Heland 1974; Marshall 1983; Amiro 1984; Gibson et al. 1987), the effect of acidification (Lacroix 1989), the effects on recruitment of populations under fishing stress (Paloheimo and Elson 1974; Randall and Chadwick 1986; Chadwick and Randall 1986); and finally to determine survival and production rates (Watt and Penney 1980; Randall and Paim 1982; Chadwick and Green 1985). In recent years, there has also been an increasing interest in the potential use of juvenile densities as an indicator of the status of Atlantic salmon populations.

Clearly there is a need for efficient and accurate methods to census juvenile salmonids in fluvial habitats. The objective of this report is to briefly review the current methods commonly used to determine the abundance of juvenile Atlantic salmon in streams and rivers of eastern Canada. Emphasis is placed on identifying the levels of precision that can be attained fram field surveys, and potential biases associated with the cencusing techniques. The discussion will be restricted primarily to techniques suitable for shallow (wadeable) waters.

## 1. Sampling gear

In past years, a variety of gear types have been used to determine the relative or absolute abundance of juvenile Atlantic salmon in streams (Table 1). Without question, however, electrofishing gear has become the most efficient and popular method of capturing and enumerating juvenile salmonids in streams. During electrofishing surveys, the capture probability of juvenile Atlantic salmon is usually high, averaging 0.4 to 0.5 , and thus large samples of fish can be collected and reasonably accurate estimates of population densities can be made (see below). Most of the following discussion refers to techniques which involve the use of electrofishing gear. Descriptions of the types electrofishing gear available, and the methodology for conducting field surveys in streams and other
habitats, are given by Reynolds (1983) and Cowx (1990).

## 2. Estimating the abundance of salmon within individual sites

### 2.1 Removal method

Background
The removal method of estimating population abundance is based on a series of 'removals' of fish from the survey area. Because fish are physically removed, each subsequent sample will result in fewer and fewer individuals being captured. The simplest form of the removal method is based on equal units of effort during each trapping period; the sequential catch data are then used to calculate maximum likelihood estimates of probability of capture ( $p$ ) and population size (N) (Moran 1951; Zippin (1956; 1958). Seber (1982; 1986) and White et al. (1982) describe the formulae and assumptions of the removal method. Application of the removal method to salmon data collected in Newfoundland and New Brunswick is described by Gibson et al. (1987) and Randall (1990), respectively.

The removal method is a closed population estimator which is based on the assumption that all individual fish have an equal and constant probability of capture on all removal or capture occasions (Table 2). Estimates are usually made during a short time (less than one day), and geographic and demographic closures are achieved by using barrier nets to delineate the survey area.

## Precision

Precision of the removal method is dependent on the proportion of the total population captured, which in turn in dependent on the capture probability of the juvenile salmon and the number of removals employed during the survey. For the removal method to be valid, the capture probability (p) should be at least 0.2, and for consistently reliable results, p should equal or exceed 0.4 (Seber 1982; White et al. 1982). For individual estimates of population size, Bohlin (1990) calculated the coefficients of variation (ratio of standard error to the population estimate, N ) for the removal estimator for different population sizes and $p$ of 0.4 and 0.5 , assuming four removals are carried out for each survey:

|  | Population size (N) |  |  |  |
| :---: | ---: | ---: | ---: | ---: |
| $\mathbf{p}$ | 50 | 100 | 200 | 400 |
| 0.4 | 10.5 | 7.4 | 5.3 | 3.7 |
| 0.5 | 5.6 | 3.9 | 2.8 | 2.0 |

For juvenile Atlantic salmon in the Miramichi and Restigouche rivers, probability of capture (p) has been shown to vary from <0.1 to 0.7 , but at most sites $p$ ranged between 0.3 and 0.6 , with means of 0.4 for age 0 salmon and 0.5
for age 1 salmon, respectively (Randall 1990). Because the catchability of juvenile salmon is reasonably high, a large proportion (usually >90\%) of the estimated total population is removed from the survey area after four removals. For population estimates within individual sites where the total catch equalled or exceeded 40, coefficients of variation averaged less than 0.10 for all age groups (Table 3). The capture probability of salmon during successive samples usually remained constant (Table 4), and thus the removal method was an appropriate estimator for most sites.

## Bias

The catchability of juvenile salmon during electrofishing surveys is sizedependent; smaller salmon characteristically have a significantly lower probability of capture than larger salmon (Randall 1990). The potential bias associated with size-dependent catchability can be eliminated by estimating the population size of each age group separately.

Assumptions of equal catchability of juvenile salmonids of comparable size, however, are also sometimes not valid during removal studies. Several researchers have noted that a proportion of salmon at any particular site have a low catchability (Bohlin and Sundstrom 1977; Peterson and Cederholm 1984) or that the catchability declines during successive removals (Cowx and Bohlin 1990). Inherent variation among individual fish in catchability, or gradual decreases in catchability over time are not always detectable from the capture data, and they can lead to consistent underestimates of fish population abundance (Fig. 1). Unequal catchability between the first and subsequent removals has sometimes been detected (example 2 in Table 5); when this is the case, there are models available which will correct this potential bias (White et al. 1982; Schute 1983; Cross and Stott 1975). In a recent review of methods for estimating animal abundance, Seber (1986) suggested that White et al. 's generalized removal method was the most appropriate model in most of these cases.

For most surveys, it is preferable to conduct at least three removal samples so that the assumption of equal catchability among sampling occasions can be tested. Two removals are advisable only if the probability of capture equals or exceeds 0.8 (White et al. 1982), but catchabilities of juvenile salmon are rarely this high during surveys in New Brunswick (Randall 1990).

### 2.2 Mark-recapture estimates

## Background

For estimating population size using a mark-recapture method, a sample of fish from a closed population is captured, marked and returned to the original population. At a later time, a second sample is captured, and the ratio of marked to unmarked fish in the sample is used to determine the population size at the time of marking. If only one tagging and one recapture sample is taken, a simple Petersen (or Petersen-Lincoln) estimate is made. As for the removal method, the Petersen estimator is appropriate if certain fundamental assumptions about
capture probabilities are met (Table 2). More complex models, involving several marking and recapture occasions, may also be used and allow relaxation of the above assumptions. Multiple mark-recapture methods will be discussed more fully below.

## Precision

Statistical precision of the Petersen and removal estimators are similar if three or more removals are made. Ratios of the coefficient of variation of the removal estimate to the Petersen estimate (from Seber 1982) are:

Probability of capture

| 0.1 | 0.2 | 0.3 | 0.4 | 0.5 |
| :--- | :--- | :--- | :--- | :--- |
| 4.36 | 3.00 | 2.39 | 2.00 | 1.73 |
| 2.14 | 1.42 | 1.09 | 0.88 | 0.73 |

Examples of the use of the Petersen method to estimate the abundance of juvenile Atlantic salmon in eastern Canada are given by Gibson et al. (1987) and Amiro et al. (1989).

If multiple tagging and recapture occasions are used (eg. the Schnabel estimator), the level of precision is greater than the Petersen estimator, although the difference is not great unless the probability of capture is low (Seber 1982: p. 568). The main advantage of a multiple tagging/recapture experiment is that assumptions of constant catchability of fish can be tested, and more appropriate models can be used if catchability is not constant (White et al. 1982).

## Bias

Most of the potential biases identified above for the removal method are also applicable to mark-recapture data. In addition, the Petersen method involves marking fish, and an important assumption is that both tagged and untagged fish have the same probability of being caught in the second sample (Table 2). Electrofishing and marking procedures may affect the catchability of fish (Bohlin and Sundstram 1977; Seber 1982; Peterson and Cederholm 1984; Mesa and Schreck 1989). If the more catchable fish tend to be marked, then the Petersen method will underestimate the population size (Cross and Stott 1975; Bohlin and Sundstrom 1977). On the other hand, if marking makes the fish 'capture shy' (Schreck et al. 1976; Mesa and Schreck 1989), marked fish will have a lower catchability, and estimates of $N$ will be biased high. For a single mark-and recapture study, unequal catchability of marked vs unmarked fish is difficult to detect, unless the recapture data can be stratified, for example by size or sex. If a Peterson/Lincoln estimator is going to be used, it may be advisable to catch and mark fish by a method other than electrofishing (Table 1) to reduce
the possibility of affecting the catchability of fish and introducing a bias. Alternatively, electrofishing could be used to provide a sample of fish for marking, and another type of gear to collect the recapture sample.

When a multiple recapture experiment is performed, the constancy of capture probabilities among samples can be tested (Seber 1982; White et al. 1982). If capture probabilities are shown to vary because of (i) behavioral (capture probability during the first sample is higher or lower than all subsequent captures), (ii) inherent heterogeneity (i.e., , probability of capture varies from fish to fish but remains constant from sample to sample) or (iii) temporal factors, appropriate models are available which will provide unbiased estimates of population size (White et al. 1982).

### 2.3 Comparison of removal and mark-recapture methods

The removal method is easy to conceptualize because of its simplicity, and the data needed are reasonably easy to obtain in one field day. The high catchability of juvenile salmon by electrofishing, particularly in small streams, makes it possible to estimate fish abundance with reasonable accuracy. Because fish do not have to be marked, the removal method involves less handling time of fish in the field, and problems associated with marking effects on catchability are avoided.

As noted above, a basic assumption of the removal method is that the probability of capture is constant among individual fish and among samples. Because this is often not the case, some statisticians (White et al. 1982) recommend the use of mark-recapture techniques over removal estimators. Unfortunately, marking of fish requires additional field effort, and it may affect their subsequent catchability. If a multiple mark-recapture procedure is used, several models are available for analyzing the data, even if heterogeneity in capture probabilities is apparent (White et al. 1982). Alternatively, the Peterson method can be modified to include a second sample consisting of a sequence of removals. This 'mark-removal method' (Seber 1986) combines the mark-recapture and removal models, and provides various tests of underlying assumptions about catchabilities (Paloheimo 1963; Skalski and Robson 1982).

### 2.4 Estimating biomass

Estimates of biomass (product of population abundance and mean weight) of juvenile salmon within study sites requires that the mean size of salmon be determined. Estimates of mean size and growth rate may be biased high if electrofishing tends to take larger individuals of a year class, and fish from the first sample only are measured. Although size-selectivity within age groups is apparently not usually a problem with juvenile salmon (Table 6), these biases can be avoided if fish from all electrofishing samples are sampled randomly for length and weight (i.e. not just the first sample).

Chapman (1978) provides formulae for estimating the variance associated with estimates of fish biomass from field data.

## 3. Estimating the total population of juvenile salmon in a stream <br> 3.1 Spatial distribution among sites

Annual mean densities and associated variances of juvenile Atlantic salmon at 15 sites in the Miramichi River, sampled from 1972 to 1989 are summarized in Table 7. For all years, estimated variances greatly exceeded the mean densities, and coefficients of variation (CV; standard deviation/mean density) were high, averaging 1.0 for age 0 salmon and 1.2 for age 1 salmon. Similar coefficients of variation were reported by Elson (1967). In both examples, however, the sites were widely separated (spatially), and covered a variety of habitat types. For more localized sites within the same stream, CV's would be less; for example, Bohlin et al. (1982) reported CV's of about 0.7 in small streams in Norway.

For the Miramichi data, the relationship between the means and estimated variances indicated that the distribution of the juvenile Atlantic salmon was highly contagious (Elliott 1977). The relationship between mean density and variance among years was described by Taylor's Power Law (Taylor 1961), where $\log \mathrm{v}=\log \mathrm{a}+\mathrm{b} \log$ mean density. Parameter a depends mainly on the size of the survey area. Parameter $b$ is an index of dispersion and can vary from 0 (regular distribution) to infinity (highly clustered distribution). The Power Law was a good fit to the data for each age group in both the Miramichi (Fig. 2) and Restigouche Rivers. For most age groups, slopes (b) were greater than unity (Table 8), confirming that the distribution of juvenile salmon was not random (Fig. 2) (Taylor 1961; Green 1979; Taylor and Taylor 1977).

The application of Taylor's Power Law to the data on juvenile salmon densities also provided information on appropriate transformations to apply to salmon counts. When the parameter $b$ is close to 2 , as it was for most age groups (Table 8), the logarithmic transformation (more specifically $\log 10[x+1]$ because of 0 counts) is an appropriate transformation to apply to the data (Elliott 1977; Green 1979). Before transformation, the distribution of counts among sites was highly skewed, but the distribution was more normally distributed after the count data is log transformed (Fig. 3). A more precise transformation is provided by the parameter $b$ from Tayor's Power Law, where each count is replaced by $x^{p}$, where $\mathrm{p}=1-\mathrm{b} / 2$ (Elliott 1977;Green 1979).

### 3.2 Sampling Strategies and the Number of Sites Required

When the objective is to determine the total abundance of juvenile salmon in a large area, it is generally not feasible to census the entire area and subsampling is required. Sample sizes (number of sites) depend on the objective of the study and the level of precision that is required.

Both Bohlin (1990) and Hankin (1984) considered the estimation of total numbers of fish in a stream to be a two stage sampling problem. At the first stage, a set of stream sections is selected for study. At the second stage, population estimates are made within each study section using removal or markrecapture estimators as discussed above. Variances associated with the first stage (variability in fish densities among sites) is always substantially greater
than variances associated with population estimates within sites (Bohlin 1990; Hankin 1984). For the following discussion of survey techniques, the latter source of variance is temporarily ignored.

Different methods can be used to select study sites within a stream:

### 3.2.1 Simple random sampling

For simple random sampling (SRS), a number of sampling units ( $n$ ) are chosen at random fram the total survey area (N) such that the total population is represented by the sampled areas (Fig. 4). Usually, each survey area ( $n$ ) is equal in size or length. Because of the large variation in densities of salmon among sampling sites, a major disadvantage of SRS is that the sampling fraction ( $\mathrm{n} / \mathrm{N}$ ) must be large to achieve reasonable levels of precision (Table 9). To calculate absolute population abundance in the stream, the total area or number of sections (N) must be known. SRS may require considerable field effort and may be prohibitively costly.

### 3.2.2 Stratified randam sampling

A significant gain in precision for the same field effort is possible if a stratified randam sampling design (STRAT) is chosen. Strata are selected such that the variance in densities among sites within strata is less than the variances among strata; that is, areas of high and low densities of fish are identified and strata boundaries are chosen accordingly. Usually the strata are unequal in area, and the number of units allocated to each stratum is proportional to the size of the strata (i.e., proportional sampling) (Fig. 4). Bohlin et al. (1982) gives an example of the gain in precision of a SIRAT versus a SRS survey design for Brown trout. Using bottam type as the strata variable, he found that the STRAT design resulted in a $14 \%$ to $52 \%$ gain in precision for the same field effort. Any enviromental factor that accounts for the variability in juvenile salmon abundance among sites (eg. riffles/pools; proximity to spawning areas or to the sea; substrate; stream order; gradient) can be used to establish strata boundaries for a survey area. However, in order for the total absolute population abundance to be estimated, the area of each strata must be known. Therefore, considerable work must be done to map the habitat variable(s) selected before the survey can be carried out.

### 3.2.3 Ratio estimator and proportional sampling

For the SRS and STRAT survey designs discussed above, the primary sampling units ( n ) are usually of equal size. Hankin (1984) argued that it may be advisable to let the size of the primary sampling units within strata vary depending on the size of natural habitat units. There are both biological and statistical advantages to having sampling sites of unequal sizes. First, nonsampling errors associated with erecting barrier nets midway within a pool or riffle would be eliminated. Fish are less likely to be disturbed and flee from the survey area if natural habitat boundaries are used as site boundaries.

Secondly, there are many two-stage designs that can be used to estimate the total abundance of fish when unequal site sizes are used, some of which may be considerably more efficient (i.e., result in increased precision with the same field sampling effort) than SRS or simple SIRAT designs.

Hankin (1984) discusses four possible models for choosing study site locations within strata; the choice of an appropriate method depends on the correlation between fish numbers and habitat unit sizes, the total number of stream sections, and the sample size that is feasible. Two methods in particular are worth noting: (1) a SRS/ratio estimator, where the size of the sampling units (which Hankin calls the auxiliary variable) is incorporated into the estimator; and (2) proportional sampling, sites are selected with a probability proportion to their sizes (PPS) (Fig. 4). Both methods may result in increased precision from SIRAT if the total mmber of fish in each site is highly correlated to the area of the site. Hankin (1984) discusses these methods in detail and provides information that is helpful for selecting an appropriate survey design.

### 3.2.4 Fixed Stations

In large rivers, where determining the absolute abundance of juvenile salmon is not feasible, it may be possible to determine an index of relative abundance. A number of survey sites, which remain fixed in location from one year to the next, can be identified and monitored (Johnson and Nielson 1983). As these authors note, 'randamess is abandoned in this case, and the data are not expected to represent accurately the average situation that would be revealed by random sampling. The use of the data is to monitor change, and the only assumption is that changes seen at permanent sampling stations reflect overall changes'.

Fixed stations have been surveyed anmully on the Miramichi and Restigouche Rivers for a number of years (the establishment of the Miramichi sites is discussed by Elson 1974). In the early 1970's, a large number of sites were surveyed annually ( 80 to 90 ), but the number of sites in recent years has been reduced substantially (15) because of manpower constraints.

Annual changes in salmon densities are often correlated among sites (pers. observation). Mean densities of salmon at the 15 sites were significantly correlated to mean densities at the higher number of sites (Randall and Schofield 1988), indicating that the subset of sites was providing a similar index. Densities of salmon at the fixed stations were significantly correlated with other indices of spawning escapement, including angling catches, trap counts of adult salmon and estimates of egg deposition (Elson 1974; Randall et al. 1990). Annual surveys at fixed sites to provide an index of recruitment of juvenile salmon seem to have considerable potential.

## RESIEARCHI NHGDB

The above summary of techniques used to estimate the abundance of juvenile salmonids in streams of eastern Canada indicated the following research needs:

1. Because of the unequal catchability of individual salmon, with a proportion of fish having a low catchability, the removal estimator is sometimes negatively biased. Research is required to determine the extent of this bias, particularly when the surveys are being conducted in different habitats.
2. The effect of marking on the subsequent catchability of salmon during electrofishing surveys requires further research. As for the removal method, the performance of the Petersen method in different habitat types needs to be verified.
3. Appropriate subsampling techniques (stratified random and proportionate sampling) need to be investigated for areas where a total stream cencus is required and only a portion of the total stream area can be surveyed.
4. In lange rivers, the use of fixed stations to determine an index of relative abundance of juvenile salmon has considerable potential as a monitoring tool. However, research is required to validate the use of index sites, possibly in small streams where both relative abundance (fixed station) and absolute abundance (stratified random sampling) estimates could be made. Careful monitoring in a study stream would also provide information on the best choice of sites. For example, should permanent sampling stations be located in prime salmon habitat or secondary habitat? How many fixed sites would be required?

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Table 1. Examples of types of gear used for population studies of salmonids in rivers and streams (excluding electrofishing gear).

| Gear | References |
| :--- | :--- |
| Seines | Elson 1962 <br> Gibson 1973 |
| Trap nets | Gibson 1973 <br> Gibson et al. 1987 |
| Snorkelling | Gibson 1973 <br> Gardiner 1984 <br> Cunjak et al. 1988 |
| Rotenone | Mahon 1980 |
| Minnow traps | Swales 1987 |
| Angling | Gibson 1973 |
| Gillnet | Gibson et al. 1987 |

Table 2. Fundamental assumptions of removal and Petersen mark-recapture methods for estimating abundance of fish populations.

| Assumption | Removal | Petersen |
| :--- | :---: | :---: |
| 1. Population is closed | X | X |
| 2. Equal catchability <br> i. among individuals <br> ii. marked vs unmarked | $\mathrm{X} *$ | $\mathrm{X} *$ |
| 3. Capture probability remains <br> constant among samples <br> (removals) | $\mathrm{X} *$ |  |
| 4. Marks are detected |  |  |
| 5. Fish do not lose their marks |  |  |

* indicates assumptions that are sometimes not valid for the removal on Peterson methods.

Table 3. Precision of estimates of juvenile Atlantic salmon abundance using the removal method. $n$ is the mmber of sites in Miramichi River where the total catch was 40 or more.

| Age | $n$ | Coefficient of variation |  |
| :--- | :---: | :---: | :---: |
|  |  | Mean | Range |
| 0 | 141 | 0.06 | $<0.01-0.89$ |
| 1 | 55 | 0.09 | $<0.01-0.73$ |
| 2 | 6 | 0.08 | $0.05-0.13$ |

Table 4. Frequency of sites in the Miramichi River where the probability of capture (p) remained constant from one electrofishing removal to another.

| Age | Number of sites | Category | Frequency | \% |
| :---: | :---: | :---: | :---: | :---: |
| 0 | 178 | p constant | 137 | 77.0 |
|  |  | p not constant | 33 | 18.5 |
|  |  | $\mathrm{p}<0.20$ | 7 | 3.9 |
|  |  | Failure | 1 | 0.6 |
| 1 | 60 | p constant | 52 | 86.7 |
|  |  | p not constant | 5 | 8.3 |
|  |  | $\mathrm{p}<0.20$ | 3 | 5.0 |
|  |  | Failure | 0 | 0.0 |

Table 5 . Numbers of juvenile salmon captured during successive electrofishing removals and associated appropriate estimators. The G-test is a goodness of fit test for constant capture probability among removals (see texst).

|  | Removal number |  |  |  | G-test | Model |
| :--- | :---: | :---: | :---: | :---: | :---: | :---: |
|  | 1 | 2 | 3 | 4 |  |  |
| 1 | 148 | 79 | 62 | 32 | $2.2(\mathrm{NS})$ | a |
| 2 | 162 | 42 | 26 | 22 | $16.6(\mathrm{~S})$ | b |
| 3 | 28 | 42 | 31 | 20 | - | C |

Appropriate models: a - constant catchability, therefore removal method; b capture probability for first sample is higher than subsequent samples, therefore generalized removal method; c - no significant reduction in mumbers, therefore removal method not applicable.

Table 6. Comparison of the mean fork lengths (cm) of juvenile salmon captured in two successive samples by electrofishing at a stream site (Sample sizes in parenthesis).

| Example | Age | Sample |  |  |  | F value |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  | 1 |  | 2 |  |  |  |
| 1 | 1 | 54.9 |  | 55.4 |  | 0.21 | (NS) |
| 2 | 1 | 66.8 | (24) | 66.1 |  | 0.20 | (NS) |
| 3 | 0 | 45.9 | (60) | 45.0 |  | 2.27 | (NS) |
| 4 | 1 | 83.1 | (22) | 83.0 |  | $<0.01$ | (NS) |
| 5 | 0 | 52.3 | (31) | 54.3 |  | 3.03 | (NS) |

Table 7. Annual mean density of age 0 and age 1 salmon parr at 15 sites in the Miramichi River, 1972 to 1989. RSE is the relative standard error (standard error/density).

|  |  |  |  |  |  |  |  |  |  |
| ---: | ---: | ---: | ---: | ---: | ---: | ---: | ---: | ---: | ---: |
| OBS | YEAR | AGE | DENSITY | VAR | SD | SE | CV | RSE |  |
|  |  |  |  |  |  |  |  |  |  |
| 1 | 1972 | 0 | 9.75 | 211.12 | 14.5300 | 3.7516 | 1.49025 | 0.384782 |  |
| 2 | 1973 | 0 | 24.94 | 1013.30 | 31.8324 | 8.2191 | 1.27636 | 0.329554 |  |
| 3 | 1974 | 0 | 34.20 | 1247.50 | 35.3200 | 9.1196 | 1.03275 | 0.266654 |  |
| 4 | 1975 | 0 | 40.04 | 798.10 | 28.2507 | 7.2943 | 0.70556 | 0.182175 |  |
| 5 | 1976 | 0 | 25.09 | 440.31 | 20.9836 | 5.4179 | 0.83633 | 0.215940 |  |
| 6 | 1977 | 0 | 51.79 | 1838.90 | 42.8824 | 11.0722 | 0.82801 | 0.213790 |  |
| 7 | 1978 | 0 | 36.37 | 2127.46 | 46.1244 | 11.9093 | 1.26820 | 0.327448 |  |
| 8 | 1979 | 0 | 19.73 | 350.36 | 18.7179 | 4.8329 | 0.94870 | 0.244954 |  |
| 9 | 1980 | 0 | 34.46 | 2846.01 | 53.3480 | 13.7744 | 1.54811 | 0.399721 |  |
| 10 | 1981 | 0 | 53.60 | 2236.58 | 47.2925 | 12.2109 | 0.88232 | 0.227815 |  |
| 11 | 1982 | 0 | 15.00 | 128.31 | 11.3274 | 2.9247 | 0.75516 | 0.194981 |  |
| 12 | 1983 | 0 | 44.52 | 1470.47 | 38.3467 | 9.9011 | 0.86134 | 0.222396 |  |
| 13 | 1984 | 0 | 19.07 | 308.22 | 17.5562 | 4.5330 | 0.92062 | 0.237703 |  |
| 14 | 1986 | 0 | 55.36 | 2267.39 | 47.6171 | 12.2947 | 0.86014 | 0.222086 |  |
| 15 | 1987 | 0 | 74.54 | 7712.32 | 87.8198 | 22.6750 | 1.17816 | 0.304199 |  |
| 16 | 1988 | 0 | 95.10 | 9172.38 | 95.7725 | 24.7284 | 1.00707 | 0.260025 |  |
| 17 | 1989 | 0 | 72.24 | 2611.28 | 51.1007 | 13.1941 | 0.70737 | 0.182643 |  |
| 18 | 1972 | 1 | 8.29 | 137.13 | 11.7103 | 3.0236 | 1.41258 | 0.364725 |  |
| 19 | 1973 | 1 | 3.01 | 45.45 | 6.7417 | 1.7407 | 2.23975 | 0.578302 |  |
| 20 | 1974 | 1 | 10.96 | 220.32 | 14.8432 | 3.8325 | 1.35430 | 0.349680 |  |
| 21 | 1975 | 1 | 12.82 | 181.87 | 13.4859 | 3.4820 | 1.05194 | 0.271611 |  |
| 22 | 1976 | 1 | 11.73 | 122.44 | 11.0653 | 2.8570 | 0.94333 | 0.243567 |  |
| 23 | 1977 | 1 | 8.43 | 144.29 | 12.0121 | 3.1015 | 1.42492 | 0.367913 |  |
| 24 | 1978 | 1 | 10.72 | 200.54 | 14.1612 | 3.6564 | 1.32101 | 0.341083 |  |
| 25 | 1979 | 1 | 9.04 | 84.31 | 9.1820 | 2.3708 | 1.01571 | 0.262256 |  |
| 26 | 1980 | 1 | 8.31 | 130.09 | 11.4057 | 2.9449 | 1.37253 | 0.354385 |  |
| 27 | 1981 | 1 | 7.03 | 91.69 | 9.5755 | 2.4724 | 1.36209 | 0.351690 |  |
| 28 | 1982 | 1 | 9.77 | 162.46 | 12.7460 | 3.2910 | 1.30460 | 0.336847 |  |
| 29 | 1983 | 1 | 6.73 | 32.74 | 5.7219 | 1.4774 | 0.85021 | 0.219522 |  |
| 30 | 1984 | 1 | 6.49 | 28.43 | 5.3320 | 1.3767 | 0.82157 | 0.212128 |  |
| 31 | 1986 | 1 | 12.23 | 172.27 | 13.1252 | 3.3889 | 1.07319 | 0.277098 |  |
| 32 | 1987 | 1 | 13.07 | 162.70 | 12.7554 | 3.2934 | 0.97593 | 0.251984 |  |
| 33 | 1988 | 1 | 13.86 | 326.36 | 18.0654 | 4.6645 | 1.30342 | 0.336542 |  |
| 34 | 1989 | 1 | 18.36 | 270.29 | 16.4405 | 4.2449 | 0.89545 | 0.231205 |  |


| VARIABLE | N | MEAN | MINIMUM VALUE | MAXIMUM VALUE |
| :---: | :---: | :---: | :---: | :---: |
| DENSITY | 17 | 41.517647 | 9.7500000 | 95.100000 |
| VAR | 17 | 2163.530000 | 128.3100000 | 9172.380000 |
| SD | 17 | 40.518953 | 11.3274004 | 95.772543 |
| CV | 17 | 1.006262 | 0.7055610 | 1.548114 |
| RSE | 17 | 0.259816 | 0.1821751 | 0.399721 |
|  | AGE=1 |  |  |  |
| DENSITY | 17 | 10.0500000 | 3.01000000 | 18.3600000 |
| VAR | 17 | 147.8458824 | 28.43000000 | 326.3600000 |
| SD | 17 | 11.6687731 | 5.33197899 | 18.0654366 |
| CV | 17 | 1.2189732 | 0.82156841 | 2.2397547 |
| RSE | 17 | 0.3147375 | 0.21212805 | 0.5783022 |

Table 8. Values of constants a and $b$ (SE's in parenthesis) for Taylors Power Law (see text) fitted to data sets of juvenile salmon in the Miramichi and Restigouche rivers. All regressions were based on 18 samples. P2 is the coefficient of determination. slopes (b) ware not significantiy different from 2 except for age 1 parr from Miramichi River.

| River | Age | a |  | b |  | R2 |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Miramichi | 0 | 0.29 | (0.31) | 1.80 | (0.20) | 0.85 |
|  | 1 | 0.78 | (0.27) | 1.35 | (0.27) | 0.62 |
|  | 2 | 0.15 | (0.15) | 2.04 | (0.32) | 0.73 |
| Restigouche | 0 | 0.63 | (0.13) | 1.51 | (0.10) | 0.93 |
|  | 1 | 0.13 | (0.17) |  | (0.23) | 0.80 |
|  | 2 | 0.28 | (0.05) | 1.74 | (0.16) | 0.88 |

Table 9. Approximate muber of sections required to reach relative standard errors ( $\mathrm{SE} / \mathrm{mean}$ ) of 0.10 and 0.20 , assuming coefficients of variation of 0.7 , 1.0 and 1.2, simple random sampling, and a total potential number of sites of 100 or > 200 (infinite). Calculated using equations from Bohlin et al. (1982).

| CV | Relative standard error |  |  |  |
| :---: | :---: | :---: | :---: | :---: |
|  | 0.10 |  | 0.20 |  |
|  | 100 | infinite | 100 | infinite |
| 0.7 | 33 | 50 | 11 | 12 |
| 1.0 | 50 | 100 | 20 | 25 |
| 1.2 | 59 | 144 | 26 | 36 |



Fig. 1. Comparison of actual versus estimated numbers of salmon at five study sites in streams. Estimated numbers (vertical bars indicate $95 \%$ confidence limits) were based on electrofishing data and the removal method (data from Bohlin and Sundstrom 1977 and Peterson and Cederholm 1984). The solid line indicates the one to one relationship between actual and estimated numbers of fish at the study sites; confidence intervals of unbiased estimates of population numbers would usually touch this line.


Fig. 2. Relationship between the sample variance and arithmetic mean for densities (number per $100 \mathrm{~m}^{2}$ ) of juvenile salmon. Each mean is based on data from 15 stream sites. The broken line represents the relationship for a random distribution (variance $=$ mean). Means and variances were log transformed.

AGE=1

|  | FREQUENCY BAR CHART |  |  | PERCENT | CUM. PERCENT |
| :---: | :---: | :---: | :---: | :---: | :---: |
| MIDPOINT |  |  |  |  |  |
| DENSITY |  | FREQ | CUM. <br> FREQ |  |  |
| 0 | 1 |  |  |  |  |
|  | \|*********************** | 114 | 114 | 42.38 | 42.38 |
|  | + |  |  |  |  |
| 8 | \|**************** | 82 | 196 | 30.48 | 72.86 |
|  | \| |  |  |  |  |
| 16 | j****** | 28 | 224 | 10.41 | 83.27 |
|  | I |  |  |  |  |
| 24 | 1**** | 19 | 243 | 7.06 | 90.33 |
|  | \| |  |  |  |  |
| 32 | \|** | 10 | 253 | 3.72 | 94.05 |
|  | I |  |  |  |  |
| 40 | 1* | 7 | 260 | 2.60 | 96.65 |
|  | \| |  |  |  |  |
| 48 | 1* | 7 | 267 | 2.60 | 99.26 |
|  | I |  |  |  |  |
| 56 | 1 | 1 | 268 | 0.37 | 99.63 |
|  | I |  |  |  |  |
| 64 | 1 | 1 | 269 | 0.37 | 100.00 |
|  | 1 |  |  |  |  |
|  |  |  |  |  |  |
|  | $\begin{array}{lllll}20 & 40 & 60 & 80 & 100\end{array}$ |  |  |  |  |
|  | FREQUENCY |  |  |  |  |

AGE=1 (transformed)
FREQUENCY BAR CHART


Fig. 3. Frequency bar charts for densities of age 1 salmon parr among 269 stream sites. Data are shown before transformation (upper) and after $\log$ transformation (lower).


Fig. 4. Schematic diagram of a section of stream showing different sampling strategies. A - study sites are chosen using simple random sampling; B the stream is divided into pool (unstippled) and riffle (stippled) habitat strata, and study sites are selected at random within each strata; C - within each strata type, study sites are unequal in size, and are chosen at random.

# Population and Biomass Estimates of Juvenile Atlantic Salmon in Lacustrine Habitat in Insular Newfoundland 

## by

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#### Abstract

In many river systems in insular Newfoundland, a substantial proportion of the total production of Atlantic salmon smolts comes from lacustrine habitat. While the relative importance of lacustrine habitat has long been recognized, to date there have been only a few attempts to quantitatively define lacustrine production. Juveniles of different sizes and ages have been shown to utilize different lentic areas to varying degrees for rearing and the relative amount of these areas can vary among lakes. Certain morphometric parameters can have a greater effect than others on productive capacity and these as well can vary among lakes. Methods of estimating population size and biomass in lacustrine habitat were reviewed. A new approach addressing production from different lentic habitats, morphometric considerations, and movements of juveniles is suggested.


## Résumé

Dans de nombreux réseaux hydrographiques de l'ile de Terre-Neuve, une bonne partie de la production totale de saumoneaux provient d'habitats lacustres. Bien que l'on connaisse depuis longtemps l'importance relative de ce type d'habitat, on s'est jusqu'ici contententé de quelques tentatives de quantitification de sa production. Il apparait que les juvéniles d'âges et de grosseurs variés utilisent à des degrés divers les milieux lénitiques en phase de grossissement et que ces milieux varient en importance d'un lac à un autre. Certains paramètres morphométriques ont des effets plus marquants que d'autres sur la capacité de production. Là encore, la situation varie d'un lac à un autre. On a examiné les méthodes d'estimation de la population et de la biomasse dans les habitats lacustres. On propose une nouvelle méthode d'évaluation de la production des divers habitats lénitiques, tenant compte de considérations morphométriques et des mouvements de juvéniles.

## Introduction

Anadromous Atlantic salmon (Salmo salar) are typically considered to be stream (predominantly riffle) dwellers (Keenleyside 1962; Gibson 1966: Elson and Tuomi 1975; Symons and Heland 1978). While this applies to the species throughout most of its North American distribution, in insular Newfoundland juveniles make extensive use of lacustrine habitat for rearing (Pepper 1976; $0^{\prime}$ Connell and Reddin 1983; Chadwick and Green 1985; Pepper et al. 1985; $0^{\prime}$ Connell 1986; Ryan 1986; $0^{\prime}$ Connell and Ash 1989). Members of the families Esocidae, Cyprinidae, and Percidae are not found in insular Newfoundland waters (Scott and Crossman (1964). The use of lacustrine habitat by Atlantic salmon juveniles can probably be attributed to the lack of these potential predators and competitors. Approximately one-third of the surface area of insular Newfoundland is comprised of freshwater, much of which is accessible to anadromous Atlantic salmon.

Some understanding of the relative importance of lacustrine habitat for rearing purposes on an individual river basis can be obtained from Table 1. Compared to certain river systems in St. Mary's Bay, some systems in Bonavista Bay are dominated to a considerable extent by lacustrine habitat. These rivers can be regarded as extremes with many river systems in insular Newfoundland falling somewhere in between.

Given the great overall importance of lake rearing, the problem arises of defining juvenile production in terms of lacustrine habitat. An approach to solving this problem and inherent difficulties associated with it are the subject of this paper. Methods of measuring population size and biomass previously employed are reviewed.

## Sampling Gear Employed

Population estimates of juvenile salmon based on mark-recapture techniques for individual ponds in insular Newfoundland have employed fyke traps (Ryan 1986). In another approach ( $0^{\prime}$ Connell and Ash 1989), modified fyke nets (installed in the stream) were used to count smolts leaving a number of ponds in a section of a river system where virtually all habitat is lacustrine. A counting fence of the type described by Anderson and McDonald (1978) can also be used in this situation instead of fyke nets which are more prone to washout.

In addition to estimating juvenile population and biomass levels in terms of units based on total lake surface area, an attempt is currently underway to define lacustrine production in terms of different lentic areas ( $0^{\prime}$ Connell et al. 1990). The first step in this process is to determine the temporal and spatial utilization of various lentic habitats. In this type of study, Lundgren multiple mesh experimental gill nets of the type described and used in Newfoundland by Hammar and Filippson (1985) have been employed. These nets are set in the benthic zone along different depth contours and also suspended in the pelagic zone.

## Estimates of Population Numbers

## Indirect Methods

The indirect designation as used here refers to the method in which lacustrine smolt production was derived; these approaches make use of population and biomass estimates determined by the direct methods presented below as primary data.

Chadwick and Green (1985) estimated that approximately $67 \%$ of the smolts leaving Western Arm Brook, a system located near the tip of the Great Northern Peninsula, Newfoundland, were produced in lacustrine habitat. Total numbers of smolts, determined by means of a counting fence located near the mouth of the river, were converted to biomass in terms of parr. The biomass of parr produced in fluvial habitat was determined by electrofishing. The biomass attributable to lacustrine habitat was the difference between total biomass and fluvial biomass. Using this approach, any deviation from a 50:50 sex ratio of parr is a possible source of error since the total production of a year-class was calculated as twice that for females. Calculated lacustrine values also depend on the representativeness of the fluvial estimates. Chadwick and Green (1985) indicated that their estimate of fluvial production was probably an underestimate and hence the lacustrine component was overestimated.

Ryan (1986) calculated the net number of juvenile salmon leaving two ponds located in the headwaters of the Gander River, central Newfoundland, as the difference between spring and fall population estimates. A significant positive relationship was obtained between the number of emigrants in a given year and adult abundance (as catch per unit of effort in the recreational fishery) one year later. Emigrants were therefore used as an index of smolt production. The precision of this approach depends to a large extent on the accuracy of population estimates. Also, it is assumed that most if not all juveniles leaving the ponds are destined to become smolts in the year in question.

For both the Chadwick and Green (1985) and Ryan (1986) studies, a possible confounding factor is movements between lacustrine and fluvial habitats.

## Direct Methods

Mark-recapture
Ryan (1986) used the Schnabel multiple mark-recapture method to estimate population levels of juvenile Atlantic salmon for several years in the two Gander River headwater ponds referred to above. Estimates were conducted in spring and fall each year. For Headwater Pond, in 5 out of a total of 10 estimates the $95 \%$ confidence limit ranged from approximately $10-25 \%$ of the population estimate; for the remaining 5 estimates, the range was $41-147 \%$. For Spruce Pond, the $95 \%$ confidence limit varied between 9 and $32 \%$ of the population estimate (total of 11 estimates); in one instance the $95 \%$ confidence limit was $57 \%$ of the estimate. Roff (1973) indicated that mark-recapture estimates can be considered reliable if confidence limits are plus or minus 10\% of the population estimate and the coefficient of variation is less than $5 \%$. Few studies however meet these criteria.

## Counting fences

Counting fences (or modified fyke nets) can be used to count smolts leaving a pond or group of ponds similar to the approach of $0^{\prime}$ Connell and Ash (1989). In this study, estimated smolt production from fluvial habitat in the area above the enumeration site was subtracted from total smolt output to give an estimate of lacustrine production for Northeast River, Placentia. The counting method was direct but the allocation of production to lacustrine habitat was indirect. Smolt production in fluvial habitat was based on parameters derived for other Newfoundland river systems where virtually all available rearing habitat is fluvial. Applying parameters developed elsewhere is always a potential source of error. Lacustrine smolt production was expressed as the number of smolts per ha of total surface area.

## Estimates of Mean Size

When converting estimates of population numbers to biomass it is absolutely essential that age composition and mean weight and length data are representative. Data obtained from counting fences collected at random over the entire smolt run can be considered reasonably representative. Data collected at Western Arm Brook (Chadwick et al. (1978) are an example of this approach. Fig. 1 shows age composition data used by Ryan (1986) in spring and fall population estimates for the Gander River ponds. There was annual variability in age composition and in the number of age groups represented each season or year. Mean weights used in the study also showed a fair amount of annual variability. Sampling for both age and size parameters was considered reasonably random and followed the same procedure each year.

Age composition of parr taken in Lundgren multiple mesh gill nets in various lentic habitats in Junction Pond, Northeast River, Placentia and Conne Pond, Conne River, are shown in Figs. 2 and 3 respectively. Length composition is shown in Figs. 4 and 5 respectively. Data are combined temporally. Overall, the gear was effective at sampling a wide range of age groups ( $0+-6+$ ) and sizes.

## Discussion

A study of the temporal and spatial distribution of parr (using Lundgren multiple mesh gill nets) in Junction Pond, Northeast River, Placentia and Conne Pond, Conne River, Newfoundland revealed that different lentic areas were used for rearing with the shoreline littoral zone accounting for 59 and $72 \%$ of occurrences overall ( $0^{\prime}$ Connell et al. 1990). While this indicates that the single most important rearing area is littoral zone, other areas (pelagic zone and deeper benthic areas) in are very important as well. The limit of the littoral zone depth in this study was 2 m for one pond and 3 m for the other. However, the amount of shoreline area to a depth of $2-3 \mathrm{~m}$ (expressed as a percentage of total surface area) can vary considerably among ponds. This is exhibited by examining percentage hypsographic curves developed for the major lakes in Northeast River, Placentia (Fig. 6). The amount of shoreline littoral
area available for rearing is also affected by shore development (an index measure of shoreline irregularity) and there can be variability in this morphometric parameter as well (Table 2). In light of this, comparisons of smolt production among lakes in terms of total surface area can be quite misleading.

Fyke nets typically are set perpendicular to the shore and extend out to a maximum depth of around 2 m . This is the area usually occupied by the littoral zone. 0'Connell et al. (1990) demonstrated that compared to the littoral zone, larger and older parr were found in the pelagic zone and in deeper areas of the benthic zone. If there is an overall tendency for certain groups of parr to occupy a particular lentic zone, then age and size composition (as well as population size) could be underestimated using fyke nets set in the littoral zone. In the study by Ryan (1986), the maximum depth of both ponds was within the littoral zone (as defined in the Junction Pond and Conne Pond study).

A variety of methods and approaches for estimating population size and biomass of juvenile Atlantic salmon in lakes has been reviewed. The methods varied in complexity and precision. It is quite evident that the most appropriate way to define production is in terms of the relative contribution of different lentic zones and that production expressed in units of total lake surface area could be misleading when making interlake comparisons. Further research is required to determine the extent of movements of parr of different sizes and ages among different lentic zones, among ponds, and between lentic and fluvial habitats.

Another possible way to express results is in terms of length of shoreline. While this addresses differences in shore development, differences in slope of basin are not accounted for, which could confound estimates.

A new approach is to combine estimates of smolt production with a knowledge of spatial and temporal habitat usage and allocate production to different lentic zones accordingly (after accounting for fluvial production). A number of stations would have to be examined in a given pond depending on the size and the results extrapolated to the whole pond. For small systems (e.g., Northeast River) all lakes can be surveyed for habitat usage; for large systems (e.g., Conne River) this can be determined for a sampling of ponds and the results applied to the whole river.

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Table 1. The ratio of lacustrine to fluvial habitat (in terms of $\mathrm{m}^{2}$ ) for selected river systems dominated by lacustrine habitat (Bonavista Bay) and those with a large proportion of fluvial habitat (St. Mary's Bay).

| River | Habitat area |  | Ratio <br> (ha) $)$ |
| :--- | :---: | :---: | :---: |
|  | Fluvial <br> $\left(100 \mathrm{~m}^{2}\right.$ units $)$ | Racustrine/Fluvial |  |
| Indian Bay |  |  |  |
| Traverse | 9,878 | 3,861 | 255.8 |
| Middle | 4,389 | 2,639 | 166.3 |
| Northwest | 4,636 | 2,640 | 175.6 |
|  | 8,489 | 4,093 | 207.4 |
| ST. MARY'S BAY |  |  |  |
| Northeast |  |  |  |
| Northwest | 29 | 262 | 11.1 |
| North Harbour | 648 | 8,467 | 7.7 |
| Branch | 63 | 912 | 6.9 |
|  | 150 | 7,670 | 2.0 |

Table 2. Morphometry of the major lakes located in Northeast River, Placentia, Newfoundland.

| Parameter | Fitzgeralds Pond | Junction Pond | $\begin{aligned} & \text { Big Gull } \\ & \text { Pond } \end{aligned}$ | Rhodies Pond (East) | Rhodies Pond (West) |
| :---: | :---: | :---: | :---: | :---: | :---: |
| Surface area, excluding islands (ha) | 129.93 | 61.92 | 291.29 | 145.51 | 102.08 |
| Volume ( $\mathrm{m}^{3}$ ) | $3.66 \times 10^{6}$ | $1.93 \times 10^{6}$ | $10.57 \times 10^{6}$ | $5.39 \times 10^{6}$ | $2.38 \times 10^{6}$ |
| Maximum length (km) | 3.18 | 1.72 | 3.77 | 2.80 | 2.85 |
| Maximum width (km) | 1.00 | 0.66 | 1.95 | 0.75 | 0.84 |
| Mean width (km) | 0.41 | 0.36 | 0.81 | 0.52 | 0.37 |
| Maximum depth (m) | 8.0 | 12.0 | 9.0 | 16.0 | 13.0 |
| Mean depth (m) | 2.82 | 3.12 | 3.63 | 3.70 | 2.33 |
| Length of shoreline (km) | 7.8 | 5.6 | 15.1 | 10.8 | 9.0 |
| Mean depth - maximum depth relation | 0.35 | 0.26 | 0.40 | 0.23 | 0.18 |
| $\begin{aligned} & \text { Maximum depth - surface } \\ & \text { relation } \end{aligned}$ | 0.007 | 0.015 | 0.005 | 0.013 | 0.013 |
| Shore development | 1.93 | 2.01 | 2.43 | 2.52 | 2.46 |
| Volume development | 1.06 | 0.78 | 1.21 | 0.69 | 0.54 |



Fig. 1. Age composition of Atlantic salmon parr in headiwater and Spruce ponds (Gancier River) as a whole at the end of population estimates each spring and fall, 1979-83. From Ryan (1986).

Atlantic salmon




Fig. 2. Age composition of Atlantic salmon parr taken in Lundgren multiple mesh experimental gill nets, Junction Pond, Newfoundland, 1987.

## Atlantic salmon



Fig. 3. Age composition of Atlantic salmon parr taken in Lundgren multiple mesh experimental gill nets, Conne Pond, Newfoundland, 1987.

## Atlantic salmon





Fig. 4. Length frequency distributions of Atlantic salmon parr taken in Lundgren multiple mesh experimental gill nets, Junction Pond, Newfoundland, 1987.

Atlantic salmon




Fig. 5. Length frequency distributions of Atlantic salmon parr taken in Lundgren multiple mesh experimental gill nets, Conne Pond, 1987.


Fig. 6. Percentage hypsographic curves for the major ponds in Hortheast River, Placentia.

# Current and Potential Adult Habitat of Northern Pike (Esox Lucius L.) in Hamilton Harbour, Lake Ontario. 

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#### Abstract

According to the criteria of the Intemational Joint Commission, the Hamilton Harbour, Lake Ontario, ecosystem is one of 42 heavily impacted Areas of Concem around the Great Lakes. Much of the original fish habitat has been destroyed by man and what remains is badly impaired. Formulation of plans to restore fish habitat requires assessment of the current supply of suitable fish habitat and prediction of the potential supply under various remedial options. We used the SPANS GIS (Tydac Technologies) to integrate existing habitat data bases for the Harbour with published knowledge of the ecology and habitat needs of northern pike, a warmwater top predator essential to the stability of the ecosystem. Then we modelled potential changes in pike habitat given improvements in water quality. The results demonstrate the central role of GIS-based approaches in fish habitat management.


## RESUME

Selon les critères de la Commission mixte internationale, l'écosystème du port de Hamilton, sur le lac Ontario, est un des secteurs les plus lourdement affectés de la région des Grands Lacs. Une bonne partie de l'habitat du poisson a été détruit par l'homme et ce qu'il en reste est en très mauvais état. Avant de concevoir des plans de reconstitution de l'habitat du poisson, il faut d'abord faire l'inventaire de l'habitat adéquat restant et prédire l'apport que sont susceptibles d'offrir à cet égard diverses mesures correctives. Nous avons eu recours au système général d'information (GIS) SPANS, de Tydac Technologies, pour intégrer les bases de données sur l'habitat existant dans le port aux données publiées sur l'écologie et les besoins du brochet du Nord en matière d'habitat. (Le brochet du Nord est un prédateur d'eau chaude essentiel à la stabilité de l'écosystème). Nous avons ensuite modélisé les changements que l'amélioration de la qualité de l'eau pourrait apporter dans l'habitat du brochet. Les résultats obtenus illustrent le rôle essentiel des GIS dans la gestion de l'habitat du poisson.

In October 1986, the Department of Fisheries and Oceans issued a "Policy for the Management of Fish Habitat" (DFO 1986) with the goal to "Increase the natural productive capacity of habitats for the nation's fisheries resources, to benefit present and future generations of Canadians." The guiding principle in the policy is "No net loss of the productive capacity of habitats." Thus there is an implicit need to quantify fish habitat in terms of productive capacity and produce a fish habitat 'budget'. Budgeting fish habitat requires unit qualitative indicators be combined with spatial measures to quantify productive capacity. The unit qualitative indices are variables such as presence-absence, numbers or biomass (relative or absolute), production, yield, etc. of preferred, critical or important species with units such as $\mathrm{kg} / \mathrm{ha}$, number/litre, etc. The spatial units refer to lengths, areas and volumes in specific locations. For example,

$$
\text { Harvest }(\mathrm{Kg}) \quad=\text { Unit Yield }(\mathrm{Kg} / \mathrm{ha}) \cdot \text { Area (ha) }
$$

The term 'Productive Capacity' relates primarily to potential exploitable production but comprehensive fish habitat budgeting will also involve measuring area with and without the capacity to produce particular species.

The DFO policy arose out of concern for past and on-going losses of important fish habitat. In the Great Lakes basin, where more than 60 million North Americans live, there has been considerable loss of habitat as a result of human developments. The International Joint Commission has identified 42 Areas of Concern (AOCs) around the Great Lakes where aquatic ecosystems have been significantly degraded as a result of eutrophication, contamination, and/or loss of physical habitat. Hamilton Harbour at the western tip of Lake Ontario is one of the most degraded areas among the 17 Canadian AOCs (Fig. 1). The harbour is a small bay ( Area $=2090$ ha) with a relatively high flushing rate ( 4 times/yr) which receives most of the waste water from two adjacent cities, Hamilton (population 360,000 ) and Burlington ( 120,000 ). All of the south shore marshes have been lost to industrial development. Wetlands in the harbour area have been significantly reduced as a result of filling, eutrophication, and sustained high water levels. The harbour's hypolimnion is deoxygenated during the summer. A detailed review of environmental conditions in Hamilton Harbour is available in the Stage I report (IJC 1989). In the past, the harbour supported one of the most important commercial and sport fisheries in western Lake Ontario.

In 1986, the IJC issued a call for Remedial Action Plans (RAPs) to be developed for all 42 AOCs, specifying the problems, the remedial options, and an implementation time-table. Among the objectives specified for the Hamilton Harbour RAP so far, is the rehabilitation of a self-sustaining, edible warmwater fishery dominated by species such as northern pike (Esox lucius) and largemouth bass (Micropterus salmoides). To support the RAP process, DFO has conducted a fisheries research program to assess the current status of the fish assemblage and fish habitat, to identify remedial options, and to predict the future status of the fishery. Much of the effort has been directed to the study of northem pike habitat requirements.

This paper describes our first efforts to contribute to the RAP process and to fulfil the restoration objective of the DFO policy using GIS technology. Specifically, we have used point observations of pike habitat selection obtained through a tracking study, and quantification of habitat supply obtained using SPANS GIS (Tydac Technologies) and on-site habitat assessment, to (i) measure summer habitat preferences of northern pike, (ii) estimate the current area of suitable habitat and (iii) predict the potential area of suitable habitat given rehabilitation of the Harbour ecosystem.

Chapman and Mackay (1984) and Diana et al. (1977) showed with tracking studies that adult non-spawning northern pike more frequently occupied locations with shallower depths, proximity to shore, and close to submerged vegetation. Chapman and Mackay also found pike preferred sand/rock bottom types. They also estimated the supply of habitat types in their study area, Seibert Lake, Alberta, and were able to measure how habitat selection differed from random. If both the frequency of selection of habitat types by fish and the availability of those habitat types are available, the ratio of observed to expected occurrences can be used as a index of habitat suitability. The indices can then be used in other situations to estimate the area of suitable habitat or to project area changes given development or rehabilitation.

In the United States, the Fish and Wildlife Service has developed an extensive system of habitat evaluation procedures (HEP) based largely on the use of habitat suitability index (HSI) models (Terrell et al. 1982). HSI models have been developed for many species, terrestrial and aquatic, including northern pike (Inskip 1982). Our approach is derived from that conceptual base with the addition of a spatial representation of the habitats involved.

## MATERIALS AND METHODS

## Database

Pike tracking observations: In the summer of 1987 six adult pike were released with implanted ultra-sonic tags. At various times between June 2 and August 27, a crew searched the Harbour with a locating-device. Whenever they found a pike, they recorded estimates of the location (these turned out to be highly inaccurate and were ignored in our analyses), depth, distance from shore and proximity to heavy or moderate vegetation. The cumulative database consists of 223 point observations.

Hamilton Harbour shoreline and depths: The shoreline and 1380 point depth measures shown on the 1:12,000 map of Hamilton Harbour (Sheet 2067 Canadian Hydrographic Service) were digitized and imported into the SPANS database. The depth were adjusted from the map datum to mean lake level for May to September at Cobourg, Port Weller and Toronto ( +0.8 metres). Because there were few depth points close to the shore, a subset (208) of the points defining the shoreline arcs were added with zero depth being sure to avoid points on artificial structures like piers and breakwaters. We used the contouring feature of SPANS to produce depth contours at the desired intervals (Fig. 1). We used the arc-to-corridor feature to generate bands of different distance from shore (Fig. 2).

Nearshore vegetation: In September 1987 and 1988, we conducted a survey of submerged vegetation in the nearshore zone of the Harbour using acoustic estimates confirmed by diver measurements along transects. Following the 1.5 metre depth contour line, all arcs were designated as belonging to one of four vegetation classes (heavy, moderate, sparse and absent). The arcs were marked in different colours on a $1: 12,000$ hydrographic map. The start and end coordinates of the different vegetation arcs were measured directly from the map and entered into the SPANS database. A series of transects at right angles to the shore showed the maximum depth of plant occurrence was about 2.5 metres. This was consistent with measures of Secchi depth. The SPANS graphics drawing feature was used to create and classify artificial polygons enclosing each vegetation arc. A map prepared from these was then cut to obtain the final map by 'imposing' a map bounded by the 0.5 and 2.5 metre depth contours (Fig. 3). We produced a similar map using the 0.5 and 4.5 depth contours to limit vegetation extent, representing potential future conditions.

## Approach

Pike habitat selection: We summarized the pike tracking data points treating the three variables of interest (depth, distance from shore, distance from moderate or heavy vegetation) separately. We ordered the points and tried to divide them into 9 equal frequency classes. With distance from vegetation, we used 3 classes because most values (169) were zero.

Habitat availability: Using the results from the summarization of the pike tracking data, we used the inputs to the SPANS database to create contour maps with the same class intervals. We did an area analysis of each map to determine the habitat availability.

Habitat suitability model: For each of the model variables, we then used the percentage of area available to estimate the number of pike sightings expected at random. The observed:expected frequency ratios were taken as measures of preference and scaled against the maximum value to produce HSI values between 0 and 1.

The HSI values were used to redefine the habitat availability maps. Finally we took the geometric mean of the three maps [map $\mathrm{D}=(\operatorname{map} \mathrm{A} * \text { map } \mathrm{B} * \text { map } C)^{* *}(1 / 3)$ ] to produce the composite habitat suitability map. If all three habitat components are necessary to determine suitability, the geometric mean should provide a useful measure. We arbitrarily selected three cut-off levels to obtain a map with three classes of suitable pike habitat (Good, $0.5-1.0 ;$ Fair, $0.1-0.5$; Poor, $>0.0-0.1$; and None, 0.0 ). This classification is similar to that described by Inskip (1982).

## Scenarios

Current: With present water quality conditions the vegetation is limited to depth less than 2.5 metres. We used the database as derived to estimate the area of suitable pike habitat.

Future: To project changes in suitable pike habitat under improved ecosystem conditions, we considered two possibilities: first, that the maximum depth extent for vegetation would increase to 4.5 metres and, second, that the areas now identified as having sparse vegetation would have moderate or heavy vegetation in the future. This scenario involves replacing the current vegetation suitability map with one based on expanded vegetation in the calculations.

## RESULTS

Pike habitat suitability indices: The frequency summaries of the pike tracking observations showed the pike selected narrow ranges of depth, distance from shore, and distance from vegetation (Table 1). For depth, all the points were between 0.8 and 12.0 metres with the majority between 0.8 and 2.1 metres. For distance from shore, all the points were within 300 metres of shore with most within 120 metres. All the distance from vegetation points were within 200 metres of moderate or heavy vegetation with the majority located within vegetation patches. Except for vegetation where the pattem is similar, the habitat selection in Hamilton Harbour is narrower than that reported for pike by Chapman and Mackay (1984) and Diana et al. (1977).

The depth, distance from shore, and distance from vegetation maps in SPANS were contoured with the same class intervals as used for the frequency analysis and the available areas determined (Table 1). The proportional availabilities of habitat were used to estimate expected point frequency distributions and the ratios of observed to expected, scaled to a maximum of one, used to estimate habitat preference, i.e. suitability (Figure 4.). The highest depth suitability is in the 1.5-1.6 metre interval with steep declines on either side. The distance from shore suitability is highest in the 50 to 60 metre interval. The irregularity on either side indicate a broad band of higher suitability in the 0 to 120 metre range. With vegetation, the suitability was confined almost entirely to within vegetation. Rather than attempt any smoothing of these suitability curves, we used the these curves for modelling purposes.

Suitability modelling: To complete the analysis of habitat suitability, we attached suitability values to the class intervals on the three maps and computed the geometric mean of suitability values. Using the class intervals selected for the resultant map, we computed the area in each class for the two scenarios (Table 2). Based on current vegetation patterns, we estimated there were 45.6 hectares of good habitat and 62.1 of poor. Expansion of the vegetation area roughly doubled the good area to 94.6 hectares and the poor to 136.5. In the west end of the Harbour, an area with relatively less disturbed and wind-sheltered shorelines and much of the natural onshore woods and brush, the potential habitat improvements are obvious (Figure 5).

## DISCUSSION

In the process of linking habitat suitability models and GIS to predict the amount of area available to pike in Hamilton Harbour, there are four aspects to be considered: the limitations of the databases, the limitations of the models, the predictions themselves, and the broader potential of approaches like this. At present, we are engaged more in the development of applicable methodology than in the operational application of models and GIS to habitat management.

Database limitations: There were substantial problems with our input database. The point locations for pike obtained by triangulation were not accurate enough for use in the SPANS database. The distances from shore and vegetation in the pike tracking data are based on visual estimates. The pike observations were not uniformly distributed in time or between individuals. The vegetation map was not based on a complete areal survey and only represented one point in time. The depth and shoreline data were derived from a map which may not register with the data collected independently.

Fisheries ecologists have not traditionally been concemed with knowing the exact coordinates of their sampling locations as they have assumed samples were independent. Clearly, if GIS-based approaches to fish habitat management are to succeed, they must pay more attention to where they are when they lift a net or sample water quality.

Model limitations: Regardless of the database limitations, there are structural and conceptual issues relating to the construction of habitat suitability models. The suitability indices used for Hamilton Harbour are unlikely to be universally applicable. Doubtless other factors, both abiotic and biotic, influence the shapes of the curves. In this instance we have assumed the three factors, depth, distance from shore and distance from vegetation are independent. The maps can be used to determine the joint occurrences of classes of the 3 factors and used as null hypothesis with the point observations of pike habitat selection and the independence of the factors tested. For example, the depth preferences may be set by the distribution of vegetation. Variations in the densities of pike and/or other fish species and prey organisms may influence suitability curves: higher densities of competitors may force more fish into marginal areas or differential gradients in food supply may modify the pattems. Only derivation of suitability models from a wider range of conditions, and perhaps through experimentation, will allow more comprehensive models to be developed.

Taking the geometric mean of three maps to produce a composite suitability map is only one of many algebraic means of combining indices and maps (Terrell et al. 1982). We expect that further model development is this field will lead to rational criteria for the combining of indices.

Predictions of adult pike habitat in Hamilton Harbour: The results showed a doubling of fair and good suitability habitat under an optimistic increase in the area of heavy and moderate vegetation. By assuming that vegetation occurs in all parts of the Harbour between depths of 0.5 and 2.5 or 4.5 metres, we have overestimated the area of habitat. The depth ranges should only apply to unstructured shores such as exist in the west end of the Harbour. Shores with breakwalls reflect wave energy and cause a wider zone without vegetation close to shore. If an additional map were prepared showing the shoreline type, we could include this factor in the suitability modelling.

In the introduction we indicated results like these can be used to estimate productive capacity. Inskip (1982) reports an optimal adult pike biomass in summer habitat of $20 \mathrm{~kg} / \mathrm{ha}$, where optimal habitat is assumed to have an HSI value of 1 . Applying the $20 \mathrm{~kg} / \mathrm{ha}$ figure scaled by the class mid-point suitabilities in Table 2 ( $20 *$ HSI*Area $=$ Pike kg ), we obtain an estimated 1700 kg of pike under current conditions and 2800 kg with expanded vegetation. Of course, this is an incomplete estimate for the Hamilton Harbour RAP site. An adjacent wetland-marsh area, Cootes Paradise, formerly supported large numbers of pike and is targeted for restoration under the RAP. The nearshore areas nearby on Lake Ontario would also contribute to Harbour pike production in other seasons. Nonetheless, this analysis does demonstrate the potential for linking HSI models and GIS to estimate productive capacity of fish habitats.

Role of models and GIS in fish habitat management: The need to quantify fish habitat is not confined to RAP sites on the Great Lakes. Quantification is needed for all fish habitat in Canada. Detailed inventories of fishery resources cannot be pursued in all locations. Habitat suitability models provide a means of estimating productive capacity with simpler variables, many of which can be derived from maps. The marriage of fish habitat models and GIS-based maps will allow a more flexible and timely analysis of fish habitats. GIS databases are being developed by numerous agencies for many situations and many will contain features which can be used in the assessment of environmental issues in general and fish habitat in particular.

## CONCLUSIONS

- Habitat suitability models can be used in conjunction with GIS databases to assess fish habitat and to estimate productive capacity.
- Fisheries ecologists need to be more concerning with accurate geo-referencing if they want to take advantage of GIS maps assembled for other purposes.
- More research will be required to develop more flexible HSI models and methods for joining indices.


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Table 1. Classification of pike observations by depth, distance from shore, and distance from moderate or heavy submerged vegetation, into similar frequency classes and estimates of the available area of each class in Hamilton Harbour.

| Variable | Class interval | N | Area ha |
| :---: | :---: | :---: | :---: |
| Depth (m) | 0.8-<1.1 | 28 | 22.0 |
|  | $1.1-<1.3$ | 28 | 25.7 |
|  | 1.3 - <1.4 | 20 | 10.3 |
|  | 1.4 - <1.5 | 14 | 10.0 |
|  | 1.5 - < 1.6 | 25 | 8.4 |
|  | 1.6 - < 1.8 | 34 | 14.5 |
|  | 1.8 - <2.1 | 21 | 17.0 |
|  | 2.1 - <3.0 | 26 | 41.5 |
|  | 3.0 - <12.0 | 27 | 555.1 |
|  | Sum $=223$ |  |  |
| Distance from shore (m) | $0-<10$ | 25 | 49.3 |
|  | 10-<20 | 24 | 39.2 |
|  | $20-<30$ | 21 | 37.2 |
|  | $30-<50$ | 25 | 68.6 |
|  | $50-<60$ | 25 | 30.1 |
|  | $60-<80$ | 20 | 56.9 |
|  | $80-<100$ | 20 | 54.5 |
|  | 100 - < 120 | 36 | 52.2 |
|  | 120->300 | 29 | 417.1 |
|  | Sum $=225$ |  |  |
| Distance from vegetation (m) | 0 | 169 | 64.2 |
|  | >0 - <40 | 26 | 54.2 |
|  | 40 - <200 | 25 | 175.9 |
|  | Sum $=220$ |  |  |

Table 2. Estimated areas (ha.) of adult pike summer habitat in three suitability classes using current conditions and an expanded vegetation scenario based on the geometric mean of depth, distance from shore, and distance from vegetation suitability maps.

|  |  |  | Suitable Habitat (ha.) |  |  |
| :--- | :---: | :---: | :---: | :---: | ---: |
|  |  |  |  |  |  |
| Scenario |  |  |  |  |  |
|  | HSI | None | Poor | Fair | Good |
|  | Range: | 0.0 | $>0.0 .1$ | $0.1-0.5$ | $0.5-1.0$ |
|  |  |  |  |  |  |
|  |  |  |  |  |  |
| Current vegetation | 1233.9 | 676.9 | 62.1 | 45.6 |  |
| Expanded vegetation | 1233.9 | 553.5 | 136.5 | 94.6 |  |



Figure 1. A shoreline map of Hamilton Harbour showing the points where depths were digitized, and the $0.5,2.5$, and 4.5 metre depth contours.


Figure 2. The shoreline of Hamilton Harbour showing the 200 and 500 metre contours of distance from shore.


Figure 3. Estimated extent of four classes of nearshore submerged vegetation in Hamilton Harbour between the 0.5 and 2.5 metre contours after combining absent with sparse and moderate with heavy.


Figure 4. Habitat suitability index curves for adult pike in Hamilton Harbour: (a) depth, (b) distance from shore, (c) distance from heavy or moderate submerged vegetation.


Figure 5. Predicted suitable adult summer habitat in the west end of Hamilton Harbour, based on the geometric mean of depth, distance from shore and distance from vegetation suitability indices, given: (a) current and (b) expanded vegetation patterns.

Synthesis of Sessions I to IV

## Session I

## factors affecting productive capacity of fish habitat

The initial part of this session advises on the specific habitat attributes that CAFSAC was requested to provide advice on. These attributes are: substrate, stream width, cover, velocity, ice scour, stream order, turbidity, total dissolved solids, pH, winter temperature, summer temperature, discharge and gradient. Additional habitat attributes which may be appropriate to record, and the circumstances when they are likely to be important, follow. A tabulation of points (Synthesis Tables 1 and 2) highlighted in presentations at the meeting is included, both to begin systematizing present knowledge, and guide in planning research to address gaps in current understanding. The session concludes with commentary on the topic.

## 1. Usefulness of specific attributes

## Substrate

Substrate is important, for spawning, for rearing, and as a link to seasonal patterns of habitat use. Substrate is a complex attribute with many properties. Grain size and permeability were noted as particularly important for characterizing Atlantic salmon habitat, but other aspects are relevant as well. Although substrate should be recorded routinely, there was no uniformity in how it has been measured by various biologists. This lack of uniformity impedes the ability to combine data from different studies, and to generalize results across studies and areas. There is a definite need to review methods used to quantify substrate and designate appropriate standards.

## Stream Width

Stream width was very important in many models discussed. The effect of stream width on Atlantic salmon is not direct, but width is a convenient surrogate for many other habitat attributes. It reflects many other features, notably discharge, depth, and cover levels. The shape of the relationship between stream width and productivity is not well known, and should be studied further. In Newfoundland, lacustrine habitats are important to Atlantic salmon. For these lakes either perimeter of the lake or width of the littoral zone are surrogate measures comparable to width of streams.

## Cover

Cover is an attribute with many components important to Atlantic salmon (see example flow diagram in Synthesis Table 2). Canopy cover should be quantified, because of its effect on water temperature. Instream cover should be measured, because of its direct function as habitat for juvenile stages. Overhanging cover and debris should be measured, because of its role as habitat for other species, which may interact with salmon and affect productivity. The depth of water itself is cover. All types of cover, and the surface turbulence some of them create, influence stream production, with important implications for many aspects of Atlantic salmon biology. There is a definite need to review methods used to quantify cover and designate appropriate standards.

## Velocity

Velocity is tied to the general hydrology of the stream. Stream velocity assists adults in redd construction, juveniles in feeding, and has other indirect effects. Its effect is dependent on many other factors, notably substrate. The mean velocity has some importance, but the salmon are exploiting the microhabitats of velocity distributed through the water-column. It is difficult to measure the distribution of microhabitats, and the aspects of it which are easier to measure are largely captured by width, depth, substrate and gradient. It is important to note whether water velocity is above some minimum (moving, not stagnant), but beyond that, only intensive site-specific studies can quantify it usefully.

## Ice Scour

Ice scour is a surrogate for range of discharge of the stream. Height of the scour mark on U-shaped stream banks reflects the indirect effects of variance in discharge levels. Ice scour marks on the sides of a flatter stream bed indicate direct disturbance of redds and substrate. Debris line and lichen growth limits may also serve as surrogates for processes reflected by ice scour height. Disruption of stream bed, spawning, and juvenile overwintering habitat by ice scouring can severely affect production of salmon. There is a definite need to review methods used to quantify ice scour and designate appropriate standards.

## Stream Order

Stream order serves as a surrogate for organizing variation in discharge, width, depth, cover and other stream attributes. The individual attributes are not likely to follow stream order closely, so measures of the attributes are more desirable than measures of stream order. Stream order does not properly capture the role of lakes in drainage basins. Moreover, its organizing property is not transferrable across regions, therefore any relationships with
productive capacity are difficult to evaluate and use. Stream order is a good and convenient organizer of stream properties intraregionally, however, and can facilitate communication among professionals. In these contexts it can be useful.

## Turbidity

Turbidity was not addressed directly in the presentations. By itself, intraregionally, turbidity is unlikely to show much contrast, or to be an important predictor of features of Atlantic salmon biology. It can reflect major anthropogenic habitat degradation, however, with increasing turbidity indicating likely increases in suspended solids which may lead to losses of spawning and feeding habitat, decreased egg survival and decreasing productivity. Turbidity due to estuarine dredging deserves special consideration because of potential interference with migration, early marine feeding, and subsequent recruitment. As noted in section 2 (other habitat attributes), direct measures of levels of suspended solids are preferable to turbidity.

## Total Dissolved Solids (TDS)

This was not directly addressed in presentations. In Atlantic Canada TDS is generally measured as conductivity, and conductivity was shown to be a poor predictor of aspects of Atlantic salmon biology. As noted in section 2 (other habitat attributes), measures of levels of specific chemicals are preferable to TDS.

## pH

pH has a role in measuring existing anthropogenic disturbance, with direct effects on productive capacity through effects on survival of life history stages. It also affects species composition of the invertebrate fauna. This could have effects on feeding, but studies have shown that when densities are reduced by low pH, surviving juvenile salmon may grow faster. If pH is low, it should always be used in conjunction with measures of dissolved organic carbon and specific metals, especially calcium and aluminum. Appropriate methods to measure pH are presented elsewhere in this report and should be used.

## Winter temperature

Winter temperature can affect egg and fry survival directly, and indirectly through effects on discharge. Cold winters lead to accumulation of acidic deposition which is released in a strong spring pulse, also with detrimental effects on salmon. Mild winters with frequent snowmelt lead to episodes of low pH lasting throughout the winter months, often with serious adverse effects on survival. Winter temperature can be a surrogate for many other factors which influence salmon survival as well, such as extent of frazil and anchor ice. Even
in cases when anthropogenic activities are unlikely to alter winter temperature, it must be recorded at least as a covariate, to interpret the results of any site-specific studies. The winter temperature regime must be considered, not just a seasonal extreme, because of the many ways in which winter conditions interplay with salmon life history.

## Summer temperature

Summer temperature has many important effects directly on growth, physiology, and mortality (upper lethal limits), and indirectly through effects on water chemistry, oxygen levels, primary production, and levels of potential predators and prey. Summer temperature should also be represented as a regime, but it was suggested that midsummer median and variance may predict aspects of salmon biology. Summer temperature regime can be impacted by many anthropogenic activities, including forestry and altering hydrological regimes. For both winter and summer temperature, studies are required to determine which aspects of the temperature regime best capture its relationship with aspects of salmon biology.

## Discharge

Discharge is definitely important. It can have direct effects on productive capacity at extreme drought or flood levels and on migration. Discharge also acts indirectly through interactions with other habitat attributes such as width, velocity, pH , oxygen, and other chemical parameters. The regime of discharge is the important feature for these indirect processes, and single estimates of discharge are of limited value.

## Gradient

Gradient is a surrogate for substrate and velocity. It can be important for extrapolating from individual sites, and can be obtained from remote sensing. It can be used to provide initial reference population estimates and for screening cases. However, at specific sites other habitat attributes will usually be required to assess the full impact of anthropogenic activities.

## 2. Other habitat attributes

A number of additional factors were identified that can affect or indicate productive capacity of Atlantic salmon habitat. The additional factors are grouped as physical, chemical, and biological, but are not listed in any particular order of priority.

## Physical:

- Depth (mean and maximum) are important dimensions.
- Ice and snow regimes (type, thickness, movement) both relate to winter habitat needs. They are linked to climatic and flow changes (and cover).
- Lacustrine dimensions (littoral/non-littoral, euphotic zone, benthic/pelagic, morphometry) influence juvenile distributions.
- For drainage basins, the ratio of lotic/lentic habitat area may indicate potential productivity. Distance from the river mouth or head of tide may influence the distribution of fish. Terrestrial conditions in the drainage basin (forest cover, soil temperature regimes, etc.) can be important.
- Suspended solids should be measured directly.
- Dissolved gases: Dissolved oxygen levels are determined by temperature conditions and turbulence/flow/gradient in most instances but might be measured in certain situations, for example, within substrates where egg survival may be threatened and where man-made releases reduce oxygen levels.


## Chemical

A set of characteristic habitat indicator groups (with some overlap) was identified:

- Productivity: There are several indicators including nutrients such as total phosphorus, total nitrogen, nitrogen oxides, ammonia, particulate (and dissolved) organic carbon (related to seston) and ionic species such as pH, alkalinity, calcium, and magnesium. Measures such as total dissolved solids, conductivity and hardness were deemed less appropriate because they are nonspecific.
- Terrestrial/Wetland/Marine: Variables which reflect natural effects in the drainage basins include organic acidity measured as humic acids, dissolved organic carbon, or colour (in an order of decreasing preference), and marine effects preferably using chloride or sodium.
- Anthropogenic: There is a selection of chemical measures reflecting man-made inputs. Sulphate is a proven indicator of acidification due to long range transportation of air pollutants (LRTAP). Other LRTAP contaminants, organic, and inorganic may be present and guidelines exist with recommendations for sample selection (water, sediment, biota, etc.), sample processing, and analysis. Local chemical releases can also be assessed. Use of these variables should be limited to suitable contexts given the expense of measuring most of the contaminants.

The information highlighted the potential importance of other fish species and of both stationary and drifting benthic invertebrates.

- Other fish species: The richness of the fish assemblages varied across the regions, and the habitat available for Atlantic salmon may be reduced where other species are present or more abundant. Further, because the habitat policy refers to all fish and fish habitat, other species will need to be considered in the broad context of net gain or loss of habitat.
- Benthic invertebrates: Benthic biomass and numbers were linked to fish abundance. Species assemblage patterns were related to habitat conditions (temperature, pH , discharge, etc.). Consideration of benthos may be restricted to major impact assessments and research studies because of the costs involved. Benthos, particularly as drift, are undoubtedly important determinants on productive capacity in streams. Lake invertebrates are also important.


## 3. Commentary

The complexity of Atlantic salmon life-history means many additional attributes of habitat may be important influences on productive capacity, but the relationships are not quantifiable with present knowledge. The complex life-histories and interactions with habitat also mean no simple research program can be implemented to place the Department of fisheries and Oceans in a position to advise on all the habitat-related requests it may receive. Many relationships we know or suspect to be important, and possibly vulnerable to anthropogenic impacts, cannot be quantified with anything short of large-scale site specific studies. Although research will continue to explore surrogates for these relationships, the advisory process on habitat issues must be robust to the serious limitations inherent in the use of surrogate variables for complex relationships.

Biological processes are interactive in many cases. The biological significance of one habitat attribute often depends on the values of other habitat attributes. Measuring single variables and relating them to productive capacity is better than no evaluation at all, but does not allow for the interactions. The interactive nature of habitat features must be addressed by work arising from sessions 2 and 3 (models and methods) and from this session (1) (attributes), where new variables must be developed which reflect complexity without losing specificity.

In light of the point made about the limitations of our knowledge, and the need to use what we do know efficiently, it is important to systematize our knowledge base, and make it operationally available. The development of tables of life history interactions and interconnections with habitat attributes is one avenue proposed. The tables would provide:
(i) guidance on the appropriate selection of habitat factors
to be measured in relation to the scope and intensity of the data collection, for example, single-visit habitat inspections through routine habitat and fishery monitoring/assessment activities to large-scale environmental assessments and research;
references to recommended/appropriate measurement methods; and
references to documentary evidence of factor-life stage interactions.

The tabular approach could be embedded into a biogeographic framework to ensure that the guidance was relevant to regional and local conditions. The type of impacts or habitat modifications could be included in the framework. Synthesis tables 1 and 2 provide a sketch of the intended approach. Synthesis Table 1 indicates an overall scheme, Synthesis Table 2 examines cover attributes and their significance to salmon life stages and other species. These tables are offered as an indicator which will draw present knowledge into a form which is operational. There is a need to complete the tabulation for other attributes and to develop research programs which will fill in the most serious gaps in the tables.

Specific deficiencies in our knowledge and data on all aspects of habitat-stock relationships were identified. These deficiencies were noted to be particularly acute for:

1) winter habitat requirements;
2) movement of life-history stages among habitats;
3) how influences of different habitat attributes interact;
4) the role of nutrient levels in the water on life-history attributes;
5) use of estuarine and marine habitats by juvenile salmon.

In addition, statistical methodologies and standards have to be identified and accepted for assessing the adequacy of sampling of habitat and stock attributes, in the contexts of geographic and spatial resolution, desired levels of precision, and appropriateness for discriminating areas used and areas preferred.

## Session_II

use of habitat suitability curves and other models to estimate changes IN PRODUCTIVITY OF FISH HABITAT

This session dealt with models which can be used to describe the relationship between certain measured or estimated habitat parameters and habitat usage. The session focussed on habitat suitability curves (HSCs), but did not deal with them exclusively.

It was noted that large quantities of data are collected for other purposes (e.g. stock assessment) and could be used to create HSCs if habitat attributes were collected concurrently.

HSCs link single attributes (e.g. stream width) to frequency of usage, and such curves are increasingly being used as estimators of availability of usable habitat. However, the habitat variables may not turn out to be those best suited to assessing the gain or loss associated with a particular type of alteration. Setting up the HSCs is a rather subjective process.

Subjectivity might be considered a strength in that it allows knowledgeable specialists to bring elements of their experience to bear on the process. Subjectivity may equally be a source of potential error, and in the worst cases could actually be misused to provide a desired outcome.

HSCs developed for juvenile Atlantic salmon in a disturbed stream were quite different from curves developed in nearby undisturbed rivers. Thus HSCs derived in one area may not apply in another. Under suboptimal conditions, the primary habitat variable for choice of the fish's position in the stream and the threshold response level of that variable may change.

When the habitat used by marked salmon (juvenile coho and chinook) was observed under three (controlled) discharge rates, the HSCs developed for coho were significantly different at different discharge rates. (The chinook HSCs also differed by similar amounts, but probably sample size was too small for significance.) It would appear then, that if the HSCs are not known to be consistent over the range of the attributes being affected by the proposed stream alteration, then the extent of variation in fish habitat preferences should be investigated before application.

HSCs in common use are generally not true representations of fish habitat preferences, but only of habitat usage. To construct a true HSC one has to establish the population of possible habitats that is available and compare that with the habitats actually selected.

## 2. Instream Flow Incremental Methodology (IFIM)

The application of an IFIM model (with HSCs) to assess the impact of a major water diversion on Atlantic salmon habitat in a Quebec river, was reviewed. The models used indicated that although natural flows will be reduced, by guaranteeing minimum flows, it would be possible to maintain and, in certain periods of low discharge, to actually improve habitat conditions after the diversion. Doubts were expressed as to the reliability of the quantitative estimate, but no better model was proposed for assessing this type of alteration.

For the Nechako River, B.C., modelling exercise IFIM was run for
each of nineteen independently produced sets of HSCs. The nineteen sets produced estimates of maximum habitat availability so variable that no decision could be justified on the basis of the results.

## 3. Probability Density Estimation (PDEs) Methods

PDEs have the advantage over HSCs of objectivity and over regression models of making no assumptions about the shape of the response curve. PDEs also have the advantage of providing a built-in error estimate with every prediction. Several attributes can be objectively combined with this type of model. Although promising, effort will have to be made to make the technique user-friendly.

## 4. A Dichotomous Model

A dichotomous model (a series of comprehensive and well thought out yes/no questions) was developed for assessing lake trout habitat in the Great Lakes. This approach could be adapted for Atlantic salmon. Model development and testing would probably require several years. The output, however, might be more appropriate for information purposes (Tier II) and whether the model is developed or not, the relevant research still had to be done.
5. Habitat Evaluation Procedure (HEP) Models

The HEP model process is a way of combining several habitat variable use functions (usually, but not necessarily HSCs). Its flexibility allows for a great deal of subjectivity, which can alter the final output of the model. HEP models are not transferable, require calibration at each application, and when tested against independent data sets have always failed to make accurate predictions.

## 6. Conclusions

There is a great deal of unconsolidated habitat information available for Atlantic salmon, and a number of models are available, each with different strengths and weaknesses. For example, IFIM was developed to assess the impact of changes in stream flow, and it would be quite inappropriate for assessing the impact of acidification.

Univariate models like HSCs, or those involving samples at a single point in time can be used with greater confidence when only one habitat factor is being investigated. Considerable care has to be exercised when several suitability curves are combined to get an estimate of "net effect" on habitat productivity.

The HEP approach was devised to assess complex multiple habitat changes, but it is probably inappropriate for quantifying a projected gain or loss of Atlantic salmon habitat productive capacity. PDEs may have considerable advantages over HSCs, because they are more objective
and easier to combine, and over linear regression models in that no simplifying assumptions are required concerning the shape of the response curves.

It is apparent that a single model cannot be employed in every situation, and that DFO Science is in an early phase as far as habitat model development is concerned.

It was concluded that at this time no specific approach can be recommended for development of advice on habitat issues, so advice must be developed using whatever knowledge and expertise can be assembled for a specific application. Efforts must be made to develop, test and modify methods which are appropriate for use, and to determine limits to their applicability. Problems which require special attention include interactions among habitat attributes, and linear constraints in current methods.

## Session III

methods to measure factors used to estimate productivity or changes in PRODUCTIVITY

1. Introduction

As was indicated in Synthesis Session 1, the choice of attributes to be measured is important and relates to the type of investigation. Equally important, however, once a series of attributes to be measured has been decided on, is the selection of methods used to measure these attributes. The information presented did not, in many cases, focus on the detailed methods used as, quite often, there are routine and accepted procedures used. Of concern is the occasional lack of consistency or comparability between data derived using differing methodologies. Results derived must be comparable between investigations in both time and space, this often requires functional relationships to be developed between techniques of differing rigour. For example, while some studies estimate the substrate composition quite subjectively, others use more quantitative techniques. A methodology is needed to "translate" between different techniques.

For monitoring purposes, consistency can only be attained if all staff involved (scientists, habitat managers, and fishery officers) are trained. As well, contracted research and/or sampling (by consultants) must adhere to the same standards if data are to be comparable. Careful review of proposed methodology by contractors is warranted.
2. Quantification of habitat

## Remote sensing

The first, and most basic, requirement of habitat managers is to know, quantitatively and accurately, the extent of physical freshwater habitat available to Atlantic salmon. In the past, distribution of juvenile Atlantic salmon within streams has been associated with a variety of physical attributes such as water depth, water velocity and substrate size. An alternative quantitative methodology is based on a combination of remote sensing (1:10,000 ortho photographs with 5 meter contour intervals) and ground-truthing. This method could quantify habitat area, using stream gradient as the criterion of suitability. Reaches with 100\% canopy were omitted from the estimates, thus a small percentage of suitable salmon habitat was probably omitted. This gradient-based methodology could be readily applied to other rivers and regions. This method could be attempted in other areas of Atlantic Canada. In addition, the usefulness of incorporating the data obtained into the Geographic Information System (GIS), a computer based data storage and retrieval system with habitat mapping capabilities, should be investigated.

## pH

The standardized methodology summarized by McCurdy and Lacroix (this volume) should be used in all areas where pH measurements are taken.

## Other Habitat attributes

From the information presented, inconsistencies were noted in the quantifying of many habitat attributes. Some physical attributes, for example, substrate, were estimated subjectively and, as previously stated, there is a need to standardize this measurement among study areas, probably by calibration to more objective methods. Consultation should occur with hydrologists/geologists regarding appropriate techniques for substrate assessment and dynamics.

To achieve consistency and quality of data collected by habitat managers, fishery officers and scientific staff, a manual should be developed outlining the specific methods to be used in the measurement of important attributes identified in Synthesis Session 1. Such a manual should include a key for investigation of various types of habitat disturbance to assist field staff in choosing the type of attributes to measure and the sampling intensity required. Subsequent to the development of such a manual, regional workshops would be required to instruct in its application. Field staff should also be provided with appropriate sampling gear to measure required parameters and collect water samples for chemical analysis. Periodic cross checks, among samplers should be implemented to assess staff compliance with instructions.

Periodic review of methods used to measure habitat attributes would also be necessary, methods should be updated and calibrated as new techniques/ technology become available.

Session IV
techniques to measure standing stock

Selection of appropriate methods for estimating standing stock of juvenile Atlantic salmon depends on the type of habitat being surveyed. Over the geographic range of Atlantic salmon in eastern Canada, there are two basic types of habitat which are important as nursery areas for juvenile Atlantic salmon-fluvial habitat (streams and rivers), which is the primary habitat type in most areas, and lacustrine habitat, which is characteristic of many drainages of insular Newfoundland, Labrador and the Quebec North Shore. A variety of methods to estimate standing stock for various types of habitat are required.

## 1. Fluvial Habitat

For fluvial habitat, appropriate techniques and survey designs for estimating salmon abundance depends on the objectives of the study. If information on salmon density is needed within selected study areas, reasonably accurate estimates can be made using electrofishing gear and either removal or Petersen mark-recapture methods. Either method is appropriate if fundamental assumptions about capture probabilities apply; tests of these assumptions should be an integral part of the survey design and application. Population estimates with coefficients of variation less than 0.20 are often attainable. Standardized guidelines for electrofishing practices at stream sites should be developed.

If the objective is to estimate the total standing stock of juvenile Atlantic salmon in large areas (eg. an entire stream or drainage), a two stage sampling design is theoretically required. For the first stage, a number of sampling sites are selected, and for the second stage, population estimates are made within the selected sites using the removal or mark-recapture estimators. Because the spatial distribution of juvenile salmon within streams is contagious, variances associated with the first stage of sampling (i.e., variance in estimates resulting from expansion of sampling sections to an entire stream) are significantly higher than variances associated with the second stage of sampling (estimates of numbers within the study sections). Selection of study sites by simple random sampling is usually not advisable because too many sites are required to estimate total salmon abundance with reasonable precision. More accurate population estimates are possible if stratified random surveys are conducted, where strata boundaries are chosen by identifying areas of high and low salmon densities.

To date, a stratified sampling design to estimate the total standing stock of juvenile salmon has been tried in only one (Nova Scotia) watershed. Stratification was based on two variables, gradient and distance of the habitat from the head of tide. The efficacy of the survey design has yet to be tested.

Estimates of juvenile standing salmon stock can also be inferred from counts of migrating smolts. Total smolt runs can be estimated using counting fences, or by using mark-recapture techniques. Counts of smolts, however, do not provide information on the population dynamics or biomass of individual age groups before smoltification, or on localized habitat effects on survival. Whether or not smolt counts are suitable depends on the objectives of the survey.

If the objective is to monitor temporal changes in juvenile salmon density, surveying the same sites annually may provide an index of abundance.

## 2. Lacustrine habitat

Various techniques have been used to estimate the abundance of juvenile salmon in lacustrine habitats of Newfoundland. Estimates of salmon abundance, using various gear types to conduct mark-recapture experiments within the ponds, or by counting smolts at fences on outlet streams, have been made. Studies of fish distribution within the lakes are necessary to provide specific information on the temporal and spatial habitat usage by individual age groups. Lacustrine standing stock has been calculated as the difference between total standing stock and that estimated for fluvial habitat. The difference between spring and fall population estimates has been used as an index smolt abundance.

It was concluded that research in the following areas should be carried out to provide better methods for estimating standing stock of juvenile Atlantic salmon in various types of habitats:

1. Current sampling designs for estimating standing stocks of juveniles should be reviewed;
2. Usage of specific lentic habitats by juvenile salmon should be further defined;
3. Extent and causes of variation in standing stock of salmon among lakes should be investigated;
4. Appropriate physical units for describing productivity of lacustrine habitat should be determined.


## SYITTHESIS TABLE 2. PLOM DIAGRAM OP VARS SIENTPICANCE TO ATLANIIC SALMON LIPE BISTORY STAGES




[^0]:    ${ }^{1}$ Smith-Root GPP 3.5, Smith-Root Inc. Vancouver Washington.

[^1]:    ${ }^{1}$ R.W. Dunfield and R.K. Sweeney, Department of Fisheries and Oceans, P.O. Box 550 Halifax, N.S.

    2 Land Registration and Information Systems, Surveys and Mapping Division, Summerside P.E.I.

[^2]:    a. Area derived using some interrpolated widths.
    b. Grade based on sub-sample of total area between contours.
    c. Not a contour to contour reach.
    d. Above remote-measurable widths.
    e. Above remote survey limit.
    $f$. Survey direction downstream.
    $g$. Does not include grades $>5.0 \%$.
    h. Includes grades $>5.0 \%$.

