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EVALUATION AND APPLICATION OF POPULATION SIZE ESTIMATORS TO ASSESS BROOK TROUT (SALVELINUS FONTINALIS) RESPONSE TO A NON-NATIVE SALMONID REMOVAL IN A SMALL MICHIGAN COLDWATER STREAM

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BY

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ABSTRACT

EVALUATION AND APPLICATION OF POPULATION SIZE ESTIMATORS TO ASSESS BROOK TROUT (*SALVELINUS FONTINALIS*) RESPONSE TO A NON-NATIVE SALMONID REMOVAL IN A SMALL MICHIGAN COLDWATER STREAM

By

Joseph P. Gerbyshak

Non-native salmonids have been stocked into the Great Lakes since the 1870s and now naturalized populations use tributary environments to reproduce and for their juvenile life stage. Historically, brook trout were the only salmonid to inhabit the tributary environment and numerous studies suggest that exotic salmonids negatively affect brook trout by competing for limited resources. Other studies have been successful at removing non-native salmonids and the native populations increased. During this project 5,320 exotic salmonids were removed from a tributary of Lake Superior from 2008 to 2010 significantly reducing their density and young-of-year brook trout density increased by 260% the year after the study suggesting interspecific competition may be occurring. In order to monitor the salmonid populations closely, three techniques were used to assess population size of this small brook trout population. Mark-recapture estimates had large confidence intervals, so changes in population size could not be detected. Depletion estimates were hampered by sample size constraints and likely underestimated the population size. Relative abundance seemed to be the least likely to be biased because sampling was done on a frequent basis, which helped identify apparent changes in capture probability making it the best option to monitor changes in population size.

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LIST OF ABBREVIATIONS

BACI: Before-After-Control-Impact

BKT: Brook trout

COH: Coho salmon

CPUE: Catch-per-unit-effort

IACUC: Institutional Animal Care and Use Committee

MLE: Maximum Likelihood Estimator

NMU: Northern Michigan University

PIRO: Pictured Rocks National Lakeshore

STH: Steelhead trout

YOY: Young-of-year

CHAPTER 1: A BRIEF LITERATURE REVEW: BROOK TROUT LIFE HISTORY, NON-NATIVE SALMONID COMPETITION AND POPULATION SIZE ESTIMATION TECHNIQUES

Species Description:

The brook trout *Salvelinus fontinalis,* speckled trout, or brook charr as it is more frequently called in Canadian literature, is a member of the Salmonidae family, subfamily Salmoninae (salmon and trout), and belongs to the sub-group of fishes, the chars, genus *Salvelinus*. In addition to the brook trout, there are two species of char native to North America: the lake trout *S. namaycush* and Arctic char *S. alpinus* (Becker 1983; Szymanski 2009).

According to Power (1980), brook trout display considerable life history variation. Brook trout that have access to Lake Superior exhibit one of three life histories defined by their migratory behavior. *Fluvial* or resident brook trout remain in tributaries their entire lives and are not considered coasters because they do not enter Lake Superior. *Adfluvial* brook trout spend part of their lives in Lake Superior and part in a tributary environment. Streams are occupied by juveniles and later in life for spawning. *Lacustrine* brook trout spend their entire lives in the lake, never entering a stream. Brook trout that inhabit Lake Superior for an ecologically significant portion of their life (i.e. adfluvial and lacustrine) are locally called "coasters" (Becker 1983; Huckins et al. 2008). Coasters tend to have a longer life spans (Huckins et al. 2008) and grow to larger sizes than fluvial brook trout; the world record brook trout from Lake Nipigon was a coaster (Behnke et al. 2002).

Stream resident brook trout have a streamlined, moderately laterally depressed body while coasters are generally grow larger than stream resident brook trout (Huckins et al. 2008). Stream resident adult brook trout average 254-305 mm in length (Scott and Crossman 1973) while coasters in Lake Nipigon average 510-550 mm (Huckins et al. 2008). Color varies throughout the range of brook trout. The species can be differentiated from other salmonids by an olive-green to almost black back with lighter wormlike overmarkings (vermiculations), while the sides are lighter with small pale spots, some red surrounded by bluish halos, and white on the belly. The anal, pelvic and pectoral fins have a distinct, white leading edge trailed by a black stripe, followed by reddish coloration. The lower flanks and belly of males become orange-red with black pigmentation on either side of the belly during the breeding season (Scott and Crossman 1973; Becker 1983).

Distribution:

Brook trout are native to Canada and the United States and have been successfully introduced to many parts of the world (Scott and Crossman 1973). They are native to eastern Canada from Newfoundland to Manitoba, as far north as Hudson Bay and south to the Great Lakes. In the United States brook trout naturally occur from the New England states, west to the Great Lakes states and south to Georgia along the headwaters of the Appalachian Mountains (Scott and Crossman 1973: Power 1980). Brook trout are native to the lakes and tributaries of Lakes Superior, Huron, Michigan, and the tributaries of Lakes Erie and Ontario (MacCrimmon and Campbell 1969). Introduced brook trout inhabit western North America, South America, New Zealand, Asia, and parts of Europe (Scott and Crossman 1973).

Habitat requirements:

Brook trout inhabit cool, clear, well-oxygenated (minimum 5ppm) headwaters of spring ponds, springs and spring-fed streams. They have been successfully stocked in deep, stratified lakes where the lower stratum remains cool and well oxygenated. The preferred temperature of adult brook trout, excluding spawning, is between 13.9-15.6 °C; however, they can tolerate temperatures up to 25 °C. Brook trout tend to be found in waters with a pH range of 6.5 to 9, but can tolerate pH as low as 4.5 or as high as 10 (Scott and Crossman 1973: Becker 1983).

Coaster brook trout:

Historically coasters were found throughout Lake Superior (MacCrimmon and Gots 1980), including Pictured Rocks National Lakeshore; however, this evidence is largely anecdotal. Coasters spawned in at least 106 tributaries throughout Lake Superior (Newman and Dubois 1996). Currently coaster distributions have been reduced to a few isolated populations that persist in waters around Isle Royale, the Salmon Trout River, Lake Nipigon, the Nipigon River region, Northeast Lake Superior, (Huckins et al. 2008) and in streams around Pictured Rocks National Lakeshore (Kusnierz 2009). In the early 1900s the coaster brook trout fishery collapsed primarily due to over fishing (Hansen 1994). Loss of suitable spawning habitat related to logging (Horns et al. 2003) and interactions with non-native salmonids may have also played a role in the loss of coaster brook trout populations. Restrictive size and bag limits now protect the coaster brook trout in many areas from overfishing (i.e. minimum size limit is 20 inches and possession limit is one in Lake Superior). However, interaction with exotic salmonids may still be preventing the coaster brook trout from recovering.

Exotic salmonids:

Species composition in the Great Lakes has changed dramatically in the past century, impacting many species, including the coaster brook trout. There are two salmonines native to the Great Lakes; lake trout occupy the lacustrine environment and brook trout occupy the fluvial and/or lacustrine environments. Exotic salmonines were first introduced into the Great Lakes basin in 1870s. Many Pacific, Atlantic and Arctic salmonids were introduced but not all introductions were successful. Introduced Pacific salmon include: coho salmon *Oncorhynchus kisutch,* Chinook salmon, *O. tshawytscha,* pink salmon *O. gorbuscha,* rainbow trout *O. mykiss*, sockeye salmon *O. nerka*, cutthroat trout *O. clarkia,* masu salmon *O. masou.* Introduced Atlantic species include brown trout *Salmo trutta*, and Atlantic salmon *S. salar*. The introduced Arctic salmon was formerly known as the Arctic charr *Salvelinus alpinus* (Crawford 2001). Many salmonines currently inhabiting Lake Superior could compete with coaster brook trout. Coho salmon, the lake migratory form of rainbow trout, known as steelhead trout, and brown trout are of primary interest because they are most commonly sympatric with coaster brook trout. On the south shore of Lake Superior in Pictured Rocks National Lakeshore typically only coho salmon and steelhead trout inhabit the fluvial environment and possibly compete with coaster brook trout.

Steelhead trout:

Steelhead may be competing with other salmonids for limited resources such as food. Steelhead reside in tributaries of Lake Superior as juveniles from the time they hatch in early summer until they smolt and migrate to the lake two years later (Becker 1983). Competitive effects of steelhead on brook trout could be amplified because there are typically many more juvenile steelhead than juvenile brook trout in tributaries on the south shore of Lake Superior (Huckins et al. 2008). Ensign (1991) reported that in environments where brook trout and rainbow trout coincide they consume similar foods and this was confirmed by Howard (2013) in our systems. Isely and Kempton (2000) studied the effect of stocking juvenile brook trout and juvenile rainbow trout together in raceways with food in excess. They found that brook trout were significantly larger than rainbow trout in length and weight when they were stocked alone; however when these species were stocked together, rainbow trout were significantly larger in length and weight (Isely and Kempton 2000). Further evidence of competition for food in a Lake Superior tributary was reported by Rose (1986) who studied the interaction between juvenile rainbow trout and juvenile brook trout. He concluded that growth was reduced in juvenile brook trout following the emergence of rainbow trout fry, suggesting that the two species compete for food. He also pointed out that decreased size from the lower growth rate can lead to lower winter survival rates.

Competition for stream position may also exist between steelhead and brook trout which could affect growth and survival. In some eastern North American streams, brook trout have been replaced in the downstream regions by introduced rainbow trout (Larson and Moore 1985). Gibson (1981) reported, based on experimental observation, that steelhead trout were more aggressive and could displace brook trout from preferred stream areas. Further evidence that rainbow trout are superior competitors for stream position was provided by Larson and Moore (1985) who showed that rainbow trout were better able to occupy shallow riffles and pools than brook trout. The shift in position by brook trout, presumably away from successful feeding stations, may increase the growth rate of rainbow trout (Krueger and May 1991). This could be an explanation for why rainbow trout gain an advantage in growth over brook trout in the spring of the second year of life (Whitworth and Strange 1983). The size advantage that rainbow trout gain should further enhance the ability of this species to occupy preferred position in streams (Krueger and May 1991).

Coho salmon:

Coho salmon are another exotic salmonid that could compete with brook trout for limited resources, such as food, in Lake Superior tributaries. Coho salmon remain in tributaries of Lake Superior from emergence in early spring until they migrate to the lake in the first fall. Stauffer (1977) studied three tributaries of Lake Superior and suggested juvenile coho salmon caused a reduction in brook trout populations. Coho salmon emerge earlier, are larger at emergence, and are larger throughout the first summer than brook trout, which may give them a competitive advantage for food (Fausch and White 1986). According to Gibson (1981), dominance of salmonids in the laboratory setting was based largely on size and the dominant species showed the best growth. Fausch and White (1986) stated that coho salmon are larger than brook trout throughout the first summer; following Gibson’s (1981) findings, coho salmon should thus be able to dominate brook trout giving them the competitive advantage for food and consequently better growth.

Fausch and White (1986) found that coho were superior competitors for stream position in a laboratory setting. Coho grew faster than brook trout in sympatry because they were able to outcompete brook trout for the most energetically favorable positions; thus, brook trout grew slower and became subordinate to the coho. Gibson (1981) also found that coho dominated brook trout in the laboratory setting. Gibson concluded that larger fish are usually dominant over smaller fish and since coho emerge earlier than brook trout they are dominant in competition for superior stream positions, yielding them a greater growth rate. These results suggest that larger size and competitive superiority of coho salmon should give them an advantage over juvenile brook trout in Great Lakes tributaries when resources become limiting.

Competition during spawning between coho salmon and brook trout may also negatively affect brook trout populations. Coho salmon and brook trout are both fall spawners, but their spawning times may differ slightly (Becker 1983). Salmonids that spawn in streams modify their habitat by digging redds in the stream bottom for the incubation of their eggs. Redd construction by prespawn coho significantly reduced invertebrate populations in a tributary of Lake Michigan and as a result, food supplies for native fishes could become limited through the spawning activities of introduced salmonids (Hildebrand 1971). Coho salmon can also disturb or destroy redds of brook trout making the redds inadequate to incubate the eggs. Coho, through modification of spawning habitats, could physically destroy eggs of native brook trout which are deposited at about the same time in the fall (Krueger and May 1991).

Exotic Salmonid Removals:

Exotic salmonid removals have been successful in other parts of the country in restoring native brook trout populations (Moore et al. 1983; Kulp and Moore 2000). Research in the Appalachian Mountains has shown that over a four year period, intensive electrofishing can reduce exotic salmonid populations. Consequently the standing crop of brook trout increased after the reduction (Moore et al. 1983). In addition rainbow trout were eradicated from a small stream in the Appalachian Mountains over a two year span (Kulp and Moore 2000).

Population Estimation:

In fisheries management, population abundance estimates are key for evaluating the status of a fishery. Reliable absolute or relative abundance estimates are essential in monitoring fluctuating populations and making informed management decisions (Rogers et al. 2003). A variety of approaches exist to monitor population size in stream environments. Three commonly used techniques are mark-recapture, depletion method, and relative abundance. It is important to know if these methods accurately monitor population size and if actions can be taken to improve population size estimation.

Mark-recapture:

Mark-recapture methods have long been used by fisheries researchers to estimate populations. A sample of fish is captured, marked and released back in the population. A second sample is taken, and the ratio of marked to unmarked fish can be used to estimate the total population (Rogers et al. 2003). There are numerous types of mark-recapture models, some used to estimate open populations and some used to estimate closed populations. A closed population remains unchanged during the period of investigation; the effects of migration, mortality and recruitment are negligible. An open population can change due to migration, mortality and/or recruitment. Various types of models are used to estimate closed populations. Some are used when there is only a single mark-recapture period such as the Peterson estimator or Chapman estimator, and other types of models are used when there are multiple mark-recapture periods such as the Schnabel estimator (Seber 1982; Rogers et al. 2003). Open populations are estimated with a broad family of models referred to as Cormack-Jolly-Seber survival models (Rogers et al. 2003).

The accuracy of the estimates generated from mark-recapture models can vary for numerous reasons. Compared with closed population models, additional parameters describing losses and additions to the population are necessary for open population models. The additional parameters necessary to describe open populations often lead to a decline in the precision of population estimates (Rogers et al. 2003). Population estimates generated from open mark-recapture models can vary significantly if any of the assumptions are violated. The assumptions are (Seber 1982):

a. Every animal in the population, whether marked or unmarked, has the same probability of being caught in the *i*th sample, given that it is alive and in the population when the sample is taken.

b. Every marked animal has the same probability of surviving from the *i*th sample to the (*i*+1)th sample and of being in the population at the time of the *i*th sample, given that it is alive and in the population immediately after the *i*th release.

c. Every animal caught in the *i*th sample has the same probability of being returned to the population.

d. Marked animals do not lose their marks and all marks are reported on recovery.

e. All samples are instantaneous, i.e. sampling time is negligible.

Open population models require stronger adherence to the model's assumptions, especially heterogeneity in capture probability, because violations can lead to bias in population estimates (Rogers et al. 2003). Capture probabilities can vary with (1) time (2) behavioral response (3) individual animal or any combination of these three (Otis et al., 1978). Capture probabilities may vary over time when electrofishing lotic systems due to many factors (i.e. weather, turbidity, crew members, conductivity, voltage, time of day, flow, number of fish, habitat complexity, depth or sampling regime). Changes in capture probability caused by environmental influences can be minimized by sampling under typical environmental conditions (Ney 1999). A behavioral response to initial capture could make fish more or less likely to be captured. Capture probability could decline if a fish develops an acute sensitivity to electricity and avoids it. Capture probabilities can vary by individual animal. Fish of different sizes may be caught with varying efficiency, sometimes as a result of selectivity of gear (Ricker 1975). Smaller fish are more difficult to see and have less surface area, which makes them less vulnerable to capture by electrofishing. Different species of fish may also be less vulnerable to capture by electrofishing (Reynolds 1996).

The topic of heterogeneity in capture probability is widely discussed in the literature (Carothers 1973; Otis et al. 1978; Pollock 1990; Rodgers 1992; Pine et al. 2003). If marked individuals are more likely to be caught than others (e.g. larger sized fish) and if the unequal capture probability persists through the experiment then the true proportion of marked individuals will be overestimated leading to a negative bias (Pollock et al. 1990). Rodgers et al. (1992) conducted a mark-recapture study on a known population and the mark-recapture estimate was 15% below the true population size.

The assumption that all marked animals have the same probability of surviving can be violated, which can lead to a positive bias in abundance estimates. Survival rates of PIT tagged fish sometimes differ. For example, small cutthroat trout (<200mm) had a lower survival rate than larger fish (>200mm) in a small Alaskan lake (Harding 1998). However, in many experiments overall survival remains high. In hatchery conditions PIT tagged juvenile Atlantic salmon had a 94% survival rate (Gries and Letcher 2002), while in a laboratory study PIT tagged juvenile steelhead had an 86% survival rate (Bateman and Gresswell 2006). A violation of the assumption of equal probability of survival can be a source of variation, but is difficult to address in an open population because it is nearly impossible to distinguish in an open system if an individual has died or emigrated.

Abundance estimates will be positively biased if marks are lost, and PIT tag retention can vary. For example, PIT tag retention was 99.8% for juvenile Atlantic salmon for nine months under hatchery conditions (Gries and Letcher 2002). However, Bateman 2009 reported lower tag retention rates ranging from 62% to 80%. Tag retention was much greater in fish that had a fork length that was less than 122mm. Tag retention was lower for larger fish because mature individuals ejected tags during spawning.

Depletion Method:

Another method to estimate population size is by the removal or depletion method. Like the mark-recapture method, it relies on multiple samples of the population. During each sampling period, fish are temporarily removed from the population. The subsequent catch declines with each sampling and the rate of decline provides the data needed to estimate the original population (Rogers et al. 2003). The depletion method model estimates can vary significantly if any of the assumptions are violated. According to Seber (1982) the assumptions are:

a. The population is in equilibrium (i.e. birth, recruitment, and immigration rates are balanced by death and emigration rates).

b. The probability of capture (p) remains constant from sample to sample.

c. The probability of capture in the *i*th sample is the same for each individual exposed to capture.

The first assumption of closure of the population (i.e. no recruitment, natural mortality, or migration) is generally not violated. The standard protocol calls for block nets to be erected on each end of the sampling reach, closing off the sampling area to migrations of any kind. The sampling period is typically short enough (i.e. less than one day) that recruitment or natural mortality are negligible.

The assumption that the vulnerability to capture is constant over time is likely violated in most experiments. Unequal capture probabilities caused by time are any variables that change over time that affect capture probability (e.g. weather, turbidity, crew skill, conductivity, time of day, water velocity, depth, or sampling regime). Heterogeneity of capture probability caused by time variation is reduced or virtually eliminated if successive removals are done in a timely manner (Otis et al. 1978). When experiments are done over the course of a day, capture probabilities caused by time variation remain fairly constant and this assumption is not likely violated. However, when comparing population estimates over longer periods of time, variables that affect capture probability can change, biasing the population estimates.

The final assumption that all members of the target population are equally vulnerable to capture is usually violated. The main problem with removal estimators is dealing with the heterogeneity of capture probability (Otis et al. 1978), which may occur for various reasons, including inherent features of each fish such as size and species (Rogers et al. 2003). In general, sampling efficiency decreases with each successive pass because fish that remain after the initial pass may be less catchable (Rosenberger and Dunham 2005). This may be due to larger fish being easier to catch than smaller fish (Reynolds 1996) or because of physiological or behavioral response to the previous electrofishing pass (Mesa and Schreck 1989). As a result of decreased sampling efficiency, the depletion method usually underestimates population size (Riley and Fausch, 1992). Rodgers et al. (1992) performed a depletion estimate on a known population and estimated 67% of the total actual abundance. Riley and Fausch (1992) suggested that removal estimates underestimated the true population size at least 50% of the time. Peterson et al. (2004) estimated that the removal method underestimated abundance by 88%.

Relative Abundance:

Catch per unit effort (CPUE) is an index to relative abundance and is based on the general assumption that the size of a sample caught from a population is proportional to the effort put into collecting the sample. This means that one unit of sampling effort is assumed to catch a fixed proportion of the population and thus a decline in population size will produce a decline in CPUE (Seber 1982). Obtaining CPUE is less labor intensive than methods for estimating absolute abundance, yet can still provide an accurate measurement of population change over time, as long as the vulnerability to the gear remains constant. Catch per unit effort is commonly used to monitor or assess stocks when the boundaries of the populations are unknown, as in streams. Most commonly, CPUE data over time is used to assess the effect of fisheries management actions (Rogers et al. 2003). The assumptions according to Seber (1982) are:

a. The population is in equilibrium (i.e. birth, recruitment, and immigration rates are balanced by death and emigration rates).

b. Units of effort operate independently (one unit of fishing gear does not interfere with other units).

c. Catchability, q, is constant throughout the sampling period.

d. Every individual in the stock has the same probability of capture. This assumption concerns the spatial distribution of fish and is met when fish are uniformly distributed within the boundaries of the stock (Rogers et al. 2003).

Violations of any of these assumptions can seriously compromise the ability of CPUE to serve as an accurate predictor of changes in abundance (Rogers et al. 2003). The first two assumptions are generally not violated. The first assumption that the population is in equilibrium holds true because the sampling period is short enough (i.e. less than a day) that births, recruitment, immigration, death or emigration are negligible. The second assumption that units of effort operate independently is not violated because only one type of sampling gear at a time is being used so there cannot be any interference.

Violation of the assumption of equal catchability throughout the sampling period can bias CPUE indices. Catchability is the probability of catching an individual fish in one unit of effort (Ney 1999) and can vary with size, sex, or other intrinsic characteristics of fish (Reynolds 1996). Seber (1982) suggested if the length structure of a population varies, catchability may best be estimated separately for individual length-classes in the sample. Catchability can also change due to environmental factors such as time of day, season, sampling site, water temperature, dissolved oxygen levels or other environmental features that affect the ability to capture fish (Rogers et al. 2003). The effect of environmental factors can be difficult to address because weather can be unpredictable. However, environmental influences can be minimized by sampling under typical environmental conditions (Ney 1999).

The fourth assumption is generally violated when estimating the relative abundance of a population (Paloheimo and Dickie 1964 as cited by Rogers et al. 2003). This assumption specifies that fish are uniformly distributed in space and that all occupied areas are accessible to the gear and are randomly sampled; however, neither fishing effort nor fish are typically uniformly distributed. Variation in catchability arises when changes occur in the spatial distribution of fish even when effort is uniform. One of the main problems with CPUE is that it is difficult to distinguish between a change in abundance and a change in distribution (Paloheimo and Dickie 1964 cited by Rogers et al. 2003). The bias of unequal distribution of fish can be minimized if sampling effort is high and random sampling occurs.

Catch per unit effort has been used in many situations as an index of changes in total abundance. Hall (1986) showed a high correlation between absolute abundance and CPUE of a largemouth bass population in an Ohio impoundment. Tsuboi and Endou (2008) found a linear relationship between the CPUE and the abundance of white-spotted char (*Salvelinus leucomaenis)* in a mountain stream. Catch per unit effort has been used to monitor the restoration efforts of lake trout in Lake Superior (Hansen et al. 1994).

The purpose of this study was to investigate the interactions of exotic salmonids and brook trout along with developing recommendations for monitoring population size of small populations. In order to accomplish this, non-native salmonids were physically removed from a tributary of Lake Superior to determine if it was feasible and to investigate whether brook trout density or growth responded to the treatment.

CHAPTRER 2: COMPARISON OF THREE TECHNIQUES USED TO ASSESS BROOK TROUT (*SALVELINUS FONTINALIS*) POPULATION SIZE IN A SMALL MICHIGAN STREAM

CHAPTER SUMMARY:

The population size of a relatively small population of brook trout was evaluated over three years via mark-recapture, the depletion method, and relative abundance (catch per unit effort). This project took place to assess the ramifications of using these commonly applied methods to estimate fish abundance and to develop recommendations for future studies on small populations. Over the course of the study, population estimates differed depending on the method used: mark-recapture estimates ranged from 407 to 542 per year, removal method estimates ranged from 118 to 341 per year and CPUE ranged from 0.015 to 0.073 fish per meter. Mark-recapture estimates had large confidence intervals due to low capture probabilities. Mean capture probability in this study was 0.20 (SE=0.05). I recommend to increase capture probability higher than was observed in this study, and to conduct the sampling in a two day timeframe to increase the number of recaptures and precision of the resulting estimate. When using the depletion estimator or relative abundance estimates, sampling should be conducted at the same time of year due to seasonal variability in capture probability. Also, when monitoring population size by the depletion method or relative abundance, calculate at least two estimates to minimize the chances of differential capture probabilities from environmental factors which could lead to biased estimates. Based on the goals of this multipurpose study, CPUE provided reliable and useful information on changing population size while the others approaches were less valuable for my application.

INTRODUCTION:

In fisheries management, population abundance estimates are key for evaluating the status of a fishery. Information gained from absolute or relative abundance estimates are essential in monitoring fluctuating populations and making informed management decisions (Rogers et al. 2003). However, small populations can make it challenging to generate reliable abundance estimates. Absolute and relative abundance estimation methods have strict assumptions and various limitations (Seber 1982) and small populations make those assumptions and limitations challenging to meet.

Little has been written regarding the difficulties of estimating the size of small populations (McKelvey and Pearson 2001), but some of these populations are the most important to study, including declining populations or endangered species (Chao 1989; Lynam et al. 2009). Our study was focused on a small population (N=~300-400) of brook trout in 2.7 km of northern coldwater stream that is of conservation concern. This population is one of the few remaining populations of brook trout on the south shore of Lake Superior known to exhibit a migratory behavior (Huckins et al. 2008), so it is important to monitor population size to ensure it is at sustainable levels.

The challenge was to accurately estimate and monitor population size while minimizing adverse effects to this sensitive population. Numerous studies suggest that repeated electrofishing events in a short time frame can have negative effects on the population. Gatz et al. (1986) concluded that seven electrofishing events per year on the same population lowered the average growth rate of juvenile salmonids. Other studies have shown mean injury rates to juvenile rainbow trout to be 5.1% (McMichael et al.1999). Yet, other studies have concluded internal injury rates to fish can be as high as

50% when examined internally (Snyder 2003). Due to the potential harmful effects of electrofishing, the method should not be used more often than needed to collect the necessary data required to make management decisions, especially when working with small populations of conservation concern.

One method that has long been used by fisheries researchers to estimate populations is mark-recapture estimation (Ricker 1975). This method entails collecting a sample of fish, permanently marking them, and releasing them back in the population. A second sample is taken, and the ratio of marked to unmarked fish can be used to estimate the total population (Rogers et al. 2003). Mark-recapture studies that last longer than a few days, as in this study, are considered “open” because the population is subject to immigration/emigration and births/deaths (Seber 1982). The primary models used in fisheries applications to estimate population size in open populations are the Jolly-Seber (Jolly 1965; Seber 1965) and related models (Pollock et al. 1990). Due to the complexity of the calculations, software programs have been developed such as Program MARK (White and Burnham 1999) to analyze mark-recapture data from open populations.

Mark-recapture can perform relatively well with small populations, but small populations may make mark-recapture estimates challenging because subtle violations of the assumptions (Table 2.1) become more extreme. For example, if one tag is lost or overlooked in a large population with a high proportion of fish marked, then the impact on the population estimate will be relatively small. However, if one tag is lost or overlooked in a small population with the same proportion of fish marked, the population estimate maybe impacted to a much greater degree. In other words, in a small population a lost or overlooked tag will result in a greater positive bias to the estimate than in a large population. The same is true if the assumption of equal probability of survival between marked and unmarked fish is violated; if one fish dies due to tagging, it will result in a greater bias in a small population.

In mark-recapture experiments, capture probability should be maximized because it drives the accuracy and precision of abundance estimates (Pine et al. 2003). However, in many fisheries studies capture probabilities are low resulting in “sparse” data (Bayley and Austin 2002). A high capture probability is even more imperative in small populations because it is challenging to capture individuals because they are rare already. Capture probability also needs to remain high during the recapture period because, if few marked individuals are recaptured, abundance estimates will have large confidence intervals and have low reliability.

Another way to estimate populations is by the removal or depletion estimation method. Like the mark-recapture method, it relies on multiple samples of the population. During successive sampling periods, the fish are temporarily removed from the population. Therefore, the catch declines with subsequent sampling and the rate at which it declines gives a measure of the proportion of the original population that has been removed (Rogers et al., 2003). The maximum-likelihood estimator (MLE) described by Junge and Libosvarsky (1965) is a common method used to analyze depletion data. Program MARK (White and Burnham 1999) is also capable of handling removal data by using the closed captures model and setting the recapture parameter to zero.

Removal estimates rely solely on the number of fish caught and the reduction in the catch per unit effort (Peterson and Cederholm 1984). The ability of the estimator to perform without bias rests on the assumption of equal capture probability. It is well documented in the literature that there is a negative bias in depletion population estimates due to a decline in capture probability with each successive sampling (Peterson and Cederholm 1984; Riley and Fausch 1992; Rodgers et al. 1992; Rosenberger and Dunham 2005). Capture probability can vary for many reasons, but it has a tendency to fluctuate to a greater degree in small populations. For example, in a small population fewer fish will be captured than in a large population with the same capture probability. Every fish that is captured or escapes in a small population has a mathematically larger impact on the capture probability and thus the population estimate.

Additionally, the assumption of equal capture probability among samples is often difficult to adhere to because of changing environmental conditions (Peterson et al. 2004; Rosenberger and Dunham 2005), making estimates non-comparable. For example, capture probability may be reduced after a large rain event because flow and turbidity may increase making fish difficult to see and leaving little time for capture before the fish are swept away. Sampling under conditions of reduced capture probability will result in a negatively biased population estimate and the population estimate will not be comparable to one obtained under ideal conditions.

Small populations can complicate the depletion method’s ability to generate population estimates at all due to sample size constraints. The depletion estimator fails if the number of fish caught in the last pass is greater than the number of fish caught in the first pass (Seber 1982). This is more likely to happen in a small population, when only a few rare fish are caught in the first pass. Furthermore, the cumulative removal needs to be greater than 30 for the MLE to estimate variance of the population estimate (Seber

1982). Without the variance, confidence intervals cannot be generated and the precision of the estimate is unknown, which makes it difficult to make management decisions regarding the population due to the uncertainty surrounding the estimate.

Catch per unit effort (CPUE) is an index of abundance used to estimate relative abundance of fishes. It is based on the general assumption that the size of a sample caught from a population is proportional to the sampling effort. It is less labor intensive than the absolute estimator approach, but can still provide an accurate depiction of the population over time as long as the vulnerability to the gear remains constant (Seber 1982). If the goal is to strictly monitor a change over time and an absolute population estimate is not needed, then CPUE may be useful (Pine et al. 2003).

The ability of CPUE to serve as an index of abundance is not affected by population size, but it is impacted by underlying assumptions (Table 2.1), including equal capture probability (Hubert and Fabrizio 2007). It is unlikely capture probability will remain constant over multiple passes and under varying sampling conditions; therefore, it can be difficult to separate changes in capture probability from changes in population size. Thus CPUE only demonstrates trends in catches, which may or may not be related to population abundance (Williams et al. 2002). However, sampling under ideal sampling conditions with a high amount of effort will decrease the chances of fluctuating capture probability (Ney 1999).

The objective of this project was to evaluate the effectiveness of using these commonly applied methods when evaluating the size of a relatively small population of brook trout and develop recommendations based on the results to improve future studies on small fish populations of conservation concern.

METHODS:

Study Site:

Pictured Rocks National Lakeshore (PIRO) is located in the northeastern portion of the Upper Peninsula of Michigan (Figure 2.1). Mosquito River, a tributary of Lake Superior, is located on the western side of PIRO (Figure 2.2). It is a third order stream with a four meter high waterfall 2.7 km upstream from Lake Superior that is a barrier for upstream passage. The substrate is mainly cobble with stretches of sand and bedrock.

Sampling:

Sampling occurred monthly, from May through November, from 2008 through 2010 on the Mosquito River. The research area consisted of 2.7 km of stream from Lake Superior to the barrier waterfalls. The research area was stratified into areas of similar habitat by gradient. There were three sections each comprised of similar habitat (lower, middle and upper); each consisted of approximately one third of the research area. Each section was further divided into six reaches of approximately 150 m in length. All sampling was performed with an electrofishing crew consisting of two individuals working upstream. ETS Electrofishing Systems LLC (Madison, WI) backpack electrofishing units were used; the exact voltage setting depended on the conditions of the stream at the time of sampling, but was set near 300 volts, 40% duty cycle, and a rate of 60 pps.

There were three different methods of sampling, each with a different purpose. *General sampling* was a stratified random sampling technique that occurred in the months of June, July, and October. All sites were randomly selected in 2008 and then kept constant in subsequent years of the study. General sampling consisted of sampling two reaches in each section of the research area (N=6 reaches) in order to obtain a representative sample throughout the research area. Each reach was sampled by a single electrofishing pass. All salmonids were collected and processed. *Sweeps* were conducted in May, August and November. During a sweep, the entire research area of the stream was sampled (N=18 reaches). This was accomplished by three, two-person, electrofishing crews, each electrofishing one third of the stream (N=6 reaches). Each electrofishing crew did a single electrofishing pass and all salmonids were collected and processed. *Three-pass depletion sampling* was performed in September of each year to obtain a population estimate. Block nets were erected at the beginning and end of each reach to act as a barrier for fish movement. In 2008, three-pass depletion sampling was conducted on one reach in each section (N=3 reaches). In 2009-2010, two reaches in each section (N=6 reaches) were sampled to obtain a more precise population estimate for the whole research area

Processing Fish:

After collection, each fish was identified, weighed (g) and measured for total length (mm). All brook trout over 100mm were scanned with a portable Texas Instruments half-duplex radio frequency identifier to check for the presence of a previously implanted passive integrated transponder (PIT tag). If a brook trout greater than 100mm had not been previously tagged, it was implanted with a 23mm PIT tag and returned to the stream. The tagging procedure was approved via the Northern Michigan University Institutional Animal Care and Use Committee (IACUC #66 and #152).

Mark-Recapture:

The POPAN formulation of the Jolly-Seber model was used in Program MARK (White and Burnham 1999) to analyze the mark-recapture data. This model was selected because it yields an abundance estimate from an open population. Mark-recapture data were used from adult (estimated one year old and greater) brook trout from May, August, and November of each year because an assumption of the POPAN model is that the sampling area must remain constant and during these sampling occasions the whole study area was sampled. Based on this dataset, one abundance estimate was obtained for August of each year for the entire study area because initial and the final abundance estimates for each year could not be cleanly estimated due to non-identifiable parameters (White and Burnham 1999). Static and time variant models, where survival, capture probability and probability of entrance were allowed to vary, were built in Program MARK to account for parameter variation over time. Models that were built were:{N, p(t), phi(t), pent(t)}, {N, p(t), phi (.), pent(t)},{N, p(.), phi (t), pent(t)}, {N, p(.), phi(.) pent(t)}, where N=abundance, p=capture probability, phi=survival, pent=probability of entrance, (.) static, and (t) time variant. Model averaging was used to address any uncertainties in model selection.

Depletion Method:

Population estimates for adult brook trout were obtained by two different approaches from the three-pass depletion data collected in September 2008-2010. The first approach was the maximum-likelihood estimator (MLE) described by Junge and Libosvarsky (1965) cited in Seber (1982):

$$N = \frac{6X^2 - 3XY - Y^2 + Y\sqrt{Y^2 + 6XY - 3X^2}}{18(X - Y)}$$

where N=estimated population size, n_i=the number removed at a given pass, $X=2n_1+n_2$ and $Y=n_1+n_2+n_3$. When the cumulative removal is relatively large (>30), the asymptotic variance of N can be calculated, and confidence intervals can be obtained (Seber, 1982). However, for this study the cumulative removal of adult brook trout never exceeded 30 individual so the variance for the estimates could not be calculated.

The second approach to estimate abundance with the three-pass depletion method used Program MARK. A closed captures model was used with the recapture parameter (c) fixed to 0 because fish that were removed could not be recaptured. A model that allows for no temporal variation ({N, p(.), c(.)}) was built for each of the sites where depletion sampling occurred. Confidence intervals were generated for each site using the profile likelihood approach which yields asymmetrical confidence intervals.

Population estimates were obtained at the reach level using both approaches. They were extrapolated to the whole research area as in Bohlin et al. (1989). In 2008, when one reach site per section was sampled, the mean population estimate was calculated for the whole research area by

$$\mu_r = \sum^{n} \frac{y_i}{n}$$

where n = number of reaches sampled in the research area and y_i = estimated population size in reach i, and $\mu_r =$ mean population estimate for a reach in the whole research area. The total population for the whole research area was then estimated by

$$Y_{tot} = N\mu_r$$

where N = number of reaches in the whole research area, Y_{tot} = total estimated population size in the whole research area. In 2009 and 2010, when two sites in each section were sampled, the total estimated population for the stream was calculated

$$\mu_s = \sum^{n} \frac{y_i}{n}$$

where μ_s= mean estimated population size for a section, and n = number of reaches in that section. The population for each section was then estimated by

$$Y_s = n\mu_s$$

where Y_s = total estimated population for a section. The population for the whole research area was then calculated as

$$Y_{tot} = \sum^{s} Y_s$$

where is s = number of sections.

A Mann-Whitney Rank Sum Test was run on the 2009 and 2010 data to determine if there was a significant difference in the population estimate when three sites or six sites were incorporated using SigmaPlot Version 11.0.

Relative abundance:

Relative abundance or catch per unit effort (CPUE) was used to estimate relative abundance for each month of sampling over a three year time period. CPUE was calculated by

$$\frac{number\ of\ fish\ caught\ in\ one\ pass}{distance\ electrofished\ (m)} = CPUE$$

First, relative abundance data (fish/m) was displayed by month and year for all reaches sampled each month from 2008 to 2010. It was also displayed by individual reach and season (May, August and November) from 2008 to 2010. A repeated measures analysis of variance was run on the CPUE data for each year by sampling month to determine if there was a significant difference in the variation of CPUE among years. A Friedman repeated measures analysis of variance on ranks was run on the CPUE to determine if there was a significant difference in season (May, August, November) during 2008 to 2010 (Figure 2.5).

RESULTS*:*

Absolute abundance estimates obtained via mark-recapture averaged 50% higher than abundance estimates obtained through the depletion method. Model-averaged abundance estimates varied, {N, p(.), phi(t), pent(t)} ranked the highest in 2008 and 2010 and {N, p(t), phi(.), pent(t)}ranked the highest in 2009 (Table 2.2). Mark-recapture abundance estimates calculated with data from August varied throughout the study from 407±354 to 542±379 with a 12% annual variation across years (Table 2.3).

September population estimates generated from depletion data via the MLE were 20% higher than results obtained from Program MARK. Depletion population estimates, depending on the calculation method, ranged from 235 to 314 in 2008, 260 to 291 in 2009 and 118 to 128 in 2010 (Figure 2.3). Confidence intervals could not be generated for these estimates so the level of precision is unknown. The greater than average precipitation received in September of 2010 (Table 2.4) may have influenced the depletion method population estimates that year. Table 2.5 shows depletion method population estimates for each site. A population estimate could not be generated for site two in 2010 because the catch did not decline. In 2009 and 2010, six sites and three sites were used to calculate stream wide population estimates. Six sites were used to obtain more precise estimates, but no difference ($p=0.99$) (Figure 2.4) was found with the additional sampling.

Absolute and relative population estimates did not show the same trend over the course of the study. The highest, middle or lowest values never occurred on the same year for each respective estimator. Catch per unit effort by month was highly variable within year, but the pattern was very similar among years (Figure 2.5). Relative abundance was low in May, rose and leveled off in the summer months, and declined in November. This seasonal (May, August, and September) fluctuation in relative abundance was significant ($F=23.470$, $df=2$, $p<0.001$) (Figure 2.6). The mean CPUE was approximately 30% higher in 2009 than in 2008 or 2010 (Figure 2.5). Relative abundance estimates, when calculated by month, did not differ among years from 2008 to 2010 ($F=1.713$; $df=2$; $p=0.222$) (Figure 2.5).

DISCUSSION:

Small populations that are of conservation concern are some of the most important to study; however, monitoring population size of small populations can be challenging. It is important to accurately monitor population size to understand how management actions affect small populations and to ensure population size remains at a sustainable level. An additional challenge is to minimize the adverse effects on these sensitive populations while obtaining an accurate estimate of population size.

Both the temporal and spatial openness of the study area likely affected the mark-recapture results, but probably did not considerably affect the depletion and relative abundance estimates. The depletion and relative abundance estimates were completed within a day, so the population was only open to migration or death for a short time period. Block nets were used when conducting the depletion estimates, so the study area was spatially closed. The study area for relative abundance estimates was not closed spatially, but few fish were seen leaving the research area while sampling. Additionally, if fish escaped upstream during sampling, there was ample amount of holding cover for them to remain in until capture. The large temporal openness (i.e. seven months) associated with the mark-recapture estimates likely lead to lower capture probabilities, which reduced the number of marked and recaptured fish and resulted in large confidence intervals. The mark-recapture estimates were generated from data collected over this timeframe because an assumption of the POPAN model is that the sampling area must remain constant (i.e. each sampling event must have the same spatial coverage to be included in the model). Due to this, only three sampling events could be used (i.e. sweeps) and there was a two month gap between sampling events leading to an average capture probability of 0.20. Many factors could have contributed to a decreased capture probability such as natural mortality, emigration/immigration, behavioral changes and changes in habitat. More sampling events were not added to the established sampling regime because other studies have shown harmful effects of repeated electrofishing events on the same population (Gatz et al. 1986; McMichael et al.1999; Snyder 2003).

Equal capture probability is an assumption of the depletion and relative abundance estimators, but capture probability likely fluctuated throughout the study affecting the ability of these estimators to monitor changes in population size as has been shown in other studies (Riley and Fausch 1992; Williams 2002; Peterson et al. 2004; Rosenberg and Dunham 2005). Capture probability likely changed when brook trout migrated out of the sampling area to find adequate spawning habitat (Swanberg 1997) or to Lake Superior (Cross 2013). Environmental factors may have also influenced capture probabilities (Bohlin et al. 1989; Speas et al. 2004), such as when flows increased due to snowmelt and large precipitation events. The CPUE results supports these reasons for changes in capture probability because each year the CPUE results were low during May, when flows were high, and November, when spawning migrations occurred. Changes in habitat, primarily the creation of beaver ponds, also may have affected capture probability. Numerous beaver dams were created, which resulted in beaver ponds that were too deep to sample and thus, reduced capture probability in those reaches. Brook trout prefer slow moving pools (Gibson et al., 1981) making the low gradient of beaver ponds likely habitat for brook trout. The decline in capture probability would have negatively biased relative abundance estimates by decreasing catch. The depletion estimates were not influenced by beaver ponds because no beaver ponds were constructed in the reaches where the depletion estimates were conducted. Depletion abundance estimates dropped by 55% in 2010 from the previous year. This is a substantial decline, but it is likely a temporary change in capture probability rather than a true change in abundance, because the relative abundance estimates did not reflect the same substantial drop throughout the year demonstrating the importance of multiple estimates within a year. Capture probability likely decreased due to environmental variables (Bohlin et al. 1989; Speas et al. 2004) such as precipitation in September 2010 that increased flow, turbidity and stream width. Additionally, inexperienced electrofishing operators during the September 2010 sampling event likely further reduced capture probability. Any factors that decrease capture probability increase confidence intervals in mark-recapture estimates, because they decrease the number of fish marked and subsequently recaptured.

The small population also compromised the ability of the depletion estimator to generate abundance estimates and confidence intervals. The MLE cannot estimate abundance when the last catch is greater than the initial catch (Seber 1982), and this is more likely to happen when estimating a small population where few fish are captured as was the case in site two in 2010. Small populations also hamper the MLE's ability to calculate variance of the population estimate when fewer than 30 fish are captured, so the precision of the estimate is uncertain (Seber 1982). During no event were at least 30 fish captured, so confidence intervals could not be generated for the MLE. The small population also hampered the ability of the closed captures model in Program MARK to estimate confidence intervals because when the catch in the final pass was zero or one, the confidence intervals estimated by Program MARK were ±zero around the estimate. These confidence intervals are likely incorrect since it is unlikely that all the fish were captured in the reach even though the final catches were so low.

Changes in the sampling design could improve estimator accuracy and precision, but they could also have negative effects on the population. A shorter timeframe between sampling events for the mark-recapture estimator would have likely increased the number of recaptures, improving the precision of estimate. Additionally, a second depletion estimate each year would help confirm changes in population size rather than a change in capture probability that resulted in a biased estimate. There are drawbacks to increasing the number of sampling events and sampling frequency. More sampling events are more labor intensive, and it can be difficult to find adequate skilled labor to assist in the field. More importantly, studies have shown that numerous electrofishing sampling events in a short timeframe may reduce growth (Gatz et al. 1986), physically harm (McMichael et al 1999; Snyder 2003) or change the behavior of fish (Mesa and Schreck 1989; Nordwall 1999). Since this project focused on a species of conservation concern, electrofishing sampling events were minimized to collect only the necessary data needed for this project.

Based the results of this project, I recommend that when using the POPAN model, it is important to increase capture probability to greater than what was observed in this study (0.20) in order to increase precision of the population estimate (Pine et al. 2003). Capture probability can be increased, if sampling is done in a shorter timeframe, reducing or eliminating many of the factors that likely contribute to lower capture probabilities. Marking should likely be done one day prior to recapture as done in other studies (Rosenberg and Dunham 2005); however, this can create consequences associated with frequent sampling especially with a technique such as electrofishing that impacts physiology (Gatz 1986). When using the depletion estimator or relative abundance estimates, the aim should be to conduct the sampling during the same time of year due to seasonal variability in capture probability, as observed in this study. Depletion population estimates have been shown to underestimate population size in other studies (Peterson and Cederholm 1984; Riley and Fausch 1992; Rodgers et al. 1992; Rosenberger and Dunham 2005) and the depletion population estimates were consistently lower than the mark-recapture estimates in this study. If the depletion method is used, it is important to be cognizant that the population estimates will likely be negatively biased. Also, when monitoring population size by the depletion method or relative abundance, it is advisable to calculate population estimates from at least two sampling events to minimize the chances of differential capture probabilities biasing estimates, as was likely observed with environmental variability in this study. Additionally, all sampling should be done when capture probability is the highest, in this project it was mid-summer, to increase estimator precision.

Based on the goals of this multipurpose study which included monitoring of the populations over time and the protection of fish of concern, relative abundance provided reliable and useful information on changing population size while the others approaches were less valuable. Mark-recapture estimates had large confidence intervals, so changes in population size could not be detected. Depletion estimates were hampered by sample size constraints and likely underestimated the population size. Relative abundance seemed to be the least likely to be biased because sampling was done on a frequent basis. Sampling on a frequent basis helped identify apparent changes in capture probability because, when the catch temporarily fluctuated in comparison to the same time the previous year or the adjacent sampling periods, the change in catch could be attributed to seasonal or environmental influences on capture probability rather than a change in population size.

Table 2.1: Assumptions for methods that were used to estimate changes in population size in a brook trout population in the Mosquito River, Pictured Rocks National Lakeshore, Michigan: mark-recapture model, the removal method and catch per unit effort (Otis et al. 1978; Seber 1982).

Assumption	Mark-Recapture	Removal Method	CPUE
All animals within a sample have an equal probability of capture	X	X	X
Equal probability of survival for marked and unmarked animals from one sampling time to the next	X		
Marks are not lost or overlooked	X		
All animals are immediately released and sampling periods have a short duration (i.e. instantaneous)	X		
Equal capture probability among samples		X	X
Closed Population (i.e. No births or deaths/ No immigration or emigration)		X	X

Table 2.2: Model ranking of Program MARK mark-recapture models of the brook trout population from 2008 to 2010 in the Mosquito River, Pictured Rocks National Lakeshore, Michigan.

Year	Model	AICc	Delta AICc	AICc Weights	Model Likelihood	Num. Par
2008	{p(.),phi(t),pent(t)}	171.99	0	0.44402	1	6
	{p(t),phi(.),pent(t)}	172.213	0.2237	0.39703	0.8942	5
	{fully time dep}	174.044	2.0547	0.15894	0.358	6
	{p(.),phi(.),pent(t)}	193.392	21.4021	0.00001	0	5
2009	{p(t), phi(.),pent(t)}	229.846	0	0.50549	1	5
	{fully time dept}	231.676	1.8302	0.20244	0.4005	6
	{p(.)phi(t)pent(t)}	231.898	2.0519	0.1812	0.3585	6
	{p(.)phi(.)pent(t)}	232.88	3.0343	0.11087	0.2193	5
2010	{p(.)phi(t)pen(t)}	296.152	0	0.66574	1	6
	{p(t)phi(.)pent(t)}	298.324	2.1723	0.22469	0.3375	5
	{fully time dependent}	300.179	4.0268	0.0889	0.1335	6
	{p(.)phi(.)pen(t)}	303.096	6.9442	0.02067	0.031	5

Table 2.3: Summary of the number of brook trout marked, recaptured, total captured, capture probability and mark-recapture abundance estimates from 2008 to 2010 in the Mosquito River, Pictured Rocks National Lakeshore, Michigan.

	2008			2009			2010		
	May	Aug	Nov	May	Aug	Nov	May	Aug	Nov
Marked	56	135	35	66	137	61	109	125	61
Recaptured		17	9		14	21		26	22
Total Captured	56	152	45	66	151	82	109	151	83
Capture Probability		0.30	0.05		0.37	0.10		0.28	0.09
Abundance		506			407			542	
Confidence Interval		±191			±354			±379	

Table 2.4: Total September Precipitation in Munising, MI from 2008 to 2010 (Source: us climatedata.com).

Total September Precipitation (mm)			
Average	2008	2009	2010
10.3	7.62	5.33	23.8

Table 2.5: Depletion estimates (number of fish per reach) for the maximum likelihood estimator (MLE) and Program MARK (MARK) from 2008 to 2010 for the Mosquito River, Pictured Rocks National Lakeshore, Michigan. Lower confidence 95% intervals (L-CI) and upper 95% confidence (U-CI) are for Program MARK. * denotes when an estimate could not be calculated for the MLE.

	2008				2009				2010			
Site	MLE	MARK	L-CI	U-CI	MLE	MARK	L-CI	U-CI	MLE	MARK	L-CI	U-CI
2					5.03	3.00	3.00	3.00	*	2.00	2.00	2.00
5	22.79	11.79	9.20	47.83	3.07	2.00	2.00	2.00	1.00	1.00	1.00	1.00
9	13.50	13.00	13.00	13.00	22.71	22.00	22.00	22.00	6.54	6.00	6.00	6.00
11					15.16	12.81	12.04	26.56	11.69	8.43	21.98	21.98
14	16.00	14.37	14.01	24.90	22.74	20.21	18.24	38.21	11.72	11.00	11.00	11.00
16					28.32	26.67	25.19	39.74	11.05	11.00	11.00	11.00

Figure 2.1: Pictured Rocks National Lakeshore, Alger County, Michigan.

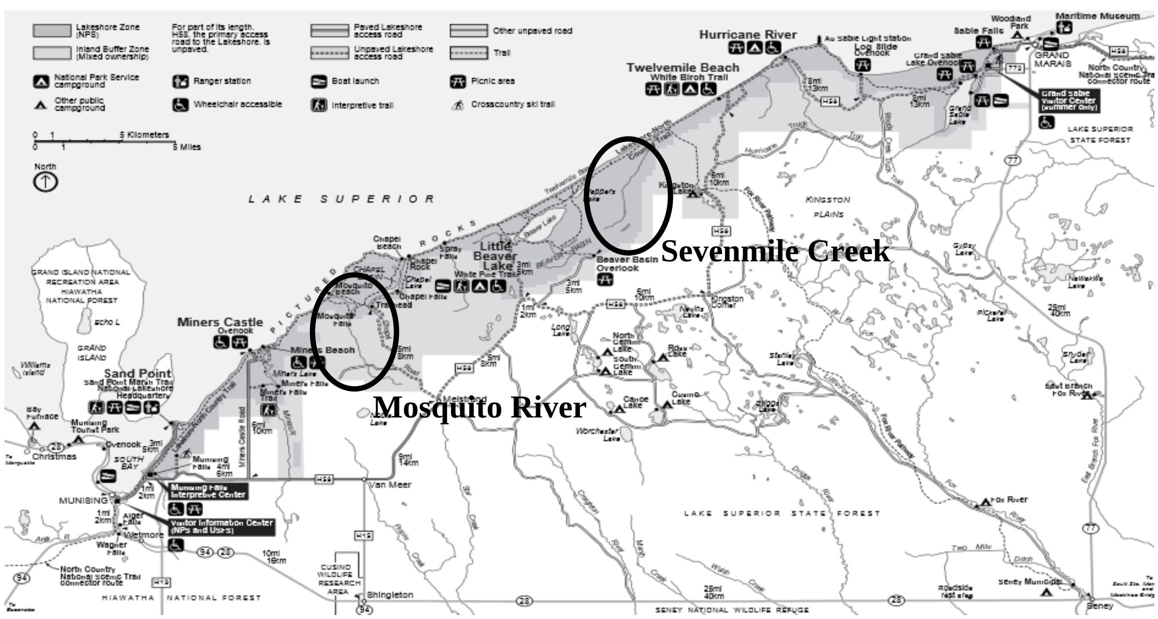

Figure 2.2: Map of Pictured Rocks National Lakeshore, Michigan with the study locations identified.

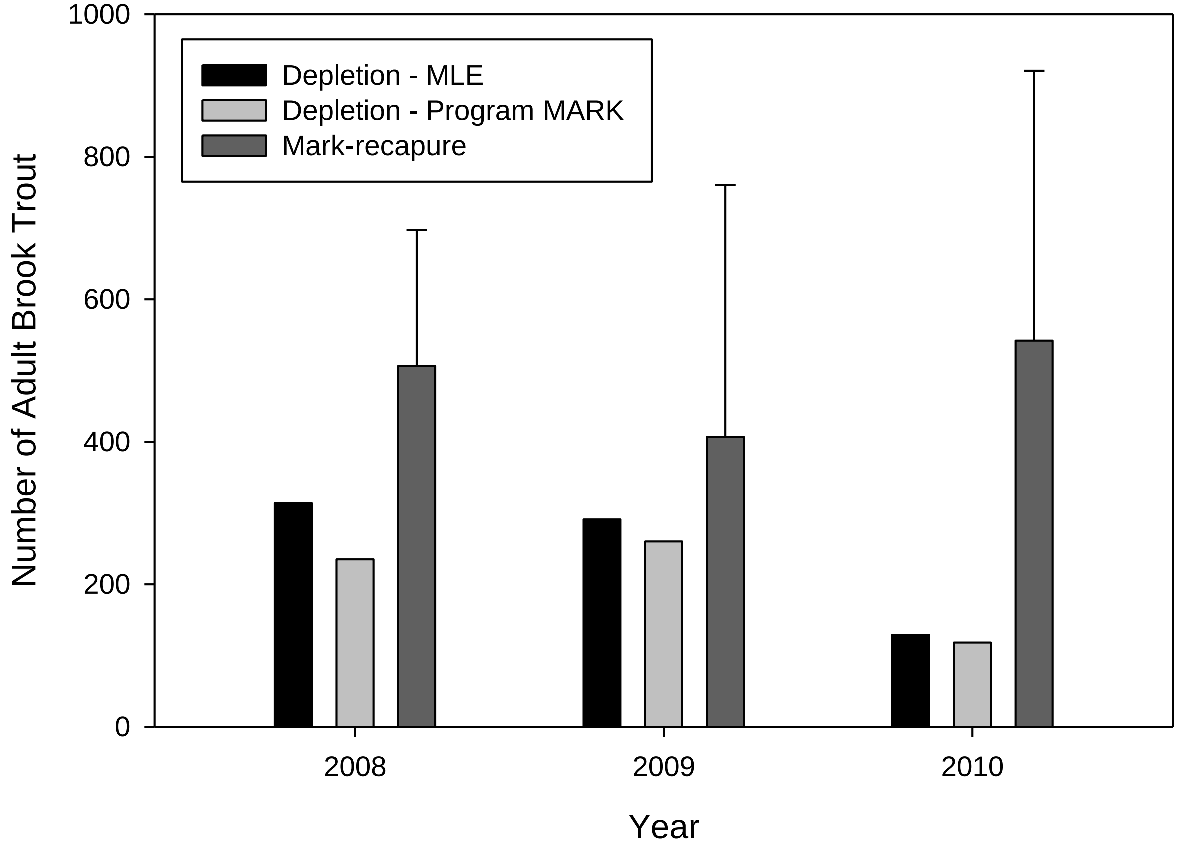

Figure 2.3: Brook trout abundance estimates for the 2.7 km study area from 2008 to 2010 from the Mosquito River, Pictured Rocks National Lakeshore, Michigan. Depletion estimates calculated in Program MARK and by the maximum likelihood estimator (MLE) were extrapolated out to the whole study area. Mark-recapture abundance estimates were created in Program MARK and are shown with the upper 95% confidence interval.

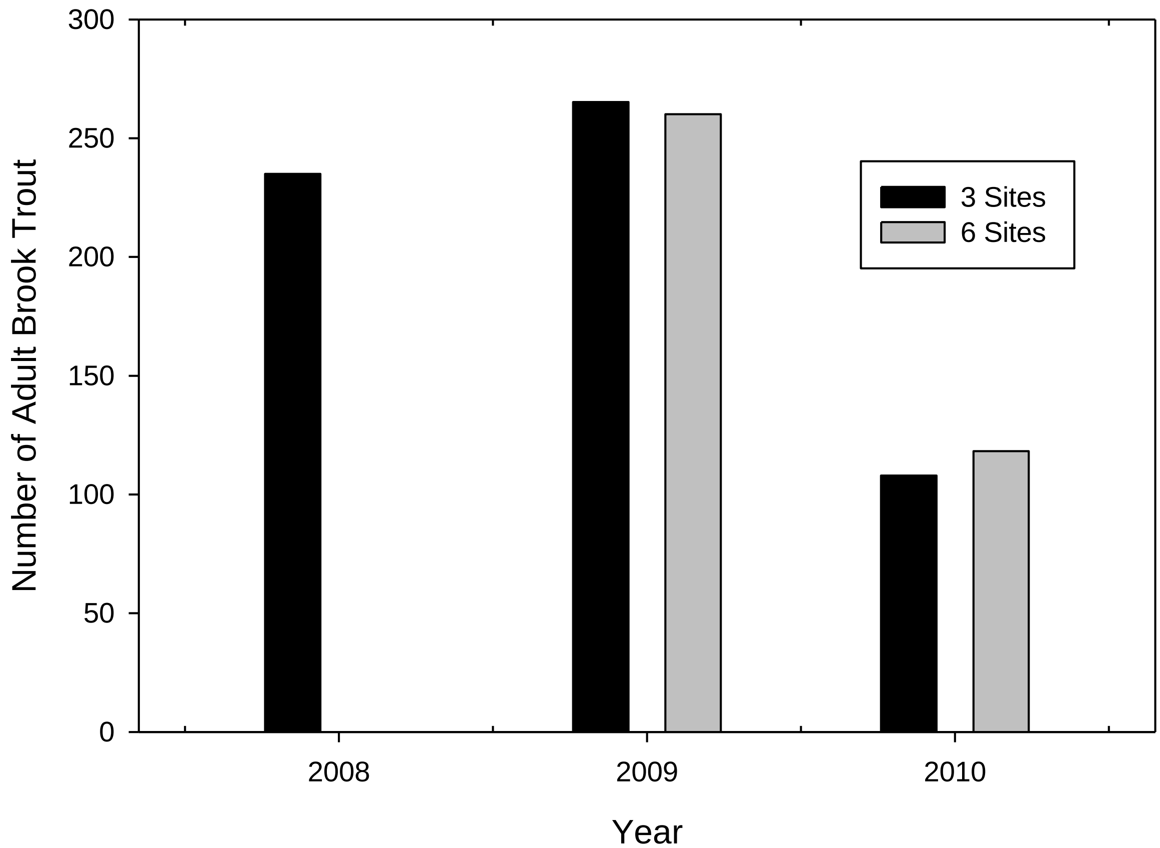

Figure 2.4: Comparison of stream wide population estimates when three or six sites were used to calculate estimates in Program MARK from 2008 to 2010 from the Mosquito River, Pictured Rocks National Lakeshore, Michigan.

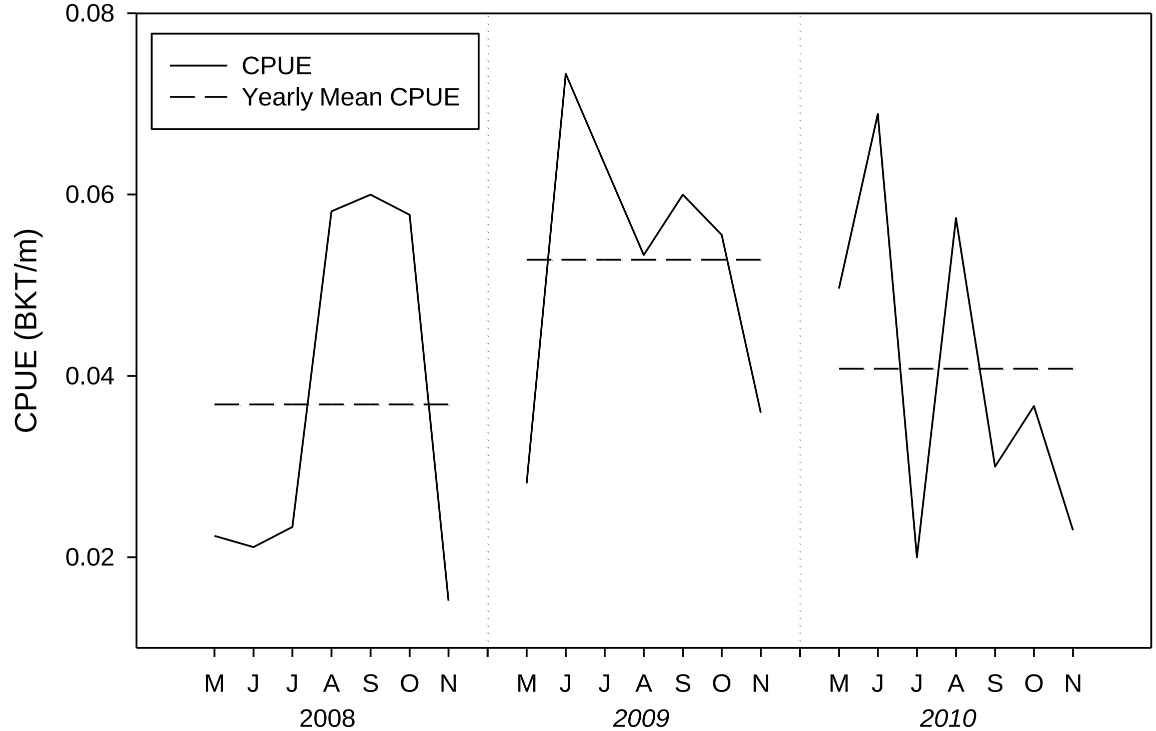

Figure 2.5: CPUE of brook trout (excluding young-of-year) by month from 2008 to 2010 from the Mosquito River, Pictured Rocks National Lakeshore, Michigan.

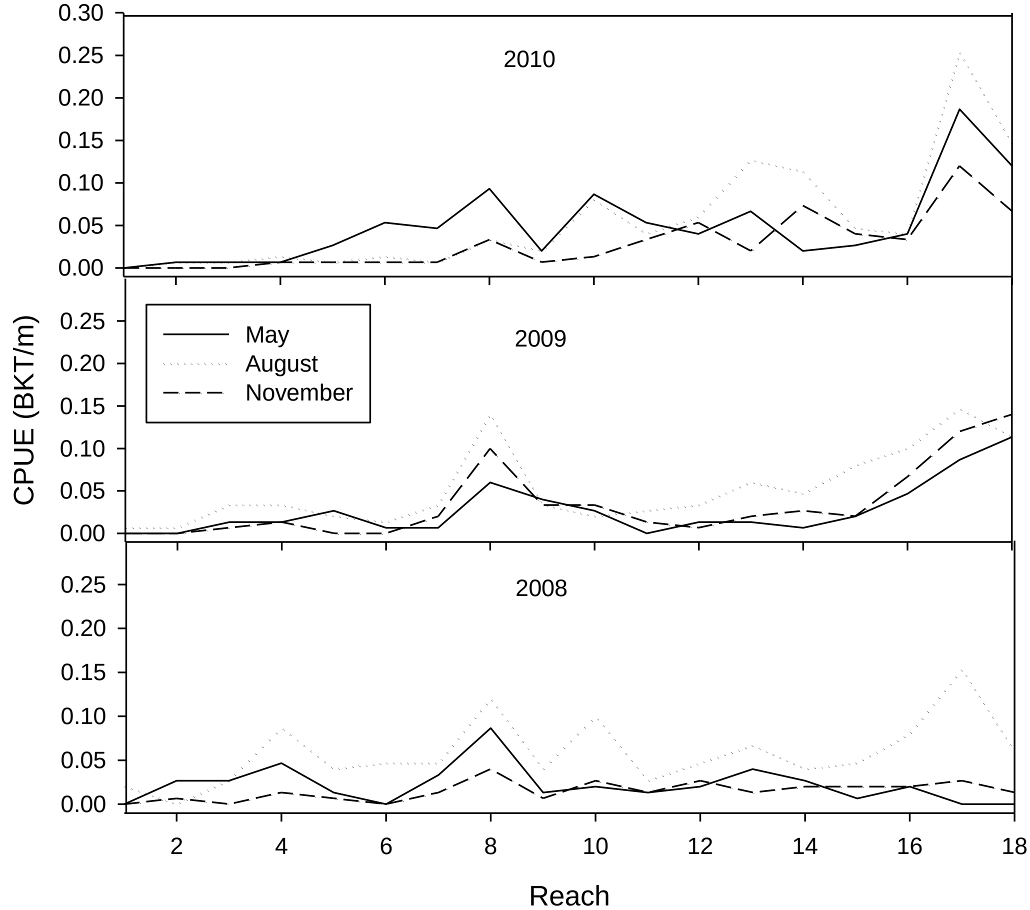

Figure 2.6: CPUE of adult brook trout by reach and season (May, August, November) from 2008 to 2010 from the Mosquito River, Pictured Rocks National Lakeshore, Michigan.

CHAPTER 3: THE EFFECT OF A NON-NATIVE SALMONID REMOVAL ON BROOK TROUT (*SALVELINUS FONTINALIS*) DENSITY AND GROWTH IN A SMALL MICHIGAN STREAM

CHAPTER SUMMARY:

In many studies non-native salmonids (e.g. *Oncorhynchus kisutch*, *O. mykiss*) have been shown to outcompete native brook trout *Salvelinus fontinalis* for limited resources. Other research has shown that exotic salmonids can be successfully removed/depleted from stream environments and that native salmonids responded positively to this intervention. In Sevenmile Creek, Pictured Rocks National Lakeshore, Michigan, an experimental brook trout rehabilitation project was conducted from 2008 to 2011. Exotic salmonids were lethally removed via backpack electrofishing and adult trapping during spawning, while salmonid populations were monitored in a nearby stream for reference. Over the course of the project, 5,320 exotic salmonids were removed from the treatment stream. The primary species removed were steelhead trout (3,138) and coho salmon (2,177). Age-one steelhead trout populations decreased by 61% in the treatment stream, while an increase of 171% was observed in the reference stream from 2008-2011. Young-of-year steelhead trout density dropped by 67% in the treatment stream and increased by 47% in the reference stream during the removal. The effect of the treatment was difficult to assess for young-of-year coho salmon density because of failed year classes in the reference stream. The year following the treatment brook trout young-of-year density increased by 260% compared to the beginning of the project, while young-of-year brook trout density declined by 57% in the reference stream. Adult brook

trout populations were variable, and showed no consistent pattern in response to the removal during the time-frame of our study. In the final years of the project, an increase in brook trout density was observed in the lower reaches, which were once primarily dominated by non-native salmonids. These data suggest that interspecific competition may be a factor limiting the brook trout population in Sevenmile Creek.

INTRODUCTION:

Brook trout (*Salvelinus fontinalis*) in the Lake Superior basin exhibit considerable life history variation. Lacustrine trout spend their entire life history in the lake, adfluvial trout reside in the lake and spawn in tributaries, while fluvial trout inhabit only the tributary environment. Brook trout that enter the lake for an ecologically significant portion of their life are locally considered “coasters” and the term encompasses both lacustrine and adfluvial life histories. Historically, adfluvial brook trout were abundant throughout Lake Superior (MacCrimmon and Gots 1980) and spawned in at least 106 streams (Newman and Dubois 1996) including tributaries in Pictured Rocks National Lakeshore; however, much of this evidence is largely anecdotal and provides little practical information for modern management (Leonard et al. 2013). In the early 1900s, the Lake Superior coaster brook trout (*Salvelinus fontinalis*) fishery collapsed, primarily due to overfishing (Hansen 1994). Loss of suitable spawning habitat related to logging activities (Horns et al. 2003) and interactions with non-native salmonids may have also played a role in the decline in coaster brook trout populations. Currently, coasters have been reduced to a few isolated populations that persist in waters around Isle Royale, the Salmon Trout River, Lake Nipigon, the Nipigon River region, Northeastern Lake Superior (Huckins et al. 2008), Pictured Rocks National Lakeshore (Kusnierz 2009;

Kusnierz 2014) and in Minnesota's Lake Superior tributaries (Ward 2008). Restrictive size and bag limits now protect coasters in some areas from overharvest and best management practices are used by the logging industry (Aust et al. 2004; USFS 2012). Interaction with exotic salmonids may impact brook trout of both forms (Fausch and White 1986; Gibson et al. 1981; Larson and Moore 1985; Krueger and May 1991). Similarly, resident brook trout in the region are highly valued and are of conservation concern in some areas.

Species composition in the Great Lakes has changed dramatically in the past century. Non-native salmonids were introduced to the Great Lakes basin as early as the 1870s to diversify sport fishing opportunities and provide biological control for alewifes (Crawford 2001). Introductions into Lake Superior that have formed naturalized, self-sustaining populations include coho salmon (*Oncorhynchus kisutch*) and a lake migratory form of rainbow trout (*O. mykiss*) known as steelhead trout. Both species spend the juvenile stage of their life in the tributary environment where competition with brook trout could occur, possibly limiting rehabilitation efforts. Brook trout and coho salmon also spawn at similar times (Becker 1983) making competition for spawning habitat another possible negative interaction. Other studies have shown that long-term persistence of native fishes can be threatened by introduced species (Allan and Flecker 1993; Rahel 2000).

Adult steelhead spawn in the spring and, once hatched, the juveniles reside in tributaries until they migrate to the lake environment two years later (Becker 1983). Competitive interactions could occur during the juvenile life stage while they inhabit the same environment as brook trout. If competition does occur, it could be amplified because there are typically many more juvenile steelhead than brook trout in tributaries on the south shore of Lake Superior (Huckins et al. 2008). In other areas, steelhead trout have been shown to compete with other salmonids, including brook trout, for limited resources, such as food or stream position. Brook trout and steelhead trout consume similar diets (Ensign et al. 1991) and steelhead trout can outcompete brook trout for food (Rose 1986; Isely and Kempton 2000) and optimal stream position (Gibson et al. 1981; Larson and Moore 1985), thus limiting brook trout growth and survival.

Coho salmon inhabit the tributaries for the first year of life (Becker 1983) and can cause a reduction in brook trout populations in tributaries of Lake Superior (Stauffer et al. 1977) . Juvenile coho salmon emerge earlier in the spring, are larger at emergence, and are generally more aggressive than brook trout (Fausch and White 1986). These factors may make coho salmon superior competitors for food and stream position, which supports faster growth than occurs in brook trout and likely results in higher coho salmon survival (Gibson et al. 1981; Fausch and White 1986). There also may be competition during spawning because, through modification of spawning habitat, coho salmon can physically destroy eggs of native brook trout which are deposited at about the same time in the fall (Krueger and May 1991).

Density dependent or density independent factors can regulate lotic fish populations (Strange et al 1992; Lorenzen and Enberg 2001; Rose et al 2001; Lobon-Cervia and Mortensen 2005). Density independent factors (e.g. environmental) such as high discharge and fluctuating stream temperature during the early life stages have also been shown to have profound impacts on recruitment and abundance of stream dwelling salmonids (Strange et al. 1992; Lobon-Cervia and Mortensen 2005). Other studies have provided compelling evidence that stream dwelling salmonid populations, including brook trout (McFadden et al. 1967), are regulated through density dependent processes (Jenkins et al. 1999; Lorenzen and Enberg 2002; Lobon-Cervia 2007).

Non-native salmonid removals have been conducted with both successes and failures. Meronek et al. (1996) reviewed fish control projects and reported success rates ranging from 33 to 57%. Electrofishing has been the most common method for removing non-native, stream-dwelling salmonids (Moore et al. 1983; Thompson and Rahel 1996; Kulp and Moore 2000;Shepard et al. 2002; Meyer et al. 2006). Many exotic salmonid removals have occurred in Rocky Mountain streams with eradication of brook trout as the objective. Meyer et al. (2006) attempted to remove non-native brook trout from an Idaho stream over a three year period and was able to remove over 80% of the brook trout population, but due to compensation, abundance of age-0 brook trout increased 789% during the two years following the removal. Thompson and Rahel (1996) significantly reduced the number of exotic brook trout in small Rocky Mountain streams, but were unable to achieve full eradication. Shepard (2002) was able to completely eradicate brook trout over the course of eight years of electrofishing and construction of a barrier on White’s Creek in the Missouri River basin; after the eradication was complete, the native westslope cutthroat trout (*Oncorhynchus clarki lewisi*) population rebounded.

Other research has focused on removing exotic rainbow trout from Appalachian Mountain streams. Moore et al. (1986) determined it took six years of electrofishing second and third order streams in the Great Smokey Mountains to greatly reduce the number of exotic rainbow trout. Brook trout populations responded positively to the non-native removal, suggesting removal of non-native rainbow trout by electrofishing could be used as a successful management tool. Kulp and Moore (2000) determined a minimum of three removals per summer were needed to eliminate reproduction of non-native rainbow trout in small southern Appalachian streams.

According to the National Park Service (NPS) management policies (NPS 1988), the NPS is mandated to protect and preserve “naturally functioning ecosystems.” Pictured Rocks National Lakeshore (PIRO), like many parks in the country, has threats from non-native species including fish that may be endangering sustainability of native fish species. Exotic salmonids were intentionally introduced in the Great Lakes, but now may threaten small brook trout populations, including coaster brook trout in PIRO. Population control of exotic salmonids has been successful in other national parks throughout the country (Moore et al. 1983; Thompson and Rahel 1996; Kulp and Moore 2000;Shepard et al. 2002; Meyer et al. 2006).

As part of a native coaster brook trout rehabilitation project, a physical removal of exotic salmonids occurred in PIRO over a three year time period while monitoring salmonid populations in a nearby reference stream. The objectives for this project were: 1) monitor all salmonid populations in the removal and reference stream; 2) reduce the density of exotic salmonids through intensive electrofishing of juveniles and adult trapping; 3) determine if brook trout density responded to the non-native salmonid removal; 4) determine if salmonid growth rates were impacted as a result of the removal; 5) determine if mean size of brook trout changed as a result of the removal.

METHODS:

Study Site Description:

The study streams, Sevenmile Creek and the Mosquito River, are typical of southeastern tributary streams of Lake Superior with similarly sized watershed areas of 26.8 and 35.9 km^2, respectively. Sevenmile Creek, the treatment stream where exotic salmonids were lethally removed, and Mosquito River, the reference stream, are both third order streams located in PIRO in the Upper Peninsula of Michigan (Figure 2.1). The study locations for Sevenmile Creek and Mosquito River began at the mouth and extended upstream 2.1km and 2.7 km, respectively (Figure 2.2). The Mosquito River has a higher gradient with substrate dominated by gravel and cobble with occasional stretches of sand and bedrock. Sevenmile Creek is mainly cobble and gravel in the lower reaches with a higher gradient (2.16%), but in the upper portion of the research area the gradient decreased (0.48%) and the habitat was dominated by beaver ponds with sand and silt substrate. Woody debris was prevalent throughout both streams and served as the primary salmonid cover. Species composition in the streams was similar and was comprised of native brook trout and naturalized, non-native salmonid populations of coho salmon and steelhead trout. Non-salmonid species that inhabited the streams included sculpin (*Cottus* spp.), central mudminnow (*Umbra limi*), dace (*Rhinichthys* spp.), suckers *(Catostomus* spp.), burbot *(Lota lota)*, and brook stickleback (*Culaea inconstans*).

Study design:

All sampling occurred annually in the ice free season, May through November. As part of a separate project, fish populations were monitored from 2004 through 2005 in Sevenmile Creek and the Mosquito River; although sampling effort was lower in 2004 these data were used as baseline data and are considered to be indicative of pretreatment conditions. The nonnative removal portion of this study occurred from 2008 through 2010 when exotic salmonids were lethally removed from Sevenmile Creek and fish populations were concurrently monitored in the Mosquito River. Post-removal populations were monitored in 2011 in both Sevenmile Creek and Mosquito River.

The research areas in both streams were stratified into three areas of similar habitat. Each section of similar habitat type was divided into reaches. Sevenmile Creek was split into 14 reaches and the Mosquito River was split into 18 reaches. Reaches were approximately 150m in length; however, when surveyed the reach length ranged from 138 to187 m and these surveyed measurements were used in the analysis of the data.

Sampling occurred once per month during the ice free season in 2004 through 2005 and in 2008 through 2011. During May, August and November, the entire study site was sampled for both Sevenmile Creek and the Mosquito River. To reduce stress on the fish from repeated sampling events, a subset of the study reaches was sampled during June, July, September and October such that two reaches were sampled in each section (N=6 reaches per stream). Reaches were chosen to minimize repeated sampling in two successive months for a single reach. Reaches were originally selected randomly within each section, but the sampling sequence was held constant in subsequent years to facilitate between year comparisons.

Relative density reported as catch per unit effort (CPUE) (fish captured/meter electrofished) was used to monitor salmonid populations for the duration of the project. Precautions were taken to ensure a representative sample was collected from the population. The majority of the sampling occurred under typical environmental conditions (e.g. average monthly flows and low turbidity) to minimize variability in catchability. Fish were likely to be distributed based on habitat, so stratified random sampling was implemented to cover all habitat types. Sampling effort was high (N=7/year) and distributed across the ice free season to compensate for unforeseen fluctuations in catchability, along with behavioral and seasonal variation in the populations.

Sampling:

Each month, sampling within reaches was accomplished by making a single electrofishing pass upstream with a crew consisting of one backpack electrofishing operator with a two netter crew. ETS Electrofishing Systems LLC (Madison, WI) backpack electrofishing units were used; the exact voltage setting depended on the conditions of the stream at the time of sampling but was set near 300 volts with a 40% duty cycle, and a rate of 60 pps. All salmonids caught were identified and measured for total length (mm) and weight (g). All exotic salmonids collected were humanely killed (2008-2010 in Sevenmile only) via cranial concussion followed by decapitation (according to NMU IACUC and NPS approved protocols) and discarded in the riparian zone.

Adult exotic salmonids were targeted via fish traps during spawning migrations in Sevenmile Creek in 2009 and 2010. Traps were placed in both upstream and downstream directions to capture migrating fish moving in either direction. Rectangular traps (1.2m x 0.61m x 0.61m) were constructed out of 10 gauge custom expanded sheet metal with 12mm diamond openings. The entrance to the trap was mesh netting extending 0.7m into the trap tapering to a circular 0.2m diameter opening. Mesh netting (2cm) was placed across the stream to guide fish into traps that were placed approximately 75m upstream from Lake Superior. Traps were positioned in the stream in mid-March and removed in mid-May to capture migrating steelhead trout. Traps were again placed in the stream from late-September through late-October to capture migrating coho salmon. After setting the traps, the study area upstream of the traps was electrofished to catch any adult non-native salmonids that may have moved into the stream prior to deployment of the traps. All exotic salmonids captured were humanely killed via cranial concussion with decapitation and discarded in the riparian zone. All native fishes were placed on the opposite side of the trap in their direction of movement.

Data Analysis:

For analysis, salmonids were sorted by species: brook trout (BKT), coho salmon (COH), steelhead trout (STH), and age group: young of year (YOY) and older resulting in five categories for analysis (BKT, BKT YOY, COH YOY, STH, STH YOY). Young-of-year fish and age one fish were determined by length frequencies. Coho salmon older than YOY (other than adult returning spawners) were not usually present in these systems since coho juveniles primarily outmigrate in the fall of their hatching year in this these systems. Catch per unit effort (CPUE) was calculated for each category of fish on a linear reach basis.

Before After Control Impact (BACI) analysis, often used to measure the effect of environmental impacts on populations (Smith, 2002; Underwood, 1991; Underwood, 1994), was used to evaluate differences in density in the treatment stream in comparison to the reference stream pre and post treatment. The CPUE data was normalized by transforming it by adding one to the CPUE data and then raising it to $\log_{10}$. Six, two-way

ANOVAs were run each using different “before” and “after” time periods (i.e. pre and post-treatment) (Table 3.3). BACI analyses 1 to 3 used 2010 as post-treatment data and BACI analyses 4 to 6 used 2011 as the “after” time period. BACI analysis 1and 4 used 2004 through 2005 as the “before” time period. The BACI analysis 2 and 5 used 2004 through 2005 and 2008 as the “before” time period. Lastly, the BACI analysis 3 and 6 used 2008 as the “before” time period. Different time periods were used for the pretreatment data because certain caveats apply to the pretreatment data: 1) sampling effort was lower in 2004, 2) coaster brook trout were stocked in 2004 through 2005, and while stocked fish were marked prior to stocking and were not included in the density data, they could have affected native trout and 3) 2008 marked the beginning of the removal treatment. Due to the possibility that these conditions added variability in the pretreatment data, different “before” time periods were used in the analyses to capture “normal” pretreatment densities. Different time periods were used for the post-treatment data because it was unknown when the salmonid densities displayed the maximum response to the treatment.

Two factors and factor interactions were assessed in the BACI analyses to determine if significant variation in the CPUE data was caused by any of the factors investigated. One factor assessed was termed *Treatment*, which indicated the study streams; Sevenmile Creek was the treatment stream and the Mosquito River was the reference stream. The other factor was the time frame before and after the removal (*Time Period*). The interaction between the *Time Period* and *Treatment* factors was key in assessing the impact of the removal because temporal environmental variation of populations was taken into account allowing for a direct evaluation of the effects of the removal. Additionally, the data from time period between the pre and post treatment periods (2009-2010) were taken into account when interpreting the results.

Instantaneous growth rates were calculated for young-of-year brook trout, young-of-year coho, and young-of-year steelhead from August to November (Figures 3.7, 3.8 and 3.9). Earlier sampling dates within a year were not used because an insufficient number of fish had recruited to the gear. Instantaneous growth rates were calculated for each section of the stream and then a stream wide mean was derived. Instantaneous growth was calculated by:

$$G = \frac{Ln(L_2) - Ln(L_1)}{t_2 - t_1}$$

where G = growth, L_2 = mean length per section November, L_1 = mean length per section August, t_2=day of the year at the November sampling date, and t_1=day of the year at August sampling date. Growth rates were not calculated for 2004 because there were an inadequate number of fish captured in November of that year. Growth rates were not calculated for adult brook trout because ages were not determined for these fish and calculating growth rate of unaged fish is problematic. However, mean length was calculated for adult and young-of–year brook trout each year.

RESULTS:

From 2008 to 2010, 5,320 non-native salmonids were removed from Sevenmile Creek. Steelhead trout comprised 59.0% of the total catch; 94.9% were less than 225mm and 5.1% were greater than 225mm. Coho salmon encompassed 40.9% of the total catch; 88.4% were less than 225mm and 11.6% were greater than 225mm. Adult pink salmon (*Oncorhynchus gorbuscha*) were caught occasionally, but comprised less than 0.01% of the catch (Table 3.1).

From 2008 to 2011, age one steelhead density decreased in the treatment stream, while it increased substantially in the reference stream, suggesting that the treatment had significant impacts on the population. In the treatment stream age-one steelhead density decreased from 0.018 fish/m (SE, 0.003) when the removal began in 2008 to 0.007 fish/m (SE, 0.001) in 2011 after the removal, a 61.0% decrease. In the reference stream densities increased nearly threefold from 0.032 fish/m (SE, 0.003) in 2008 to 0.087 fish/m (SE, 0.008) in 2011, a 171% increase (Figure 3.1). The coefficient of variation was 41% for the yearly average density during the study (Table 3.2). Significant differences in the relationship between the control and reference streams were seen in BACI analyses 3, 5 and 6, $p<0.001$, $p=0.004$ and $p<0.001$, respectively (Table 3.3). These data suggest that environmental influences were favorable for age-one steelhead production during the project and that the treatment may have suppressed age one steelhead density in Sevenmile Creek.

Young-of-year steelhead density appeared to be heavily impacted by the treatment (Figure 3.2). Young-of-year steelhead density dropped in the treatment stream from 0.107 fish/m (SE, 0.017) in 2008 to 0.049 fish/m (SE, 0.008) in 2009 and 0.035 fish/m (SE, 0.006) in 2010, a decrease of 67.2%. Upon cessation of the project in 2011, density quickly rebounded to 0.085 fish/m, (SE, 0.015) nearly to the 2008 density level. During the project, while young-of-year steelhead density was declining in the treatment stream, density was increasing in the reference stream from 0.097 fish/m (SE, 0.014) in 2008, to 0.146 fish/m (SE, 0.018) in 2009 with a decrease to 0.142 fish/m (SE, 0.015) in 2010, a 47.0% increase overall, followed by a decline to 0.089 fish/m (SE, 0.012) in 2011 (Figure 3.2). Due to the rapid rebound of YOY STH density in 2011, only BACI analyses 1 to 3 showed significant differences (all were $p<0.001$) in the relationship between the densities in the treatment and reference streams (Table 3.3). The coefficient of variation was 52% for the yearly average density during the study (Table 3.2). Both the inverse density trends in the treatment stream compared to the reference stream during the project and the immediate recovery in the treatment stream suggest that the decrease in young-of-year steelhead trout density in the treatment stream was due to the project and not environmental influences. These data also highlight the extremely rapid recovery in young-of-the year steelhead trout production after cessation of the treatment.

Young-of-year coho density did not serve as a good indicator of the influence of the treatment on their population due to failed year classes in the reference stream. In the treatment stream, density increased slightly during the project, with the greatest variation from 0.055 fish/m (SE, 0.008) in 2008 to 0.062 fish/m (SE, 0.009) in 2009, a 13.9% increase. There is no suggestion that we were able to significantly deplete coho salmon as a result of the treatment, despite the large number of individuals removed. Density in the reference steam decreased sharply from 0.049 fish/m (SE, 0.007) in 2008 to 0.004 fish/m (SE, 0.002) in 2009, a 91.4% decrease. Density stayed low in 2010, 0.004 fish/m (SE, 0.001) and then increased to 0.023 fish/m (SE, 0.009) in 2011 (Figure 3.3). There were significant differences in BACI analyses 1,2,3, and 6 ($p=0.002$, $p<0.001$, $p<0.001$, and $p=0.049$, respectively) (Table 3.3). The coefficient of variation was 28% for the yearly average density during the study (Table 3.2). The low young-of-year coho densities in the reference stream in 2009 and 2010 were likely due to failed year classes resulting from low water levels which impeded adult upstream migration at the river mouth.

Adult brook trout densities in both the treatment and the reference streams were quite variable showing no discernible trends (Figure 3.4), making it difficult to assess the influences the removal of non-native salmonids had on the adult brook trout population. The density in the treatment stream ranged from a low of 0.023 fish/m (SE, 0.003) in 2009 to a high of 0.053 fish/m (SE, 0.007) in 2010, a 136% increase, followed by a decline to 0.040 fish/m (SE, 0.005), a 25.1% decrease in 2011, resulting in a final density that was higher than initially observed. The density in the reference stream ranged from 0.033 fish/m (SE, 0.004) in 2008 to 0.044 fish/m (SE, 0.005) in 2009, a 32.1% increase staying fairly constant between this range (Figure 3.4). The coefficient of variation in the treatment stream was 31% for the yearly average density during the study. There were no significant differences in relative density of adult brook trout between the two streams in the BACI analyses (Table 3.3).

Young-of-year brook trout density in the treatment stream showed an increasing trend throughout the treatment, while young-of-year brook trout in the reference stream steadily declined, indicating the treatment may have had significant effects on the young-of-year brook trout population. At the beginning of the project, in the treatment stream, density was 0.012 fish/m (SE, 0.003) in 2008, then varied moderately in 2009 and 2010, then jumped to 0.043 fish/m (SE, 0.007) in 2011, a 260% increase from the start of the project. Throughout the project, density in the reference stream showed a steady decrease, from 0.023 fish/m (SE, 0.004) in 2008 to 0.010 fish/m (SE, 0.002) in 2011, a 56.6% decrease (Figure 3.5). The coefficient of variation was 62% in the treatment stream for the yearly average density during the study (Table 3.2). The dramatic increase of young-of-year brook trout densities observed in 2011 in the treatment stream was significant in BACI analyses 1,2,4,5, and 6 ($p<0.001$, $p=0.005$, $p=0.002$, $p<0.001$, and $p<0.001$, respectively) (Table 3.3).

Instantaneous growth rates calculated from August to November were fairly stable for most species. Young-of-year brook trout instantaneous growth rates in Mosquito River ranged from 0.0022 Ln(mm)/day (SE, 0.0005) in 2005 to 0.0012 Ln(mm)/day (SE, 0.0002) in 2009. Young-of-year instantaneous growth rates in Sevenmile Creek had similar range as the reference stream, ranging from 0.0024 Ln(mm)/day (SE, 0.0004) in 2009 to 0.0014 Ln(mm)/day (SE, 0.0001) in 2011 (Figure 3.6). The lowest growth rates occurred at peak or near peak fish density. The young-of-year coho instantaneous growth rates showed the same pattern in both streams, growth rates were the highest in 2005 and the lowest in 2009 (Figure 3.7). Young-of-year steelhead growth rates were generally the highest of the young-of-year salmonids, ranging from 0.0022 Ln(mm)/day (SE, 0.0004) to 0.0028 Ln(mm)/day (SE, 0.0005) in Sevenmile Creek and from 0.0018 Ln(mm)/day (SE, 0.0002) to 0.0028 Ln(mm)/day (SE,0.0002) in the Mosquito River (Figure 3.8).

Mean August lengths of young-of-year brook trout decreased throughout the project in Sevenmile Creek and the Mosquito River, from 79.4 mm (SE, 1.46) to 74.0 mm (SE, 0.92) and from 75.4 mm (SE, 0. 87) to 71.2 mm (SE, 1.54) in each stream respectively (Figure 3.9). Mean August lengths of adult brook trout showed no discernable trend in either stream and, for the most part, mean August lengths in both streams tracked together. Mean August lengths of adult brook trout in the Mosquito River were more variable ranging from 138 mm (SE, 2.62) in 2005 to 166 mm (SE, 2.17)

2008. Mean August lengths of adult brook trout were quite stable in Sevenmile Creek ranging from 155 mm (SE, 2.11) in 2010 to 163 mm (SE, 2.97) in 2008 (Figure 3.10).

The number of brook trout captured in Sevenmile Creek steadily increased as the study progressed; 433 brook trout were captured in 2008 and 864 were captured in 2011, a 50% rise . Young-of-year brook trout were responsible for the majority of the increase in catch observed in 2011 when 477 were caught. The number of brook trout greater than 200mm fluctuated throughout the study and showed no steady trend; however, the greatest number of brook trout over 200mm, 37, were captured in 2011 comprising 9.6% of the adult catch (Table 3.4).

DISCUSSION:

Exotic salmonid densities were significantly reduced when compared with the reference stream and subsequently an increase in brook trout density was observed. As has been suggested by other studies (Moore et al. 1983; Thompson and Rahel 1996; Kulp and Moore 2000;Shepard et al. 2002; Meyer et al. 2006), these data confirm that exotic salmonids can be depleted via intensive, manual effort, which may reduce the effect of interspecific density-dependent regulation. When the exotic salmonid densities were reduced, the remaining salmonids likely had a compensatory response attempting to promote a numerical increase in their populations (Rose 2001). This was the intent of the project for the brook trout population; however, it made the removal of the exotic salmonids even more challenging.

A more pronounced decline in non-native salmonid densities could have been offset by a compensatory exotic salmonid population response. According to Ricker (1975), when populations are experiencing an increase in exploitation (i.e. the treatment), a new equilibrium is reached because the decrease in abundance allows the remaining fish to respond with a greater rate of growth, reduced natural mortality, or greater rates of reproduction and/or survival of young. However, direct measurement of compensation in the field can be difficult (Rose et al. 2001). Since adult exotic salmonids spend the majority of their life in the lake environment (Becker 1983) where the treatment had no effect, it is unlikely that an increased rate of reproduction due to the treatment was observed. A likely mechanism that may have altered the observed effect of the removal was an increased in juvenile growth rate as observed in the juvenile steelhead population. Young-of-year steelhead instantaneous growth rates showed a positive trend and were consistently higher in the treatment stream during the removal (Figure 3.9). Faster growth rates have led to an increase in survival in numerous other studies (Jenkins et al. 1999; Lorenzen end Enberg 2002; Lobon-Cervia 2007). Meyer (2006) concluded that the compensatory response of the exotic salmonids likely contributed to his lack of success in an exotic salmonid removal project in the Rocky Mountains and suggested the increase in exploitation was sufficiently compensated by the reduction of natural mortality. A compensatory mechanism, such as an increase in growth rate potentially leading to increased survival, could have contributed to the ability of the exotic salmonid populations to cope with the increased exploitation from the treatment and may have been a reason a more pronounced decline was not observed.

Despite the compensatory ability of salmonids, juvenile steelhead densities were negatively impacted by the treatment. The treatment likely prevented a significant increase in age one steelhead as observed in the reference stream. Additionally, young-of-year steelhead density dropped 67% during the treatment. Many studies have provided evidence that juvenile exotic salmonids can outcompete brook trout for limited resources in stream environments (Stauffer et al. 1977; Gibson et al. 1981; Larson and Moore 1985; Fausch and White 1986; Rose 1986; Isely and Kempton 2000), which would presumably lead to slower growth and higher mortality during the juvenile life stage. However, while exotic salmonid densities were reduced in the treatment stream, brook trout instantaneous growth rates declined and mean size of young-of-year brook trout decreased. This was to be expected in 2011, when young-of-year brook trout densities were high, but not during 2009 and 2010 when young-of-year brook trout densities were at lower levels. These results suggest that decreasing juvenile steelhead density does not increase young-of-year brook trout growth rate; however, it may increase density.

Adult brook trout densities in both the treatment and the reference streams were quite variable, making it difficult to assess whether the treatment had an influence on adult brook trout densities in Sevenmile Creek (Figure 3.5). According to Dauwalter et al. (2009), the average annual coefficient of variation for trout (brook, brown, rainbow) population size in North America was 49%; the coefficient of variation in brook trout density in Sevenmile Creek was 31% (Table 3.2), well within the North American average for trout populations. Given this known variability in the trout populations, a numerical change may not be the best indicator to measure their response. The adult brook trout population may have displayed other compensatory responses such as an increase in reproductive success or redistribution within the stream to habitat once used by exotic salmonids.

An increase in reproductive success was likely one of the primary mechanisms responsible for the increase in young-of-year brook trout density in 2011. Density has been linked to many compensatory factors that affect reproductive success such as fecundity, maturation, spawning frequency, egg quality, and competition for spawning habitat (Rose et al. 2001). It is unlikely that spawning frequency changed because only one year class was observed each year and, due to the relatively short time period of the study, age at maturity likely did not change. Fecundity (i.e. larger eggs and/or more eggs) has been linked to adult size in other salmonid populations (Taube 1976; Ojanguren et al.1996). For an increase in fecundity to be responsible for an increase of young-of-year brook trout in 2011, the average body size of adult brook trout would have been expected to have increased in 2010, but this was not the case. It is possible that egg quality increased, which lead to an increase larval survivorship as a result of the study. While the exotic salmonid densities were reduced, brook trout may have had increased access to food resources or they could have been no longer forced into subpar habitat which had a higher energetic cost. This could have led to more nutrients and energy allocated toward reproduction, specifically gamete quality, which could help explain the surge of young-of-year brook trout observed in 2011.

Brook trout potential and reproductive success could also have been limited by competition for spawning habitat prior to the removal. In the Great Lakes, coho salmon have been known to have large spawning runs that tend to overlap with brook trout egg deposition (Krueger and May 1991). Since spawning habitat may be limited in small streams, later spawners could superimpose their redds on previously constructed nests displacing and/or destroying eggs deposited by earlier spawners (Krueger and May 1991). Redd superimposition can be a major cause of mortality for salmonid eggs and embryos, causing fry production to be inversely density dependent (McNeil 1964;

Fukushima et al. 1997; Taniguchi 2000). For example, redd superimposition by rainbow trout caused a 94% reduction in spawning success of brown trout in a small New Zealand tributary (Hayes 1987). Further studies between salmonid species have shown that females prefer to spawn on existing redd sites, dislodging eggs that have been placed there previously (Essington et al. 1998). Trapping of adult coho salmon during the 2010 spawning run was extremely successful. Prior to spawning, 157 adult coho were trapped from Sevenmile Creek, a 233% increase from the previous year. The following year brook trout young-of-year density significantly increased (260%) compared to the beginning of the project. These data suggest that spawning competition between coho salmon and brook trout may be a significant factor limiting reproductive success of the brook trout population in Sevenmile Creek.

Leonard et al. (2012) examined overall brook trout distribution (adult and young-of-year) in our study area throughout each stream during the project using instream GIS techniques. At the beginning of the project, the lower reaches were dominated almost exclusively by non-native salmonids, while the upper sites were occupied mainly by brook trout. As the project proceeded and non-native densities were reduced, brook trout density increased throughout the stream. The largest absolute increase in density was observed in the upper sites of the study area, where brook trout were already abundant. The lower reaches, once dominated by non-native salmonids, showed a larger proportional increase than the upper reaches. This suggests that the reduction in steelhead trout and coho salmon densities from the lower reaches allowed brook trout density to increase in the newly vacant habitat. Other studies have found density dependent movement by individuals toward lower quality habitat which may have resulted in higher mortality or slower growth (Gibson et al. 1981; Fausch 1984; Larson and Moore 1985). These data imply that interspecific competition may be limiting brook trout distribution by forcing them into subpar habitat for the non-natives which could be reducing brook trout production in the lower reaches.

To help bring together the individual results observed in this project into a larger stream wide picture, I propose a hypothetical model. Prior to the non-native salmonid removal, interspecific competition rather than intraspecific competition played a bigger role in regulating the brook trout population. Redd superimposition and decreased larval survivorship may have been the interspecific competitive mechanisms that controlled the brook trout population size, although more data are needed to confirm this effect. Because the brook trout population was limited by interspecific competition, the density was lower, growth rates were higher, and the brook trout were restricted to the upper reaches by the presence of the non-native salmonids. Once the non-native salmonid densities were reduced, intraspecific competition became the primary driver limiting the brook trout population. In the absence of the interspecific mechanisms that limited the brook trout population, densities increased, growth rates declined, dispersal to the lower reaches of the stream increased and coasting behavior may have been facilitated (Cross 2013). Even though this is only a hypothetical model, it helps explain many of the results seen in this project and offers pathways for future research and management actions.

Exotic salmonid densities were reduced via electrofishing of juveniles and trapping of adults, subsequently positive results were seen in the young-of-year brook trout density. If the project could have continued to monitor the pulse of the 2011 young-of-year brook trout cohort, a subsequent increase may have been seen in adult brook trout density. It took Shepard (2003) seven years to eradicate exotic brook trout from a Rocky Mountain stream and the western cutthroat trout population rebounded. Moore et al. (1986) spent six years removing exotic rainbow trout from streams in the Great Smokey Mountains to greatly reduce density, which promoted an increase in native brook trout populations.

The results from this project have management implications for other brook trout populations where interspecific competition between salmonids may occur. Prior to providing aquatic organism passage allowing exotic salmonids access to upstream habitat, managers should consider the potential impacts on native fish community. Along with providing fish passage to exotic salmonids, fisheries managers will have to decide if the pros outweigh the cons in regard to stocking coho salmon or steelhead trout into bodies of water where brook trout are present. Additionally, agencies such as the U.S. Fish and Wildlife Service and the National Park Service whose primary mission is to conserve, protect and restore native species, could use the methods described in this project, perhaps concurrent with altered angler regulations, to remove or reduce exotic salmonids with the ultimate goal of restoring native brook trout populations.

In summary, exotic salmonid populations in the treatment stream were reduced in comparison to the reference stream via intensive electrofishing and trapping of adults, and during this same time the brook trout population increased. The treatment likely had the greatest effect on the young-of-year steelhead population, reducing the density by 67% from the beginning of the removal. The treatment likely significantly influenced the age-one steelhead population by suppressing an increase in density similar to that observed in the reference stream. The decline in non-native salmonid density may have led to an increase in brook trout density in the lower reaches previously occupied primarily by exotic salmonids. The decrease in both the juvenile steelhead density and the number of spawning adult coho likely had a two pronged effect, which accounted for a significant increase of the young-of-year brook density. First, a possible reduction in competition for vital resources may have led to an increase in egg quality or larval survival. Secondly, due to the successful trapping of adult coho in 2010, there may have been a reduction in competition for spawning habitat, which could have led to greater nest success. These results suggest that there may have been interspecific competition occurring between salmonid species in Sevenmile Creek, that may have limited brook trout densities, and fisheries managers may want to take these results into consideration when managing these populations.

Table 3.1: Exotic salmonids removed from 2008 to 2010 in Sevenmile Creek, Pictured Rocks National Lakeshore, Michigan.

	Exotic Salmonids Removed from Sevenmile Creek						
	2008		2009		2010		
Species	(<225mm)	(>225mm)	(<225mm)	(>225mm)	(<225mm)	(>225mm)	Total
Coho Salmon	467	49	783	67	654	157	2,177
Pink Salmon	0	1	0	0	0	4	5
Steelhead Trout	1,210	7	990	50	777	104	3,138
Total	1,734		1,890		1,696		5,320

Table 3.2: Average annual density (fish/m) for each class of fish and the associated coefficient of variation for 2004, 2005 and 2008 to 2011 in Sevenmile Creek, Pictured Rocks National Lakeshore, Michigan.

Year	BKT	BKT YOY	COH	STH	STH YOY
2004	0.0526	0.0327	0.0593	0.0331	0.0363
SE	0.0060	0.0074	0.0124	0.0047	0.0068
2005	0.0471	0.0167	0.0223	0.0258	0.0354
SE	0.0053	0.0033	0.0038	0.0036	0.0052
2008	0.0298	0.0118	0.0545	0.0182	0.1050
SE	0.0045	0.0030	0.0079	0.0026	0.0172
2009	0.0228	0.0205	0.0621	0.0225	0.0492
SE	0.0028	0.0040	0.0091	0.0030	0.0081
2010	0.0539	0.0068	0.0552	0.0265	0.0349
SE	0.0065	0.0014	0.0088	0.0037	0.0064
2011	0.0382	0.0425	0.0565	0.0071	0.0852
SE	0.0049	0.0069	0.0073	0.0013	0.0146
CV	31.0980	61.5090	28.3680	40.0590	52.2590

Table 3.3: Before-After-Control-Impact (BACI) analysis comparing CPUEs and post exotic salmonid removal in Sevenmile Creek to the Mosquito River, Pictured Rocks National Lakeshore, Michigan. Factors: Time Period=years used for before and after (located in column one), Treatment=Stream (treatment and control).

BACI	**Factors**	**BKT**	**BKT YOY**	**COH YOY**	**STH**	**STH YOY**
#1	Time Period	0.427	0.226	0.127	0.056	<0.001
2004-2005 = Before	Treatment	0.012	0.636	<0.001	<0.001	<0.001
2010=After	Time Period x Treatment	0.827	**<0.001**	**0.002**	0.107	**<0.001**
#2	Time Period	0.049	0.143	0.396	0.216	0.004
2004-2008 = Before	Treatment	0.024	0.045	<0.001	<0.001	<0.001
2010=After	Time Period x Treatment	0.398	**0.005**	**<0.001**	0.407	**<0.001**
#3	Time Period	0.002	0.184	<0.001	<0.001	0.412
2008 = Before	Treatment	0.369	<0.001	<0.001	<0.001	<0.001
2010=After	Time Period x Treatment	0.103	0.835	**<0.001**	**<0.001**	**<0.001**
#4	Time Period	0.101	0.005	0.002	<0.001	<0.001
2004-2005 = Before	Treatment	0.107	<0.001	<0.001	<0.001	0.528
2011=After	Time Period x Treatment	0.385	**0.002**	0.268	0.786	0.956
#5	Time Period	0.486	0.004	0.249	0.359	0.008
2004-2008 = Before	Treatment	0.244	<0.001	<0.001	<0.001	0.676
2011=After	Time Period x Treatment	0.736	**<0.001**	0.071	**0.004**	0.927
#6	Time Period	0.258	0.036	0.096	<0.001	0.367
2008 = Before	Treatment	0.956	0.013	0.006	<0.001	0.923
2011=After	Time Period x Treatment	0.442	**<0.001**	**0.049**	**<0.001**	0.658

Table 3.4: Age classes and length distribution of brook trout captured in 2004, 2005 and 2008 to 2011 in Sevenmile Creek, Pictured Rocks National Lakeshore, Michigan.

	2004	2005	2008	2009	2010	2011
BKT >200mm	33	34	16	8	27	37
% of Adults >200mm	8.5%	6.0%	5.2%	3.3%	4.7%	9.6%
BKT Adult	386	566	305	243	578	387
BKT YOY	198	98	128	256	86	477
BKT Total	584	664	433	499	664	864

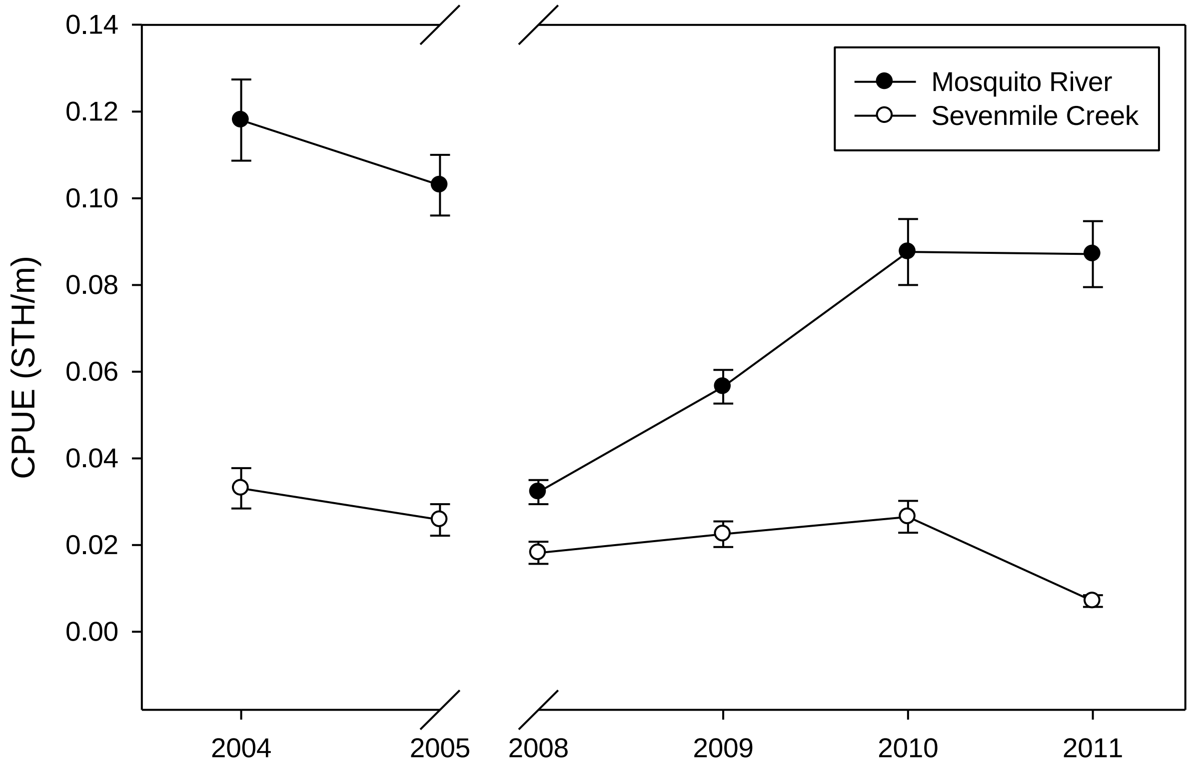

Figure 3.1: Age-one steelhead mean CPUE (fish/m) in the Mosquito River and Sevenmile Creek before (2004-2005), during (2008-2010), and after (2011) and exotic salmonid removal in Sevenmile Creek, Pictured Rocks National Lakeshore, Michigan. The bars on CPUEs are standard errors.

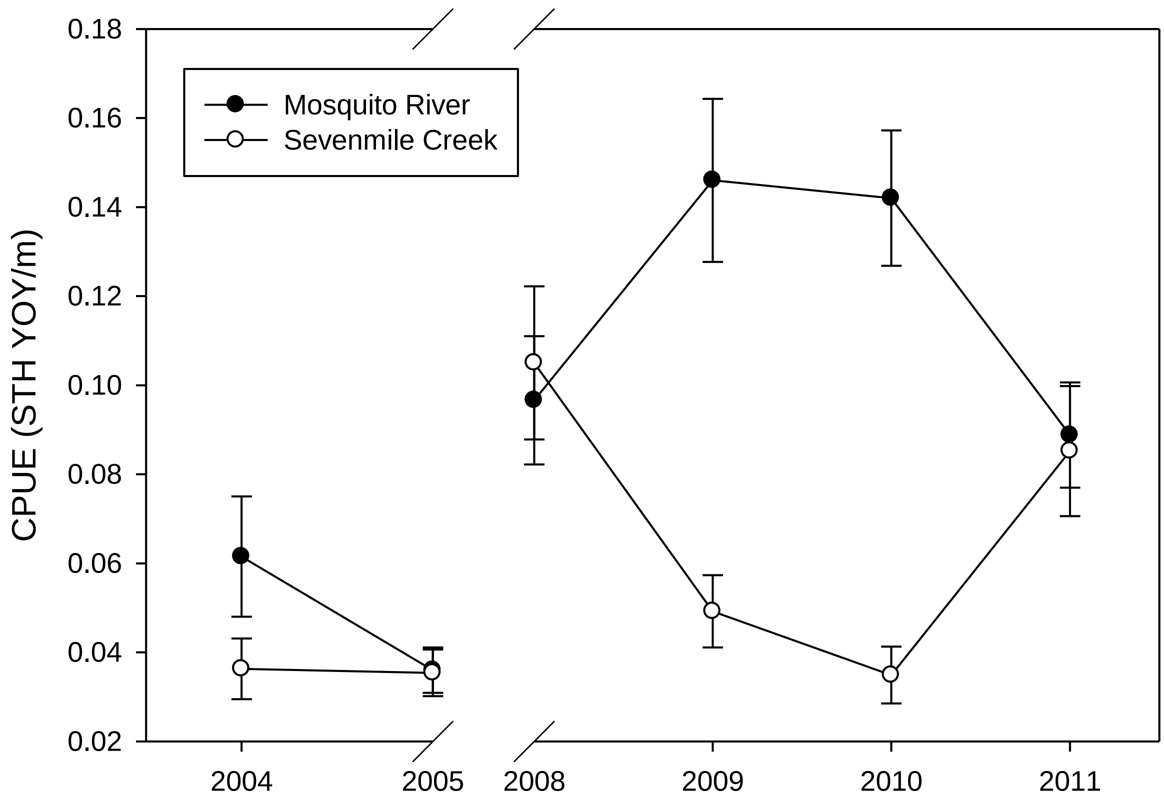

Figure 3.2: Young-of-year steelhead mean CPUE (fish/m) in the Mosquito River and Sevenmile Creek before (2004-2005), during (2008-2010), and after (2011) and exotic salmonid removal in Sevenmile Creek, Pictured Rocks National Lakeshore, Michigan. The bars on CPUEs are standard errors.

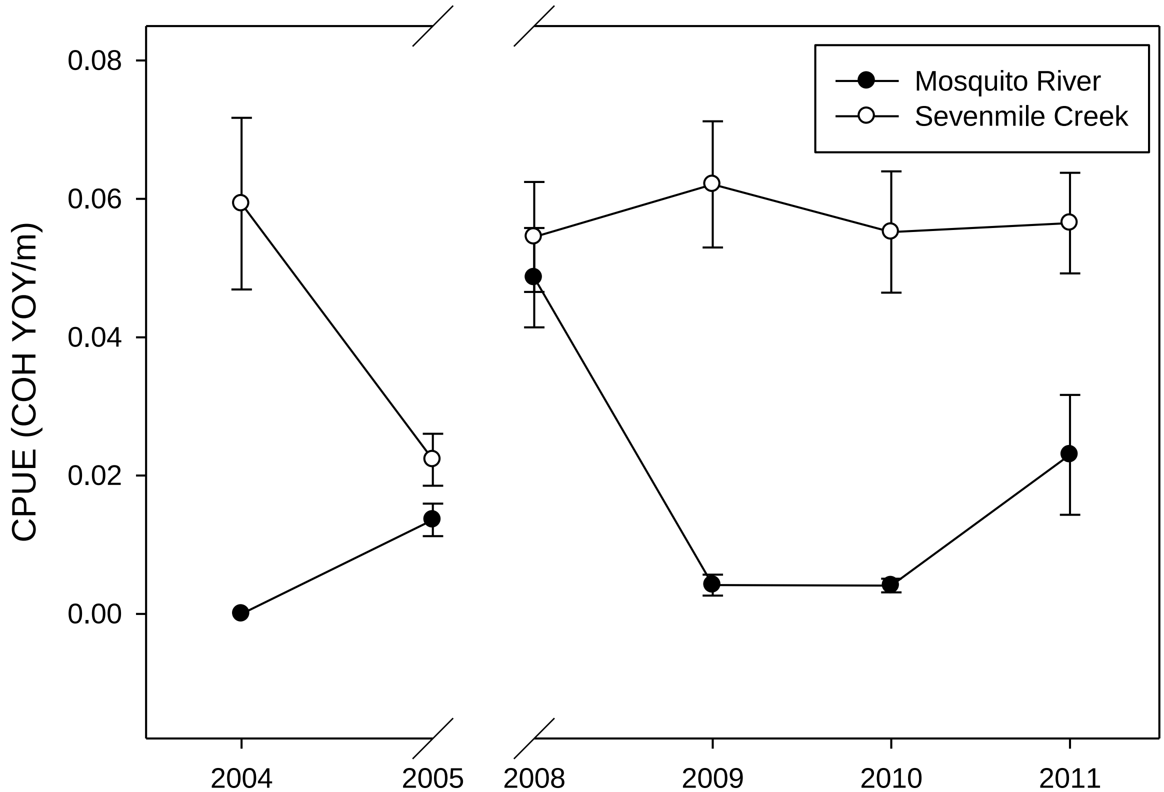

Figure 3.3: Young-of-year coho mean CPUE (fish/m) in the Mosquito River and Sevenmile Creek before (2004-2005), during (2008-2010), and after (2011) and exotic salmonid removal in Sevenmile Creek, Pictured Rocks National Lakeshore, Michigan. The bars on CPUEs are standard errors.

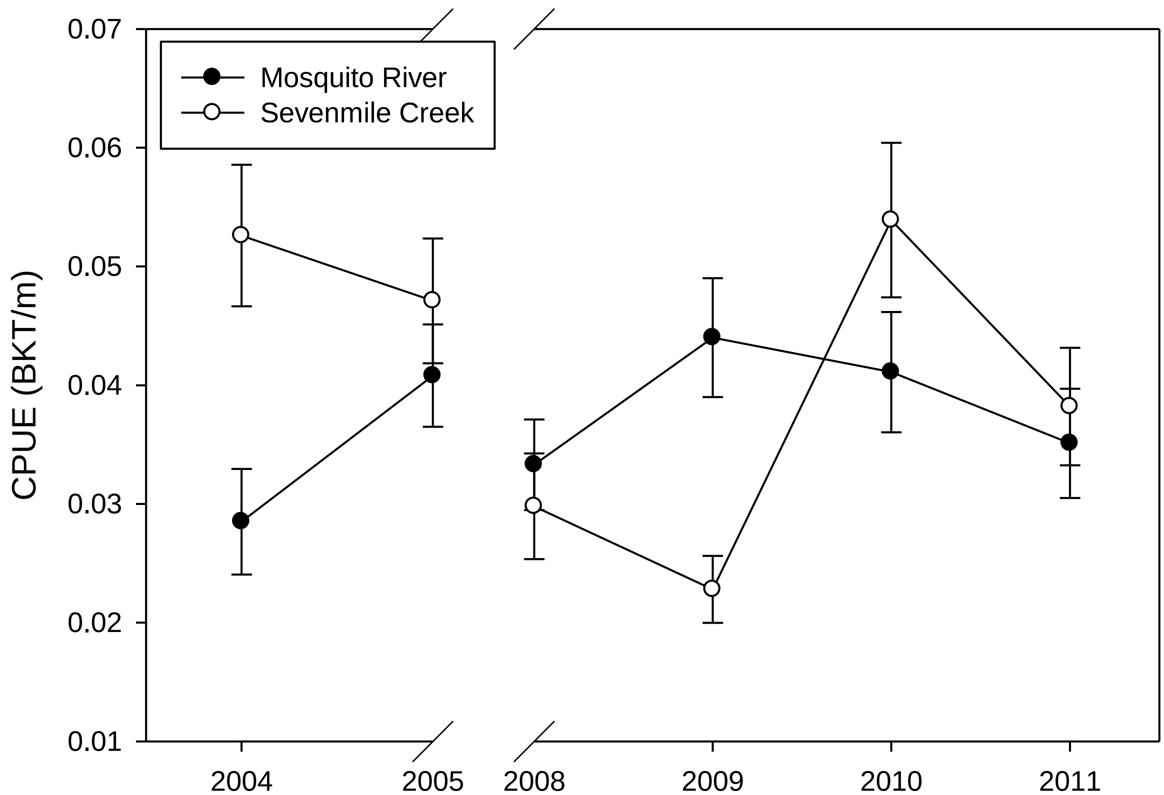

Figure 3.4: Adult brook trout mean CPUE (fish/m) in the Mosquito River and Sevenmile Creek before (2004-2005), during (2008-2010), and after (2011) and exotic salmonid removal in Sevenmile Creek, Pictured Rocks National Lakeshore, Michigan. The bars on CPUEs are standard errors.

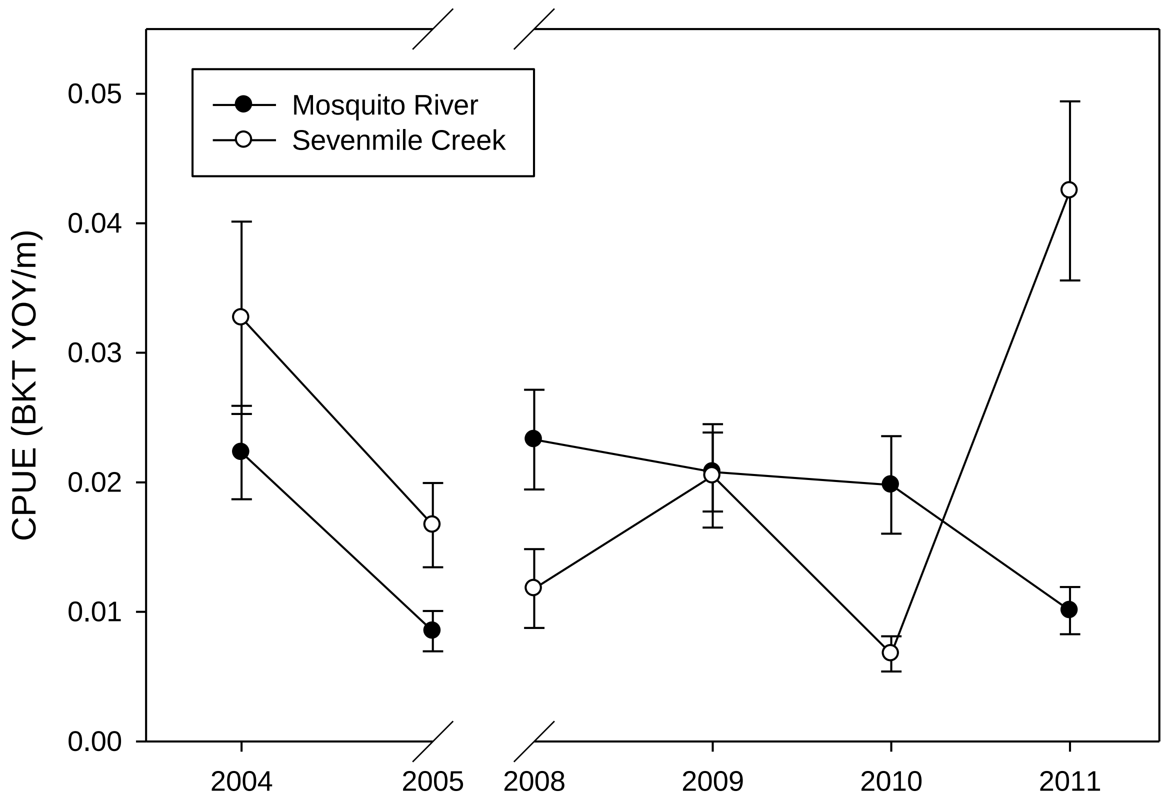

Figure 3.5: Young-of-year brook trout mean CPUE (fish/m) in the Mosquito River and Sevenmile Creek before (2004-2005), during (2008-2010), and after (2011) and exotic salmonid removal in Sevenmile Creek, Pictured Rocks National Lakeshore, Michigan. The bars on CPUEs are standard errors.

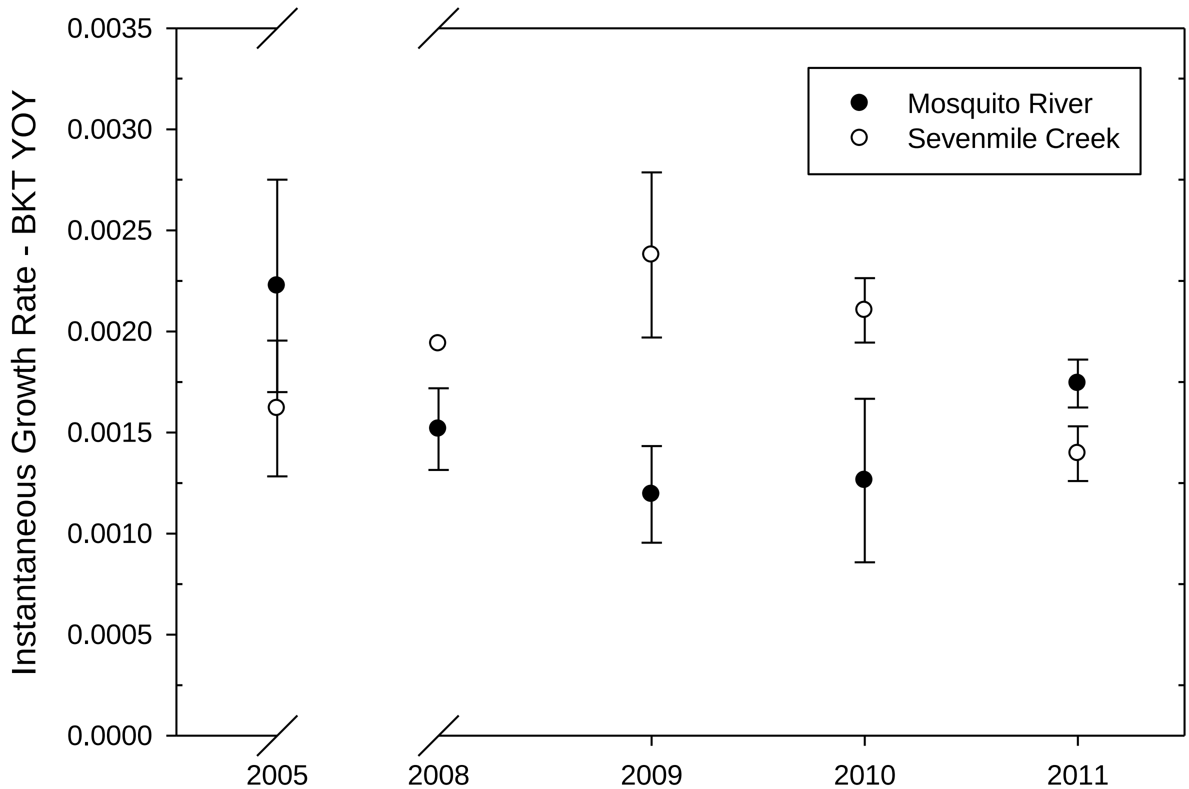

Figure 3.6: Instantaneous growth rates of young-of-year brook trout from August to November for 2005 and 2008 to 2011 in the whole study area in Sevenmile Creek and the Mosquito River, Pictured Rocks National Lakeshore, Michigan. The bars on the instantaneous growth rates are standard errors.

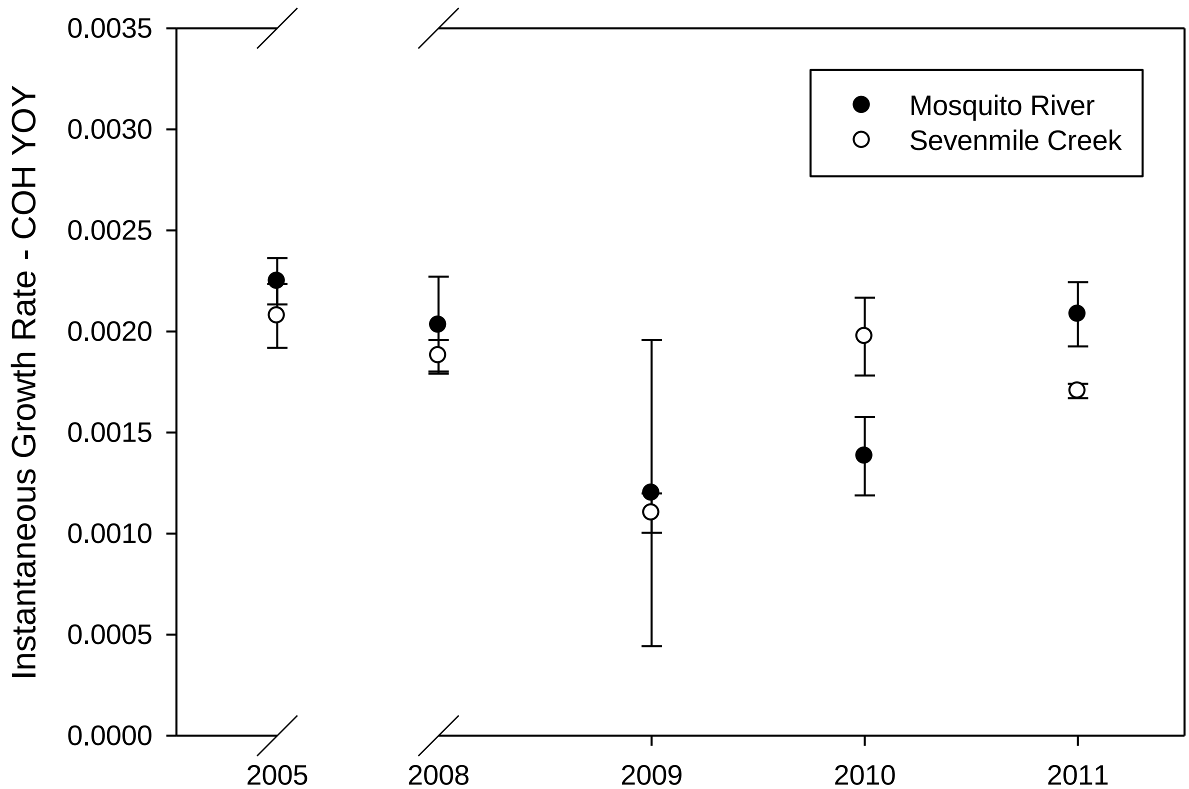

Figure 3.7: Instantaneous growth rates of young-of-year coho salmon from August to November for 2005 and 2008 to 2011 in the whole study area in Sevenmile Creek and the Mosquito River, Pictured Rocks National Lakeshore, Michigan. The bars on the instantaneous growth rates are standard errors.

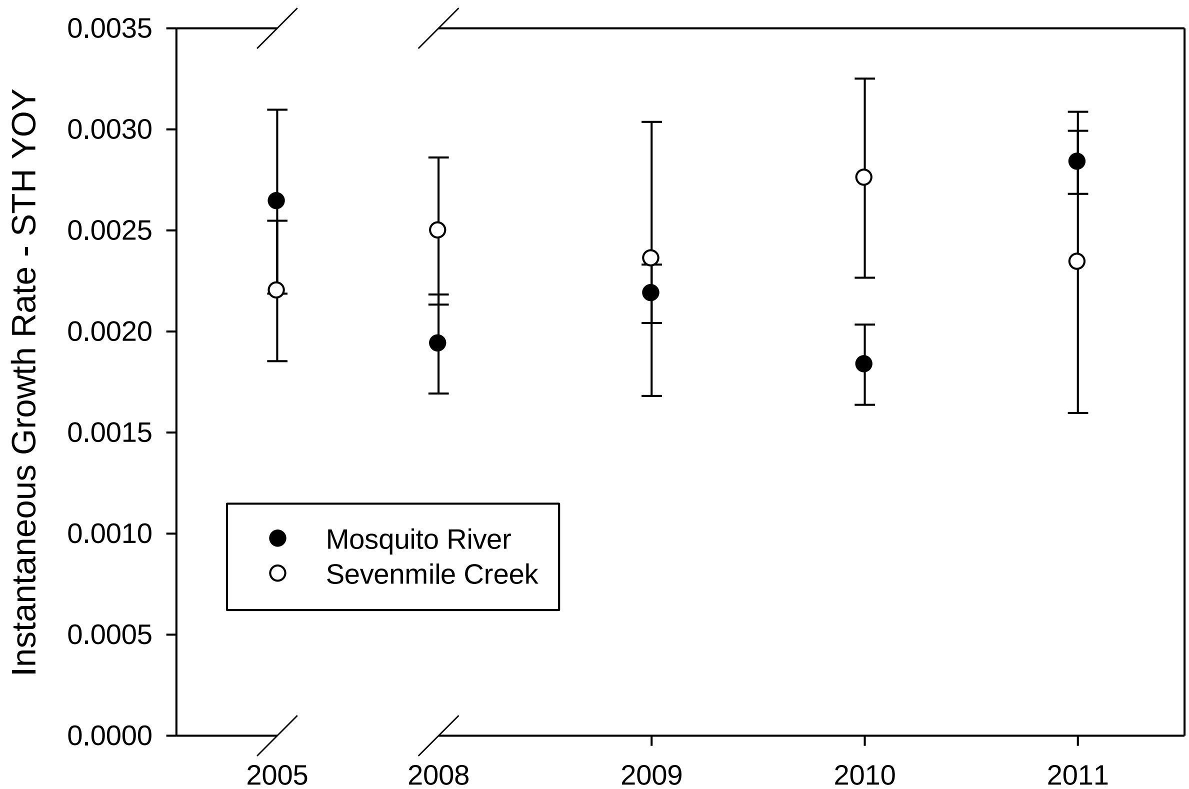

Figure 3.8: Instantaneous growth rates of young-of-year steelhead from August to November for 2005 and 2008 to 2011 in the whole study area in Sevenmile Creek and the Mosquito River, Pictured Rocks National Lakeshore, Michigan. The bars on the instantaneous growth rates are standard errors.

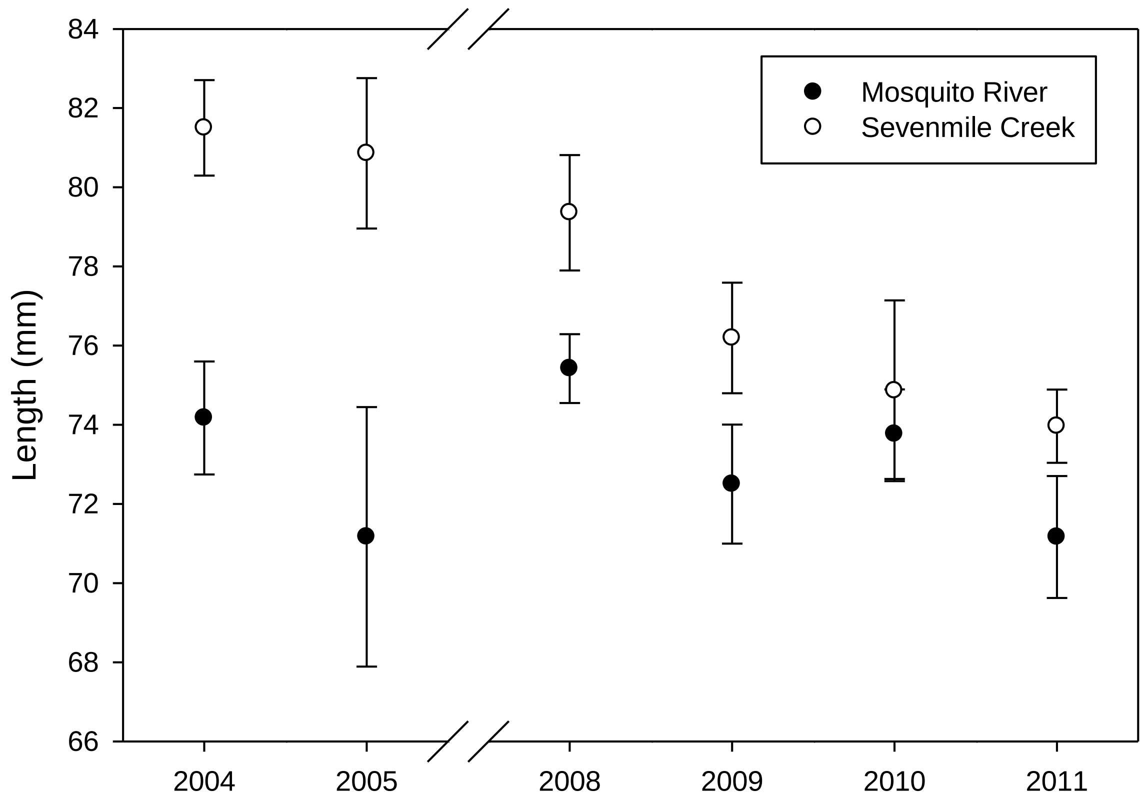

Figure 3.9: August mean length of young-of-year brook trout for 2004, 2005 and 2008 to 2011 in the whole study area in Sevenmile Creek and the Mosquito River, Pictured Rocks National Lakeshore, Michigan. The bars on the mean lengths are standard errors.

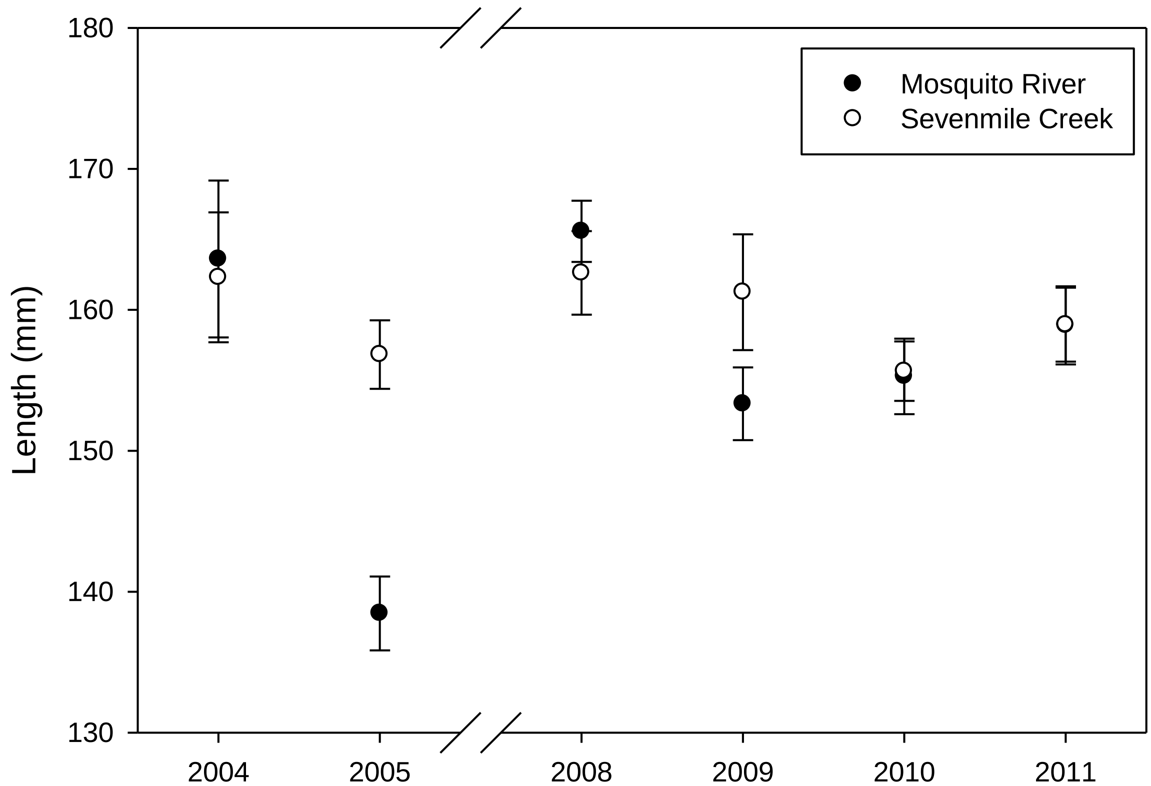

Figure 3.10: August mean length of adult brook trout for 2004, 2005 and 2008 to 2011 in the whole study area in Sevenmile Creek and the Mosquito River, Pictured Rocks National Lakeshore, Michigan. The bars on the mean lengths are standard errors.

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CHAPTER 4: CONCLUSIONS

During this project non-native salmonid densities in the treatment stream were reduced and a positive response in young-of-year brook trout densities was observed. Also, the small population of brook trout in the reference stream was monitored with multiple methods and recommendations were developed to aid in assessing the size of a small population.

These recommendations were developed to aid in monitoring the size of a small population of stream trout. When the POPAN model is used, our recommendation is to increase capture probability by marking one day prior to recapture (Rosenberg and Dunham 2005) to minimize of the influence of factors that can reduce capture probabilities, such as environmental or behavioral changes. Additionally, it is recommended to increase the capture probability to greater than 0.20 in order to increase precision of the population estimate. Pine et al. (2003) recommended increasing capture probability to as high as possible to obtain the most precise estimates. To help minimize seasonal variability in capture probability when using the depletion estimator or relative abundance estimates, sampling at the same time of year is recommended. If the depletion method is used, it is important to be cognizant that the population estimates will likely be negatively biased. The depletion method likely underestimated the population in this study and it has been shown to underestimate population size in other studies (Peterson and Cederholm 1984; Riley and Fausch 1992; Rodgers et al. 1992; Rosenberger and Dunham 2005). It is helpful to estimate population size at least two times per year when using the depletion method or relative abundance to minimize the chances of differential capture probabilities biasing estimates. Relative abundance estimates seemed to be the least likely to be biased because data was collected over the entire season and apparent changes in capture probability could be identified because catches could be compared across and within years.

Over three years 5,320 non-native salmonids were physically removed from Sevenmile Creek. Three BACI analyses showed significant effects of the removal on the young-of-year steelhead density, which was reduced by 67% from the beginning of the removal. An increase in age-one steelhead density, as observed in the reference stream, was likely suppressed by the removal in the treatment stream, which was significant in three BACI analyses. Prior to spawning, 157 adult coho were trapped in 2010 from Sevenmile Creek, a 233% increase from the previous year, although there was little change in young-of-year coho density in the treatment stream.

The brook trout population increased during the removal, which could possibly have been a result of the reduced competition with non-native salmonids. Redistribution within the stream was observed by the final years of the project. An increase in brook trout density was observed in the lower reaches, which were previously occupied almost exclusively by non-native salmonids, suggesting that brook trout were being forced into subpar habitat. The year following the project, brook trout young-of-year density significantly increased by 260% compared to the beginning of the project. The decrease in both the juvenile steelhead density and the number of spawning adult coho may have contributed to a significant increase of the young-of-year brook density in 2011. The reduction in the competition for vital resources may have led to an increase in egg quality of the adult brook trout population resulting in healthier or more viable eggs increasing larval survivorship. Also, due to the successful trapping of adult coho in 2010, there was a reduction in competition for spawning habitat leading to greater nest success. These results suggest that interspecific competition was occurring between salmonid species in Sevenmile Creek, possibly limiting brook trout densities, and fisheries managers may want to take this into account when managing these populations.

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APPENDIX A

Northern Michigan University

Continuing Education & Sponsored Programs
1401 Presque Isle Avenue
Marquette, MI 49855-5325

MEMORANDUM

January 15, 2008

TO: Dr. Jill Leonard
Department of Biology

FROM: Cynthia A. Prosen, Ph.D. CR
Dean of Graduate Studies & Research

RE: **Application to use Vertebrate Animals**

Application # IACUC 066
Approval Period: 8/15/2007-8/15/2010

The Institutional Animal Care and Use Committee have approved your application to use vertebrate animals in research, "Examination of resident and coaster brook trout response to exotic salmonid removal in Sevenmile Creek, Pictured Rocks National Lakeshore".

If you have any questions, please contact me.

kjm

Telephone: 906 227-2102 ■ FAX: 906 227-2108
E-mail: conteduc@nmu.edu ■ Web site: www.nmu.edu/ce

Northern Michigan University

Continuing Education
1401 Presque Isle Avenue
Marquette, MI 49855-5301

MEMORANDUM

October 12, 2010

TO: Dr. Jill Leonard
Department of Biology

FROM: Terrance Seethoff, Ph.D. RS
Dean of Graduate Studies & Research

RE: **Application to use Vertebrate Animals**
Modification to Application # IACUC 152
Approval Period: 01/25/2009-05/31/2011

The Institutional Animal Care and Use Committee, has approved the modifications to your application to use vertebrate animals in research, "Examination of resident and coaster brook trout response to exotic salmonid removal in Sevenmile Creek Pictured Rocks National Lakeshore".

If you have any questions, please contact me.

kjm

E-mail: conteduc@nmu.edu ■ Web site: www.nmu.edu/ce

Northern Michigan University

Continuing Education
1401 Presque Isle Avenue
Marquette, MI 49855-5301

MEMORANDUM

August 27, 2010

TO: Dr. Jill Leonard
Department of Biology

FROM: Terrance Seethoff, Ph.D. TLS
Dean of Graduate Studies & Research

RE: **Application to use Vertebrate Animals**
Application # IACUC 152
Approval Period: 08/15/2010-08/14/2013

The Institutional Animal Care and Use Committee have approved your application to use vertebrate animals in research, "Examination of resident and coaster brook trout response to exotic salmonid removal in Sevenmile Creek, Pictured Rocks National Lakeshore".

If you have any questions, please contact me.

kjm

E-mail: conteduc@nmu.edu ■ Web site: www.nmu.edu/ce