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PRODUCTIVE CAPACITY OF THE BETSIE RIVER WATERSHED FOR STEELHEAD SMOLTS

By

Tammy Joann Newcomb

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ABSTRACT

PRODUCTIVE CAPACITY OF THE BETSIE RIVER WATERSHED
FOR STEELHEAD SMOLTS

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Steelhead (*Oncorhynchus mykiss*), a Pacific salmonid and popular sportfish, were introduced into Michigan rivers in 1864. Since their initial stocking, they have become naturalized in many rivers and produce wild offspring. Knowledge of the abundance of wild juvenile steelhead and the factors influencing their mortality is important when considering the management options for harvest and sustainability of the fishery. Temperature can be an important influence on the distribution, growth, and mortality of juvenile steelhead. The Betsie River in northwestern Michigan is a marginal trout stream and temperatures regularly exceed the growth and lethal limits for juvenile steelhead. Objectives for this study included: estimating the number of juvenile steelhead leaving the Betsie River and comparing the results of three monitoring techniques, describing the distribution, density, and mortality of juvenile steelhead in the Betsie River and its tributaries, describing the thermal regime in the watershed, and developing temperature models to evaluate the effects of management alternatives for lowering instream temperatures in the summer. I used mark-recapture, visual observation, and time-lapse videography to monitor and estimate the number of juvenile steelhead emigrating from the Betsie River each spring, 1993-1996. I used stratified random sampling (18 sites) to determine the distribution and abundance of juvenile steelhead in the Betsie River and its tributaries. In July and October, 1993-1996, I used electrofishing and multiple pass depletion methods to estimate the number of juvenile steelhead. Continuous recording data loggers monitored hourly stream temperatures at 7 sites in the watershed. Mean daily temperatures were calculated to evaluate the thermal regime and empirical and physical process models were developed to predict instream temperature in the Betsie River and evaluate dam removal or hypolimnetic siphon options. The number of juvenile steelhead leaving the Betsie River ranged from 1,143 ± 266 in 1996 to 2,151 ± 286 in 1993. Mark-recapture over-estimated the number of fish leaving the river while visual observations and time-lapse videography gave similar results. Juvenile steelhead were found in the greatest densities in the tributaries and lower main channel Betsie River. In terms of total abundance, approximately 50% of the juvenile fish were found in the tributaries which comprised only 11% of the total channel area considered. The density of juvenile steelhead was negatively correlated with daily mean maximum summer temperature and mortality was positively correlated with daily mean minimum winter temperature. The number of juveniles emigrating in spring was negatively correlated with prior winter severity. Removal of a dam in the Betsie River will result in lower instream temperatures, but they would still exceed the optimal growth and preferred temperatures for juvenile steelhead. A hypolimnetic siphon would reduce instream temperatures to within the preferred and optimal growth limits for the main channel.

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INTRODUCTION

Steelhead in Michigan

Steelhead (*Oncorhynchus mykiss*) were introduced into Michigan rivers in the late 1800's and anglers widely accepted them as a popular sportfish. The first steelhead eggs were brought by private interests from California to Bay City, Michigan in 1876 and the fry were stocked in the Au Sable River. The Michigan Fish Commission received 2,000 eggs from the McCloud River in California in April 1880 that were incubated in the Pokagon Fish hatchery (Latta 1974). Steelhead have been stocked annually throughout the Great Lakes since their introduction, but many populations are naturalized and their progeny have the opportunity to contribute significantly to the sport fishery (Latta 1974, Seelbach 1989, Rand et al. 1993, Peck 1992). In the Great Lakes, adult steelhead provide multiple and diverse fishing opportunities that include offshore fishing in the summer, pier and surf fishing in the spring, summer, and fall, and instream fishing for driftboat and wading anglers during the spawning migrations in the spring and fall.

Naturalized steelhead production (the product of instream spawning) has occurred in varying and largely unquantified levels since their introduction. Some streams, such as the Little Manistee River are not stocked but they experience large and highly variable returns of adults (Figure 1). In contemporary fisheries science in the Great Lakes, managers and researchers are investigating these naturalized

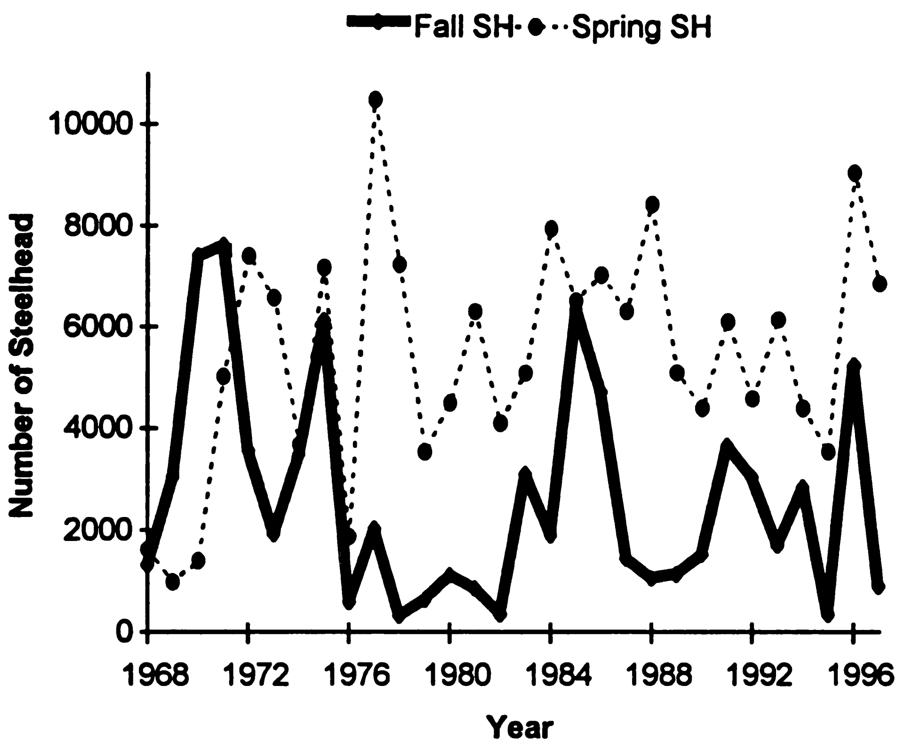

Figure 1. Fall and spring adult steelhead returns to the Little Manistee River weir, Michigan, 1968-1997 (Data obtained from the Michigan Department of Natural Resources, Fisheries Division, Lansing).

populations and consider them a valuable resource for the maintenance and sustainment of the steelhead fishery. Furthermore, watershed capacities to produce steelhead parr are increasing as a result of recent improvements in water quality, efforts to decrease erosion and sand bedload, and instream habitat improvements (Seelbach 1989, Rand et al. 1993). Production of juvenile steelhead can be very large in streams of good quality trout habitat that contain cold thermal regimes, such as the Little Manistee River (Seelbach 1993). Little is known however, about smolt production from streams that contain habitat with limiting thermal regimes for salmonid species. Many streams in southern Michigan that are classified as marginal host large runs of adult steelhead, but very little is known about the quantities in the resulting smolt cohort. Determination of the contributions of marginal streams to the lakewide steelhead fishery and knowledge of the conditions which may inhibit natural production and successful recruitment of steelhead is necessary for the management of a sustainable population of Great Lakes steelhead.

Life History of Steelhead

Adult steelhead begin immigrating into Great Lakes tributary streams in late autumn (October-November) through the early spring (February-April) after 1 to 3 years of residence in the lake (Biette et al. 1981, Seelbach 1993). Hypothetically, fall-run adults that overwinter in the river may spawn earlier than the early-spring immigrating adults. In Great Lakes tributary watersheds, the spawning period lasts for 4 to 7.5 weeks (Biette et al. 1981). Steelhead typically home and enter their natal stream to spawn. In Great Lakes tributaries, studies indicate that rates of straying vary but can be as low as 10% (Biette et al. 1981). Unlike coho (*O. kisutch*) and chinook salmon *(O. tschawchtya)*, post-spawning survival is probable and steelhead may return to the lake and return to spawn in subsequent years (Seelbach 1993). Throughout the Great Lakes, the proportion of repeat spawning ranges from 16% to 61% and the majority of the repeat spawners are 2nd year spawners (Biette 1981, Seelbach 1989, Seelbach and Miller 1993).

Adult female salmonids construct saucer shaped redds in gravel of 20-30 mm in diameter, velocities greater than 15 cm/s, and depths between 5 and 50 cm. An individual female may construct one or more redds and may deposit eggs more than once per spawning season (Crisp and Carling 1989). Fecundity varies with the length of the female and location of the fish stock. In general, a 57.1 cm fish will produce between 2,424 and 4,964 eggs (Bulkley 1967). The eggs of rainbow trout require approximately 50 days at 7.3°C or 25 days at 12°C to fully incubate and hatch (Bardach et al. 1972). Water temperature can influence year-class strength throughout a watershed as different hatching times may result from a difference in temperatures at redd sites throughout the watershed (Bardach et al. 1972).

During stream residency, steelhead parr use a variety of microhabitat conditions diurnally, seasonally, and with increased size. Within the stream environment, steelhead parr are subject to temperature extremes in both summer and winter and to competition for space and food. Certain time periods during stream residency may act as bottlenecks for their recruitment to the next year class. Water temperatures required for parr growth range between 13 and 19°C with an optimum range of 17 to 19°C (Wismer and Christie 1987). No growth occurs above 22°C or below 8°C and water temperatures greater than 24°C are lethal (Hokanson et al. 1977).

Newly emerged steelhead fry inhabit shallow water, such as that found along the stream margin or in shallow riffles, and they use crevices in logjams and boulders as cover (Hartman 1965, Cambpell and Neuner 1984). In the fall, steelhead parr seek shelter in log jams and boulders (Hartman 1965). With increased size, steelhead parr use deeper and faster water (Everest and Chapman 1972, Waite and Barnhart 1992, Campbell and Neuner 1984). Age-0 parr prefer depths of less than 0.15 m and velocities less than 0.15 m/s, whereas age-1 parr use depths from 0.60 to 0.75 m and velocities 0.15 - 0.3 m/s (Everest and Chapman 1972).

Seelbach (1987) documented consistent production of age-1 steelhead parr in the Little Manistee River, however the numbers of outmigrating juvenile steelhead varied from 10,000 to 72,000 fish per year. One hypothesis regarding the variability in yield of smolts was in relation to environmental conditions, particularly winter severity. Overwinter mortality estimates were 13 to 90 % and a significant relationship (r^2 = 0.54) existed between the severity of the presmolt winter and the number of returning adults. Further evidence suggests that instream habitat for overwintering parr may be very critical to the success of individual year classes (Bjornn et al. 1991). Critical winter habitat for steelhead includes groundwater fed side channels and boulder or rubble strewn stream margins with water less than 30 cm deep and very slow water velocities (Everest and Sedell 1983, Campbell and Neuner 1984). Below 11°C, rainbow trout parr tend to hide in the interstices under and between rocks and reduce their activity and their selection of instream habitat may be crucial (Hearn and Kynard 1986, Everest and Sedell 1983). In Oregon streams, parr occupied low energy environments that were less than 0.15 m deep and had water velocities that were less than 0.15 m/s. Parr were more concentrated in the favorable habitats in winter than in summer (Everest and Sedell 1983). In Michigan streams, where substrates are predominantly composed of sand or small gravel, overwinter habitat in the form of cobbles and boulders may be limited. Furthermore, low velocity areas that are not sufficiently buffered by groundwater may experience ice and freezing conditions lethal to juvenile steelhead.

Steelhead parr spend 1 to 3 years in the river during which they grow to a size suitable for smoltification, typically greater than 180 mm (Seelbach 1993). During smoltification, juvenile steelhead metamorphose, becoming longer and thinner and turning a solid silver color as their parr marks diminish (Wedemeyer et al. 1980, Hoar 1988). Smolts begin to school and emigrate nocturnally from the river to the lake or ocean (Brege et al. 1996). Temperature, photoperiod and discharge are the primary environmental factors that influence the timing and rate of smoltification and the outbound movement of smolts from a watershed. The majority of smolt movement occurs right after dusk. For example, from 6 years of observations of smolt movement through the John Day Dam on the Columbia, Brege et al. (1996) found rates of nocturnal movement to range from 66.2 to 87.4 % (mean 77.9%) of the total steelhead smolts passing the dam. Similarly in Michigan, Stauffer (1968) found that over 93% of migrating juvenile steelhead moved between 17:00 and 08:30 hours and peaks of migration occur between May 21-June 30 in Lake Michigan tributaries (Stauffer 1968, Biette et al. 1981). Steelhead smolts begin migrating when the water temperatures approach 10°C (Stauffer 1968) and smoltification behavior begins to decline when water temperature warms to greater than 13°C (Zaugg et al. 1972, Zaugg and Wagner 1973, Zaugg 1981). Increases in stream discharge usually result in increased rates of smolt movement and decreases in smolt travel time (Berggren and Filardo 1993, Hvidsten and Johnsen 1993, Dempson and Stansbury 1991).

Size of steelhead smolts may also influence their survival and timing of emigration. Steelhead smolts range in length from 120 mm to over 200 mm (Ward et al. 1989, Seelbach 1993), and numerous studies report greater survival rates for larger smolts (Seelbach et al. 1994, Ward et al. 1989). Survival rates for smolts greater than 180 mm are much higher (17- 33%) than 140 mm smolts (<5%)(Ward et al. 1989).

The majority of naturalized steelhead observed from rivers throughout the Great Lakes smolted at the age of 2, and age 1 smolts were more prevalent than age 3 smolts (Biette et al. 1981, Seelbach 1993).

Because smolt movement is predictable and concentrated through a short window of time during the spring the annual event gives fishery managers an opportunity to examine the cohort before they disperse into the vast lake environment. Estimation and observation of smolt emigrations provide an excellent opportunity to obtain information on the future potential of the fishery as well as the productive capacity of the watershed.

Monitoring Techniques For Estimating Smolt Production

Several methods have been used with varied success to monitor and estimate the abundance of emigrating smolts. Typical means of enumeration include: 1) trapping either a portion or the entire run (Raymond 1979, Davis et al. 1980, Seelbach et al. 1994), 2) videotaping through counting windows, 3) using a combination of trapping and mark-recapture (Dempson and Stansbury 1991), or 4) use of resistivity counters (Beach 1978, Reddin et al. 1992). Several of the trapping methods require the use of fyke-nets, which can be problematic with debris loads, or construction of a low head dam or similar apparatus which can be costly to implement (Davis et al. 1980, Seelbach et al. 1984). Fence counts which involve constraining fish movement through an area where they can be counted are often used as reliable methods with adult escapement estimates (Cousens et al. 1982). Success of the methods also varies with discharge: typically as flow increases, enumeration becomes less efficient (Raymond 1979).

One method of trapping, the inclined screen smolt trap is a successful method (Seelbach et al. 1984, Dubois et al. 1991). The method incorporates a dam or weir in its design. An inclined screen is attached to the sill of the dam or weir and guides fish into a floating barge trap. Drawbacks to this approach are that it requires an instream structure, its efficiency is limited at higher flows, and it requires handling of the physiologically sensitive smolts.

Methods most desirable for enumerating smolts would feature minimum labor requirements, cost effectiveness, reliable and accurate counts or estimates, and would not require handling or other stresses to smolts. Time-lapse videography is a technological approach that is effective for enumerating returning adult salmonids (Dexter and Ledet 1994). As the cost for the required equipment decreases (Collins et al. 1991), a time-lapse videographic approach may be valuable for estimating the number of emigrating smolts. Advantages to the development of such a system include: collection of a permanent record, minimizing stress to the smolts, and making more efficient use of labor than in trapping efforts.

Temperature Modeling

Temperature plays an integral part in the function of a stream biota and can have many interacting influences on organisms throughout a stream. As water temperature increases, the metabolic rates of instream organisms also increase (Adams and Breck 1990). Properties of water such as viscosity, sediment load, and concentrations of both nutrients and dissolved oxygen are influenced by temperature.

Factors such as groundwater inflow, geomorphology, and stream shading influence the thermal regime found in streams and these influences can be predictably modeled. Using a stream-reach temperature model, Pajak (1992) discovered that stream side shading had the most influence on maximum temperature in Wisconsin streams and estimated that 75% shading would maintain summer temperatures in a stream suitable for brook trout. Temperature models can also be used in diagnosing problem areas in a watershed. In the John Day River basin in Oregon, Li et al. (1994) used a model to illustrate that stream temperatures were 5-11°C warmer due to increased levels of insolation as a result of a loss of vegetation from cattle grazing.

Stream hydrology as well as land uses, shading, bank stability and severity of the winter are all contributing factors to the overall physical suitability of a stream for steelhead production. Furthermore, accumulative factors throughout the watershed such as tributary influences may provide better indications for a watershed's capacity to produce steelhead.

Stream temperature models are of two types, empirical models and physical process models. Empirical models are statistically based and result from tests for relations between measured variables (such as air temperature) and instream water temperatures, usually through simple or multiple regression (Barthalow 1989). Empirical models are useful for inferring water temperatures when they are not measured but tend to be limited in predicting responses to the system in the event of a physical change, such as in stream width or changes in shading. Furthermore, in some cases, when such physical changes occur, new data often must be collected for the development of a new empirical model.

Physical process models incorporate the physical parameters that act to produce the instream water temperatures. These models usually incorporate measures to predict the heat flux within the modeled reach (Theurer et al. 1984). Conceptually, the heat flux model can be presented as:

Net Heat Flux = SR + AR + VTS + EV + CV + CD + F - WBR

where,

SR = solar radiation,
AR = atmospheric radiation,
VTS = vegetative and topographic shading,
EV = evaporation,
CV = convection,
CD = conduction,
F = friction, and
WBR = water back radiation.

Physical process models are powerful in that they can be used to predict likely changes that would occur with changes in the river morphology or thermal dynamics. Models such as this are often used to make management decisions in regulated systems with hydropower dams. These models can be used to alter dam operations regarding flow and temperature to benefit the fishery. In unregulated systems, these models can be used to evaluate management alternatives for the thermal environment such as the effects of increased shading or changes in channel morphology. Once developed for a stream or watershed, a physical process model can be used in a pro-active approach for predicting changes in water temperature as a result of changes in land or water use.

Betsie River Watershed

The Betsie River is located in northwestern Michigan (86°N, 44°W) (Figure 2) and flows through Grand Traverse, Benzie, and Manistee Counties draining a watershed of 67,126 hectares. The river originates as an unregulated surface outflow from Green Lake near Interlochen and flows approximately 79 km from its source to where it empties into Betsie Lake and ultimately into Lake Michigan at Frankfort.

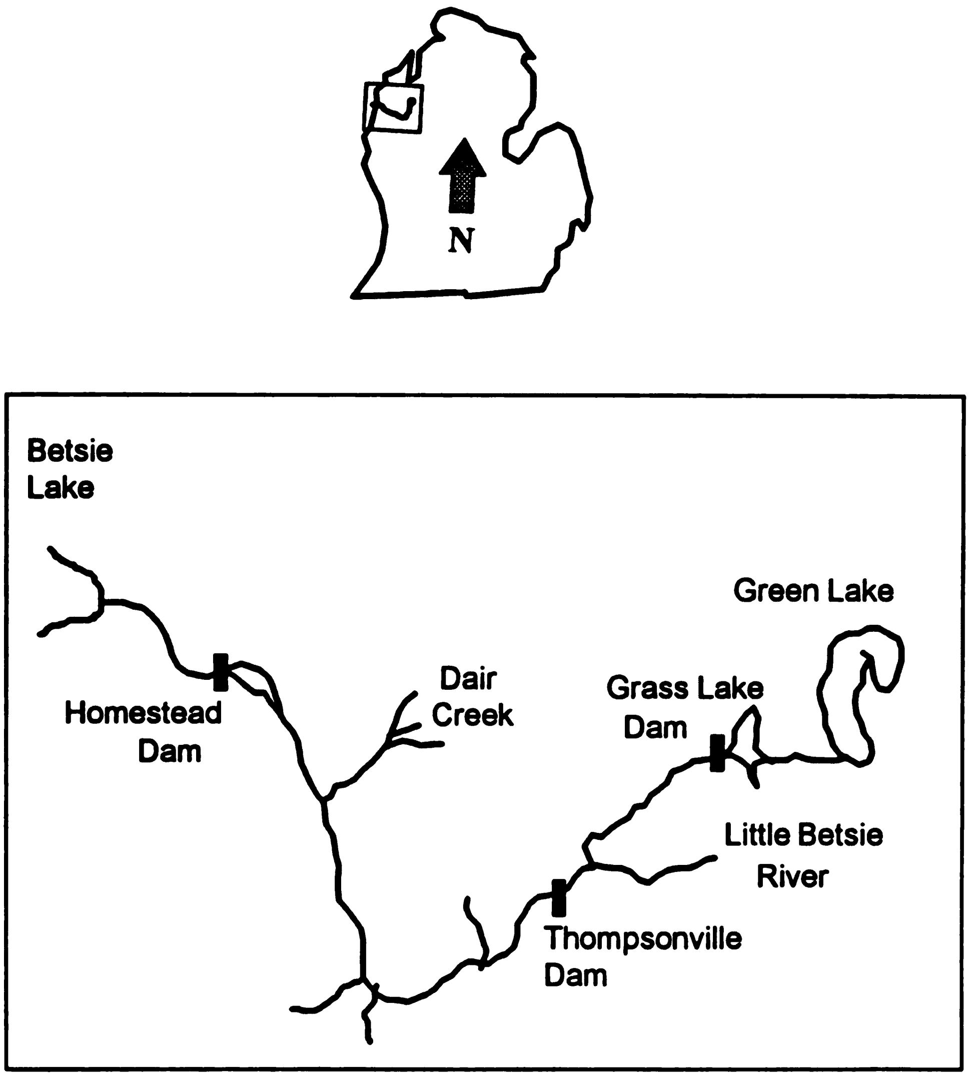

Figure 2. Location and map of the Betsie River watershed in northwestern Michigan.

Land use in the Betsie River Watershed is hilly, with several glacial moraines present and the watershed is ranked as one of the highest in the state in percent composition of glacial outwash, a highly permeable material. The surficial geology in the Betsie River watershed consists of glacial outwash (52%) comprised of thick deposits (73.8 - 220.6 m) of stratified sand and gravel (Hendrickson and Doonan 1972, Gooding 1995). Watershed soils are comprised of 55% sandy soil type, which is less than the nearby Boardman and Manistee watersheds which contain more of the sandy soil type (71% and 86% respectively) (Gooding 1995). Soils in the Betsie River watershed are comprised of 20% loamy soil type, a mixture of sand, silt and clay which gives it a larger capacity to hold water than the sandy soil type Gooding 1995).

Land use in the watershed consists of orchards, tree farms, abandoned farms, forests and several small towns and villages. Between 1966 and 1994, total forested land increased with the watershed presently containing 57% forest. In 1994, 52% of the forested area was in deciduous forest and 5% of the forested land was in coniferous forest (Gooding 1995, Letherberry 1994, Jakes 1982, Pfeifer and Spencer 1966). Approximately 32% of the land is under agricultural use and only 0.5% is considered as urban (Gooding 1995). The watershed receives 78.7 cm of mean annual precipitation, and a mean annual temperature of 7.2°C, mean July temperature of 20.5°C (Gooding 1995).

The Betsie River is classified as a marginal trout stream by the Fisheries Division of the Michigan Department of Natural Resources because of limiting high temperatures that exist during the summer (Wicklund and Dean 1958). Historically, there were 3 dams in the Betsie River that impounded the river flow and altered water quality by increasing water temperature. Two of these dams also precluded potamodromous fish runs (such as steelhead), prevented access to the largest coldwater tributary in the watershed and probably warmed the water of the river. Two of the three dams no longer exist or are modified to allow fish passage and the river, along with the largest coldwater tributary, is available to potamodromous fish.

The first 7 km of river below Green Lake are impounded by Grass Lake Dam which is 1.2 m high and floods nearly 465 hectares of land (Carbine 1945). This dam was established in 1945 based on the location of a logging dam present in the early 1800's. Grass Lake Dam was designed to maintain a head of 1.2-1.8 m and to flood the Grass Lake area and several small spring-fed tributaries. The motive for impounding this area was to increase duck hunting and northern pike (*Esox lucius*) fishing opportunities in Grass Lake (Wicklund and Dean 1958).

Fourteen kilometers downstream and 0.5 km above the site of Thompsonville Dam, the Little Betsie River joins the main channel of the Betsie River. Previous reports (Wicklund and Dean 1958, Bullen 1972) identified Thompsonville Dam as the site that contributed most to the warming of the mainstem of the Betsie River and proposed its removal. The dam maintained a head of 3.1 m and the impounded area covered approximately 5.7 hectares. As early as 1958, Thompsonville Dam pond was full of sand and silt that had eroded from unstable banks in the upstream channel. Temperatures recorded at the outflow of the dam reached 27°C. Placement of this dam also precluded potamodromous fish from entering the largest cold-water tributary in the watershed, the Little Betsie River. Thompsonville Dam fell into disrepair and collapsed in March, 1989, after a tremendous rainstorm. Remains of this dam still stand along the stream margins downstream of the dam. Raw, eroding streambanks in the former impoundment area appear to contribute large amounts of sand and silt to downstream areas as the river cuts a new channel. Failure of the dam resulted in heavy sedimentation in downstream spawning gravel and increased the number of eroding banks. Presently a sand trap is in place downstream of Thompsonville Dam but no efforts were made to stabilize the former impoundment area.

Homestead Dam, 35 km downstream of Thompsonville Dam, was converted to a lamprey weir in 1972. Potamodromous fish can pass over the dam, but the design prevents passage of other migratory fish such as sea lamprey (*Petromyzon marinus*), catostomids, and burbot (*Lota lota*). A small, shallow backwater area still remains upstream as a result of the weir. Temperature surveys do not indicate significant warming in this backwater.

Limited information exists on the Betsie River fishery. However, both the salmon and steelhead fisheries are regarded highly by anglers. From 1985 - 1988, a limited (2 months per year) creel study collected information on the returning steelhead fishery. Estimates of harvest ranged from 2,600 ± 1,401 steelhead in 1986 to a low of 1,129 ± 1,119 steelhead in 1988 (Rakoczy and Rogers 1987, 1988, 1990).

Dissertation Overview

The Betsie River provides an opportunistic setting to obtain information on production of naturalized steelhead smolts from a marginal watershed, effects of temporal and spatial thermal variation on steelhead parr throughout a watershed, and to evaluate management alternatives to thermal limitations. The general objectives of my research were to:

1) Estimate the abundance of naturalized steelhead from the Betsie River watershed by developing a cost effective, labor efficient, time-lapse video system.

2) Investigate the production of parr throughout the watershed, identify and quantify production areas, and identify the environmental limitations such as temperature that influence population dynamics.

3) Determine the current thermal regime throughout the watershed and evaluate management alternatives, such as a change in source water or removal of Grass Lake Dam, and their potential effects on altering the thermal regime and making it more conducive to steelhead trout production.

Each of these objectives is presented as an independent manuscript in chapters 1-3, and the final chapter is a summary, integration, and application of the results regarding the steelhead population in the Betsie River.

Chapter 1

EVALUATION OF ALTERNATE METHODS FOR ESTIMATING NUMBERS OF OUTMIGRATING STEELHEAD SMOLTS

Abstract

The annual emigration from streams tributary to the Great Lakes by migratory juvenile salmonids provides an opportunity for fishery managers to estimate the abundance of the smolting cohort. This information is valuable for modeling recruitment potential to the fishery and for estimates of the returning broodstock. The purpose of this study was to compare the results of several techniques for estimating the number of steelhead *Oncorhynchus mykiss* smolts emigrating from the Betsie River in northwestern Michigan. I also compared the production of steelhead smolts from this river with other rivers in the state. In May and June, 1993-1996, I monitored smolts by use of visual observations, time-lapse videography and mark-recapture to estimate the number of steelhead smolts migrating past a lamprey weir located 18 km upstream of the river mouth. In 1993 and 1994, two observers counted smolts passing over the weir for 20 minutes out of each hour from 21:00 to 05:00 hours. In 1995 and 1996, black and white video cameras and time lapse videocassette recorders continuously monitored the passing smolts through the night. Three reviewers watched the videotapes and counted the number of smolts passing over the weir from 21:00 to 05:00 for each night, also collecting counts from hourly 20 minute samples similar to the visual observations in 1993 and 1994. A subsample of videotapes were reviewed

a second time to measure the count variation between the reviewers. In all four years, a constriction weir was constructed every 5th night during the smolt run to capture emigrating fish and to quantify species composition and the origin of steelhead (hatchery or wild). Steelhead smolts comprised 30-61% of all the emigrating salmonid juveniles sampled. Other species present were brown trout *Salmo trutta*, coho salmon *Oncorhynchus kisutch*, and chinook salmon *Oncorhynchus tshawytscha*. Wild steelhead comprised 12-52% of the steelhead smolts. Mark-recapture estimates and 95% confidence intervals for steelhead smolts (hatchery and wild combined) ranged from 13,837 (12,583-15,215) in 1985 to 56,661 (46,036-69,703) in 1993 and were 2-9 times greater than the estimates from visual observation and videography. However, the results from time-lapse videography and visual observation were similar and gave the most reliable estimates for steelhead smolts produced from the watershed. Estimates of steelhead smolt numbers from observation methods ranged from 2,198 (± 512) in 1996 to 9,645 (± 1,111) in 1994. The Betsie River produced fewer wild steelhead smolts (12-22/ha) than other streams studied in Michigan .

Introduction

Steelhead were introduced to Michigan streams in 1876 and since then, have established naturalized populations (Latta 1974, Biette et al. 1981). Great Lakes steelhead retain the life history characteristics of their Pacific counterparts and spend 2-3 years growing to sexual maturation in one of the Great Lakes. In late autumn and early spring, adult steelhead return to their natal stream and spawn in the spring. Juveniles reside in the stream for 1-3 years before smolting and emigrating back to the lake (Biette et al. 1981, Seelbach 1993).

The smolting period is a sensitive and critical phase in steelhead life history. Wild smolts reach a size of 160-200 mm before the necessary changes for smoltification, including the ability to hypoosmoregulate and migratory behavior, occur (Wagner 1974, Seelbach 1987). Although Great Lakes steelhead do not encounter saltwater, they follow the same smoltification process as steelhead in Pacific streams: juvenile steelhead smolts lose their parr marks, become silvery, and emigrate in water temperatures of 10-15°C during the months of April-June (Biette et al. 1981).

Estimation of steelhead smolt numbers is an important component in the management of steelhead fisheries. The abundance of the smolting cohort, coupled with expected survival rates, can aid in forecasting recruitment to the fishery and give indices of expected returns for escapement and stock sustainment (Raymond 1988). Furthermore, because steelhead parr are resident in the stream for more than one year and require good water quality (high levels of dissolved oxygen, suitable water temperatures for survival and growth, low sedimentation rates, an adequate forage base, and instream habitat for refuge and cover), the number of smolts produced annually may provide a relative index of the overall health of a watershed for cold water species.

Many methods have been used to monitor emigrating smolts, and each method has produced various levels of success in providing accurate measures of emigrants. Smolts are difficult to monitor because they migrate at night. In addition, spring high flows from snowmelt and rain can be problematic for some methods, especially in larger rivers. Methods to monitor and estimate smolt abundance include: 1)mark-recapture for trapping efficiency (Seelbach 1987, Dempson and Stansbury 1991) or for total run estimates (Macdonald and Smith 1980), 2) trapping a portion or all of the smolt run with fyke nets and box traps (Davis et al. 1980, Seelbach and Miller 1993) or inclined screen smolt traps (Wagner et al. 1963, Lister et al. 1969, Seelbach et al. 1984, DuBois et al. 1991, Seelbach and Miller 1993), 4) electronic fish counters (Appleby and Tipping 1991), and 5)camera monitoring (Cousens et al. 1982). When a dam is present, collection methods at turbine intake gatewells or bypass collection methods are often used (Raymond 1979, Giorgi and Sims 1987, Peven and Hays 1989, Peven et al. 1994).

Problems inherent with various sampling methods for smolts include gear function at varying flows, debris loading of nets and screens, trap avoidance by smolts, and mortality associated with handling stress and gear design. Modifications to the inclined-screen smolt trap make this a promising method for a variety of applications. However, trapping efficiency can be highly variable or very low, introducing large amounts of error to the estimates and a low head dam must be in place to use inclined screen traps (Dubois et al. 1991, Seelbach 1993). Fyke nets quickly become clogged with debris (Davis et al. 1980). Electronic counters often require that fish pass through a very constricted area, and are most accurate when fish are of a uniform length (Appleby and Tipping 1991). Although uniform length and monitoring in a controlled and confined area are applicable to hatchery settings, both of these conditions are not usually attainable for monitoring smolts in a field setting.

Time-lapse videography has been used successfully to record escapement of adult salmonids (Dexter and Ledet 1994, Hatch et al. 1994). The advantages of this non-intrusive methodology include establishment of a permanent record, low labor requirements, ability to review and automate counts, and fairly low cost of equipment as videographic technology advances (Collins et al. 1991). In the Dexter and Ledet (1994) study, fish were viewed through windows in a fish ladder which allowed for identification of species, but also involved an additional construction cost. Irvine et al. (1991)

developed a computerized video-camera system to monitor coho smolts by use of cameras trained on tunnels through which smolts passed. Although this early prototype of the method was promising and provided the additional benefit of smolt length measurements, it was a highly controlled situation and not generally applicable to typical field settings.

Steelhead have been naturalized in the Great Lakes Region for over 125 years, yet few estimates of emigrating smolts exist. Yearly production of smolts is known to vary widely in high quality trout streams in Michigan (Seelbach 1987, Seelbach and Miller 1993), but little is known about their production from watersheds with marginal trout habitat. In southern Michigan, for example, rivers tributary to Lake Michigan which are stocked with steelhead receive large returns of adults, a fraction of which are wild. Although some natural reproduction is known to occur, it has yet to be quantified (Seelbach 1987). The tendency of rivers in this region to warm significantly during the summer makes them marginal or unsuitable for salmonid production (Seelbach et al 1994).

The objectives of this study were to:

1) develop a time-lapse, surface-view videographic system for monitoring emigrating smolts

2) estimate the number of steelhead smolts emigrating from the Betsie River by use of direct observation, time-lapse videography, and mark-recapture and compare results among the three methods, and

3) compare the number of emigrating smolts in the Betsie River watershed with other rivers in Michigan.

Study Site

The Betsie River watershed is tributary to northern Lake Michigan. The main channel of the Betsie River is 79 kilometers long and the watershed drains an area of

67,100 hectares (Figure 3). Land use in the watershed consists primarily of tree farming, some fruit farms and a portion is owned by the State of Michigan and managed as a State Forest. Approximately 15-35% of the watershed is in agricultural use, 4-14% is covered by coniferous forest, and 45-60% of the watershed is deciduous forest (Gooding 1995). The river also flows through two small municipalities, with little development or industry along the river. Sandy soil types dominate a large portion of the watershed (35-66%) (Gooding 1995) and glacial moraines are dominant landscape features throughout the watershed. Several permanent flowing springs and seeps supply groundwater directly into the river, especially in the mid-reaches of the watershed. The two largest tributaries that empty into the Betsie River are Dair Creek and Little Betsie River, and both of these are spring-fed.

The Betsie River has no permanent U.S. Geological Survey gaging stations. To determine discharge, I measured river stage weekly by use of temporary staff gages from 1993-1996. Staff gages were installed and rated according to standard hydrological methods (Gordon et al. 1992, Appendix A). The mean May-June measured discharge ranged from 4.3 to 11.3 cubic meters per second (m^3/s) during this period. Stream gradient is generally low, and averages 0.002% (range 0.001 - 0.005%). Historical water temperature data showed that the Betsie River warms significantly in the summer months (Wicklund and Dean 1958) and for this reason, the river is considered a marginal trout stream. The fish community of the Betsie River includes rockbass *Ambloplites rupestris*, creek chub *Semotilus atromaculatus*, common shiner *Notropis comutus*, hornyhead chub *Nocomis biguttatus*, blacknose dace *Rhinichthys atratulus*, mottled sculpin *Cottus bairdi*, brown trout *Salmo trutta*, and juvenile steelhead. Dair Creek and Little Betsie River fish communities consist of cold

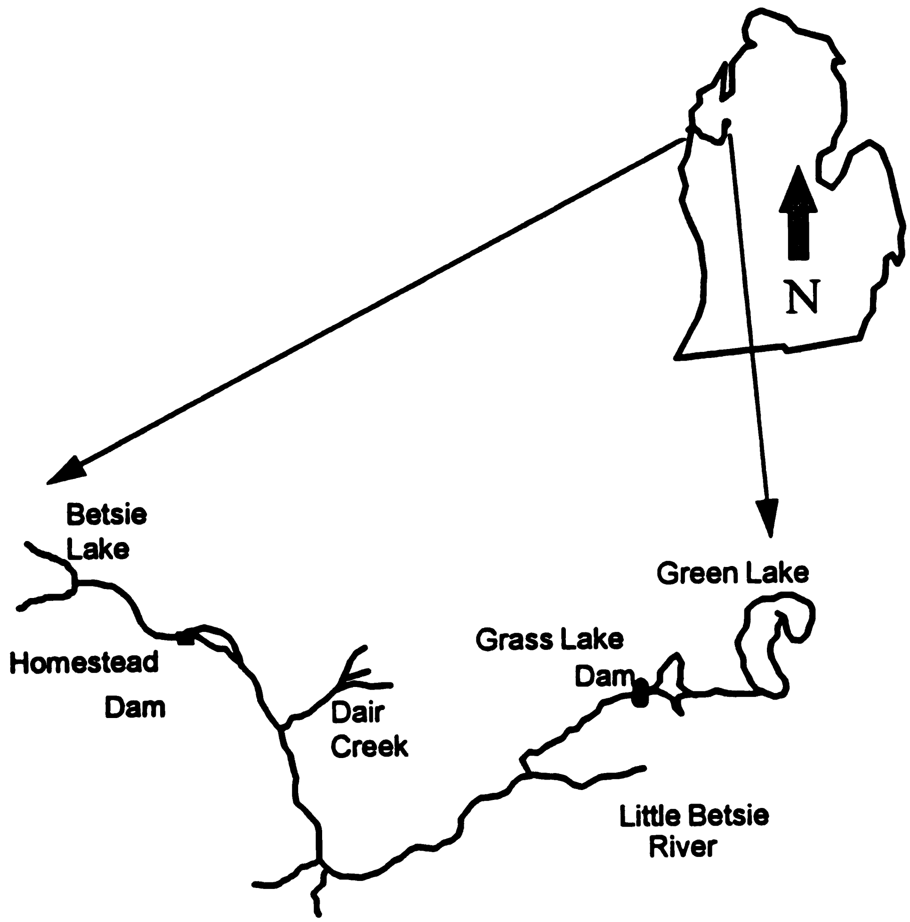

Figure 3. Location and map of the Betsie River watershed in northwestern Michigan.

water species, including brown trout, slimy sculpin *Cottus cognatus*, mottled sculpin, steelhead, and coho salmon parr. Although the mainstem Betsie River is classified as marginal for trout, it supports a popular fishery for returning salmon and steelhead (Rakoczy and Rogers 1987, 1988, 1990).

Since 1991, angling interest organizations in cooperation with the Michigan Department of Natural Resources, Fisheries Division, have raised juvenile steelhead in a hatchery adjacent to the Betsie River and linked to the river by a small flowing spring. Yearling steelhead were released volitionally, beginning in April. Most of the smolts from this hatchery were clipped with either a right ventral (1993-1995) or right pectoral clip (1996). The number of steelhead released from this facility has ranged from 20,000 (1991) - 55,000 (1996).

I monitored emigrating smolts at a lamprey weir located in the lower river reach, 17.7 km upstream from Lake Michigan. Formerly the site of Homestead Dam, a low-head hydroelectric dam, the site now consists of a lamprey weir which is divided into two sills (10 m wide each) that are staggered, with one sill approximately 5 m further downstream than the other (Figure 4). The weir has a head of approximately 2 m which is stepped into 1 m jumps on the most upstream sill and there are no steps on the downstream sill.

Methods

Beginning the first week in May and ending in late June, 1993-1996, I monitored outmigrating smolts by use of three methods: visual observation, time-lapse videography, and mark-recapture. In 1993 and 1994, two observers counted smolts passing over the downstream sill of the weir for 20 minutes out of every hour from

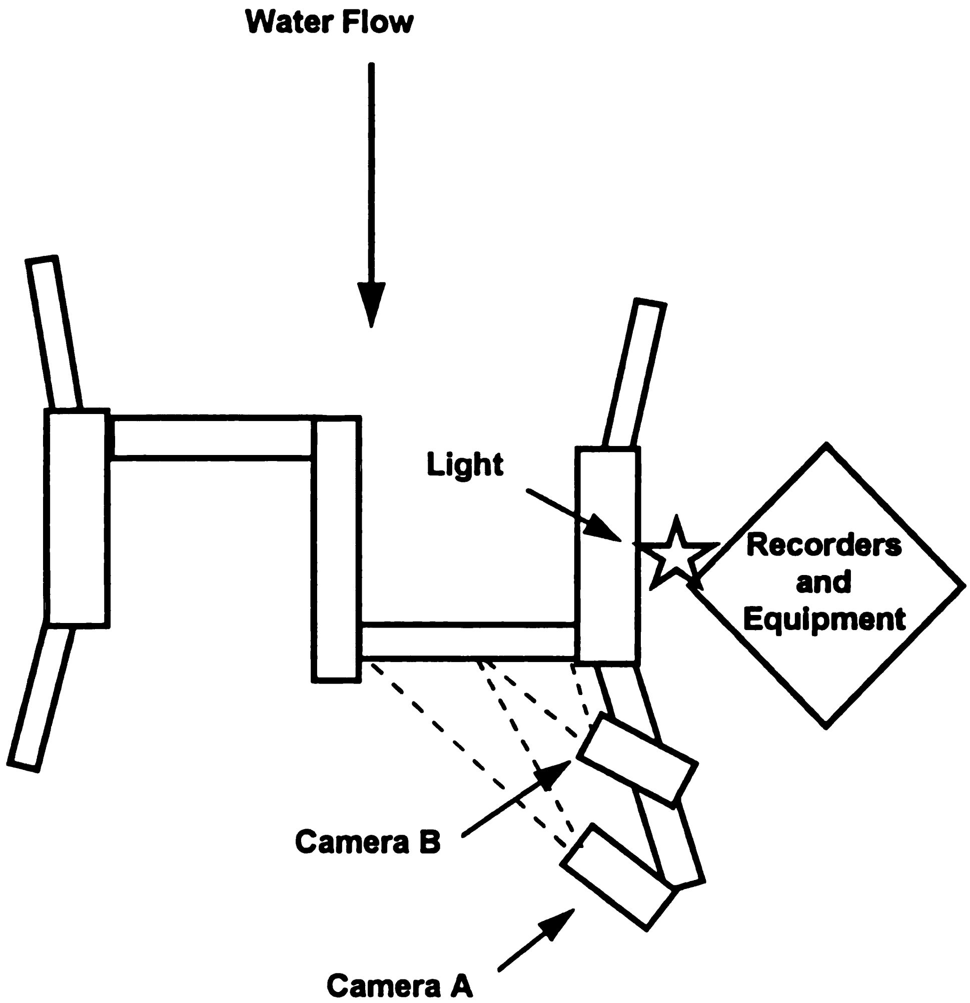

Figure 4. Diagram of equipment used at the Homestead Lamprey Weir to monitor emigrating smolts in the Betsie River, Michigan.

21:00 to 05:00 hours, approximately dusk to dawn . To check for daytime emigration, I randomly sampled during daytime hours approximately every other day during the smolt run. Visual observations in 1993 and 1994 indicated that the majority (80%) of the smolt movement occurred over the most downstream sill. Therefore, the visual and video observations were concentrated on this sill and estimates were adjusted accordingly. During night sampling, each viewer used a hand-held light to illuminate the area and monitored half of the 10 m wide sill to count the smolts, which were readily visible passing over the weir. In 1995 and 1996, visual observations were corroborated with camera observations and similar 20 minute observations were obtained from the videotapes. For each night of observation, a mean 20-minute sample was calculated and multiplied by 24 to calculate a nightly estimate for 8 hours of smolt movement. Because a few nights were not sampled due to equipment malfunction, weather interference, and flooding, a weekly estimate for the smolt run was determined from:

$$C_\tau = (N/n) * (\sum C_i + EF) \qquad \text{(Equation 1)}$$

and,

$$Var\,(C_\tau) = N(N-n)1/(N-1)^* \sum (C_i - C_{avg}) \qquad \text{(Equation 2)}$$

where,

C_τ = total number of smolts for the week,
N = total number of nights of smolt movement,
n = number of nights sampled by electrofishing or observation,
C_i = the total estimate for a single night,
EF = the number of smolts captured by electrofishing, and
C_{avg} = the average of all night estimates for the week.

The estimate of total smolts was adjusted by the proportion of steelhead captured by trapping in that week to obtain an estimate of steelhead smolts. All C_t and Var(C_t)

estimates were summed for an annual estimate of smolt numbers and variance of annual smolt numbers, respectively.

In addition to estimating smolt numbers with 20 minute observations, I evaluated estimates and the error rates associated with different frequencies of this type of sampling. I used the original estimate, derived from all days of sampling, as the estimate for 100% of observations through the duration of the smolt migration. Estimates and standard errors were then derived for each of the four years by using the values obtained from periodic sampling schedules including: every-other day (50%), every 3rd day (30%), and by obtaining a random sample of 80% and 20% of the nights during the smolt run in 1993-1996. Estimates and variances were calculated according to Equations 1 and 2.

In 1995 and 1996, I used surface-view, time-lapse videography to continuously monitor and record emigrating smolts. Two videocassette recorders (Sony SVT-100) stationed in housing next to the river and connected to two solid state black and white cameras (Vicon 12.5 mm F1.3 (A) and 25 mm F1.3 (B), housed in Environmental Enclosures (Pelco EH4500)) each monitored and recorded smolts passing over half of the sill (Figure 4). Five frames per second were recorded, resulting in 12 hours of actual smolt movement condensed to 1 hour of videotape. The sill of the dam was continuously illuminated with a 250 watt halogen light. Due to the remote location of the monitoring site, gasoline powered generators were used to provide electricity. Twice each night, recording was interrupted for less than 2 minutes to refuel the generators. Cameras were set up each night before dark and removed in the morning to prevent vandalism and theft. A small video monitor located in the housing next to the stream assisted with camera placement and focus. A fishing bobber, anchored at the middle of the sill aided in positioning each camera's view over half the sill and avoided overlap and duplicate recording of smolts.

Video recordings of smolts passing over the wild were watched by 3 reviewers using a professional editing videocassette recorder (Panasonic AG1980P). This equipment provided for frame by frame analysis and reverse playback of the videotapes. All reviewers worked together initially to develop a consensus on smolt images and counting methods. Fish were distinguished from flotsam on the videotapes by their fusiform shape and thickness, and subsurface presentation. In addition, smolt behavior while passing over the weir typically consisted of smolts backing downstream tail first and approaching the sill. Smolts also exhibited a slight bending to form a C-shape, movement that was slower and often lateral to the water current. Smolts usually were present in two or more frames. All tapes were reviewed from the beginning of the evening, 21:00 hours, to the end of the nightly smolt movement at 05:00 hours.

Camera A and camera B counts were summed to obtain an estimate of smolts for the night. Similar to the 20 minute estimates, a weekly estimate of smolts from the videotape counts was calculated by the equation:

$$C_{T(AB)} = N_{(A)} / n_{(A)} * \sum C_{i(A)} + N_{(B)} / n_{(B)} * \sum C_{i(B)} + EF \qquad \text{(Equation 3)}$$

where A and B designate the cameras of observation. Each weekly estimate was then adjusted by the proportion of steelhead in the captured smolts and weeks were summed to obtain an annual estimate for 1995 and 1996.

In consideration of the viewer subjective interpretation that was required to review videotapes and enumerate smolts, I conducted an error analysis from a stratified, random sampling of the videotapes. Because rates of smolt movement varied hourly, daily, and weekly through May and June, I randomly selected one night from each week during the smolt run for recounting. The selected night was divided into three equal time blocks, and I randomly selected one 30 minute period per time block to recount by a second reviewer. A paired t-test was used to test for differences between 2 viewers and variances of the daily estimates based on viewer interpretation were calculated according to the following equation:

$$Var\ (C_i) = (K\ C_i)^2 \qquad \text{(Equation 4)}$$

where,

C_i = total number of smolts observed for night i, and
K = the coefficient of variation (CV) among viewers.

The CV was calculated by:

$$CV_i = (sqrt(\Sigma\ ((X_i - X_{i(mean)})^2\ )/\ n_i - 1)/\ X_{i(mean)}) \qquad \text{(Equation 5)}$$

where,

X_i = the sum of the 2 reviewers counts for a night,
$X_{i(mean)}$ = the nightly average from the two reviews, and
n_i = number of reviews of night (i) = 2.

A mean CV was calculated for each camera from the nightly variance among viewers. The CV was assumed to be constant over time and a variance of the estimate of the numbers of smolts based on viewer interpretation was calculated by:

$$Var\ (C_T) = N(N-n)(1/N-1) * \Sigma\ (C_i - C_{(mean)})^2 + N/n * \Sigma\ Var\ (C_i) \qquad \text{(Equation 6)}$$

Separate estimates of smolt numbers and variances were determined for each camera and summed to determine a yearly smolt estimate and variance.

To determine species composition of the outmigrants and also to identify the origin of the steelhead smolts, I constructed a temporary constriction weir upstream of the lamprey weir approximately every 5th night during the outmigration period. The weir consisted of construction netting angled from both streambanks to a 3 m downstream opening (Figure 5). A large block net was placed in the opening to collect the smolts. Beginning at 21:00 hours, I used a 250 volt DC barge electrofishing unit within the block net every two hours until 05:00 hours to capture smolts. All fish were identified to species, measured for total length, weighed, and released below Homestead Dam. In addition, I checked all steelhead for either a right pelvic (1993-1996) or right pectoral (1996) fin clip and removed scale samples from a subsample of hatchery (fin-clipped) steelhead and all steelhead without a fin clip. I analyzed scale samples by inspecting the circuli surrounding the first annulus (Seelbach and Whelan 1988) to determine if the smolt was hatchery or wild.

I calculated a Peterson mark-recapture estimate of the steelhead smolts leaving the Betsie River watershed by assuming that the number of stocked hatchery fish each year comprised the marked population, and that clipped fish were equally likely to be recaptured in the constriction weir as the unmarked wild portion of the population (Ricker 1975).

Summer electrofishing surveys (Chapter 2) revealed that a large number of hatchery steelhead did not leave the watershed (Figure 6). I estimated the number of stocked steelhead that remained in the river (residualized fish) in July to determine the actual number of marked migratory steelhead that migrated downstream for the smolt mark-recapture estimates. I stratified the watershed into reaches based on hydrology, gradient, and thermal regime and randomly sampled equally in the reaches for a total of 21 sites. I used electrofishing and multiple-pass depletion methods to determine the number of residual steelhead (Zippen 1958). An estimate of residual steelhead abundance was calculated from the mean density of hatchery fish for the sample reach and the channel area in the reach. Because residual hatchery fish are known to remain

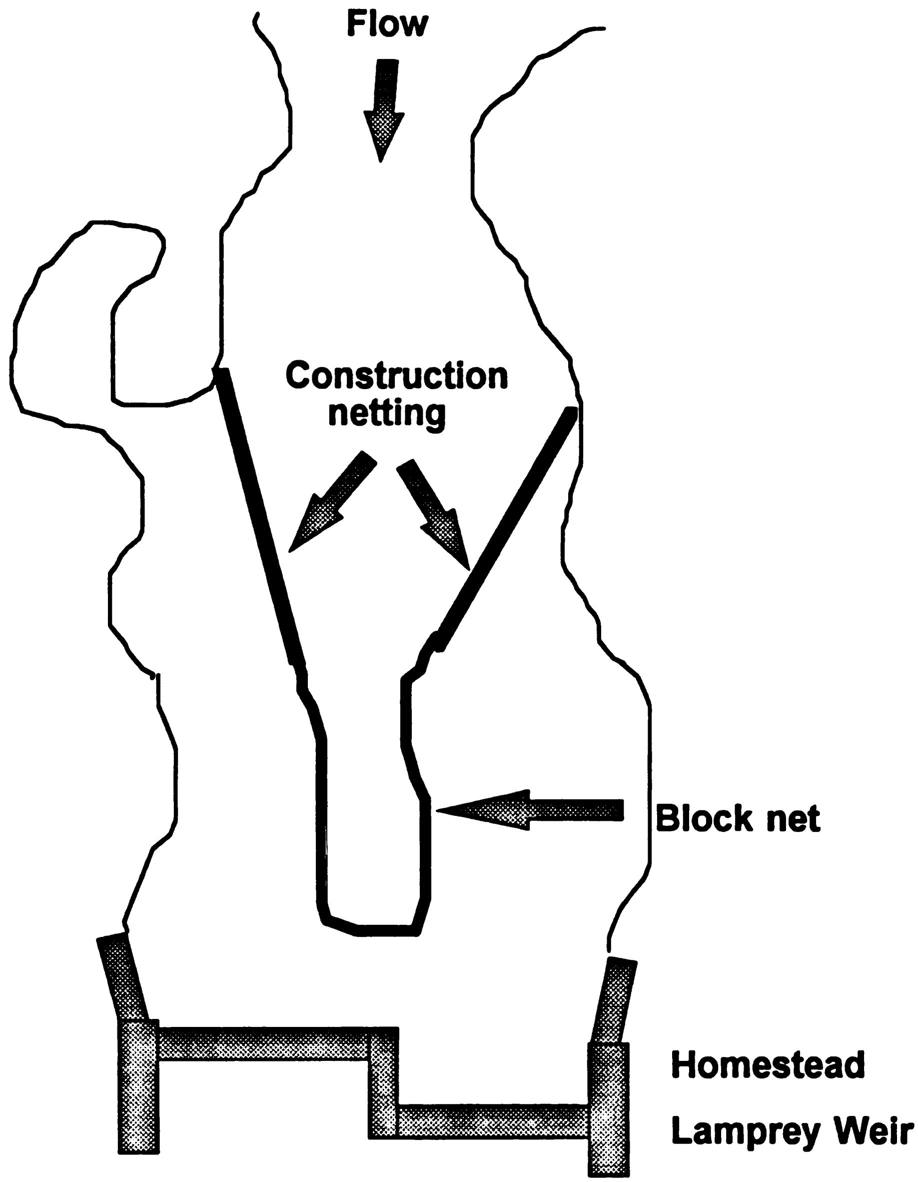

Figure 5. Constriction weir and block net design for capture of emigrating in the Betsie River, Michigan.

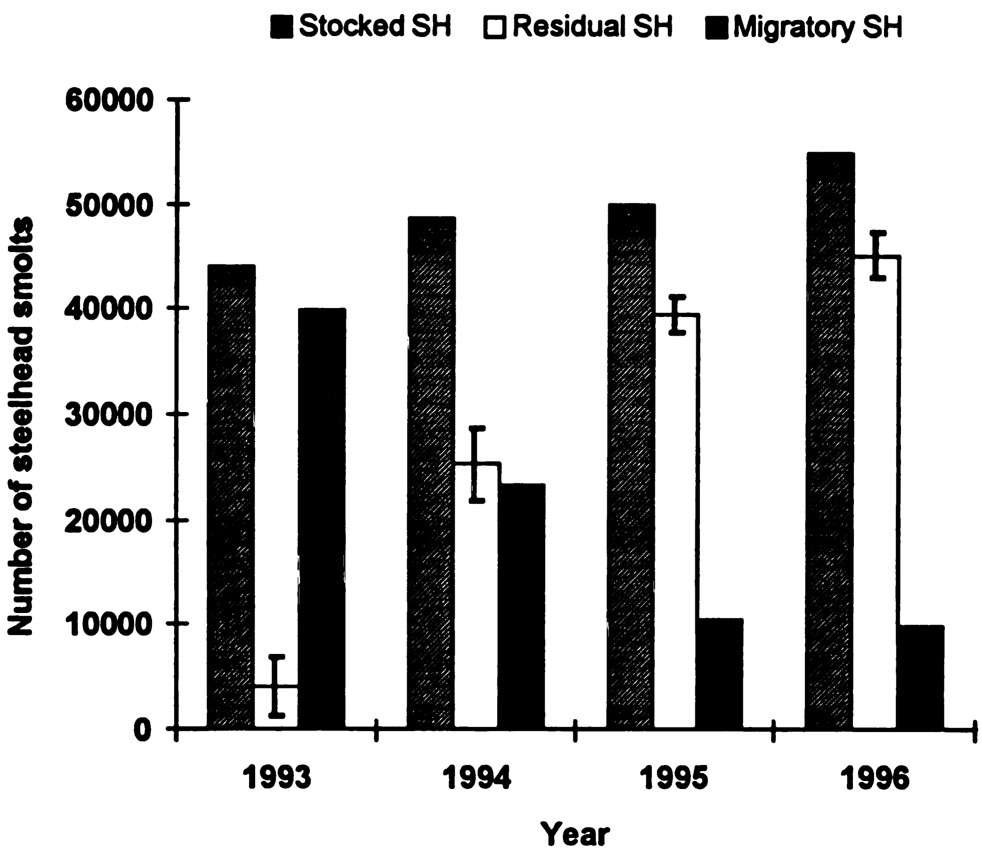

Figure 6. Number of steelhead smolts stocked, estimated number that remained as residuals in the river, and resulting number of potential migratory steelhead smolts in the Betsie River, Michigan, 1993-1996.

within the area where they were stocked (Seelbach 1987), I estimated the density of residual fish from a site sampled near the hatchery to determine the abundance of hatchery fish within 2 miles upstream and downstream of the stocking location. All reach estimates were then summed to obtain a total watershed estimate for residual hatchery steelhead. The estimate of residual steelhead was subtracted from the number of fish stocked from the hatchery to determine the number of marked, migratory fish that was assumed to be randomly mixed with the wild migrating smolts (Figure 6). I used standard methods to calculate the Peterson estimate and variance (Ricker 1975).

Results

Observations of smolt movement commenced at the beginning of May each year and continued until mid- to late June, with the exception of 1993 when sampling was discontinued early due to logistical complications. Steelhead smolts ceased migratory activity in the last two weeks of June and the duration of the smolt run ranged from 39 to 51 days (Table 1). From 1993-1996, I captured 1,755 steelhead smolts of

Table 1. Duration of observations and smolt movement at Homestead Dam, Betsie River, Michigan, 1993-1996.

Date of First Observation	First Smolt Observed	Last Smolt Observed	Date of Last Observation	Duration of Smolt Movement
5-8-93	5-10-93	6-16-93	6-17-93	39 days
5-5-94	5- 7-94	6-22-94	6-23-94	47 days
5-1-95	5- 2-95	6-19-95	6-22-95	48 days
5-1-96	5- 7-96	6-27-96	6-28-96	51 days

which 365 were determined to be wild on the basis of scale analysis. Annual mean length of wild steelhead smolts in 1993-1996 ranged from 193.6 to 195.0 mm. Over half of the wild emigrating smolts were age-1 (55-72%) each year, less than half were age-2 (26-45%), and fewer than 3% of the emigrating steelhead smolts were age-3.

Smolts migrated past the monitoring site in a consistent nightly pattern. The greatest number of smolts passed over the Homestead Weir between dusk and midnight. Over 80% of the smolt passage occurred before 02:00 hours (Figure 7) on most nights.

Four species of fish were captured in the constriction weir above Homestead Dam during the hours of 18:00 through 05:00. Recently released hatchery brown trout (fin erosion indicated hatchery origin), wild coho salmon smolts, wild and hatchery steelhead smolts, and wild chinook salmon smolts were present in the catch. The proportions of each species captured varied between years; steelhead comprised 30% to 61% of the spring smolt run (Figure 8).

Each year the number of marked fish stocked into the Betsie River increased. The number of residual fish also was very large in some years, which accounted for between-year variance in the number of marked migratory steelhead (Figure 6). Peterson mark-recapture estimates that were not adjusted for residual fish resulted in estimates 2-5 times greater than those adjusted for residual fish (Figure 9). Total estimates for hatchery and wild steelhead passing the weir ranged from 13,837 (12,583-15,215) in 1995 to 56,661 (46,036-69,703) in 1993 (Table 2). Wild steelhead smolts comprised 30%, 12%, 24%, and 52% of the captured steelhead smolts in 1993-1996, respectively.

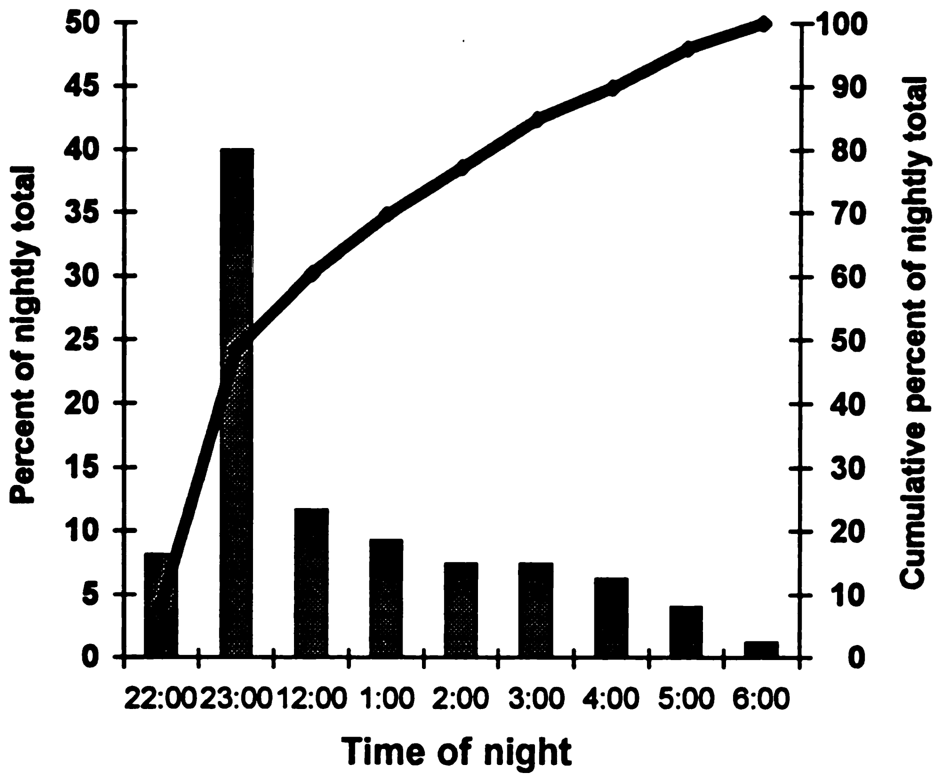

Figure 7. Temporal distribution of outmigrating steelhead smolts in the Betsie River, Michigan. Hourly values represent the mean for observations during that time period.

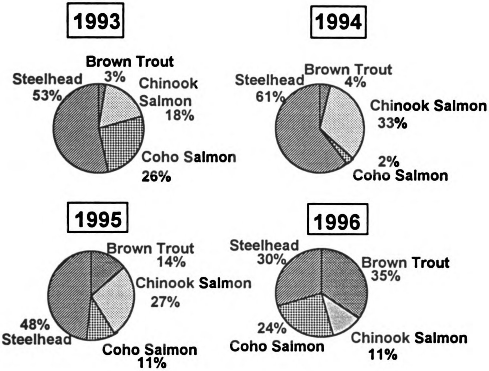

Figure 8. Species composition of the juvenile salmonids captured at Homestead Weir, Betsie River, Michigan, 1993-1996.

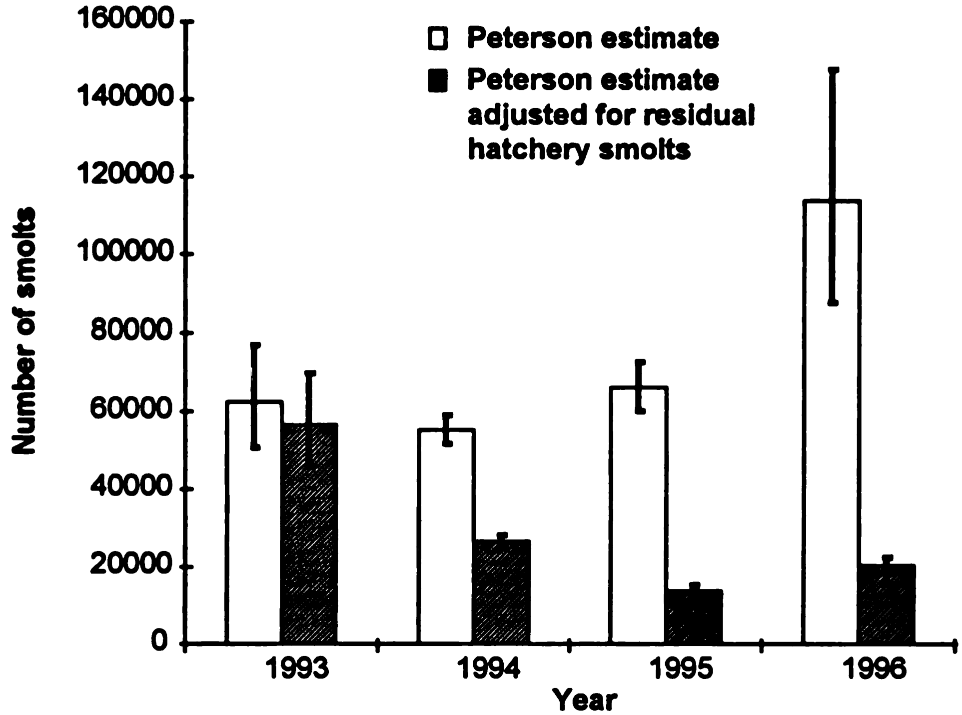

Figure 9. Comparison of mark-recapture estimates and 95% confidence intervals of steelhead smolt numbers with and without adjustment for residual steelhead in the Betsie River, Michigan, 1993-1996.

Table 2. Peterson mark-recapture estimates of the number of steelhead in the Betsie River smolt run.

Year	# Marked Migratory Steelhead	Total # of Steelhead Captured	# of Hatchery Smolts Recaptured	Estimate and 95% C. I. of All Steelhead	Estimate and 95% C. I. of Wild Steelhead
1993	39,888	124	87	56,661 46,036-69,703	16,998 15,057-18,939
1994	23,335	977	860	26,507 27,795 - 28,337	3,181 3,107 - 3,255
1995	10,463	561	424	13,837 12,583 - 15,215	3,321 3,164 - 3,478
1996	9,832	13	54	20,381 15,687 - 26,448	10,598 8,581- 12,615

The mean number of steelhead outmigrants was even lower based on the 20 minute visual sampling and videographic methods. The number of wild and hatchery steelhead passing the weir ranged from 2,328 (± 1,249) in 1996 to 9,645 (± 1,111) in 1994. The number of wild steelhead smolts was determined from the overall estimate of steelhead smolts and varied only slightly among years from 1,211 (± 649) in 1996 to 2,151 (± 286) in 1993 (Table 3). I used videographic methods in the last two years of the study, and smolt numbers were similar to the estimates derived from direct observation, 20 minute sampling (Table 3).

Table 3. Estimate of the number of smolts from 20 minute direct observation sampling and continuous time-lapse videography at the Homestead Lamprey Weir, Betsie River, Michigan.

Year	Estimate and 95% Confidence Limit for all Steelhead Smolts	Estimate and 95% Confidence Limit for Wild Steelhead Smolts
20 Minute Observations		
1993	7,169 ± 954	2,151 ± 286
1994	9,645 ± 1,111	1,157 ± 133
1995	7,120 ± 1,282	1,709 ± 308
1996	2,198 ± 512	1,143 ± 266
Time Lapse Videography		
1995	5,259 ± 3,328	1,262 ± 799
1996	2,328 ± 1,249	1,211 ± 649

With a few exceptions, estimates of smolt numbers from subsamples of the nights of observation were similar to those from sampling the entire smolt run (Figure 10). Large differences occurred however, in the 95% confidence intervals and standard error (Figures 10 and 11). Standard error increased with decreasing levels of effort. Effort involving every other day (50%) and an 80% random sample had similar error rates, and sampling efforts of every 3rd day (30%) and a random 20% were much larger in error (Figure 11).

Discussion

Steelhead smolts in the Betsie River followed similar migratory patterns to those described from other Michigan streams and elsewhere in their timing, and nocturnal movement (Stauffer 1968, Seelbach 1993). Because I observed no significant smolt movement during the hours between 05:00 and 21:00, our efforts were concentrated on nocturnal movement. Brege et al. (1996) monitored salmonid smolt movement on the Columbia River over 6 years and nocturnal movement of steelhead smolts

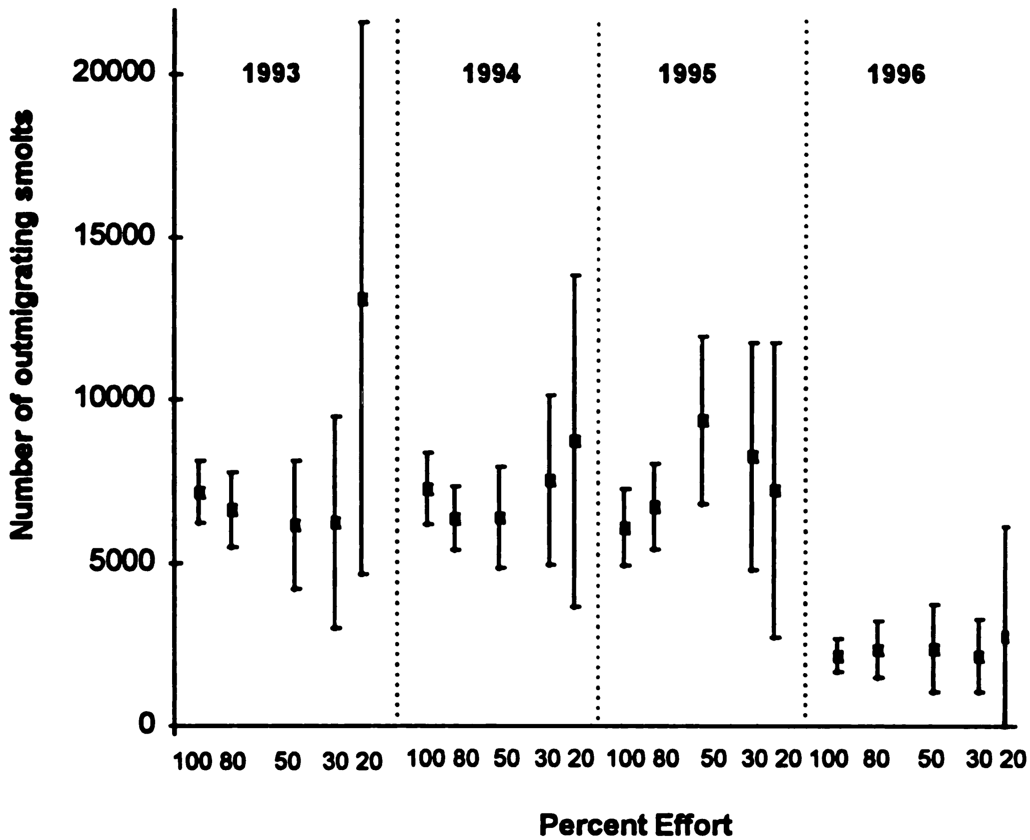

Figure 10. Comparison of the number of outmigrating smolts and 95% confidence intervals calculated from variable sampling efforts ranging from 100% to 20% of the days during the smolt run.

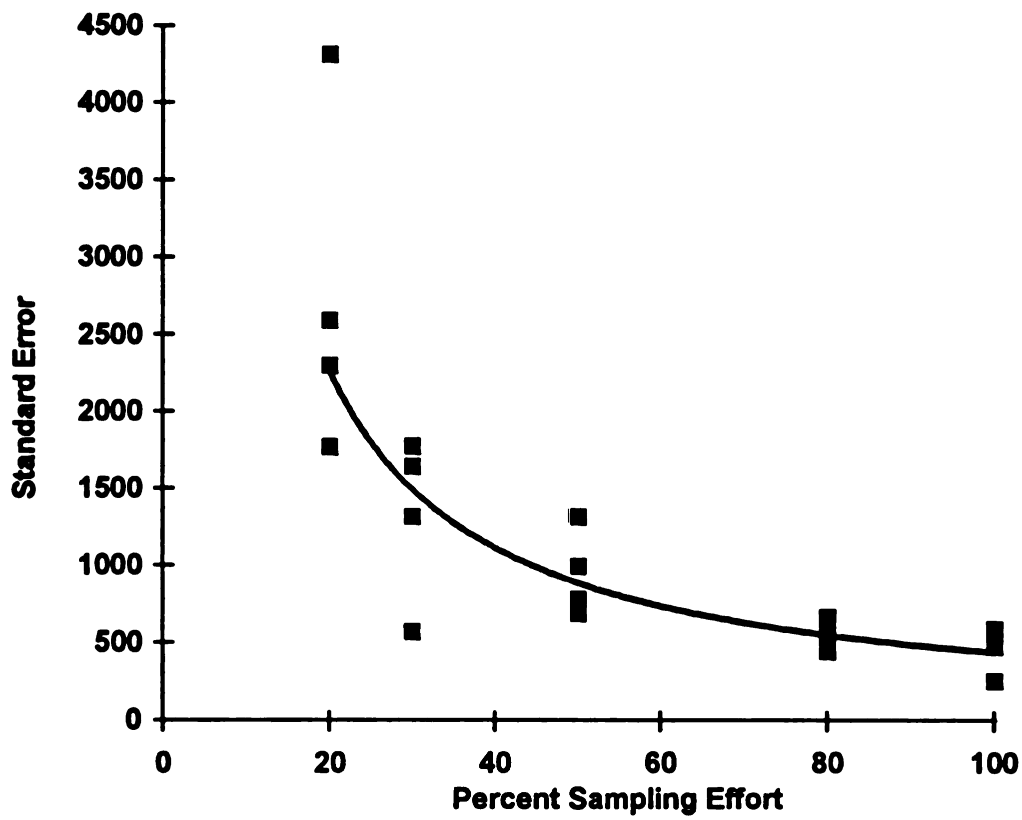

Figure 11. A comparison of the standard error of number of outmigrating steelhead smolts as affected by sampling effort. Fitted line is a logarithmic relationship with $r^2 = 0.7697$.

accounted for 66.2% to 87.4% of the daily observations, with a mean of 77.9% (Brege et al. 1996). Using this information, our estimates could be increased by 22% to account for any daytime passage that may have occurred. However, from periodic checks of smolt activity during the day, only once, in 1995 was any movement observed between the hours of 07:00 and 21:00.

The mark-recapture estimates of outmigrating smolt numbers were 2-9 times greater than those determined from the videography and direct observation methods (Figure 12). Assumptions of mark-recapture methodology include: 1) animals do not lose marks, 2) sampled fish are classified correctly, 3) either the population is closed, or there is neither recruitment or immigration, and 4) sampling is random with regard to marks, so that every fish has an equal chance of capture (Cormack 1968, Pollock 1981). Because fin clips were used to identify hatchery fish, marks were not lost. However, fish marking was not consistent (up to 35% of the smolts were not clipped in one year). To avoid an error in the assumption of correct classification, I used scale analysis to verify classification of smolts as either hatchery or wild. I also assumed that the migratory population of smolts was closed. This was a reasonable assumption as this was a watershed assessment and smolts were not able to move upstream past the Homestead lamprey weir.

Unexpectedly large numbers of marked fish did residualize and I adjusted the estimate of the marked migratory population accordingly. It is possible that I underestimated the number of residual smolts: e.g., they may have concentrated in large pools below the hatchery which were not amenable to sampling with our electrofishing procedures. In a different approach to estimating residual steelhead, I calculated the number of hatchery steelhead that were likely to residualize based on lengths observed in the hatchery in April. From measures of 100-200 individual fish in

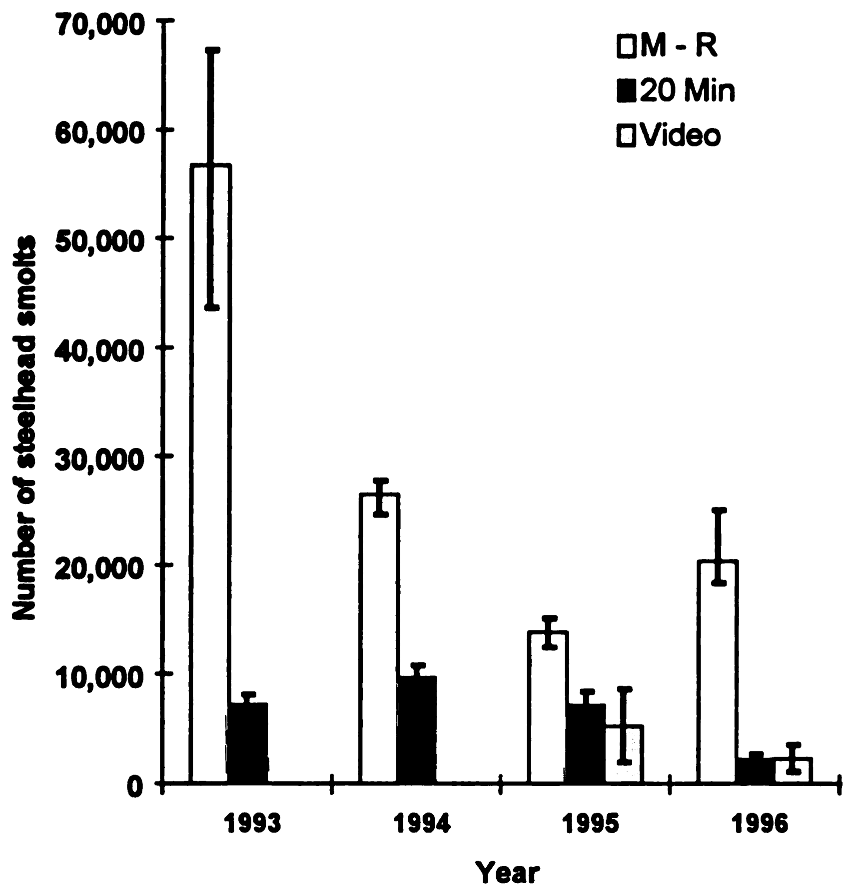

Figure 12. Comparison of the steelhead smolt numbers determined from mark-recapture (M-R), 20 minute observations (20 Min) and continuous recording (Video) in the Betsie River, Michigan, 1993-1996. Error bars represent 95% confidence intervals.

hatchery in 1994-1996, I determined that 97%, 81%, and 88% (respectively) of the steelhead in the hatchery were less than 180 mm, the length at or above which steelhead smoltification is likely to occur (Seelbach 1987). Using this method, the estimates of residual steelhead were 47,142, 40,500, and 48,400 in 1994-1996. In this approach, the estimates for 1995 and 1996 are similar to the estimates of residual fish determined from electrofishing throughout the watershed in July (Figure 6). However, the 1994 estimate from the hatchery approach was nearly twice as large as that determined from the watershed estimates. This may indicate that I underestimated the number of residual fish for that year with electrofishing, or that other factors, such as release dates or hatchery crowding may have influenced the emigration of these fish (Wedemeyer et al. 1980).

A failure to recapture a sufficient number of marked fish for a precise estimate would have been indicated by large confidence intervals (Dempson and Stansbury 1991). The confidence intervals determined in this study ranged from 10-24% of the estimates, which may be considered as fairly reliable estimates (Roff 1973, Cousens et al. 1982)

The most likely reason for the seemingly disproportionate mark-recapture estimates were due to violations of the assumption of equal chance of capture (Pollock 1981). Inherent in this assumption is that marked and unmarked fish mix randomly and that the probability of capture is equal for either classification. Juvenile steelhead were released volitionally from the hatchery, which was located approximately 29 km upstream of the Homestead lamprey weir. Volitional releases resulted in the hatchery fish being released at different rates between years. In some years, smolts left the hatchery one to two weeks after a peak in the number of outmigrating wild smolts and as the river was warming above the upper limit (13°C; Zaugg and McClain 1973) for steelhead smolting behavior.

In a similar mark-recapture application to estimate the number of outmigrating smolts (Peven and Hays 1989), the estimates of smolt numbers were corroborated with a life-history method (estimating smolt production based on spawner escapement values, fecundity rates, and survival to age information) to determine the number of smolts. Peven and Hays (1989) found similar results between the two methods, with mark-recapture of smolts giving a likely and reasonable estimate. The difference in this application from the Betsie River is that Peven and Hays (1989) verified that marked smolts were migrating when they were marked, but I could not verify that the marked fish were migrating when they entered the Betsie River.

Other estimates, using mark-recapture and maximum-likelihood methods, may have yielded better estimates than the single census mark-recapture (Cormack 1968, Dempson and Stansbury 1991, Schwarz and Dempson 1994, Pollock 1981, Macdonald and Smith 1980). However, I could not obtain estimates of the number of smolts released on any given day and smolts were released at varying times (e.g. weekdays only and not on holidays), which precluded the use of any method other than the single census Lincoln-Peterson index.

Mark-recapture estimates are used often to estimate trap efficiency for smolt collection methods (Raymond 1979, Seelbach 1987). Raymond (1979) concluded that the marked steelhead smolts were equally mixed with the nonmarked smolts because of an equal capture rate at a downstream location. When used to estimate escapement by adult salmonids, the accuracy of mark recapture ranged from ± 25% to ± 30% of the estimated value (Cousens et al. 1982). Based on my experience in the Betsie River, mark-recapture methods overestimated the number of emigrating smolts and I would caution against the use of single-census Peterson estimates alone to obtain total smolt abundance estimates with stocked, marked hatchery fish. At the very least, an estimate of residual marked fish should be determined to make adjustments in the number of marked migrants.

The time-lapse videography used in this study has several advantages that make it feasible in many different field applications. Use of such monitoring systems is labor efficient in that many stations can be operated by a single individual, allowing a large amount of information to be recorded during this time-specific life history phase for later analysis. The system can be used in remote locations, as demonstrated by our use of gasoline generators which provided sufficient power and did not cause interference with the cameras or recorders. A surface view approach provided an effective and inexpensive viewing of emigrating smolts. Although this application involved a lamprey weir, I could have easily established a viewing station on a hydraulic control or riffle; the system should work at any location where fish could be guided over a narrow backdrop presented at a depth where smolts could still be viewed. Drawbacks to the surface view approach are that it requires an additional trapping effort when multiple species are migrating and that turbidity of the water can impede observations. However, in our study, turbidity prevented visual observations on only 3 days in four years (185 days) of viewing. A ruled backdrop and slightly higher resolution of the images (which would require at least one additional camera in this application) could have allowed us to use inference from lengths to determine species composition among coho salmon, chinook salmon and steelhead based on the differences in mean length of smolts of each species.

Although the video monitoring was relatively labor efficient, manual review of the videotapes was tedious, but produced accurate results. A single taping event for one camera included 8 hours of recording and required 3-4 hours to carefully review and count smolts. The efficiency of this effort could be enhanced by automated viewing techniques (Irvine et al. 1991). Similar to Hatch et al. (1994), comparisons between direct observation estimates and videotape results were similar. In addition, no significant differences were found between the 3 trained reviewers in counting smolts from the videotapes.

Because of the nocturnal movement of smolts, lighting was required to illuminate the area enough to provide a view through the water to the sill. I used 250 watt diffuse halogen lighting to illuminate the area for monitoring. Because the lighting was constant, it is not likely to have affected the passage of smolts. The change in environment for the emigrating smolts from the dark upstream areas to the lighted sill area may have caused hesitation in passage over the weir. A typical pattern observed by the smolts was for them to back down to the front of the sill and school together. The school would swim back and forth across the front of the sill and continually increase in numbers until a time when the large numbers of smolts would pass over the sill either together or sequentially. Artificial lighting was not likely to have caused this behavior, because the same behavior was observed during dusk when lighting was not required and also during the few daylight observations of smolt passage. One problem with the use of halogen lighting was that it attracted clouds of emerging insects. At times, this caused the cameras to focus on the insects rather than the sill and sometimes the insects were so thick that they prevented counting of the smolts. This was usually an early evening phenomenon, and occurred when air temperatures were warm. However, it could be avoided by the use of infra-red lighting, without an effect on fish behavior (Beach 1978).

The abundance of smolts passing through any given night was variable. In a telemetry study of coho smolts, Moser et al. (1991) found that movement was in a saltatory fashion with movement in the direction of the current and then extended periods of holding in areas of low current velocity. This behavior could explain some of the variability in the abundance of smolts observed between nights, especially when considering the small number of smolts that emigrated from the watershed.

Subsamples of the number of nights required to establish an estimate of smolts indicated that sampling between 60-100% of the nights during smolt outmigration would give similar estimates for the effort incurred. An effort of observation that included less than 60% of the nights of smolt movement would likely result in an estimate with an extremely large error. Of course, in any smolt monitoring application, determination of the beginning and end of smolt movement is required to determine the extent of sampling required. In some cases, the onset and cessation of smolt movement may be approximated with water temperature trends (Wedemeyer et al. 1980, Hoar 1984).

The Betsie River watershed produced few smolts per hectare when compared to other streams in Michigan. The Huron River, a tributary of Lake Superior and a smaller river than the Betsie River, produced 9,141 and 1,031 wild steelhead smolts in 1987 and 1988 or 46-262 steelhead smolts/ha (Seelbach and Miller 1993). The Little Manistee River is a high quality trout stream tributary to Lake Michigan, south of the Betsie River, with a drainage of 59,000 ha and a mean annual flow of 5-6 m^3/s (Seelbach et al. 1984). This river produces a greater number of wild smolts annually, ranging from 11,845 to 86,425 smolts/year with a density of 425/ha (97-713/ha).

Conclusions and Management Implications

Currently, the Betsie River watershed produces fewer than 3,000 wild steelhead smolts, far less than steelhead smolt abundance estimated in a high quality trout stream such as the nearby Little Manistee River. However, this level of production may reflect the productive capacity that could be expected in the marginal trout streams that experience high summer temperatures and moderate to severe winter temperatures.

Mark-recapture methods largely overestimated the numbers of smolts leaving this watershed. A single census, Peterson estimate should be used with great caution and the assumptions of equal mixing and equal probability of capture should be addressed for conclusive results.

The surface-view time-lapse videography and 20 minute observational sampling provided estimates similar to one another. Both are inexpensive (relative to equipment requirements) and could be conducted on weirs and natural hydraulic controls in smaller rivers. The videography has additional advantages of providing a permanent record and requiring less labor. Improvements to the video system I used could include electricity at the site, infra-red lighting to decrease interference with the view frame by flying insects, and a contrasting backdrop to help clearly illuminate the fish. Further efficiency could be accomplished by increasing the resolution of the cameras and adding a ruled backdrop to measure emigrating smolts.

Chapter 2

DISTRIBUTION, DENSITY, AND SURVIVAL OF STEELHEAD PARR IN A THERMALLY DIVERSE WATERSHED

Abstract

In watersheds with diverse instream habitats, such as the Betsie River in northern Michigan, production of early life history stages of fishes can be concentrated into a few critical habitat units. The Betsie River watershed is atypical of the neighboring watersheds due to its thermal properties. The main channel supports a popular migratory salmon and steelhead fishery, but is marginal habitat for resident trout because summer temperatures are high. Tributaries in the watershed are proportionately greater in groundwater flow than the main channel and therefore provide colder habitats in summer. The objectives of this study were to describe and quantify the distribution, density, and mortality of steelhead (*Oncorhynchus mykiss)* parr throughout the Betsie River and its tributaries and to evaluate the influence of different thermal macrohabitats on steelhead parr production. From 1993-1996, I estimated parr abundance at 14 sites throughout the watershed by use of depletion sampling and I determined the number of emigrating smolts by visual methods. Densities of steelhead parr ranged from 0/ha to 3000/ha at sampling sites. I found 62% of age-0 parr and 50% of the age-1 parr in the main channel. The remaining parr were found in the tributaries which comprise only 11% of the total stream area in the

watershed. Smolts out-migrated at annual yields ranging from 11.9/ha to 22.0/ha for the watershed channel area. Annual instantaneous mortality rates (Z) ranged from 0.710 to 3.578 and were the greatest for age-1 and age-2 parr. Mortality was greater during severe winters and at sites in the main channel. Because many parr were found in the few tributaries sampled, and because survival was higher in these streams, the small tributaries are highly valuable to the overall production of steelhead from this watershed.

Introduction

Steelhead were introduced into the Great Lakes Basin in 1867 and since their introduction, naturalized populations have become established in many Great Lakes tributaries (Latta 1974, Biette et al. 1981). Great Lakes steelhead migrate into streams in the fall, winter, and spring and spawn in the spring. Juveniles typically spend 1-3 years resident in the tributary watershed before migrating to the lake, where they grow for 1-3 years before reaching sexual maturity (Biette et al. 1981, Seelbach 1993). In most streams, the majority of juvenile steelhead spend 2 years in the river while the remainder spend only 1 or 3 years in the watershed (Biette et al. 1981).

Water temperature can limit the distribution and abundance of a population of salmonids by high summer temperatures or low winter temperatures that exceed the tolerance limits for survival (0 - 25°C; Wismer and Christie 1987). Effects of temperatures approaching or exceeding the limits may include mortality, metabolic challenges resulting in decreased growth (Adams and Breck 1990), or fish may distribute themselves throughout the watershed in the most preferred thermal areas

(Nielson et al. 1994, Peterson and Rabeni 1996) which may lead to competition for space or a carrying capacity that is a function of thermal macrohabitat availability. Many of the southern streams in Michigan are marginal for resident trout due to high summer temperatures, yet returns of both wild and hatchery steelhead to these rivers can be very large (Seelbach 1989, Dexter and Ledet 1993). Currently, little information exists on the numbers of juvenile steelhead that can be produced or their population dynamics in watersheds dominated by marginal thermal habitat (Seelbach 1989, 1993).

Severe winters can be detrimental to a cohort of steelhead with mortality of 50-90% reported in some studies (Maciolek and Needham 1951, Seelbach 1987). Partial explanation for the mechanisms behind winter mortality may be limiting microhabitat. In water below 8°C, salmonids reduce activity and seek cover (Hartman 1963, Campbell and Neuner 1983). Preferred winter habitat for steelhead parr includes groundwater fed side-channels, complex woody debris, interstitial spaces within boulders or rubble along stream margins, under cobble, and in water less than 30 cm deep with near-0 water velocities (Everest and Sedell 1983, Campbell and Neuner 1984, Heifetz et al. 1986, Swales et al. 1986). In northern streams, low temperature conditions can exist for months at a time and in predominantly sandy substrate streams, overwintering habitat may be critically limiting (Bustard and Narver 1975, Swales et al. 1986).

Many stream rehabilitation or restoration projects focus on the improvement of instream habitat and reduction of erosion throughout the watershed. Knowledge of the critical production areas for wild steelhead in streams can help to prioritize and implement important watershed management activities such as protection of riparian vegetation, a reduction in erosion, and the addition or manipulation of instream habitat.

The objectives of this study were to:

1) describe and quantify the spatial and temporal distribution of steelhead parr throughout the Betsie River and its tributaries,

2) estimate reach densities and total annual production of the number of age-0, age-1, age-2 juveniles in the watershed,

3) evaluate mortality rates and their relationship with thermal conditions in the Betsie River and its tributaries.

Study Site

The Betsie River watershed is located in northwestern corner of the lower peninsula of Michigan (Figure 13). The watershed is one of the smaller watersheds in Michigan that is tributary to Lake Michigan (67,126 ha). The river begins via an unregulated, natural surface outflow from Green Lake and flows for approximately 79 km before reaching Betsie Lake, which flows into Lake Michigan at Frankfort (44°W, 86°N). Annual mean discharge is 2.31 m^3/s at Green Lake and 7.45 m^3/s near the mouth of the river. The gradient is low (mean gradient = 0.113%, ranging from 0.007% to 0.568%) and the river flows through a landscape dominated by glacial moraines surrounding a broad glacial outwash valley. Land use in the watershed is predominantly light agriculture and timber production, and very little urban development exists (Gooding 1995). Several small tributaries drain into the Betsie River. The Little Betsie River and Dair Creek are the largest with mean flows of 0.51 and 0.44 m^3/s . Small spring-fed tributaries that are approximately 1.0 m wide and 5-30 cm deep are prevalent throughout the watershed. The river has no permanent discharge gages, but

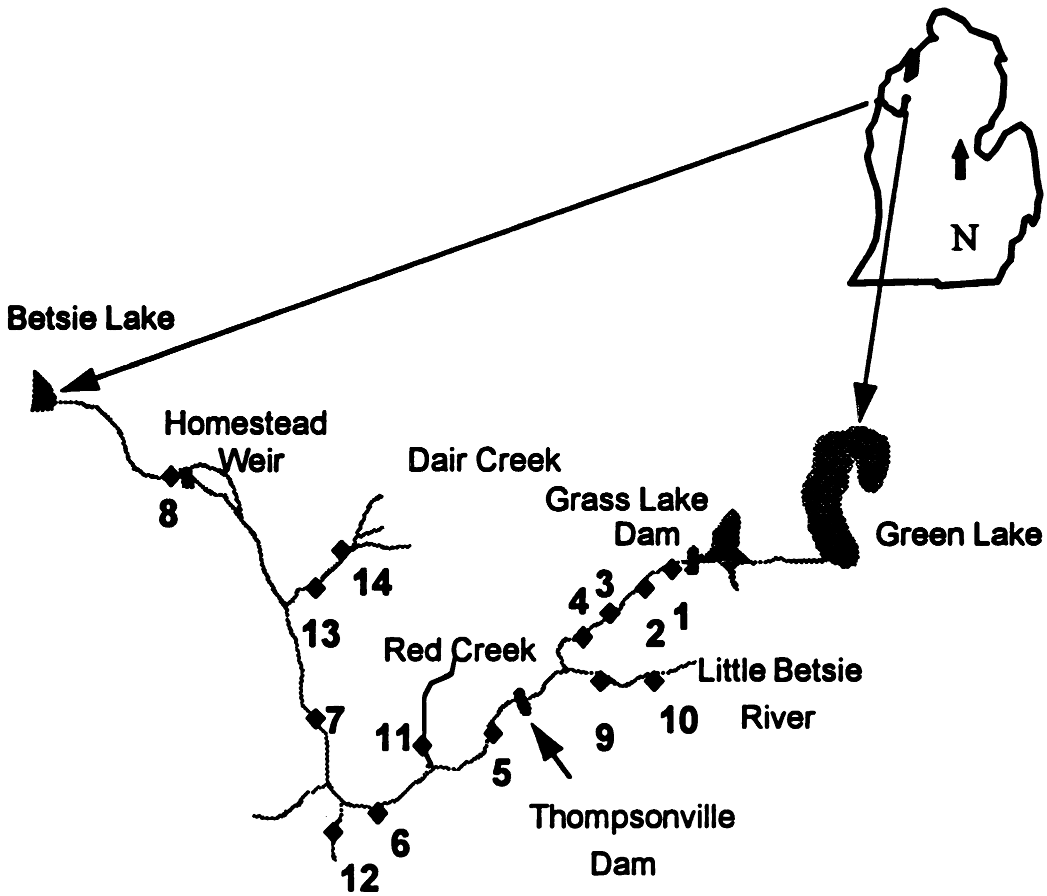

Figure 13. Sampling sites for electrofishing in the Betsie River watershed, Michigan, 1993-1996.

summer flows measured through the course of the study ranged from 4.0 to 12.7 m^3/s with a mean flow of 5.5 m^3/s .

Fish surveys indicate a predominance of white suckers (*Catostomus commersoni*), hornyhead chubs (*Nocomis biguttatus*), creek chubs (*Semotilus atromaculatus*), and blacknose dace (*Rhinichthys atratulus*) (Wicklund and Dean 1958, MDNR, Fisheries Division Stream Surveys, 1990) in the fish community, yet the Betsie River supports popular steelhead, chinook salmon (*Oncorhynchus tshawytscha)* and coho salmon (*Oncorhynchus kisutch*) sport fisheries. The steelhead fishery is supplemented by stocking of hatchery fish in the river, but neither coho or chinook salmon are stocked in the Betsie River.

Recent changes in the watershed have increased interest in the potential production of wild steelhead. Thompsonville Dam, which was located in the mid-reaches of the watershed (Figure 13) failed in March, 1989. Downstream sedimentation and streambank erosion is still evident from this event. However, the upper 22 km of the watershed is now accessible to spawning migratory salmonids and resident juvenile salmonids, including steelhead.

Methods

I stratified sections of the watershed according to differences in temperature regimes, discharge, and gradient, resulting in the following reach designations: upper Betsie River (sites 1-4), lower Betsie River (sites 5-8), Little Betsie River (sites 9-10), Dair Creek (sites 13-14), and small tributaries (sites 11-12)(Figure 13). Sites were randomly selected within the reaches and all 14 sites were sampled in the same two weeks in late July (1993-1996) and mid-October (1994-1995) to determine steelhead parr abundance. I used a 250 volt DC electrofishing barge unit and multiple pass depletion methods to capture parr and estimate site abundance (Zippen 1958). Block nets were placed at the upstream and downstream margins of the sampling sites prior to electrofishing to meet the assumption of a closed population. At the time of capture, I anaesthetized steelhead parr with MS-222, and measured total length (mm) and weight (g), and removed scale samples from an area above the lateral line, below and posterior of the dorsal fin origin (Jearld 1983) . Fish were checked for fin clips to identify their origin (hatchery or wild) and were released downstream of the sample site. I determined the age structure of the population by use of length-frequency analyses and scale pattern analysis. Scales were also inspected to determine if the unclipped steelhead parr had been produced in a hatchery and then remained in the river after stocking (Seelbach and Whelan 1988). All hatchery parr were excluded from this analysis.

Maximum likelihood estimates of abundance and confidence intervals were calculated for each site from the removal data by use of Microfish 3.0 computer software (Van Deventer and Platts 1983, 1985, 1986). Because of small estimates for steelhead parr at many sites, a total steelhead estimate was calculated for each site and adjusted by the proportions represented at the site in each age-class captured to obtain an age-specific site estimate and variance according to

$$n_{ac} = P_{ac} * m, \quad Var(n_{ac}) = P_{ac}^{2} * Var(m) \qquad \text{(Equation 1 and 2)}$$

where,

n_{ac} = the number in each age class
P_{ac} = the proportion captured in each age-class, and
m = the estimate of the total number of steelhead parr in the site.

Density and variance estimates per hectare by age-class were calculated by

$$d_p = n_p / a_p, \text{ and } Var(d_p) = 1/a_p^2 \, \theta^2_p \qquad \text{(Equations 3 and 4)}$$

where,

d_p = the age-specific density at site p,
a_p = the channel area in hectares at site p,
θ^2_p = the variance for the estimate at site p.

Mean densities for the Betsie River, Little Betsie River, Dair Creek, and the two small tributaries were calculated by

$$D_t = \Sigma \, d_i / n, \; Var(D_t) = 1/n^2 \qquad \text{(Equations 5 and 6)}$$

where,

D_t = the density for reach or tributary t, and
n = the number of sites sampled within reach or tributary t.

Extrapolations for reach or tributary (Z) abundance of parr and a variance were calculated by

$$Z_t = a_t * D_t, \; Var(Z) = A^2 * (Var\, D_t) \qquad \text{(Equation 7 and 8)}$$

where

Z_t = reach or tributary abundance, and
a_t = area in the reach or tributary t.

And finally, the overall watershed estimate of abundance (N) was calculated for each age-class by

$$N = \Sigma \, Z_t, \; Var(N) = \Sigma \, Var\, Z_t \qquad \text{(Equation 9 and 10)}$$

where N = total watershed abundance for age-class.

I used a general linear model procedure (SAS) to investigate differences in mean lengths among the channel reaches for the summer and fall data.

Estimates of the number of outmigrating steelhead smolts were determined each May-June, 1993-1996, by use of a visual method (Chapter 1). In addition to estimating total abundance of smolts, I determined the species composition and origin (hatchery/wild) of the smolts by use of traps every fifth day in the smolting period (Chapter 1). Hatchery smolts were excluded from consideration in this study.

Instantaneous mortality rates were calculated according to Ricker (1975) for sample reaches and the watershed. Mortality estimates that I considered included mean annual mortality and within year rates for winter (October-July) and summer (July-October) for the watershed. I assumed that by October, the parr population had experienced the full effects of summer conditions and the population represented pre-winter abundance. I also assumed that by July, all age-0 steelhead were large enough to be captured by electrofishing gear and that all emigrating fish had left the watershed.

I monitored temperature at 7 sites in the watershed that corresponded to electrofishing sites. Continuous recording equipment (Stowaways™, Onset Instruments, Pocasset, Massachusetts) documented hourly water temperatures in those sites. Prior to deployment and at the end of the study, the Stowaways™ were tested in a water bath using an ASTM thermometer to check the accuracy and to assess the need for calibration adjustments. Stowaways™ were anchored in the channel where the water was continuously flowing, out of direct sun, and approximately 3 cm above the substrate. I calculated daily, weekly, and monthly means from the hourly records and also extracted the daily minimum and maximum values. A few missing data points were filled in with regression analysis from upstream and downstream locations. For each year, I also calculated mean summer (June, July, and August) and winter (December, January, and February) temperature and determined maximum and minimum temperatures for each monitoring location. In addition to water temperature, I obtained air temperature data from the weather station in Cadillac (NCDC) and calculated winter severity as a function of number of days with a mean daily temperature less than -12°C (similar to Seelbach 1987).

Results

Reach Characteristics

The channel morphometry, discharge, and thermal characteristics varied widely among the sample reaches (Table 4). Discharge doubled between the upper and lower reaches of the Betsie River with only a 2.7 m increase in mean channel width and a

Table 4. Hydrological and morphological characteristics of reaches sampled in the Betsie River watershed.

Reach & Sites	Mean July Discharge (m^3/s)	Mean Channel Width (m)	Mean Channel Depth (m)
Upper Betsie River (1-4)	2.2	14.4	0.4
Lower Betsie River (5-8)	5.5	17.1	0.5
Little Betsie River (9-10)	0.5	6.7	0.3
Dair Creek (11-12)	0.4	5.5	0.2
Small Tributaries (12-13)	0.1	1.7	0.1

0.1m increase in mean channel depth. Mean summer discharge varied annually. In 1995, very little snow melt and lack of spring rain resulted in low flow conditions. In 1993 and 1996, the opposite occurred with large snow melt flows followed by high volumes of spring rain and higher summer flows for both years (Figure 14).

Water Temperature and Environmental Variability

Water temperature regimes varied spatially and temporally throughout the watershed. Winters of 1993-1994 and 1995-1996 were unusually cold, with over 20 days of mean daily air temperature less than -12°C (Seelbach 1987, Tables 6 and 7). Summer, 1995 was a drought year and the mean air and water temperatures were the highest recorded in four years of study (Tables 5 and 6).

Table 5. Summer and winter air temperature from the Cadillac, Michigan weather station.

Year	Summer Mean Temperature (°C) (Jun-Jul-Aug)	Winter Mean Temperature (°C) (Dec-Jan-Feb)*	# Days < -12°C
1993	18.6	- 7.9	8
1994	18.1	-11.8	29
1995	20.4	- 6.3	9
1996	18.0	- 8.8	20

* December is from the prior year

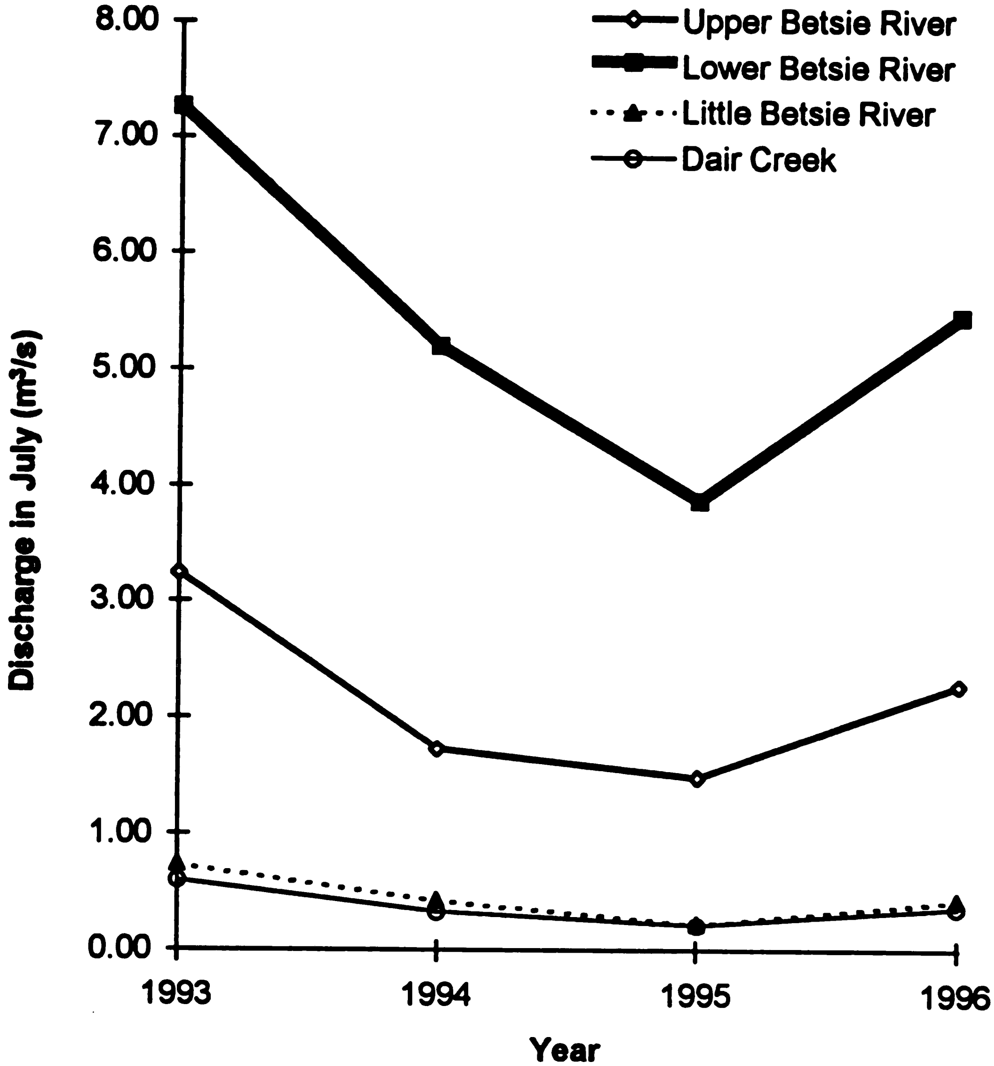

Figure 14. Mean discharge measured in July at sampling reaches in the Betsie River watershed, 1993-1996.

Table 6. Mean summer and winter water temperature in the Betsie River at Homestead Weir and Little Betsie River, 1993-1996.

	Summer Mean Temperature(°C)		Winter Mean Temperature (°C)	
Year	Little Betsie	Betsie	Little Betsie	Betsie
1993	16.0	18.3	not measured	
1994	15.9	17.6	1.0	0.9
1995	17.5	19.0	1.9	1.6
1996	16.0	17.8	0.7	0.4

Mean and maximum summer temperatures from stations throughout the watershed were as much as 6 to 8°C different on any given day. The upper Betsie River exhibited the greatest extremes in temperature with the warmest summer (>25°C) and coldest winter temperatures (< 0°C). The lower Betsie River reaches were not as extreme, but exhibited a wider range of temperatures than the tributaries, Dair Creek and Little Betsie River (Table 7). Dair Creek was the tributary most dominated by groundwater and was typically the coldest in the summer and the warmest in the winter. The Little Betsie River was similar, but usually slightly warmer in the summer and cooler in winter (Table 7).

Table 7. Summer and winter water temperatures in reaches sampled in the Betsie River watershed.

Reach and Sites	Water Temperature (°C) Summer Mean	Summer Max[1]	Winter Mean	Winter Min[2]
Upper Betsie River (sites 1-4)	20.9	24.8	0.70	0.30
Lower Betsie River (sites 5-8)	18.4	20.6	1.03	0.05
Little Betsie River (sites 9-10)	16.2	18.1	1.14	0.44
Dair Creek (sites 11-12)	14.7	16.7	1.41	0.59
Small Tributaries (sites 12-13)	7.2-12.8[3]			

[1]mean daily maximum temperature in July (1994-1996)
[2]mean daily minimum temperature in January (1994-1996)
[3]periodic measurements during the summer (1993-1996)

Abundance and Distribution of Parr Throughout the Watershed

From 1993 through 1996, I captured 4,130 steelhead parr in summer and autumn electrofishing. Length-frequency and scale analyses (699 scale samples) from electrofishing in July and October confirmed the predominance of two age classes (Figures 15 and 16). Age- 0 parr comprised a mean of 63% of the catch, age-1 parr made up 36% and fewer than 1% were age-2 parr.

Density estimates of parr varied widely throughout the watershed (Appendix B). Values of 0 - 20 parr/ha were common in the upper Betsie River, sites 1-4. The lower sites in the Betsie River (5-8) yielded higher densities, ranging from 0 to 1677 parr/ha (Table 8). The greatest densities of parr were recorded consistently in Little Betsie River, Dair Creek, and the small tributaries, with densities frequently more than 500 parr/ha. Age-2 parr ranged in density from 16/ha to 77/ha when they were present, but I encountered them only in the Little Betsie River and Dair Creek in 1994 and 1995.

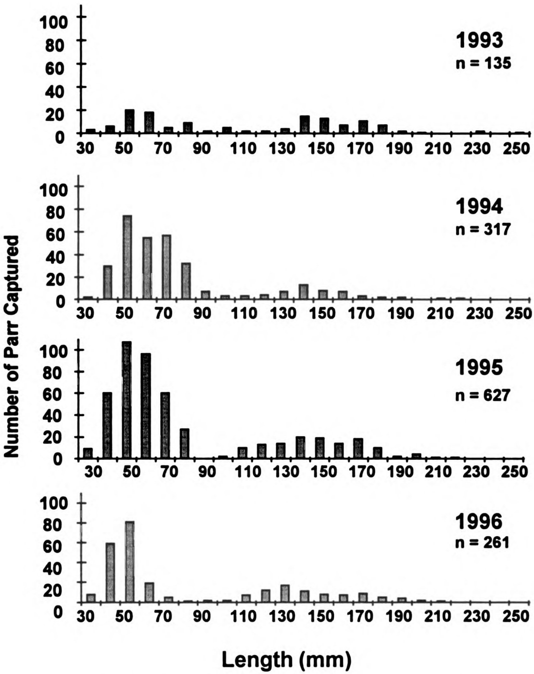

Figure 15. Length-frequency distribution of steelhead parr captured in July in the Betsie River watershed, 1993-1996.

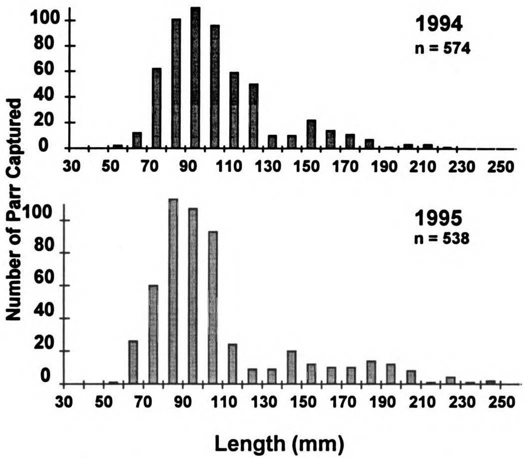

Figure 16. Length-frequency distribution of steelhead parr captured throughout the Betsie River watershed in October, 1994-1995.

Table 8. Density estimates and 95% confidence intervals for age-0 and age-1 steelhead parr at sites sampled by electrofishing and depletion methods throughout the Betsie River watershed, 1993-1996.

	SAMPLING SITES													
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)	(10)	(11)	(12)	(13)	(14)
SUMMER (AGE-0)														
1993	0	0	-	-	157.5	3.1	485.7	180.6	350.0	-	1600.0	2200.0	-	1000.0
					± 89.6	± 3.5	±303.0	±31.9	±20.7		±339.5	±195.3		±86.9
1994	0	0	0	0	25.0	58.7	499.0	38.9	1973.8	1327.6	1050.0	1800.0	1250.0	523.0
					*	±90.9	±21.5	±39.2	±87.1	±183.0	±72.7	±21.6	±43.3	±306.4
1995	0	80.0	16.7	9.1	85.3	360.9	1378.4	175.0	1532.1	360.6	1167.0	3090.9	1350.0	757.1
		±84.3	±4.4	±4.2	±119.0	±80.8	±73.3	±53.2	±30.4	±139.4	±1583.9	±352.3	±63.2	±110.5
1996	0	0	0	-	12.3	41.7	32.5	11.1	157.1	61.5	369.2	2775.0	800.0	2296.9
					±32.9	±36.5	*	*	±11.4	±25.4	±18.8	±273.7	±35.0	±89.6
SUMMER (AGE-1)														
1993	0	0	-	-	22.5	0	709.8	18.1	1200.0	-	0	0	-	200.0
					±12.8		±442.8	±3.2	±70.8					±17.4
1994	0	0	0	0	8.3	39.2	69.9	0	943.0	113.3	0	0	1750.0	31.4
					*	±30.9	±3.0		±79.9	±15.6			±393.9	±105.0
1995	0	0	8.3	36.4	24.4	68.2	75.1	19.4	978.8	106.1	166.7	103.0	750.0	108.2
			±2.2	±16.9	±34.0	±15.3	±4.0	±5.9	±19.6	±41.0	±115.4	±11.7	±35.1	±15.7
1996	0	0	7.7	-	0	16.7	71.5	16.7	257.1	23.1	138.5	102.8	600.0	423.1
			*			*	±1.6	*	±18.7	±9.5	±7.1	±10.1	±26.3	±16.5
FALL (AGE-0)														
1994	0	13.5	0	0	145.8	15.4	1676.5	-	1421.9	970.2	250.0	800.0	1326.5	714.3
		±8.0			±75.8	*	±56.8		±135.0	±59.5	*	±77.4	±128.2	±10.9
1995	0	0	8.3	-	25.0	234.2	1399.4	-	1320.0	500.0	400.0	700.0	650.0	1000.0
			±8.2		±2.8	±39.3	±63.4		±25.1	±15.5	±92.9	±77.4	±21.0	±28.8
FALL (AGE-1)														
1994	0	13.5	0	0	31.3	7.7	111.8	-	634.4	14.9	0	0	1071.4	0
		±8.0			±16.3	*	±3.8		±60.2	±0.9			±103.6	
1995	0	40.0	0	-	33.3	81.1	138.0	-	700.0	128.6	0	0	850.0	83.3
		±7.1			±3.7	±13.6	±6.3		±13.3	±4.0			±27.4	±2.4

* indicates no variance calculated, all fish caught on first pass

Reach estimates were calculated from the mean of site abundance estimates to determine the contribution of each specific reach to the watershed production of steelhead parr. I calculated reach abundance of steelhead parr for the upper and lower Betsie River, Dair Creek, Little Betsie River, and the small tributaries. Density estimates for age-0 parr in Dair Creek, Little Betsie River, and the small tributaries were 2.5 to 18 times greater than the densities observed in the Betsie River (Figure 17). Estimates of age-1 parr densities for the tributary reaches ranged from 2.6 to 18 times their density in the Betsie River. The small tributaries were the exception in 1993 and 1994, when no age-1 parr were found.

Within the Betsie River, the upper and lower reaches also differed in parr density estimates. In all years, densities for both age-0 and age-1 parr were greater in the lower reach. In July, 1994 and 1996, very few parr were found in the upper reaches of the watershed and fewer parr were estimated in the lower reaches (Figure 18). A similar trend occurred in the October sampling. However, more parr were observed in the upper reach in October than in July (Table 9).

Table 9. Density (parr/ha) of age-0 and age-1 steelhead parr in the upper (1-4) and lower (5-8) reaches of the Betsie River, October 1994-1995.

Year	Age-Class	Upper River	Lower River
1994	Age-0	31.9 ± 15.3	846.0 ± 28.4
	Age-1	9.0 ± 3.6	59.8 ± 1.9
	Age-2	1.4 ± 0.8	0
1995	Age-0	8.3 ± 2.2	816.8 ± 37.3
	Age-1	18.3 ± 2.0	109.5 ± 7.5
	Age-2	7.1 ± 0.9	0

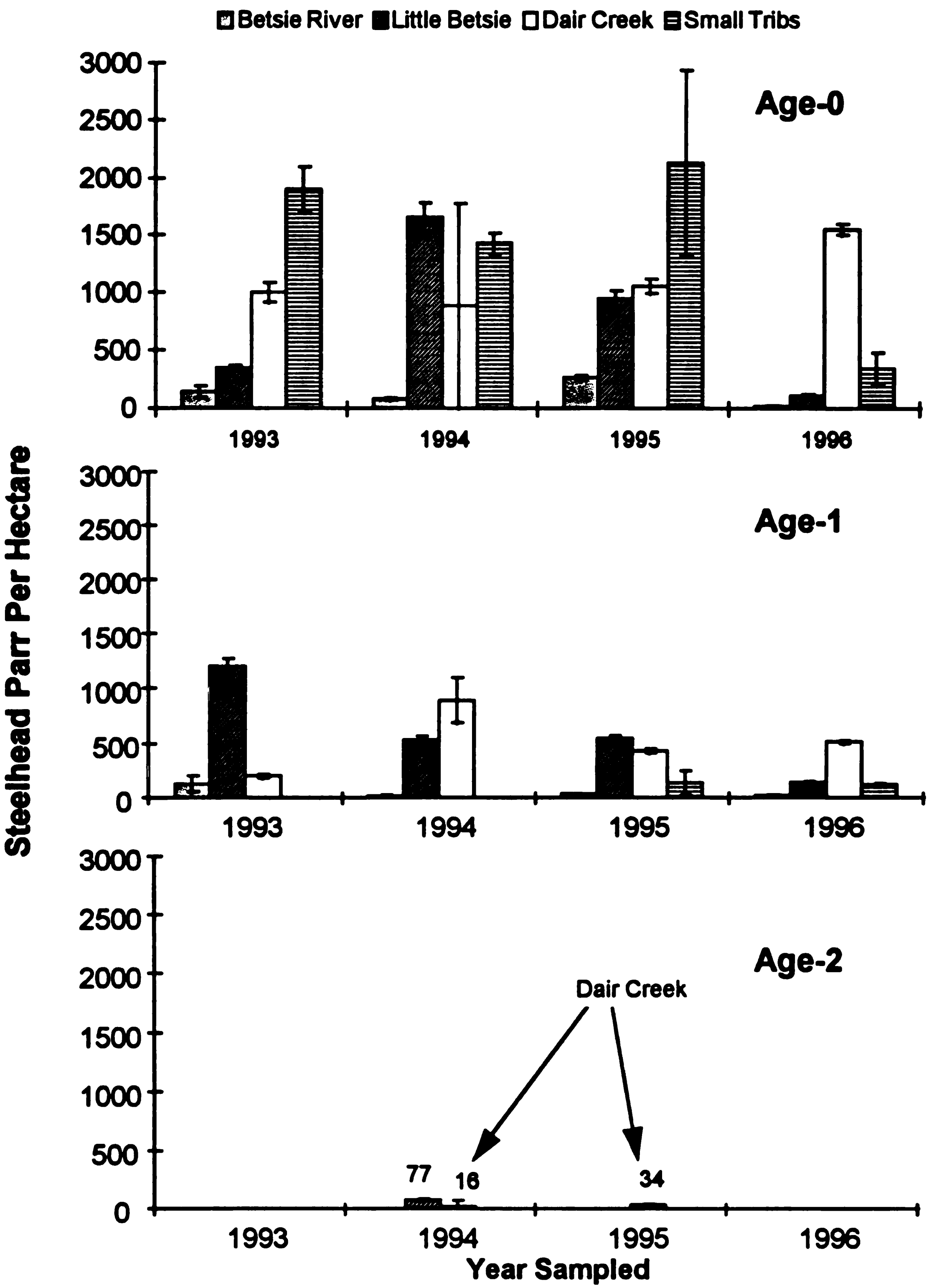

Figure 17 . Comparison of age-0, age-1, and age-2 steelhead parr densities between the Betsie River and its tributaries, July, 1993-1996.

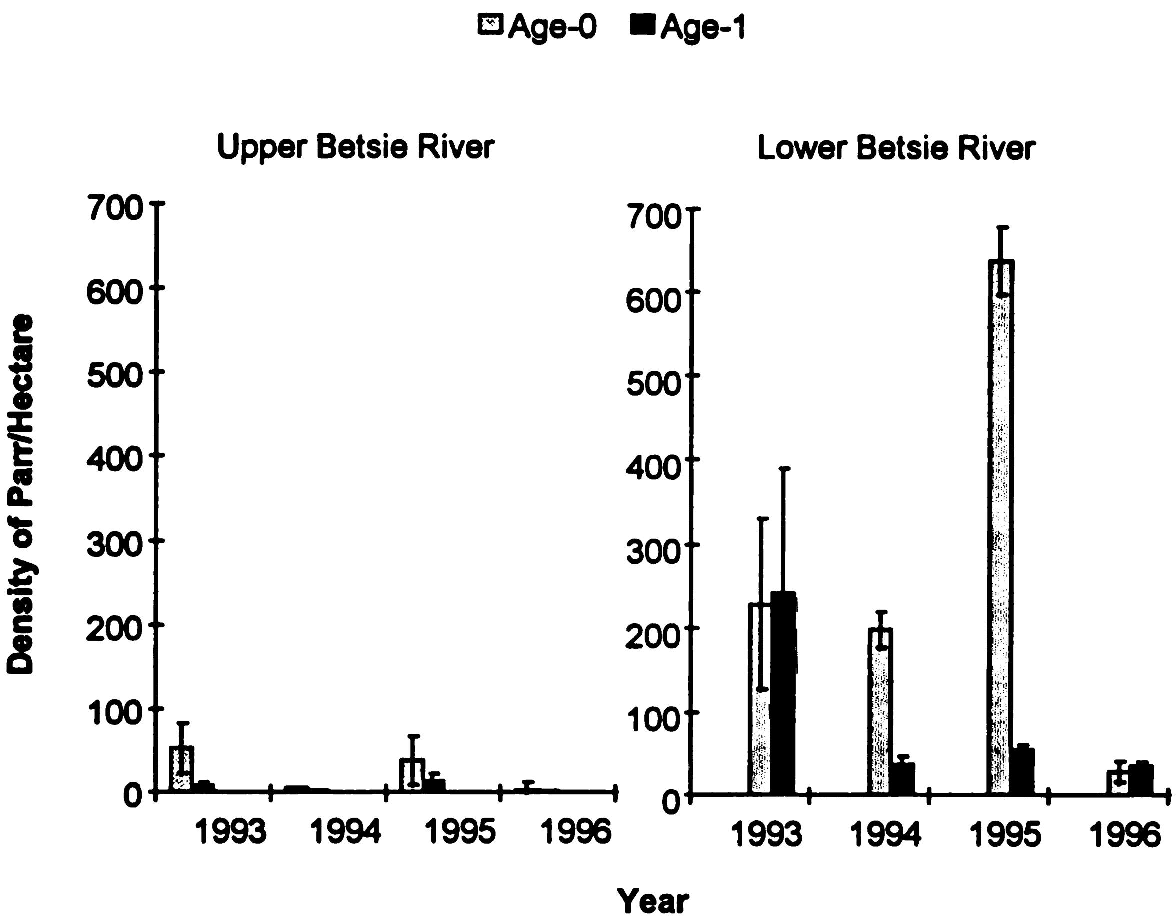

Figure 18. Comparison of the densities of age-0 and age-1 steelhead parr in the upper and lower reaches of the Betsie River, 1993-1996.

Although densities of parr were highest in the tributaries (Figure 17), when the estimates were extrapolated by area to account for total watershed production, the main channel Betsie River produces more age-0 steelhead (Figure 19). Using channel areas of 84.0 ha for the Betsie River, 5.9 ha for the Little Betsie River, and 4.5 ha for Dair Creek, the estimates of age-0 abundance for the Betsie River main channel range from 1,171 to 11,812 age-0 parr while the abundance in the Little Betsie River, Dair Creek and the small tributaries ranged from 314 ($\pm$ 41.2) to 9739 ($\pm$ 1462) age-0 parr (Figure 19). For age-1 parr, the differences in total abundance between the tributaries and the Betsie River are less and in some cases, a tributary contributed more age-1 parr. In the Betsie River, the total abundance of age-1 parr ranged from 1,232 ($\pm$ 2,610) to 10,505 ($\pm$ 3,721). The total abundance estimates in the Little Betsie River and Dair Creek range from 827 ($\pm$ 124) to 7,080 ($\pm$ 418 parr). In 1994 and 1995, the Little Betsie River abundance of age-1 parr was greater than in the Betsie River. The small tributaries held no age-1 parr in 1993 and 1994 and only a few in 1994 and 1995. When abundance estimates for all the sampled tributaries were combined, they contributed a mean of 56% of the total for age-0 parr and 67% of the total for age-1 parr, yet they comprised only 11% of the total channel area in the study reach (Figure 20).

In terms of watershed production, the overall watershed abundance of each age-class of steelhead parr varied throughout 1993-1996 (Table 11). Age-0 parr increased from 18,795 in 1993 to 32,896 in 1995, but fewer than 10,000 age-0 parr were estimated in 1996. In 1993, the largest abundance of age-1 parr was observed, 18,485, which was more than twice the estimates for 1994-1996. However, the large

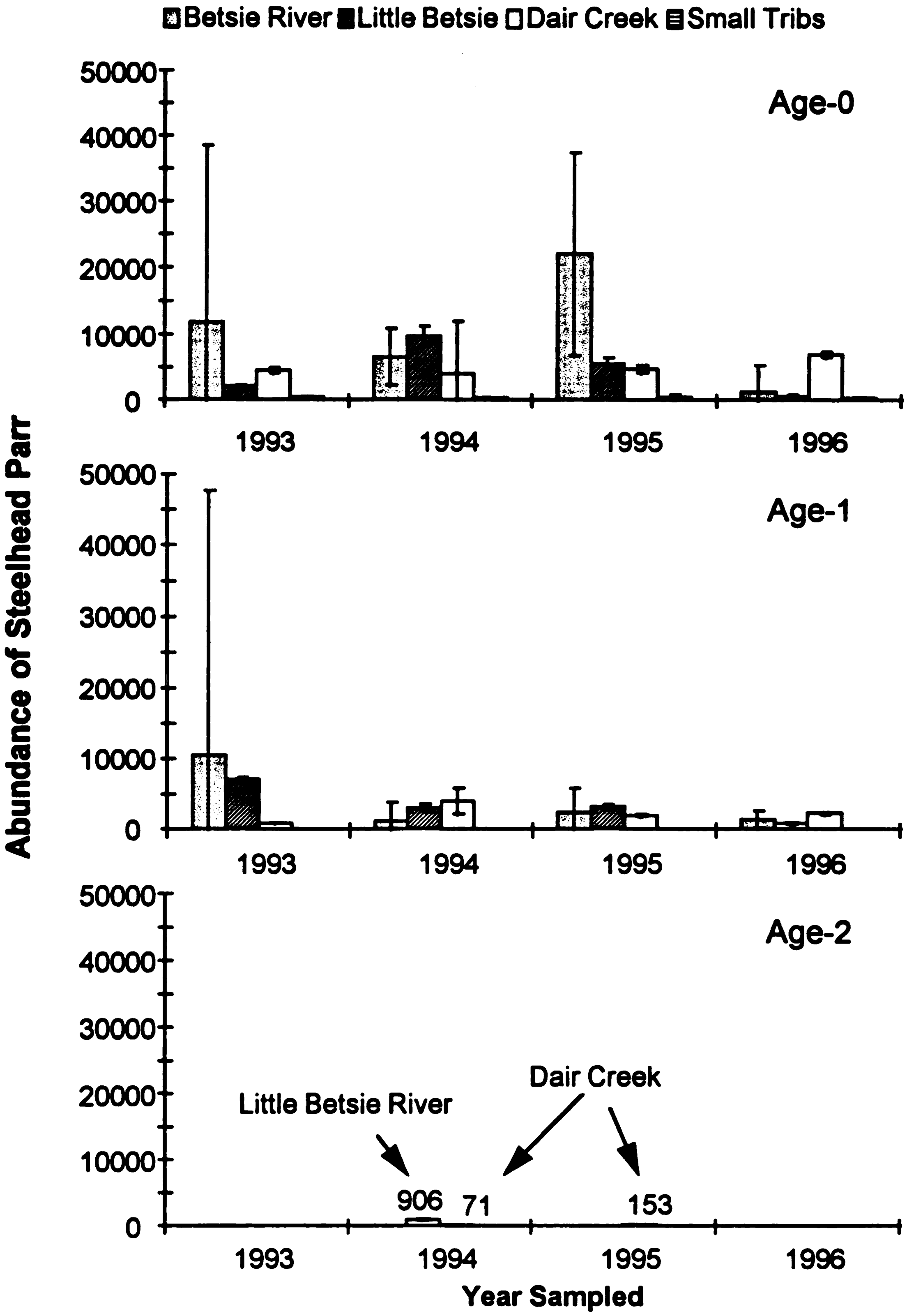

Figure 19. Estimate of abundance of steelhead parr in 4 reaches throughout the Betsie River watershed, July, 1993-1996.

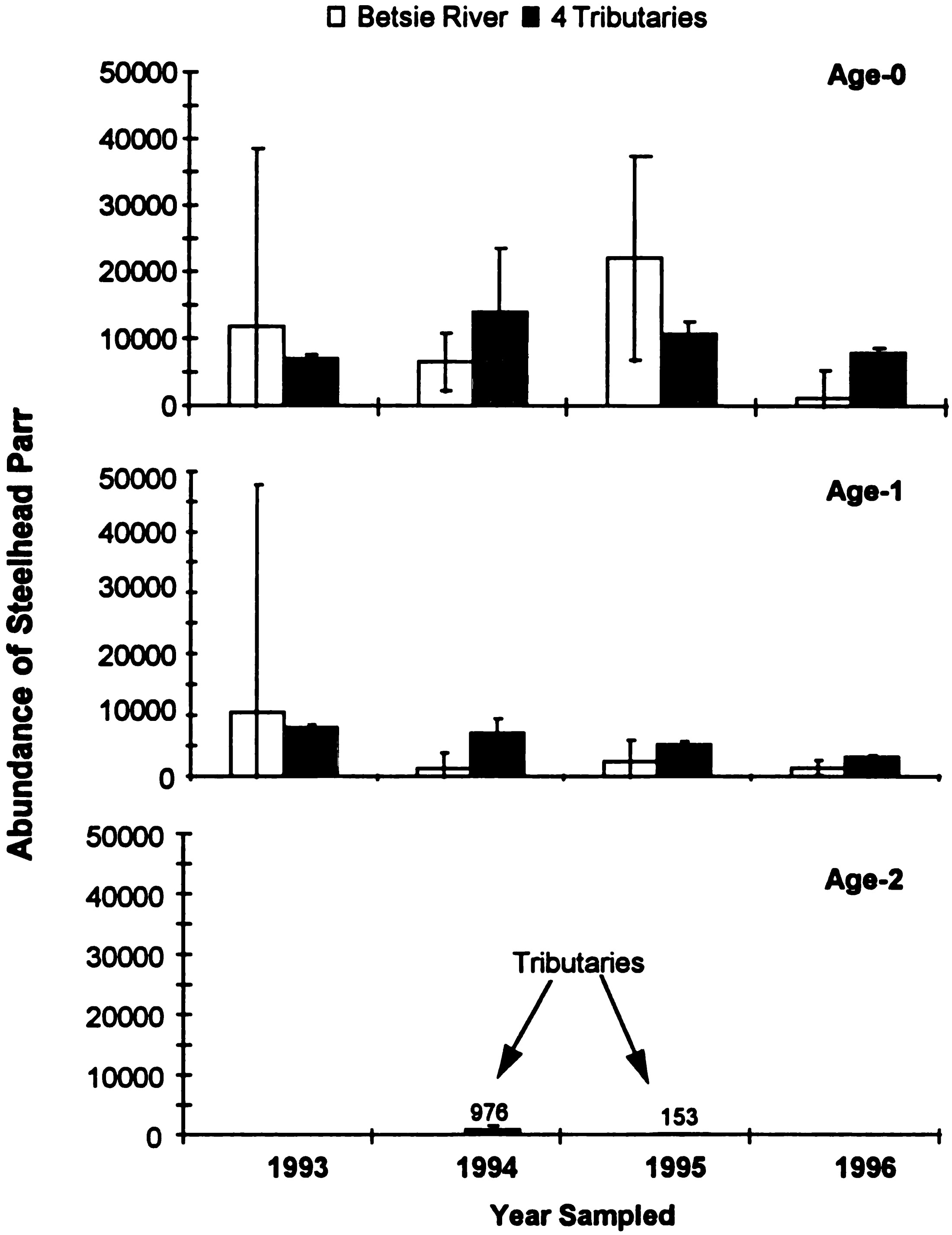

Figure 20. Comparison of abundance of steelhead parr in the Betsie River and the four tributaries that were sampled in the watershed, July, 1993-1994.

confidence interval should be noted for 1993. Watershed production estimates for age-2 parr were less than 1,000 fish.

Table 10. Watershed abundance estimates and 95% confidence intervals for age-0, age-1, and age-2 steelhead parr calculated from stratified reach estimates throughout the Betsie River watershed, July, 1993-1996.

Year of Sampling	Age-0 Parr	Age-1 Parr	Age-2 Parr
1993	18,795 ± 27,275	18,485 ± 37,708	0
1994	20,569 ± 13,778	8,357 ± 4,924	976 ± 549
1995	32,896 ± 17,081	7,621 ± 3,953	153 ± 16
1996	9,130 ± 4,789	4,505 ± 1,499	0

Number of Outmigrating Smolts

In May and June (1993-1996), I captured a total of 1,775 steelhead smolts by electrofishing. The mean length of wild smolts ranged from 188 to 191 mm and the age-structure of the yearly emigration was comprised of 48-54% age-1 fish, 40-46% age-2 smolts and 0-10% age-3 smolts. The spring of 1996 was different in that over 78% of the smolts captured were age-1 and only 20% were age-2. Less than 3% of the smolts in 1996 were age-3. Fewer than 3,000 wild steelhead smolts left the watershed each year (Table 8). The Betsie River produced smolts at watershed densities ranging from 11.9/ha in 1996 to 22.2/ha in 1993 with a mean of 17.5 smolts/ha.

Table 11. Age structure and watershed yield of wild steelhead smolts migrating from the Betsie River watershed, May-June, 1993-1996.

Year	Total	% Age-1	% Age-2	%Age-3	Smolts/Ha
1993	2,096 ± 716	54	46	0	22.2
1994	1,847 ± 1,152	48	44	8	19.6
1995	1,534 ± 1,804	53	40	7	16.3
1996	1,125 ± 1,441	78	21	2	11.9
MEAN	1,651	58	38	4	17.5

Mortality Rates

Annual instantaneous mortality rates (Z) were calculated for the juvenile steelhead in the Betsie River Watershed. I excluded losses due to spring emigration by including the number of smolts estimated each year from the cohort (Table 12). Annual mortality rates were higher for age 1-2 parr with a mean of 1.943. The mean annual mortality rate for parr age-0 to age-1 was 1.059. Mortality for age 2-3 parr was very large at 2.448.

Parr estimates of mortality were problematic for summer to fall in that more age-1 and age-2 parr were estimated for fall than late summer. This may be attributed to the fact that parr were more widely distributed throughout the watershed in the fall than in the summer. Watershed estimates of mortality from summer to fall were much larger in 1994 and 1995 and were also greater than for age-1 and 2 parr in 1994 (Table 12). In 1995, low flow and high summer temperatures most likely constrained parr to a few areas throughout the watershed and the larger fall estimates were due to larger abundance estimates throughout the entire watershed.

Table 12. Annual, winter, and summer instantaneous mortality rates (Z) for steelhead parr cohorts observed in the Betsie River watershed, 1993-1996. The annual mortality rate includes an adjustment for the cohort's emigrating smolts.

Year	Mortality (age-0 to age-1)	Mortality (age-1 to age-2)	Mortality (age-2 to age-3)
ANNUAL			
1993-1994	0.710	2.395	1.881
1994-1995	0.892	2.489	2.817
1995-1996	1.810	3.576	3.507
MEAN	1.137	2.820	2.735
JULY-OCTOBER			
1994	3.839	1.215	2.166
1995	0.185	*	*
OCTOBER-JULY			
1994-1995	1.842	3.212	1.395
1995-1996	2.396	4.633	4.396

*indicates more fish captured in October than in July

Within year rates of instantaneous winter mortality were calculated from October to July for years 1994-1995 and 1995-1996 (Table 12). An adjustment was also added to the summer estimates to reflect cohort survival in the emigrant smolts. Winter mortality rates were greater in 1995-1996 than in 1994-1995 for all age classes.

In addition to total mortality from the watershed level, mortality rates were also calculated individually for the Betsie River, the Little Betsie River, Dair Creek, and the small tributaries (Table 13). In this analysis, I could not adjust for emigrating smolts or

Table 13. Annual instantaneous mortality rates (Z) throughout the Betsie River watershed without adjustment for outmigrating smolts, 1993-1996.

		Mortality Age-0 to Age-1	Mortality Age-1 to Age-2
Betsie River			
	1993-1994	2.260	**
	1994-1995	0.987	**
	1995-1996	2.795	**
	MEAN	2.014	
Little Betsie River			
	1993-1994	*	2.056
	1994-1995	1.105	**
	1995-1996	1.910	**
	MEAN	1.508	
Dair Creek			
	1993-1994	0.116	2.545
	1994-1995	0.726	3.265
	1995-1996	0.722	**
	MEAN	0.521	

* larger estimate of age-1 than age-0
** no observations of age-2 parr

parr moving between reaches, therefore these rates should be interpreted as loss rates from the reach rather than mortality rates. Loss rates were the greatest for age-0 parr in the Betsie River and the least in Dair Creek. Loss rates for age-1 parr were high in the Little Betsie River and Dair Creek.

Steelhead Density, Mortality Rates and Temperature Variables

I analyzed several relations between parr density, smolt abundance and mortality rates with discharge and several indices of summer and winter severity. Significant negative correlations were found between the summer densities of age-0 and age-1 parr and the mean and maximum summer temperature (Figures 21 and 22).

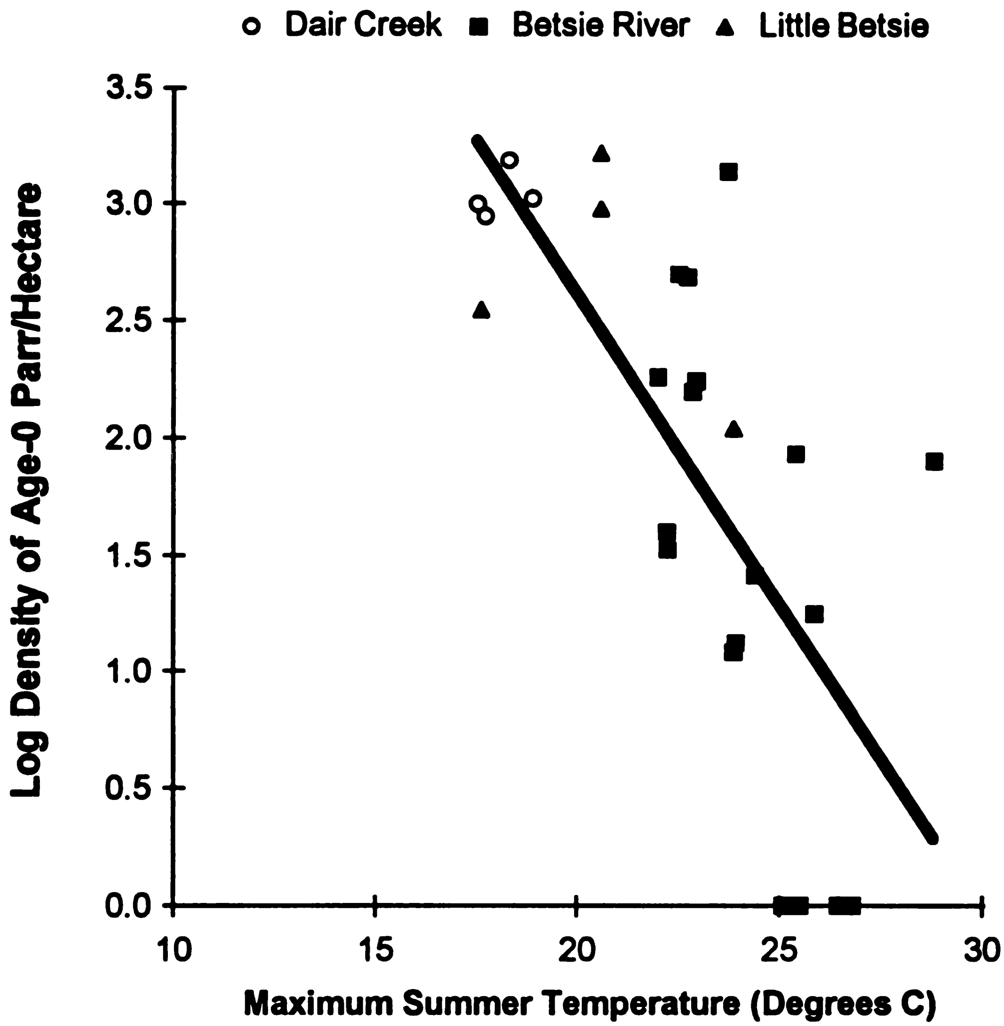

Figure 21. Correlation between the density of age-0 steelhead parr and the maximum summer water temperature at sites throughout the Betsie River Watershed, 1993-1996 (adjusted $r^2 = 0.4981$, $p = 0.00002$).

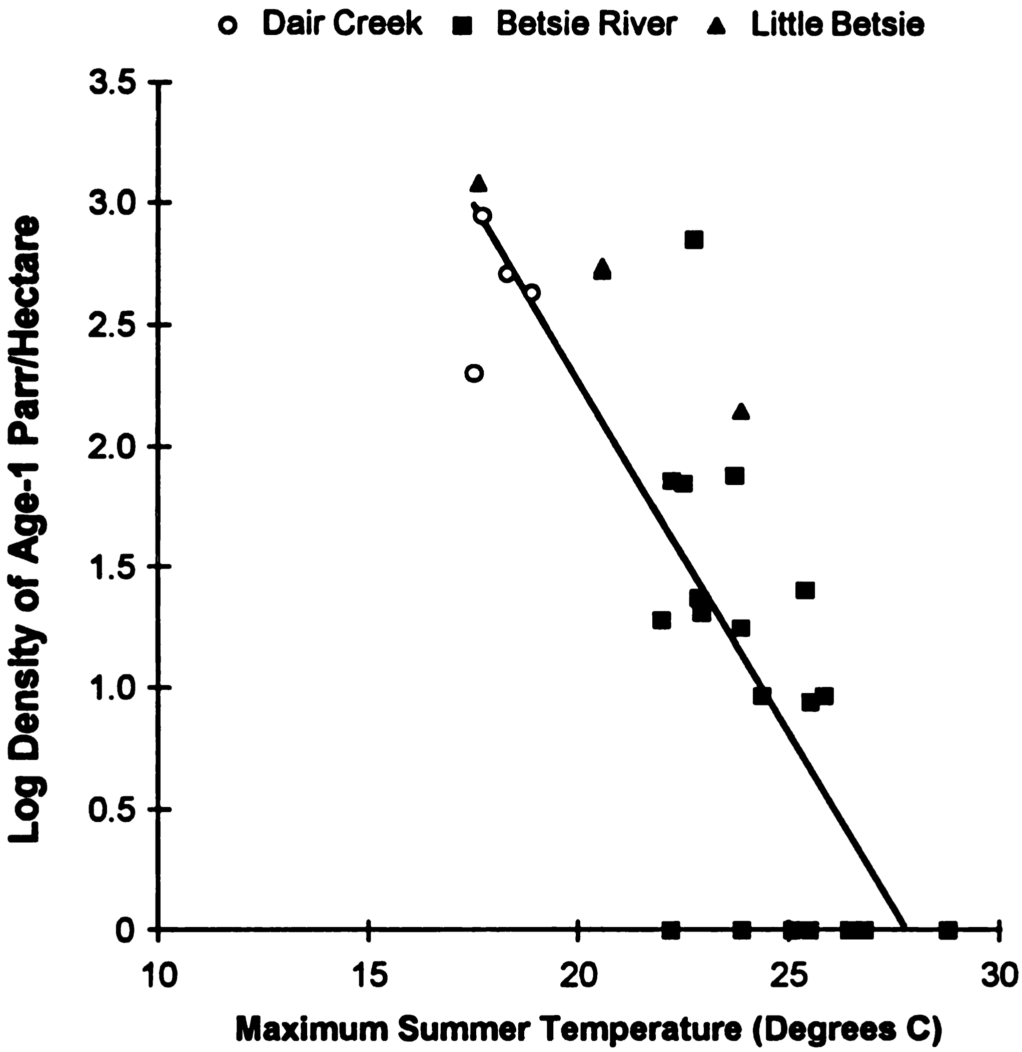

Figure 22. Correlation between the density of age-1 steelhead parr and the maximum summer water temperature at sites throughout the Betsie River Watershed, 1993-1996 (adjusted $r^2 = 0.6225$, $p = 0.000007$).

The correlations were the strongest with age-1 parr and maximum summer temperature.

Significant correlations also were observed with indices of winter severity. Weak relations were observed between the density age-1 parr and the prior winter minimum temperature (Figure 23) and mean winter temperature and the instantaneous mortality rate for reaches throughout the watershed (Figure 24). Using the number of days less than -12°C as a winter severity index, a significant negative relationship (adjusted $r^2 = 0.946$, $p = 0.018$) was observed between severe winters and the estimate of outmigrating steelhead smolts (Figure 25).

To evaluate the effects of spring flow during steelhead fry emergence, the summer estimate of age-0 steelhead was examined for a relationship with mean spring discharge in May and June, during steelhead fry emergence. I found a negative relationship between spring flow and the number of age-0 steelhead ($r^2 = 0.771$ and $p = 0.079$) (Figure 26).

Observed Differences in Fish Lengths

I used length as an index of growth, and tested for differences in length within a reach through the 4 years and among reaches each year for age-0 and age-1 steelhead parr. Too few age-2 steelhead were sampled to analyze differences in mean length. Because I found a significant interaction between year and reach, we analyzed the two factors separately. For age-0 parr, there were no significant differences in within-reach mean lengths for the Betsie River, Dair Creek, or the small tributaries from

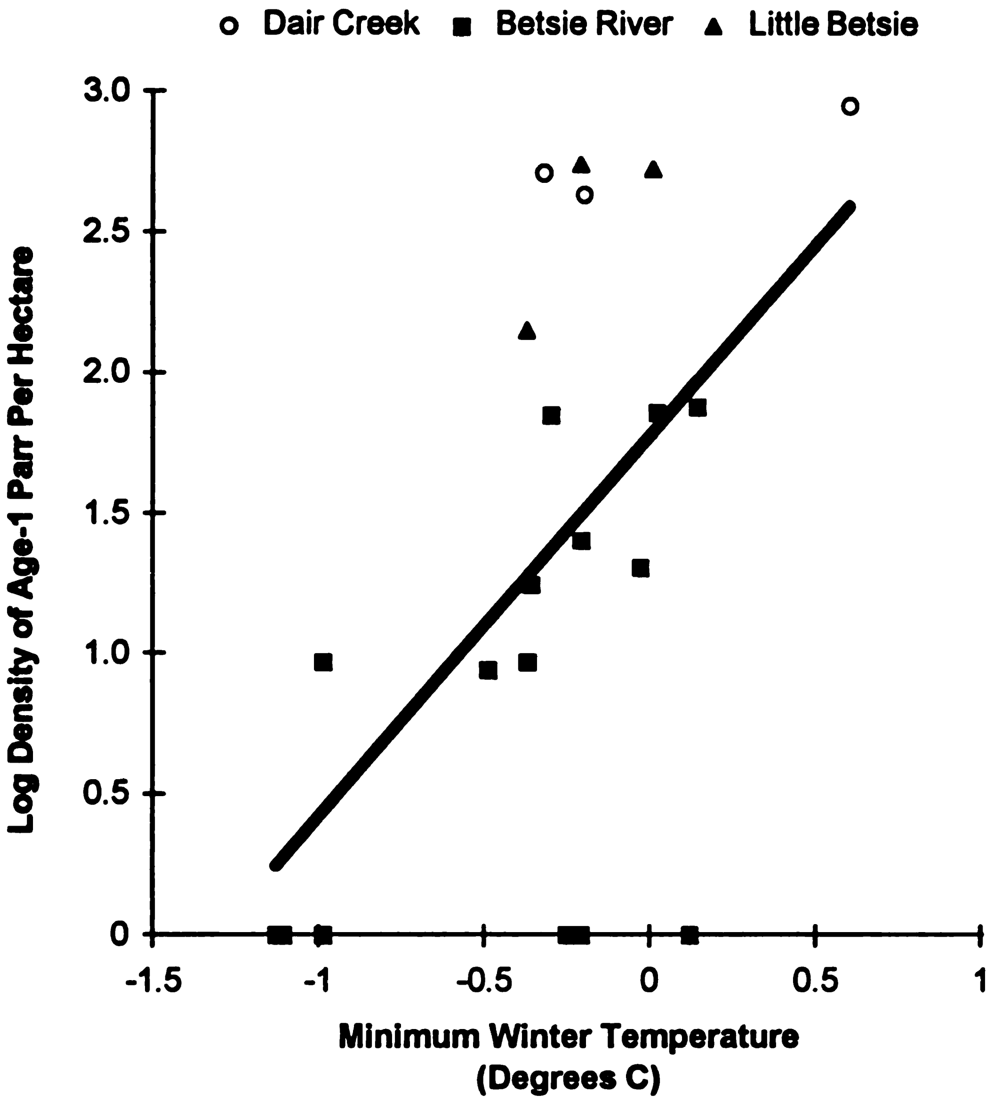

Figure 23. Correlation between the density of age-1 steelhead parr and the minimum water temperature at sites throughout the Betsie River Watershed, 1993-1996 (adjusted $r^2 = 0.3391$, $p = 0.0275$).

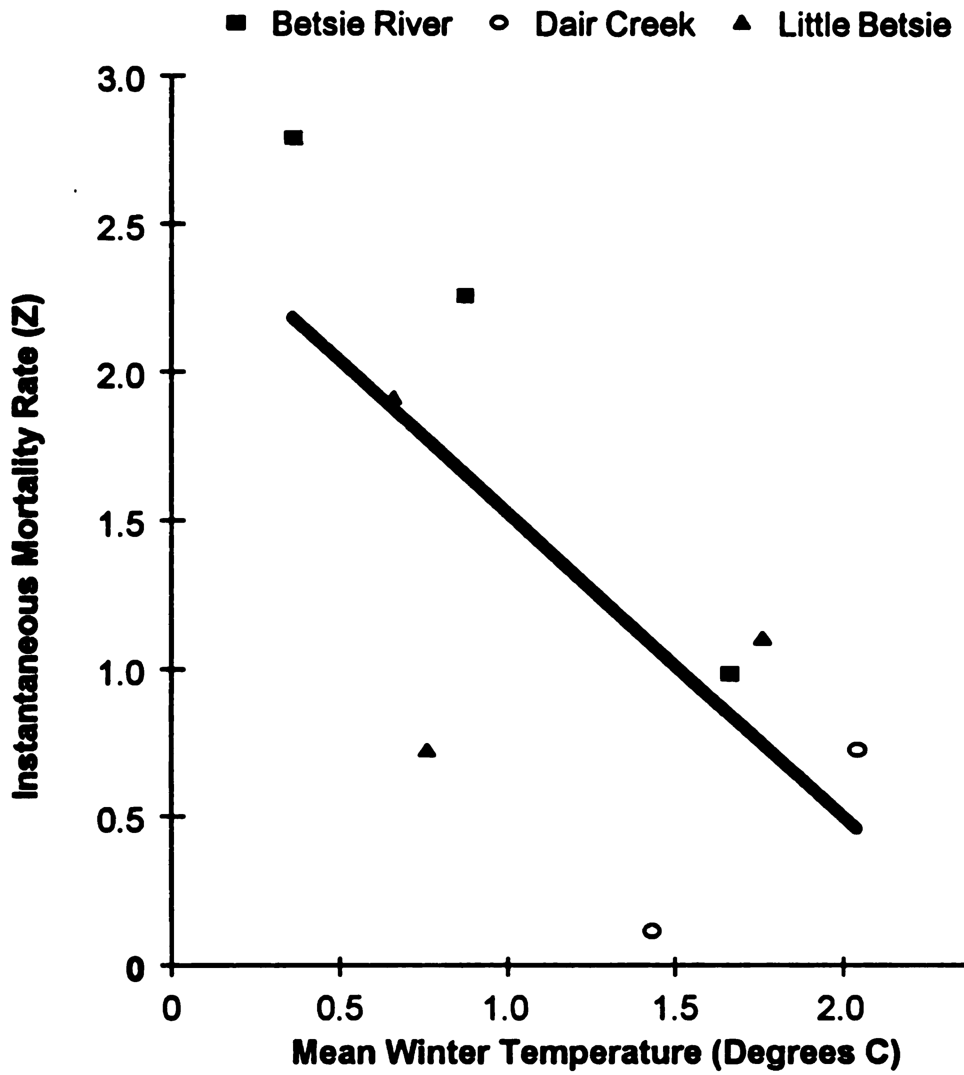

Figure 24. Correlation between mean winter temperature and the annual mortality rate for steelhead parr (age-0 to age-1) at sites throughout the Betsie River Watershed, 1993-1996 (adjusted $r^2 = 0.389$, $p = 0.0581$).

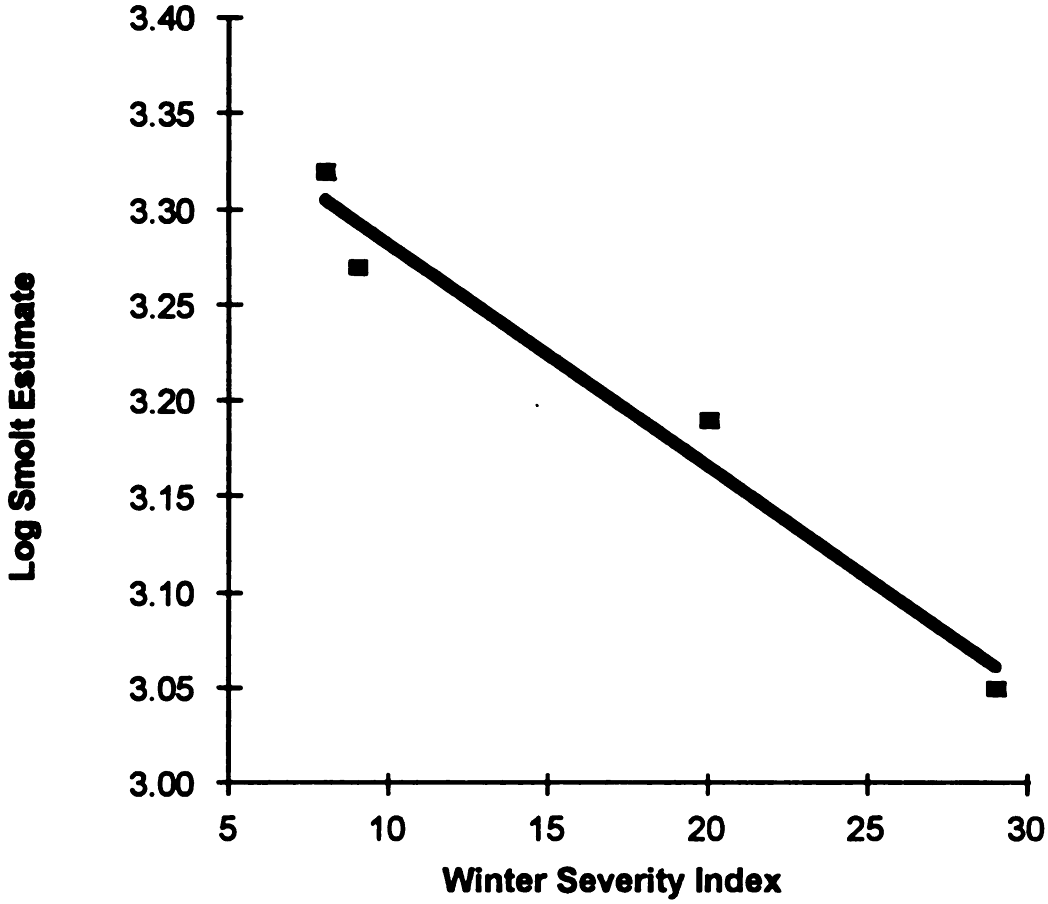

Figure 25. Correlation between the winter severity index (number of days when air temperature is less than -12°C) and the estimate of smolts leaving the Betsie River Watershed, 1993-1996 (adjursted $r^2 = 0.946$, $p = 0.018$)

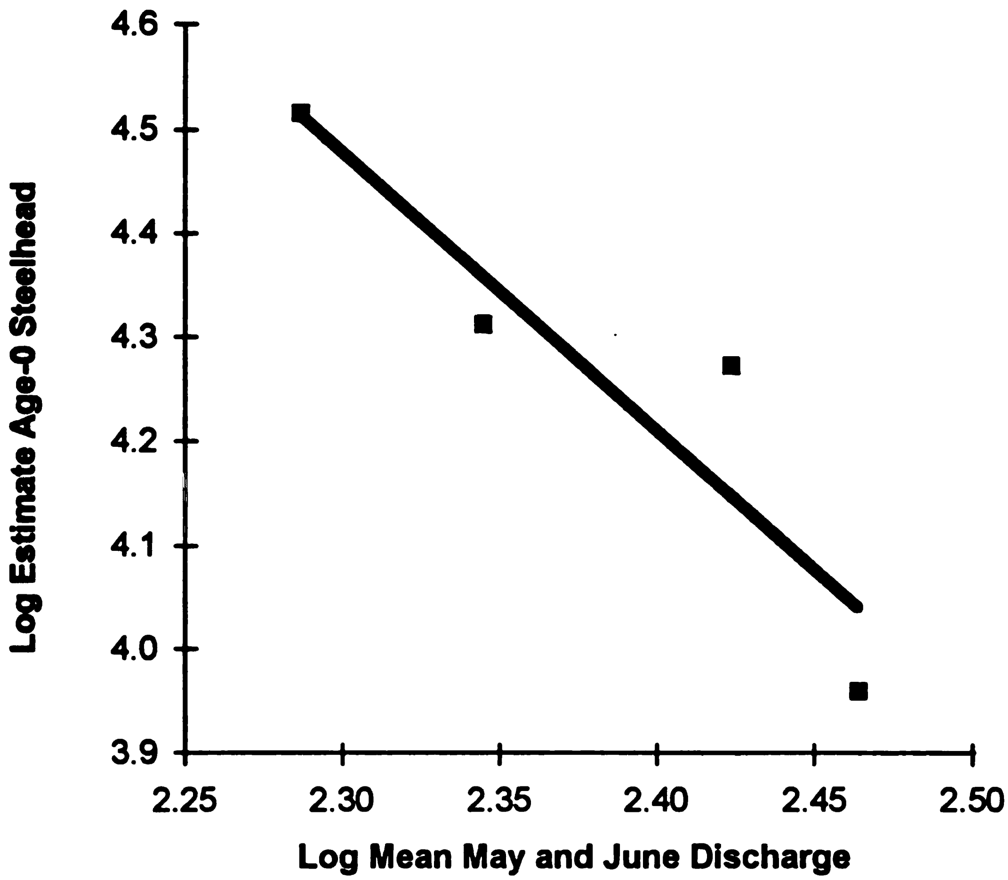

Figure 26. Correlation between the abundance of age-0 steelhead in July and the mean May and June discharge in the Betsie River watershed, 1993 - 1996 (adjusted $r^2 = 0.771$, $p = 0.079$).

1993 to 1996 (Table 15). In the Little Betsie River, mean lengths were significantly larger in 1993 and significantly smaller in 1996 than in 1994 and 1995. Between reaches, mean length was significantly larger in the Little Betsie River than the other three reaches. In 1993, mean length in Dair Creek was also significantly larger than in the other reaches.

Age-1 steelhead parr were more variable in their mean length. Within reaches, mean lengths were significantly larger in 1993 and 1995 in the Betsie River and in 1996, mean lengths were significantly larger in the Little Betsie River (Table 14). Between reaches, in 1995, age-0 and age-1 parr in Dair Creek and the small tributaries were smaller than the Betsie River and the Little Betsie River, while in 1996, age-1 parr were significantly larger in the Little Betsie River and significantly smaller in the small tributaries.

Table 14. Mean lengths and 95% confidence intervals for age-0 and age-1 steelhead parr sampled from four reach categories throughout the Betsie River watershed, 1993-1996.

Year	Betsie River	CHANNEL Little Betsie River	Dair Creek	Small Tributaries
SUMMER				
AGE-0				
1993[1]	55.8 ± 8.4	77.0 ± 12.2*	80.6 ± 10.6*	53.2 ± 5.8
1994	50.9 ± 7.3	62.4 ± 6.8	47.2 ± 8.0	49.7 ± 6.2
1995	52.9 ± 6.4	61.7 ± 6.0	48.5 ± 9.4	47.6 ± 7.5
1996	50.9 ± 6.5	52.1 ± 4.5*	43.8 ± 7.4	44.6 ± 7.2
AGE-1				
1993[1]	155.2 ± 8.4*	148.1 ± 8.7	128.0 ± 6.2	0 age-1 parr
1994	137.8 ± 8.2	145.9 ± 8.2	149.6 ± 8.0	0 age-1 parr
1995	152.3 ± 8.8*	152.8 ± 8.9	132.1 ± 8.7	127.3 ± 11.0
1996	138.3 ±10.0	168.4 ± 9.5*	132.3 ± 8.5	118.5 ± 6.3
FALL				
AGE-0				
1994	89.1 ± 7.4*	94.4 ± 7.3*	73.9 ± 7.9	74.2 ± 7.7
1995	83.3 ± 6.6	87.2 ± 6.8	71.2 ± 6.9	67.5 ± 7.7
AGE-1				
1994	154.4 ± 8.5	152.9 ± 7.3	139.1 ± 9.2	0 age-1 parr
1995	157.5 ± 10.8	166.2 ± 10.1*	150.2 ± 10.3	0 age-1 parr

[1] Sampling dates 10 days later in the summer than 1994-1996.

* indicates a significant difference (P < 0.05) within the reach among the years for the age-class

Different thermal rearing conditions may explain some of the observed differences in parr length. I combined site specific temperature with optimum growth rate information to compare the amount of time spent in the optimal growth conditions between sites (Figure 27). Based on information found in references (Wismer and Christie 1987, Hokanson et al. 1977), I defined the optimal growth range as 15-17°C with an upper growth limit of 23°C and lower lethal limit of 0°C. In the upper Betsie River (Sites 1 and 4), steelhead parr spent very little time in the optimal growth zone and more time in water with temperatures above the optimum limits, approaching the zero growth limit. In the lower Betsie River (Sites 7 and 8), the trend was similar, yet parr spent more time during the summer near and slightly above the upper optimum growth limit. In the tributaries (Sites 9 and 14), the difference in thermal regimes had varied results for parr. In these sites, parr spent the most time in their optimum growth zone (> 4 months) and never exceeded the upper growth limit. During the winter months, there were few differences in time spent near 0°C in the lower Betsie River and the tributaries, and slightly more time was spent near the 0°C limit in the upper Betsie River. Little Betsie River was warmer than Dair Creek which may have resulted in the larger mean lengths observed there.

Discussion

Life history characteristics which vary from the norm may indicate that rearing conditions such as extremes in thermal variability are facilitating trends in the population. Over half of the emigrating smolts left the Betsie River watershed at age-1. In most other Michigan streams, the majority of smolts leave at age-2 (Biette et al.

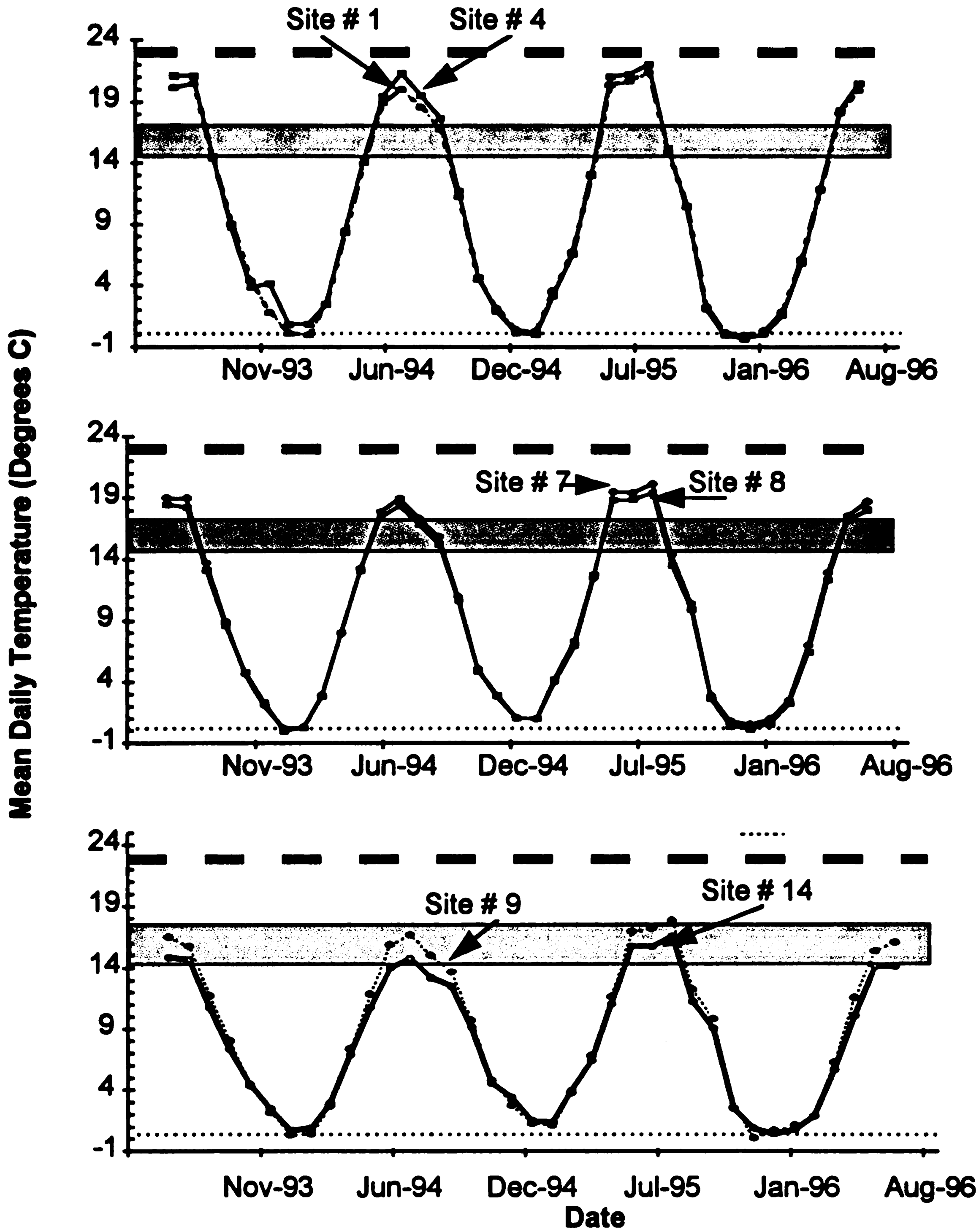

Figure 27. Comparison of mean monthly water temperature among 6 sites in the Betsie River watershed. The gray shaded area respresents the optimal growth range for juvenile steelhead, the heavy dashed line is the upper zero growth level (23° C), the light dashed line is the lower lethal level (0°C).

1981, Seelbach and Miller 1993). Furthermore, although the Betsie River smolts are leaving at a younger age, their mean length (188-190 mm) is typical of the physiological size for steelhead smoltification (Hoar 1988) and also similar to lengths observed for populations in which most fish leave at age-2 (Biette et al. 1981, Seelbach 1993). These population attributes combined with the high rate of homing often observed in steelhead, may indicate that the Betsie River steelhead has the potential to become a discrete stock (Biette et al. 1981). Further study is required to determine if the faster growth observed in parr throughout the watershed is a bioenergetic manifestation or the result of phenotypic selection for faster growth rates in an environment where survival beyond year 1 is highly improbable .

Large smolts are advantageous and can result in a high rate of return as adults. For example, Ward et al. (1989) found that mean lengths of outmigrating smolts from the Keogh River, age-2 age-3, and age-4 were 153, 177 and 218 mm. However, back-calculation of the returning adults showed that mean lengths for age-2, age-3, and age-4 were 177, 196, and 220 mm. Survival for smolts averaging 140 mm was only 2-3% , but was 37% for 220 mm smolts. Similarly, Seelbach et al. (1994) observed a return rate of 10- 30% for smolts stocked at lengths greater than 200 mm. In the late 1980's, the Betsie River fishery was found to be over 90% wild fish even though 20,000 steelhead juveniles were stocked in the river each year (Seelbach and Whelan 1988). Betsie River smolts at age-l, 2, and 3 average 170, 188, and 203 mm and it is likely that these larger wild smolts also return at high rates.

Densities of steelhead parr varied widely throughout the Betsie River watershed and can be compared with other studies and tributaries to the Great Lakes (Table 15).

The production of parr in the Little Betsie River and Dair Creek is comparable with Sand Creek and Silver Creek located in southern Michigan (Dexter 1993a, 1993b). These streams were also found to have parr with growth rates greater than the average growth found throughout the state. We found lower densities of in the main channel Betsie River than what were found in the Little Manistee River in northern Michigan and also in the southern Michigan streams, Sand and Silver Creek.

Table 15. Comparison of steelhead parr densities in select streams.

General Location	Stream	Mean Q (cms)	Total Steelhead Parr/ha	Density per hectare Age-0	Age-1	Age-2
Southern Michigan	Sand Creek	0.14	1,536	1,045	476	15
	Silver Creek	0.14	1,818	436	1,345	36
Ontario Canada	Normandale Creek	0.14	600 - 2,200			
Northern Michigan	Pine Creek	0.60	4,500			
	Little Manistee	5.0-6.0		2,295	663	10
	Little Betsie	0.5		764	604	19
	Dair Creek	0.4		1,123	508	25
	Betsie River	5.5		124	46	0

Incorporation of naturalized production of steelhead in Great Lakes fishery management plans is a fairly recent approach. The basic premise in management of naturalized steelhead populations is that the protection of spawning and rearing habitat can maintain a population and is cheaper and more cost effective than restoration or rehabilitation that often has mixed results (Biette et al. 1981, Peck 1992). Knowledge of the important rearing areas and the limiting mechanisms to juvenile steelhead production throughout the Betsie River watershed can help to prioritize protective and rehabilitative actions to preserve and improve these critical habitat units. By using a watershed approach and considering all tributaries as well as the main channel in estimating parr abundance, I was able to determine the spatial distribution of juvenile steelhead and identify specific reaches of the watershed that were important to overall production. In a watershed similarly limiting by high water temperatures, Roper et al. (1994) also used a basin approach to determine the distribution and abundance of salmonids. They found that abundance estimates based on densities from a limited area in the watershed could have overestimated the juvenile steelhead population by a magnitude of 5 and that significant proportions (25%) of the older juvenile steelhead population were constrained within 12% of the total stream area. While they sampled throughout the entire length of the stream, they did not incorporate tributary streams in their analysis. Similarly in the Betsie River, over 50% of the age-0 and age-1 steelhead parr were found in the tributaries which comprise only 11% of the total channel area.

Tributaries are key to the production of juvenile steelhead in the Betsie River watershed. Not only are the preferred thermal environments provided in the tributaries, but there may also be an advantage to growth. Peterson and Rabeni (1996) observed 2 populations of sunfish in a mainstem river and a tributary and found that the fish that both resided in and seasonally moved into the thermally moderated tributary fed more frequently and consistently which resulted in larger fish. Similarly in the Betsie River watershed, larger juvenile steelhead were consistently observed in the Little Betsie tributary and in some years, movement from the mainstem into the tributaries seemed likely.

Steelhead smolt production is highly variable in some cases. Production of smolts in the Little Manistee River ranged from 11,845 to 86,425 smolts per year (97 - 713 smolts/ha), yet a consistent number of age-1 parr were produced each year (Seelbach 1993). Huron River, tributary to Lake Superior, is in a harsher winter environment, but is a stream similar in size to the Little Betsie. This stream was found to produce between 1,031 and 9,141 steelhead smolts (46 - 262 smolts/ha) (Seelbach and Miller 1993). In contrast, the Betsie River produced a limited number of smolts, 1,125 - 2,096 but the abundance estimates were fairly consistent among years. Density estimates for smolt production from the Betsie River watershed were less than half of the Huron River estimates, ranging from 11.9 to 22.0 smolts/ha.

When compared with high quality trout streams, the production of steelhead in the Betsie River is very limited, yet historically this population has supported a popular steelhead fishery. Because the watershed is recovering from a major system disruption as a result of Thompsonville Dam failure in 1989, the current production estimates may underestimate prior production or future potential in the mainstem Betsie River.

After a similar experience with a dam collapse in the Pigeon River, Michigan, high silt loading and sand introduction reduced the population of trout < 200 mm by 52% (Alexander and Ryckman 1986). Suspended solids have also been shown to cause sublethal stress effects in yearling steelhead (Redding et al. 1987). In the Betsie River, in addition to the initial impact of Thompsonville Dam failing, resulting in a channel disequilibrium, stream banks continue to erode and contribute sand which is highly mobile and observable as bedload movement during short time frames (Newcomb, unpublished data). This mobile sand bedload may be acting to suppress invertebrate populations (Alexander and Hansen 1986) and in concert with high temperatures, could result in metabolic limitations for the population of steelhead parr in the main channel. Through time, as the channel bedload reduces as a result of the channel again approaching equilibrium, the mobile bedload may diminish and juvenile steelhead production may increase. Further inquire is required to determine if in fact, this scenario is occurring.

Mean maximum summer water temperatures appeared to limit the distribution of juvenile steelhead throughout the watershed. Age-1 fish in particular showed a stronger relationship between maximum summer temperature and site densities. These results are similar to Roper et al. (1994) and are expected given the ease of fish movement throughout the Betsie River. For example, at site 2, few age-1 steelhead were found in July, yet in October 1994 and 1995, large numbers of age-1 wild and hatchery fish were found here. This site is 10 km upstream from the hatchery stocking location and from areas of significant densities of age-1 parr. Spawning activity and steelhead redds were observed in this upper reach, yet few age-0 parr were captured here in July.

In the drought summer, 1995, summer temperatures were greatly increased throughout the watershed. This year also had the largest age-0 year class measured in the 4 years with a very low mortality estimate for summer to fall of 1995. Parr mean length in the fall for age-0 fish was smaller than lengths measured in 1994 and larger for age-1 parr. Although the exact mechanisms for this observation are not known, it may be that either density dependent factors limited the growth of age-0 fish or that growth was limited by the high metabolic costs of surviving in warm water. On the other hand, a larger number of age-1 parr reside in the Little Betsie River and Dair Creek, which approached the upper optimal growth limits without going above and incurring the higher metabolic costs.

Although summer temperatures may limit distribution of steelhead parr in the Betsie River and its tributaries, winter effects may limit smolt abundance and cohort size. In studying a Michigan stream, Kocik (1992) found mortality rates (Z) of 1.07 during the growing season and higher winter rates of 1.57 for age-0 steelhead. His results compare favorably with the mortality rates in the Little Betsie River for age-0 steelhead which average 1.51 and results from the Betsie River as annual instantaneous rates for the entire watershed ranged from 0.71 to 1.81 with a mean of 1.14. Early winter acclimatization to cold temperatures has been shown to have a large influence on juvenile salmonids due to physiological limitations in early winter (Cunjak 1988). Furthermore, the more dramatic and sudden the winter cooling the greater the negative influence on the fish and this may give insight to the differences in mortality rates observed in the tributaries, where groundwater moderates the rapidly cooling air temperature, versus the mainstem Betsie River. Areas influenced by groundwater may be the key to winter habitat throughout the Betsie River watershed and those areas are confined to the tributaries and the main channel reach between sites 5-8.

Winter microhabitat may also be limiting in the Betsie River watershed. The important winter microhabitat observed in other streams such as cobble-boulder rubble along stream margins (Everest and Sedell 1983, Meyer and Griffith 1997) are limited in the Michigan watershed. Boulders and cobble are rare anomalies and woody debris along stream margins is really the only form of complex habitat available (Newcomb, unpublished data). Complex woody debris has been shown to be important to overwintering steelhead (Heifetz et al. 1986), the woody debris in the Betsie Main channel is not abundant. The absence of complex woody debris can be attributed I part due to the logging history of the watershed that has left few old stands in the riparian zone to contribute as windfall and clearing activities for canoe passage.

Other studies on the effects of winter severity on juvenile salmonid abundance have had mixed results. Overwinter mortality in the Little Manistee River ranged from 13 to 90% for juvenile steelhead and was related to the severity of winter (Seelbach 1987). In the Au Sable River, Michigan, Nuhfer et al. (1994) could not find a significant relationship between standing stocks of age-0 brown trout and winter severity as an index of air temperature. However, they did not analyze any older age-classes and the winter occurred during the egg incubation phase.

The effects that adult returns or broodstock size has on the population of juvenile steelhead in the Betsie River is unknown. There are no recent surveys to estimate fish harvest or return. In the late 1980's, creel harvest rates were estimated to range between 1,000 and 3,000 steelhead per year (Rakoczy and Rodgers 1987, 1988, 1990). However, the adult spawning population or total numbers in the return was not surveyed.

Conclusions

The production of wild juvenile steelhead in the Betsie River watershed is limited when compared with other steelhead producing streams. Although densities of juvenile steelhead vary both spatially and temporally throughout the watershed, consistent numbers of smolts are produced. Tributaries are a major component to the overall production of steelhead in this river that can be thermally marginal for trout and winter temperatures appear to influence mortality rates which are especially large for older fish. By the use of a watershed approach to assessing production, we were able to quantify areas critically important for steelhead based on thermal limitations and parr distribution.

CHAPTER 3

ASSESSMENT OF MANAGEMENT ALTERNATIVES FOR ALTERING THE THERMAL REGIME OF THE BETSIE RIVER, MICHIGAN

Abstract

The Betsie River in northern Michigan is classified as a marginal trout stream because in some reaches, instream summer temperatures exceed the tolerance limits for trout species. In 1989 Thompsonville Dam failed, which resulted in an additional 22 km of main channel accessible to migratory steelhead (*Oncorhynchus mykiss)* adults and juveniles. However, water temperatures in the upper watershed may still limit the production and distribution of these fish. The objectives of this study were to: 1) describe the current thermal regime throughout the Betsie River watershed and characterize it in relation to juvenile steelhead life history requirements, 2) evaluate empirical water temperature models based on air temperature to predict instream temperatures throughout the watershed, 3) develop a watershed physical process model to describe and quantify juvenile steelhead macrohabitat based on thermal criteria, and 4) develop a stream-reach physical process temperature model to evaluate the thermal effects of removing a remaining low-head dam in the headwaters or, alternatively, the addition of cold water from the hypolimnion of the source lake. Under current channel conditions, summer temperatures in the upper Betsie River routinely exceed the optimal growth limits for steelhead and sometimes the upper incipient lethal level, with mean summer temperatures from 21-23°C and maximum daily temperatures

up to 28°C. Although instream water temperatures at sites in the watershed were strongly correlated with air temperatures (r^2 = 0.81- 0.92), air temperature was a poor predictor of winter water temperatures. The stream network model provided temporal and spatial predictions of the weekly mean thermal regime in the Betsie River. During summer months, the preferred thermal habitat for juvenile steelhead was limited to less than 52% of the main channel area. The stream-reach model predicted instream summer temperatures on a daily time-step and provided for simulation of management alternatives for reducing instream summer temperatures. Removal of Grass Lake Dam would likely result in mean daily summer water temperatures 2°C lower than under current conditions in both typical-flow and low-flow water years for the reach from Green Lake to Grass Lake. In the Grass Lake to Thompsonville reach, mean daily water temperatures were predicted to be less than 1°C lower than under current conditions in a typical and low-flow year. The addition of hypolimnetic water would result in temperatures 3.6°C lower than under current conditions in the Grass Lake to Thompsonville reach. Although these management alternatives may provide better thermal habitat for juvenile steelhead during the summer in this 22 km reach of river, the trade-offs, including changes in the level of Green Lake, the loss of wetland habitat and fishery and recreational boating opportunities in the current Grass Lake impoundment areas should be considered.

Introduction

Water temperature is an important and controlling influence on fish ecology. Temperature influences species distributions, timing of life history events, and growth and survival (Hokanson et al. 1977, Adams and Breck 1990, Griffith 1993). When salmon and trout reside in the upper limits of their thermal tolerance, their growth rate can be very high, but growth rates rapidly decline when temperature exceeds the upper threshold (Griffith 1993, Elliott et al. 1995).

The Betsie River, a tributary to Lake Michigan, located in northwestern lower Michigan (Figure 28), drains approximately 67,126 ha comprised mainly of glacial outwash with predominantly sandy soils (Gooding 1995). The main river channel is classified as a marginal trout stream because summer temperatures in reaches of the river exceed the limits for growth and survival of most trout species. Temperature in the two largest tributaries in the watershed, Dair Creek and Little Betsie River, is better suited for trout species because large groundwater contributions moderate ambient temperature. Although brown trout (*Salmo trutta)* are stocked in the Betsie River, their numbers are very low (Wicklund and Dean 1958, MDNR Fish Surveys 1990). Rather, the greater portion of the fish species assemblage in the Betsie River is comprised of suckers (*Catostomus commersoni*, *Moxostoma erythrurum*), chubs (*Semotilus atromaculatus, Nocomis biguttatus*), and minnows *(Notropis cornutus*, *Rhinichthys atratulus*) (Wicklund and Dean 1958, Bonham 1975). In spite of this marginal status, the river has historically supported a popular fishery for migratory adult steelhead (Wicklund and Dean 1958, Rakoczy and Rogers 1987, 1988, 1990).

Prior to the early 1970's, the river was impounded by 3 dams: Homestead, Thompsonville, and Grass Lake. Homestead Dam, a former power generating

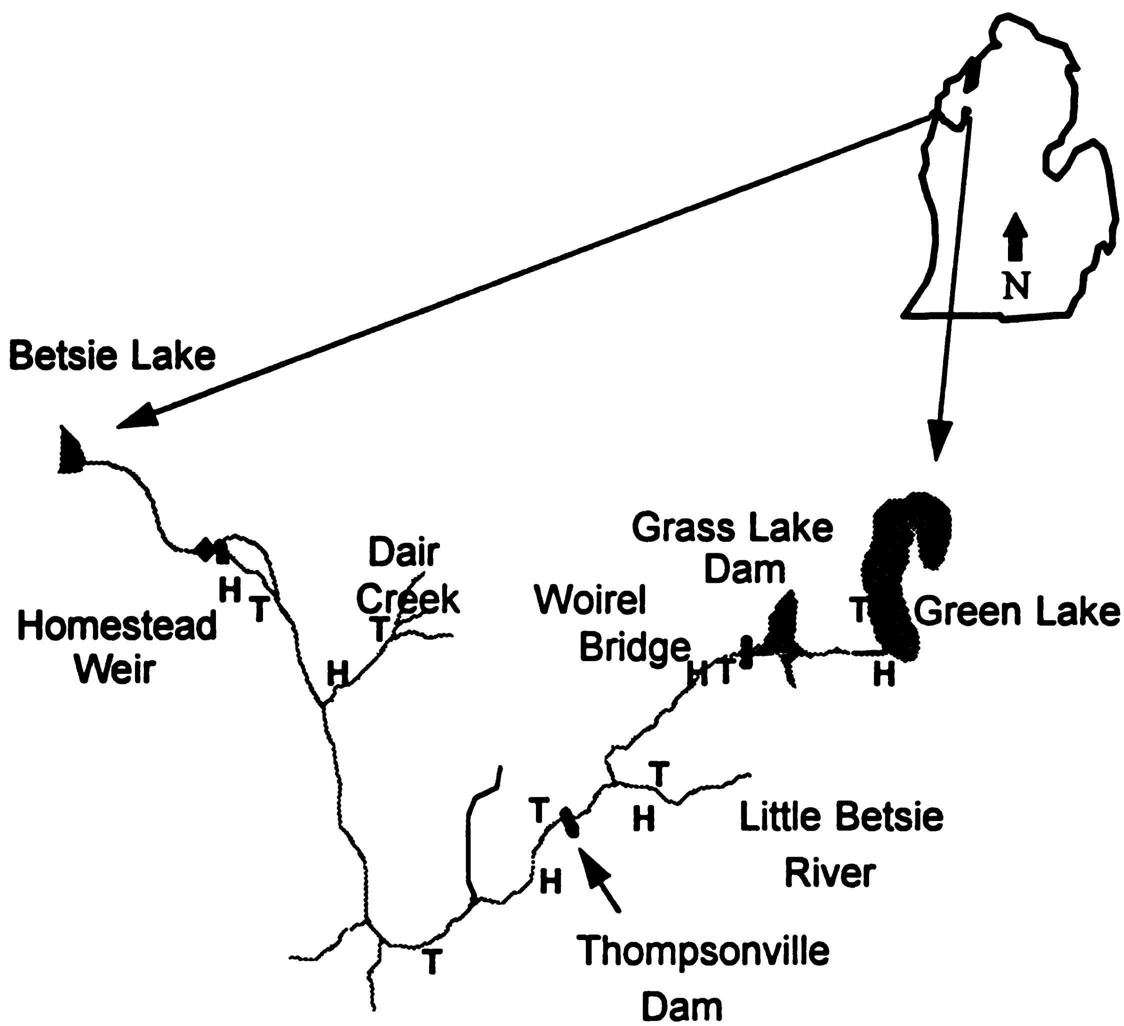

Figure 28. Location of the Betsie River watershed and temperature monitoring locations (T) and flow monitoring locations (H).

structure, was removed in 1974 and replaced with a 2 m high lamprey weir. The original dam contained a crude fish ladder for passage of migratory salmonids but its functionality or success for fish passage is unknown. The lamprey weir currently in place is designed to allow for upstream passage of migratory salmonids but other species can only pass in a downstream direction.

Thompsonville Dam, also a former power generating structure, was approximately 3.1 m high with an impoundment around 5.7 ha (Bullen 1972). From anecdotal reports of local residents, a few migratory salmonids were known to pass over this dam, but it prevented the passage of most migratory fish. Prior to 1972, Thompsonville Dam was identified as one of the sources of heating in the Betsie River and its removal was recommended (Wicklund and Dean 1958, Bullen 1972). However, no action was taken on the recommendation and during a flood in March 1989, the river breached Thompsonville Dam which resulted in its collapse. As a result of the dam's collapse, migratory salmonids were permitted access to the upstream reach of the river. This newly accessible reach contains several high quality riffles and some of the steepest gradients in the watershed. In addition to the main channel, juvenile and spawning steelhead gained access to the Little Betsie River, which enters the Betsie River upstream of the Thompsonville Dam. This is the largest tributary to the Betsie River, contributing approximately 10% of the total flow to the Betsie River at this point in the watershed. Thermal habitat limitations in the main channel still appear to limit the population of steelhead as evidenced by the low number of wild steelhead smolts produced from the watershed (Chapter 1) and the large differences in the spatial distribution and densities of age-0, age-1 and age-2 steelhead parr (Chapter 2).

Current thermal limitations in the Betsie River may, in part, be the result of the remaining low head dam. Grass Lake dam was constructed in 1951 to improve wetland habitat for waterfowl and northern pike (*Esox lucius*). Grass Lake Dam is located approximately 6.7 km downstream of the river source at the outlet of Green Lake. The impoundment area floods approximately 463.6 ha of wetlands (Carbine 1945) which are currently dominated by cattail (*Typha latifolia)*.

Steelhead, naturalized in Michigan since 1876, possess a complex life-history that includes a 2-3 year residency in the river before they emigrate to the receiving ocean or lake (Latta 1974, Biette et al. 1981). In the Great Lakes, steelhead spawn in the late winter and early spring (Biette et al. 1981) in water temperatures ranging from 10-15.5° C (Scott and Crossman 1973). Juvenile steelhead parr fed to satiation experience optimal growth at temperatures between 15 and 17.3° C, and their upper incipient lethal temperature is 25-26°C (Hokanson et al. 1977). In behavior studies, juvenile steelhead preferred temperatures in the range of 11-21° C (Coutant 1977, Cherry et al. 1977). Growth ceases for parr in temperatures greater than 22° C and less than 8° C (Hokanson et al. 1977). Steelhead parr metamorphose into smolts and outmigrate in the spring in water temperatures from 10 to 13° C . At water temperatures greater than 13° C, smolts lose the capacity to smolt, revert back to parr and may remain in the watershed for another year (Zaugg and Wagner 1973).

Interest in improving the Betsie River steelhead fishery by management of the thermal regimes has focused on 2 options: 1) removal of Grass Lake Dam, and 2) changing the source of the initial water supply from the surface of Green Lake, to a hypolimnetic siphon that is diverted around the Grass Lake impoundment (Bullen 1972). Both of these options incur unique biological and financial costs. Removal of Grass Lake Dam would result in a large loss of wetlands, waterfowl habitat, a largemouth bass (*Micropterus salmoides)* and northern pike fishery, and recreational boating opportunities. In addition, the water level of Green Lake would likely decrease with removal of the structure as backwater effects from the control structure can be observed upstream to the Green Lake outlet. The second option would require installation of a water control structure at the Green Lake outlet, where none currently exists, and would involve a large engineering effort to transport water around the Grass Lake flooding. A water diversion would also alter the thermal and flow regime of the river's source above Grass Lake Dam. An evaluation of the effects of these alternatives before implementation is needed in order to predict the potential benefits of such actions.

Methods used to evaluate thermal regimes in streams include two basic approaches which differ in levels of complexity and predictability. The two general categories of stream temperature models are empirical models and physical process models (Barthalow 1989). Empirical models are statistical models that incorporate measured observations and regression or harmonic analysis (Steele 1974, Dyar 1985, Dyar 1997). Empirical models can be used to fill in missing data, determine historical temperatures, and can also be useful for determining thermal suitability of water temperature for target organisms (Stoneman and Jones 1996). Although straightforward, empirical models generally do not consider heat flux or heat transport and are thus limited in their abilities to predict changes in thermal regimes as a result of physical changes in the stream or surrounding landscape.

Physical process models incorporate relationships in the energy budget to predict instream temperatures. By incorporating an energy budget, physical process models predict water temperature on the basis of gains and losses in thermal energy from processes such as radiation, convection, conduction, and evaporation. Additional physical parameters (e.g. stream gradient, discharge, humidity, and shading) are incorporated into physical process models to address the energy flux processes. Because of their predictive abilities in response to a change in the ecosystem, physical process models are very powerful. The trade-off however, is that physical process models are usually much more difficult to develop and require more data than statistical models (Barthalow 1989).

The stream reach model and the stream network model are two physical process models with software established by the U. S. Fish and Wildlife Service (Theurer et al. 1987, Barthalow 1989). Both models are similar in their algorithms and prediction of water temperatures based on an energy budget. The stream segment model is used to consider smaller, uncomplicated reaches of a river over a limited number of time periods. It is a straightforward interactive model that presents a simplified modeling approach and is useful in sensitivity analyses. The stream reach model can be very useful in predicting temperatures over a limited reach for definition of thermal habitat characteristics.

The stream network model is more complicated to initiate than the stream reach model, largely due to the development of a conceptual network model and establishment of the input data files. The network model is very powerful when considering several channel reaches with various input sources over several time periods. The network model is also powerful in that the software provides post-simulation statistical evaluations. Common applications of the stream network model include assessment of the effects of altered thermal regimes as a result of hydropower operations (Theurer et al. 1982, Barthalow 1987, Barthalow 1991, Zedonis 1997).

The objectives of this study were to: 1) describe the current thermal regime throughout the Betsie River Watershed as it compares to the temperature requirements of juvenile steelhead, 2) develop simple empirical statistical models that relate water temperature at sites in the watershed to meteorological influences such as air temperature, 3) develop a watershed physical process temperature model to describe and quantify juvenile steelhead thermal habitat in a normal flow year, and 4) develop a stream-reach physical process model to evaluate management alternatives for changing the thermal regime in the Betsie River.

Methods

I monitored water temperature at 7 locations in the Betsie River watershed (Figure 28, Table 16) on an hourly schedule by use of continuous logging temperature recorders (Stowaways™, Onset Instruments). All recorders were tested with an ASTM thermometer to check for accuracy prior to and after deployment. Daily, weekly, and seasonal (winter: Dec-Feb, spring: Mar-May, summer: Jun-Aug, autumn: Sep-Nov) means were calculated for each station to use in describing the current thermal conditions from July 1993 to July 1996.

Table 16. Sites and reaches used for monitoring flow and temperature in the Betsie River watershed.

Site	Distance upstream of river mouth (rkm)	Model Reach
Green Lake	74.7	Segment A
Grass Lake	68.0	
Woirel Bridge	64.4	Segment B
Little Betsie River	53.8	
Thompsonville	53.1	
M115	46.9	
Dair Creek	24.0	
Homestead	17.7	

I developed empirical univariate regression models of stream temperature at 7 sites in the watershed with the independent variable of air temperature. Information from the Manistee, Frankfort, Cadillac, and Traverse City weather stations provided mean daily air temperature (National Climate Data Center 1993-1996). Mean daily water temperature at each site was the dependent variable in each regression model. I used SAS statistical software (SAS 1988) to obtain an r^2, significance level, y-intercept, and slope for each model.

The Betsie River has no permanent gages for recording river stage or discharge. I established five gaging stations in the watershed and measured discharge at low, intermediate and high flows to establish stage-discharge relationships (Figure 28, Appendix A) (Gordon et al. 1992). Stage levels were checked and manually recorded weekly at each of the gaging sites. The Betsie River substratum is comprised predominantly of sand and therefore, there was a large potential for changes in channel morphology which could have resulted in a change in the stage-discharge relationship. To ensure accurate flow ratings, discharge was measured at the gaging stations each year to monitor for a shift in the stage-discharge relationship, and establish a new rating curve if necessary (Gordon et al. 1982).

I developed a stream network model of the Betsie River that also incorporated the flows and thermal regime of the Little Betsie River with software developed by the U.S. Fish and Wildlife Service (Theurer et al. 1984). Because the model is fairly complex, I will briefly explain its required components and logic for a general understanding of the model. The stream network model incorporates 6 features: 1) a heat transport model to predict water temperature as a function of stream distance, 2) a heat flux model to predict the energy balance between the water and surrounding environmental conditions, 3) a solar model for the prediction of solar radiation penetrating the water as a function of latitude, time of year, and meteorological conditions, 4) a shade model that predicts the effects on solar input from shade resulting from topographic and vegetative conditions, 5) meteorological corrections as a function of changing elevation within the watershed, and 6) regression aids for filling in missing water temperatures (Theurer et al. 1987). Components 5 and 6 were not necessary in this application of the model.

Information processed by each of the model components is supplied by independent data files. Seven input data files are required for the analysis: stream geometry data, time period information, meteorology information, study site information, hydrology site information, hydrology data, and shade data. Each file has a number of specified parameters that must be supplied (Table 16).

I defined the watershed by a number of "nodes" recognized by the network model software as location descriptors and site specific available information for processing information. The model of the Betsie River began with a headwater node at the Green Lake outlet and concluded with an end node at Homestead Weir (Figure 29). The Little Betsie River was the only tributary node included in the network model because it was the only tributary with a contribution of flow of 10% or more of the main channel (Barthalow 1989). The network structure for the model also incorporated 3 validation nodes of measured water temperature for model calibration and verification (Figure 29). In addition to the nodes illustrated on Figure 29, 28 output (O) nodes were included every 2 km from Green Lake to Homestead Weir to predict instream temperatures along the length of the study reach. The stream network software processed the input files through a series of programs to execute the model and estimate the output values (Barthalow 1989).

To address assumptions inherent in the network model, travel time for water from the start to the end of the study reach must be considered as the time step for the model. Based on the low gradient of the stream channel, the reach distance from Green Lake to Homestead Weir, and a low mean water velocity, a time step of 1 week was used to evaluate instream temperatures. The model was supplied with mean weekly water temperature and flow information for a normal flow water year, 1994. Groundwater contributions in Michigan streams are known to have a significant effect in moderating the thermal regime (Hendrickson and Doonan 1972). In this model, groundwater contributions are added by accumulating lateral flow in the channel between two defined hydrologic nodes.

After the initial data entry and first successful run of the model, I conducted a sensitivity analysis to evaluate the sensitivity of the model output to changes in several parameters for the Betsie River network. Parameters evaluated in the sensitivity analysis included air temperature, humidity, windspeed, solar radiation, percent

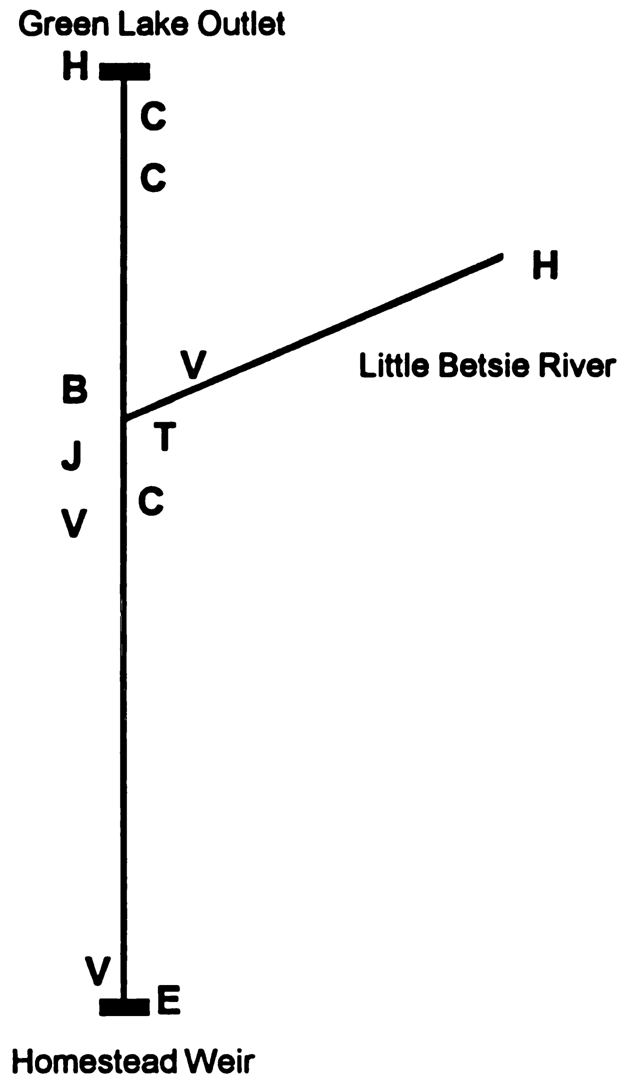

Figure 29. Skeleton framework and nodes for the Besie River Watershed network temperature model. Nodes H = headwater, C=change in shading or stream width, T= terminus of tributary, J = junction below tributary, B= branch above tributary, V= validation node, E = end of study reach.

Table 17. Input files and associated parameters required for the stream network model.

Input File Name	Parameters	Sources
Stream Geometry	Site Latitude & Elevation	U.S.G.S. Topographic Map
	Manning's N	Stream Observations
	Stream Width Coefficient	Derived[1]
	Stream Width Exponent	Derived[1]
	Stream Shading Minimum	Observed & Measured
	Stream Shading Maximum	Observed & Measured
	Ground Temperature	U.S.G.S.
	Streambed Thermal Gradient	Program Default
Time Period	Time Period Name	Defined as 1 week
	Julian Dates for Time Periods	
	Dust Coefficient	Calculated
	Ground Reflectivity	Barthalow (1987)
Meteorology	Average Annual Air Temperature	N.C.D.C.-Frankfort Station
For all time periods:		
	Air Temperature	N.C.D.C.-Frankfort Station
	Wind Speed	N.C.D.C.-Frankfort Station
	Humidity	Calculated
	Sunshine Ratio	Estimated
	Solar Radiation	SSSolar Output[2]
Study File	Node Configuration for the Network (Table 17)	
Hydrology Data File	Flow Data	Measured
	Water Temperature	Measured
	Lateral Inflow Temperature	Mean Annual Air Temp.
Hydrology Node File	Hydrology Node Configuration	
Shade File	Stream Reach Azimuth	U.S.G.S. Topographic Map
	Stream Width	Measured
	East and West Bank Topographic Altitude	Measured
	East and West Bank Vegetation:	
	Height	Measured
	Offset	Measured
	Density	Estimated from Observations

sunshine, and discharge. I used results from the sensitivity analysis when considering potential parameters to adjust in calibrating the model. After the model was fully calibrated, mean weekly temperatures from the output nodes were evaluated at the to characterize the thermal habitat for juvenile steelhead in the Betsie River.

To evaluate thermal habitat management alternatives such as dam removal and a siphon draw, I modeled two stream segments (A and B) with stream-reach physical process model software also developed by the U.S. Fish and Wildlife Service (Theurer et al. 1984, Barthalow 1989). Segment A extended 6.7 km from the Green Lake outlet to Grass Lake Dam and Segment B included 14.9 km from Grass Lake Dam to Thompsonville Dam (Table 16). Similar to the network model, the stream-reach model required 3 types of information including stream geometry (upstream and downstream elevations, stream width, Manning's N, and shading), meteorology (solar radiation, air temperature, relative humidity, cloud cover, daylength, and wind speed), and hydrology (stream discharge in and out of the reach and temperature of the water entering the reach). Wind speed and solar radiation were obtained from the Traverse City National Weather Service Station and cloud cover was estimated from measured solar radiation. Relative humidity was calculated from measures of the dew point according to:

$$R_h = [(112 - 0.1{*}T_A + T_{dp}) / (112 + 0.9{*}T_A)]^8$$

where R_h = relative humidity, T_A = air temperature, T_{dp} = dew point temperature. Shading was defined according to Barthalow (1989) as the amount of the stream channel covered by shade in mid-day. Based on measurements of the channel and stream bank vegetation, I estimated shading to be 0% for segment A and 5% for segment B. Because of the short travel distance, a daily time step was appropriate for this application in which I used mean daily temperature and flow information.

I calibrated 4 models to evaluate the two stream reaches for typical and low flow water years. Calibrated models were developed for Segment A and Segment B for a typical- and low-flow year, 1994 and 1995 respectively. Humidity was the primary calibration parameter and it is known to vary by as much as 20% away from the measurement site (Barthalow 1989). Solar radiation was altered from the measured values in Traverse City within reasonable limits (± 10%) to also assist in calibration. Results from the stream reach models included minimum, mean, and maximum outflow temperatures, width, depth, and slope of the channel with varying flows, and heat flux components which indicate atmospheric and ambient conditions that contribute to heat gains and losses. After calibrating the model for the typical- and low-flow measured conditions, segment A input parameters were altered by increasing the gradient, decreasing channel width, and decreasing Manning's N values, to simulate the channel morphology that would exist with the removal of Grass Lake Dam. Initially, high Manning's N values were used because of the large amount of vegetation present in the existing channel. With the dam removed, the channel would return to a silt, sand, and gravel substratum which would indicate a lower Manning's N value than vegetation. Outflow water temperatures predicted from the simulation of Segment A with the dam removed were then used as starting temperatures at the upstream reach of segment B to evaluate the effects of dam removal on this downstream reach.

Evaluation of the hypolimnetic siphon alternative required changing the water temperature at the inflow of segment B to reflect a proposal to withdraw hypolimnetic water from Green Lake, divert it around the Grass Lake flooding and discharge it into the Betsie River immediately below Grass Lake Dam (Bullen 1972). I assumed that the water diversion occurred in an enclosed pipe rather than a channel and that ambient conditions had no effect on the temperature of the diverted water. I simulated the siphon scenario using a water temperature of 7.2°C as the starting temperature for segment B and a constant mean summer monthly flow for a normal- (1994) and low - flow (1995) year, reflecting the regulated flow as a result of water control structures.

Results

Current Thermal Regimes Measured in the Betsie River Watershed

Water temperature varied spatially and temporally throughout the watershed. The upper Betsie River at Green Lake and Woirel Bridge (sites 1 and 2) experienced the warmest water temperatures in summer while the tributaries (sites 6 and 7) had the coldest water temperatures in summer (Table 18). Mean winter water temperature was warmer in the upper river and in the tributaries (sites 1,2, 6, and 7) in relation to main channel locations Thompsonville, M115, and Homestead (sites 3, 4, and 5).

Table 18. Mean water temperature by season and year measured at sites in the Betsie River and its tributaries.

	Fall			Winter			Spring			Summer			
Site #	93	94	95	93	94	95	94	95	96	93	94	95	96
1	11.8	14.9	11.9	1.6	1.8	1.1	5.4	6.1	6.3	23.2	21.4	23.4	21.0
2	9.7	12.9	10.7	1.2	0.9	0.4	7.9	9.0	9.0	22.5	21.6	23.4	21.2
3	9.3	11.7	9.3	0.7	1.0	0.2	8.3	7.8	6.7	20.5	19.2	20.9	19.1
4	9.1	11.3	9.3	0.9	1.7	0.8	8.1	7.8	7.5	19.0	18.1	19.8	18.1
5	8.7	11.0	8.7	0.9	1.7	0.4	8.0	8.1	7.0	18.3	17.6	19.1	17.6
6	7.5	9.3	7.6	1.4	2.0	0.8	6.9	7.2	5.9	14.7	14.0	16.1	14.1
7	8.1	10.0	8.2	1.0	1.8	0.7	7.3	7.4	6.6	15.9	15.9	17.4	15.7

Sites: 1= Green Lake, 2 = Woirel Bridge, 3 = Thompsonville, 4 = M115, 5 = Homestead, 6 = Dair Creek, 7 = Little Betsie River.

Cumulative probability plots of temperature (exceedence curves) were developed with mean daily temperature data from water years 1993-1995 to illustrate temporal patterns in which the water temperature was likely to exceed the preferred temperature range for juvenile steelhead (Figure 30, Table 19). In the upper watershed, (Green Lake and Woirel Bridge), the preferred temperature range and optimal growth temperatures were exceeded 20 and 30% of the time, respectively. At Thompsonville and Homestead, the preferred temperature range was exceeded between 1 and 7% of the time and optimal growth limits were exceeded 12-20% of the time. In the tributaries, Dair Creek and Little Betsie River, the preferred temperature limits were never exceeded and the optimal growth limits were exceeded only 3% of the time in Dair Creek and 7% in the Little Betsie River. Generally, all six sites were similar in their probabilities to be less than the lower preferred temperature (11°C) ranging

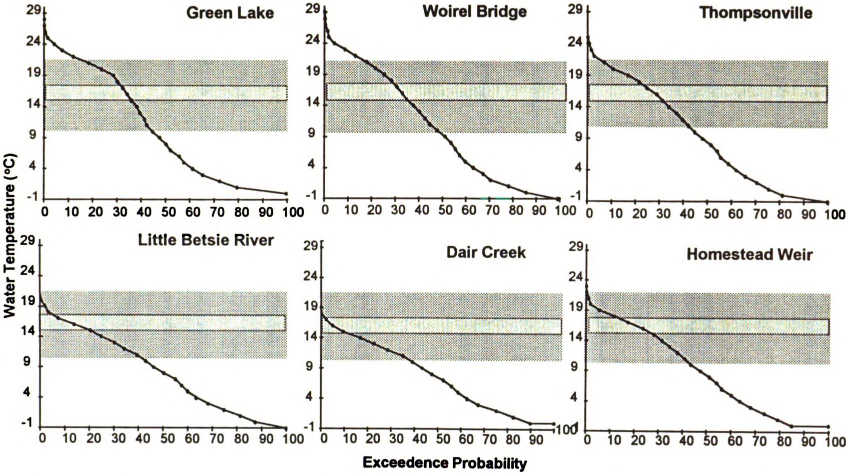

Figure 30. Temperature exceedence curves for sites in the Betsie River watershed, water years 1993-1995. The large rectangle indicates temperatures preferred by juvenile steelhead and the smaller rectangle indicates the temperature range of optimum growth of juvenile steelhead (Hokanson et al. 1977, Wismer and Christie 1987).

from 52% at Woirel Bridge to 61% in Dair Creek. The largest difference between sites was the duration of time (0 - 18.6%) when the preferred thermal range for juvenile steelhead was exceeded (Table 19).

Table 19. Probability of exceeding, meeting, or remaining below preferred and optimal growth temperatures for juvenile steelhead at sites throughout the Betsie River watershed. Calculated from mean daily temperatures recorded in water years 1993-1996.

	SITE					
	Green Lake	Woirel Bridge	T-ville	Homestead	Little Betsie	Dair Creek
Percent of time						
likely within optimum growth range	3.6	4.2	6.8	10.6	13.1	7.3
likely within preferred temperature range	23.7	26.7	35.0	39.7	39.5	35.1
likely to exceed the preferred range	18.6	17.9	10.6	1.1	0.0	0.0
likely to be less than the preferred range	55.3	52.3	55.3	56.3	57.2	60.8

Temperature and photoperiod interact to stimulate smoltification and emigration of juvenile steelhead. Photoperiod initiates the physiological changes, while temperature directs the rate and duration of smoltification (Wedemeyer 1980). Water temperature between 7° and 13°C is optimal for smoltification. Above this, smolts may revert back to parr and spend another year in the stream environment (Zaugg and Wagner 1973). In the Betsie River in most years, the lower smolting temperature was reached in mid-April and the upper threshold was exceeded by mid-May (Figure 31).

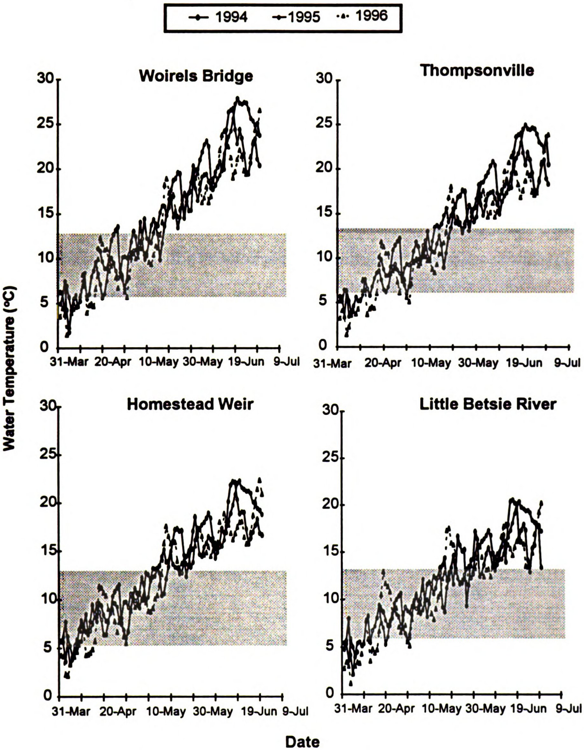

Figure 31. Spring water temperatures during steelhead smolt emigration in the Betsie River watershed, 1994-1996. Shaded area denotes smoltification temperatures, above which smolts may revert to parr.

In the Little Betsie River, the lower smolting temperature threshold occurred in mid-April and suitable temperatures for smolting extend into mid-June (Figure 31).

Empirical Models of Water Temperature

Air temperature data from neighboring climatological stations were used to predict daily mean water temperature at Betsie River stations. I initially tested data from stations at Cadillac, Frankfort, Manistee, and Traverse City (NCDC 1993-1996). Data from all climatological stations produced significant relationships ($p < 0.001$) with an adjusted $r^2 > 0.80$. The Frankfort station accounted for the greatest variability in water temperature with r^2 values ranging from 0.826 to 0.908 at sites throughout the watershed (Table 20). Water temperature at tributaries and downstream sites were more strongly related to air temperature than at the upstream sites, Woirel Bridge and the Green Lake outlet.

Table 20. Results of regression modeling water temperature as a function of air temperature measured at Frankfort, Michigan. All results are significant, $P < 0.001$.

Site	R^2	Coefficient (Slope)	Y-intercept
Green Lake	0.826	0.756	4.964
Woirel Bridge	0.866	0.781	4.931
Thompsonville	0.897	0.717	4.261
M115	0.906	0.656	4.549
Homestead	0.910	0.634	4.417
Little Betsie River	0.903	0.571	4.164
Dair Creek	0.908	0.511	4.015

Predicted temperatures were plotted with observed temperatures to evaluate the empirical models (Figures 32 and 33). In all cases, the empirical models were poor predictors at very low air temperatures, from -10 to -5°C, likely due to the effects of ice

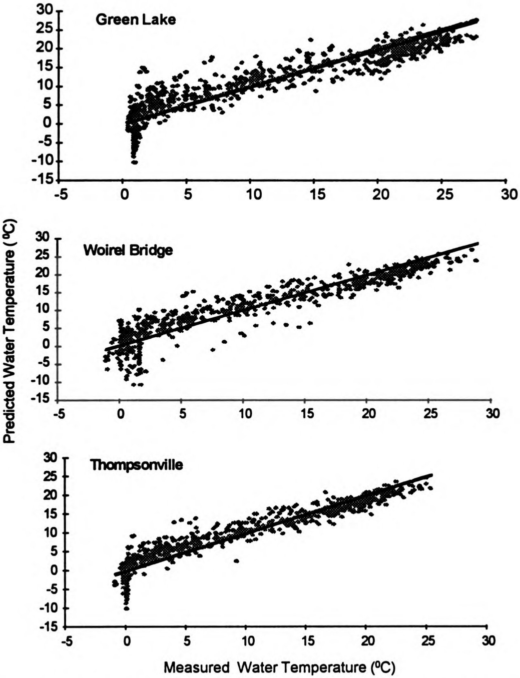

Figure 32. Predicted water temperature (symbols) as a function of air temperature and mean daily water temperature (line) at sites in the Betsie River, 1993-1996.

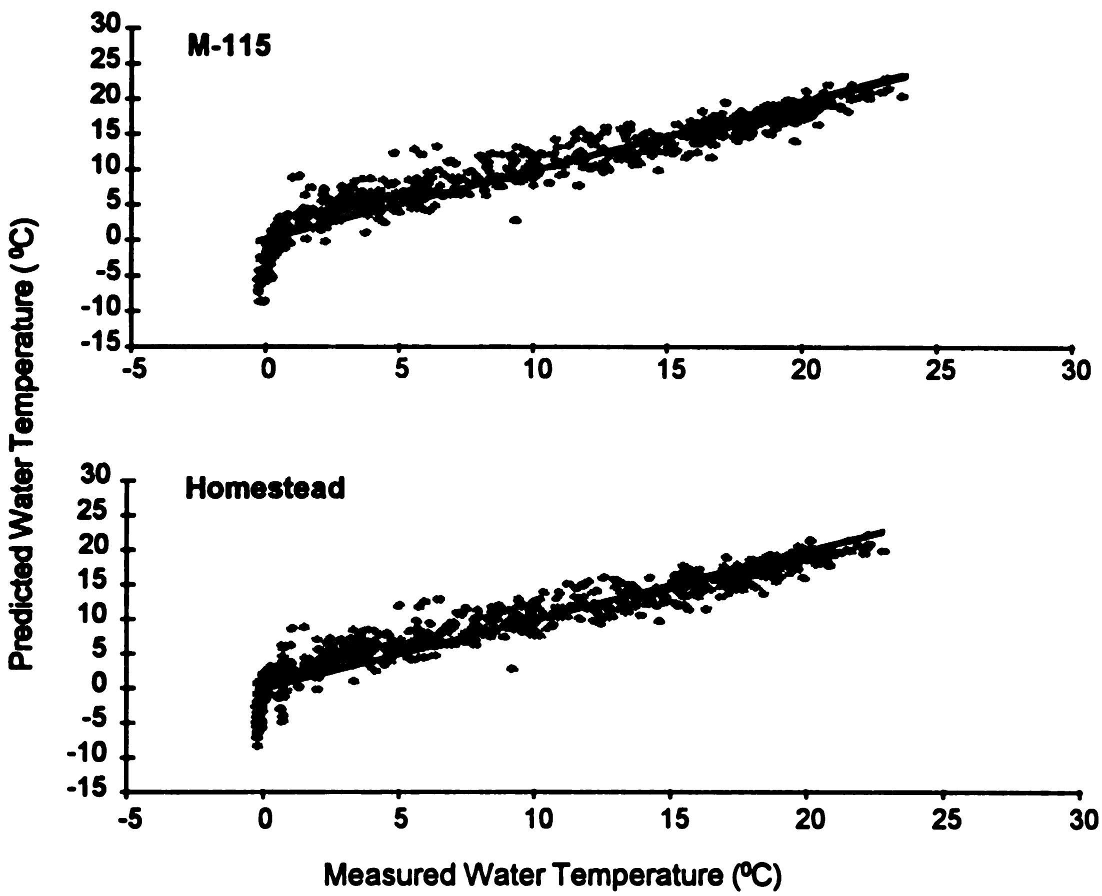

Figure 32 (continued). Predicted water temperature (symbols) as a function of air temperature and mean daily water temperature (line) at sites in the Betsie River, 1993-1996.

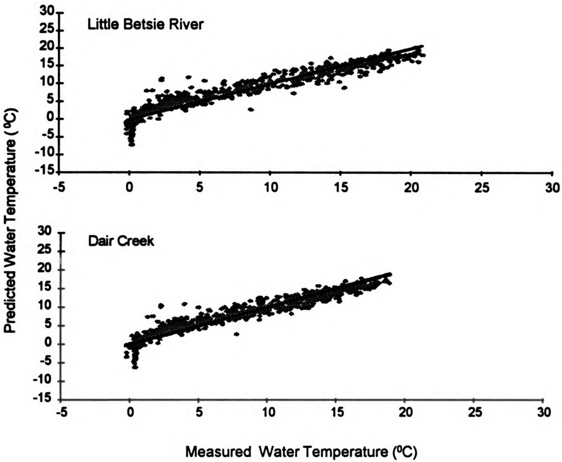

Figure 33. Predicted water temperature (symbols) as a function of air temperature and mean daily water temperature (line) at tributary sites in the Betsie River watershed, 1993-1996.

and snow cover. The largest error from the observed values was observed at the Green Lake monitoring location (most likely due to the heat capacity of the lake) and became progressively smaller in downstream locations, M115 and Homestead Weir. With a few exceptions, air temperature was a very good predictor of water temperature in the tributaries, Dair Creek and Little Betsie River at air temperatures above -7.0°C (Figure 33).

Stream Network Temperature Model Results

All data input files were assembled and entered for the stream network file (Appendix C). Results of the sensitivity analysis indicated that changes to air temperature and humidity produced the greatest changes in the predicted temperatures (Table 21). Discharge was also sensitive although in all cases, the resulting mean error was less than 1 degree. The model was relatively insensitive to other variables, such as percent sunshine, solar radiation, and windspeed and changes in these parameters resulted in little change in the temperatures predicted.

Table 20. Mean error at 3 validation nodes in the Betsie River and Little Betsie River resulting from 5% increases and decreases a single parameter.

	Site					
	Little Betsie Hwy 669		Betsie River Thompsonville		Betsie River Fred's Landing	
	+ 5%	-5%	+5%	-5%	+5%	-5%
Parameter						
Air Temperature	-0.64	0.64	-0.46	0.46	-0.68	0.67
Humidity	-0.27	0.27	-0.24	0.24	-0.34	0.34
Windspeed	-0.04	0.04	0.02	-0.03	0.02	-0.02
Solar Radiation	0.01	0.01	-0.12	0.12	-0.12	0.12
% Sunshine	0.00	0.00	0.01	0.01	0.01	-0.02
Discharge	0.16	0.64	0.48	-0.08	0.41	0.39

I evaluated model performance on the basis of 3 error values for each validation node. Mean error values were the averages of the differences between observed and predicted water temperatures at each validation node. Median error represented the value in which at least 50% of the predicted water temperatures were within this range. Maximum error was the greatest deviation of predicted temperature from the observed water temperature. I determined my decision criteria for calibration as:

1) no more than 10% of the predicted temperatures greater than 1°C from observed temperatures,

2) no single predicted temperature greater than 1.5°C from observed temperatures,

3) the mean error for the validation node should be less than 0.5°C, and

4) no spatial or temporal trends in error (Barthalow 1987).

Although the results for the initial run of the network model were fairly good (Table 22), for several weeks, predicted temperatures ranged above the values defined by my decision criteria. For weeks and locations where the mean weekly values were beyond the defined calibration decision criteria, I altered selected input parameters to calibrate the model. Humidity and solar radiation were altered in most cases for calibration and discharge was altered within ± 20% for the Little Betsie River in which gage measurements were the least certain, but water temperature was known. The resulting calibrated model predicted stream temperatures within the 4 standards outlined in the decision criteria (Figure 34).

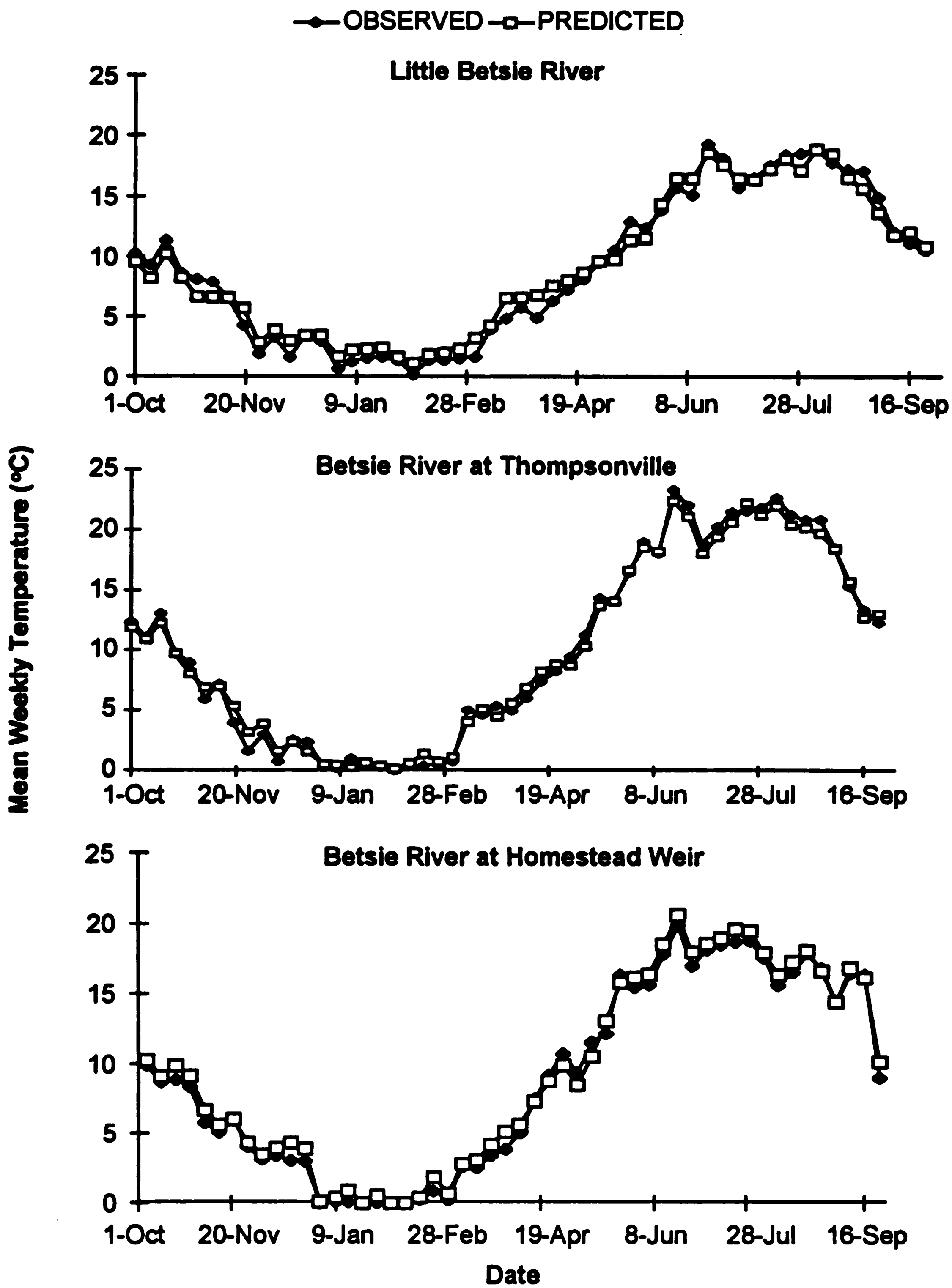

Figure 34. Comparison of mean weekly observed water temperature and network model predicted temperature at validation sites in the Betsie River and Little Betsie River.

Table 21. Mean, probable, and maximum error for the stream network model predictions at 3 validation sites in the Betsie River watershed.

	Site			
	Little Betsie Hwy 669	Betsie River Thompsonville	Betsie River Fred's Landing	Entire Stream Network
Initial Model Run				
Mean Error	-0.51	0.36	1.68	0.51
Probable Error	2.53	2.01	1.38	2.10
Maximum Error	-8.62	6.83	-5.89	-8.62
Final Calibration Run				
Mean Error	0.16	-0.07	0.40	0.16
Probable Error	0.64	0.48	0.35	0.52
Maximum Error	1.89	1.58	1.31	1.89

Once calibrated, mean weekly temperature results were predicted from the stream network model for every 2 km in the Betsie River from Green Lake to Homestead Weir. Spatial and temporal patterns emerged in the annual thermal regime of the Betsie River. At the beginning of the water year (October) from Green Lake downstream to river kilometer (rkm) 55, the river was warmer and in the preferred thermal range for juvenile steelhead (Figure 35). By November, all sites in the watershed were below the preferred range and these conditions persisted until April (Figures 35-37). Rapid cooling occurred in December and by January and through February, mean weekly temperatures hovered near 1-2°C throughout the length of the river (Figure 36). Warming patterns emerged in early spring (Figure 37). In March, the water quickly warmed from 1°C to 5°C downstream of rkm 50, staying slightly cooler in the reach near Green Lake. A similar pattern was observed for April as water

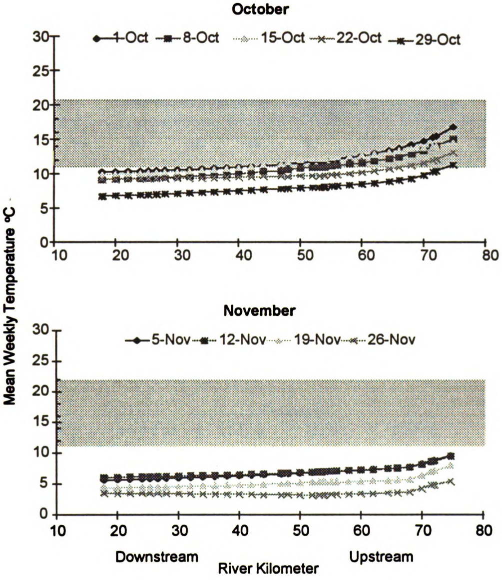

Figure 35. Longitudinal distribution of mean weekly water temperature as predicted for the Betsie River during the fall and the start of the water year, October - November in a normal flow year. The shaded box is the preferred temperature range for juvenile steelhead.

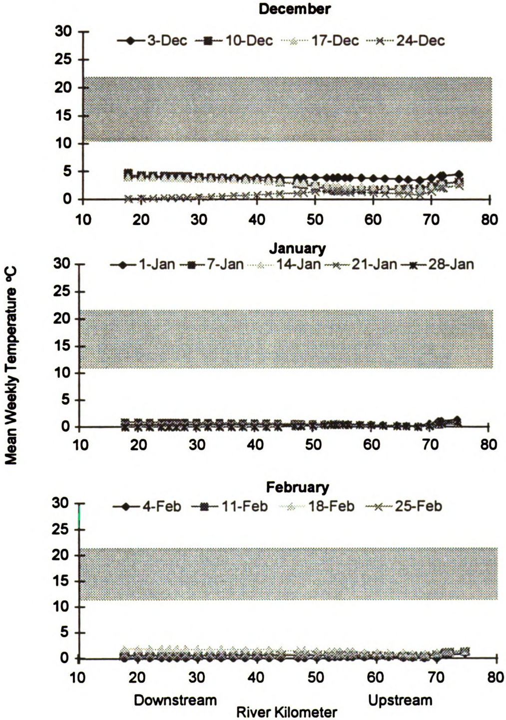

Figure 36. Longitudinal distribution of mean weekly water temperature as predicted for the Betsie River during winter, December - February, in a normal-flow year. The shaded rectangle is the preferred temperature range for juvenile steelhead.

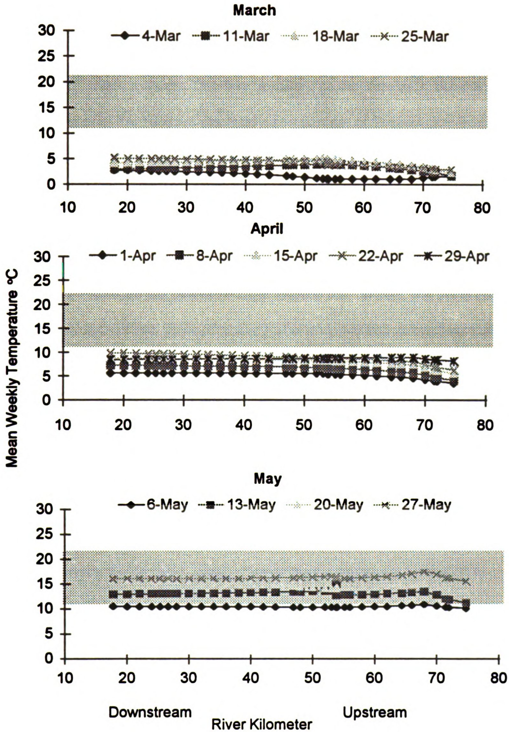

Figure 37. Longitudinal distribution of mean weekly water temperature as predicted for the Betsie River during spring, March - May, in a normal flow year. The shaded box is the preferred temperature range for juvenile steelhead.

temperatures warmed to 10°C below rkm 50, and while remained 2-4°C colder in the river upstream of rkm 50. Water temperatures increased quickly between weeks in May and water temperatures entered the preferred range for juvenile steelhead, gaining 3-4°C within a week. The river remained cooler upstream, but the area decreased to only the 10 km immediately downstream of Green Lake. In May, the entire length of the river was within the preferred range for juvenile steelhead (Figure 37).

Summer temperatures throughout the river sometimes exceeded the preferred range for up to 47% of the stream length. By late June, upstream of rkm 62, the water temperature exceeded the preferred limits (Figure 38). In July, the trend continued with the river upstream of rkm 50 beyond the preferred limits and downstream of rkm 50 within the preferred range. A cooling pattern emerged in August, however, upstream of rkm 50, when the water temperature reached and exceeded the upper preferred temperature limit (Figure 38). In September, nearly the entire length of the Betsie River was within the preferred temperature range (Figure 39).

Stream Segment Water Temperature Model Results

Based on the limited 4 years of water temperature and hydrological observations, 1994 represented the typical flow and climatic conditions in the Betsie River watershed and 1995 represented a low-flow year. I used data for these two summers to develop a stream reach temperature model to evaluate management alternatives. The model for segment A (Green Lake to Grass Lake) was calibrated with the water temperature monitoring station 2.4 km downstream of Grass Lake Dam. I assumed that only minor differences in water temperature and discharge existed

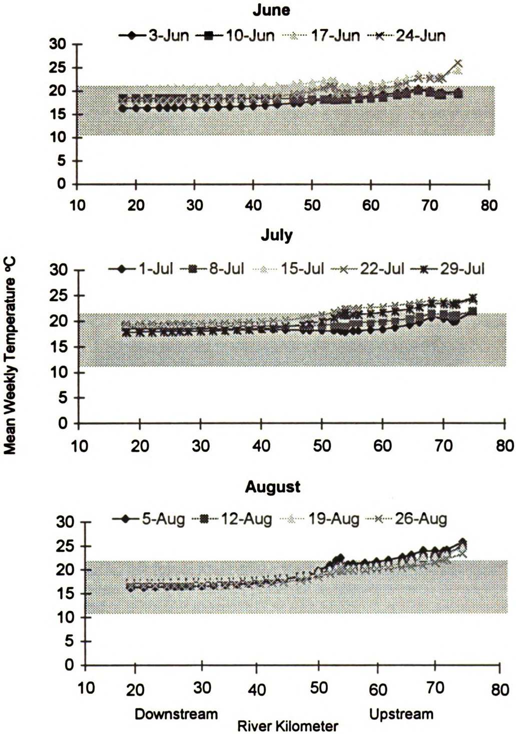

Figure 38. Longitudinal distribution of mean weekly water temperature as predicted for the Betsie River during summer, June - August, in a normal-flow year. The shaded rectangle indicates the preferred temperature range for juvenile steelhead.

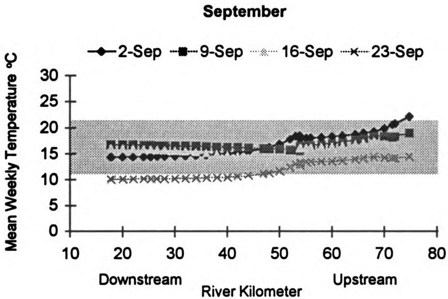

Figure 39. Longitudinal distribution of mean weekly water temperature as predicted for the Betsie River during September in a normal-flow year. The shaded rectangle indicates preferred temperature range for juvenile steelhead.

between the Grass Lake Dam outlet and Woirel Bridge, because discharge measurements did not differ between the sites.

The calibrated temperature model for segment A had an average error of 1.57°C and 1.52°C for 1994 and 1995 and most errors were less than 2°C different (Figure 40). Predicted temperatures for Segment B had mean differences of 0.58°C and 0.34°C for 1994 and 1995 (Figure 41). In all cases, the errors were randomly distributed and the models did not consistently over- or under-predict the measured values.

In the first simulation, I modeled the effects on water temperature as a result of removing of Grass Lake Dam by reducing the channel width and increasing the gradient, both of which would result with the removal of the structure. Mean daily water temperature declined in this scenario (Figure 42, Table 21). In segment A, during the typical-flow year, daily mean water temperature ranged from 0.88 to 6.51°C (mean = 3.72°C) less than under current conditions. In the low-flow year, daily mean water temperatures averaged 3.94°C less and ranged from 0.88 to 7.37°C less than under current conditions. For the Grass Lake to Thompsonville reach, water temperature averaged 1.22°C less in the typical year and 0.55°C less in the low-flow year with the dam removed (Figure 43).

A siphon from the hypolimnion of Green Lake to the outlet of Grass Lake has been suggested as a remedy to the thermal limitations in the upper Betsie River (Bullen 1972). Logistics in this plan included routing the water around the Grass Lake impoundment and directing it through either a pipe or open culvert into the Betsie River channel below the Grass Lake Dam. I simulated this alternative with the reach model using the climatic conditions observed for the typical and low-flow year. Flows were

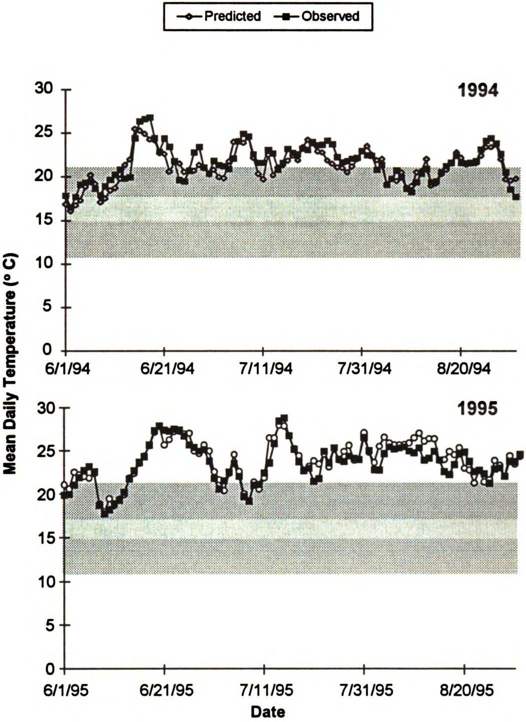

Figure 40. Measured and predicted mean daily summer temperatures from the stream reach temperature model for segment A, Green Lake to Grass Lake, in the Betsie River during a typical-flow (1994) and low-flow water year (1995). The large shaded area indicates water temperatures preferred by juvenile steelhead and the inner rectangle identifies optimum growth temperatures for juvenile steelhead (Hokanson et al 1977, Wismer and Christie 1987).

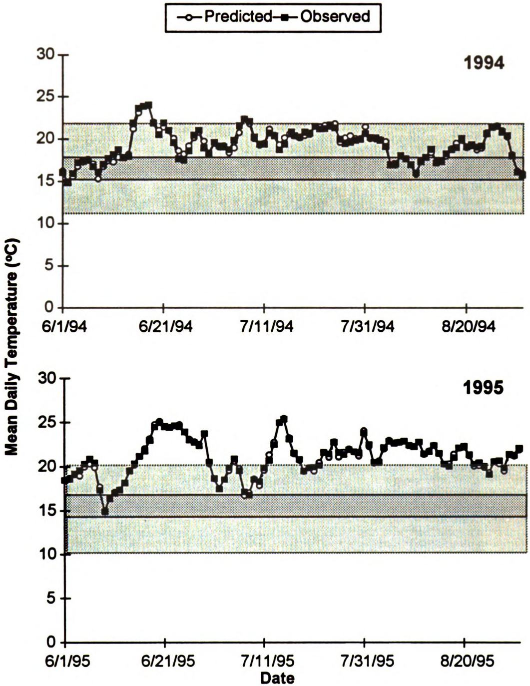

Figure 41. Measured and predicted mean daily summer temperatures from the stream reach temperature model for segment B, Grass Lake to Thompsonville, in the Betsie River during a typical-flow (1994) and low-flow water year (1995). The large shaded area denotes temperatures preferred by juvenile steelhead and the smaller inner rectangle indicates the optimum growth temperatures for juvenile steelhead (Hokanson et al 1977, Wismer and Christie 1987).

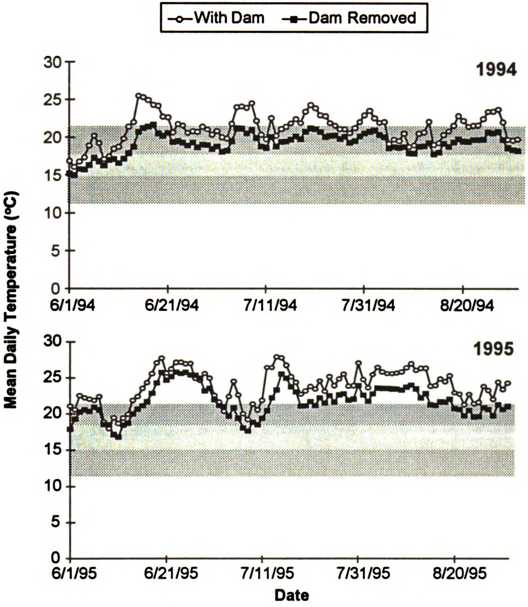

Figure 42. Comparison of mean daily summer water temperatures under current conditions with Grass Lake Dam and simulated conditions of dam removal in a typical- flow (1994) and low-flow (1995) water year for Segment A. The shaded area indicates the temperature range preferred by juvenile steelhead and smaller rectangle depicts optimal growth temperatures (Hokanson et al. 1977, Wismer and Christie 1987).

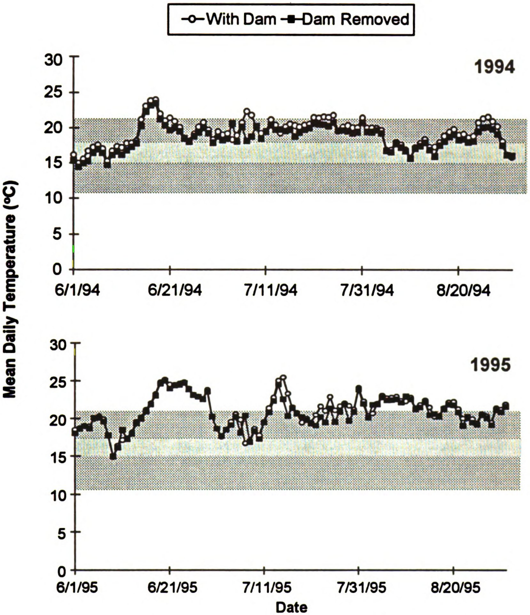

Figure 43. Comparison of mean daily summer water temperatures from Grass Lake Dam to Thompsonville under current conditions with Grass Lake Dam present and simulated conditions with the dam removed for a typical-flow (1994) and low-flow (1995) year. The large shaded area indicates the preferred temperature range for juvenile steelhead and small inner rectangle depicts optimum growth limits (Hokanson et al. 1977, Wismer and Christie 1987).

averaged from measurements in 1993-1996 to obtain mean monthly flows of 57.6, 59.1, and 63.9 for June, July, and August, and I designated 7.2°C as the outlet temperature. This was the measured temperature during the summer months for the hypolimnion of Green Lake (Newcomb, unpublished data). This scenario resulted in mean daily water temperatures less than those simulated with the dam removal (Table 23, Figure 44).

Table 22. Mean daily water temperature in June, July, and August for the river at the downstream section of Segments A and B, in the Betsie River under current conditions and under simulated conditions of removing Grass Lake Dam and releases of hypolimnetic water from Green Lake at Grass Lake Dam. Data from sites at M-115, Homestead, and the Little Betsie River are also included for comparison.

Management Alternative	Typical-Flow Year (1994) °C			Low-Flow Year (1995) °C		
	June	July	August	June	July	August
Current Conditions						
Segment A	21.2	22.7	21.0	24.0	24.2	25.2
Segment B	19.0	20.3	18.7	21.0	20.7	21.5
M-115	17.9	19.0	17.4	19.6	19.5	20.2
Homestead	17.4	18.4	16.7	18.8	18.8	19.4
Little Betsie River	15.9	16.7	15.0	17.1	17.4	17.9
Dam Removal						
Segment A	18.4	19.9	19.3	21.8	21.5	21.9
Segment B	18.3	19.6	18.1	20.9	20.1	21.2
Hypolimnetic Discharge						
Segment B	15.2	15.6	13.7	18.9	18.3	19.0

During both the typical-flow year and the low-flow year, the hypolimnetic release resulted in water temperatures ranging from 12 to 21°C with a mean of 18.2°C at Thompsonville. When compared with the dam removal alternative, the cold hypolimnion contribution provided mean water temperatures 2.7-3.6°C cooler than

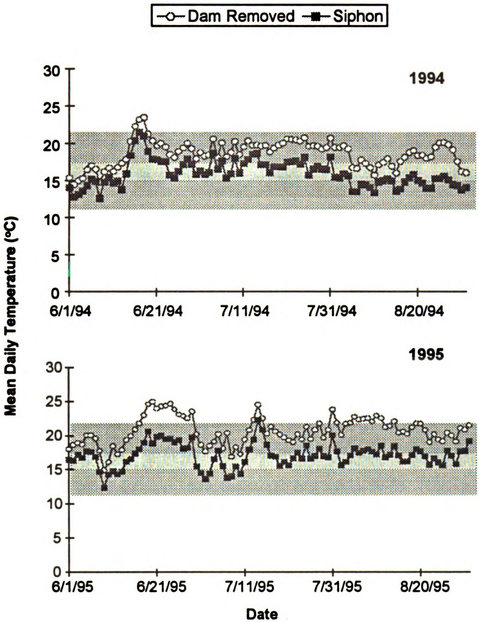

Figure 44. Comparison of mean daily water temperatures simulated for the dam removal and the addition of cold hypolimnion water in the Grass Lake to Thompsonville reach of the Betsie River in a typical-flow (1994) and low-flow (1995) water year. The large shaded area is the water temperature range preferred by juvenile steelhead and smaller rectangle depicts optimum growth ranges (Hokanson et al. 1977, Wismer and Christie 1987).

temperatures simulated without Grass Lake Dam in Segment B (Figure 44). In warm, low-flow years such as 1995, instream temperatures in the Thompsonville reach still exceed the optimum growth temperatures for juvenile steelhead (Table 23).

Discussion and Conclusions

Both empirical models and physical process models are useful tools in assessing instream temperature. Empirical models are most valuable for predicting water temperature based on one or a few other variables. However, empirical models can be limited in predicting water temperature as a response to morphological changes within the channel or other variables, such as riparian shading, which would alter the heat dynamics within the river. Physical process models are necessary to model the changes in thermal regime that result from morphological or riparian changes.

Both the stream network model and the stream reach model were sensitive to changes in air temperature. Several researchers have documented the dominating effect of air temperature on stream water temperature (Smith and Lavis 1975, Crisp and Howson 1982, Barthalow 1989, Stoneman and Jones 1996). The regression models provided in this manuscript can be used to predict water temperature in future modeling applications in the Betsie River under the current instream conditions. If there were significant changes in channel morphology or if Grass Lake Dam were removed, new empirical models would need to be derived.

Multivariate empirical models that incorporate an additional aspect, such as discharge, may predict water temperature more accurately than single univariate models. The empirical models for water temperature in the Betsie River watershed were simple univariate models (air temperature = independent variable, water temperature = dependent variable), resulting in variable but high correlation values throughout the watershed (0.826-0.910). Improvement to these models might occur if an additional variable, such as a lag time value were incorporated to account for the ongoing heat gains and losses in preceding days.

Both physical process models, the stream network model and the stream reach model were useful for examining the thermal regime in detail in the Betsie River and evaluating consequences of changes in the watershed. The stream network model results provided for a detailed evaluation of thermal habitat availability based on temperature data collected at 7 stations.

Juvenile steelhead residing in marginal environments can face a number of consequences. In those environments that exceed the optimal growth limits defined by temperature, fish can experience a decline in growth rates or exhibit zero growth (Hokanson et al. 1977). Alternatively, fish will move out of an area that is too warm and into refuge areas which may already be occupied by resident fish (Bilby 1984), suggesting a temporal density dependent limitation and competition for space. Finally, when temperatures are near or at the lethal levels, mortality can occur either as a direct result of the heat, or from a secondary cause, such as infection or starvation (Coutant 1976).

On the other hand, juvenile steelhead residing in locations where the thermal regime remains below the optimal growth limits may experience slower growth and later age of smoltification. Although an additional year of residency in the stream may subject these fish to mortality in the stream environment, they are less likely to succumb to heat associated mortality or stress.

Under the current channel and flow conditions, the most thermally suitable area for juvenile steelhead in the summer is between Thompsonville and Homestead Weir in the main channel of the Betsie River and in the tributaries. In this reach, groundwater contributions, as evidenced by increases in downstream discharge, provide a cooling effect to moderate summer temperatures. Locations above this reach, receive proportionately less groundwater and may provide marginal thermal habitat in typical-flow years, but during low-flow years, conditions can exceed the lethal limits. Thermal conditions in the Little Betsie River are optimal for juvenile steelhead: summer temperatures only rarely warm above the optimal growth rates. Dair Creek temperatures are lower than optimal growth rates for much of the year, but this is not likely to limit the number of juvenile steelhead produced in this stream.

Spring temperature regimes in the Betsie River give a very short window of opportunity for the process of smoltification and emigration to occur, particularly when the warming trend occurs rapidly. This short duration may cause fish to stay in the river as the water temperature increases above the required temperature for smoltification (Zaugg and Wagner 1973, Wedemeyer 1980) and may have serious implications for stocking fish. If smolts are stocked too late or too far upstream in the watershed, they will not have time to emigrate before reverting back to parr and residualizing within the river. This would put hatchery fish in direct competition for space with wild produced fish and this competition would be most extreme in marginal habitats during years of low flow.

If Grass Lake Dam were removed, the largest improvement in temperature would be observed in the reach from Green Lake to Grass Lake. Because of the lack groundwater contributions, little improvement to the thermal regime would be observed in the Grass Lake to Thompsonville reach during either a typical-flow or low-flow year (Figures 36 and 37). The hypolimnetic flow diversion and release would provide no thermal benefit for trout species in the Green Lake to Grass Lake reach but it would decrease the mean summer temperature in the Grass Lake to Thompsonville reach (Figure 38). When compared with downstream locations such as M-115, Homestead and Little Betsie River, mean monthly water temperature under the dam removal scenario is still greater than the mean monthly temperature at sites with greater densities of juvenile steelhead (Chapter 2). Because the river begins as a surface outflow from a lake, the upper river water temperatures are dominated by the lake surface temperatures which tend to be fairly stable as a result of the ability of the large body of water to retain heat. This stable tendency, combined with the large outflow of water that begins the river (approximately 0.9 - 3.3 m^3/s during the summer) and daily climatic conditions is what directs the thermal regime in the upper watershed until enough groundwater enters into the channel and decreases the water temperature.

Areas where groundwater is seeping into the stream channel may provide important micro-thermal habitat areas that give refuge to fish during periods of high water temperature. Bilby (1984) identified 4 types of micro-thermal habitat areas as lateral seeps, pool bottom seeps, cold tributary mouths and flow through the streambed. In areas of the Betsie River watershed, where the river flows near glacial moraines, lateral seeps are evident at the surface near the channel, although their influence has not been quantified other than as an increase in discharge. These types of features are limited in the reach between Grass Lake and Green Lake, and may be too few to provide refuge to a large number of juvenile steelhead. One potential habitat management alternative is to identify these features in this reach and provide instream structures to prohibit mixing of the warm and cold water as much as possible (Bilby 1984). Objects, such as woody debris, that would divert the warm water or create a small pool effect may increase these potential refuge areas. Additional research would need to be conducted to assess the existence of these cooler micro-thermal areas and the feasibility of their development.

Other research found thermally stratified pools to be an important microhabitat for adult steelhead in the summer months (Nielson et al. 1994). However, the reach between Grass Lake and Thompsonville has a wide channel, likely a remnant effect from logging in the 1800's, with a steep gradient. Pools are scarce nor are they deep enough to stratify.

SUMMARY

The results of my investigation on the Betsie River steelhead resources and thermal limitations provided information on juvenile steelhead population dynamics in a thermally marginal watershed. From 1993-1996, the watershed produced a limited number of wild steelhead smolts, <3,000 per year, of which approximately 50% were age-1. Although the smolts emigrated at a young age, they were comparable in size to smolts observed in the Little Manistee River, 180-200 mm (Seelbach 1983). Time-lapse videography was a useful approach to obtain an estimate of smolts, although the 20 minute direct observation samples were also effective. In both cases, sampling 60% of the days throughout the duration in the smolt run would have produced estimates and errors similar to those obtained when sampling the entire duration.

I found stark differences among the juvenile steelhead parr densities at sites throughout the Betsie River watershed in the summer months. Estimates ranged from 0 to over 3,000/ha at sampling sites in the watershed. The densities of juvenile steelhead were lowest in the upper reach of the river above Thompsonville and temperature studies indicated a summer thermal range above the desired thermal limits in this reach of river. The largest densities of juvenile steelhead were in the tributaries and in the Betsie River below Thompsonville. Results of the temperature modeling study further indicate that preferred thermal habitat is limited to the 33 km above Homestead Weir during the peak of the summer heat in July. This reduction of available habitat based on thermal suitability may indicate temporal, density dependent competition, especially for the larger, age-1 and age-2 juvenile steelhead.

The densities that I observed for juvenile steelhead in the tributaries were similar to those found in Sand Creek and Silver Creek in southern Michigan (Dexter 1993a and 1993b). These results may suggest similar levels of productivity that could be used to assess production of juvenile steelhead from different watersheds. In particular, the results from the Betsie River are most applicable in watersheds where the main channel is marginal thermal habitat for juvenile steelhead.

Although summer thermal habitat appears to limit the distribution of juvenile steelhead, a shortage of winter habitat may influence mortality rates. Annual instantaneous mortality for juvenile steelhead in the watershed ranged from 0.710 to 3.576 and net channel losses were greatest in the Betsie River and least in the tributaries. Mortality rates were not related to summer temperatures, but were significantly related to minimum winter mean daily temperatures and winter severity. After the coldest winters, the highest mortality rates were observed and the fewest smolts emigrated in the spring.

I used a watershed approach in evaluating the juvenile steelhead abundance and distribution in the Betsie River and its tributaries. Had I not sampled the tributaries, I would have neglected a large portion of the juvenile population. In terms of total watershed abundance, I found that almost 50% of the total juvenile steelhead abundance occurred in the 4 tributaries sampled, yet these tributaries comprised only 11% of the total channel considered in this study. The tributaries are the best rearing environment for juvenile steelhead with the most desired thermal regime and lowest loss rates. This evidence can now be used to prioritize management activities in the watershed to further protect riparian habitat and decrease soil erosion, particularly in the tributaries which, due to their small size, can be sensitive to small changes in bedload.

In evaluating the thermal regime for water temperature in the watershed, I developed several empirical models and two physical process models for evaluating temporal and spatial patterns in thermal regime. In addition, a stream reach physical process model was used to assess management alternatives for altering the summer temperature regimes.

The empirical models predicted water temperature reasonably well with the exception of very cold weather. It is likely that surface ice or snow as well as the physical properties of water at very low temperatures caused this reduced predictive power. However, these models can be used to determine water temperature in the Betsie River unless major landscape features change within the watershed.

The two physical process models gave insight to the spatial and temporal thermal dynamics on a daily time-step for summer in the stream reach model and on a weekly time step for the annual stream network model. Removal of Grass Lake Dam would result in decreased summer water temperatures from Green Lake to Thompsonville, although in low-flow, hot summers, the benefits would be minimal. Even with the removal of Grass Lake dam, water temperatures in this reach still could rise above the optimum growth limits and at times exceed the preferred and zero-growth limits for juvenile steelhead. Consideration should be given to the trade-offs between the benefit to the fishery and losses in recreational opportunities if this dam were removed.

The siphon from the hypolimnion of Green Lake would lower water temperatures to the preferred range for juvenile steelhead in the reach from Grass Lake to Thompsonville. However this option would require the construction of a dam to control water flow at the outlet of Green Lake and construction of a diversion. Again, a cost benefit analysis would be prudent if this management option were considered. Other management options that might result in potential thermal benefits include additional shading in the reach above Thompsonville and the addition of habitat diversity in several reaches that are basically homogenous runs. Both of the physical process models could be used to guide future management options in the Betsie River and were extremely useful tools in assessing stream thermal dynamics.

Future research in the Betsie River could include several topics. Bioenergetically, the watershed proves to be taxing in some areas and entirely beneficial in others. A study of the bioenergetics from the watershed perspective would provide biological insight to the conditions that these juvenile steelhead face. The mobile sand bedload that resulted from the Thompsonville Dam collapse may be suppressing invertebrate populations and this combined with temperatures above the preferred for juvenile steelhead could partially be the mechanism behind the watershed's limited capacity to produce smolts. A bioenergetic investigation would illuminate some of these potential forage and growth limitations. Additionally, research into the rates and periodicities of movement conducted by this juvenile population, coupled with a bioenergetics study, may lead to an understanding between the watershed thermal dynamics and fish behavior regarding thermal optimization.

In terms of the steelhead fishery, neither the number of adult steelhead returning to the river or the number of steelhead harvested from within the watershed is presently unknown. This information is crucial to determining the appropriate levels of stocking in combination with wild production. Wild production, while seemingly small from 1993-1996, comprised a large portion (90%) of the fishery in the late 1980's (Seelbach and Whelen 1987) when harvest was estimated to be approximately 1,200 -

2,600 adult steelhead (Rakoczy and Rogers 1987, 1988, and 1990). Conservative survival and return rates (7%) from the mean number of wild smolts leaving the watershed in 1993-1996 (1,540) predict that approximately 108 wild adults will return to the river annually in years 1995-1999. One reason for the low number of wild smolts observed throughout the course of my study may be due to environmental extremes: the winter of 1993-1994 was the coldest winter in 22 years, the drought conditions of 1995 had not occurred in 17 years, and a large flood occurred in 1996. Alternatively, the low number of smolts could reflect the stream's limited carrying capacity due to changes in habitat that resulted from the collapse of Thompsonville Dam. If this is the case, wild smolt abundance may increase as the channel stabilizes following this systemic perturbation. Finally, because their growth rates are high, the Betsie River steelhead may experience higher rates of survival than observed in other streams which would yield a larger return to the fishery than I have predicted.

Future management efforts of the Betsie River watershed should incorporate actions that may moderate the effects of extreme temperatures. For winter, options for reducing the severity of the stream temperatures may be limited. Increased shading may result in less radiant heat loss and subsequently less frazil and anchor ice formation. Maintenance of loose gravel and cobble substrates, either by introduction of the substrate or reduction of the sand/silt bedload, could also provide additional habitat during the winter.

For the extreme summer temperatures, one possibility for increasing summer habitat could be assessed regarding the feasibility of protecting the cooler micro-thermal areas in the stream above Thompsonville. Most certainly, protection of the tributaries and areas of the watershed that introduce groundwater will provide continued sustainment of the wild production. In particular, maintenance of the riparian and reductions in sediment loads from road crossings or other activities could provide large benefits for the continued production of juvenile steelhead from these areas. However, given the unique hydrology of the watershed in that groundwater contributions are not significant until below Thompsonville Dam, in low-flow, hot years, there may not be any actions that could moderate temperatures. Variability is inherent in natural resource management and some water years will lead to better habitat and greater juvenile steelhead production than others.

APPENDICES

APPENDIX A

APPENDIX A

Procedures for Establishing and Maintaining Staff Gages and Developing Rating Curves in the Betsie River Watershed

Gaging Sites and Comments on Gage Quality

1. Green Lake: DNR established gage, good quality, good place to measure
2. Grass Lake: DNR gage attached to Grass Lake Dam, poor rating curves
3. Woirel Bridge: my gage, vandalized early in study, good flow and gage relationship prior to its loss
4. Liitle Betsie River: ruler gage along Highway 669, good gage and flow measures
5. Thompsonville: several different gage placements, channel changed frequently, poor rating curve, good flow measurements
6. Homestead Weir: DNR gage on fishing platform above dam, good relationships between flows both upstream and downstream of the dam

Gage Reading and Protocols for Discharge Measurements

During May and June, the Homestead Weir gage was read daily. All other gages were recorded on a weekly basis throughout the summer. Volunteers read the gages during the winter. Discharge was measured several times throughout the course of the study. I tried to capture low, median, and high stream flows to develop the rating curves.

Rating Curves

Measured discharges and gage readings for each site were log transformed and I used regression analysis to determine the relationship that best fit the data. Extreme outliers were considered individually and removed when it became apparent that either a gage was misread or discharge measures were in error. This only occurred on 3 different instances at different gages in different years.

Rating Equations	r^2 value	significance level
Green Lake		
Log (Q) = Log(GH) * 5.6137-2.042	0.913	0.0019
Grass Lake Dam		
Log(Q) = Log(GH)*9.3299 - 6.1032	0.956	0.0005
Woirel Bridge		
Log (Q) = Log (GH)*0.221 + 1.907	0.961	0.0020
Little Betsie River		
Log(Q) = Log(GH)*0.480 + 0.766	0.871	0.0040
Thompsonville		
Log(Q) = Log(GH)*0.7005 + 1.2867	0.649	0.6490
Homestead Weir		
Log(Q) = Log(GH)*0.2196 + 1.7547	0.972	0.0001

Where, Q = discharge, GH= gage height.

Missing Data

I used regression analysis to determine relationships between gaging stations for filling in missing data. The following equations were used to fill data when necessary.

Regression Equation	r^2	P- value
Thompsonville (TV)		
Log (TV-Q) = Log (HS-Q) + Log (LBR-Q) -0.6661	0.931	0.0003
Green Lake (GL)		
Log (GL-Q) = Log (TV-Q)*1.2145 - 0.7380	0.701	0.0004
Homestead Weir (HS)		
Log(HS-Q)= Log(TV-Q)*0.5626+1.1421	0.758	0.0001
Little Betsie River (LBR)		
Log(LBR-Q) = Log(HS)*5.0382 + Log (TV)*4.8067+2.920	0.971	0.0001
Dair Creek (DC)		
Log (DC) = Log(LBR-Q)*0.4610+Log(HS)*1.0619-1.8828	0.974	0.0120

APPENDIX B

Appendix B. Esimates of steelhead parr abundance, standard errors, and 95% confidence intervlas for each site sampled throughout the Betsie River watershed, 1993-1996.

	SUMMER 1993			SUMMER 1994			SUMMER 1995			SUMMER 1996		
Site	Steelhead Estimate	Standard Error	95% C. I.	Steelhead Estimate	Standard Error	95% C. I.	Steelhead Estimate	Standard Error	95% C.I.	Steelhead Estimate	Standard Error	95% C.I.
BETSIE RIVER												
Woirels Bridge (East)	0	.	.	0	.	.	0			0		
Woirel's Bridge(West)	2	.	.	1	.	.	12	6.45	9-26	39	24.60	23-89
Haze Road	0	.	.	.	.	.	.			.		
King Road Upper	.	.	.	1	.	.	5	1.19	5-8	0	.	.
King Road Lower	.	.	.	0	.	.	4	0.54	4-6	3	.	.
Thompsonville	9	2.61	9-15	4	.	.	19	13.53	12-47	8	10.92	5-34
Highway M115	99	31.51	48-193	313	6.66	300-326	464	12.59	439-489	652	7.24	638-667
Psutka Road	3	0.27	3-4	14	5.64	14-26	62	7.09	53-76	10	4.47	10-22
Homestead Weir	13	1.17	7-15	7	3.60	6-16	35	5.43	30-46	5	.	.
River Road	0	.	.	0	.	.	0	.	.	0	.	.
LITTLE BETSIE RIVER												
Hwy669	8	1.40	8-9	.	.	.	.			.		
Bentley Road West	31	0.93	31-33	159	6.87	145-173	127	1.29	126-130	29	1.08	29-32
Bentley Road East	.	.	.	102	7.17	90-116	56	11.05	44-78	13	2.73	9-11
DAIR CREEK												
M115	.	.	.	60	6.89	55-66	44	1.05	44-46	28	0.63	28-30
Landis Road	12	0.53	12-13	41	70.03	15-103	53	3.95	49-66	136	2.71	38-43
SMALL TRIBUTARIES												
Red Creek	8	0.87	8-9	21	0.74	21-23	30	20.78	18-72	12	0.31	125-152
Steeves	14	0.63	14-15	21	1.03	21-23	34	1.98	33-38	37	1.86	36-41

Appendix B (Continued). Estimates of abundance, standard errors, and 95% confidence intervals for steelhead parr at each sampling site throughout the Betsie River watershed, 1993-1996.

	FALL 1994			FALL 1995		
Site	Steelhead Estimate	Standard Error	95% C.I.	Steelhead Estimate	Standard Error	95% C.I.
BETSIE RIVER						
Woirels Bridge (East)	0			0	.	.
Woirel's Bridge(West)	13	3.90	11-21	13	1.17	12-5
Haze Road	.	.	.	.	.	.
King Road Upper	0	.	.	2	1.00	2-15
King Road Lower	0	.	.	0	.	.
Thompsonville	25	6.63	20-39	11	0.63	10-12
Highway M115	385	6.66	372-398	349	8.07	333-365
Psutka Road	5	0.00	.	40	3.43	37-47
Homestead Dam	.	.	.	.		
River Road	.	.	.	.	.	.
LITTLE BETSIE RIVER						
Bentley Road East	70	2.19	64-73	44	0.70	43-44
Bentley Road West	105	5.09	97-115	101	0.98	99-106
Hwy669	.	.	.	.		
DAIR CREEK						
M115	50	2.47	50-51	32	0.53	29-32
Landis Road	50	0.39	50-51	65	0.96	65-69
SMALL TRIBUTARIES						
Red Creek	5	.	.	8	0.95	8-9
Steeves	8	0.40	8-9	7	0.40	7-9

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