ASSESSING THE IMPACTS OF STREAM HABITAT AND LAND USE VARIABLES ON POPULATION AND COMMUNITY STRUCTURE OF SALMONIDS OF PRINCE EDWARD ISLAND

A Thesis

Submitted to the Graduate Faculty

in Partial Fulfilment of the Requirements

for the Degree of

Master of Science

in the Department of Biology

Faculty of Science

University of Prince Edward Island

Karen L. Gormley

Charlottetown, P. E. I.

July, 2003

Your file Votre référence
ISBN: 0-612-93856-5
Our file Notre référence
ISBN: 0-612-93856-5

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ABSTRACT

The farming of potatoes, the main agricultural crop on Prince Edward Island (PEI), leaves land susceptible to erosion, and uses large amounts of fertilizers and pesticides. These chemicals can enter surface waters through runoff or through ground water (which supplies surface water) through leaching. To determine the impact of land use practices and run-off events on salmonid communities, I examined salmonid populations (Atlantic salmon, *Salmo salar*, brook trout, *Salvelinus fontinalis*, and rainbow trout, *Oncorhynchus mykiss*) at twenty-seven sites on nine rivers located across Prince Edward Island throughout the summers of 2001 and 2002. Multiple regression models were developed using various stream and watershed characteristics to predict biotic variables (density, percent habitat saturation, and condition factors) and stable isotope values. Five rivers were sampled in 2002 to use in a cross-validation study to test the models. Six models were developed (brook trout density, variability in total density, total percent habitat saturation, variability in total percent habitat saturation, 1+ brook trout condition factor, and $\delta^{15}N$ values). Although all models explained a significant proportion of the variability in the nine original rivers (r^2 values ranging from 0.374 to 0.985), only that describing $\delta^{15}N$ values proved valid when tested in the cross-validation study.

At least forty-two pesticide runoff-related fish kills have occurred on PEI since 1962. Two pesticide runoff events resulting in the death of thousands of salmonids occurred 9 July and 19 July 2002 on the Wilmot River, one of the nine original rivers. Populations of salmonids on 12 July at the affected site had

decreased by 40% from those of the previous year. Brook trout populations were over 80% lower than in 2001, while rainbow trout populations were higher. Populations were sampled on 20 July after the second runoff event, and a different site was affected. At this site, brook trout populations decreased by 98% from those seven days earlier, while rainbow trout decreased by 66%. Within species, 0+ fish were more affected than larger age classes. Five additional sampling sites were added after the second pesticide runoff event to better determine the extent and effects of the pesticide runoff. The populations were examined throughout the summer and into the fall (November/December). There was some movement of fish into affected areas; most of this increase was due to 0+ salmonids. These sites will also be monitored in the future to determine long-term effects and changes in community and population structure.

ACKNOWLEDGEMENTS

First, I would like to thank my co-supervisors, Dr. Kevin Teather and Daryl Guignion, for their extensive input into this project. Kevin, I greatly appreciate the time you took to go through the numerous drafts and revisions of this thesis. Daryl, thank you very much for the help you gave in selecting my sites and improving my general knowledge of salmonids of PEI (and teaching me how to fly fish!). My thanks also goes to my committee member, Dr. Donna Giberson, and her help with the stats aspect of this project, general limnological information, and numerous constructive criticisms during the first draft of this thesis. Thank you also for the use of your flow meter, solar pathfinder, and surveying equipment.

This project would not have been possible without the numerous people who helped me out with field work. Todd Dupuis, Daryl, and Kevin were a great help with choosing study sites, and came out numerous times to help me sample. Erin Duffy, Eddie Francis, Carrie Jardine, and Ben Moore were all wonderful to work with, and stayed cheerful (usually) even with the long hours, deer flies, and bubba fish. Lisa Purcell also helped a great deal with field work, as well as with many other aspects of this project. Thank you also to the many people who came out to help me sample: Melissa Abbott, Nancy Beaton, Craig Durant, Rachael Eedy, Nick Gautreau, Bonnie Gillis, Floyd Gormley, Nick Gormley, Dr. Doug Holdway (a.k.a. "the rock"), Neil Mater, and Ben Hoteling's Holland College Summerside students.

Thank you to the UPEI Department of Biology, and the generous use of various water chemistry testing equipment. Thank you very much to the technical staff: Kaye Ross for her help with keeping things organized and dealing with various aspects of the university, Gilbert Blatch for his help with the vans, Pat Doyle for his help with equipment, and Terrie Hardwick and Liz Rostant for their help around the department.

Thank you to Dr. Marva Sweeney Nixon, who agreed to take time out of her schedule and sit on my defence committee, as did my external reviewer, Dr. David Speare, who agreed to sit on the committee at such short notice. Thank you also to Dr. Wayne Peters, who represented the Graduate Studies Committee for my defence.

The initial funding for this project came from the Prince Edward Island Department of Fisheries, Aquaculture and Environment. In addition, the Department provided in-kind support of water chemistry testing (Water Resources Division) and use of some equipment (Fish and Wildlife Division). The Department of Agriculture and Forestry generously allowed the use of their GIS layers. Additional funding came through the Olin Fellowship (Atlantic Salmon Federation), and personal support was provided by a Natural Sciences and Engineering Research Council Post Graduate Scholarship-A, as well as the Margaret MacMillan Pratt Post Graduate Scholarship in the Sciences.

Finally, thank you very much to my family and friends who provided much-needed moral support throughout this degree. Mom, Dad, Nick, and Julie, thank you very much. Thanks especially to Nick and Dad for helping me out with

sampling and finding data loggers. The other graduate students at UPEI also helped make this a memorable experience.

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INTRODUCTION AND LITERATURE REVIEW

1.1 INTRODUCTION

There is increasing concern regarding populations of salmonids on Prince Edward Island (PEI). Atlantic salmon (*Salmo salar* Linnaeus) are declining (Guignion et al. 2002), as they are throughout much of their range (Parrish et al. 1998), and rainbow trout (*Oncorhynchus mykiss* Walbaum) are invading rivers and competing with native populations (Guignion et al. 2002). Brook trout (*Salvelinus fontinalis* Mitchill) are still widespread (Guignion et al. 2002) but, as with salmon and rainbow trout, populations have likely been affected by recent fish kills. These fish kills have been associated with intense agricultural practices (Mutch et al. 2002), which is the main land use issue on PEI as almost 40% of the province is in agricultural production (PEI DAF 2002). In addition to causing extensive fish mortality, intensive agriculture can have sublethal effects on fish populations. Pesticides, nutrients, and increased siltation in the rivers due to surrounding land use practices may affect fish growth, condition, or population size and structure (Platts and Nelson 1988, Bergstedt and Bergersen 1997, Peterson and Kwak 1999, Bradford and Irvine 2000, Wang et al. 2001). Changes to the landscape may also affect the food web, resulting in changes in fish stable isotope ratios (e.g., France 1996, Lake et al. 2001).

Although pesticide related fish kills have been studied in individual rivers on PEI (e.g., Johnston and Cheverie 1980), neither lethal nor sublethal effects of land use practices on salmonid populations have been extensively studied. This study examined the effects of land use practices on salmonid size, condition, population density, percent habitat saturation, and brook trout stable isotope ratios. In addition, the extended effects of two major fish kills on salmonid population and community structure were examined.

1.2 LITERATURE REVIEW

1.2.1 Effects of Land Use on Fish Communities

Salmonid populations are declining in many parts of the world (Parrish et al. 1998, WWF 2001). The factors contributing to this decline are not completely understood, but it is clear that human land use activities within the watershed may influence the health of fish populations by affecting various physical and biotic features of the aquatic habitat (Platts and Nelson 1988, Bradford and Irvine 2000). Understanding the relationships between surrounding land use patterns, stream characteristics, and fish community health is essential in predicting the influence of changes in land use on fish communities.

1.2.1.1 Assessing Fish Community Health

The health of fish communities may be examined at the individual, population, and community levels of organization. At the individual level, weights and body sizes of fish can provide valuable information concerning differences in growing conditions between rivers. Condition indices [(weight/length^3) x 100] have been particularly important in indicating the health of fish. Low condition factors have been found in rainbow trout, brown trout (*Salmo trutta* Linnaeus), and mountain whitefish (*Prosopium williamsoni* Girard) exposed to increased levels of sediment (Bergstedt and Bergersen 1997). Low condition factors have been linked to high mortality rates related to starvation in laboratory and natural populations of Atlantic cod (*Gadus morhua* Linnaeus) (Dutil and Lambert 2000), so are a valid tool in assessing fish health.

Measurements of individuals in long term studies are often used to provide information concerning general population structure. For example, Peterson and Kwak's (1999) investigation of smallmouth bass (*Micropterus dolomieu* Lacépède) in Illinois over a 24 year period allowed them to assess the stability of the population. Individual fish were measured and weighed, and age classes were determined by a combination of these measurements and scale analysis. The relative numbers in each age class were determined, and annual survival was calculated for each age cohort. The relative proportions of different age classes provides information on population dynamics; for a population to be stable, a certain proportion of the population must be young of the year and these fish must survive to the next generation (Megrey 1989). Tracking these age classes over numerous seasons is necessary to better understand population fluctuations, and may be important in implementing management plans if deemed necessary (Megrey 1989).

Rather than determining population numbers or density, some studies examine population biomass of the system (e.g., Chapman and Knudsen 1980). Biomass is estimated by multiplying the mean weight of a fish species or age class by its population density estimate for the area. The biomass of all fish species present can be summed to generate a biomass for the system. Biomass values are useful for determining the productivity of fish in a system (Mertz and Myers 1998), and may be divided by trophic group (e.g., Randall et al. 1993), species, and age class (e.g., Mertz and Myers 1998).

Biomass assumes that two small fish are biologically equivalent to a single fish that is twice their size. However, food consumption and metabolic rate are not linear relationships, so this assumption is not valid (Grant et al. 1998). Another method that has been developed to examine salmonid populations that takes these non-linear relationships into account is the percent habitat saturation (PHS). PHS assumes a salmonid will defend a certain-sized territory based on its size (fork length) (Grant and Kramer 1990). Although the territory size is inversely related to food availability, between 69 and 97% of the variation in territory size is explained by the size of the fish (Grant et al. 1998). Once the length of fish in the site is determined, it is multiplied by the density, and the amount, or percent, of the habitat being used is determined, with the equation:

$$PHS = 100 * \sum_{i=1} D_i * T_i \qquad (1)$$

where D_i is the density (number/m^2) and T_i is the territory size, determined from

$$\log_{10} T_i = 2.61 \log_{10} \text{fork length (cm)} - 2.83 \qquad (2)$$

(2.83 is a correction factor) (Grant and Kramer 1990). Age classes should be summed, as should different salmonid species, unless their territories do not affect one another's. This model is most valid when examining salmonid populations in shallow, riffle areas, as salmonids defend a two-dimensional habitat in these areas (Grant and Kramer 1990).

Another way of examining the health of fish communities is to calculate species richness, species diversity, and/or indices of biological integrity (IBI). Species richness refers to the number of species in the area (Randall et al. 1993), while species diversity takes into account both the number of species and the equitability, or evenness, of the populations of different species. The index of biological integrity incorporates variables such as species richness, species diversity, presence/absence of fish tolerant/susceptible to anthropogenic stressors, and condition factors, and is a good indicator of fish community health (Stauffer et al. 2000). Each of the categories is given a rating, and the sum of the ratings is the IBI score. The ratings may be based on a model developed for a particular region (e.g., Wang et al. 2001), or they may be used in relation to a reference site (e.g., Stauffer et al. 2000).

1.2.1.2 Assessing the Impact of Land Use

Investigations examining how surrounding land use may affect fish communities vary widely in the detail to which they subdivide land use activity. Wang et al. (2001), for example, examined 47 streams in southwestern Wisconsin and divided the land use types into sixteen different categories. However, it may not always be feasible or necessary to examine the land use in such detail. Often certain categories, such as road density, forested land, amount of logging, extent of urbanization, or amount of agriculture, provide reasonable indicators of land use (Chapman and Knudsen 1980, Hartman et al. 1996, Peterson and Kwak 1999). As these are expected to be most important on PEI, I examine these categories in more detail.

1.2.1.2.1 Road Density

Increased road density is associated with increased human disturbance in the area and, as such, is used as an indicator of land use in many studies of fish population or community dynamics. For example, coho salmon (*Oncorhynchus kisutch* Walbaum) smolt abundance in fourteen streams in western Washington was found to be adversely affected by the amount of human disturbance, indicated by road density, in the watershed (Sharma and Hilborn 2001). In the Thompson River, British Columbia, the rate of decline of coho salmon populations was related to the density of roads in the watershed, as well as to the amount of agriculture and urban land use (Bradford and Irvine 2000). In a similar study, Paulsen and Fisher (2001) examined the effects of land use on parr to smolt survival of chinook salmon (*Oncorhynchus tshawytscha* Walbaum) in the Snake River in Idaho and Oregon. Data were collected for up to seven years at twenty different sites. Results of this relatively long term study indicated that survival rate of salmon smolts was negatively correlated with road density; wilderness (undisturbed) areas were found to have the highest survival rate of fish.

1.2.1.2.2 Logging

The removal of trees in an area may have numerous detrimental effects on fish. Many aquatic food webs are based on leaf litter and other material entering the stream from the trees bordering the water; the loss of these trees can reduce the food availability in the system (France 1996). Trees adjacent to streams also provide cover for fish from predators and solar radiation (Jones M.L. et al. 1996). The removal of trees has been found to increase both the average temperature and the diurnal fluctuations in temperature of the stream, due to the decreased shading (e.g., Holtby 1988, Johnson and Jones 2000). Increased temperature reduces the oxygen carrying capacity of water (Cole 1994), which could cause the negative effects of increased water temperature that have been observed in salmonid populations (Hotlby 1988)

Removing trees may also leave soil susceptible to erosion if proper management steps are not taken (Hartman and Scrivener 1990). Sediment entering the stream can have lethal or sublethal effects on all life stages of fish (Newcombe and MacDonald 1991). Of particular concern for salmonids is the effect sedimentation may have on breeding success. Sediment on a stream bottom settles in the spaces between the gravel and cobble, causing increased embeddedness (Cunjak 1996). These spaces are essential for salmonid reproduction, as they allow water flow around the eggs for gas exchange, and provide a route through which young fry can leave the stream bed and enter open water (Mills 1989). Juvenile salmon also use these spaces as overwintering habitat (Cunjak 1988). Without these spaces, fry and juvenile survival are drastically reduced (Mills 1989, Heaney et al. 2001).

Exposure to sediment has also been found to be associated with reduced growth rates in rainbow trout (Shaw and Richardson 2001) and brook trout (Sweka and Hartman 2001). This appears to be due to the increased amount of foraging effort required due to decreased visibility, as there was no decrease in the numbers of invertebrates present (Shaw and Richardson 2001) or the amount of food consumed (Sweka and Hartman 2001).

The relationship of salmonid populations to logging on Carnation Creek in British Columbia was studied by Hartman et al. (1996) over a twenty year period. All species examined (coho salmon, chum salmon, *Oncorhynchus keta* Walbaum, rainbow trout, and cutthroat trout, *O. clarki* Richardson) were affected by logging, with populations and/or biomass of each species reduced in deforested areas. Logging had some positive effects on coho salmon. By reducing population numbers, competition was reduced and individuals grew larger. These larger individuals also experienced greater overwintering success than fish in previous years. However, these fish underwent smolt transformation at a younger age (95% at age 1+ as compared to 50-75% at age 1+ prior to logging), resulting in lower survival at sea, and leading to a negative impact on the population in the long-term.

1.2.1.2.3 Urbanization

Urban areas are major contributors of point and non-point source pollution. This pollution may be from roads (de-icing products, automobile by-products), sewage, or industrial run-off (Carpenter et al. 1998). The amount of urbanization in the watershed, followed by the amount of land in agriculture, has been found to be the best predictor in the variation in nitrate concentrations in streams in Alabama (Basnyat et al. 1999). It is also a contributing factor, along with amount of land in agriculture and orchards, to the amount of sediment in these streams, likely through runoff from roads and deforested areas.

1.2.1.2.4 Agriculture

When land is used for agricultural purposes, any resident trees are generally cut down. As mentioned above, this can lead to problems with decreased shading, increased temperature, and increased sedimentation in streams. The more direct problems associated with agriculture depend on the type of farm (i.e., livestock or crop), as well as the farming practices (i.e., grazing vs. penned livestock; type of pesticide and or fertilizer application).

In some areas, grazing livestock are allowed access to streams, resulting in increased erosion of banks and sedimentation in the streams (Chapman and Knudsen 1980), as well as increased nitrate levels as a result of animal waste (Carpenter et al. 1998). Chapman and Knudsen (1980) related salmonid biomass in streams in Puget Sound, Washington, to livestock impacts. Coho salmon and cutthroat trout were sampled in early and late summer. Livestock access areas did not differ from control areas in the early part of the summer; in the later part of the summer salmonid biomass was lower in areas with livestock access. This was thought to be associated with the decreased quality of the habitat (i.e., less cover, more sedimentation) in the livestock access areas.

Many crops in large-scale agriculture operations require large amounts of fertilizer and pesticides. The unintentional addition of fertilizers to a stream ecosystem (e.g., through runoff) can increase the nitrate and phosphorus levels (Carpenter et al. 1998, Heaney et al. 2001, Vuorenmaa et al. 2002). When the amount of nitrogen is increased in streams with a large proportion of silt in the streambed, water may not be able to percolate through the silt, creating anaerobic conditions (Massa et al. 2000). In the absence of oxygen, nitrogen and nitrates will convert to ammonia and nitrite, which are both more toxic to fish than nitrogen and nitrates. These increased ammonia and nitrite levels, along with the decrease in dissolved oxygen, has been found to reduce egg and fry survival of brown trout in agricultural watersheds in France (Massa et al. 2000). Similarly, increased sedimentation in laboratory studies was found to reduce egg-to-fry survival in Atlantic salmon (Heaney et al. 2001). Smolt production in the River Bush in Northern Ireland has decreased in recent years with the increase in land in agricultural production in the watershed; many sites in the river have higher sediment loading as well as increased nitrate and phosphorus levels (Heaney et al. 2001).

Pesticides can adsorb to silt particles and enter surface waters with silt runoff. If pesticides enter the waterways in large quantities they can cause extensive fish mortality (e.g., Saunders 1969, Johnston and Cheverie 1980, Mutch et al. 2002). If pesticides enter at lower concentrations they may have other, sublethal, effects, which may affect long-term survival or reproduction. For example, exposure to the organophosphate insecticide malathion has been found to reduce physical activity in both brook and rainbow trout (Post and Leasure 1974), and the organophosphate insecticide fenitrothion has been found to reduce the swimming speed of brook trout (Peterson 1974). Reduced activity levels could lead to reduced ability to evade predators or capture prey (Little and

Finger 1990). Diazinon, an organophosphate insecticide, has been found to inhibit olfactory-mediated alarm responses and homing behaviour in chinook salmon, which could reduce survival and reproduction (Scholz et al. 2000).

Many pesticides are also endocrine disruptors. Endocrine disrupting chemicals have three modes of action. They can block the endogenous hormone, trigger the endogenous hormone to have an abnormal response, or mimic an endogenous hormone (Fenner-Crisp 1997). In extreme cases, endocrine disruptors can affect sexual development. For example, snakeheaded murrel (*Channa punctatus* Bloch) exposed to phenthoate experienced reduced ovarian weight (Bhattacharya 1993). Additionally, exposure to 0.5 to 5 μg/L estradiol-17β resulted in feminization of male masu salmon (*Oncorhynchus masao* Brevoort) (Nakamura 1981). Any disruption to sexual development is expected to adversely affect the population.

In general, agricultural activity is known to negatively affect fish communities. For example, indices of biological integrity scores for fish communities at 18 sites on the River Raisin in southeastern Michigan were found to be negatively correlated with the amount of agricultural land in the watershed (Lammert and Allan 1999). This relationship was strongest when only a 50 m area on each side of the stream was used to determine the upstream and adjacent land use. Peterson and Kwak (1999) found a reduction in smallmouth bass populations over time that was attributed to changes in flow and discharge patterns related to increased agriculture and urbanization in the watershed.

1.2.1.3 Impoundments

Impoundments, or dams, are areas where water flow is blocked or decreased. There are two main sources of artificially created impoundments: human and beaver. Impoundments can disrupt the normal functioning of a river through changing water flow, blocking passage for fish movement and migration, increasing water temperature and pH, decreasing dissolved oxygen content, and changing other water chemistry variables (ASE Consultants Inc. and Department of Biology, UPEI 1997, Jager et al. 2001, Margolis et al. 2001, Zabel and Williams 2002). Impoundments can increase water quality downstream by collecting and retaining sediment from the river, and decreasing nitrate concentrations in streams through nitrate reduction (ASE Consultants Inc. and Department of Biology, UPEI 1997, Margolis et al. 2001). The general effects of impoundments are understood, but the specific effects of individual impoundments on their watersheds need to be evaluated (ASE Consultants Inc. and Department of Biology, UPEI 1997).

1.2.1.4 Minimizing the Damage of Land Use

There are different steps that can be taken to minimize the impact of land use practices on watersheds. With respect to agriculture, an extended crop rotation with more legumes will allow soil to retain more organic matter, which reduces erosion and leaching of excess nutrients (Honisch et al. 2002). When planting potatoes, farming smaller fields, with strip cropping (alternating crop types), terracing (planting grass in strips down a slope), and hedgerow installation has been found to reduce nutrients in ground and surface water (Honisch et al. 2002).

The riparian zone has a major role in the health of the river, and can be used as a buffer between potentially deleterious land use practices and the aquatic habitat. Basnyat et al. (1999) found that as the proportion of forest inside the riparian zone of Fish River, Alabama, increased, nitrate levels downstream decreased. Using stable nitrogen isotope analysis, Konohira et al. (2001) found that most of the denitrification of water traveling to surface water occurs in the ground water of the riparian zone. Riparian zones have been found to outperform both lakes and wetlands in reducing nitrate and phosphorus concentrations in water (Kuusemets and Mander 2002). This improved water quality has impacts on the organisms in the river. Stauffer et al. (2000) examined 20 streams in the Minnesota River Basin, Minnesota. Sites varied in riparian zone cover (wooded or open) and in runoff potential (high or low) of the surrounding soil. Mean IBI (index of biological integrity) scores were highest at sites with wooded riparian cover and low runoff potential (based on soil characteristics), and lowest at sites with open riparian cover and high runoff potential. Species richness was also correlated positively with riparian cover and negatively with runoff potential. The size of the riparian zone required to be effective in reducing land use impacts varies with the slope, vegetation, soil, and hydrology in the watershed (Haberstock et al. 2000).

In Minnesota, best management practices (BMPs) have been initiated on some farms to reduce negative impacts on the environment. Nerbonne and Vondracek (2001) examined 27 sites in the Whitewater River watershed to determine if there were differences in sites with upstream BMPs farming compared to sites with upstream conventional farming. All of the sites had upstream agricultural land use. Sites were separated into warm- and coldwater sites, with different fish expected at each; therefore, the researchers developed two different IBIs, and analyzed cold- and warmwater sites separately. Sites with BMP farming upstream had less fines in the substrate and less embeddedness; the differences in fish populations were not significant for the different types of upstream land use.

1.2.2 Stable Isotope Analysis

Stable isotope analysis is a technique that is currently being used in aquatic studies to examine attributes such as food chains (e.g., Kidd et al. 1999, Vander Zanden et al. 2000) and landscape effects (e.g., France 1996, Lake et al. 2001). The analysis is performed by measuring the stable isotopes of certain elements, typically carbon or nitrogen, but also sulfur, hydrogen, and oxygen. These isotopes naturally occur in certain ratios; much knowledge can be gained on food source and food web structure by examining the ratios that are present in the plants or animals tested (Doucett et al. 1999).

Stable isotope ratios are examined using the equation

$$[(R_{sample}/R_{standard}) - 1] \times 1000 \quad (3)$$

where R is the ratio of the heavier isotope divided by the lighter isotope (i.e., for carbon, R = $^{13}C/^{12}C$). $R_{standard}$ is the international ratio standard for that element, while R_{sample} is the ratio that was measured. The difference between the two is expressed as parts per thousand (ppt or ‰) (Krouse 1987, Doucett et al. 1999). A positive result indicates the sample is enriched, or heavy, while a negative result indicates the sample is depleted, or light (Krouse 1987). The symbol indicating the ratios of the isotopes is $\delta^{13}C$ for carbon and $\delta^{15}N$ for nitrogen (i.e., the heavier isotope is indicated).

Stable isotope ratios in fish are affected by the food web structure. The basis of aquatic food webs can be from autochthonous (within the stream) or allochthonous (outside the stream) material (Doucett et al. 1996). The relative amounts of these materials may change depending on surrounding land use. For example, a stream in a forested area with trees in the riparian zone will have very different types and amounts of allochthonous material than a stream through an agricultural area with a grassed riparian zone. Stable carbon isotopes are the most commonly-used isotopic analyses to examine differences in primary food sources, as carbon isotopes are fixed differently during photosynthesis, and do not change a great deal through the food web (see Doucett et al. 1996). Stable nitrogen isotopes are used most often to examine changes in food web structure, as they are enriched (ratio increases) with increased trophic position (see Doucett et al. 1996). However, stable nitrogen isotope ratios may also be affected by the surrounding land use. Stable nitrogen ratios in the clam (*Potamocorbula amurensis* Schrenck) were found to be higher in San Francisco Bay in areas receiving large amounts of sewage (Fry 1999), which demonstrates the importance of anthropogenic inputs on stable nitrogen isotope ratios.

1.2.3 Habitat Use by Salmonids

1.2.3.1 Atlantic salmon (*Salmo salar*)

In the fall, Atlantic salmon return from the sea to their natal river to breed. Although historical runs of salmon on PEI started as early as June (Stewart 1806), present-day runs generally do not begin until October (Johnston and Dupuis 1990). The female salmon digs a nest, known as a redd, that is typically located in a swift-moving riffle area with a high proportion of gravel-cobble substrate and good aeration (Smith 1983, Bardonnet and Baglinière 2000). In October or November the female and male release eggs and sperm simultaneously, and the female, producing 1600-1800 eggs/kg of fish, covers the fertilized eggs with gravel (Scott and Crossman 1973, Smith 1983, Bardonnet and Baglinière 2000). Eggs typically hatch in April (Scott and Crossman 1973, Jobling 1995). The young salmon, or alevins, are about 15 mm at hatching, and have a yolk sac which provides nourishment for the first few weeks of life (Smith 1983). Once salmon use up their yolk sac, they emerge from the gravel in May or June and are known as fry. Fry are referred to as parr once they acquire darkly coloured lateral bars and reach 5-8 cm in length . Parr are mainly found in riffles, usually within 100 m of the redd from which they originated. Young parr

(<1 year old) are generally found in water that is not as swift or deep as the water in which older parr are found (DeGraaf and Bain 1986, Morantz et al. 1987). However, both groups prefer water velocities between those found in a clear-running riffle-pool sequence, and those in a murky slow-flowing area (DeGraaf and Bain 1986).

There is less information available concerning the overwintering habitat of young salmon. Cunjak (1988) studied the daytime winter behaviour and microhabitat of salmon parr in Nova Scotia, and found that rock shelters provide important refuges. The majority of the salmon were found in riffle-run habitats, and were beneath rocks that had available access underneath (i.e., were not compacted by silt). Interestingly, many of the parr were found in redds that had been excavated the previous fall. Although redds composed only 10% of the transect area, they were home for 40% of the salmon found. Stomach content analysis showed that salmon continued to eat throughout the winter, although the diversity of the prey was low. Cunjak (1988) noted that no salmon were observed above the rocks, indicating that foraging must occur overnight. Whalen and Parrish (1999) investigated winter nocturnal location and behaviour of Atlantic salmon in Connecticut. They observed salmon above the substrate, with most of the salmon resting in contact with the silt-sand and gravel substrates. Older parr preferred deeper water than young of the year parr, and both groups preferred water velocity of 12-21 cm/s, slower than the average velocity of 44-57 cm/s for the section of the river examined (Whalen and Parrish 1999).

Atlantic salmon are anadromous, and in spring some of the salmon parr will undergo physiological changes to prepare for migration to sea. These salmon are now referred to as smolts, and migrate to sea at the same time as reconditioned salmon spawners from the previous fall (kelts). The parr-smolt transformation is thought to be under control of a circannual rhythm, and modulated by photoperiod and temperature (Hoar 1988), but the timing may be variable, even among similarly aged individuals (Metcalfe 1998). Smolting is indicated externally by a silver coloration in the skin of the salmon. Internally, there is a change in the production of many hormones, including thyroid hormones, growth hormone, prolactin, and corticosteroids. The gills, kidney, urinary bladder, and intestine undergo morphological and physiological changes to deal with the transition from a freshwater to a saltwater environment (Hoar 1988). Salmon that have been at sea for one year are referred to as grilse, and are 1.5 to 2 kg; salmon that have spent more than one year at sea (multi-sea winter fish, MSW), range from 4 to 14 kg, averaging 5 kg in Canada (Smith 1983).

Atlantic salmon were originally widespread on Prince Edward Island, and found in most, if not all, watersheds (Stewart 1806). However, overharvesting caused decreased populations, and by 1780 an act was passed by the Legislature to regulate the salmon fishery (Statutes of Prince Edward Island, *cited from* Dunfield 1985). This act was not enforced, and by 1820 Atlantic salmon had disappeared from many Prince Edward Island streams (Warburton 1923). The province-wide status of this species has been recently investigated (Guignion et al. 2002), and salmon were found to be absent from many watersheds in which they were present only a decade ago.

1.2.3.2 Brook trout (*Salvelinus fontinalis*)

Brook trout are widespread throughout Prince Edward Island (Guignion et al. 2002). They prefer cool, clear, well-oxygenated streams, and will actively seek lower temperatures when the water warms up (≥20°C) (Power 1980).

Brook trout typically reach sexual maturity at three years of age, but some may breed at two years (Power 1980). Spawning occurs through October and November in upwelling areas in gravel beds. Some of the spawners are sea-run trout (i.e., are anadromous), but others remain in the stream throughout their lives. *Salvelinus* species that go to sea generally do not spend more than a few weeks in saltwater, and return to freshwater every year in late summer and early autumn (Smith and Saunders 1958, Jobling 1995). When breeding, the female trout, sometimes assisted by the male, constructs a redd by clearing away an area of debris and silt. The fish release sperm and eggs simultaneously, and the female covers the redd with gravel (Power 1980).

Brook trout are less tolerant than salmon to high temperatures, and generally cannot live above 25°C (Fry et al.1946, *cited from* Power 1980, Scott and Crossman 1973). High temperatures below 25°C can still cause sublethal effects in brook trout. Hokanson et al. (1973) found that male brook trout did not become sexually mature and produce spermatazoa at temperatures above 19°C, and ovulation and spawning did not occur above 16°C. Incubation time of the eggs depends on the temperature of the water; the optimum incubation temperature is 6°C (Hokanson et al. 1973). Water temperatures above 11.7°C results in increased mortality of eggs; 100% mortality of eggs was observed above 18°C (Hokanson et al. 1973, Scott and Crossman 1973).

Brook trout have requirements similar to Atlantic salmon, but generally live and spawn in lower velocity water (Gibson 1966, Guignion and MacFarlane 1991). As with salmon, older trout are often found in deeper, faster water than younger fish (Cunjak and Power 1986). Although trout generally are not found in large groups during the summer, they will aggregate in pools close to groundwater upwellings in winter months (Cunjak and Power 1986).

1.2.3.3 Rainbow trout (*Oncorhynchus mykiss*)

In contrast to Atlantic salmon and brook trout, rainbow trout are not native to Prince Edward Island. Rainbow trout are a west coast, Pacific Ocean species, and were first introduced to the east coast in Newfoundland in 1887 (Scott and Crossman 1973). The first record for rainbow trout introduction to Prince Edward Island was in 1925 (Scott and Crossman 1973). Since that time, rainbow trout have colonized many of the rivers in the province, due to stocking, escape from aquaculture facilities, and natural movements (Guignion et al. 2002).

The biology of rainbow trout in the Maritime provinces has not been studied as extensively as that of Atlantic salmon and brook trout. However, the life history of rainbow trout in its native western streams has been well documented. Contrary to Atlantic salmon and brook trout, rainbow trout are spring spawners (Zimmerman and Reeves 2000). The eggs are laid in redds that the female excavates, and then covers once the eggs are laid and fertilized. The redds are excavated in a gravel bottom with good aeration and water movement. The time to hatch depends on the temperature of the water, with rainbow trout eggs requiring 375 degree days to hatch (Jobling 1995).

Growth in rainbow trout is variable between populations, and is dependent on food availability, territory availability, and temperature (Linton et al. 1998). Individuals typically become sexually mature in 3-5 years, with males generally maturing a year earlier than females; the age of maturity can depend on temperature and food availability (Van Winkle et al. 1997). Rainbow trout can be anadromous, but some populations remain in freshwater throughout their lives. As with brook trout there can be both anadromous (anadromous rainbow trout are called steelheads) and non-anadromous fish within a population. Where fish from both of these life histories co-exist, they have been found to be reproductively isolated (Zimmerman and Reeves 2000).

Rainbow trout are thought to compete with salmon, as they are found in the same habitat, and occupy a similar niche (Volpe et al. 2001). In British Columbia, Atlantic salmon have been introduced, so the concern in that area is that Atlantic salmon may out-compete the native trout. Volpe et al. (2001) examined the effects of prior residency on inter- and intraspecific competition in rainbow trout and Atlantic salmon. The endpoints examined were foraging, agonism, cruising, and change in weight and length. They found that, regardless of species, the resident had an advantage over the intruder. However, rainbow trout outperformed Atlantic salmon overall, with more acts of aggression, more cruising, more foraging, and less weight loss or more weight gain in rainbow trout. Cunjak and Green (1983) investigated the habitat preference of rainbow trout and brook trout in streams in Newfoundland, and found that the two species occupied separate niches, with rainbow trout preferring faster water with less cover than brook trout. However, limited evidence suggests that rainbow trout on PEI may be more successful than brook trout in recolonizing an area after a fish kill (Johnston and Cheverie 1980). These studies indicate that the effects of rainbow trout on native salmonid populations on Prince Edward Island should be investigated further.

1.2.4 Objectives

The objectives of this study were to examine the:

- relationships between land use and salmonid populations of Prince Edward Island
- effects of two major fish kills on the population structure of salmonids in the Wilmot River.

Information from these objectives will permit the development of more successful models to predict the impact of current land use activity on salmonid communities in the province, and allow for the development of better guidelines and regulations if required.

CHAPTER 2:

GENERAL MATERIALS AND METHODS

2.1 Fish Community Sampling

2.1.1 Sampling Method Background

Many studies on fish populations are conducted by electrofishing (e.g., Chapman and Knudson 1980, Barlaup and Åtland 1996, Kahler et al. 2001). Electrofishing is a sampling method that utilizes an electric current to temporarily immobolize fish to make them available for collection. Electrofishing can have negative impacts on fish, so it is important that all precautions are taken to minimize adverse effects, such as stress and physical injuries including hemorrhages, fractures, and mortality. Using the lowest required power output when electrofishing reduces the chance of injuring the fish and is a more efficient use of the available power (Kolz et al. 2000), therefore all electrofishing in this study was done using the lowest possible power setting.

2.1.2 Sampling Procedure

Approximately 100 m^2 was sampled at each site. Since the rivers were of varying width, the length of the sampled area varied for each site. All sites had a barrier net placed at the upstream site boundary to prevent the escape of any fish that swam ahead while sampling. Any sites that were 4 m or greater in width also had a downstream barrier net in place, as it was more likely that in streams this wide fish would be able to swim around the individuals netting the fish. In 2002, barrier nets were placed at both ends of all sites to reduce the loss of fish from the study site. The fish were sampled by electrofishing with a Smith-Root backpack electrofishing unit, Model 12 POW. Three consecutive sampling runs were performed, with the fish captured during each run placed into separate 40-50 L tubs. Keeping the fish separate allowed the determination of population estimates using the program POPDN3, version 1.3 (1985), which uses the maximum likelihood method to produce the estimate, then tests the estimate with the G-test. The program provides the population estimate and probability of capture, as well as the standard error and 90% confidence limits for these values.

In 2001 fish were sedated with ENO® (536 mg sodium citrate/g) to reduce injury during handling. In 2002 clove oil (1.25×10^{-2} ppt) was used instead of ENO® as the anaesthetic agent, as it is thought to have fewer deleterious effects on fish (see Anderson et al. 1997, Keene et al. 1998). All salmonids were identified to species, measured (tip of rostrum to fork in tail) to the nearest mm and weighed to the nearest 0.1 g. Condition factors (weight/length^3 x 100) were determined for each fish, and percent habitat saturation was determined for each site (Equation 1, p. 6, Chapter 1). After processing, fish were placed in a bucket of clear water and released into the stream when fully recovered. Nonsalmonids were immediately placed into the bucket of clear water and released when sampling was completed.

CHAPTER 3:

THE INFLUENCE OF STREAM HABITAT AND SURROUNDING LAND USE PATTERNS ON SALMONID POPULATIONS OF PRINCE EDWARD ISLAND

3.1 INTRODUCTION

The landscape of Prince Edward Island has changed dramatically since the arrival of European settlers. Prior to this, forests covered most of the island, and were composed of large, shade-tolerant climax species, such as beech, yellow birch, and sugar maple. Much of the forest was hardwood or mixed wood, with some isolated stands of single species such as cedar and red pine (Sobey 2002). Much of PEI was cleared of its forests during the 1800s, and by 1900 only 31% of the province was forested. The harvesting of trees during this period diminished the quality of the forests, as the highest quality trees were selected to be used for shipbuilding and to be sent to Europe (MacDonald 2001). The percentage of forested land has increased in the last 100 years with the abandonment of some farms (Cairns 2002a), and in 1990 49% of the province (2800 km^2) was forested (PEI DAF 2002). By the year 2000, the amount of forested land decreased by six percent, to 2568 km^2, with the increased clearing of land for potato and blueberry production. The amount of land in agricultural production increased during this period by eight percent from 2050 to 2222 km^2 (PEI DAF 2002).

Potatoes are the major agricultural crop on Prince Edward Island (DeHaan 2002); in the summer of 2000 8.5% (477.4 km^2) of land on PEI was in potato production (see Fig. 3-1; determined using MapInfo Professional Version 6.5, © MapInfo Corporation 1985-2001, from 2000 Corporate Resource

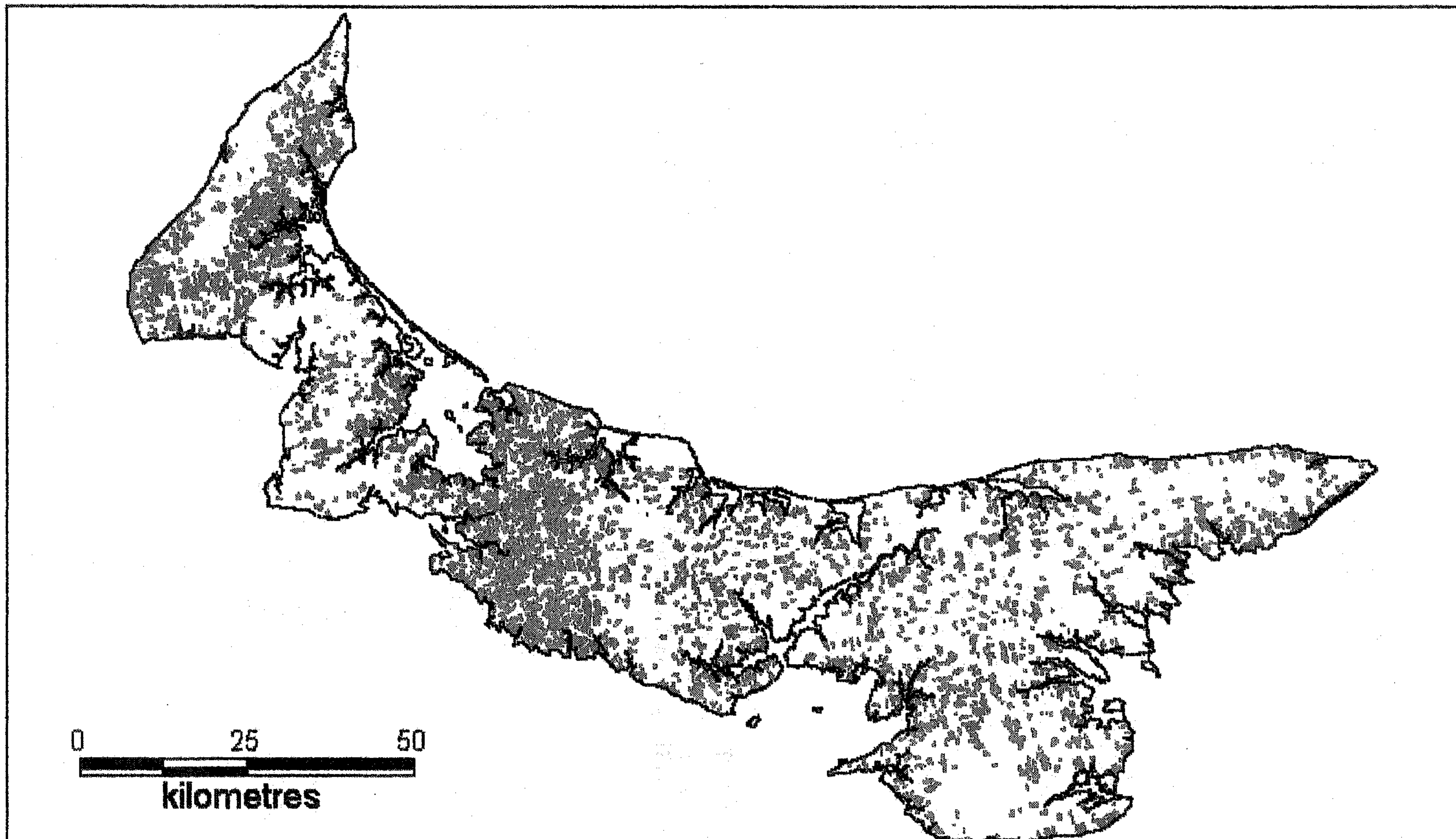


Figure 3-1. Potato production on Prince Edward Island. Fields that were growing potatoes in 2000 are indicated in brown.

Inventory GIS layers obtained from PEI Department of Agriculture and Forestry). Assuming a 3-year crop rotation (now mandatory on Prince Edward Island unless a management plan is in place; PEI Provincial Government 2002), approximately 25.5% of land on PEI is in potato rotation. Industrial potato farming uses a substantial amount of pesticides, and current potato farming practices can facilitate soil erosion (see Edwards et al. 1998), especially in PEI where the soils are mainly acidic sandy loams that are high in silt and fine sand content (DeGrace 1999). In fact, more than 80% of land currently in potato production is considered to be at high to severe risk of waterborne soil erosion (see Cairns 2002a). As noted previously, silt entering waterways can have a direct negative impact on fish communities, as well as provide an entryway for pesticides.

Short- and long-term changes in the environment due to different types of land use are expected to affect the water quality of streams, and subsequently impact the fish species present. Potential changes in fish health may be assessed through monitoring changes in population density (e.g., Peterson and Kwak 1999), percent habitat saturation (Grant et al. 1998),and fish condition (e.g., Bergstedt and Bergersen 1997). Examination of stable isotope ratios in fish tissue may also be used to investigate changes in the environment of the fish (e.g., Lake et al. 2001). The objectives of this study are to:

- describe the community structure of salmonids (Atlantic salmon, brook trout, and rainbow trout) in selected rivers of Prince Edward Island.
- determine if population densities, percent habitat saturation, body condition, and stable isotope ratios are influenced by surrounding land

use practices.

- test a predictive model developed from the relationships emerging from the second objective.

3.2 MATERIALS AND METHODS

3.2.1 Study Locations

Through consultation with individuals from the Atlantic Salmon Federation, PEI Wildlife Federation, and UPEI Biology Department, watersheds that represented a wide range of land use patterns were selected to investigate potential impacts of land use practices on salmonid communities. The preliminary selection of sites involved the examination of aerial photographs, and a 1990 Forestry Road Atlas was used to estimate the amount of forest in the watershed. These sites were visually inspected to determine suitability for salmonids and sampling methods. Final sites were chosen to represent good spawning habitat for salmonids (shallow with riffle areas, gravel-cobble substrate, some pools). Surrounding land use was also noted, and degree of difficulty in accessing the site was taken into consideration. Following these procedures, nine rivers, with three sites on each river, were selected for study in 2001. Five additional rivers with one site on each river were chosen in June 2002 to test predictive models generated from the original 27 sites. These additional sites varied in both geographic location and land use activity. In July 2002, two separate fish kills affected one of the nine original rivers (Wilmot River). Therefore, five additional sites were added on this river to monitor fish recovery and movement after the fish kills (see Chapter 4). Fig. 3-2 shows the locations of all 37 sites; Table 3-1 gives site and river abbreviations. Latitude and longitude for each site are given in table A-4, Appendix A.

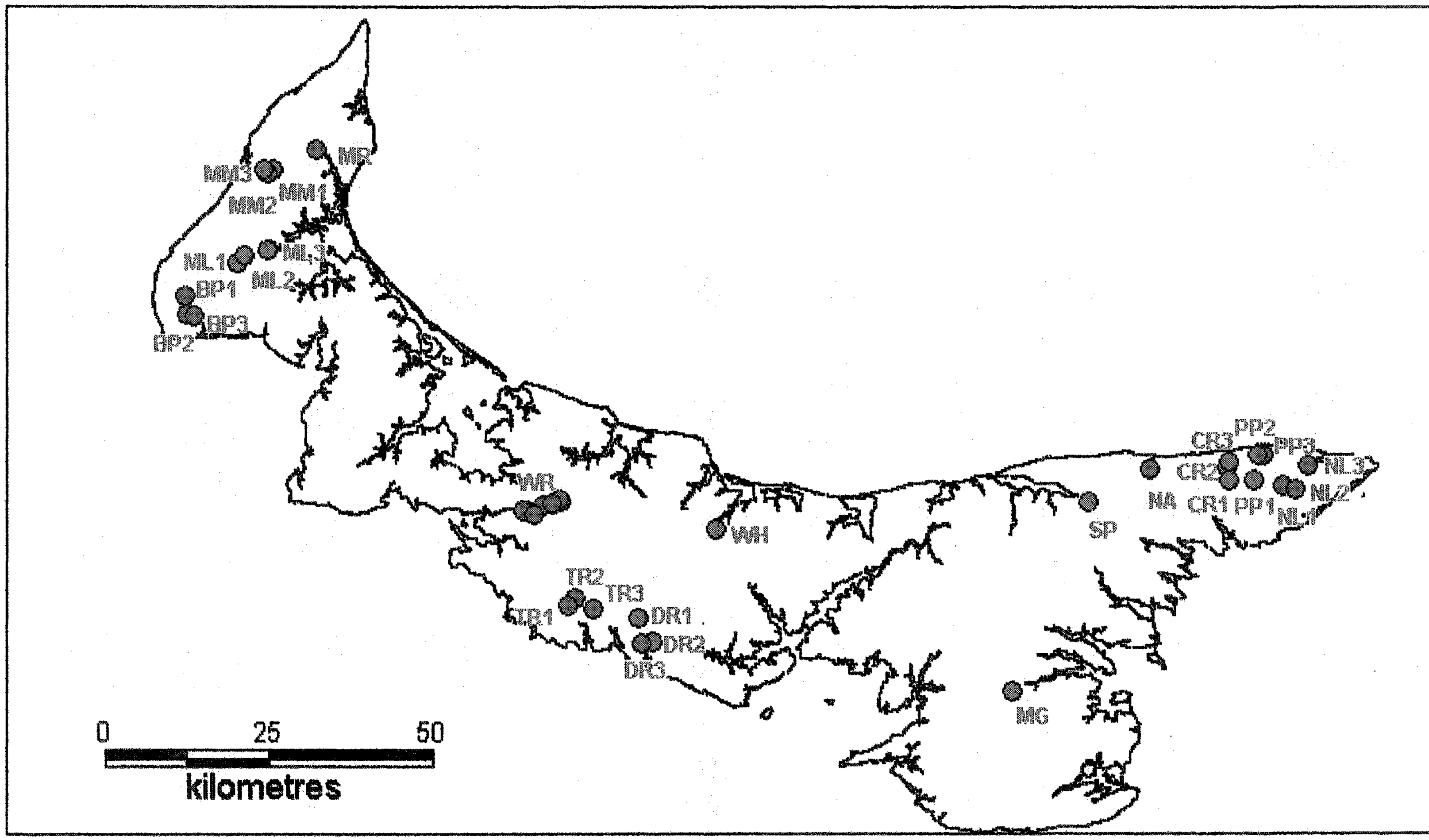


Figure 3-2. Location of the 37 sites sampled. WR (Wilmot River) sites are shown in more detail in Figure 3-1. Site abbreviations are given in Table 4-1. Full site names are given in Table 3-1.

Table 3-1. Study rivers, their abbreviations, and study sites on the rivers. Sites for each river are numbered upstream (1) to downstream (3).

River	Abbreviation	Sites
Big Pierre Jacques River	BP	BP1, BP2, BP3
Cross River	CR	CR1, CR2, CR3
DeSable River	DR	DR1, DR2, DR3
Mill River	ML	ML1, ML2, ML3
Miminegash River	MM	MM1, MM2, MM3
Montague River	MG	MG
Montrose River	MR	MR
Naufrage River	NA	NA
North Lake Creek	NL	NL1, NL2, NL3
Priest Pond Creek	PP	PP1, PP2, PP3
St. Peters River	SP	SP
Tryon River	TR	TR1, TR2, TR3
Wheatley River	WH	WH
Wilmot River	WR	WR1, WR2, WR3, WR4, WR5, WR6, WR7, WR8

3.2.2 Fish Community Sampling

All salmonids were sampled by electrofishing as described in Chapter 2. In 2001, all 27 sites were sampled over a two-week period at four week intervals in June (June 18 - July 2), July (July 16 - 26), and August (August 13 - 23). As it was observed that not all young of the year rainbow trout and Atlantic salmon were out of the gravel in June 2001, sampling times were delayed in 2002. Sampling periods during this year were July (July 4 - 16, and 20-25 for Wilmot River additional sites), August (August 1 - 12), and September (August 26 - September 2). Fish were processed, and condition factors, population density, and percent habitat saturation were determined as described in section 2.1.2. Additionally, the percent variability in density and percent habitat saturation between sampling periods was determined with

$$[(\text{max}-\text{min})/\text{min}]*100 \qquad (4)$$

3.2.3 Stable Isotope Analysis

Adipose fin clips were taken from live young of the year brook trout at all sites where they were present during the September 2002 sampling to use for carbon and nitrogen stable isotope analysis. Adipose fins were collected from young of the year brook trout at sites on rivers not used in this study during August 2002 to determine the number of fins needed to obtain 0.5 mg dry weight (the amount required for analysis). It was determined that 2-3 fins would yield enough tissue for analysis. Up to 20 fin samples were collected at each site to account for variation in individual fish and acquire a representative sample of the site. The fins were put in 2 mL glass vials and kept on ice while in the field. The tissue samples were subsequently dried in an oven at 60°C for 24-48 h, and sent to the Stable Isotopes in Nature Laboratory at the University of New Brunswick in Fredericton, New Brunswick, for analysis.

3.2.4 Water Chemistry

Air and water temperatures were monitored using a combination of methods. Temperature was recorded at each site on each sampling occasion using a thermometer. In addition, 11 of the 27 sites in 2001 and 30 of the 37 sites in 2002 had data loggers installed in early July (2001) or June (2002) to record the water temperature every hour. In 2001, all data loggers were Onset Optic StowAway™, and were located at the most downstream site of each river. As the Tryon River has two main branches, a data logger was located in each of these branches (sites TR1 and TR3). Priest Pond Creek has a large dam near its source, immediately upstream of one of the sites (PP1) and a data logger was installed here in addition to the data logger at the most downstream site (PP3). In 2002, data loggers were installed in all sites except MM1, PP2, WR1, WR3, WR4, WR6, and WR8, and were either Onset Optic StowAway™, Vemco Minilog-TR, or Onset Hobo-Temp. Data loggers were removed in November of each year. Table 3-2 summarizes which sites had data loggers in each year.

For July and August 2001 and all of 2002, conductivity (μS) was

Table 3-2. Location of data loggers to record temperature in 2001 and 2002. Sites that had a data logger are indicated by a check mark. The data logger at TR2 (indicated by *) was not successfully recovered.

Site	2001	2002	Site	2001	2002
BP1		✓	NL2		✓
BP2		✓	NL3	✓	✓
BP3	✓	✓	PP1	✓	✓
CR1		✓	PP2		
CR2		✓	PP3	✓	✓
CR3	✓	✓	SP		✓
DR1		✓	TR1	✓	✓
DR2		✓	TR2		✓*
DR3	✓	✓	TR3	✓	✓
MG		✓	WH		✓
ML1		✓	WR1		
ML2		✓	WR2		✓
ML3	✓	✓	WR3		
MM1			WR4		
MM2		✓	WR5		✓
MM3	✓	✓	WR6		
MR		✓	WR7	✓	✓
NA		✓	WR8		
NL1		✓			

determined with a YSI Model 33 Salinity-Conductivity-Temperature meter, and dissolved oxygen was determined with a YSI Model 55 dissolved oxygen meter. In July 2001, pH, ammonia, nitrate, and alkalinity were determined on-site using a HACH® kit. Alkalinity was determined by titration; all others were determined using the colour-wheel method. In August 2001, pH was again determined using the HACH® kit colour wheel method. Additional water chemistry variables were analyzed by the Water Resources Division of the Prince Edward Island Department of Fisheries, Aquaculture, and Environment (see Table 3-3) to determine if they fell within normal ranges.

In 2002, water temperature, dissolved oxygen, and conductivity were determined at all sites for all sampling periods using the methods described above. pH was determined on site using a Fisher Scientific accumet® portable AP61 pH meter. Water samples from each site for each sampling period were sent to the Water Resources Division of the Prince Edward Island Department of Fisheries, Aquaculture, and Environment, for determination of nitrate, phosphorus, and alkalinity.

3.2.5 Habitat Descriptions

Three types of habitat descriptions were performed: (a) instream, (b) riparian zone, and (c) entire watershed. Except where indicated below, habitat descriptions were performed in September 2001.

Table 3-3. Water chemistry variables collected at each site for each sampling period. A check indicates the test was performed in the corresponding month. Tests with an * were analyzed by Water Resources, Prince Edward Island Department of Fisheries, Aquaculture, and Environment. pH was also analyzed in the field. Units are given for all measurements; the detection limits are given for the variables analyzed at Water Resources. Sites that had data loggers to record the temperature are indicated in Table 3-2, page38.

Variables	Units/Detection Limits	June	July	August	2002
air temperature	°C	√	√	√	√
water temperature	°C	√	√	√	√ (+ data loggers)
conductivity	μS/cm		√	√	√
dissolved oxygen	mg/L		√	√	√
alkalinity	5 mg/L $CaCO_3$		√	√*	√*
ammonia	mg/L		√		
cadmium	0.005 mg/L			√*	
calcium	0.01 mg/L			√*	
chloride	1 mg/L			√*	
chromium	0.05 mg/L			√*	
copper	0.02 mg/L			√*	
iron	0.1 mg/L			√*	
lead	0.002 mg/L			√*	
magnesium	0.01 mg/L			√*	
manganese	0.02 mg/L			√*	
nickel	0.05 mg/L			√*	
nitrate-nitrogen	0.2 mg/L		√	√*	√*
pH			√	√*	√
total phosphorus	0.02 mg/L			√*	√*
potassium	0.05 mg/L			√*	
sodium	0.1 mg/L			√*	
sulfate	1 mg/L			√*	
zinc	0.02 mg/L			√*	

3.2.5.1 Instream habitat variables

Substrate and flow were examined in various ways with the instream habitat evaluations. Cross stream transects were examined every 5 m starting at 0 m (the downstream site boundary). For each transect, the water depth (cm), bank full depth (high water mark, cm), and dominant substrate (silt, gravel, cobble, boulder, or bedrock/hardpan) were recorded every 1 m across the stream (see Fig. 3-3). (These descriptions were obtained in 2002 for the ten additional sites). The substrate was given a rating using the modified Wentworth scale (Bain et al. 1985). This scale rates the substrate on a scale of 1-6, based on its size, with smaller substrate having a lower rating (Table 3-4). A low mean score indicates smaller substrate; a low standard deviation indicates a homogeneous substrate. In 2002, a more quantitative substrate score was obtained by measuring the three dimensions of 99 randomly-selected rocks to the nearest 0.1 cm.

In 2002, the velocity of the stream was determined at 10 intervals in a transect across the stream using a flow meter (Flo-mate™ Model 2000 Portable Flowmeter, Marsh-McBirney, Inc., and Global FlowProbe FP101-FP201, Global Water Sensors Samples Systems). The stream-bed slope was determined using SAL Series Automatic Levels (Task ® Tools). The stream-bed slope and bank full water depth were used to determine the tractive force (τ) which is a measure of the ability of the river to move objects, including substrate. The tractive force is determined using the equation (Gordon et al. 1992):

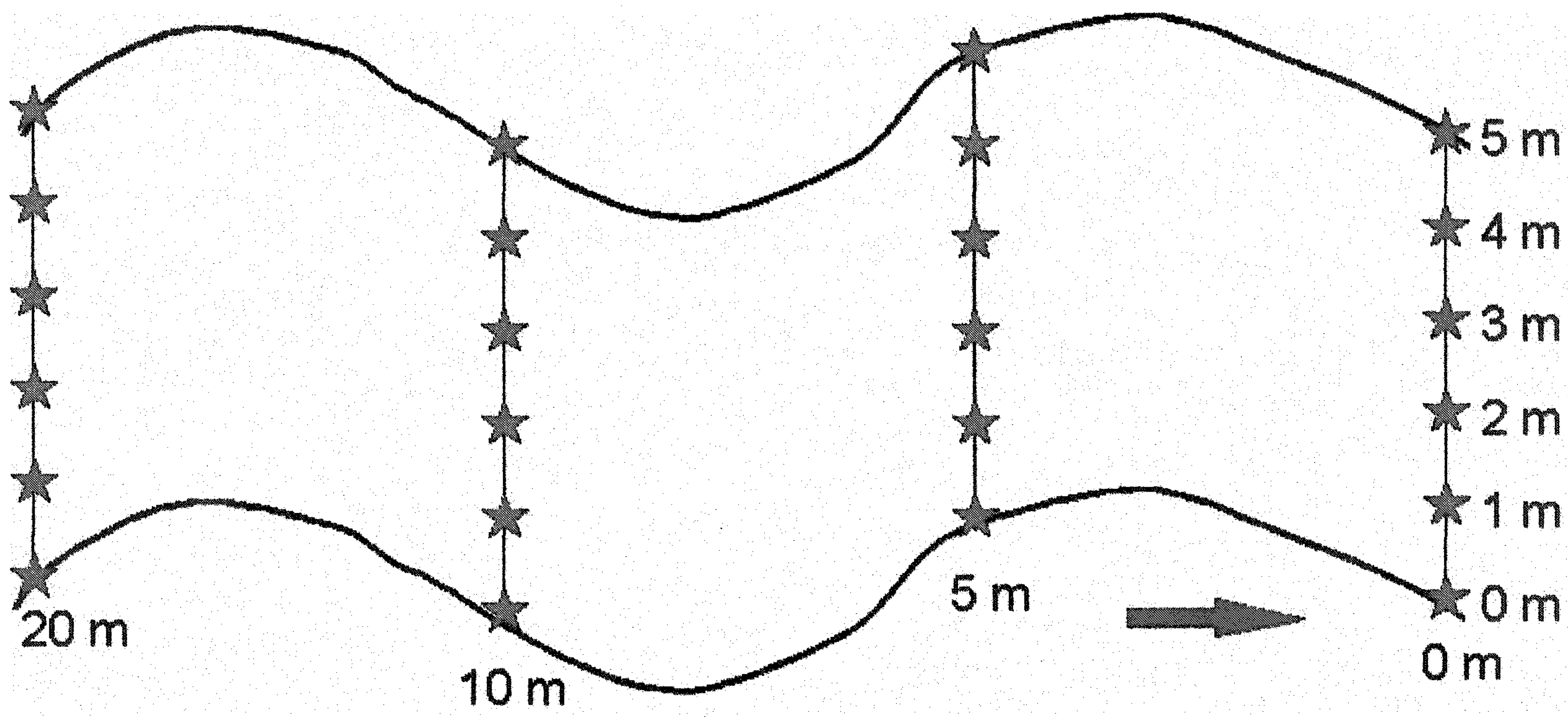


Figure 3-3. Representation of transects taken for stream habitat characteristics. The blue arrow indicates water flow direction. Measurements started at 0 m at the downstream edge of the site, and were taken every 1 m across the river starting at the left hand bank (indicated by red stars). Transects were taken every 5 m upstream until the end of the site was reached.

Table 3-4. Modified Wentworth classification system used to rate substrate size (from Bain et al. 1985).

Substrate type	Size class (cm)	Code
Smooth surfaces:		
Flat bedrock		1
Sand, silt	<0.2	1
Gravel	0.2-1.6	2
Pebble	1.7-6.4	3
Cobble	6.5-25.6	4
Boulder	>25.6	5
Irregular Bedrock		6

$$\tau\ (kg/m^2) = 1000\ kg/m^3 \times slope\ (m/m) \times depth\ (m) \qquad (5)$$

The tractive force was compared to the median substrate size to determine stream bed stability, using the equation (Gordon et al. 1992):

$$\tau\ (kg/m^2) = \text{diameter (cm) of particle that will be moved/stable} \qquad (6)$$

Additionally, a score for each site was determined in 2001 (and 2002 for additional sites) using the Stream Reach Inventory and Channel Stability Evaluation, which evaluates the stream bed and banks (Pfankuch 1975).

3.2.5.2 Riparian zone variables

In September 2001, a 60 m transect extending perpendicular from the downstream edge of the site was examined on both banks for each site. At every 10 m of the transect the slope and dominant vegetation were determined. If agricultural fields were observed in the surrounding area the crop type and distance from the stream were noted. In 2002, the tree type (coniferous or deciduous) and the diameter at breast height (1.5 m; DBH) of all trees in a 1 m wide strip was determined for a 30 m transect extending perpendicular from the stream on both banks. To determine the amount of solar radiation reaching the stream bed of the site, a Solar Pathfinder (Solar Pathways, Inc.) was placed in the centre of the stream in the approximate middle of the site. This instrument allows the quantification of shading from the riparian vegetation that blocks direct solar radiation, and is adjusted for latitude and time of year (Platts et al. 1987).

3.2.5.3 Watershed variables

For large scale analysis, MapInfo Professional Version 6.5 (© MapInfo Corporation 1985-2001) was used to analyze geographical information system (GIS) layers, obtained from the PEI Provincial Department of Agriculture and Forestry. The watershed boundaries upstream of each site were drawn based on 2-m contour lines (see Fig. 3-4). Within the watershed, the watershed size (km^2), the length of the river and roads (km), and the percent forest, wetland, potato rotation (potatoes, hay, and grain), other agriculture (blueberries, cranberries, pasture, feedlot, other agriculture), and urban (farmstead, residential, institutional, industrial, excavation pits, and commercial) land use types were determined from the 2000 Corporate Resource Inventory layer (see Fig. 3-5).

3.2.6 Data Analysis

Data from all samples from each site were averaged to generate one value for each site. Rivers with more than one site had the data from all the sites averaged to give one value for each river. For this analysis, only WR2, WR5, and WR7 (the three original sites selected) on the Wilmot River were combined. Before the data from the sites on each river were combined, they were checked for outliers using box-plot diagrams.

Figure 3-5. Map demonstrating determination of land use types within watershed (black line) of river (blue line). Land use types and their corresponding colours are: forested: green; clear cut: bright green; wetland: pale blue; abandoned: dark brown: potatoes: brown hatch-marks; hay: red hatch-marks; grain: orange hatch-marks; pasture: khaki dots; other agriculture: yellow: residential: bright pink; farmsteads: black; hedgerows: bright blue; industrial and institutional: white.

separately. Models were considered valid if $p \leq 0.05$. This test provides an overall r^2 value, which is the coefficient of multiple determination, and which indicates the strength of the relationship (how much of the variance in the dependent variable is explained by the independent variables in the equation). The partial regression coefficients for each independent variable (i.e., the "m" in $y = mx + b$) are given. Additionally, the standard deviation of the dependent and independent variables and their partial regression coefficients are determined. These standardized regression coefficients show the relative weight of each independent variable in contributing to the variation in the dependent variable. The models were developed using the mean values from each of the original nine rivers. The models were tested in a cross-validation study using the mean values from the five additional "test" rivers added in 2002. To test the model, the independent variables were multiplied by their respective partial regression coefficients to generate dependent values. These estimated values were compared to the actual dependent values with a correlation analysis (Spearman's or Pearson, depending on normality of variables).

3.2.6.1 Relationships between independent variables

A correlation analysis was used to assess the relationships between the independent variables (habitat and water chemistry values), and reduce the number of variables for later analysis. As not all variables were normally distributed, Spearman's correlations were used. If variables were highly correlated, only one of the variables was used for future analyses. Variables were considered highly correlated when $p<0.05$ and $r > 0.750$ ($r^2 > 0.563$).

3.2.6.2 Relationships between dependent and independent variables

A stepwise multiple regression was used to determine if a dependent variable (biotic variables) could be predicted by the multiple independent variables (habitat and water chemistry values), and to generate an equation of a line, or model, that best explained this relationship. This test provides an overall r^2 value, which is the coefficient of multiple determination, and which indicates the strength of the relationship (how much of the variance in the dependent variable is explained by the independent variables in the equation). The partial regression coefficients for each independent variable (i.e., the "m" in $y = mx + b$) are given. Additionally, the standard deviation of the dependent and independent variables and their partial regression coefficients were determined. These standardized regression coefficients show the relative weight of each independent variable in contributing to the variation in the dependent variable (Grimm and Yarnold 1995).

The models were developed using the mean values from each of the original nine rivers. The fifteen predictor variables determined from the correlation analysis were used as the independent variables. The dependent variables that were tested were total density, variability in total density, brook trout density, variability in brook trout density, total percent habitat saturation (PHS), variability in PHS, brook trout PHS, variability in brook trout PHS 0+ brook trout condition factor, 1+ brook trout condition factor, and all stable isotope analysis results ($\delta^{13}C$, $\delta^{15}N$, %C, and %N). Each dependent variable was tested separately. Models were considered valid if $p \leq 0.05$. The models were then tested in a cross-validation study using the mean values from the five additional "test" rivers added in 2002. To test the model, the independent variables were multiplied by their respective partial regression coefficients to generate predicted values for each dependent variable. These estimated values were compared to the actual dependent values with a correlation analysis (Spearman's or Pearson, depending on normality of variables).

3.3 RESULTS

3.3.1 Description of Salmonid Populations

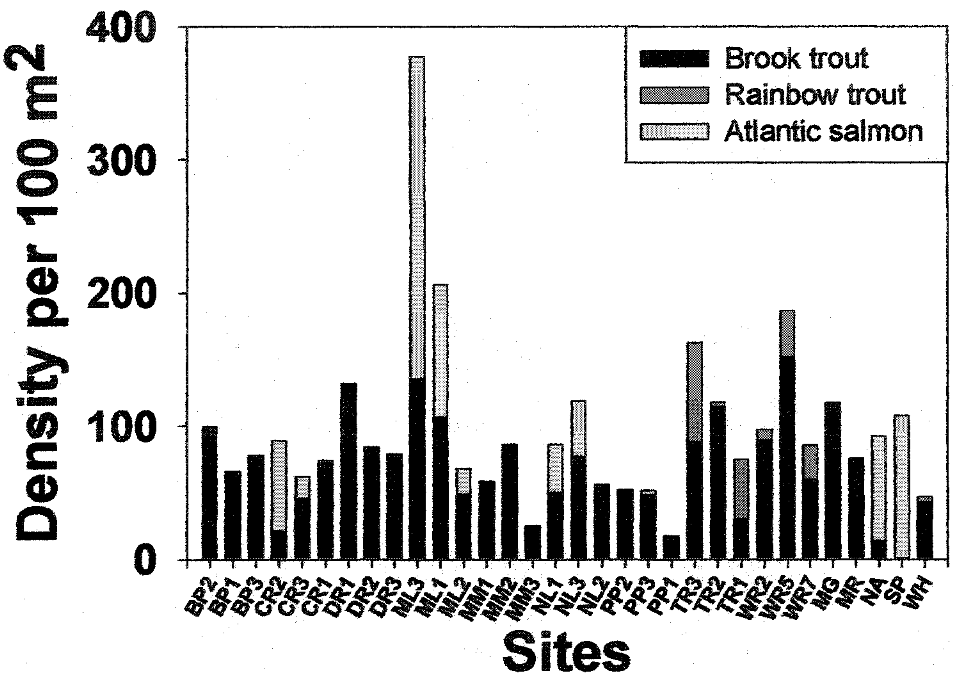
Three salmonid species were captured in streams of Prince Edward Island (Table 3-5; see Fig. 3-2, p. 35 for site locations). Brook trout were found in all 14 rivers examined. Atlantic salmon were found in eight of the rivers, although were rare (mean of samples <1 fish/100m^2) in three of these (PP, TR, and WR). Rainbow trout were found in five of the rivers. All three salmonid species were found in only two rivers (TR and WR); however, the density of salmon at the sites on these rivers never exceeded two individuals/100 m^2 in any sample, and salmon were captured in less than 15% of the samples of the rivers. Higher density populations of salmon were found in the eastern (NL, CR, NA, SP) and western (ML) sections of the province; rainbow trout populations were highest in the central region of the province (TR, WR, WH). The rivers in the figures are organized geographically from west to east; within each river, sites are numbered from upstream to downstream.

The average density of salmonids for each site for the six sampling periods (three sampling periods for sites MG, MR, NA, SP, WH) ranged from 18 to 378 fish/100m^2 (Fig. 3-6a). Overall, brook trout were the dominant species, but Atlantic salmon were more numerous than brook trout in four of the sites (CR2, ML3, NA, and SP). Rainbow trout were the dominant species by density in only one (TR1) of the thirty-seven sites. Mean densities of salmonids for each river (sites combined) ranged from 39 to 217 fish/100m^2 (Fig. 3-6b). Brook trout

Table 3-5. Salmonid species captured (presence indicated by a check mark) in the 14 rivers sampled. Two-letter river abbreviations are given after river name.

River	Atlantic salmon	Brook trout	Rainbow trout
Big Pierre Jacques (BP)		✓	
Cross River (CR)	✓	✓	
DeSable River (DR)		✓	✓
Mill River (ML)	✓	✓	
Miminegash River (MM)		✓	
Montague River (MG)		✓	✓
Montrose River (MR)		✓	
Naufrage River (NA)	✓	✓	
North Lake Creek (NL)	✓	✓	
Priest Pond Creek (PP)	✓	✓	
St. Peters River (SP)	✓	✓	
Tryon River (TR)	✓	✓	✓
Wheatley River (WH)		✓	✓
Wilmot River (WR)	✓	✓	✓

A



B

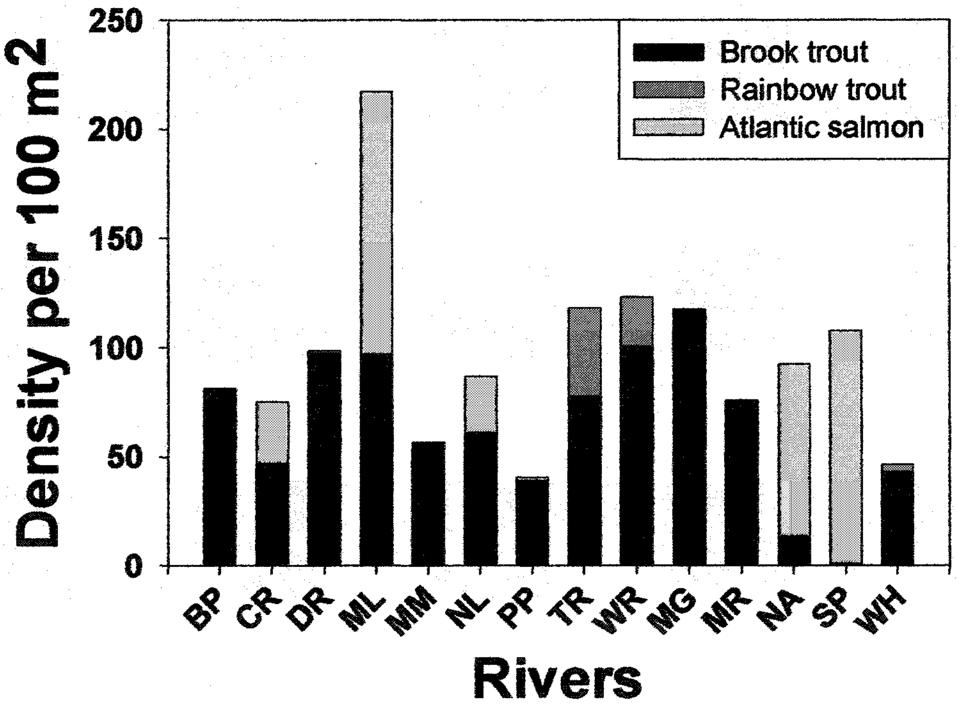


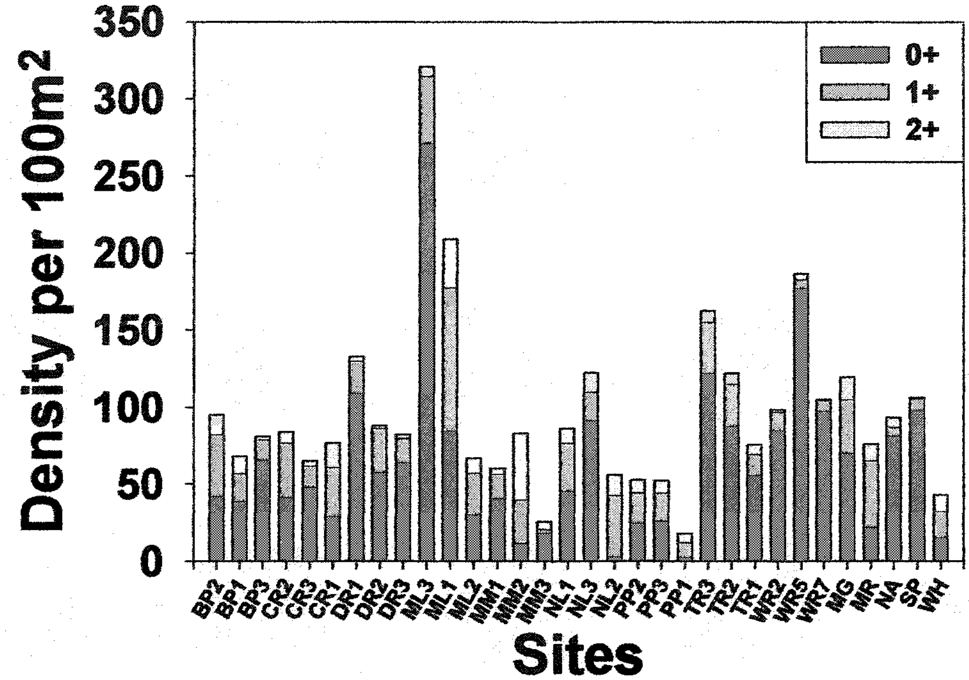
Figure 3-6. Average density of brook trout (black), rainbow trout (red), and Atlantic salmon (green) in sites (A) and rivers (B) sampled during 2001 and 2002.

had the highest density in all but three rivers (ML, NA, and SP), which were dominated by Atlantic salmon. The majority of salmonids captured at most sites (Fig. 3-7a) and rivers (Fig. 3-7b) were young of the year (0+ age).

The average percent habitat saturation (PHS) for each site for the six sampling periods (three for sites MG, MR, NA, SP, WH) ranged from 16-144% (Fig. 3-8a). Brook trout occupied the largest percentage of the habitat in all but the four sites (CR2, ML3, NA, and SP) in which Atlantic salmon dominated. Rainbow trout were not the dominant salmonid in any of the sites. The mean PHS for each river ranged from 17 to 60%. Brook trout had the highest PHS rating in all but two rivers (NA and SP) in which Atlantic salmon were higher (Fig. 3-8b).

The condition factors of the three salmonid species differed significantly from one another ($p<0.001$, ANOVA; $p<0.001$, Tukey's paired comparisons; Fig. 3-9). Rainbow trout had the highest condition factor ($1.29 \pm 4.08 \times 10^{-3}$), followed by Atlantic salmon ($1.21 \pm 1.17 \times 10^{-2}$), and lastly brook trout ($1.15 \pm 8.71 \times 10^{-3}$). Comparisons between sites therefore were limited to within-species comparisons. Within each species, the different age classes also had different condition factors (Fig. 3-10), which also limited comparisons between sites to specific age classes. Brook trout 0+ fish had a significantly higher condition factor ($p<0.001$, ANOVA) than 1+ ($p<0.001$, Tukey's) and 2+ ($p=0.001$, Tukey's) fish. There was no difference between 1+ and 2+ brook trout. 0+ Atlantic salmon had a significantly lower condition factor than 2+ salmon

A



B

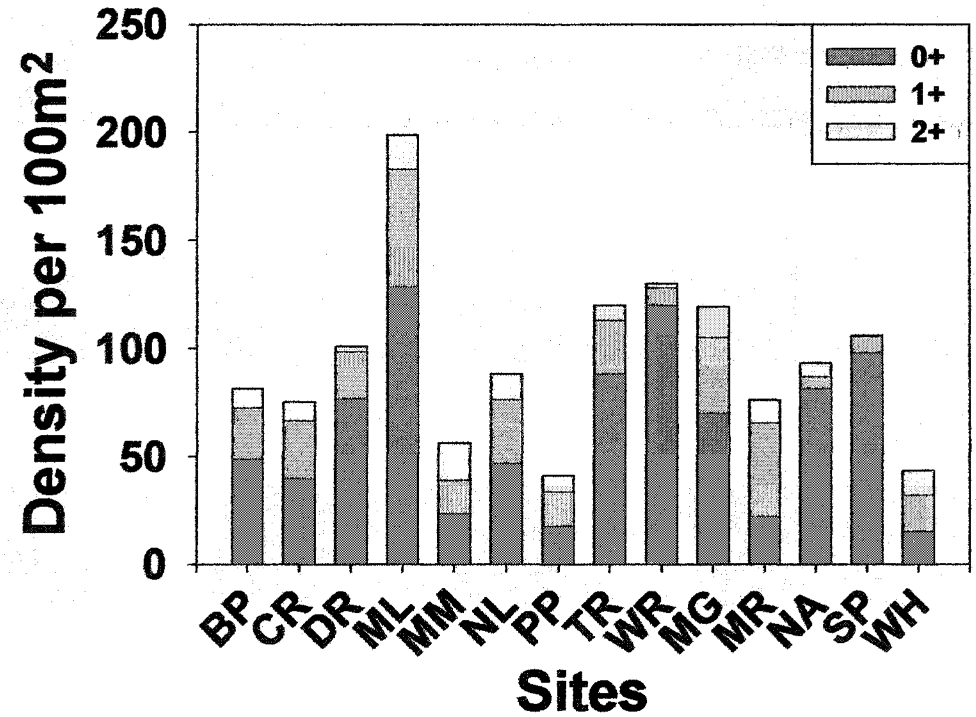


Figure 3-7. Average density of 0+ (black), 1+ (red), and 2+ (green) salmonids in sites (A) and rivers (B) sampled during 2001 and 2002.

A

B

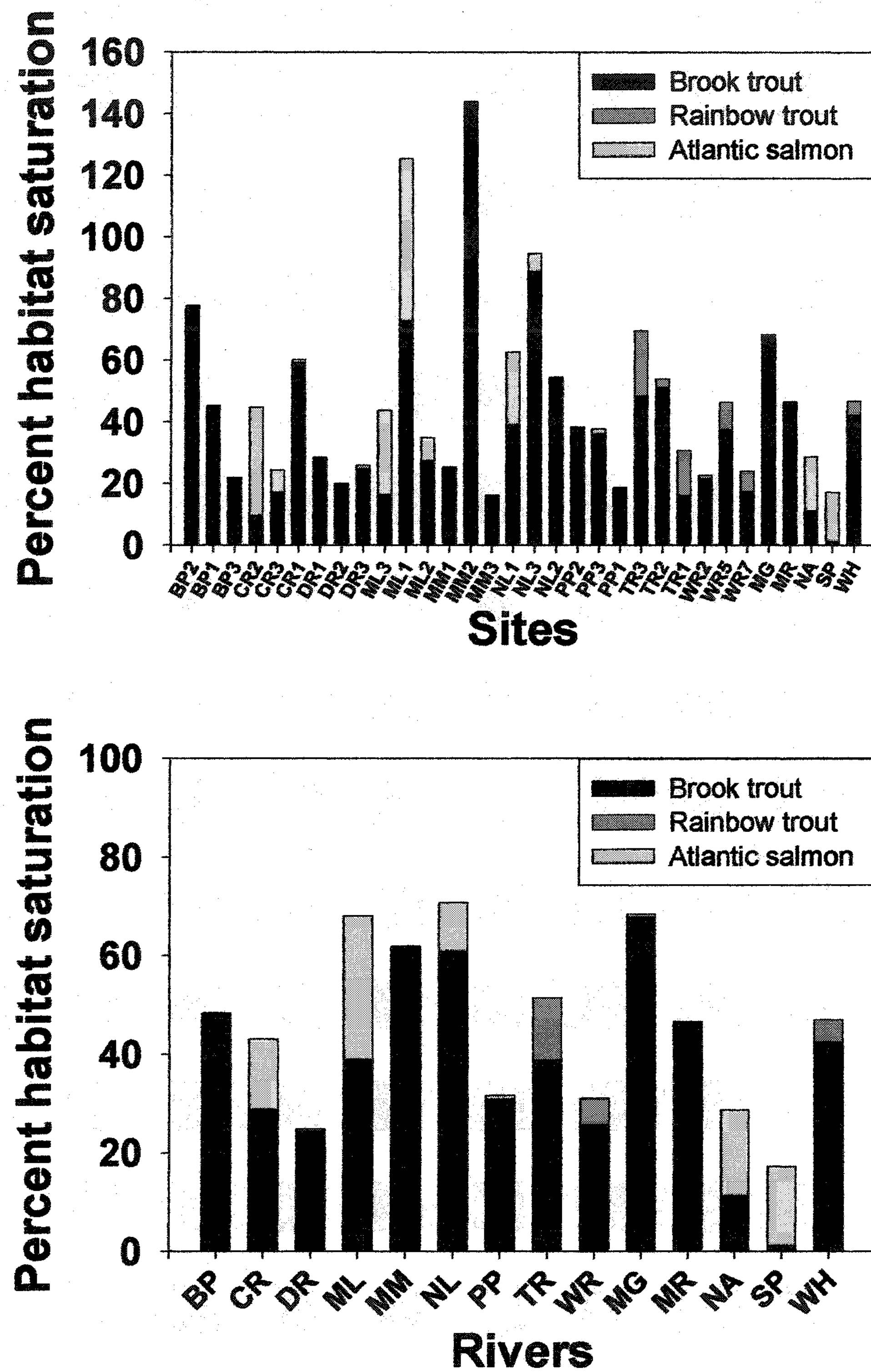


Figure 3-8. Average percent habitat saturation (PHS) of brook trout (black), rainbow trout (red), and Atlantic salmon (green) in sites (A) and rivers (B) sampled during 2001 and 2002.

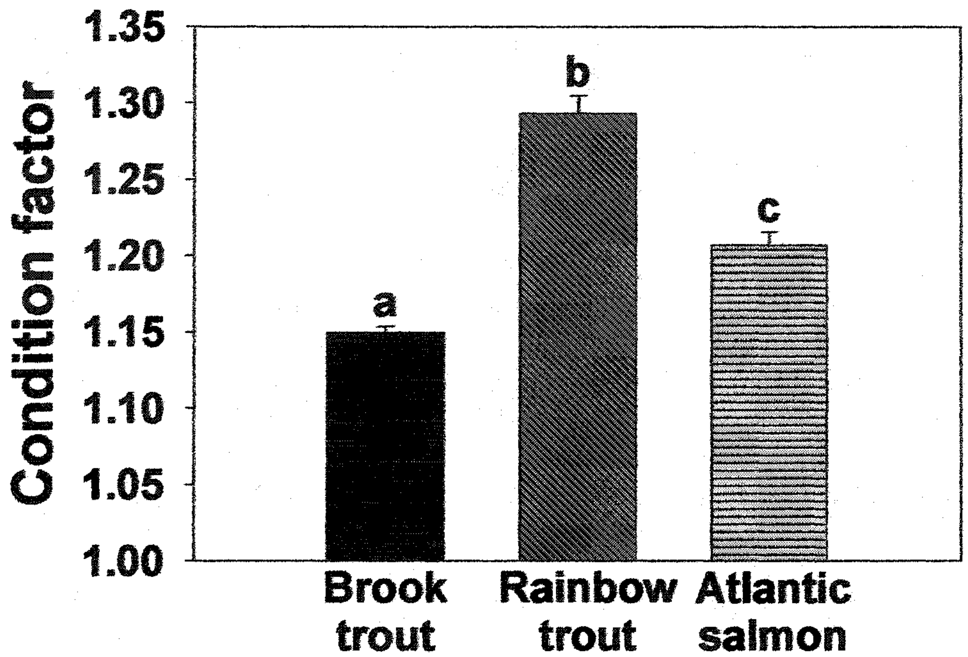


Figure 3-9. Average condition factors (± SE) for brook trout, rainbow trout, and Atlantic salmon captured during 2001 and 2002 sampling. Bars with different letters are significantly different from one another ($p<0.001$, ANOVA; $p<0.001$, Tukey's paired comparisons).

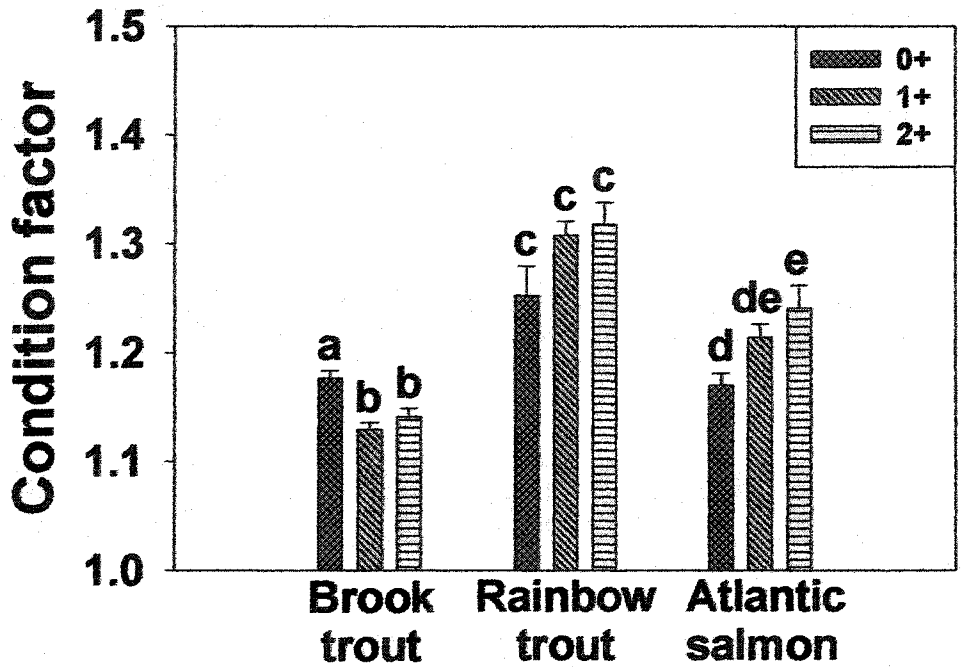


Figure 3-10. Average condition factors (± SE) for 0+ (blue), 1+ (green), and 2+ (yellow) brook trout, rainbow trout, and Atlantic salmon captured during 2001 and 2002 sampling. Within each species, bars with different letters are significantly different from one another ($p<0.01$, ANOVA; $p<0.01$, Tukey's paired comparisons).

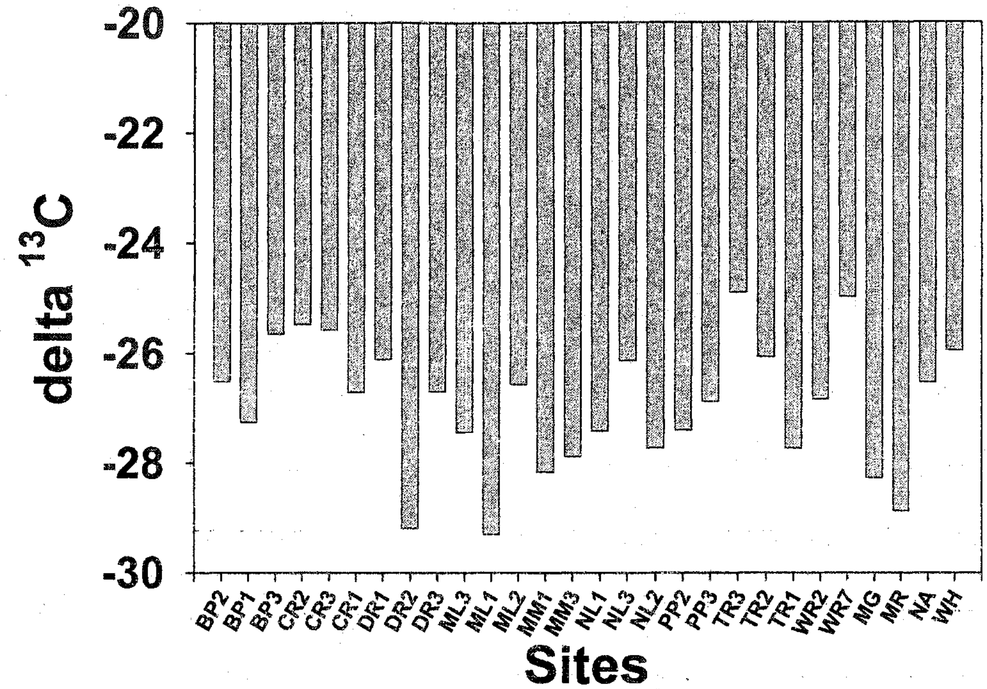
(p<0.01, ANOVA, p<0.01, Tukey's). There was no difference between 1+ salmon and any other age class. There was no significant difference between age classes in rainbow trout (p=0.061, ANOVA).

The $\delta^{13}C$ and $\delta^{15}N$ ratios were obtained with stable isotope analysis for the 27 sites where sufficient young of the year tissue samples were collected. (There was insufficient tissue collected from sites MM2, PP1, and SP to perform the analysis.) Additionally, the percent carbon (%C), percent nitrogen (%N), and carbon to nitrogen (C:N) ratio were determined. The mean $\delta^{13}C$ ratio was -27.03 ± 0.27 (SE) (Fig. 3-11A). This negative result indicates the sample is depleted, or light, compared to the international standard. The mean $\delta^{15}N$ value was 7.10 ± 0.30 (SE) (Fig. 3-11B). The positive values for $\delta^{15}N$ indicate the sample is enriched, or heavy, compared to the international standard. The percent of carbon in the samples averaged 44.52 ± 0.76 (SE) (Fig. 3-12A), while that of nitrogen averaged 12.94 ± 0.22 (SE) (Fig. 3-12B).

3.3.2 Description of Abiotic Factors

The mean values in each river for the water chemistry variables, instream habitat variables, riparian zone variables, and watershed variables are given in Appendix B. Values for Priest Pond Creek (PP) use the mean of only two sites (PP2 and PP3: see below). The mean values for each site are given in Appendix A.

A



B

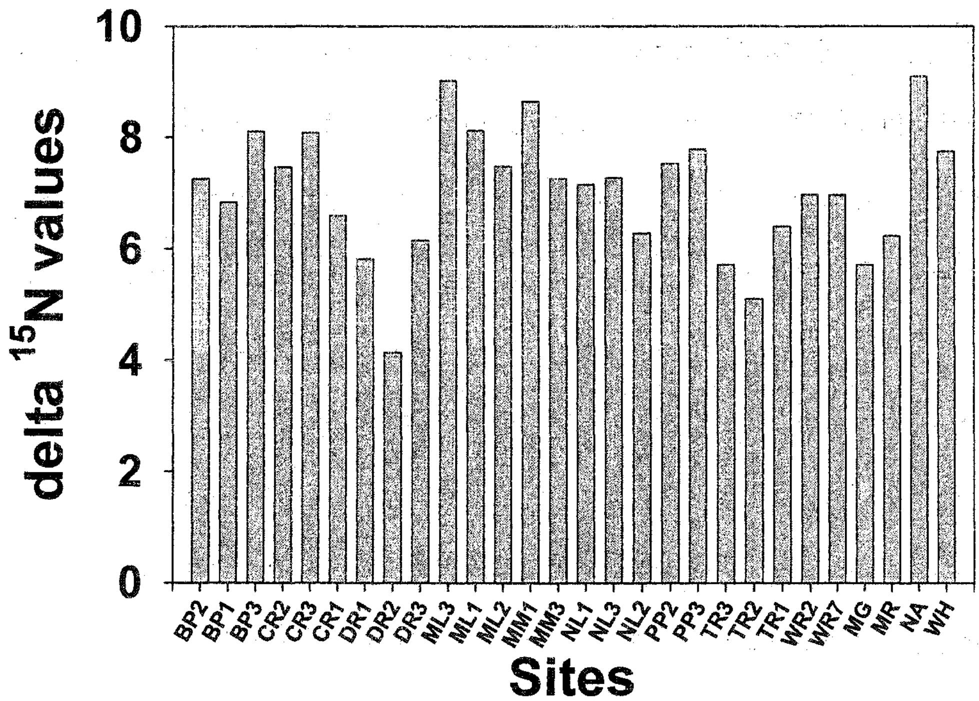


Figure 3-11. (A) δ^{13}C and (B) δ^{15}N values from stable isotope analyses from 0+ brook trout tissue samples.

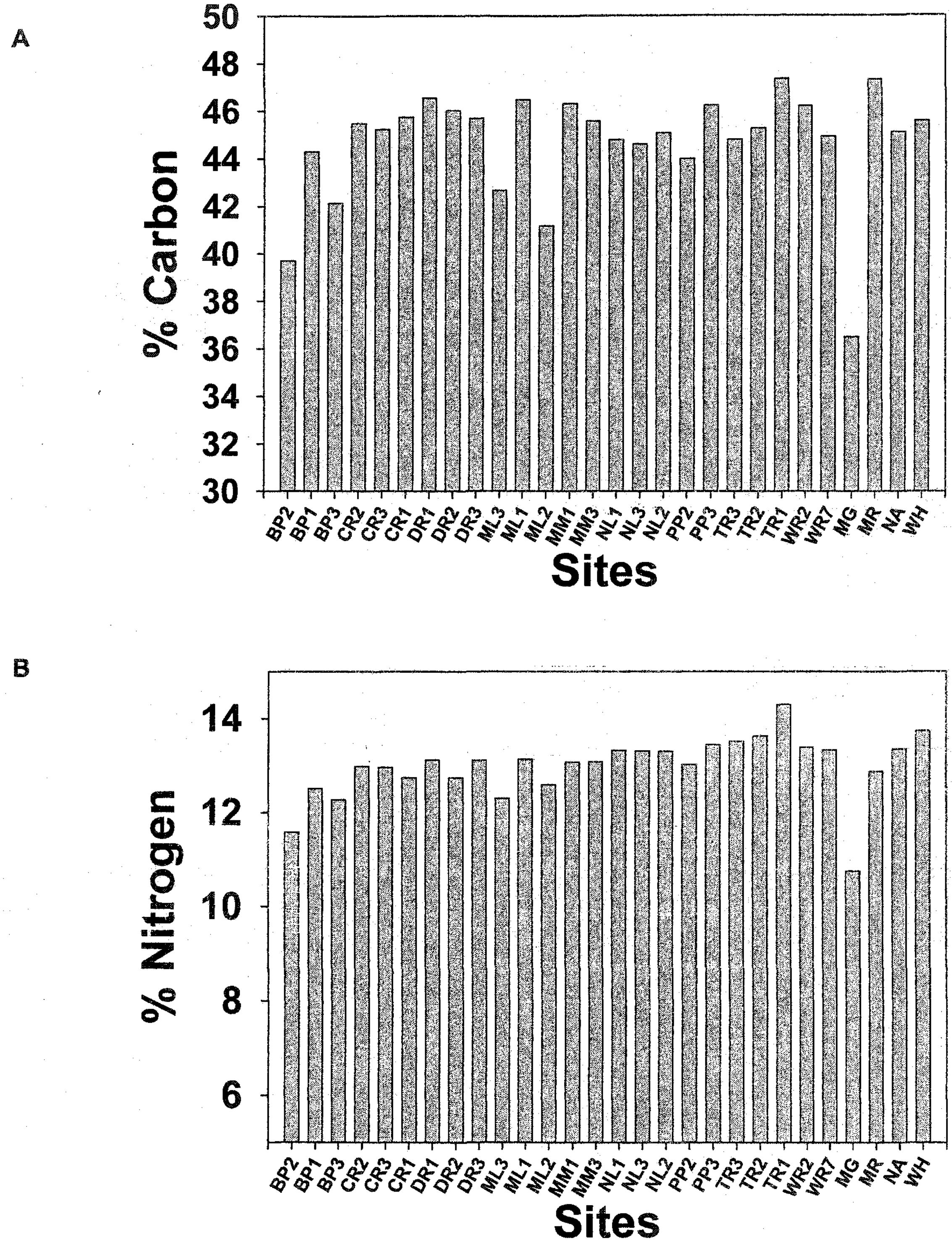


Figure 3-12. (A) Percent carbon and (B) percent nitrogen values from stable isotope analyses from 0+ brook trout tissue samples.

3.3.3 Outlier Analysis

When sites for each river were combined, only one site was found to be an outlier (based on examination of box-plot diagrams). This site, PP1, differed in both physical (high temperature, small substrate, few trees in riparian zone) and biological (as low as 1.3 brook trout per 100 m^2) attributes from the other sites. A beaver dam was removed from this site in June 2002. There is also a large impoundment above the site that warmed the water up to 28.5 °C. This site was removed from analysis for model development, and the mean of the other two sites on Priest Pond Creek (PP) was used.

3.3.4 Model Development

3.3.4.1 Relationships between independent variables

Intercorrelations among variables ($p<0.05$, $r^2 \geq 0.55$, Spearman's correlation) meant that the original 43 variables were reduced to 15. The 15 variables that were retained, as well as the variables that were correlated with them, are given in Table 3-6. Diameter at breast height (DBH) and number of trees in riparian zone were discarded, as it was concluded that a 1 m wide transect did not provide a representative sample of the riparian zone. Phosphorous readings were also discarded, as it was suspected that the inconsistently high readings (up to 0.240 mg/L) may have been due to contamination in the sample bottle or during sampling. None of the biotic (dependent) variables were correlated with one another.

Table 3-6. Relationships among independent variables. Variables that were retained for use in regression are in the first column; variables that were correlated with these retained variables ($p<0.05$, $r^2 \geq 0.55$, Spearman's correlation) are in the second column. The direction of the relationship is indicated by + (positive) or - (negative). Bankfull depth, dissolved oxygen, and alkalinity did not have any other variables correlated with them.

Retained variable	Correlated variable(s)
Rocksize	% urban
Wentworth rating: walk	Wentworth rating: transect (+); % silt: walk (-); % silt: transect (-); stability (+); Pfankuch evalution score (-)
Solar radiation: all	Solar radiation summer (+); solar radiation winter (+)
Bank full depth (cm)	
Watershed size (km^2)	river length (+); discharge (+); mean width (+); mean wetted width (+)
Road length (km/km^2)	
% wetland	stability (+); Pfankuch evalution score (-)
% potato rotation	% forest (-); nitrate (+)
% other agriculture	Pfankuch evalution score (+)
Dissolved oxygen (mg/L)	
pH	discharge (+)
Conductivity (μmhos)	Average riparian slope over 60 m: average of both banks (+)
Maximum temperature (°C) (June-Septempber)	Mean June temperature (°C) (+); Mean July temperature (°C) (+); Mean August temperature (°C) (+); Mean September temperature (°C) (+); discharge (+)
Velocity (m/s)	
Alkalinity (mg/L)	

3.3.4.2 Models

Models explained a significant proportion of the variation ($p<0.05$) for six of the biotic variables: total PHS, brook trout density, 1+ brook trout condition factor, $\delta^{15}N$, variability in total density, and variability in percent habitat saturation (Table 3-7).

Brook trout density was explained by only one measured factor, the amount of land in the watershed in potato rotation (positive relationship, +) (Table 3-7). Although this was a significant correlation ($p<0.05$), only 37.4% of the variation in density was explained by the model (Fig. 3-13A). The brook trout densities predicted by this model were not significantly correlated with the observed values when the five test rivers were used ($p>0.05$, $r^2 = 0.213$, Pearson correlation; Fig.3-13B).

The percent variability in total salmonid density was explained by the amount of solar radiation reaching the stream bed (+) and the amount of land in potato rotation (-) ($p=0.001$, adj. $r^2 = 0.859$; Table 3-7; Fig. 3-14A). When this model was tested using the five test rivers the correlation between the actual and estimated densities were not significant ($p>0.05$, $r^2 = 0.048$, Pearson correlation; Fig. 3-14B).

The total percent habitat saturation was best explained by the velocity of the water (-), the amount of wetland in the watershed (+), and the Wentworth rating determined from the random walk (+) (Table 3-7). The model that was developed had an adjusted r^2-value of 0.978 ($p<0.001$; Fig. 3-15A). The

Table 3-7. Dependent variables and the predictor variables and partial regression coefficients used in the models developed. The adjusted r^2-value of each model is also given.

Dependent variable (y)	**Equation**							**r^2 of model**
	Predictor 1 (x_1)	Partial regression coefficient 1 (m_1)	Predictor 2 (x_2)	Partial regression coefficient 2 (m_2)	Predictor 3 (x_3)	Partial regression coefficient 3 (m_3)	Constant (b)	
brook trout density	% potato rotation in watershed	0.581					56.284	0.374
variability in total density	% solar radiation	4.677	% potato rotation in watershed	1.544			-154.289	0.859
total percent habitat saturation	% wetland in watershed	3.122	velocity (m/s)	-175.88	wentworth rating (walk)	11.997	60.901	0.978
variability in total percent habitat saturation	% solar radiation	4.516	watershed size	-3.547			-34.474	0.800
1+ brook trout condition factor	dissolved oxygen (mg/L)	0.37					0.744	0.563
$\delta^{15}N$ value	alkalinity (mg/L)	0.430	maximum temperature (°C)	0.269	mean rock size (walk)	0.091	-2.401	0.985

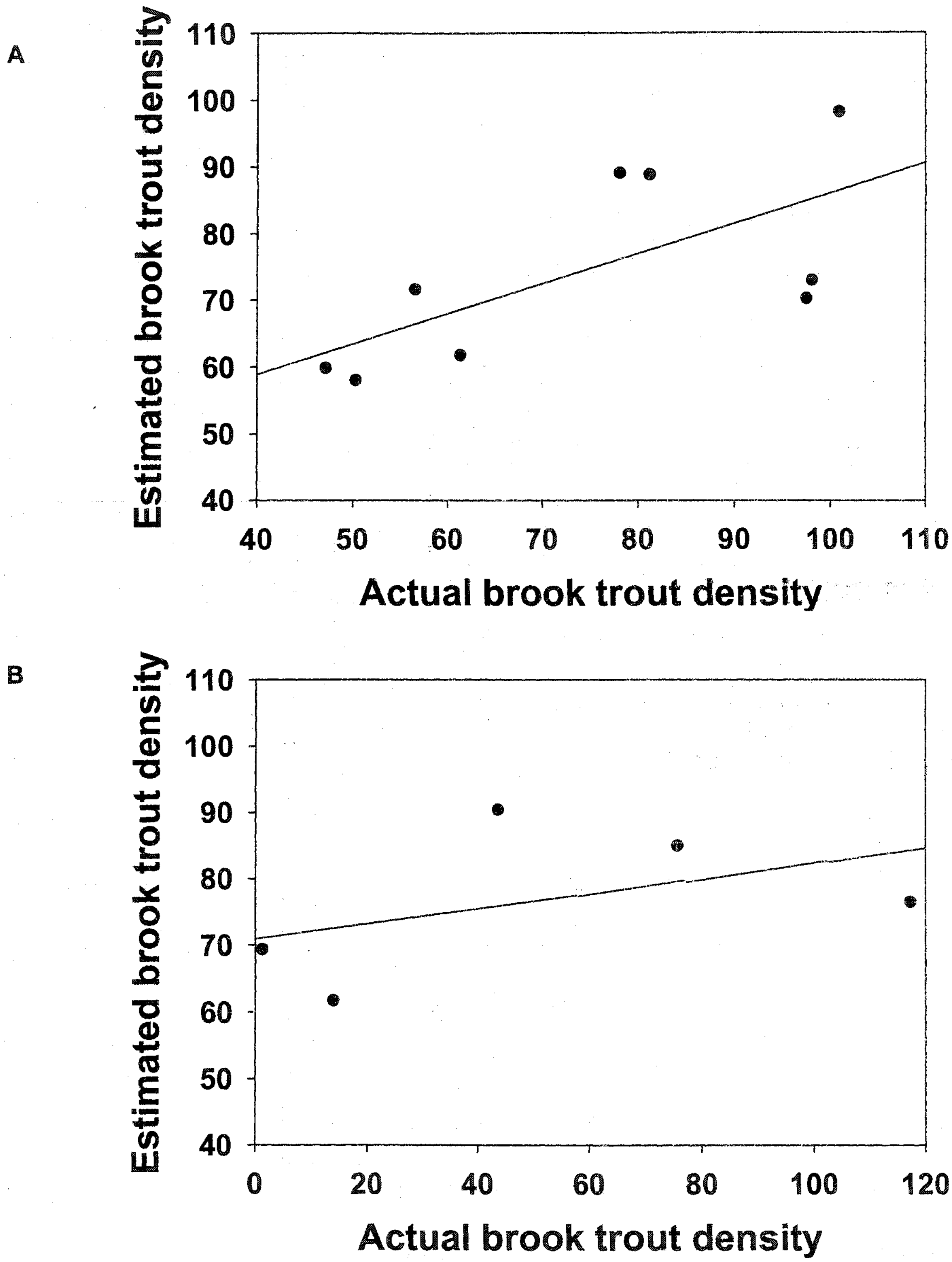
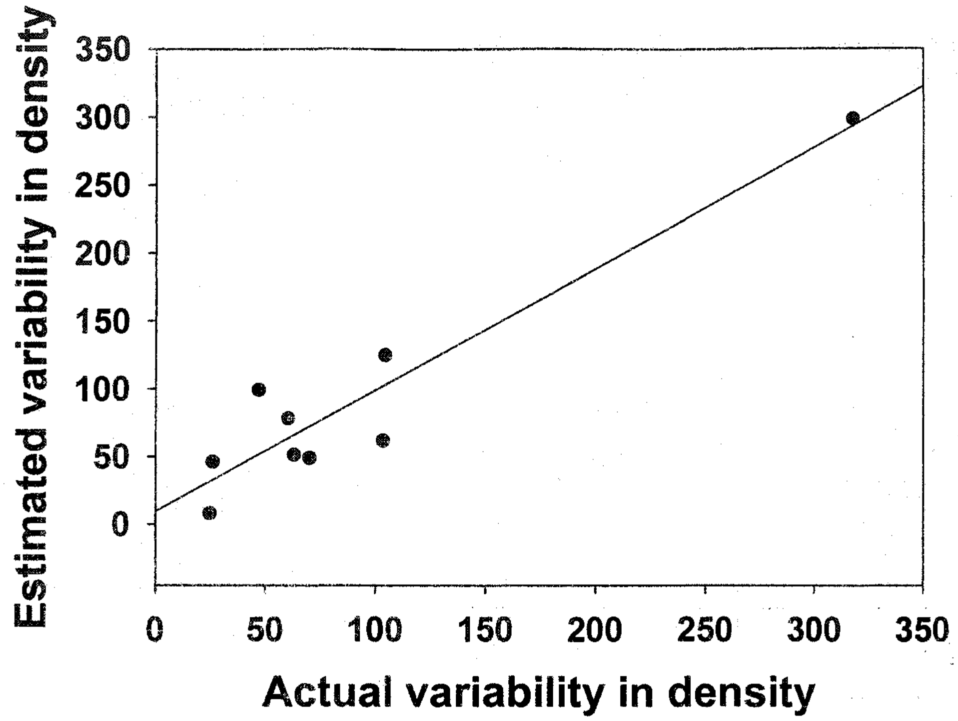


Figure 3-13. Brook trout density (fish/100m^2) models for (A) the nine rivers used to develop the model ($p<0.05$, r^2 = 0.374) and (B) the five rivers used to test the model ($p>0.05$ Pearson correlation).

A



B

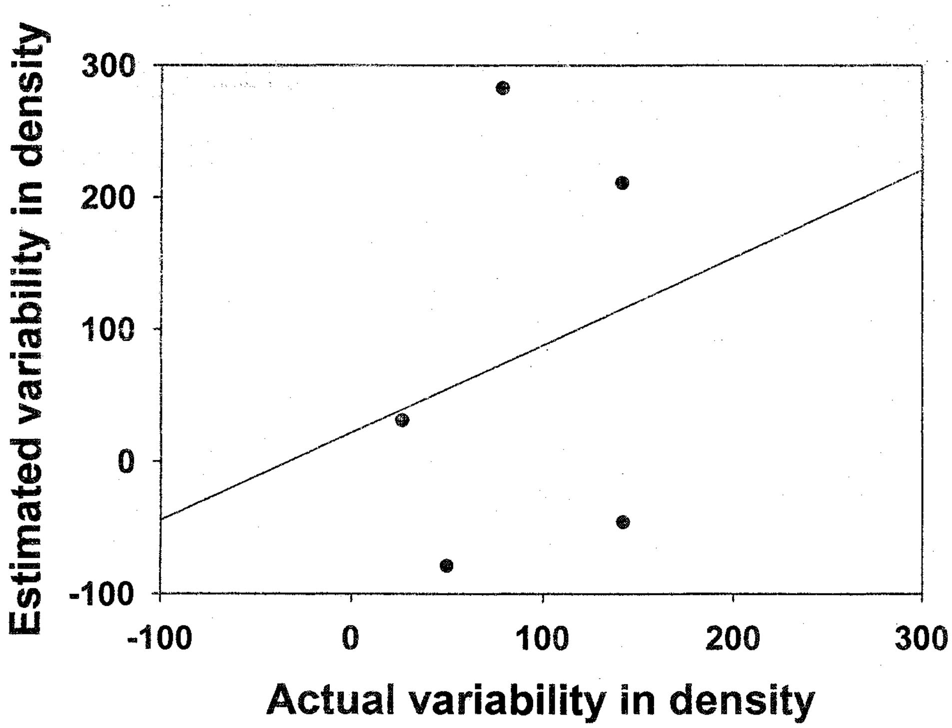


Figure 3-14. Variability in density models for (A) the nine rivers used to develop the model ($p=0.001$, $r^2 = 0.859$) and (B) the five rivers used to test the model ($p>0.05$, Pearson correlation).

A

B

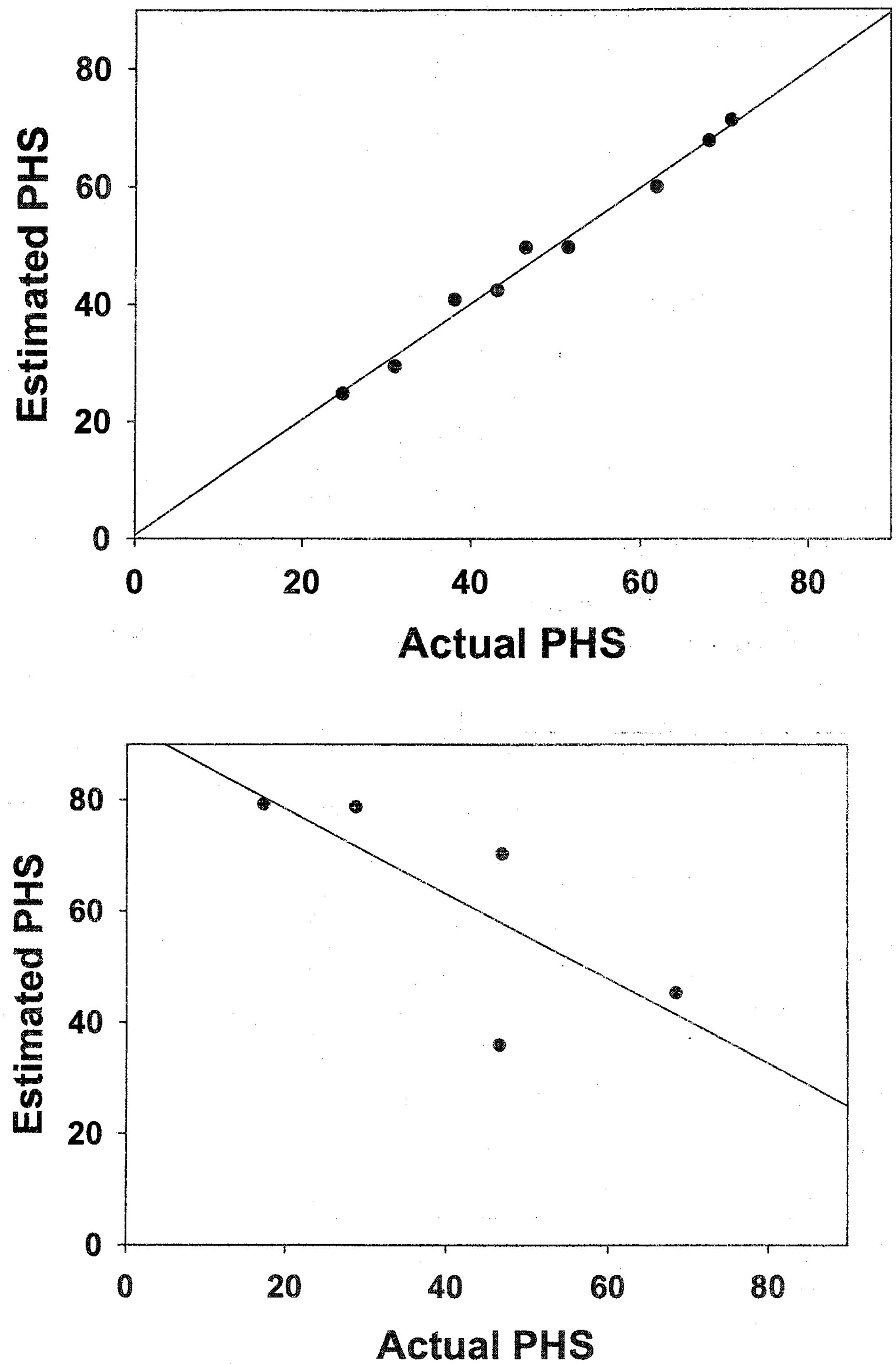


Figure 3-15. Percent habitat saturation (PHS) models for (A) the nine rivers used to develop the model ($p<0.001$, $r^2 = 0.978$)and (B) the five rivers used to test the model ($p>0.05$ Pearson correlation).

percent habitat saturation was not significantly predicted for the five rivers used to test the model ($p>0.05$, $r^2 = 0.555$, Pearson correlation; Fig.3-15B, p. 67).

The percent variability in total PHS was explained by the amount of solar radiation reaching the stream bed (+) and the size of the watershed (-) ($p<0.01$, adj. $r^2 = 0.800$; Table 3-7, p. 64; Fig. 3-16A). This model did not prove valid when the values from the model were compared to the observed values ($p>0.05$, $r^2 = 0.490$, Pearson correlation; Fig. 3-16B).

The condition factor of 1+ brook trout was positively related to the amount of dissolved oxygen in the water ($p<0.05$, adj. $r^2 = 0.563$; Table 3-7, p. 64; Fig. 3-17A). This model did not prove applicable when the generated condition factors were compared to the actual values for the five test rivers ($p>0.05$, $r^2 = 0.132$, Pearson correlation; Fig.3-17B).

The $\delta^{15}N$ ratios obtained from the 0+ brook trout tissue samples were explained by the maximum temperature (+), alkalinity (+), and mean rock size (+) ($p<0.001$, adj. $r^2 = 0.985$; Table 3-7, p. 64; Fig. 3-18A). When the four test rivers that had brook trout tissue samples were used in the model a significant correlation was found between the actual and estimated $\delta^{15}N$ values ($p<0.01$, $r^2 = 1.000$, Spearman's correlation; Fig. 3-18B).

A

B

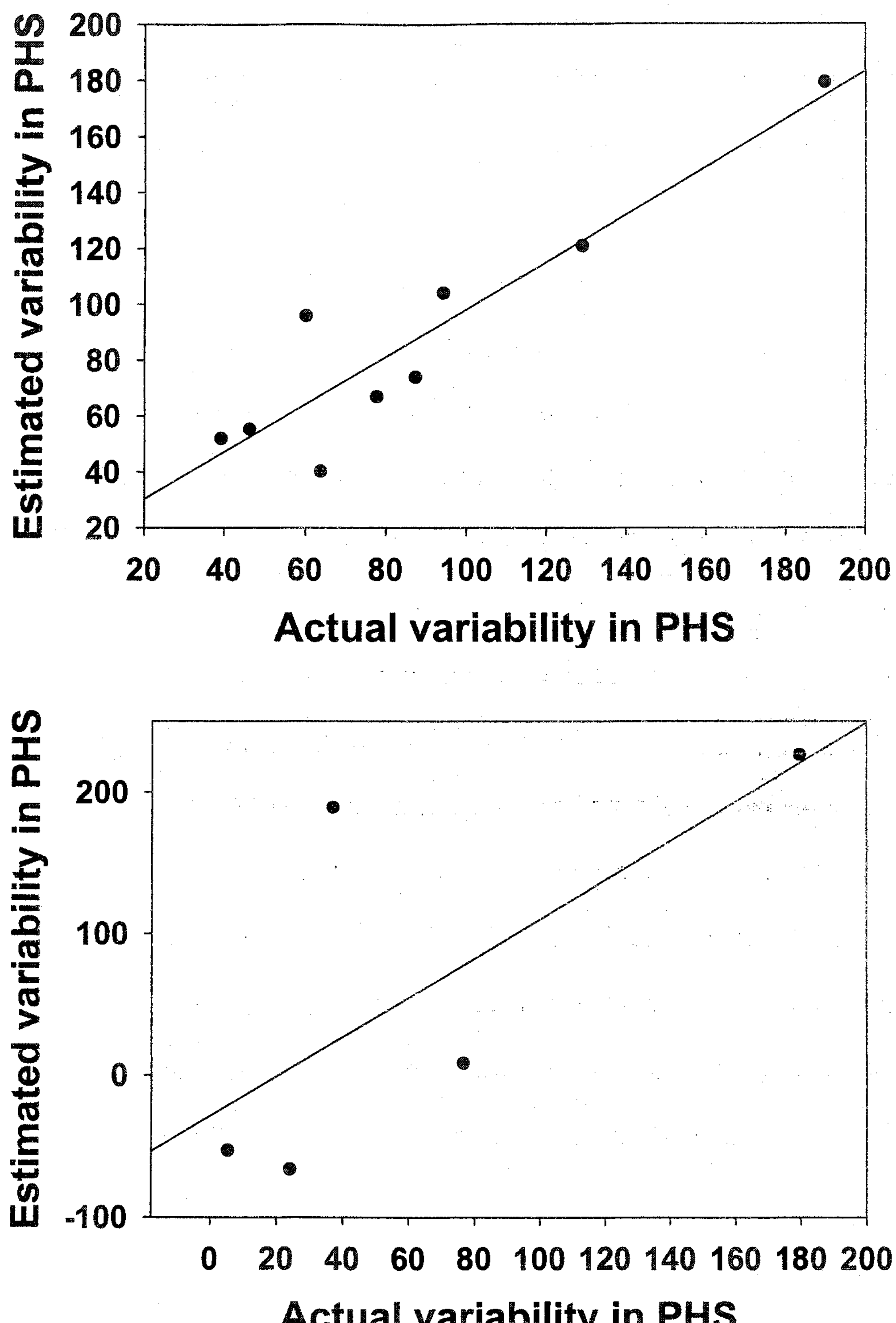


Figure 3-16. Variability in percent habitat saturation (PHS) models for (A) the nine rivers used to develop the model ($p < 0.01$, $r^2 = 0.800$) and (B) the five rivers used to test the model (p>0.05Pearson correlation).

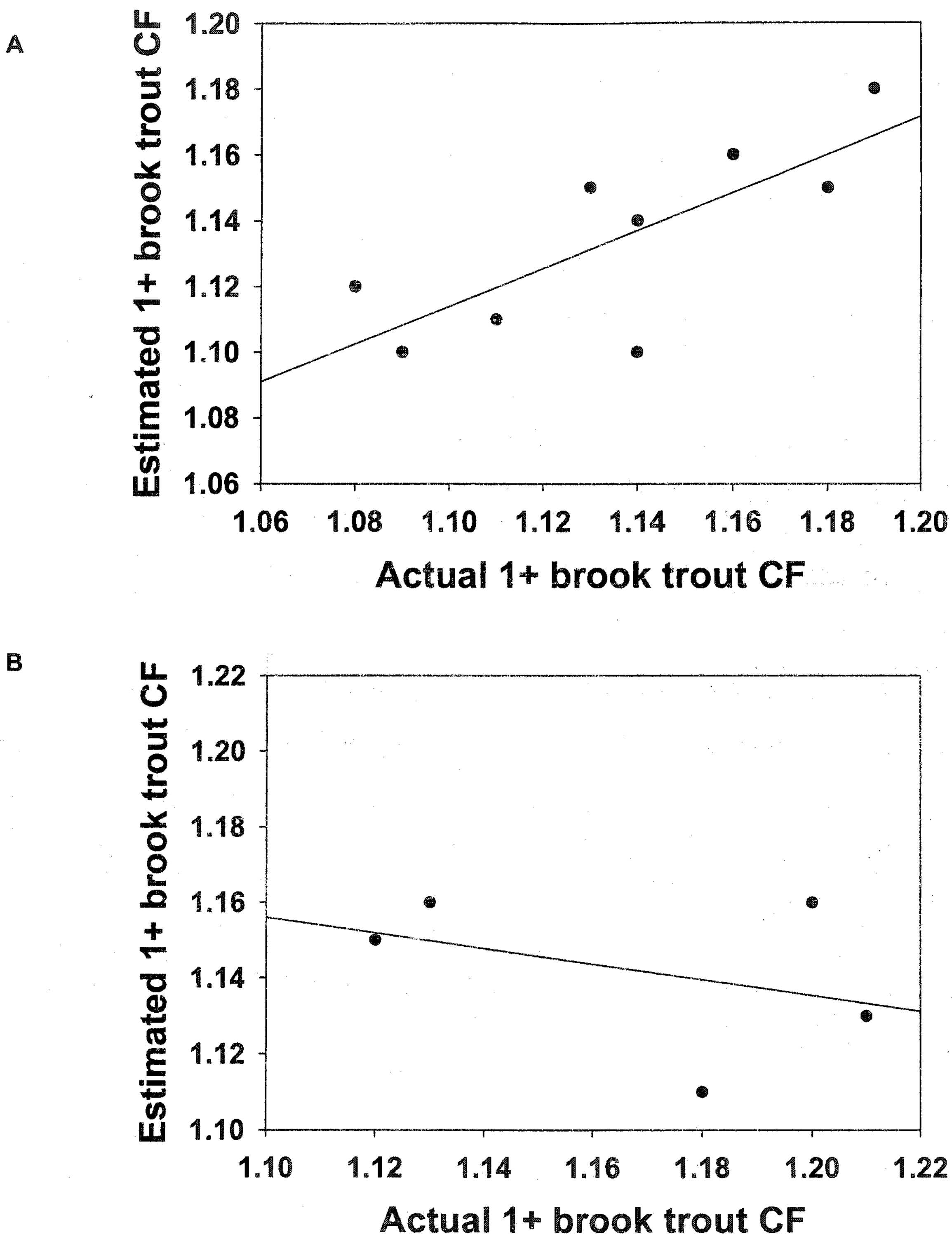
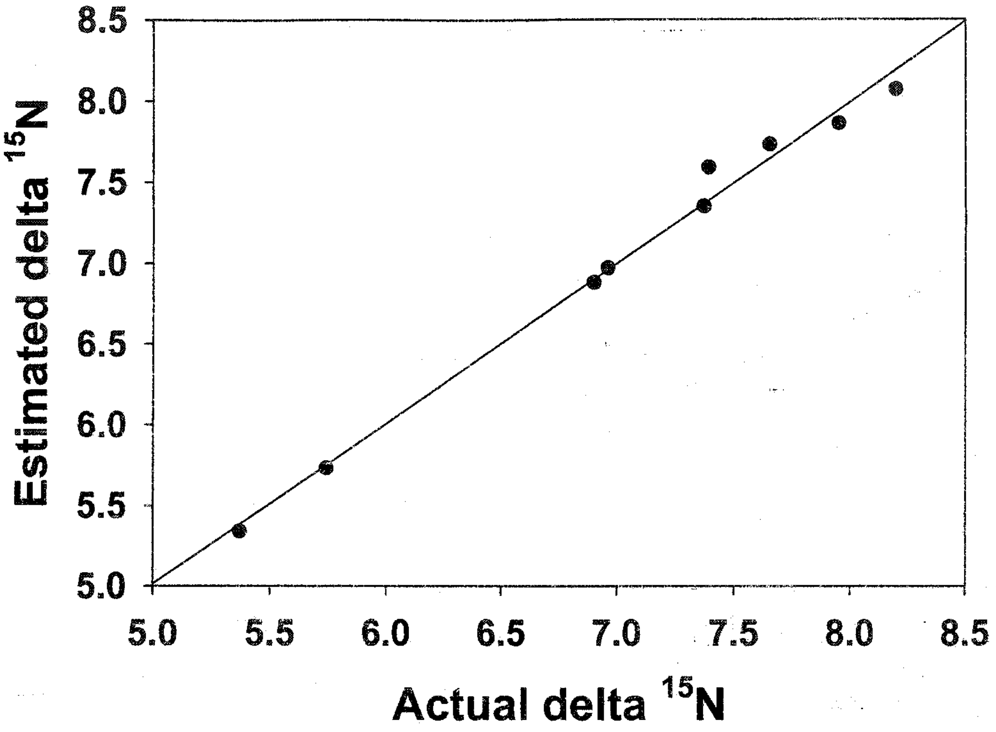


Figure 3-17. Brook trout condition factor (CF) models for (A) the nine rivers used to develop the model ($p<0.05$, $r^2 = 0.563$) and (B) the five rivers used to test the model ($p>0.05$, Pearson correlation).

A



B

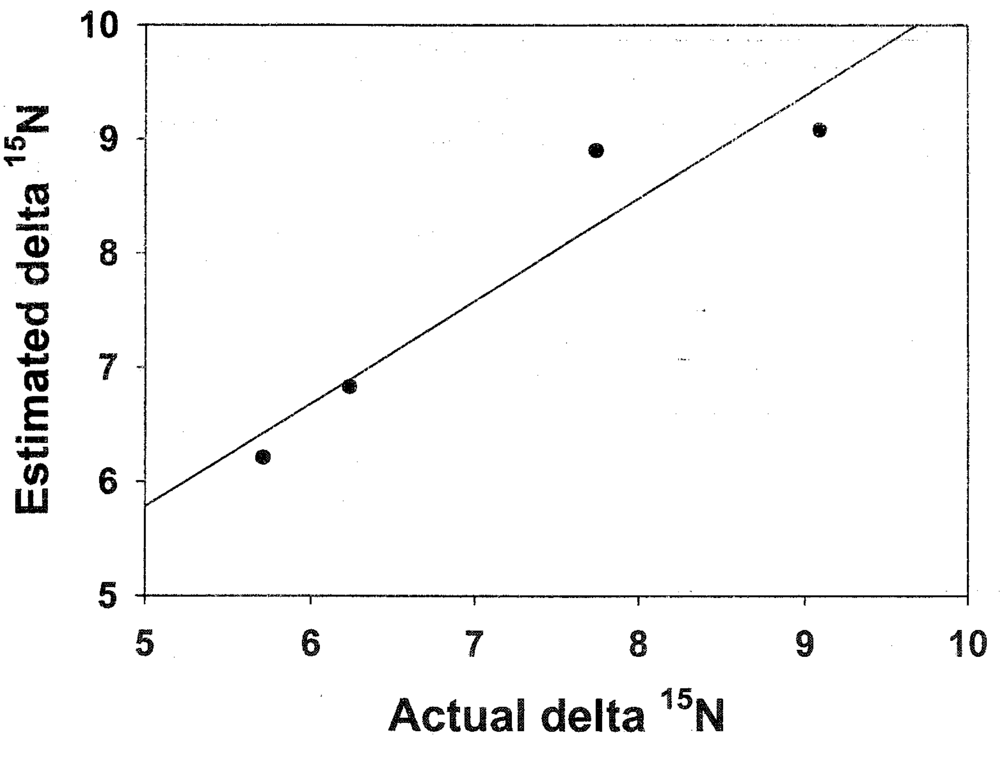


Figure 3-18. $\delta^{15}N$ (from 0+ brook trout stable isotope analysis) models for (A) the nine rivers used to develop the model ($p<0.01$, $r^2 = 0.985$) and (B) the five rivers used to test the model ($p<0.01$, $r^2 = 1.000$, Spearman's correlation).

3.4 DISCUSSION

The results of this study add significantly to our understanding of factors affecting the abundance and distribution of salmonids on Prince Edward Island. Brook trout were found in all streams examined; Atlantic salmon and rainbow trout were more limited in their distribution, and their populations may be affected by historical or current anthropogenic effects. Regression models using various stream and watershed variables explained a significant proportion of the variation in a number of characteristics describing salmonid populations. However, when these models were applied to five "test" rivers, they were largely unsuccessful in predicting salmonid condition, density, or habitat saturation.

3.4.1 Distribution of Salmonids

Brook trout are known to be widespread throughout Prince Edward Island (Cairns 2002b, Guignion et al. 2002). Their populations appear to be healthy in all areas except those affected by anthropogenic effects such as impoundments (Priest Pond Creek, PP1) and agriculture-associated pesticide runoff (Wilmot River, WR; see Chapter 4). Brook trout are historically found throughout much of North America (Scott and Crossman 1973), and have widespread, stable populations throughout Atlantic Canada (Gibson et al. 1993). The brook trout populations within Atlantic Canada (PEI, New Brunswick (NB), Nova Scotia (NS), and Newfoundland (NFLD)) are believed to have originated from the same population after the glaciers left the region. Mitochondrial DNA analysis of brook trout populations in PEI, NB, and NS indicate that these populations are derived from the same original population (Jones M.W. et al. 1996). Indeed, most (>50%) of the variation between these groups was evident between branches within the same drainage basin. Therefore, brook trout populations throughout PEI are expected to behave similarly to one another, and to populations throughout Atlantic Canada.

Atlantic salmon populations have been declining throughout much of their range in recent years (Parrish et al. 1998). On the east coast of North America, salmon populations are now limited to the more northern section of their range, although they have also been extirpated in some of these rivers. This species was once widespread throughout PEI (Stewart 1806), but is now limited in its range (Guignion et al. 2002). In this study, salmon were found consistently in only five of the fourteen rivers examined. The local extirpations from streams on PEI may be due to regional historical disturbances, such as impoundments that blocked passage to and from the ocean, which is a common cause for extirpation of salmon (MacCrimmon and Gots 1979). For example, in the western branch of the Tryon River, impoundments that blocked the passage of anadromous fish were only recently removed (D. Guignion, Dept. of Biology, University of Prince Edward Island, Charlottetown, PEI , pers. comm.). This branch of the river had no salmon captured during this study. Salmon populations may also be more recently affected. The Wilmot River, which experienced two fish kills in 2002 (pers. obs.), was observed to have Atlantic salmon returns in recent years (S. Hill, PEI Dept. of Fisheries, Aquaculture, and

Environment, Charlottetown, PEI, pers. comm.). However, in this study I found only remnant populations; in 2002 only one 0+ salmon was captured.

As in other areas of eastern North America, rainbow trout are found in areas where they have been introduced intentionally or have escaped from aquaculture facilities. On Prince Edward Island, rainbow trout were stocked in several rivers (e.g., Dunk River, Vernon River, Westmoreland River; Guignion et al. 2002); in addition, rainbow trout escaped from hatcheries to colonize other rivers (e.g., Cardigan River, Montague River, Murray River; Guignion et al. 2002). Their populations have been increasing and expanding in recent years; this is cause for concern as they may out-compete native populations (Guignion et al. 2002).

Atlantic salmon and rainbow trout were rarely found together in rivers across PEI. In the two sites where all three salmonid species were captured, rainbow trout had healthy populations, while only one Atlantic salmon was captured. There is widespread concern that introduced species such as rainbow trout and brown trout may be able to out-compete Atlantic salmon and thus contribute to their decline (Fausch 1998). In a study on intra- and interspecific competition of juvenile Atlantic salmon and rainbow trout, rainbow trout were found to be more aggressive (measured as number of attacks, charges or chases) than Atlantic salmon (Volpe et al. 2001). In addition, rainbow trout and salmon are both found to occupy riffle areas in cool streams, and are both thought to feed primarily on invertebrate drift (Hearn and Kynard 1986), so occupy similar niches. When the two species have been found together, salmon have not occupied as many areas throughout the stream as when they were alone; rainbow trout distribution throughout the stream did not change in the presence of salmon (Hearn and Kynard 1986). In addition, rainbow trout spawn in the spring, which is after Atlantic salmon and brook trout (which both spawn in the fall). This means that rainbow trout could spawn on top of the areas where Atlantic salmon or brook trout spawned (Volpe et al. 2001). This superimposition of one redd on top of the other has been found to reduce the spawning success of the first redd (Hayes 1987).

Atlantic salmon populations may also be affected by the presence of other salmonid species. There is more evidence available on the interactions between Atlantic salmon and brown trout, another introduced species. This species has been reportedly captured on Prince Edward Island, although electrofishing samples have not revealed any individuals in this study or in that of Guignion et al. (2002). Brown trout have been found to compete with salmon for food and shelter (Harwood et al. 2001, 2002), and salmon change their foraging behaviours both spatially and temporally when brown trout are present (Harwood et al. 2001). Atlantic salmon on PEI are always found with brook trout. Although Atlantic salmon are generally more aggressive than brook trout (Gibson 1973 *cited from* Chiasson et al 1990), the removal of brook trout was found to increase growth in Atlantic salmon in areas where food was limiting (Gibson and Dickson 1984 *cited from* Chiasson et al 1990). Less information is available on the effects of exotic species on brook trout, as this species is generally thought to prefer slower, deeper areas than the previous species mentioned.

3.4.2 Effects of Land Use on Biotic Variables

Salmonid densities in rivers on PEI in this study (18 to 378 fish/100 m^2) are comparable to those in other rivers in which this group is found. Brown trout densities in the River Douglas in Ireland range from 10-98 fish/100 m^2 (Lehane et al. 2002). Salmonid density (Atlantic salmon and brown trout) in four rivers in Finland was found to be more variable, with 2-242 fish/100 m^2 captured during the various sampling periods (Mäki-Petäys et al. 2002). Streams in southeastern Idaho and western Washington were found to have lower average salmonid densities (cutthroat trout, brook trout, and brown trout), with values ranging from 3-65 trout/100 m^2 (Isaak and Hubert 2000). In the Maritimes, Cairns (2002a) compared salmonid densities of PEI streams to those that had been found in New Brunswick and Nova Scotia streams. He found similar total densities of salmonids (mean 57.5/100 m^2 on PEI; 54.3/100 m^2 in New Brunswick and Nova Scotia); however, the sites sampled in New Brunswick and Nova Scotia had a higher proportion of salmon, while those in PEI were mainly composed of brook trout. In this study, I found higher overall densities than Cairns (2002a), but similar densities to those found by Guignion et al. (2002; 0-416.5 salmonids/100 m^2).

Increased land in potato rotation in the watershed was positively correlated with brook trout density in this study. Increased potato rotation was also negatively correlated with the amount of forest land and positively correlated with nitrate concentration. These relationships are not surprising, and are consistent with other studies involving agriculture and water quality (e.g., Berka et al. 2001, Randall and Mulla 2001, Borin and Bigon 2002, Honisch et al. 2002). Increased densities of brook trout in regions of intensive agriculture are probably related to the increased productivity of these habitats. Experimental and nonexperimental studies support this hypothesis. For example, the Kuparuk River in Alaska was artificially fertilized with nitrogen and phosphorus to examine the effects on algae, macroinvertebrates, and fish populations (Deegan et al. 1997). With increased nutrient availability, algae production increased. This resulted in increased numbers of macroinvertebrates, which presumably led to the increased growth and survival observed in Arctic grayling (*Thymallus arcticus* Pallas). Similar results have been found in British Columbia (Perrin et al. 1987 *cited from* Cairns 2002b). Green frogs in PEI also appear to grow better in areas of intense agriculture, probably for the same reason (K. Teather, Dept. of Biology, University of Prince Edward Island, Charlottetown, PEI, pers. comm.). Thus, it is quite probable that the increased nutrient loading in streams of PEI in heavily farmed regions provides favourable growing conditions for some species. The long term effects of increased productivity on these populations, and on the entire community, are not known. Coho salmon in Carnation Creek, British Columbia, grew larger and faster in areas where the population density had been decreased (Hartman et al. 1996). These fish smolted earlier than fish from the same area at a higher density. Higher population density in rivers on PEI in agricultural areas could therefore lead to increased competition, and slower growth and maturation.

Variability in density, as determined by repeated sampling at the same sites, increased in areas with more solar radiation reaching the stream bed and with more land in potato rotation. Higher levels of solar radiation indicates less cover for fish from surrounding riparian vegetation. Fish in these areas may therefore have to find other sources of cover (e.g., instream vegetation), which can change throughout the season. As mentioned above, the amount of land in the watershed in potato rotation is negatively correlated the amount of forest, and positively correlated with the nitrate concentration. Potato production is responsible for entry of both nutrients and pesticides (see Chapter 4) into the surface waters. Although it appears that increased nutrients may lead to higher salmonid densities, increased levels of pesticides may have the opposite effect. Changes in the relative amounts of nutrients and pesticides entering the surface water could therefore lead to a greater amount of variability in salmonid densities. Also, increased nutrient loading is not always expected to provide favourable conditions for salmonids. High nitrate levels, for example, are known to negatively affect salmonids. Exposure to 2.3 mg/L of nitrate was found to cause 31% mortality in rainbow trout eggs, and 15% mortality in rainbow trout fry (Kincheloe et al. 1979 *cited from* Rouse et al. 1999). Average nitrate concentration in sixteen of the thirty-two sites (seven of the fourteen rivers) sampled in this study exceeded 2.3 mg/L, indicating potential deleterious effects on salmonid eggs and fry in these areas.

There was a large amount of variability in the percent habitat saturation (PHS) found between sites. The two sites that had a PHS value over 100%

(ML1 and MM2) had large pools in the site. PHS is designed to work in shallow, riffle areas, where fish are expected to occupy two-dimensional habitats (Grant and Kramer 1990). In pools, fish occupy three-dimensional habitats, which inflates the PHS, and makes the numbers misleading.

The percent habitat saturation in this study was higher in areas with lower velocity. Low velocity can occur when the river deepens or widens out, and as mentioned above, PHS is less reliable in areas with deeper water (Grant and Kramer 1990). Although the PHS was not correlated with the mean depth of the water, the three sites with the highest PHS (MM2, ML1, and NL3) all had deep (>1 m) pools (pers. obs.), which therefore could have inflated the PHS.

PHS was also positively correlated with the amount of wetland in the watershed. In this study, sites with a higher percentage of wetland in the watershed had a larger substrate size, higher substrate stability, and lower Pfankuch score than sites with lower amounts of wetland. These variables would be expected to be related to an increase in fish habitat quality (Mebane 2001). Wetlands can act as purifiers, and have been found to decrease the amount of nitrogen in the system through plant uptake and denitrification by anaerobic bacteria (Saunders and Kalff 2001). They are also areas of slow moving water, which retains sediment and keeps it from entering the river. As mentioned earlier, increased levels of silt have a negative impact on salmonid habitat quality (Mebane 2001) through reduction in habitat quality for eggs, young fry, and juveniles (Cunjak 1988, Mills 1989, Heaney et al. 2001). Increased sedimentation is easily observed and measured in streams, and has been correlated with decreased survival to emergence in brook trout in the Wilmot and West Rivers on Prince Edward Island (MacNeill and Curry 2002b). In various rivers on Prince Edward Island, Atlantic salmon and brook trout densities were found to be negatively correlated with the proportion of stream bottom covered by fine particles (Cairns 2002b, MacNeill and Curry 2002a). Percent habitat saturation was also higher in areas with larger substrate, as indicated by the Wentworth rating obtained from the random walk. The Wentworth rating classifies substrate on a scale of one to six, with the score increasing with substrate size. Optimum substrate for salmonids is around four (Bain et al. 1985, DeGraaf and Bain 1986). This variable is negatively correlated with the percent of silt in the site and the Pfankuch score. The Pfankuch score is heavily rated by the substrate; the percent of silt would directly influence the Wentworth rating.

As with variability in density, variability in percent habitat saturation was positively related to solar radiation, probably for the same reasons mentioned above. Additionally, variability in PHS was higher in smaller watersheds than in larger ones. As watershed size was correlated with river length, discharge, mean width, and mean wetted width, variability in PHS also increased with smaller values of these variables. Smaller systems are found to have higher variability and wider extremes in habitat variables (Jackson et al. 2001), which could explain the increased variability in PHS found there.

As there were significant differences in condition factor depending on salmonid species and age class, these categories were examined separately.

Only 0+ and 1+ brook trout age classes were compared between sites as they were most widespread. The only significant relationship was a positive correlation between the condition of 1+ brook trout and the concentration of dissolved oxygen. Trout are known to prefer cool, well-oxygenated areas (see Steedman and Kushneriuk 2000), and low dissolved oxygen levels are known to stress fish (Jackson et al. 2001). It is possible that lower dissolved oxygen might result in decreased food intake or respiratory metabolic rates, both of which have been found to negatively affect condition in stressed rainbow trout populations (Bergstedt and Bergersen 1997).

The use of stable isotope analysis to assess the impacts of anthropogenic activities on fish communities has increased in recent years (e.g., Lake et al. 2001). In this study, the only stable isotope variable correlated with the surronding habitat quality was $\delta^{15}N$. The $\delta^{15}N$ ratios were higher (more enriched) as alkalinity, rock size, and temperature (which was correlated with discharge) increased. Nitrogen isotope ratios have been affected by inputs from the surrounding land in other studies. Both chemical and natural fertilizers can increase the amount of nitrogen in the aquatic system, and the isotope ratio of these anthropogenic sources of nitrogen differs from those from natural sources (Lake et al. 2001). For example, Lake et al. (2001) examined seventeen freshwater sites in Rhode Island, USA, that varied from those that were almost pristine to those that were highly impacted by human activity. Nitrogen isotope ratios were examined in all fish species, unionid mussels (*Elliptio complanata* Lightfoot), water, and sediment to determine if isotope signatures corresponded to different land use patterns. They found that sites with greater land development in the surrounding area had higher $\delta^{15}N$ ratios in the organisms and sediment samples, presumably from the increased sewage in the system.

Higher discharge, which was correlated with higher temperature, is associated with larger rivers. These larger rivers may have more inputs of organic material through increased areas for runoff. Increased alkalinity can be found in polluted rivers with increased organic anions (Cole 1994). As it is expected that increased $\delta^{15}N$ ratios result from the impact of some sort of pollution, the observed relationship between alkalinity and $\delta^{15}N$ could be through a third, unmeasured, factor.

The higher $\delta^{15}N$ ratios observed in areas with larger substrate do not support a land-use related theory. The opposite trend would more likely be observed if this was the case, as decreased substrate size (and therefore increased silt) could indicate land use problems (e.g., runoff from roads and agriculture), and therefore more nitrogen entering the system. Rather, the observed ratios seem to indicate trends related to trophic level. Increased substrate size could result in higher numbers of macroinvertebrates, including large, predatory macroinvertebrates, than would be found in areas with smaller substrate. Young of the year fish have been found to consume predatory macroinvertebrates (see Sutherland 1998). If these macroinvertebrates were already at a higher trophic level, the trophic level of the 0+ brook trout would be increased accordingly, explaining the observed increase in $\delta^{15}N$ ratios.

Stable isotope analysis is a useful technique to study changes in fish populations. As it is not necessary to sacrifice the fish numerous samples are easily obtained, which increases the validity of the sample. Indeed, the model developed for $\delta^{15}N$ was the only model in this study that was successful in predicting the dependent variable in the test rivers, and should be investigated further. Examination of stable isotope ratios within the food web (vegetation, macroinvertebrates, and different age classes of salmonids), as well as from known fertilizers, would create a more complete picture of the changes in $\delta^{15}N$ throughout the food web, and possibly further identify the cause of the differences between sites.

3.4.3 Use of Models in Predicting Biotic Variables

The only model that proved useful in predicting values in test streams was the one developed for $\delta^{15}N$ ratios. These values were determined from 0+ brook trout which, as young fish, are expected to move less than larger age classes, and thus be a better reflection of the habitat they are in. The isotopic signature of the fish is reflective of their life history, and the food they have eaten (Wayland and Hobson 2001), and therefore is a reliable measure of the fish in a certain area. The other biotic variables that were measured in this study may be affected by factors that were outside the scope of this study. The density (and presence) of salmonid species in rivers depends on both the current conditions and the history of the river. If there previously had been impassable structures in place, certain populations may have been reduced or extirpated. This could affect the density and the percent habitat saturation (which includes density in its determination) of the river. Additionally, condition factors of the fish also depend on the density of fish in the river, as increased density causes increased competition and therefore decreased condition of individual fish (Chiasson et al. 1990).

Another possible explanation for the low success of the models in predicting biotic variables could be the low sample size. Only nine rivers were used to develop the model, and five to test the model. The nine rivers used to develop the model had three sampling sites per river, and each site was sampled six times. The five rivers used to test the model only had one sample site per river, and these sites were sampled three times. There was a large amount of variability found between sites on the same river, and between samples at the same sites. This variability was accounted for in the nine test rivers; however, with only one site on each test river, the low statistical power and high natural variability may have masked trends that would have been evident if more sites had been sampled. In some of the models, the trends in the test sites seemed to match the original model (e.g., brook trout density), but there were not enough points to make the relationship significant. If more sites were sampled, the brook trout density predicted from the model may prove significant.

3.4.4 Conclusion

Populations of salmonids across Prince Edward Island show some clear relationships with land use patterns. Brook trout are still widespread and have healthy populations in most areas, but Atlantic salmon are becoming limited in parts of their range, while rainbow trout are expanding their range and increasing in density. The effects of habitat variables on the fish population density, percent habitat saturation, and condition are not easily determined, as there are numerous factors that may not be measurable, but which may strongly impact these biotic variables. Models developed to explain the impact of various stream and watershed variables on population density, percent habitat saturation, and condition were not successful in predicting these biotic variables in five test rivers, but did show that land use variables are important in structuring the fish communities. The lack of predictive ability may have been because certain variables were not included (e.g., pesticide concentration), because some variables may not have reflected true impacts of land use (e.g., amount of land in potato rotation does not indicate land use management practices), or because of small sample size. There are numerous other variables that may not have been measured, or may not even be measurable, but that could be affecting the biotic variables. Alternatively, the small sample size could be masking trends that are there. It would be extremely useful to test a larger number of rivers, focussing on the variables used in these models. Stable isotope ratios ($\delta^{15}N$) appear to be the most easily modelled variable of any collected. These ratios are affected by the surrounding land use and food web structure, and their use in examining differences in salmonid populations of Prince Edward Island should also be further investigated.

CHAPTER 4:

CHANGES IN SALMONID COMMUNITIES AS THE RESULT OF PESTICIDE RUNOFF EVENTS IN THE WILMOT RIVER, PRINCE EDWARD ISLAND

4.1 INTRODUCTION

4.1.1. Effects of Fish Kills on Salmonid Populations

Fish kills are episodes of massive fish mortality over a short period of time USEPA 2002). They are most commonly the result of exposure to natural (e.g., dinoflagellate toxins, USEPA 2002) or anthropogenic (e.g., toxic wastes SPCC 1978 *cited from* Moran 1984) toxins, high temperature (Fry et al.1946, *cited from* Power 1980, Scott and Crossman 1973), or low oxygen (USEPA 2002). In addition to the acute lethal response, fish kills may result in longer-term mortality, changes in population structure, or changes in community structure.

Longer-term (delayed) effects of mortality on fish populations after a fish kill may occur due to a number of causes. For example, stressors responsible for fish mortality may result in mortality of macroinvertebrates (Wallace et al. 1973, Whiles and Wallace 1995), a main food source for many fish (Sweka and Hartman 2001). Exposure to contaminants may also result in disruptions to swimming behaviour, which may result in decreased ability to escape predation or capture prey (Little and Finger 1990). For example, one year old rainbow trout exposed to sublethal concentrations of two herbicides, diquat and simazine, exhibited reduced rheotaxis (movement and orientation into a current) and swimming speeds (Dodson and Mayfield 1979). Juvenile chinook salmon exposed to the pesticide diazinon for two hours had inhibited olfactory-mediated alarm responses at 1.0 µg/L, and reduced homing behaviour ability at 10 µg/L (Scholz et al. 2000). The spillage of two pesticides, nabam (disodium ethylene bisdithiocarbamate) and endrin (1,2,3,4,10,10-hexachloro-6,7-epoxy-1,4,4a,5,6,7,8,8a-octahydro-1,4,-endo,endo-5,8-dimethanonaphthalene) into the Mill River, Prince Edward Island, resulted in the death of at least 5000-8000 brook trout and undetermined numbers of salmon (Saunders 1969). Fish that were still alive were observed swimming erratically, with fish repeatedly surfacing, swimming from side to side across the stream, and propelling themselves out of the water, causing them to become stranded. These fish were also found to have a reduced startle response, as they did not respond to pebbles thrown in amongst them. Therefore, although these fish may not have been killed directly by the pesticides, the exposure changed their behaviour and likely made them more susceptible to predation.

Within a species, different life stages may have differential sensitivity to many contaminants, leading to changes in population structure after the introduction of a contaminant. When DDT was aerially applied over a forest in New Brunswick in the 1960s to control spruce budworm, Atlantic salmon parr (2+ fish) numbers decreased by 40%, but young of the year (0+) salmon numbers decreased by 98% (Elson 1967). This could result in longer-term effects to the population, as most of one age class is missing.

Changes in community structure may result after fish kills if all species present are not affected the same. Different species may require different concentrations of the same compound to induce effects or mortality. For example, coho salmon exposed to the pesticide malathion required 2.4 times the amount of the pesticide to induce the same reduction in brain acetylcholine esterase levels as did brook trout (Post and Leasure 1974). Rainbow trout required 1.4 times the level of brook trout to induce the same reaction. After the accidental spillage of the pesticide disulfoton at Basle on the River Rhine in 1986, it was found that the European eel (*Anguilla anguilla* Linnaeus) population decreased dramatically (Arnold and Braunbeck 1994). Subsequent toxicity studies found that eels had a 96-hr LC_{50} value of 37 μg/L for this compund, many times lower than that found for rainbow trout (1.85-3.0 mg/L; Arnold and Braunbeck 1994).

4.1.2 The Wilmot River

The Wilmot River is one of the province's longest rivers at 78.3 km, and has a watershed area of approximately 166 km^2. In the year 2000, over 77% of the land in the watershed was devoted to agriculture; of this, 36% was in potato production (determined using MapInfo Professional Version 6.5 from 2000 Corporate Resource Inventory GIS layer obtained from the PEI Dept. of Agriculture and Forestry).

Two runoff events from adjacent potato fields, resulting in significant fish mortality, occurred on the Wilmot River in 2002. On 10 July 2002, runoff from three sites on two potato fields entered Wilmot River above WR1 and below WR2 (see Fig. 4-1) during a heavy rainstorm. On 19 July 2002, another rainstorm occurred, and runoff again entered the Wilmot River, this time between WR5 and WR7. Analysis of water samples by Environment Canada after both runoff events revealed that azinphos-methyl (O,O-dimethyl S-[(4-oxo-1,2,3-

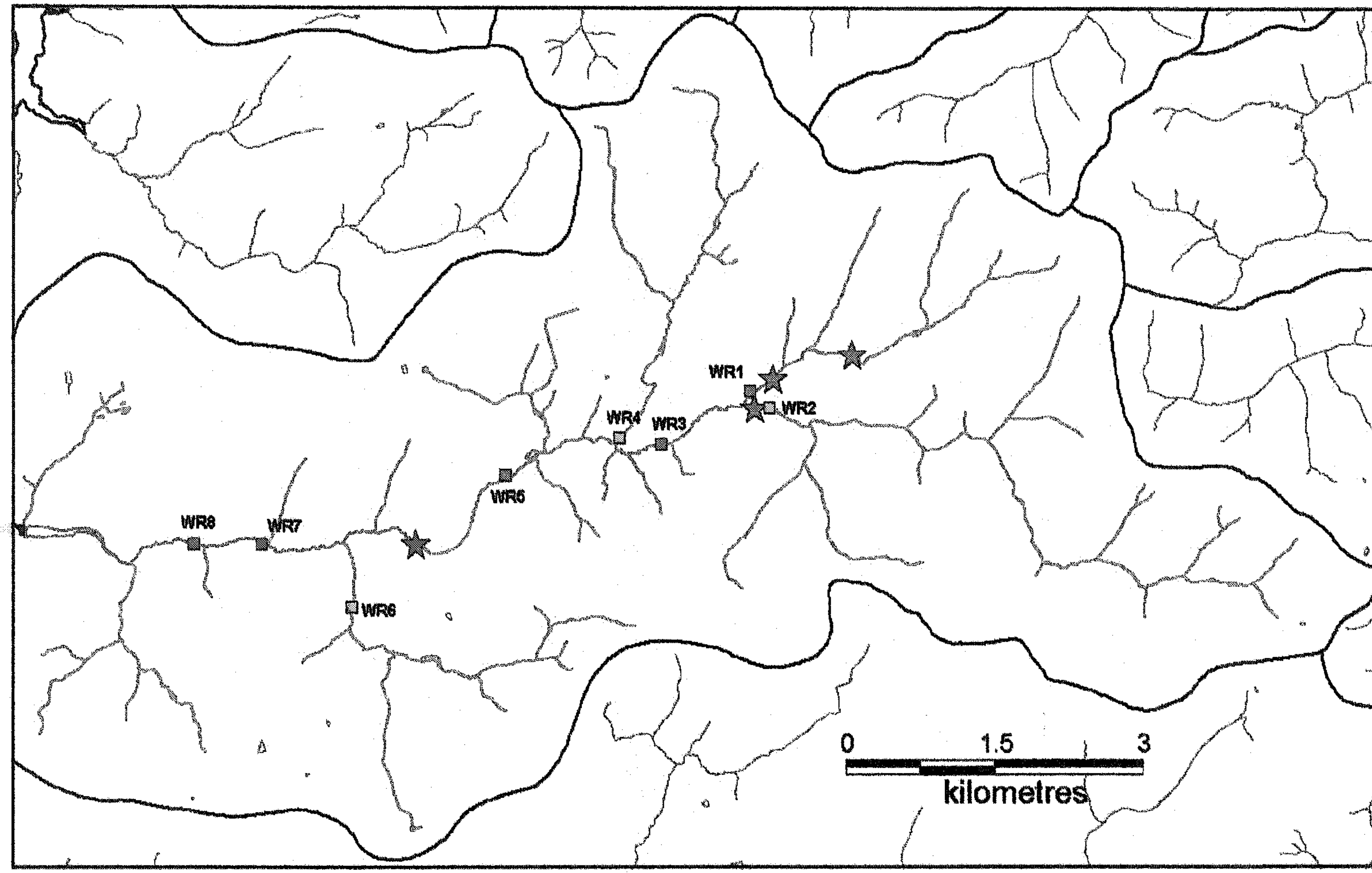


Figure 4-1. Location of sampling sites (WR1-WR8, square boxes) on the Wilmot River. Sections of the river in red were affected by the first pesticide runoff event (11 July), sections of the river in pink were affected by the second pesticide runoff event (19 July), and sections of the river in blue were thought to be unaffected by pesticide runoff. Entry points of the pesticide runoff are indicated by red (11 July) and pink (19 July) stars.

benzotriazin-3(4H)-yl)methyl] phosphorodithioate) was present (B. Birch, Environment Canada, Dartmouth, NS, pers. comm.). This chemical is highly toxic to fish (USEPA 1998), and was likely responsible for the observed fish mortality (B. Birch, pers. comm.).

After the second fish kill event, five more sampling sites were added on the Wilmot River to examine fish movement and recovery in both affected and unaffected areas of the river (see Fig. 4-1, p.90). Sites WR1, WR3, and WR8 are all downstream of where runoff entered during the first fish kill. Site WR8 is also downstream of where runoff entered during the second fish kill. Sites WR4 and WR6 are on tributaries that were not thought to be affected by the runoff during either of the fish kills.

These runoff events on a river where salmonid populations prior to the event were known provided a unique opportunity to study the effects of pesticide runoff on salmonid population and community structure. The objectives of this study were to:

- describe temporal and spatial patterns of fish mortality after two pesticide runoff events on the Wilmot River.
- examine differences in community and population structure of salmonids before and after pesticide runoff events, and compare areas that were affected and unaffected.
- examine extended (four months) effects of the pesticide runoff events on the salmonid community and population structure.

4.2 MATERIALS AND METHODS

4.2.1 Salmonid Sampling

The salmonid populations of the three original sites on Wilmot River (WR2, WR5, WR7, Fig. 4-1, p.90) were sampled by electrofishing in June, July, and August 2001 as described in the general methods (section 2.1). The three original sites (WR2, WR5, and WR7) were sampled immediately after the first pesticide runoff event (10 July 2002). Site WR2 was sampled 12 July 2002; WR5 and WR7 were sampled 13 July 2002. After the second pesticide runoff event (19 July 2002) sites WR5 and WR7 were sampled by electrofishing (20 July 2002). One additional site was sampled between the runoff events (WR3, sampled 14 July), and the other additional sites were sampled after the second runoff event (WR1, WR4, WR6, and WR8, sampled 22-25 July 2002).

The salmonid populations of all eight sites were sampled by electrofishing 11-12 August, 1-2 September, and 31 October (WR2, WR5, and WR7), 1 November (WR1, WR3, and WR4), and 29 November (WR6 and WR8) 2002 (see Fig. 4-2).

4.2.2 Water Chemistry

Water temperature, dissolved oxygen, conductivity, nitrate, phosphorus, and alkalinity were collected as described in Chapter 2. In addition, the average, maximum, and minimum temperatures for the five days preceding sampling were determined from the data loggers (which recorded the temperature every hour

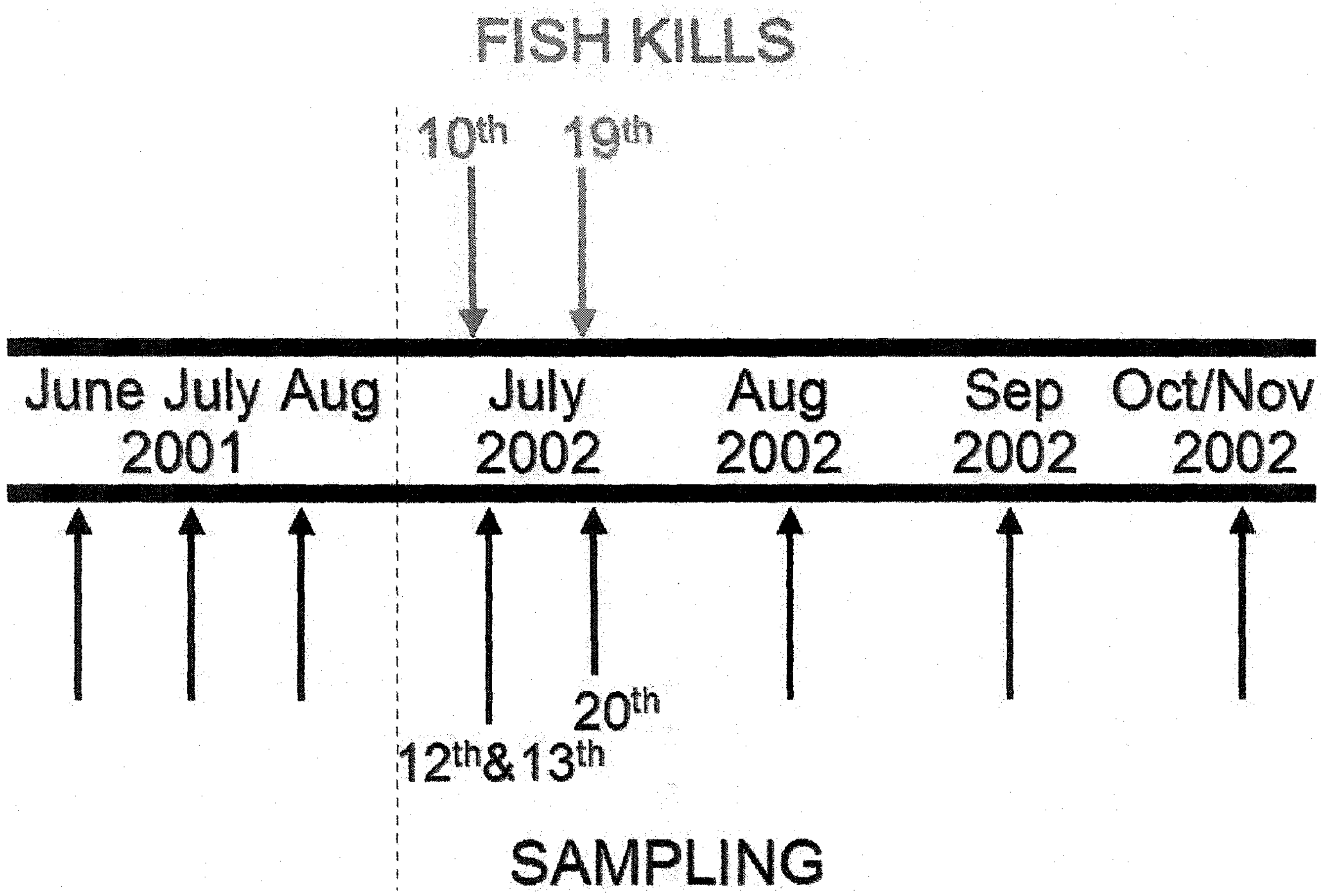


Figure 4-2. Timeline of fish kill events (top arrows) and fish sampling (bottom arrows) on the Wilmot River in 2001 and 2002.

from May to November) installed at WR2, WR5, and WR7 to ensure the temperature did not exceed levels acceptable for salmonids.

4.3 RESULTS

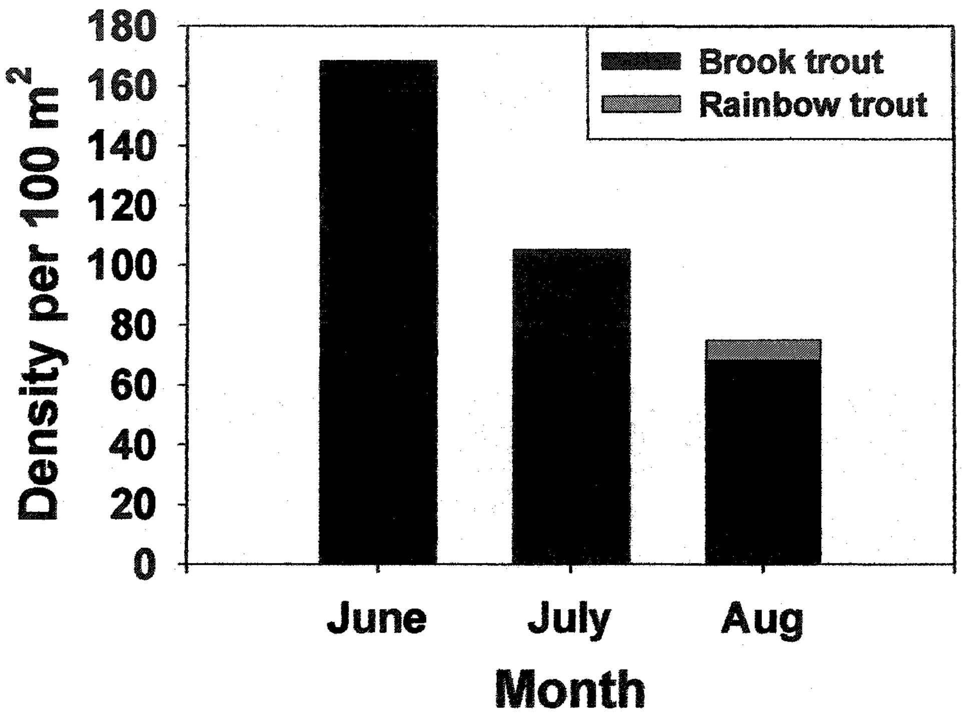
4.3.1 Comparison of Salmonid Data Between 2001 and 2002

The three sites on the Wilmot River (WR2, WR5, and WR7) that were sampled in both 2001 and 2002 allowed for the comparison of salmonid data at these sites between the two years, before and after the fish kills. WR2 was located on a branch that was upstream of all pesticide runoff sites (Fig. 4-1, p. 90), approximately 57 m upstream of one of the three runoff sites that contributed to the first pesticide runoff event (10 July 2002). WR5 was downstream of all three of the pesticide runoff sites that contributed to the first runoff event (Fig. 4-1, p. 90). It was 3.10 km downstream of where the two branches that contained pesticide runoff joined, and was therefore 3.38, 3.55, and 4.55 km downstream of the three entry points. This site was upstream of the pesticide runoff site that caused the second runoff event (20 July 2002) by 1.51 km. WR7 was downstream from WR5 by 3.35 km, and therefore was > 6.7 km downstream of all three sites that contributed to the first runoff event (Fig. 4-1, p. 90). This site was located 1.84 km downstream from the entry point that caused the second pesticide runoff event (20 July 2002).

4.3.1.1 Salmonid Communities and Population Structure

The salmonid community at WR2 (the upstream, unaffected site) was composed primarily of brook trout in both 2001 (Fig. 4-3A) and 2002 (Fig. 4-3B). The population density decreased throughout the summer in both years, but did

A



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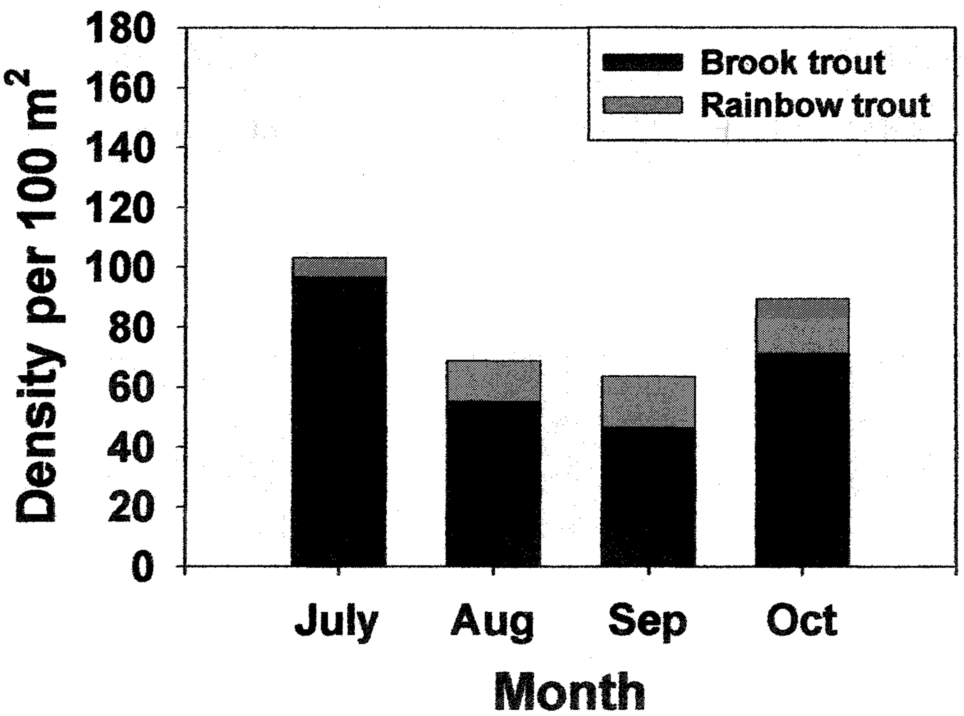


Figure 4-3. Population density (number/100 m^2) of brook trout and rainbow trout during each sample period at WR2 in (A) 2001 and (B) 2002.

increase in October 2002. Brook trout density was composed primarily of 0+ fish in both 2001 and 2002 (Fig. 4-4 A and B), but there were more 1+ fish captured 2002 than 2001. Four of the five rainbow trout captured in 2001 were 0+; all of the rainbow trout captured in 2002 were 0+.

The salmonid community at WR5 (3.38-4.55 km downstream of first runoff event entry points) was composed primarily of brook trout in 2001 (Fig. 4-5A). Population density decreased between June and July, but remained relatively stable between July and August. In July 2002, three days after the pesticide runoff event, the population had decreased by 40% from the previous year (Fig. 4-5B). Brook trout density had decreased by over 80%, while rainbow trout, which were now the dominant species, had increased in density. Density of both brook and rainbow trout decreased at the same proportion throughout the summer, but both increased between September and October.

Brook trout populations at WR5 were composed primarily of 0+ fish. During the 2001 sampling, 97-100% of the brook trout captured were 0+ (Fig. 4-6A). In 2002, after the pesticide runoff event, 0+ fish still made up the greatest percentage of the brook trout population, but varied from 47-83% of brook trout captured (Fig. 4-6B). Rainbow trout populations in 2001 had 0, 88, and 95% 0+ fish in June, July, and August, respectively (Fig. 4-7A). In 2002, 0+ rainbow trout made up 79-99% of the population captured (Fig. 4-7B).

A

B

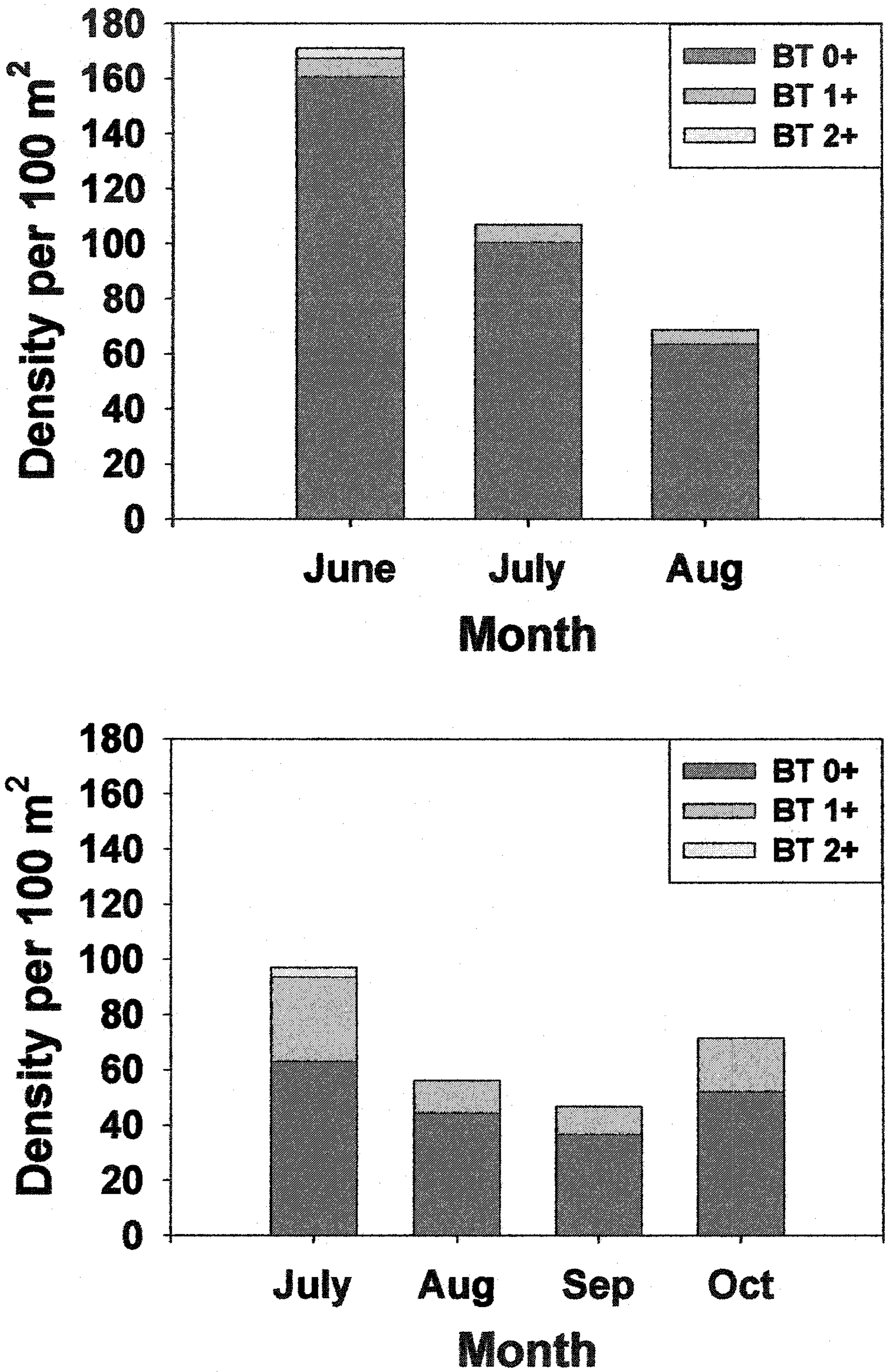


Figure 4-4. Population density (number/100 m^2) of brook trout age classes during each sample period at WR2 in (A) 2001 and (B) 2002.

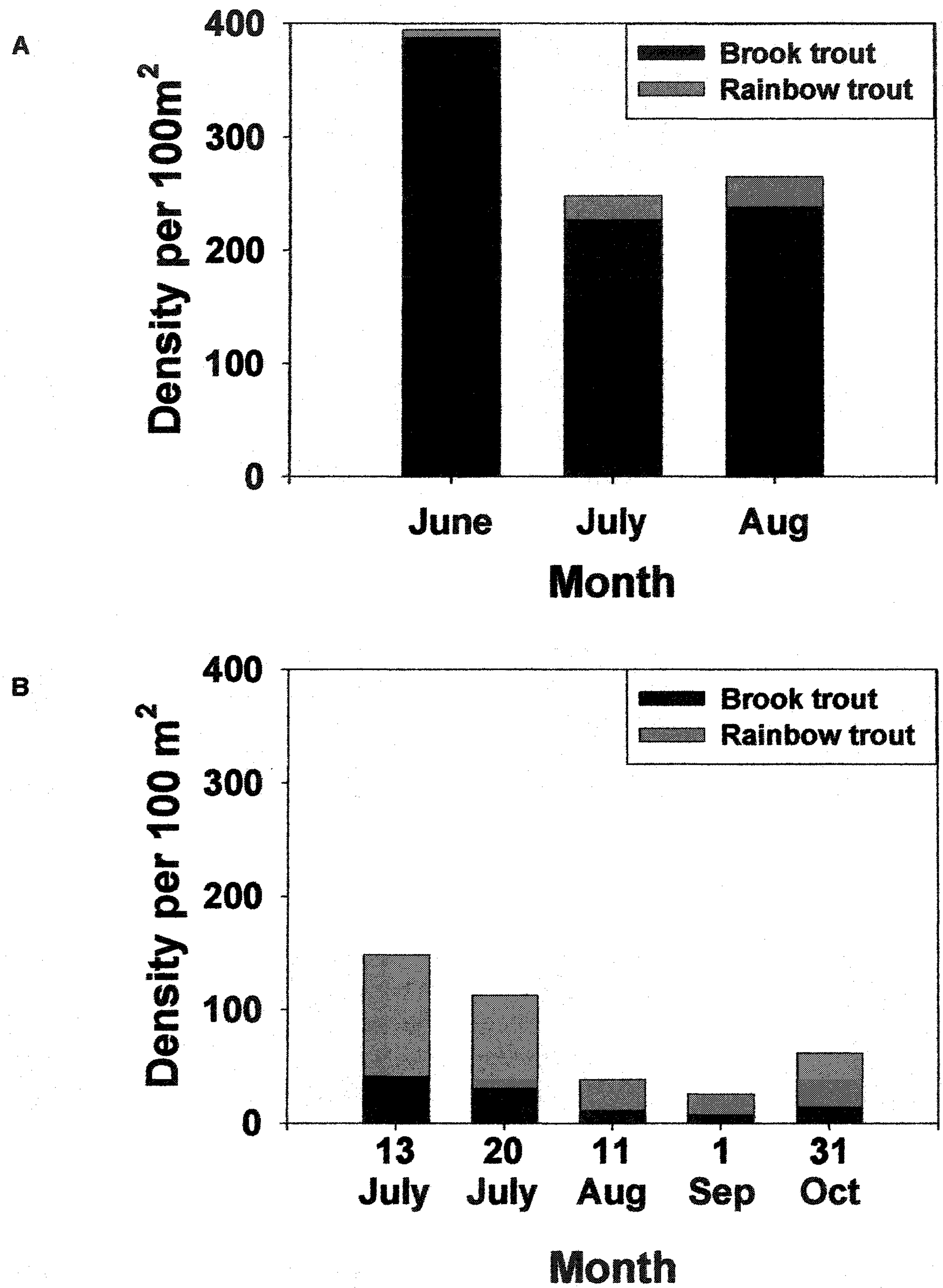


Figure 4-5. Population density (number/100 m^2) of brook trout and rainbow trout during each sample period at WR5 in (A) 2001 and (B) 2002.

A

B

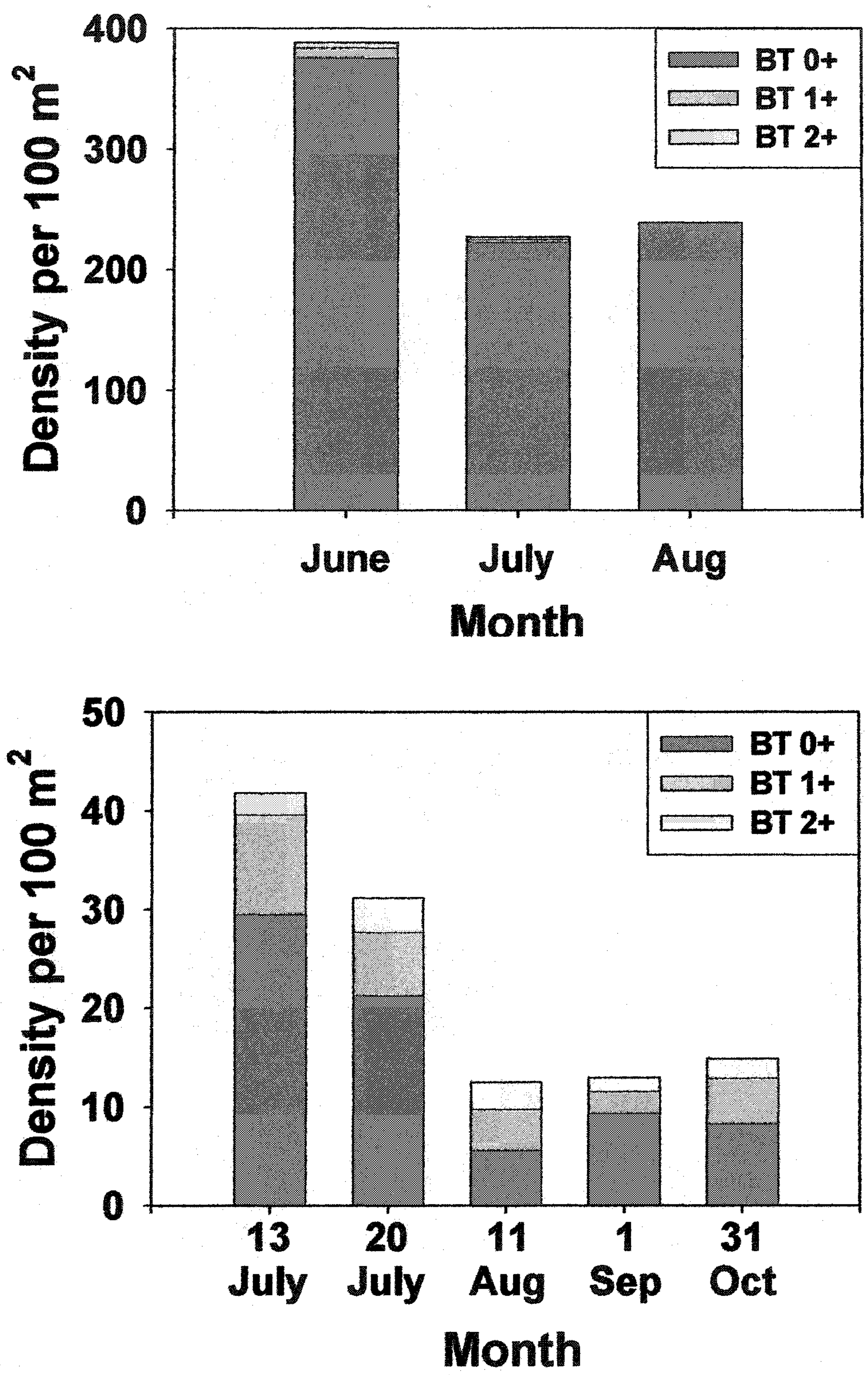
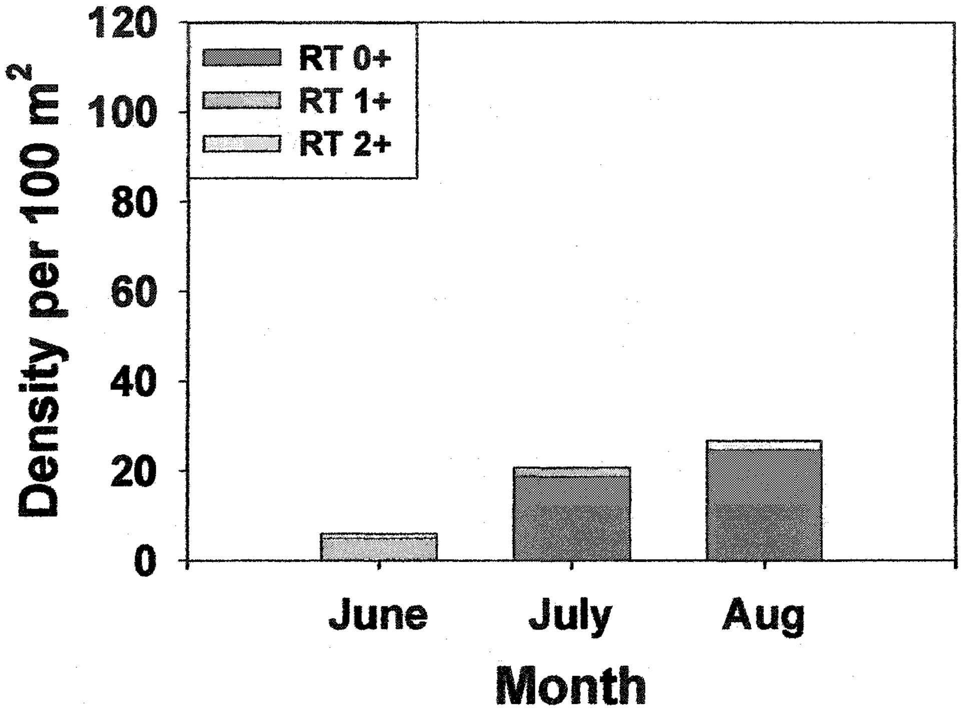


Figure 4-6. Population density (number/100 m^2) of brook trout age classes during each sample period at WR5 in (A) 2001 and (B) 2002.

A



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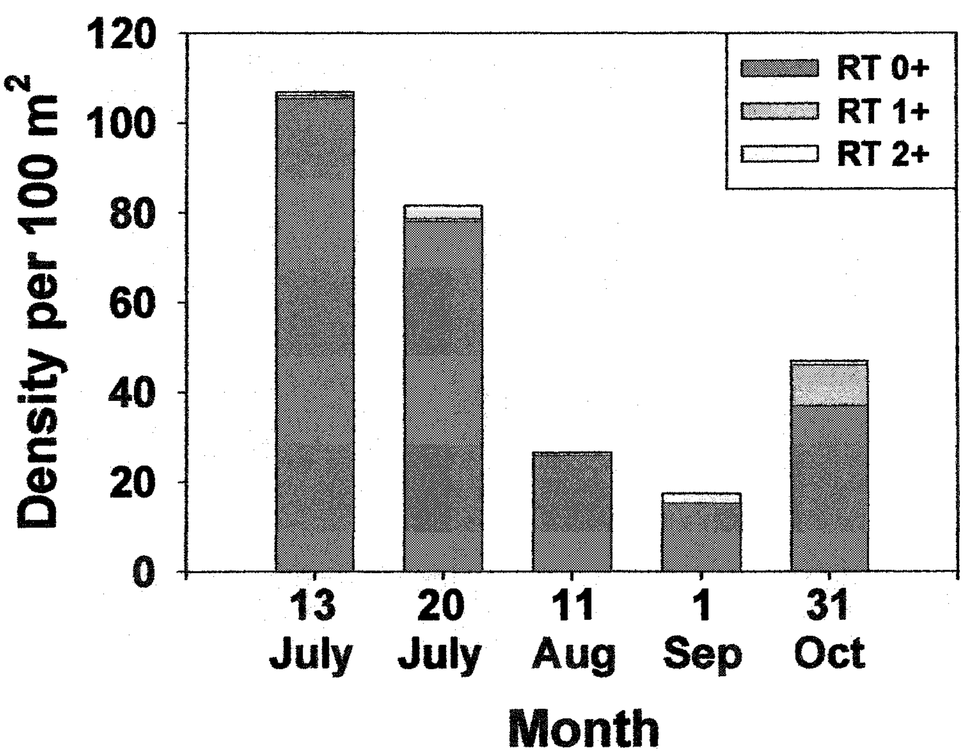
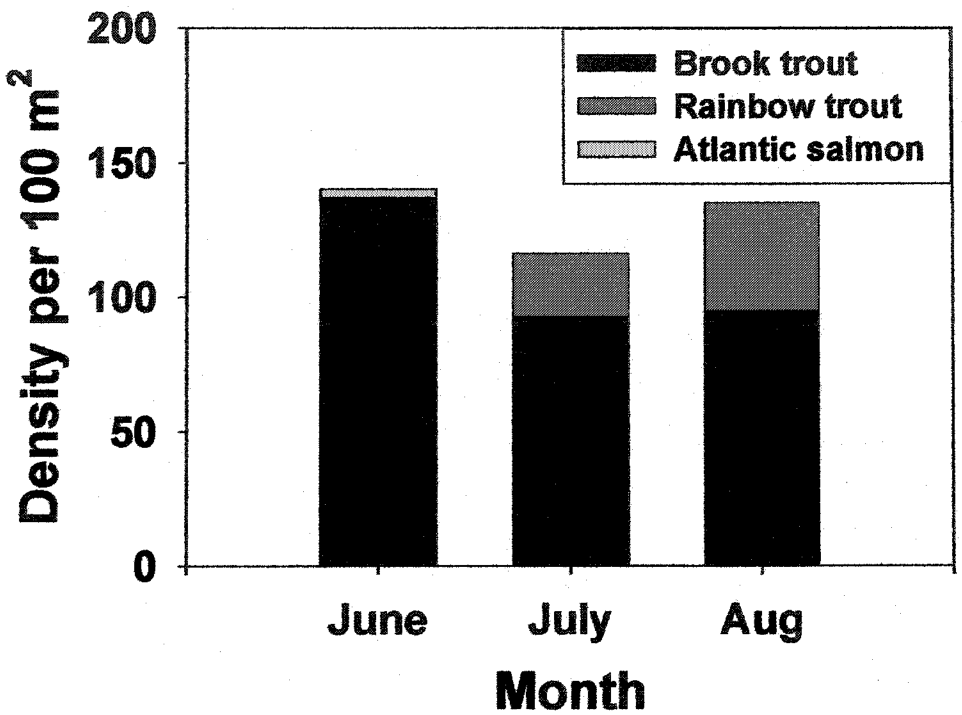


Figure 4-7. Population density (number/100 m^2) of rainbow trout age classes during each sample period at WR5 in (A) 2001 and (B) 2002.

Brook trout were the dominant species at WR7 (>6.7 km downstream of first runoff events entry points, 10 July; 1.84 km downstream of second runoff event entry point, 19 July) in 2001 (Fig. 4-8A) and 13 July 2002 (Fig. 4-8B). After the pesticide runoff event on 19 July 2002, salmonid density had decreased by 90% from seven days previous, with a 98% decrease in brook trout density (from 145 fish/100 m^2 to 2 fish/100 m^2) and a 66% decrease in rainbow trout density (from 44 fish/100 m^2 to 15 fish/100 m^2). This resulted in rainbow trout becoming the dominant species at this site (Fig. 4-8B). Salmonid density remained low for August, and while it doubled between August and September, it was still 82% less than in had been 13 July. This increase was mainly due to an increase in rainbow trout density.

The age class structure at WR7 before the pesticide runoff event was similar to other sites on the Wilmot River in that the population was dominated by 0+ fish. In 2001, 0+ brook trout comprised 95-96% of the brook trout density (Fig. 4-9A). Similarly, on 13 July 2002, before the closest pesticide runoff event, 0+ brook trout accounted for 96% of the brook trout population density of 145 fish per 100 m^2 (Fig. 4-9B). On 20 July, one day after the closest pesticide runoff event, there were no 0+ brook trout captured. Of the two brook trout that were captured, one was a 1+ fish and one was a 2+. There was one brook trout captured in August (0+), three in September (two 0+ and one 1+) and nine in October (six 0+, two 1+, and one 2+).

A



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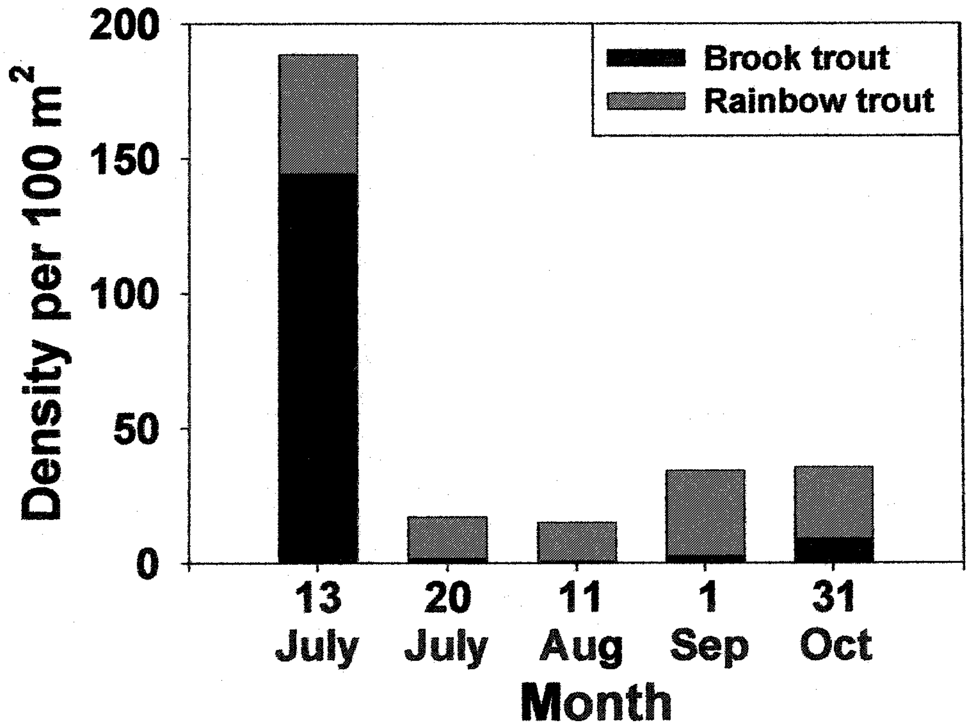


Figure 4-8. Population density (number/100 m^2) of brook trout and rainbow trout during each sample period at WR7 in (A) 2001 and (B) 2002.

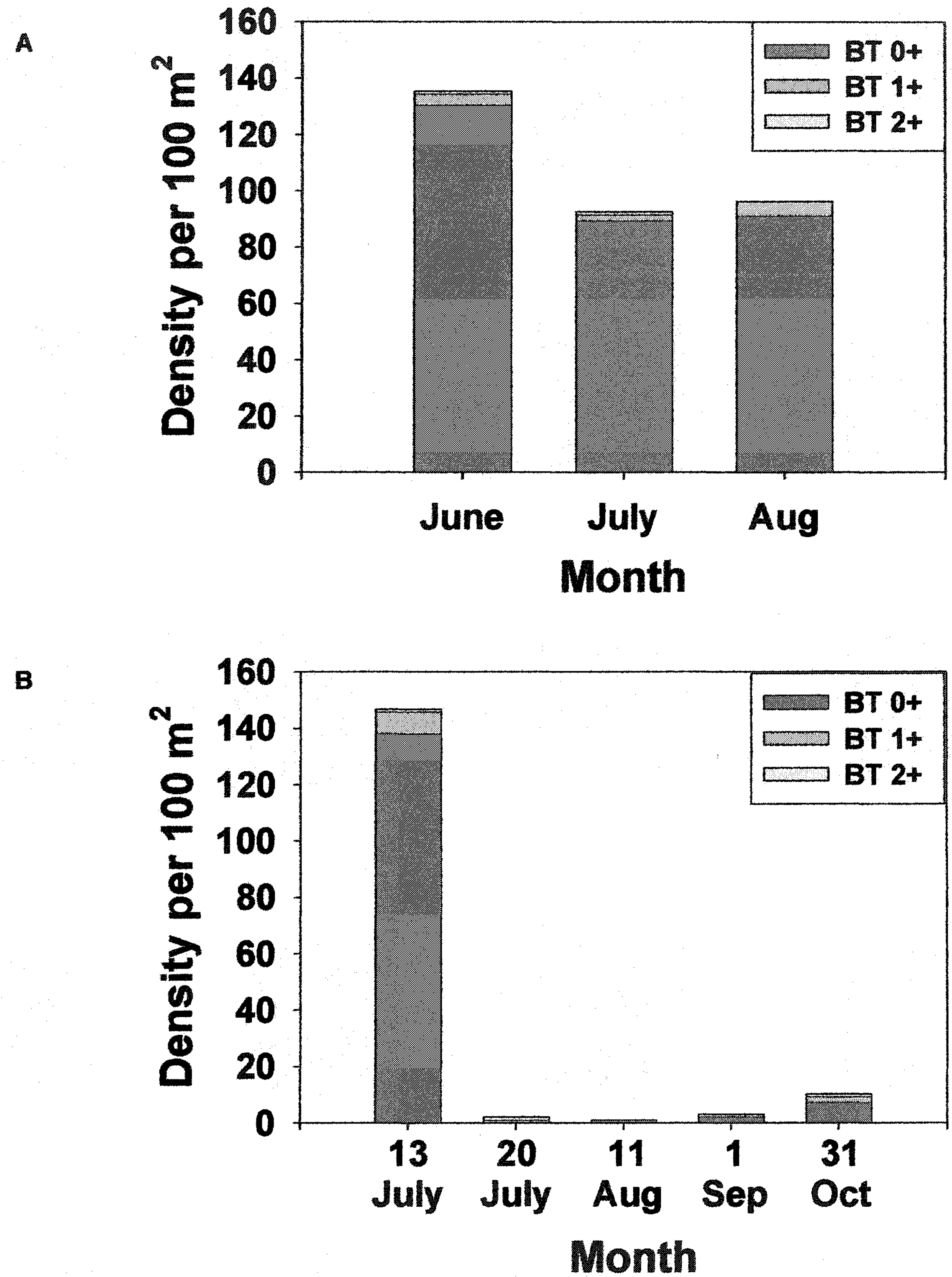


Figure 4-9. Population density (number/100 m^2) of brook trout age classes each sample period at WR7 in (A) 2001 and (B) 2002.

In 2001, 93-96% of the rainbow trout captured were 0+ fish (Fig. 4-10A). On 13 July 2002, before the closest pesticide runoff event, 0+ fish accounted for 89% of the total rainbow trout density. On 20 July 2002, one day after the pesticide runoff event, 0+ rainbow trout accounted for only 54% of the remaining population (Fig. 4-10B). By August, this percentage had increased to 87%, and stayed above 90% for the rest of the year. The total density of 0+ rainbow trout in September and October was up to 80% and 76% of that found on 13 July 2002, respectively. This was an increase from the samples on 20 July and August, in which the 0+ rainbow trout densities were at 21% and 31% of those found 13 July. To examine if this increase in 0+ rainbow trout was from movement into the area or from new fish hatching, the lengths of the fish at the different sample times were compared (Fig. 4-11). The lengths were signifcantly different ($p<0.001$, Kruskal-Wallis). There was no overlap in 0+ rainbow trout length between 13 July and 11 August ($p<0.05$, Dunn's Multiple Comparison Test), suggesting that the increase was from movement of same-aged fish into the area, rather than new, young fish emerging from the gravel bed.

The two pesticide runoff events had major impacts on the salmonid populations. Population density decreased dramatically, with brook trout populations suffering larger declines than rainbow trout populations. Within species, 0+ age classes were most affected.

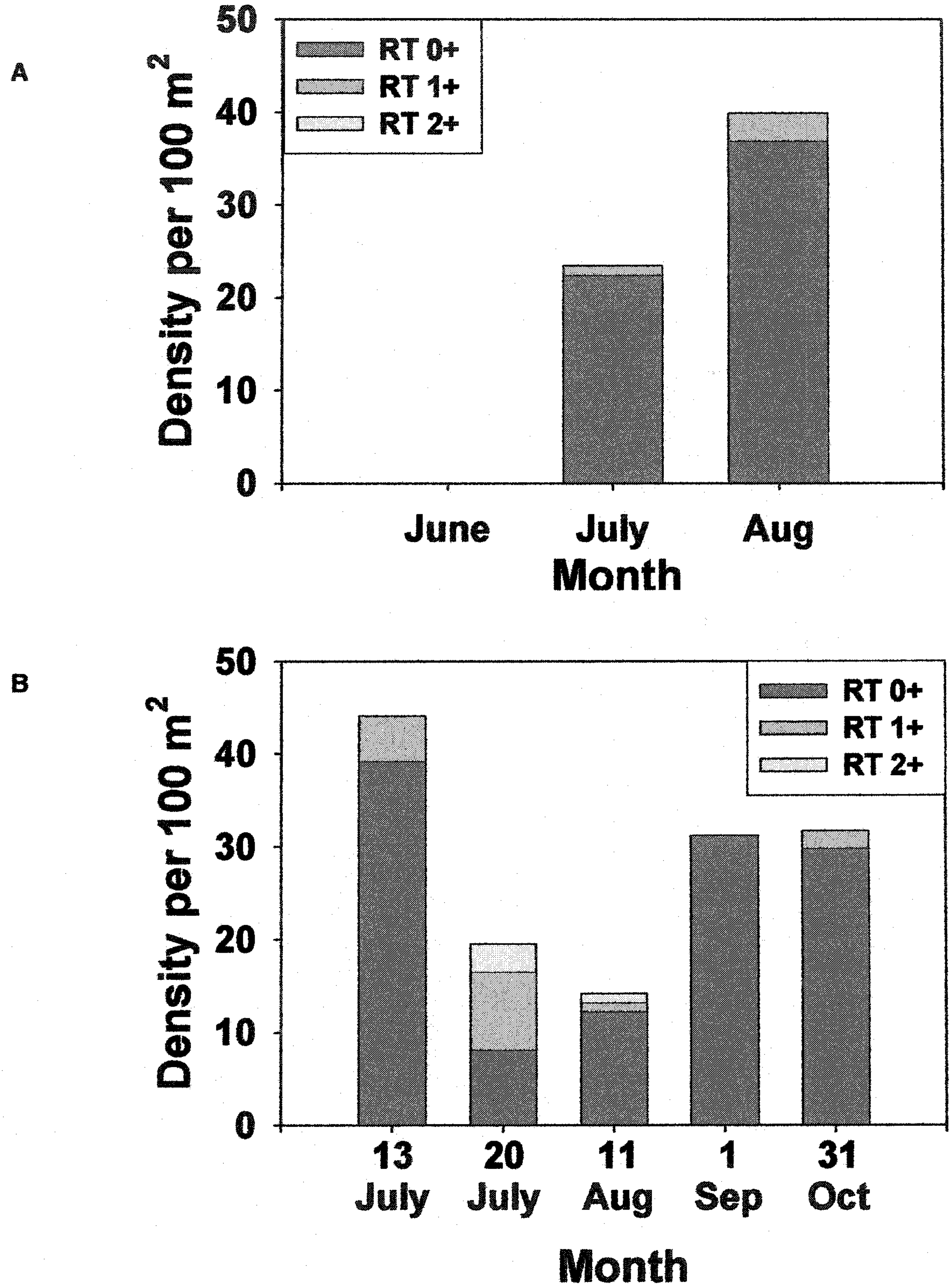


Figure 4-10. Population density (number/100 m^2) of rainbow trout age classes during each sample period at WR7 in (A) 2001 and (B) 2002.

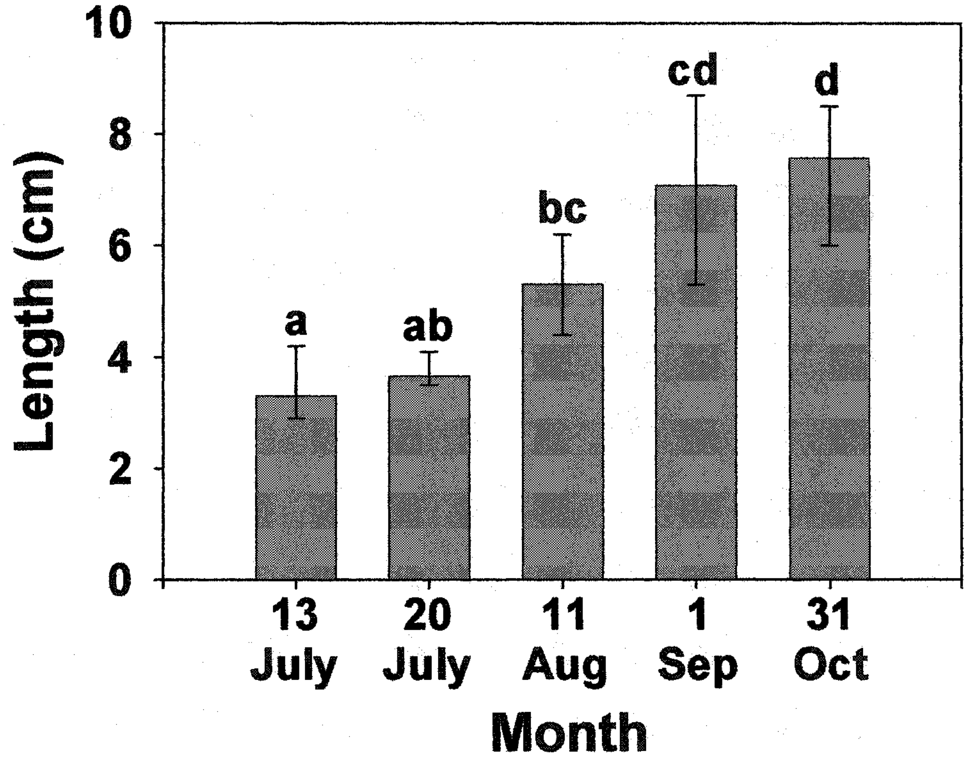


Figure 4-11. Lengths (cm) of 0+ rainbow trout during each sample period in 2002 at WR7. Mean lengths are indicated by height of bars; error bars show minimum and maximum lengths of fish captured. Bars that have different letters are significantly different from one another ($p<0.05$, ANOVA; $p<0.05$, Dunn's Multiple Comparison Tests).

4.3.2 Examination of Additional Areas in 2002

WR1 was located on a branch that had two runoff entry points during the first pesticide runoff event (10 July 2002). These entry points were 0.34 and 1.33 km upstream of the site. The site was located 0.11 km upstream of where it joined with the main branch (Fig. 4-1, p. 90). Only one salmonid, a 2+ brook trout, was captured during 22 July 2002, twelve days after the pesticide runoff event. Salmonid density increased slightly throughout August and September, with salmonid densities of 6.7 and 4.4 fish/100 m^2, respectively (Fig. 4-12). One rainbow trout was captured during August; all other salmonids captured during the summer were brook trout. There was a substantial increase in salmonid density by the 1 November sample, with densities of 23.9 brook trout and 4.8 rainbow trout per 100 m^2 found.

The age class structure at WR1 was highly variable, as densities of less than ten salmonids per 100 m^2 were found during the first three sampling periods (Fig. 4-13A and B). The majority of salmonids captured 1 November were 1+ brook trout; all of the rainbow trout captured at this time were 0+ fish.

WR3 was located 1.12 km downstream of where the two branches that contained pesticide runoff during the first runoff event (10 July) combined. It was therefore 1.40, 1.57, and 2.56 km downstream of the three pesticide runoff entry points (Fig. 4-1, p. 90). Only one salmonid, a 1+ brook trout, was captured during the 14 July 2002 sample. Density increased slowly during subsequent months, with a high of 16.4 salmonids/100 m^2 captured 1 November 2002

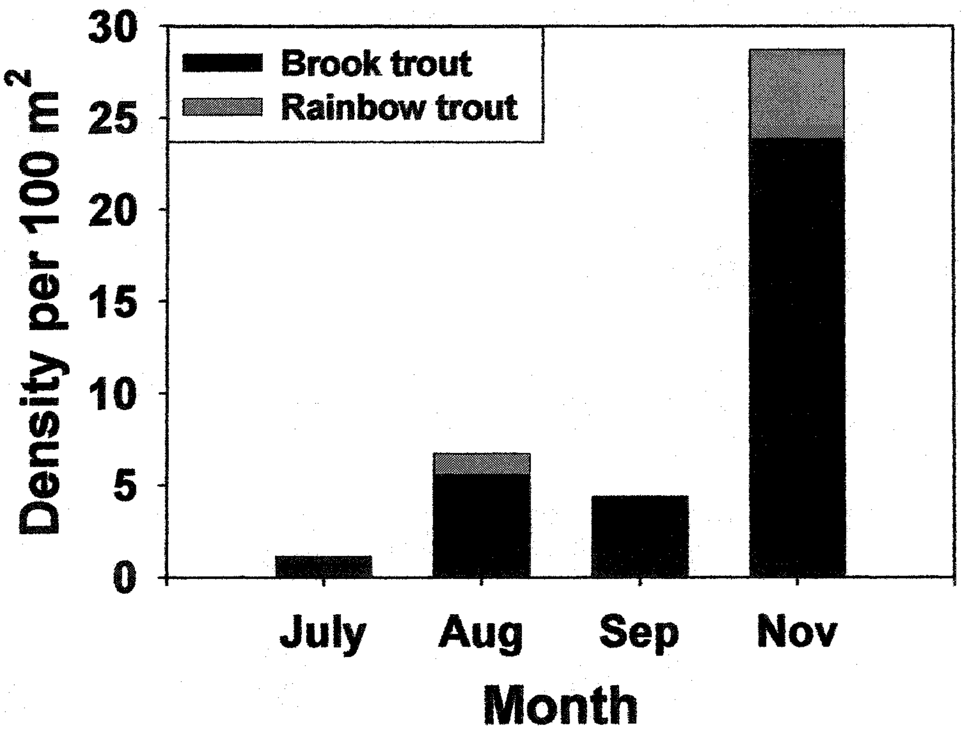
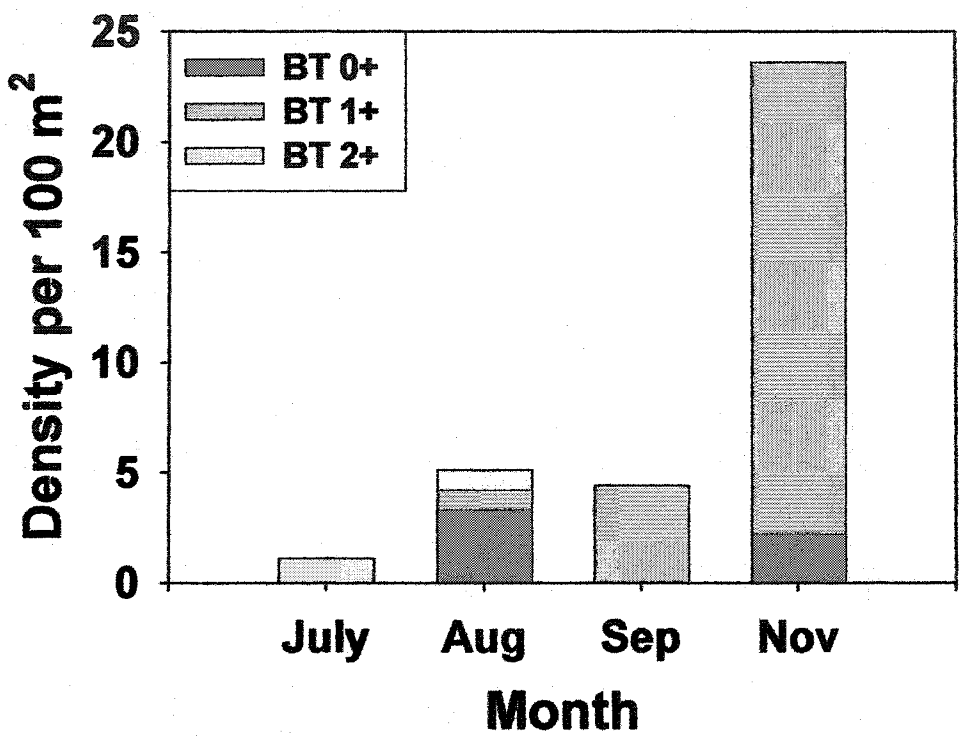


Figure 4-12. Population density (number/100 m^2) of brook trout and rainbow trout during each sample period at WR1 in 2002.

A



B

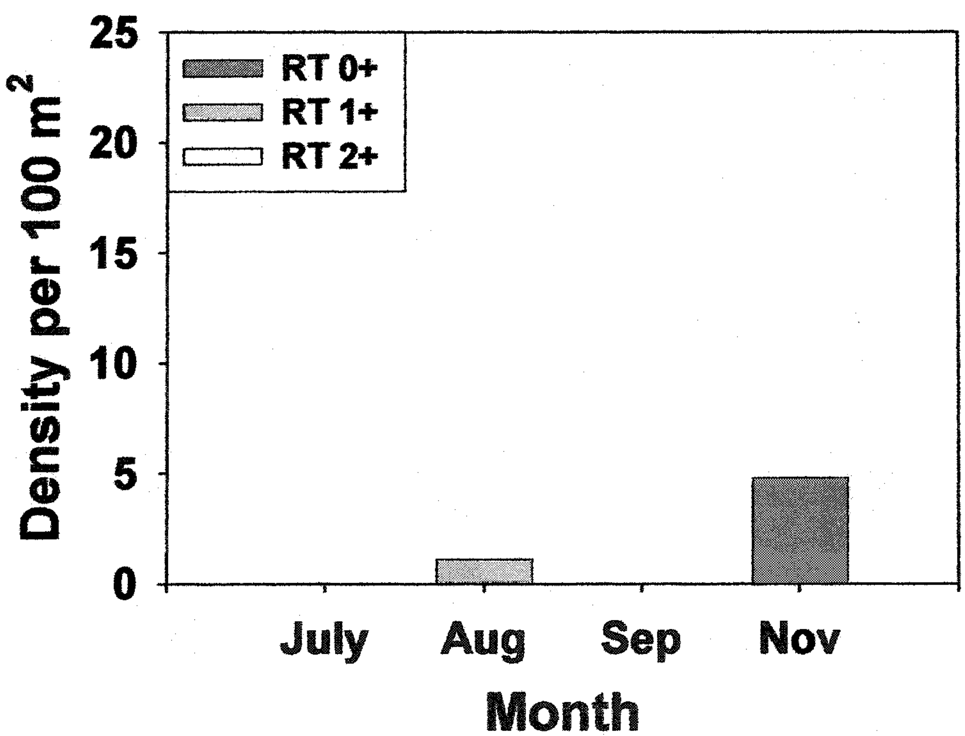


Figure 4-13. Population density (number/100 m^2) of (A) brook trout and (B) rainbow trout age classes during each sample period at WR1 in 2002.

(Fig. 4-14). Only brook trout were captured during July and August sampling. During the September sampling, equal densities of brook and rainbow trout were found, but at the 1 November sampling, 80% of the salmonids captured were rainbow trout. The majority of brook trout captured for all except the 14 July sample were 0+ fish (Fig. 4-15A). Only 0+ rainbow trout were captured (Fig. 4-15B).

WR4 was located on a branch separate from the main stem, and was not affected by runoff from either reported pesticide runoff event. It was located 62 m upstream from the main stem, from a point that was 0.66 km downstream of WR3 (Fig. 4-1, p. 90). Salmonid density was 41.4 and 47.1 fish/100 m^2 for 25 July and 11 August, respectively. Only brook trout were captured during these two samples (Fig. 4-16). On 1 September, brook trout density was only 1.7 fish/100 m^2, and rainbow trout were captured, at a density of 6.9 fish/100 m^2. Total salmonid density remained constant between 1 September and 1 November, with a slight increase in brook trout and decrease in rainbow trout..

The age class structure at WR4 was different than the other sites on Wilmot River in that the majority of brook trout captured in July, August, and November were 1+ fish (Fig. 4-17). All of the brook trout captured in September were 0+ fish, but the density was only 1.7 fish/100 m^2. All of the rainbow trout captured during sampling were 0+ fish.

WR6 was also located on a side branch of the Wilmot River that was unaffected by runoff, but the branch connected farther downstream

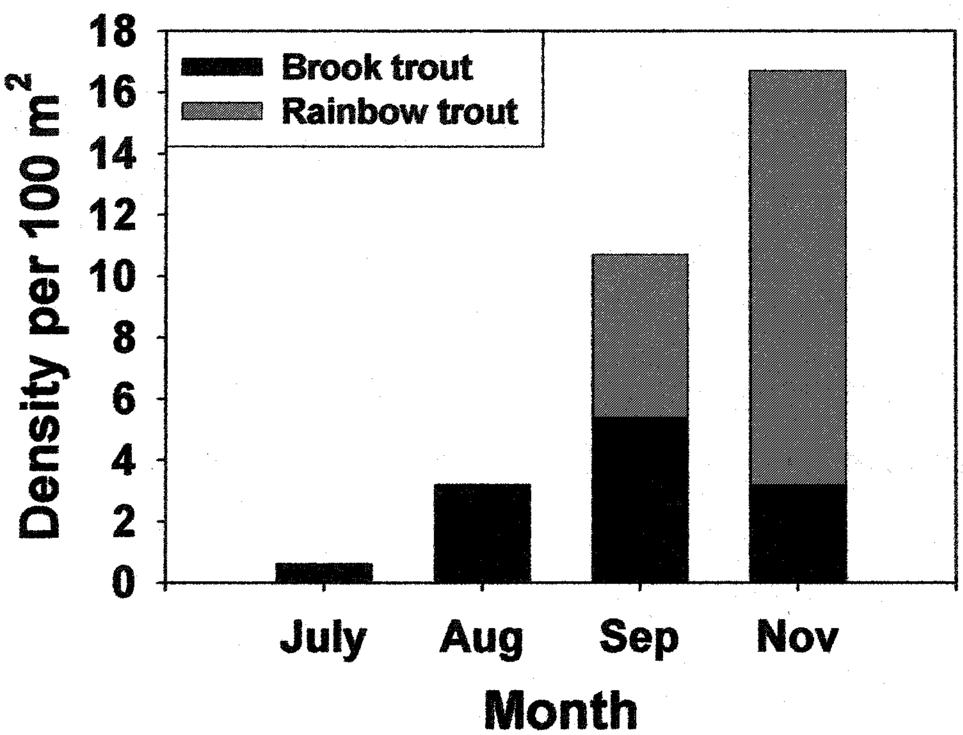
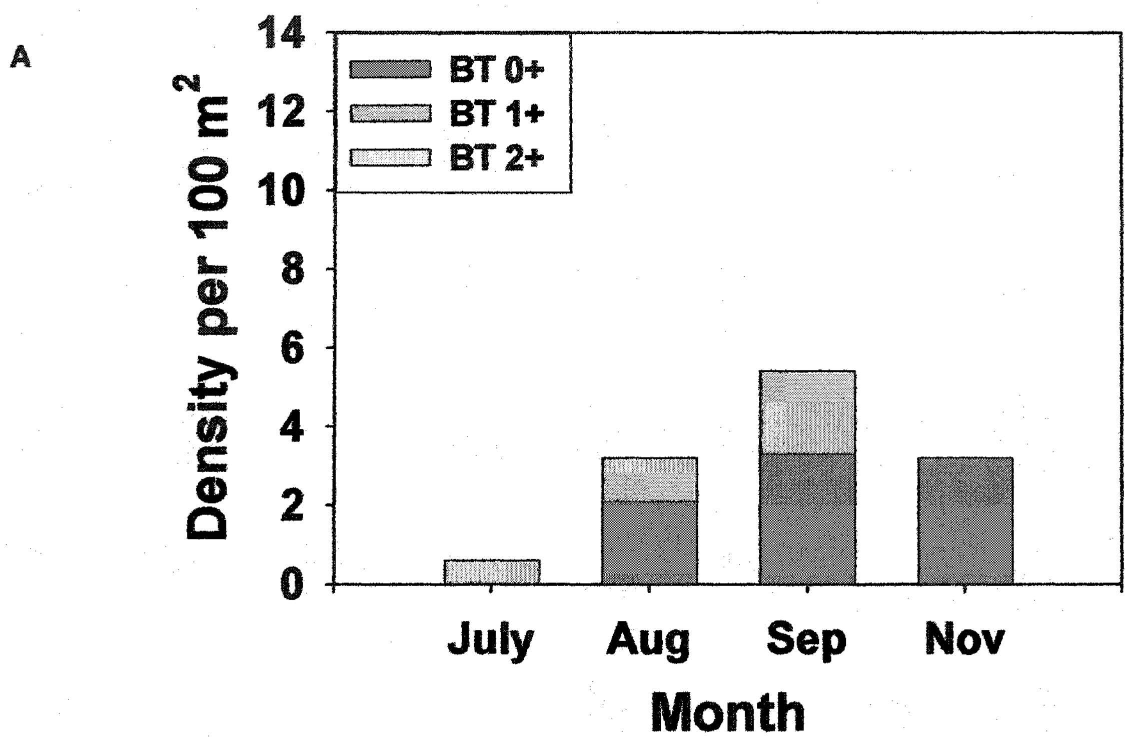


Figure 4-14. Population density (number/100 m^2) of brook trout and rainbow trout during each sample period at WR3 in 2002.



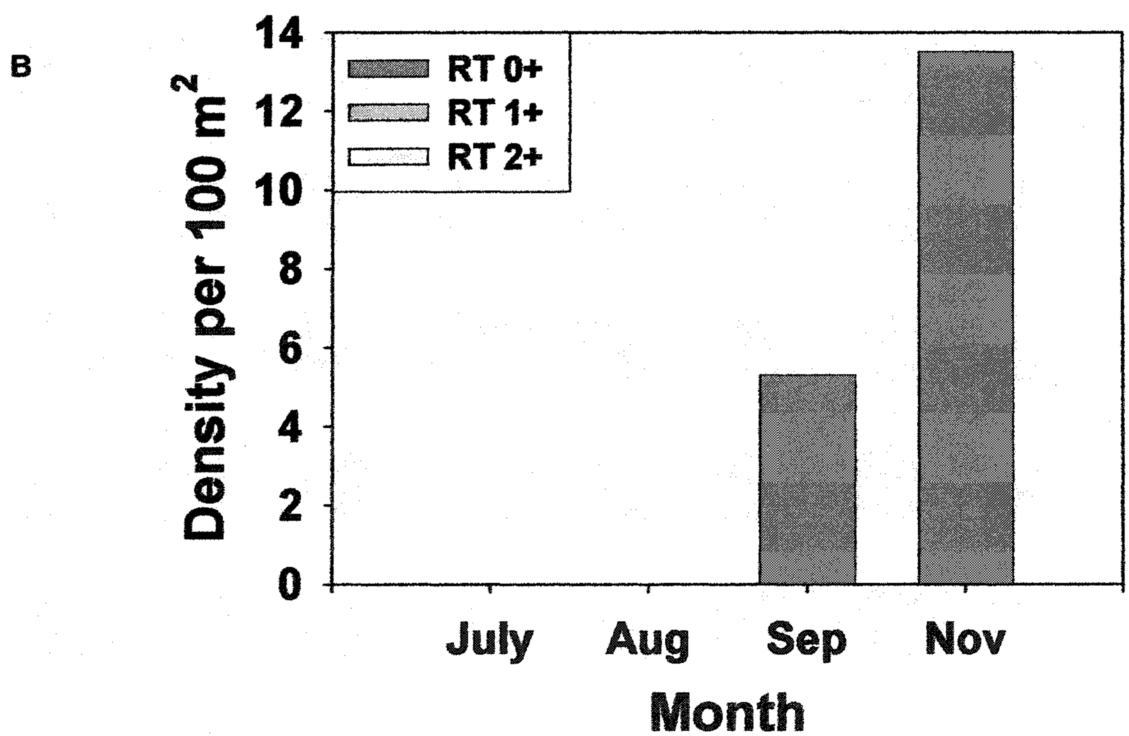


Figure 4-15. Population density (number/100 m^2) of (A) brook trout and (B) rainbow trout age classes during each sample period at WR3 in 2002.

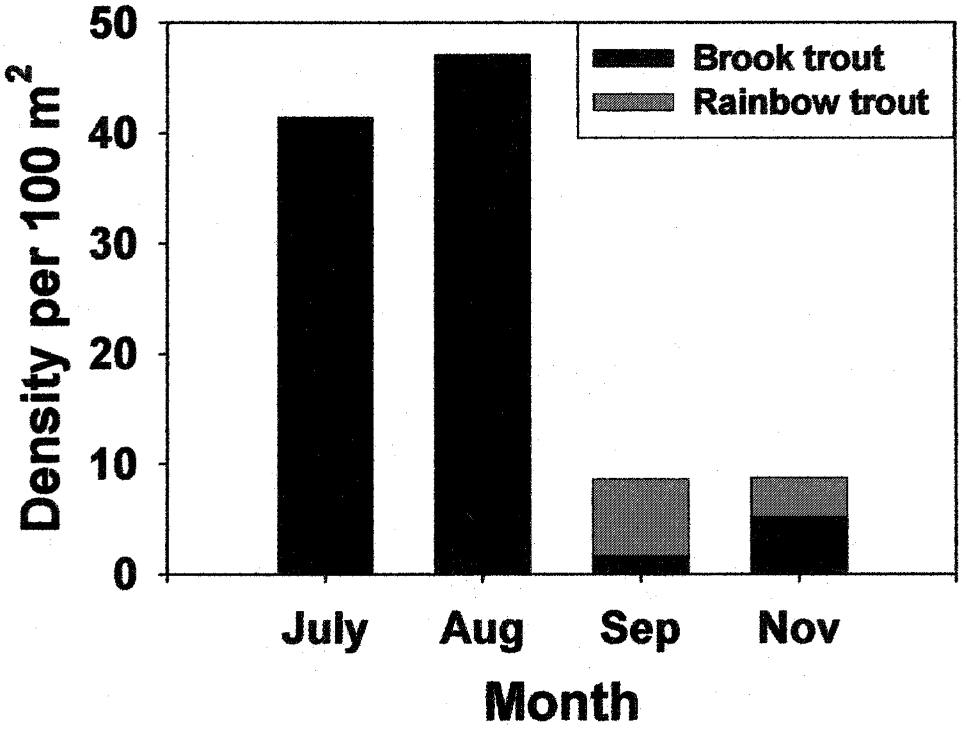


Figure 4-16. Population density (number/100 m^2) of brook trout and rainbow trout during each sample period at WR4 in 2002.

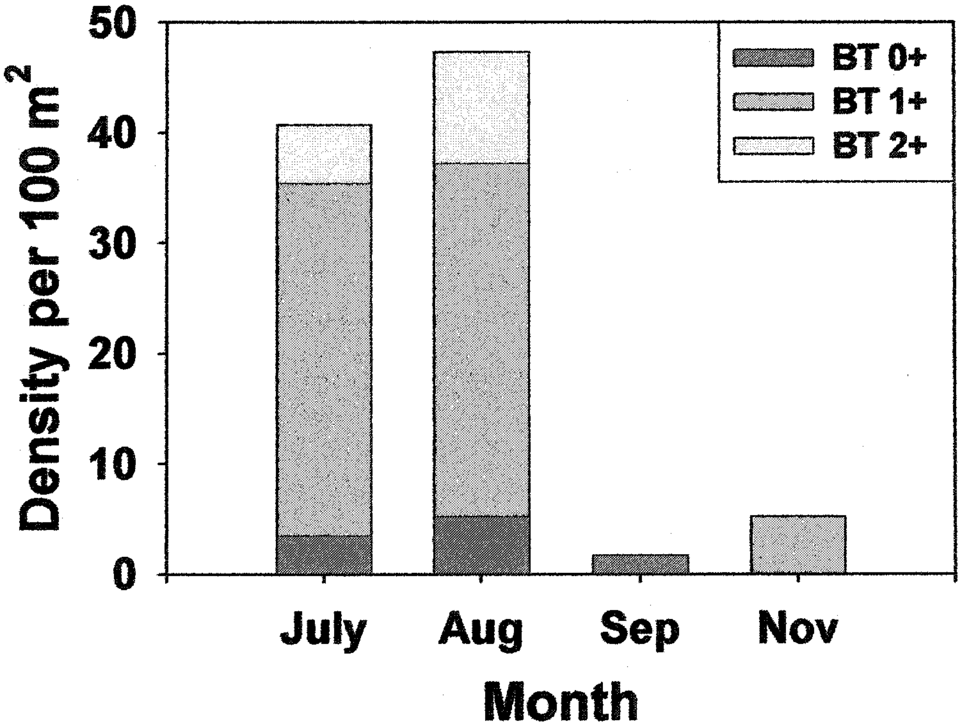


Figure 4-17. Population density (number/100 m^2) of brook trout age classes during each sample period at WR4 in 2002.

(Fig. 4-1, p. 90). The branch connected to the main stem of the river 0.89 km downstream of the pesticide runoff site that caused the second pesticide runoff event (20 July). The sampling site was located 0.69 km upstream from where the branch connected with the main stem. Salmonid density was 181.1 fish/100 m^2 23 July 2002, and decreased consistently throughout the summer and fall (Fig. 4-18). Only brook trout were captured July-September. Rainbow trout were captured 29 November, but their density was low (3.5 fish/100 m^2). The majority of brook trout captured at all sample dates were 0+ fish (Fig. 4-19). The proportion of larger age classes increased slightly during the 29 November sampling. Of the rainbow trout that were captured, a density of 2.5 0+ fish and 1.1 2+ fish per 100 m^2 was found.

WR8 was located on the main stem downstream of where pesticide runoff entered from both pesticide runoff events (Fig. 4-1, p. 90). It was 0.79 km downstream from WR7, and 2.623 km downstream from where the runoff from the second pesticide runoff event entered the Wilmot River. WR8 was >7.5 km downstream from all three entry points for the first pesticide runoff event. Salmonid density at WR8 was less than 20 fish/100 m^2 for the 23 July sample, and 23 fish/100 m^2 for the 12 August sample (Fig. 4-20). The density almost doubled between 12 August and 2 September, but decreased dramatically for the 29 November sample. The majority of salmonids captured during all sampling periods were rainbow trout. 0+ salmonids had the highest density in both brook and rainbow trout for the July and September samples

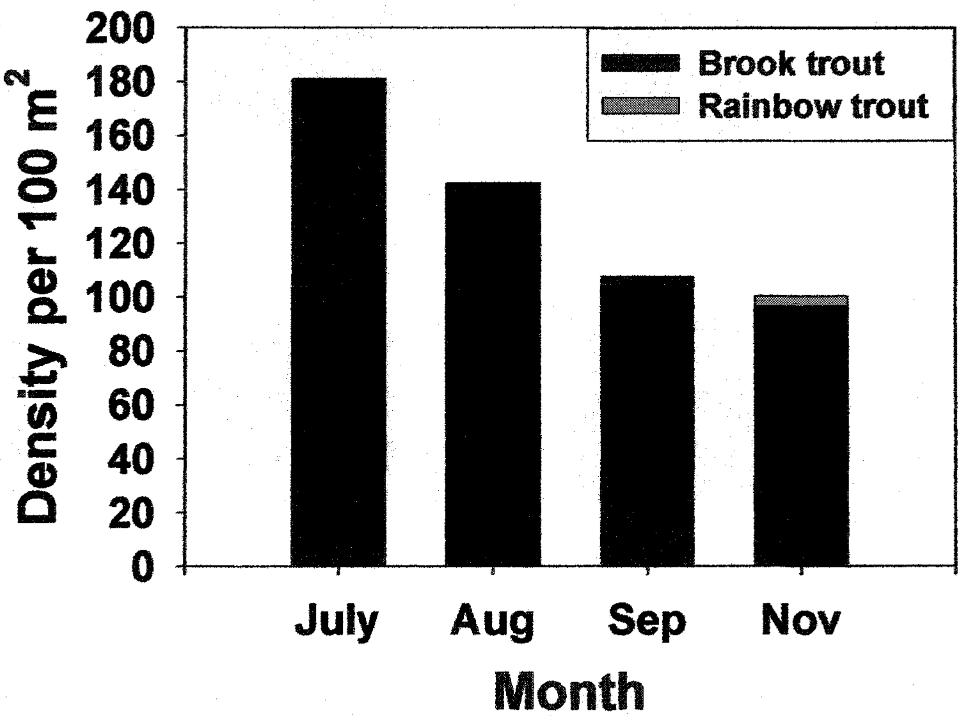


Figure 4-18. Population density (number/100 m^2) of brook trout and rainbow trout during each sample period at WR6 in 2002.

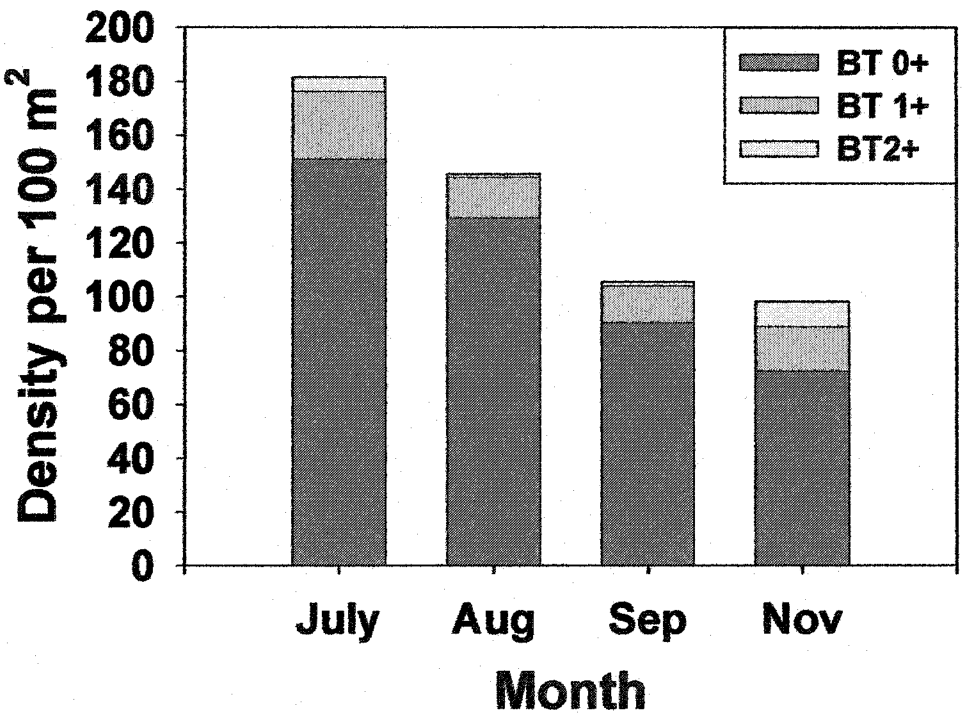


Figure 4-19. Population density (number/100 m^2) of brook trout age classes during each sample period at WR6 in 2002.

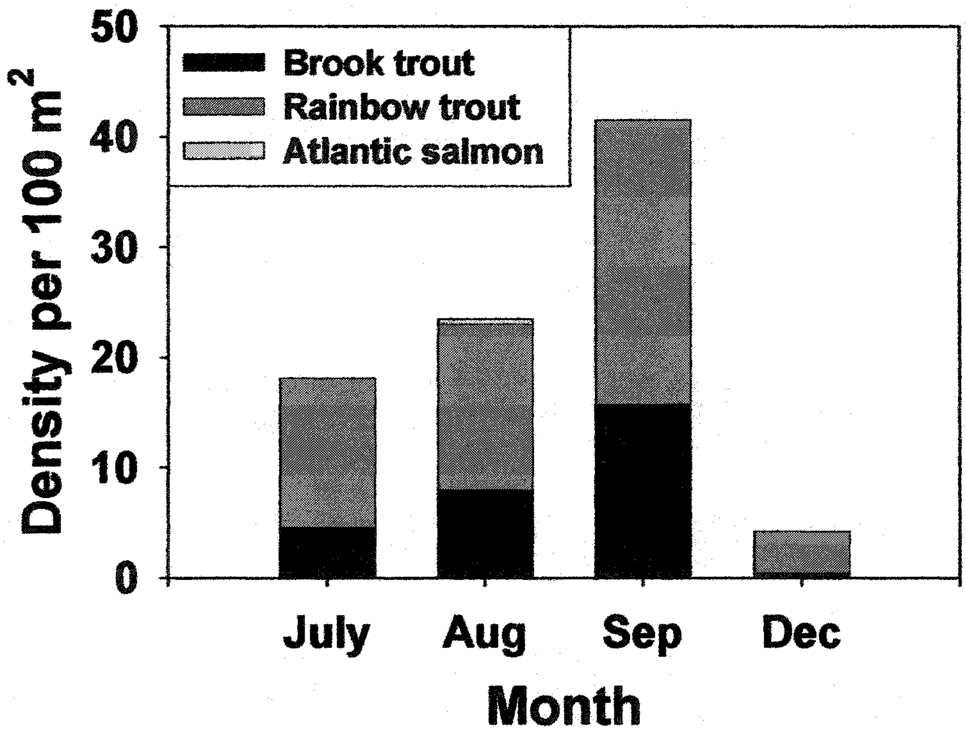


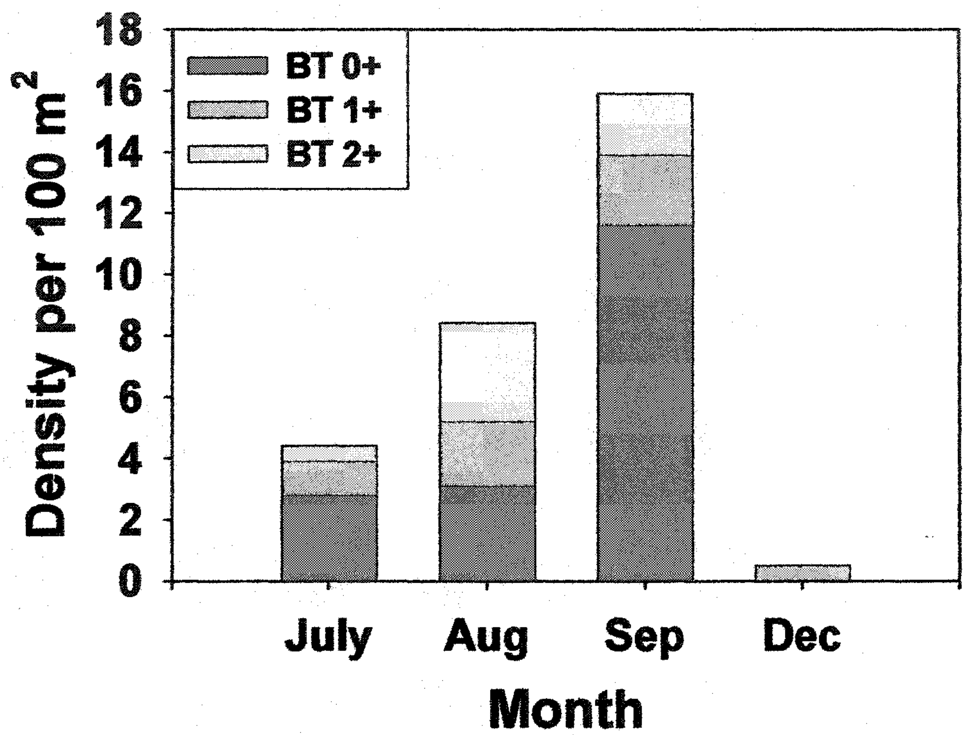
Figure 4-20. Population density (number/100 m^2) of brook trout, rainbow trout, and Atlantic salmon during each sample period at WR8 in 2002.

(Fig. 4-21 A and B). Larger age classes had higher densities in both brook and rainbow trout in the August sample. In the November sample, only 1+ brook trout and 0+ rainbow trout were captured.

4.3.3 Water Chemistry Results

The water chemistry results are given in Table 4-1 and 4-2. All parameters have values within normal ranges, although nitrate levels are higher than in most other streams on PEI. The average, maximum, and minimum water temperatures at WR2, WR5, and WR7 for each sampling period are given in Table 4-3.

A



B

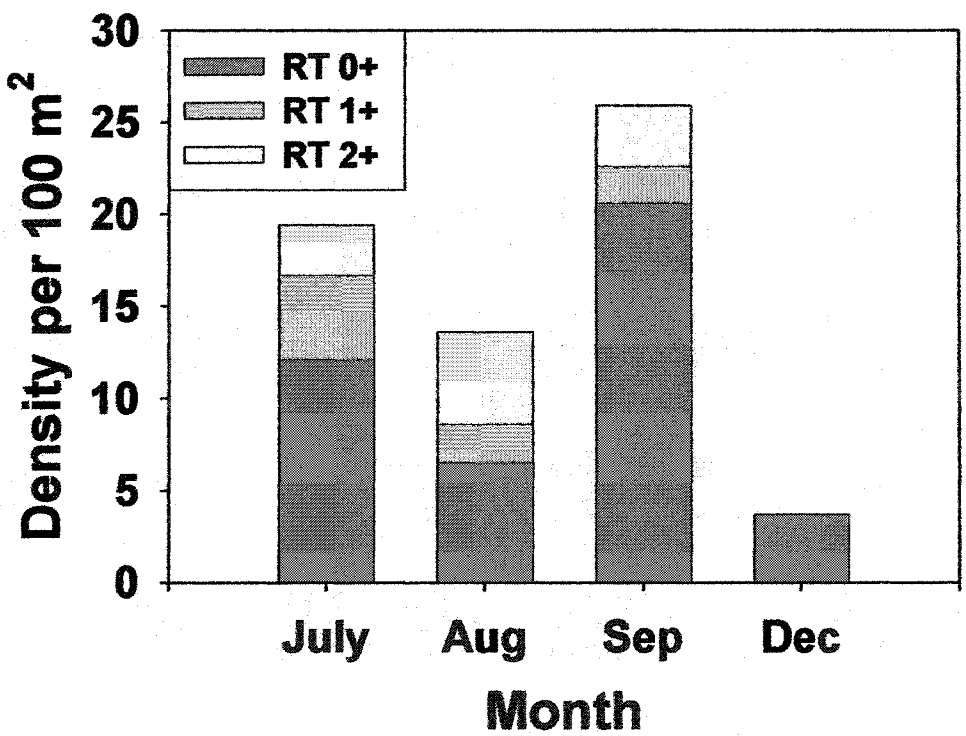


Figure 4-21. Population density (number/100 m^2) of (A) brook trout and (B) rainbow trout age classes during each sample period at WR8 in 2002.

Table 4-1. Water temperature (June, July, Aug), dissolved oxygen (July, Aug), pH (July, Aug), conductivity (July, Aug), nitrate (Aug), phosphorus (Aug), and alkalinity (Aug) values obtained during each sampling event during 2001.

Site	Date	water temp (°C)	dissolved oxygen (mg/l)	pH	conductivity µS/cm	nitrate-N (mg/l)	phosphorus (mg/L)	alkalinity (mg/l)
WR2	June	9.9						
	July	13.8	10.2	8.5	265			
	Aug	11.7	10.8	8.0	245	7.8	0.054	92.2
WR5	June	16.4						
	July	17.8	10.9	8.5	285			
	Aug	15.6	11.2	8.5	250	6.9	0.000	87.2
WR7	June	16.0						
	July	17.9	10.8	8.5	290			
	Aug	14.2	11.3	8.0	250	7.2	0.027	99.0

Table 4-2. Water temperature, dissolved oxygen, pH, conductivity, nitrate, phosphorus, and alkalinity values obtained during each sampling event during 2002.

Site	Date	water temp (°C)	dissolved oxygen (mg/l)	pH	conductivity (µS/cm)	nitrate-N (mg/l)	phosphorus (mg/L)	alkalinity (mg/l)
WR1	22 July	11.6	12.0	7.7	260	8.7	0.00	87.6
	11 Aug	12.0	11.8	7.9	290	7.4	0.04	83.0
	1 Sep	9.8	11.6	8.2	250	8.5	0.05	75.0
	1 Nov							
WR2	12 July	10.1	11.3	7.9	250	7.8	0.04	87.5
	11 Aug	13.6	11.0	7.7	280	7.0	0.05	85.2
	1 Sep	10.1	11.5	8.2	225	7.9	0.054	76.7
	31 Oct							
WR3	14 July	N/A	N/A	N/A	N/A	N/A	N/A	N/A
	11 Aug	12.9	10.8	7.8	280	7.2	0.05	89.9
	1 Sep	8.9	11.9	8.3	250	8.1	0.047	76.2
	1 Nov							
WR4	25 July	10.4	11.0	7.5	270	9.1	0.02	92.9
	11 Aug	9.7	11.4	7.5	280	8.2	0.04	81.8
	1 Sep	7.8	12.1	8.2	260	8.8	0.042	73.1
	1 Nov							
WR5	13 July	15.2	11.4	9.0	275	7.1	0.05	89.4
	20 July	15.8	10.2	8.4	265	N/A	N/A	N/A
	11 Aug	15.4	11.1	8.4	280	6.4	0.03	89.5
	1 Sep	13.9	12.3	9.3	220	6.3	0.153	61.9
	31 Oct							

continued...

Table 4-2. continued.

Site	Date	water temp (°C)	dissolved oxygen (mg/l)	pH	conductivity (µS/cm)	nitrate-N (mg/l)	phosphorus (mg/L)	alkalinity (mg/l)
WR6	23 July	14.7	9.6	7.6	255	7.2	0.00	88.7
	12 Aug	12.5	9.7	7.5	240	6.5	0.03	94
	1 Sep	12.0	9.5	7.7	250	6.6	0.025	80.4
	29 Nov							
WR7	13 July	14.4	9.8	8.7	260	7.2	0.05	89.4
	20 July	15.1	11.0	8.6	250	N/A	N/A	N/A
	11 Aug	16.7	11.4	8.4	270	6.4	0.03	94.2
	1 Sep	14.5	11.2	9.0	250	6.5	0.024	74.6
	31 Oct							
WR8	23 July	18.4	13.0	8.8	270	6.9	0.00	87.6
	12 Aug	15.8	12.4	8.4	255	6.2	0.04	87.4
	2 Sep	13.2	12.9	8.8	240	6.8	0.028	69.9
	29 Nov							

Table 4-3. Average, maximum, and minimum temperatures for the day sampling occurred and the four days previous.

Site	Date	5-day avg temp (°C)	5-day max temp (°C)	5-day min temp (°C)
WR2	12 July	10.8	12.8	8.8
	11 Aug	11.5	14.0	9.5
	1 Sep	10.5	12.2	7.8
	31 Oct	6.5	8.1	3.2
WR5	13 July	15.2	19.6	12.8
	20 July	16.5	19.8	14.6
	11 Aug	16.3	19.4	14.0
	1 Sep	15.4	17.8	13.6
	31 Oct	5.5	7.7	3.9
WR7	13 July	13.9	17.1	11.5
	20 July	14.7	17.4	12.9
	11 Aug	14.9	17.1	12.9
	1 Sep	13.3	15.7	10.7
	31 Oct	6.0	7.1	3.2

4.4 DISCUSSION

The fish kill events of 2002 dramatically altered the structure of the salmonid communities of Wilmot River. The population structure of salmonids was altered with the differential effect of the pesticide on different age classes. Similarly, the community structure of the river was altered through the differences in the response of the salmonid species to the pesticide.

4.4.1 Changes in Population Structure

For both species, young of the year (0+) fish initially suffered a larger decline in population density than older age classes. Smaller fish are generally found to be more susceptible than larger fish to toxicants (McKim 1977, Van Leeuwen et al. 1985, Murty 1986). However, throughout the summer, the 0+ salmonid density increased in some areas, particularly WR7. As the 0+ fish were larger each month, and showed little overlap in length between sampling periods, it seems that the increase in 0+ fish resulted from 0+ fish from other areas recolonizing the affected areas. Conversely, density of larger age class salmonids decreased in many affected areas (WR5, WR7). This may have been due to movement, as the larger fish tried to find areas with more food or cover. It may also have been due to delayed mortality from decreased food availability and/or increased predation pressure.

4.4.2 Changes in community structure

After the entry of pesticides into the Wilmot River, brook trout populations decreased substantially; rainbow trout populations also decreased, although not to the same extent as brook trout. This resulted in a change in the community structure from brook trout dominated sites to rainbow trout dominated sites. Throughout the summer, this change in the proportion of each species present remained the same.

This study is not the first on PEI to find differences in how brook and rainbow trout populations are affected after pesticides enter a stream. In August 1975 the pesticide endosulfan entered into North Brook, a tributary of the Dunk River, PEI, in quantities sufficient to cause significant fish mortality (Johnston and Cheverie 1980). A site on this river had been sampled annually from 1973-1976. The 1975 sampling was performed prior to the fish kill; sampling was not undertaken in 1975 after the fish kill as it was thought this might disrupt spawning. The 1976 sampling revealed populations of brook and rainbow trout that were 38% and 73% the size of the average population of the previous three years, indicating a possible change in the relative abundance of salmonid species. Fish tagging at this and other sites along the Dunk River in previous years revealed that trout moved up to 5-6 km downstream the main stem of the river and then up into the affected tributary to recolonize the area.

In 1999 azinphos-methyl entered the west branch of the Tryon River, resulting in a major fish kill (Mutch et al. 2002). Data on fish populations before the fish kill are not available, but the population was dominated by rainbow trout

(89% of population density in August, pers. obs.) two years after the event (2001). Population density was also extremely low (36 salmonids/100m^2) during this sampling period. Sampling three years after the event (August 2002) revealed brook and rainbow trout densities that were closer to equal (62% rainbow trout). Population density also greatly increased (110 salmonids/100m^2; pers. obs.). Although both species appear to be recovering in this stream, there is still a higher proportion of rainbow trout than is typical of other streams across PEI (see Chapter 3, Guignion et al. 2002). No other stream in this study experienced a recent fish kill event (i.e., within the past five years; see Table 4-5 and Table 3-1, p. 35).

The consistently higher proportion of rainbow trout found in areas of rivers affected by pesticide runoff indicates a difference in the response of the two species to pesticide exposure. There are two possible mechanisms that may account for this difference: behavioural and physiological. Brook and rainbow trout, when exposed to a toxin, may behave differently. Brook trout are known to swim upstream when startled. If they swam upstream during a pesticide runoff event, their exposure to the pesticide could be increased, resulting in a higher mortality. If rainbow trout behaved oppositely, and swam downstream, they would be expected to suffer lower mortality than brook trout. However, it is unlikely that fish would swim into a region of higher pesticide concentration, as avoidance of chemicals is well-documented, and generally occurs at

Table 4-4. Documented fish kills on Prince Edward Island. If certain factors have been implicated in the fish kill it has been noted under "probable cause". References are given at the end of the table.

Date	River	Probable Cause
4 August 1962[a]	Mill River	pesticide spill: nabam and endrin
6 August 1966[b]	Tryon River (east branch)	pesticide dithane (mancozeb) can found nearby
28 June 1967[b]	Trout River (Coleman)	
July 1967[b]	Black Pond, Greenvale	
4 August 1967[b]	North River	pesticide: endrin
15 August 1967[b]	Newton River	
29 August 1967[b]	Bradshaw River	
August 1967[b]	Morell River	
August 1968[c]	Dunk River	pesticides
August 1968[c]	Orwell River (Kinross)	pesticides
15 August 1968[c]	Morell River	pesticides
25 August 1969[d]	DeSable River	pesticide: Dithane M-45 and DDT
19 June 1971[b]	West River	pesticide: endrin
14 August 1975[b]	Valleyfield River	pesticide: endrin
28 August 1975[e]	Dunk River, North Brook	pesticide spill: endosulfan
28 June 1977[b]	Dunk River	pesticide premerge (3 DinitroAmine) containers found nearby
summer 1990[b]	Westmoreland River	pesticide: endosulfan
27 July 1994[b]	Big Pierre Jacques River	pesticides: carbofuran and azinphos methyl
summer 1994[b]	Westmoreland River	
21-22 July 1995[b]	Big Pierre Jacques River	
25 July 1995[b]	Long Creek (Profit's Pond)	pesticide spill: mancozeb
20 July 1996[b]	Long Creek (Profit's Pond)	pesticide: chlorothalonil
23 July 1998[b]	Huntley River	pesticides: azinphos methyl and endosulfan
11 July 1999[b]	Valleyfield River	pesticides: azinphos methyl and endosulfan
12 July 1999[b]	Orwell River	

continued...

Table 4-4. continued

Date	River	Probable Cause
14 July 1999[b]	Souris River	pesticides: endosulfan and dithiocarbamate
19 July 1999[b]	Tryon River (west branch)	pesticide: azinphos methyl
20 July 1999[b]	Westmoreland River	pesticides: chlorothalonil and endosulfan
21 July 1999[b]	Clyde River	
29 July 1999[b]	Orwell River	pesticides: azinphos methyl and endosulfan
13 August 1999[b]	Trout River (Tyne Valley)	pesticides: azinphos methyl and chlorothalonil
20 July 2000[f]	Souris River	pesticide: dithane (mancozeb)
9 August 2000[g]	Indian River	pesticides: azinphos methyl, chlorothalonil, and endosulfan
9 August 2000[h]	French River	pesticide: azinphos methyl
9 August 2000[i]	Fullerton's Creek	pesticide: chlorothalonil
9 July 2002[j]	Wilmot River (Murphy's Bridge)	pesticide: azinphos methyl
19 July 2002[j]	Wilmot River	pesticide: azinphos methyl
19 July 2002[j]	North River	
19 July 2002[j]	Clyde River	
21 July 2002[j]	Trout River (Coleman)	
25 July 2002[j]	Huntley River	
20 August 2002[j]	Westmoreland River (west branch)	
20 August 2002[j]	Westmoreland River (east branch)	

Data obtained from [a] Saunders 1969, [b] Mutch et al. 2002, [c] Yeo 1968, [d] Editor 1969a and 1969b, [e] Johnston and Cheverie 1980, [f] PEI DFAE 2000c, [g] PEI DFAE 2000d, [h] PEI DFAE 2000b, [i] PEI DFAE 2000a, [j] pers. obs. and local news reports.

concentrations lower than that required to cause mortality (e.g., Hansen 1969 *cited from* Murty 1986, Hansen et al. 1972, Atchison et al. 1987). For example, after the pesticides nabam and endrin were spilled into the Mill River in 1962, both brook trout and Atlantic salmon were observed swimming downstream, away from the source of the spill (Saunders 1969). A two-way counting fence at the head of tide confirmed this observation, with a drastic increase in the movement of fish into the estuary. Thus, it is unlikely that brook and rainbow trout would have different behavioural responses to a pesticide runoff event.

Brook trout and rainbow trout may also have different physiological processes that cause them to have a different sensitivity to certain chemicals. This could be due to differences in uptake or metabolism of the compound. Regardless of the mechanism, a difference in sensitivity could explain the difference found in mortality in this study. Azinphos-methyl, which was the pesticide responsible for the fish kills on the Wilmot River, has been found to have different toxicity values for different salmonid species. Ninety-six hr LC_{50} values average 5.28 μg/L for rainbow trout (number of studies = 5), 2.87 μg/L for Atlantic salmon (number of studies = 7), and 1.2 μg/L for brook trout (number of studies = 1) (USEPA 1998). These LC_{50} values support the theory that the observed differential mortality between brook and rainbow trout after the pesticide runoff events is due to a difference in sensitivity of these species to the chemical. LC_{50} values of other pesticides that have been linked to other pesticide runoff associated fish kills on PEI are given in Table 4-5.

Table 4-5. LC50 values for Atlantic salmon, brook trout, and rainbow trout of the six pesticides associated with fish kills on Prince Edward Island since 1990.

Chemical	Species	Test	Concentration
Azinphos-methyl	Atlantic salmon	96-hr LC_{50}	2.87 µg/L[a]
	Brook trout	96-hr LC_{50}	1.2 µg/L[a]
	Rainbow trout	96-hr LC_{50}	5.28 µg/L[a]
		48-hr LC_{50}	4 µg/L[b]
Carbofuran	Rainbow trout	96-hr LC_{50}	380 µg/L[c]
Chlorothalonil	Rainbow trout	96-hr LC_{50}	42.3 µg/L[d]
Dithiocarbamate			
Endosulfan	Rainbow trout	48-hr LC_{50}	1 µg/L[b]
Mancozeb	Rainbow trout	48-hr LC_{50}	2200 µg/L[e]

Data obtained from [a] USEPA 2001, [b] Pimental 1971 *cited from* Brown 1978, [c] Johnson and Finley 1980, [d] USEPA 1999, [e] Extoxnet 1996.

4.4.3 History of Pesticide Associated Fish Kills on Prince Edward Island

Since 1962, at least 44 documented fish kills that were either proved or suspected to have been caused by pesticides occurred on Prince Edward Island (see Table 4-4, p. 132); sixteen of these kills occurred between 1994 and 2000. There were no reported fish kills in 2001, but there was very little rain during July and August of 2001, which reduced the amount of runoff during this time of intensive pesticide use. Eight fish kills thought to be associated with pesticides occurred throughout July and August 2002, including two separate fish kills on the Wilmot River.

On Prince Edward Island, 16 of the fish kills since 1994 have had specific pesticides attributed to them; of these 16, 10 involved azinphos-methyl (see Table 4-5), including both of the reported Wilmot River fish kills (B. Birch, pers. comm.). As of the year 2000, the pesticide azinphos-methyl was determined responsible for 143 fish kill incidents in the United States, which is 21% of all reported pesticide-associated fish kills in that country (USEPA 2001).

Azinphos-methyl is an organophosphate insecticide used on a number of food crops, including potatoes. It has a half-life of 26 d at 30°C and pH 7. Its metabolites (dimethylphosphorothioic and dimethylphosphoric acids, desmethyl azinphosmethyl, and azinphos-methyloxon) are thought to be more toxic than the parent compound, as toxicity increases after metabolism (WHO 2002). Azinphos-methyl is highly soluble in water (25.10 mg/L at 25°C), and does not readily leach through soil. It therefore is likely to be in runoff associated with rainfall (USEPA 1998).

Azinphos-methyl is an acetylcholinesterase inhibitor, and thus acts on the nervous system (USEPA 1998). Acetylcholine is thought to affect various fish behaviours, such as schooling, temperature selection, respiration, feeding, and rheotropism (Smith 1984). Disruption of some of these behaviours (e.g., respiration) could cause acute mortality; disruption of other behaviours (e.g., feeding, rheotropism) could lead to mortality over a longer period of time. The long-term effects of short-term exposure to azinphos-methyl runoff into rivers could have further implications for the fish populations.

4.4.4 Conclusions

Population and community structure of salmonids of the Wilmot River were severely impacted by the two fish kills that occurred in July 2002. Brook trout suffered higher mortality than rainbow trout, and young of the year fish suffered higher mortality than larger age classes. This resulted in rainbow trout outnumbering brook trout, whereas in unaffected (i.e., WR2 and WR6) or previously unaffected (i.e., WR5 and WR7 before pesticide runoff events) sites, brook trout outnumbered rainbow trout. As rainbow trout are an exotic, this is cause for concern for the native brook trout. Losing a large proportion of the young of the year salmonids from the Wilmot River may impact the populations for years to come. Young of the year fish are not only responsible for providing fish for future generations, but are also an important food source for the older fish in the streams.

The long-term impacts of pesticide runoff events are not well-understood, as data on fish community and population structure before the insult are often not readily available. Without this information, decreases of high percentages of fish populations may not be realized, and the extent of known fish kills may go unrecognized. This study is important, in that the data on salmonid populations before the pesticide runoff events are available to compare to the populations after the event. Routine sampling of rivers in areas at high-risk for fish kills is therefore important as data will be available in case of a reported fish kill event. Additionally, this routine sampling may pick up unreported fish kill incidents. There were only two reported fish kill incidents on the Wilmot River in 2002. However, at one of the sample sites (WR4), brook trout populations decreased by over 96% between the August and September samples. Although no fish kill was reported for this site, it is located in an isolated area, and therefore a pesticide-related kill may go unnoticed. The watershed above this site is 74% agriculture, with 46% of this (34% of watershed) in potato production, so the site is in an area at high risk for pesticide runoff. If this decrease in salmonid densities was the result of a pesticide runoff event, this is an example of one of the number of unreported fish kills that may be occurring every year on PEI.

The long-term impacts of the fish kills on the Wilmot River will not be known until further sampling is done in subsequent years. This will reveal long-term impacts on survival and reproduction. However, the preliminary results from 2001 and 2002 are cause for concern. Atlantic salmon is declining through much of its range (Parrish et al. 1998); the input of pesticides into historical salmon rivers provides a further stressor to populations. Rainbow trout, an exotic species, are apparently not as affected as brook trout, a native species, by pesticides that are causing fish kills in PEI. This indicates that pesticide input into streams on PEI may select for a non-native species. Conservation efforts for preserving native populations of Atlantic salmon and brook trout should target reduction of pesticides into rivers of Prince Edward Island.

CHAPTER 5:

CONCLUSIONS

5.1 SUMMARY

All fourteen rivers examined on Prince Edward Island had healthy populations of salmonids in at least one of the sites sampled. Brook trout are widespread throughout the province, and were found at all thirty-seven sites sampled. Atlantic salmon were much more restricted in their range, and only five of the fourteen rivers examined contained healthy populations. Rainbow trout populations were high in three of the fourteen rivers examined, but were found at lower population densities in three other rivers. The three species were found together in only two sites, on two different rivers. There was never more than one salmon captured at either of these sites. The ranges of Atlantic salmon and rainbow trout appear to be mostly limited by historical factors. Atlantic salmon are not found in rivers where impassable barriers previously limited migration. Rainbow trout are found in rivers in which they were stocked, or in which aquaculture facilities from which they escaped were located. They are also found in rivers adjacent to these, where they likely migrated from their original colonies. As Atlantic salmon and rainbow trout are known to occupy similar niches, it is expected that there will be competition between the two in areas where they both exist. Since rainbow trout have been found to be more aggressive than Atlantic salmon, it is possible their introduction could be contributing to the decrease in Atlantic salmon populations.

The only biotic variables explained by the stream and watershed variables examined were brook trout density, variability in total density, total percent habitat saturation, variability in total percent habitat saturation, 1+ brook trout condition factor, and $\delta^{15}N$ values. Of these, only the model for $\delta^{15}N$ values was useful in predicting values from the five test rivers. Stable isotope analysis may therefore prove to be a useful tool in examining differences between salmonid populations in different rivers of Prince Edward Island. These values are expected to be less influenced by historical factors than density and percent habitat saturation, and not as related to density and competition as condition factors are.

Fish kills related to pesticide runoff that result in the deaths of thousands of salmonids are becoming a common occurrence on Prince Edward Island during the months of July and August when pesticide applications are most intensive. The extended effects of these events on salmonid population and community structure are not well-understood, as data on the population before the runoff event is usually not available. On the Wilmot River, two pesticide runoff events in July 2002 were found to affect the native brook trout populations more than the introduced rainbow trout populations. Additionally, within the species, the 0+ fish were affected more than the older age classes. The long-term effects of these events on subsequent reproduction and population structure will be further investigated over the next few years.

5.2 FURTHER STUDY

Stable isotope analysis appears to be the most reliable tool in examining differences between populations of salmonids. Using this method would require a more in-depth study, as the food web (autochthonous and allochthonous vegetation, various macroinvertebrates, and different fish species and age classes) in these rivers must be understood to fully interpret these results. The other models developed may prove useful in predicting the biotic variables if more study sites were used. These models should be reassessed using a larger number of sites and/or rivers.

Although pesticide runoff events are more common in the watersheds with extremely high levels of potato rotation, they can occur in any area where row crops are farmed. All of the watersheds examined in this study had some potato farming, so their streams could potentially be at risk in the future. The timing and locations of these events are unpredictable, as are their long-term effects on the salmonid populations. To more fully understand the scope of the problems associated with these events, monitoring of salmonid populations in watersheds across Prince Edward Island should be continued, especially in areas under intense potato production.

CHAPTER 6

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APPENDIX A:

MEAN WATER CHEMISTRY, INSTREAM, RIPARIAN ZONE, AND WATERSHED VARIABLES FOR EACH SITE EXAMINED

Table A-1. Mean water chemistry values for each site obtained during sampling periods or from data loggers.

Variable	BP2	BP3	BP1	CR2	CR3	CR1	DR1	DR2	DR3	ML3	ML1	ML2
Alkalinity (mg/L)	99.6	119.2	107.6	102.4	90.8	100.8	79.6	73.0	84.5	104.9	106.5	93.6
Conductivity (µmhos)	231.3	302.0	259.0	203.4	188.6	185.4	157.2	115.2	158.0	241.8	262.0	178.8
Dissolved oxygen (mg/L)	8.63	10.98	10.49	11.50	10.63	9.97	10.58	11.19	10.99	10.26	10.37	8.16
pH	7.76	7.92	8.10	8.60	8.35	7.91	7.99	7.93	8.35	8.29	8.37	7.81
Phosphorous (mg/L)	0.019	0.022	0.012	0.036	0.032	0.062	0.059	0.055	0.046	0.017	0.239	0.034
Nitrate (mg/L)	3.43	7.63	5.03	0.48	0.45	0.25	2.33	0.68	2.33	3.15	4.18	0.15
Max temp (°C) (June-Septempber)	18.6	17.7	18.8	20.6	19.3	17.9	15.7	13.3	15.5	20.8	20.1	22.2
Mean June temp (°C)	13.6	9.6	10.6	15.0	11.6	11.0	10.7	8.7	9.3	12.4	12.8	17.6
Mean July temp (°C)	13.4	10.9	12.6	15.7	14.5	13.2	10.7	10.1	11.1	15.3	15.3	17.2
Mean August temp (°C)	14.3	11.8	13.5	12.8	14.9	13.3	11.0	10.5	11.6	15.8	15.7	17.5
Mean September temp (°C)	12.4	10.6	11.9	8.4	12.4	11.7	10.2	9.8	10.3	12.8	12.9	14.0

continued...

Table A-1. Continued.

Variable	MM1	MM2	MM3	NL1	NL3	NL2	PP2	PP3	PP1	TR3	TR2	TR1
Alkalinity (mg/L)	127.4	132.7	122.6	74.6	89.7	60.6	97.8	99.7	68.5	87.0	73.4	104.4
Conductivity (μmhos)	277.0	274.0	265.0	176.0	188.0	146.0	148.7	181.0	162.0	195.0	168.0	271.0
Dissolved oxygen (mg/L)	8.88	10.23	9.58	10.17	11.12	8.77	10.80	11.42	7.83	12.74	11.69	10.97
pH	7.89	8.09	8.24	8.15	8.40	7.74	8.28	8.15	7.96	8.75	7.80	7.84
Phosphorous (mg/L)	0.171	0.053	0.163	0.074	0.059	0.051	0.074	0.047	0.057	0.039	0.193	0.052
Nitrate (mg/L)	0.10	1.00	0.53	0.40	0.55	0.00	0.10	0.10	0.00	3.33	3.78	7.53
Max temp (°C) (June-Septempber)	N/A	20.0	23.8	23.8	17.6	22.2	N/A	18.2	28.5	19.2	N/A.	20.2
Mean June temp (°C)	N/A	15.8	14.1	12.3	10.5	12.2	N/A	10.9	17.6	10.9	N/A	9.4
Mean July temp (°C)	N/A	14.4	17.2	16.1	12.8	14.2	N/A	13.0	20.3	13.5	N/A	10.7
Mean August temp (°C)	N/A	14.3	17.6	16.9	13.2	16.0	N/A	13.5	20.9	14.0	N/A	11.2
Mean September temp (°C)	N/A	12.2	13.9	13.5	11.3	13.1	N/A	11.6	17.0	11.7	N/A	10.4

continued...

Table A-1. Continued.

Variable	WR1	WR2	WR3	WR4	WR5	WR6	WR7	WR8	MG	MR	NA	SP	WH
Alkalinity (mg/L)	81.0	85.4	83.1	82.6	81.8	87.7	89.3	81.6	95.1	117.7	95.6	88.2	94.3
Conductivity (µmhos)	266.7	253.0	265.0	270.0	262.0	249.0	265.0	254.3	185.0	238.3	200.7	223.3	233.3
Dissolved oxygen (mg/L)	7.93	10.93	8.03	7.69	11.36	7.55	10.96	8.64	11.14	10.96	9.84	10.42	11.12
pH	11.80	7.96	11.33	11.51	8.74	9.59	8.46	12.72	8.33	8.12	8.56	8.16	8.25
Phosphorous (mg/L)	0.030	0.050	0.049	0.034	0.058	0.018	0.033	0.023	3.600	26.07	1.767	4.067	5.433
Nitrate (mg/L)	8.20	7.63	7.65	8.70	6.68	6.77	6.83	6.63	2.33	3.13	0.20	1.20	3.67
Max temp (°C) (June-Septempber)	N/A	15.1	N/A	N/A	23.1	N/A	19.2	N/A	14.90	14.90	24.90	23.20	21.60
Mean June temp (°C)	N/A	9.7	N/A	N/A	14.2	N/A	12.3	N/A	10.8	10.1	18.1	15.9	12.4
Mean July temp (°C)	N/A	11.3	N/A	N/A	16.3	N/A	14.6	N/A	11.0	9.8	20.7	17.1	13.4
Mean August temp (°C)	N/A	11.9	N/A	N/A	17.4	N/A	15.0	N/A	11.5	10.0	19.2	17.6	14.2
Mean September temp (°C)	N/A	11.1	N/A	N/A	14.2	N/A	13.0	N/A	10.6	9.7	15.5	13.9	11.9

Table A-2. Mean instream habitat variables for each site.

Variable	BP2	BP1	BP3	CR2	CR3	CR1	DR1	DR2	DR3	ML3	ML1	ML2
Bank full depth (cm)	23.49	31.47	33.67	22.79	29.62	41.85	33.68	40.39	40.03	41.78	23.30	32.67
Mean width (cm)	260	290	426	385	770	378	255	220	575	948	250	315
Mean wetted width (cm)	226	300	347	353	604	502	387	316	743	786	419	254
Stream slope x 10^3	1.23	0.84	2.38	5.54	7.07	3.37	3.35	18.13	5.43	0.38	2.97	2.62
Velocity (m/s)	0.0419	0.254	0.257	0.31	0.343	0	0.529	0.434	0.175	0.372	0.363	0.0895
Discharge	0.02	0.05	0.17	0.21	0.38	0.00	0.13	0.07	0.23	0.42	0.11	0.03
Stability (%)	0.0	47.4	34.1	100.0	44.7	10.1	26.3	41.4	49.5	91.7	81.6	31.6
Rock size (cm)	0.01	5.69	6.53	8.21	2.53	0.77	0.89	8.37	2.46	5.90	6.90	1.94
Wentworth rating: walk	1.00	2.34	1.59	3.57	2.37	1.45	1.61	3.34	2.44	3.13	3.10	1.68
Wentworth rating: transects	1.00	1.61	1.27	2.87	2.20	1.00	1.22	2.50	1.92	2.73	3.12	1.04
% silt: walk	100.0	52.6	65.3	0.0	21.3	69.4	61.6	5.1	11.1	8.3	18.4	68.4
% silt: transects	100.0	71.9	64.1	3.3	31.9	00.0	85.3	14.3	42.3	20.0	20.0	97.2
Pfankuch evalution score	134.0	123.0	126.0	94.0	80.5	118.0	131.0	107.0	118.0	80.0	72.5	132.0

continued...

Table A-2. Continued.

Variable	MM1	MM2	MM3	NL1	NL3	NL2	PP2	PP3	PP1	TR3	TR2	TR1
Bank full depth (cm)	46.59	41.94	55.03	21.79	34.27	32.09	27.81	31.00	21.10	42.98	29.48	31.48
Mean width (cm)	180	272	470	300	725	150	320	460	230	345	195	370
Mean wetted width (cm)	269	344	706	425	650	130	302	422	217	474	283	298
Stream slope x 10^3	1.23	8.14	1.35	2.41	0.18	12.92	12.18	10.47	6.33	13.25	7.29	13.23
Velocity (m/s)	0.177	0.192	0.163	0.288	0.446	0.307	0.474	0.308	0.252	0.197	0.398	0.186
Discharge	0.02	0.03	0.08	0.16	0.43	0.03	0.13	0.16	0.03	0.08	0.07	0.06
Stability (%)	21.7	13.3	85.7	81.8	77.8	38.4	37.4	69.8	58.7	30.5	32.6	44.6
Rock size (cm)	0.96	1.90	25.15	3.55	3.77	4.39	3.33	18.34	3.10	5.61	2.55	18.43
Wentworth rating: walk	1.41	1.99	2.11	2.69	2.71	2.70	2.61	2.65	2.36	3.11	1.93	2.14
Wentworth rating: transects	1.25	1.91	1.70	2.20	2.18	1.83	2.00	2.73	2.00	1.95	2.10	1.23
% silt: walk	78.3	44.9	11.2	18.2	22.2	18.2	20.2	11.5	30.4	2.1	54.3	26.4
% silt: transects	83.3	47.8	50.0	30.0	21.2	48.1	40.7	13.3	36.7	39.4	46.7	84.6
Pfankuch evalution score	133.0	95.0	104.0	85.0	75.0	129.0	82.0	86.0	102.0	99.0	126.0	128.0

continued...

Table A-2. Continued.

Variable	WR1	WR2	WR3	WR4	WR5	WR6	WR7	WR8	MG	MR	NA	SP	WH
Bank full depth (cm)	47.09	26.38	50.91	46.36	29.08	44.47	22.90	83.84	22.52	46.54	28.98	32.5	29.0
Mean width (cm)	217	345	389	317	390	397	650	596	385	225	650	700	668
Mean wetted width (cm)	232	363	381	368	455	264	613	610	407	300	631	726	680
Stream slope x 10^3	0.68	2.31	0.28	1.46	9.65	10.66	4.87	1.10	8.63	10.00	1.36	6.41	2.53
Velocity (m/s)	0.118	0.284	0.248	0.091	0.355	0.100	0.432	0.284	0.302	0.257	0.381	0.26	0.23
Discharge	0.04	0.14	0.19	0.06	0.26	0.05	0.51	0.36	0.11	0.04	0.41	0.35	0.30
Stability (%)	0.0	17.2	0.0	0.0	43.4	5.2	60.6	38.1	70.1	13.6	86.3	88.7	96.0
Rock size (cm)	0.01	0.29	0.01	0.01	3.80	1.04	14.14	4.03	5.68	1.73	7.34	6.03	15.8
Wentworth rating: walk	1.00	1.44	1.00	1.00	2.57	1.57	2.17	1.99	2.63	1.58	3.15	3.21	3.90
Wentworth rating: transects	1.00	1.00	1.00	1.00	2.08	1.44	1.55	1.96	2.78	1.33	2.33	3.42	4.11
% silt: walk	100.0	58.6	100.0	100.0	24.2	65.6	28.3	60.8	24.5	77.3	13.7	3.1	4.0
% silt: transects	100.0	100.0	100.0	100.0	36.0	76.5	61.3	66.7	14.8	77.8	40.0	3.2	14.3
Pfankuch evalution score	124.5	137.0	126.0	132.0	90.0	107.0	122.0	115.0	83.0	136.5	72.5	74.0	70.0

Table A-3. Riparian zone habitat variables for each site.

Variable	BP2	BP1	BP3	CR2	CR3	CR1	DR1	DR2	DR3	ML3	ML1	ML2
Average riparian slope over 60 m: average of both banks	0.8	2.6	4.2	15.8	17.7	6.3	17.4	13.1	15.9	12.8	5.3	3.4
% solar radiation reaching stream bed in summer	28.3	6.5	36.5	25.5	31.0	35.0	36.7	3.5	25.8	35.3	45.8	15.7
% solar radiation reaching stream bed in winter	13.9	40.0	47.3	25.6	81.7	72.9	100.0	54.9	36.7	78.9	98.6	32.7
% solar radiation reaching stream bed all year	16.8	21.7	38.8	22.8	51.9	48.5	63.6	30.6	27.4	52.3	67.5	19.8
DBH of trees (cm) in 1m wide transect (average of both banks)	7.12	10.70	5.23	8.10	16.72	7.16	3.03	11.40	7.28	10.91	3.70	5.83
Number of trees in 1m wide transect (both banks)	45	21	38	53	31	42	61	27	26	22	28	50

continued...

Table A-3. Continued.

Variable	MM1	MM2	MM3	NL1	NL3	NL2	PP2	PP3	PP1	TR3	TR2	TR1
Average riparian slope over 60 m: average of both banks	4.8	8.3	5.5	8.7	11.3	5.0	11.8	16.3	14.6	11.5	4.8	2.8
% solar radiation reaching stream bed in summer	40.5	3.7	22.2	5.8	22.0	24.7	7.8	18.3	45.2	18.3	32.7	8.5
% solar radiation reaching stream bed in winter	100.0	2.1	50.4	22.7	92.6	100.0	44.9	71.1	36.1	37.4	100.0	56.4
% solar radiation reaching stream bed all year	65.8	3.8	35.4	15.9	55.2	58.9	27.4	40.6	36.5	26.9	63.0	34.0
DBH of trees (cm) in 1m wide transect (average of both banks)	4.37	5.23	14.41	6.79	5.23	6.58	5.53	8.64	4.79	6.50	7.22	4.20
Number of trees in 1m wide transect (both banks)	43	58	23	36	80	61	25	31	63	43	20	56

continued...

Table A-3. Continued.

Variable	WR1	WR2	WR3	WR4	WR5	WR6	WR7	WR8	MG	MR	NA	SP	WH
Average riparian slope over 60 m: average of both banks	2.3	2.7	5.5	7.1	6.0	14.1	4.1	0.0	21.3	11.5	12.0	12.4	7.7
% solar radiation reaching stream bed in summer	85.5	27.2	26.8	22.2	80.2	16.7	56.5	69.5	5.2	29.5	23.5	18.5	55.7
% solar radiation reaching stream bed in winter	100.0	100.0	100.0	100.0	100.0	12.6	98.7	100.0	2.1	100	49.6	13.4	100
% solar radiation reaching stream bed all year	88.3	60.6	61.1	59.2	85.2	14.9	74.0	80.8	4.9	62.0	36.8	16.1	74.3
DBH of trees (cm) in 1m wide transect (average of both banks)	0.84	3.71	5.75	4.70	1.57	7.30	10.02	2.54	8.45	3.73	5.44	8.50	5.17
Number of trees in 1m wide transect (both banks)	5	66	28	30	3	28	14	9	30	71	50	44	37

Table A-4. Mean watershed variables for each site.

Variable	BP2	BP1	BP3	CR2	CR3	CR1	DR1	DR2	DR3	ML3	ML1	ML2
Latitude	46.655	46.680	46.653	46.452	46.459	46.434	46.249	46.216	46.215	46.743	46.726	46.735
Longitude	64.346	64.351	64.330	62.272	62.269	62.267	63.442	63.411	63.434	64.186	64.245	64.232
Watershed size (km^2)	4.43	7.10	23.74	30.27	38.07	15.59	12.93	7.62	24.06	51.55	29.51	11.07
River length (km)	4.40	5.76	27.90	30.94	40.72	15.62	25.69	15.03	45.86	66.45	32.55	14.10
Road length/watershed area (km/km^2)	1.72	1.86	1.75	2.24	2.22	2.57	2.43	1.15	2.14	1.37	1.25	1.19
% forest	48.91	15.54	38.38	81.96	82.78	79.35	44.61	59.73	40.90	50.48	47.03	75.72
% wetland	0.30	0.35	0.71	2.82	3.18	3.08	0.42	0.00	0.50	7.16	8.94	6.54
% potato rotation	40.69	75.01	49.84	5.64	4.93	5.24	26.21	28.79	28.59	29.80	32.23	7.47
% other agriculture	3.61	0.62	2.64	1.13	0.89	1.57	16.69	6.74	15.43	0.67	0.71	0.00
% other (urban, industrial)	1.15	3.59	1.97	0.57	0.53	0.34	0.48	0.59	2.32	2.13	1.83	0.00

Table A-4. Continued.

Variable	MM1	MM2	MM3	NL1	NL3	NL2	PP2	PP3	PP1	TR3	TR2	TR1
Latitude	46.852	46.847	46.852	46.427	46.453	46.422	46.469	46.469	46.435	46.260	46.274	46.266
Longitude	64.176	64.187	64.196	62.161	62.113	62.137	62.212	62.199	62.218	63.531	63.566	63.579
Watershed size (km^2)	13.46	11.16	33.24	12.54	30.39	5.66	12.37	13.17	4.76	11.14	6.73	8.58
River length (km)	48.04	15.36	80.92	12.74	28.24	4.53	11.36	12.61	2.82	15.04	7.61	8.18
Road length/watershed area (km/km^2)	2.39	1.17	1.74	2.57	2.12	1.98	1.71	1.75	1.88	2.21	1.93	1.72
% forest	66.45	58.70	61.39	77.46	75.92	58.29	85.07	84.46	75.42	33.03	43.99	19.02
% wetland	2.49	2.70	2.59	4.43	8.40	24.66	2.97	2.79	5.30	0.65	1.59	3.39
% potato rotation	17.47	32.18	26.82	6.02	7.69	11.97	2.07	2.32	2.48	51.28	45.96	69.49
% other agriculture	0.00	2.04	1.84	1.90	0.78	0.00	0.00	0.00	0.00	1.77	0.00	1.19
% other (urban, industrial)	4.86	0.72	2.49	0.13	0.42	1.62	0.25	7.85	0.64	2.35	1.19	2.56

Table A-4. Continued.

Variable	WR1	WR2	WR3	WR4	WR5	WR6	WR7	WR8	MG	MR	NA	SP	WH
Latitude	46.408	46.406	46.403	46.403	46.400	46.388	46.394	46.394	46.149	46.880	46.451	46.406	46.37
Longitude	63.598	63.596	63.609	63.615	63.630	63.650	63.662	63.671	62.698	64.093	62.425	62.548	63.29
Watershed size (km^2)	5.55	15.89	22.43	7.21	35.24	7.17	48.05	49.45	11.47	5.47	34.83	29.45	31.61
River length (km)	8.30	18.19	28.30	8.09	45.97	7.94	60.00	61.28	13.19	9.12	54.22	37.16	45.63
Road length/watershed area (km/km^2)	2.70	2.11	2.23	1.96	2.21	2.02	2.07	2.02	1.82	2.49	2.05	1.73	1.81
% forest	9.18	8.82	8.83	12.53	9.23	9.52	9.29	9.91	44.77	38.44	74.56	52.05	18.35
% wetland	1.82	1.65	2.00	1.44	2.40	0.90	2.28	2.28	1.92	0.41	15.08	8.41	0.84
% potato rotation	67.89	70.26	70.03	76.02	70.77	76.47	72.92	73.30	33.96	48.62	8.51	21.61	58.96
% other agriculture	5.92	5.53	5.38	1.93	4.66	2.90	4.05	4.08	6.22	2.52	0.76	1.19	9.30
% other (urban, industrial)	3.99	8.43	6.96	1.63	5.35	2.78	4.63	4.50	2.63	4.21	1.32	3.65	3.49

APPENDIX B:

MEAN WATER CHEMISTRY, INSTREAM, RIPARIAN ZONE, AND WATERSHED VARIABLES FOR EACH RIVER EXAMINED

Table B-1. Mean water chemistry values for each river obtained during sampling periods or from data loggers.

Variable	BP	CR	DR	ML	MM	NL	PP	TR	WR	MG	MR	NA	SP	WH
Alkalinity (mg/L)	108.8	98.0	79.0	101.7	127.6	75.0	98.7	88.3	85.5	95.1	117.7	95.6	88.2	94.3
Conductivity (μmhos)	264.1	192.5	143.5	227.5	272.0	170.0	165.9	211.3	260.0	185.0	238.3	200.7	223.3	233.3
Dissolved oxygen (mg/L)	10.03	10.70	10.92	9.60	9.56	10.02	11.11	11.80	11.08	11.14	10.96	9.84	10.42	11.12
pH	7.93	8.29	8.09	8.1	8.07	8.10	8.22	8.13	8.39	8.33	8.12	8.56	8.16	8.25
Phosphorous (mg/L) x 10^2	1.747	4.333	5.325	9.658	1.289	6.125	6.025	9.492	4.683	3.600	26.07	1.767	4.067	5.433
Nitrate (mg/L)	5.36	0.39	1.78	2.49	0.54	0.32	0.10	4.88	7.04	2.33	3.13	0.20	1.20	3.67
Max temp (°C) (June-Septempber)	18.37	19.27	14.83	21.03	14.60	21.20	18.20	13.13	19.13	14.90	14.90	24.90	23.20	21.60
Mean June temp (°C)	11.3	12.5	9.6	14.3	15.0	11.7	10.9	10.2	12.1	10.8	10.1	18.1	15.9	12.4
Mean July temp (°C)	12.3	14.5	10.6	15.9	15.7	14.4	13.1	12.0	14.1	11.0	9.8	20.7	17.1	13.4
Mean August temp (°C)	13.1	13.7	11.0	16.3	15.9	15.4	13.6	12.5	14.8	11.5	10.0	19.2	17.6	14.2
Mean September temp (°C)	11.6	10.8	10.1	13.2	13.0	12.7	11.9	11.1	12.7	10.6	9.7	15.5	13.9	11.9

Table B-2. Mean instream habitat variables for each river.

Variable	BP	CR	DR	ML	MM	NL	PP	TR	WR	MG	MR	NA	SP	WH
Bank full depth (cm)	29.54	31.42	38.03	32.58	47.85	29.38	29.40	34.65	26.12	22.52	46.54	28.98	32.5	29.0
Mean width (cm)	483	603	557	584	621	467	485	425	567	385	225	650	700	668
Mean wetted width (cm)	291	486	482	486	439	402	362	352	477	407	300	631	726	680
Stream slope x 10^3	1.49	5.32	8.97	1.99	3.57	5.17	11.32	11.26	5.60	8.63	10.00	1.36	6.41	2.53
Velocity (m/s)	0.184	0.327	0.379	0.275	0.177	0.347	0.345	0.260	0.357	0.302	0.257	0.381	0.26	0.23
Discharge	0.082	0.295	0.144	0.188	0.044	0.207	0.148	0.068	0.301	0.109	0.045	0.409	0.35	0.30
Stability (%)	27.2	51.6	39.1	68.3	40.2	66.0	53.6	35.9	40.4	70.1	13.6	86.3	88.7	96.0
Rock size (cm)	4.08	3.84	3.91	4.91	9.34	3.90	10.84	8.86	6.08	5.68	1.73	7.34	6.03	15.8
Wentworth rating: walk	1.64	2.46	2.46	2.64	1.84	2.70	2.63	2.39	2.06	2.63	1.58	3.15	3.21	3.90
Wentworth rating: transects	1.29	2.02	1.88	2.30	1.62	2.07	2.37	1.76	1.54	2.78	1.33	2.33	3.42	4.11
% silt: walk	72.6	30.2	25.9	31.7	44.8	19.5	15.8	27.6	37.0	24.5	77.3	13.7	3.1	4.0
% silt: transects	78.7	45.1	47.3	45.7	60.4	33.1	27.0	56.9	65.8	14.8	77.8	40.0	3.2	14.3
Pfankuch evalution score	127.7	97.5	118.7	94.8	110.7	96.3	84.0	117.7	116.3	83.0	136.5	72.5	74.0	70.0

Table B-3. Mean riparian zone habitat variables for each river.

Variable	BP	CR	DR	ML	MM	NL	PP	TR	WR	MG	MR	NA	SP	WH
Average riparian slope over 60 m: average of both banks	2.5	13.2	15.5	7.2	6.2	8.3	14.0	6.3	4.2	21.3	11.5	12.0	12.4	7.7
% solar radiation reaching stream bed in summer	23.8	30.5	22.0	32.3	22.1	17.5	13.1	19.8	54.6	5.2	29.5	23.5	18.5	55.7
% solar radiation reaching stream bed in winter	33.7	60.0	63.9	70.0	50.9	71.8	58.0	64.6	99.6	2.1	100	49.6	13.4	100
% solar radiation reaching stream bed all year	25.7	41.1	40.5	46.5	35.0	43.4	34.0	41.3	73.3	4.9	62.0	36.8	16.1	74.3
DBH of trees (cm) in 1m wide transect (average of both banks)	7.68	10.66	7.24	6.81	8.00	6.20	7.09	5.97	5.10	8.45	3.73	5.44	8.50	5.17
Number of trees in 1m wide transect (both banks)	34.7	42.0	38.0	33.3	41.3	59.0	28.0	39.7	27.7	30	71	50	44	37

Table B-4. Mean watershed variables for each river.

Variable	BP	CR	DR	ML	MM	NL	PP	TR	WR	MG	MR	NA	SP	WH
Watershed size (km^2)	11.76	27.98	14.87	30.71	19.29	16.20	12.78	8.82	33.06	11.47	5.47	34.83	29.5	31.6
River length (km)	12.69	29.09	28.86	37.70	48.11	15.17	11.98	10.28	41.39	13.19	9.12	54.22	37.2	45.6
Road length/watershed area (km/km^2)	1.77	2.34	1.91	1.27	1.77	2.22	1.73	1.95	2.13	1.82	2.49	2.05	1.73	1.81
% forest	34.28	81.36	48.41	57.74	62.18	70.56	84.76	32.01	9.11	44.77	38.44	74.56	52.0	18.4
% wetland	0.45	3.02	0.31	7.55	2.60	12.50	2.88	1.87	2.11	1.92	0.41	15.08	8.41	0.84
% potato rotation	55.18	5.27	27.86	23.17	25.49	8.56	2.20	55.58	71.32	33.96	48.62	8.51	21.6	58.0
% other agriculture	2.29	1.20	12.95	0.46	1.2	0.89	0.00	0.99	4.75	6.22	2.52	0.76	1.19	9.30
% other (urban, industrial)	2.24	0.48	1.13	1.32	2.69	0.73	4.05	2.03	6.13	2.63	4.21	1.32	3.65	3.49