Stocking Trout of Wild Parentage to Restore Wild Populations: An Evaluation of Wisconsin's Wild Trout Stocking Program

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ABSTRACT—The Wisconsin Department of Natural Resources (WDNR) manages trout streams using a combination of stream habitat protection and improvement, fishing regulations, and stocking of hatchery-reared trout. The WDNR initiated a wild trout stocking program in 1995 to improve the quality of hatchery-reared brook and brown trout by raising offspring of wild parentage. The goals of the wild trout stocking program are to increase the survival and longevity of trout stocked in streams and to ultimately develop self-sustaining populations of wild trout. It is thought that hatchery trout of wild parentage maintain the genetic diversity and better embody the characteristics found in wild populations and may therefore improve restoration success. I collectively analyzed evaluations of wild trout stocking across Wisconsin to determine whether program goals were being fulfilled and to identify any research gaps. Preliminary analyses indicated survival rates 2-4 times greater for stocked trout of wild versus domestic parentage, and some increases in natural reproduction have been observed. Habitat, however, may be limiting the restoration of self-sustaining populations in some streams. Future research will address habitat limitations to survival and reproduction of stocked wild trout and the long-term viability of source populations for the wild trout stocking program.

Introduction

The Wisconsin Department of Natural Resources (WDNR) manages trout fisheries in 10,371 miles of classified trout streams using a combination of stream habitat protection and improvement, fishing regulations, and stocking of hatchery-reared trout. About 40% of the trout stream mileage (Class I) support natural reproduction sufficient for the maintenance of wild trout populations, but populations in 45% (Class II) require supplementation by stocking and populations in 15% (Class III) are wholly dependent on stocking. Through WDNR management efforts, miles of Class I and II streams have increased and miles of Class III streams have decreased in recent years. Trout habitat has been improved by land conservation measures, which have reduced siltation from erosion and improved groundwater flow (Gebert and Krug 1996), and by the restoration of damaged stream habitat (Hunt 1993). Hunt (1988) and Avery (2004) documented a half century (1953-2000) of evaluations of trout stream habitat improvement projects, which have been supported by annual trout stamp sales since 1978.

Trout stocking has a long history in Wisconsin dating back to the 19th century. Significant changes, however, have occurred in recent years. The WDNR initiated a wild trout stocking program in 1995 in contrast to its long-standing domestic trout stocking program. The idea of stocking trout of wild parentage has been around at least since the 1960's but rearing wild trout in a hatchery was impractical at that time (Flick and Webster 1964; Mason et al.

1967). Domestic strains of hatchery-reared trout often failed to sustain a fishery beyond the early season and failed to contribute to natural reproduction in Wisconsin streams (Avery et al. 2001). Poor survival of domestic trout stocked in Wisconsin streams was observed in the early 1990's during a time of severe drought and harvest prohibition (Avery et al. 2001). This prompted interest in developing a wild trout stocking program to improve the quality of hatchery-reared trout and their potential for sustaining stream fisheries.

Wisconsin's wild trout stocking program involves taking wild brook trout *Salvelinus fontinalis* and brown trout *Salmo trutta* from streams, spawning them at a fish hatchery, and later returning the spawned trout to the streams from which they came. Offspring of the wild parents are raised at reduced densities in a hatchery and stocked elsewhere in the state as spring or fall (autumn) fingerlings or spring yearlings. It is thought that these trout of wild parentage better maintain the genetic diversity found in wild populations and embody the characteristics of wild trout as compared to offspring from domestic broodstock. To help maintain "wildness" in the trout, human contact is minimized by partially shading tanks at the hatchery and feeding trout continuously throughout the day using an automatic feeder. The 2003-2004 wild brook trout stocking quota included 95,500 spring fingerlings and 7,600 fall fingerlings; the wild brown trout stocking quota included 40,700 spring fingerlings, 28,250 fall fingerlings, and 6,980 spring yearlings. Wild brown trout were stocked statewide and wild brook trout were only stocked in the southwestern genetic management zone.

The goal of the wild trout stocking program is to use hatchery-reared trout of wild parentage to develop self-sustaining populations of brook trout and brown trout in waters that lack such populations. Specific objectives include increasing the survival and longevity of stocked trout in streams and establishing natural reproduction. This program has become an integral part of trout management in the state. It is generally acknowledged that overall the trout fisheries in Wisconsin today have improved over what they were in the past, and it is thought that the wild trout program has played a key role in this recovery. The goal of this study was to evaluate Wisconsin's wild trout stocking program by collectively analyzing evaluations of wild trout stocking across the state to determine whether program goals and objectives were being fulfilled.

Methods

I distributed a memorandum to WDNR fisheries mangers and biologists requesting them to provide any available data, analyses, and reports pertaining to the evaluation of wild brook and brown trout stocking. I also requested that they include similar information that may be available concerning the evaluation of domestic brook and brown trout stocking prior to the start of the wild trout stocking program, particularly for streams that later received wild trout. I received unpublished data and reports on wild brown trout stocking in 15 streams and wild brook trout stocking in 1 stream. Included is one published report on wild and domestic brown trout stocking in two streams, the Waupaca River and the West Fork Kickapoo River (Avery et al. 2001).

I evaluated the data, analyses, and reports to determine if stocked wild trout had higher survival and longevity rates compared to domestic trout and to determine if natural reproduction was occurring as a result of wild trout stocking efforts. Survival was estimated by comparing densities of trout over time (number per mile); densities were estimated using a single marking run and a single recapture run unless noted otherwise. I reported survival rates as apparent survival rates because comparisons of densities more accurately reflected losses, which could be attributable to mortality or movement. The initial density used in apparent survival calculations was the initial stocking density. However, in some reanalyses of the data as noted in the results section, I used the estimated density after stocking had occurred as the initial density.

Results

There were sufficient data to evaluate survival, longevity, or reproductive success for stocked wild versus domestic trout in 9 of 15 streams stocked with brown trout (Table 1).

Table 1. Summary of wild brown trout stocking results in terms of survival, longevity, and reproductive success compared to domestic brown trout stocking. If there were insufficient data or results were inconclusive for a stream then the column was left blank.

Stream name	Stocked wild trout survival greater than domestic trout survival?	Stocked wild trout longevity greater than domestic trout longevity?	Successful reproduction from stocked wild trout?
Hunting River		Yes	Yes
McCaslin Brook	No	No	No
North Branch Oconto River	No	No	No
Onion River			Yes
Peshtigo River	Yes	Yes	
Rocky Run	Yes		
Rowan Creek	Yes		
Waupaca River	Yes		
West Fork Kickapoo River	Yes		

Apparent Survival

There was evidence that apparent survival rates of stocked wild brown trout exceeded apparent survival rates of domestic brown trout in five streams. Avery et al. (2001) evaluated the performance of stocked wild, domestic, and optimum domestic brown trout in the Waupaca and West Fork Kickapoo rivers. Optimum domestic trout were reared under conditions similar to those for hatchery wild trout: little human contact and at about half the density of standard hatchery protocol. Wild, optimum domestic and domestic fall fingerlings (age 0) brown trout were stocked in the Waupaca River in fall 1993 and 1994. Apparent survival ϕ was about 2–4 times greater for wild (ϕ = 0.22–0.34) versus optimum domestic (ϕ = 0.06–0.13) or domestic trout (ϕ = 0.10) after one year and about 4–8 times greater for wild (ϕ = 0.08) versus optimum domestic (ϕ = 0.02) or domestic trout (ϕ = 0.01) after two years. Wild, optimum domestic, and domestic spring yearling (age 1) brown trout were stocked in the West Fork Kickapoo River in spring 1994 and 1995. Densities in electrofishing stations increased for wild trout (indicating recruitment) and decreased for optimum domestic and domestic trout between the spring stocking and fall. I recalculated apparent survival rates using density in the fall after the spring stocking as the initial density and found that apparent survival from age 1 to age 2 was about 5 times greater for wild (ϕ = 0.53) versus optimum domestic (ϕ = 0.10) or domestic trout (ϕ = 0.11). A similar calculation for the Waupaca River showed that apparent survival from age 1 to age 2 was about 1.4–2.3 times greater for wild ($\phi = 0.23$) versus optimum domestic ($\phi = 0.17$) or domestic trout ($\phi = 0.10$). Apparent survival in the West Fork Kickapoo River from age 1 to age 2.5 (fall 1994 to spring 1996) was about 5-31 times greater for wild ($\phi = 0.31$) versus optimum domestic ($\phi = 0.06$) or domestic trout ($\phi = 0.01$).

Wild brown trout were stocked as spring fingerlings in six small Columbia County streams in southwestern Wisconsin. Apparent survival from age 0 (fall) to age 1 (fall) was about 2 times greater for stocked wild versus domestic brown trout (Tim Larson, WDNR, unpublished data). Average apparent survival was about 0.19 for stocked wild trout (2001-2003) versus less than 0.10 for domestic trout (1984-1987) in Rocky Run and Rowan Creek. Average apparent survival for stocked wild trout in four other streams was about 0.22 (Dell, Honey, Jennings, and Leech creeks, 2001-2003), but there were no data on apparent survival rates for domestic trout in these streams for comparison.

Apparent survival of stocked wild brown trout in the Peshtigo River was greater than that of domestic brown trout as was evident by the densities of age 2 and age 3 and older trout (David L. Brum, WDNR, unpublished data). There was no evidence of survival of domestic trout to age 3 and older in 1988 and 1997, even though many of those trout were stocked as spring yearlings. The number of domestic brown trout per mile based on single-pass electrofishing samples was 35 (age 1) and 7 (age 2) trout in 1988 and 26 (age 1) and 2 (age 2) trout in 1997. There was evidence of survival to age 3 and older for stocked wild brown trout in 2001-2003 (wild trout stocking began in 1998), with densities higher than densities of age-2 domestic trout. Average densities of stocked wild brown trout for 1998-2003 were 149 (age 0), 104 (age 1), 16 (age 2), and 9 (age 3 and older). A comparison of densities of stocked wild brown trout showed apparent survival rates of 0.11-0.15 from age 1 to age 2.

Stocked wild brown trout did not survive better than domestic brown trout in McCaslin Brook and the North Branch Oconto River (Lee Meyers, WDNR, unpublished data). McCaslin Brook is a tributary of the North Branch Oconto River. Wild brown trout were first stocked in the North Branch Oconto River in 1996 and in McCaslin Brook in 1997. Historic population estimates showed brown trout densities of 139 per mile in 1973, 385 per mile in 1988, and 74 per mile in 1996 in McCaslin Brook. Brook trout were also present (10 per mile in 1973 and 55 per mile in 1988). After wild brown trout were stocked in 1997, densities increased to 545 per mile in 1997 and 1,300 per mile in 1998. However, warm water temperatures of 26.7 °C and higher were recorded on three occasions in July 1999, and the brown trout density had decreased to 218 per mile by August 1999. The brook trout density, however, had not decreased (63 per mile in 1999). Similar population estimates were not available for the North Branch Oconto River, but a creel survey confirmed that stocked wild brown trout did not provide for a significant fishery. Concerns that stocked wild brown trout were not surviving prompted a return to stocking domestic yearling brown trout in 2000. A creel survey in 2000 found that the trout harvest included 71% domestic brown trout, 25% wild brook trout, and 4% wild brown trout (stocked or naturally occurring; all from the North Branch Oconto River).

Apparent survival could not be estimated for brown trout in the Hunting River, Onion River, and Pine River and brook trout in the West Branch Eau Claire River because stocked wild naturally produced and domestic fish or age classes could not be separated in the data. Therefore, these evaluations were inconclusive on the question of whether stocked wild versus domestic trout survive better in these streams.

Longevity

Longevity of stocked wild brown trout was not greater than longevity of domestic brown trout in McCaslin Brook and the North Branch Oconto River because survival of stocked wild brown trout was poor as outlined above. There was, however, evidence of increased longevity for stocked wild brown trout in the Hunting and Peshtigo rivers. The 1,000 wild brown trout stocked in the Hunting River in 1996 at age 0 had a year-specific fin clip and two of these fish were recaptured in 2003 at age 7 (David A. Seibel, WDNR, unpublished data). In the Peshtigo River there was no evidence of survival of domestic brown trout past age 2 in 1988 and 1997, but there was evidence of stocked wild brown trout surviving to age 3 and older (2001-2003 with stocking beginning in 1998). Longevity of stocked trout could not be evaluated for any of the other streams because age classes could not be separated in the data or the length of the study was too short.

Reproduction

Evidence of reproduction consistent with wild trout stocking was observed in the Hunting and Onion rivers. Young-of-year brown and brook trout were observed from 1999 to 2003 in the Hunting River, indicating that natural reproduction had occurred. Population estimates also suggested that natural reproduction had occurred (Table 2). The density of stocked wild brown trout (last stocked in 2001) generally decreased from 1998 to 2003, whereas the density of stocked domestic (last stocked in 1999) and naturally produced brown trout, which were confounded in the data, increased from 1998 to 2000. Domestic and naturally-produced brown trout decreased thereafter, but densities were 5 to 9 times greater than stocked wild brown trout. The increase in density of domestic and naturally produced brown trout in 2000 and sustained higher densities thereafter were consistent with reproduction that may have resulted from stocked wild brown trout. Reproduction from domestic trout cannot, however, be ruled out.

Table 2. Number of stocked wild and domestic brown trout by age and density (number per mile (No./mi)) of stocked wild brown trout and combined stocked domestic and naturally produced brown trout (age 1 and older).

Age	1995	1996	1997	1998	1999	2000	2001	2002	2003
				Number of stocked wild brown trout					
0		1,000	2,000	1,000	500	2,000	1,000		
No./mi				**482**	**125**	**134**	**83**	**32**	**17**
				Number of stocked domestic brown trout[1]					
0		1,000							
1	1,000		1,000	1,000	1,000				
2				644					
No./mi		**67**		**414**	**467**	**546**	**439**	**201**	**159**

[1] *Density (No./mi) is combined stocked domestic and naturally-produced brown trout*

Evidence of reproduction in the Onion River was observed in an increase in the number of unclipped brown trout over time (Table 3) (John E. Nelson, WDNR, unpublished data). The Onion River was historically stocked with domestic brown trout. Spring yearling domestic brown trout were stocked in 1995 and were last stocked in 1997 along with some age-2 trout. The stocking of wild brown trout started in 1997 with the transplant of age-1 wild brown trout from two streams in the Coon Valley watershed. These fish received an adipose clip. Fall fingerling wild brown trout were also stocked in 1997, but these did not receive an adipose clip. No fish were stocked in 1998, spring yearling wild brown trout were stocked in 1999 (adipose clip), and fall fingerling wild brown trout were stocked in 2000 (adipose clip). There have been no subsequent stockings in the Onion River watershed. The number of unclipped trout per mile (single-pass electrofishing counts) increased from 100 in 1999 to 281 in 2001. This increase in unclipped trout was consistent with the potential spawning of wild trout stocked in 1997 (adult transfers and fall fingerlings). Evidence of reproduction was found in the unclipped trout observed in 1999-2001, many of which were less then four inches in total length.

Table 3. Number of unclipped and clipped brown trout per mile from 1997 to 2001 in the Onion River.

Year	Number of unclipped trout per mile	Number of clipped trout per mile
1997	89	
1998	161	80
1999	100	110
2000	180	31
2001	281	56

Reproduction of stocked wild brown trout was not observed in McCaslin Brook and the North Branch Oconto River because survival of stocked wild brown trout was poor as outlined above. Reproductive success of stocked trout could not be evaluated for any of the other streams because stocked fingerlings were unmarked and could not be distinguished from naturally-produced trout or it was not an objective of the study (e.g., Avery et al. 2001).

Discussion

Early investigations of the performance of stocked wild trout versus domestic trout showed higher survival rates for stocked wild trout. Flick and Webster (1964) investigated differences in survival during the first year after stocking for spring and fall fingerling brook trout of domestic versus wild parentage. Oversummer survival was greater for wild (0.65-0.76) versus domestic (0.43-0.53) brook trout fingerlings; overwinter survival did not differ but was likely confounded with the larger size advantage of domestic trout. Mason et al. (1967) investigated survival of domestic, wild, and domestic/wild hybrid brook trout stocked in five central Wisconsin streams as fall fingerlings. Domestic brook trout had a higher overwinter survival rate (0.38) than stocked wild brook trout (0.25); however, after one complete year, stocked wild brook trout had a higher survival rate (0.10) than domestic brook trout (0.007).

The study by Avery et al. (2001) on wild trout stocking in the Waupaca and West Fork Kickapoo rivers was initiated to further quantify the field performance of stocked wild versus domestic trout specifically in Wisconsin streams. Early results in this study were positive in favor of wild trout stocking, and the wild trout stocking program was spread to other streams throughout the state. The original intent of wild trout stocking was for it to be a temporary management action towards establishing self-sustaining populations. This goal may have been achieved in the Hunting River. Although survival rates could not be determined from the data for stocked wild versus domestic brown trout in the Hunting River, there was evidence of longevity in the observation of age-7 stocked wild trout and there was evidence of reproduction. No stocking will occur in the Hunting River from 2002 to 2006, whereupon the need to resume stocking will be evaluated. The Onion River has also had successful reproduction since being stocked with wild brown trout. There has been no stocking in the Onion River since 2000; future evaluations of the trout population will determine the ultimate success of wild trout stocking in the Onion River.

Apparent survival rates of stocked wild trout have exceeded apparent survival rates of domestic trout as long as habitat was not a limiting factor. Apparent survival rates were generally at least two times greater for stocked wild versus domestic brown trout from age 0 to age 1 or from age 1 to age 2. Survival rates to older ages were even greater for stocked wild trout and have resulted in increased longevity. Stream habitat may, however, determine just how much greater the survival of stocked wild trout versus domestic trout may be. For example, the apparent survival from age 1 to age 2 for stocked wild versus domestic brown trout was about 5 times greater in the West Fork Kickapoo River as compared to about 2 times greater in the Waupaca River. The West Fork Kickapoo River is a highly fertile river compared to the Waupaca River and it has been suggested that the higher growth rates observed in the West Fork Kickapoo River were responsible for the higher survival rates (Avery et al. 2001).

Habitat was a limiting factor in McCaslin Brook and the North Branch Oconto River, where summer maximum water temperature exceeded 26 °C on several occasions in 1999. The wild trout stocking observations from these streams underscores the importance of stream habitat to supporting wild trout populations. A wild trout stocking program cannot substitute for quality trout habitat. Wisconsin's active stream habitat restoration program, which has a dedicated funding source via the sale of trout stamps, and Wisconsin's land conservation measures have helped to improve trout stream conditions such that the wild trout stocking program serves as a viable management tool.

Apparent survival of stocked trout can be improved by using trout of wild parentage, but successful reproduction may not necessarily follow. Many streams may support juvenile and adult trout but fail to provide adequate spawning habitat. Here again, habitat limitations need to be surpassed before the goal of establishing self-sustaining trout populations can be realized. However, the question is raised as to whether wild trout stocking may be preferred over domestic trout stocking in situations where successful reproduction may not be realized. If stocked wild trout can survive from year to year in streams that lack spawning areas, then wild trout stocking will work for supporting trout fisheries in those streams. Fisheries managers will have to determine whether the costs of a wild versus domestic trout stocking program are justified for such streams and their fisheries.

Brook trout are the only stream salmonid native to Wisconsin. Brown trout have been successfully introduced throughout the state and coexist with brook trout in many streams. However, brown trout have also displaced brook trout in many streams. Successful wild brown trout stocking may therefore be an impediment to protecting or restoring brook trout populations. Mixed brook trout and brown trout populations were present in the Pine River, McCaslin Brook, and the North Branch Oconto River. Brook trout were not stocked in McCaslin Brook and the North Branch Oconto River, but they were self-sustaining and constituted about 25% of the trout fishery. Interestingly, whereas high summer water temperatures limited stocked wild brown trout in 1999, wild brook trout persisted, possibly by finding suitable refuge. Future wild brown trout stocking will be avoided in McCaslin Brook, the North Branch Oconto, and other similar streams with brook trout populations in northeastern Wisconsin (Lee S. Meyers, WDNR, personal communication).

The potential for brook trout restoration should also be a consideration when deciding to stock wild brown trout in streams without self-sustaining trout populations. David M. Vetrano (WDNR, personal communication) has commented that wild brown trout populations have been established in westcentral Wisconsin streams that at the time would not have supported brook trout. Subsequent improvements in land use have improved groundwater flow such that those streams would now have been suitable for brook trout. The presence of brown trout is now an obstacle to brook trout restoration.

When developing a wild trout stocking program, consideration should be given to the genetics of the source populations. Stocking trout derived from wild parents helps to avoid overwhelming native genetic diversity and to prevent the loss of genetic diversity. Fields and Philipp (1998) documented levels of genetic diversity consistent with distinct stocks of brook trout. Therefore, different source populations for brook trout are needed for different parts of the state. Brook trout from the Ash Creek source population are currently stocked in the southwestern genetic management zone. Brook trout source populations for the northern part of the state were recently identified in Dority Creek and the South Fork of the Hay River (Heath M. Benike, WDNR, personal communication). Other source populations have been used in the early stages of the wild trout stocking program but have been discontinued from use due to disease issues. Genetic analyses of Wisconsin brown trout populations have determined that wild brown trout from the southwestern Timber Coulee Creek source population can be stocked statewide. However, stocked wild brown trout from northeast source populations (West Branch White River and Brule River) were found to have survival rates about four times greater than those from the southwestern population when stocked in the Waupaca River in northeastern Wisconsin (Al Niebur, WDNR, unpublished data).

Future studies of wild trout stocking are needed to better understand how habitat conditions determine the improvement in survival for wild stocked trout versus domestic stocked trout and therefore in which types of streams better success can be expected. Large annual variation in salmonid survival is common (Needham et al. 1945; Hunt 1969; Seelbach 1993; Mitro and Zale 2002); therefore, long-term studies may be necessary. Study designs should ensure that stocked wild trout and domestic trout can be distinguished from each other and from naturally-produced trout in the study stream, and if possible, among cohorts. Batch tags such as visible implants of fluorescent elastomer (Northwest Marine Technology, Inc.) used in different colors and locations on the trout are suitable for this purpose. Clipping fins can also be used to distinguish batches of fish, but batch codes are obviously somewhat limited. Consistent use of the same electrofishing stations from year to year will ensure that valid comparisons of densities can be made among years.

I am currently initiating a study to investigate the long-term viability of wild brook trout and brown trout populations as source populations for Wisconsin's wild trout stocking program. Wild brook trout have been obtained from Ash Creek since 1999 and wild brown trout have been obtained from Timber Coulee Creek since 1995. Little is known about the trout population in either stream. Each stream supports a wild trout population protected from harvest by a no-kill regulation. A sufficient number of trout have been captured to meet yearly egg quotas (about 198,000 brook trout eggs and 114,000 brown trout eggs in 2002) for the wild trout stocking program. However, we do not know what effect the annual removal of reproductive output from each stream has had on the long-term viability of each source population. Dr. Brian Sloss (Wisconsin Cooperative Fisheries Research Unit, University of Wisconsin-Stevens Point) is initiating a companion study to examine the potential and realized genetic impacts of wild broodstock selection in Wisconsin's wild trout stocking program. Together, these studies will result in a quantitative understanding of the effects of broodstock selection and egg collection on the source populations for the wild trout stocking program and will aid management decisions such that the viability of the source populations for the wild trout stocking program can be maintained.

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Quirk Creek Brook Trout Suppression Project

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ABSTRACT— A brook trout *Salvelinus fontinalis* suppression project utilizing anglers was initiated in 1998 to facilitate recovery of native westslope cutthroat trout *Oncorhynchus clarkii lewisi* and bull trout *Salvelinus confluentus* populations in Quirk Creek, Alberta. Participating anglers were required to pass a fish identification test, harvest all brook trout caught and release all other fish. Only 15 of the 7955 fish harvested were not brook trout. Although brook trout catch rates remained relatively high (1.0-2.5 fish/angler-h) and brook trout dominated the angler catch (54-73%), brook trout density has declined since 2000 while the abundance of juvenile cutthroat trout increased in 2003, resulting in a decline in the proportion of brook trout in the electrofishing catch.

Introduction

Brook trout *Salvelinus fontinalis*, although not native to Alberta, are present in many montane and foothills waters as a result of extensive stocking. In southern Alberta, brook trout populations have generally increased while native westslope cutthroat trout *Oncorhynchus clarkii lewisi* and bull trout *Salvelinus confluentus* populations have declined. Brook trout life history attributes (early spawning age, reduced longevity and low catchability) have resulted in the replacement of native bull trout and cutthroat trout fisheries with fisheries for smaller, less-catchable, non-native brook trout.

Management programs to reduce or eliminate non-native trout populations often involve pesticides and/or electrofishing (Moore et al 1983; Buktenika 1997; Kulp and Moore 2000). However, Larson et al. (1986) suggested that experimental angling programs might offer a cost-effective, alternative method for reducing densities of non-native trout. Although Larson's study only ran nine weeks, it appeared that anglers reduced the non-native trout population by about 10%.

Since pesticides are only suitable in certain situations and there are insufficient resources to attempt removal of non-native trout by electrofishing in all streams where native trout populations appear to be threatened, the option of selectively removing non-native trout by angling provides an appealing alternative. Our objective in this study was to determine whether angling could be an effective method for reducing densities of non-native brook trout in Quirk Creek, Alberta, to facilitate recovery of the native trout population.

Study Area

Quirk Creek is located 50 km southwest of Calgary in a designated off-highway vehicle (OHV) area. A good dirt road comes within 0.5 km of the creek for most of its length. Anglers participating in this project were allowed direct vehicle access to this road by fording the Elbow River, but only on supervised outings under the direction of the volunteer coordinator. A locked gate prevents anglers on unsupervised outings from crossing the Elbow River by vehicle.

Most of Quirk Creek meanders through a large wet meadow dominated by grasses and low (< 1 m) shrubs. Although cattle and OHVs have degraded streambanks in a few areas, most of the streambanks are undamaged and provide good fish habitat, consisting of deeply undercut banks with overhanging terrestrial vegetation. The lower 2 km of creek flows through a narrow valley before joining the Elbow River at an elevation of 1530 m. There are no permanent barriers on the creek, although beaver dams up to 1.5 m high are scattered along the creek.

Brook trout colonized Quirk Creek subsequent to their introduction to the Elbow River watershed in 1940. Although native cutthroat trout and bull trout were the only fish captured in Quirk Creek in 1948, brook trout had colonized the lower 3 km of the creek by 1978, comprising 35% of the fish population, and spread throughout the entire creek by 1995, comprising 92% of the fish population. These changes occurred despite the implementation of reduced bag limits and minimum size limits designed to provide more protection for native trout (Stelfox et al. 2001a). Since 1998, harvest of all fish has been prohibited in Quirk Creek, except by anglers participating in the project.

A bridge divides Quirk Creek into upper and lower reaches, with the lower reach serving as a control during the first two years of the project. Surface areas of the upper and lower reaches were estimated to be 1.65 and 1.8 ha, respectively, by extrapolating the mean widths of the creek (3.3 and 3.6 m) within the respective electrofishing sites to the approximate lengths of each reach (5 km).

Methods

Fish Identification Education

To participate in the project, all anglers had to pass a fish identification test on an annual basis to demonstrate their ability to identify the three fish species found in Quirk Creek. If a person failed the test on their first attempt, they were given a dichotomous key with pictures of the key-identifying features (a list of key-identifying features in 1998) and were permitted to take the test a second time with the key (list) in front of them. For a more detailed discussion of the fish identification test, refer to Stelfox et al. (2001b).

Angling

Participating anglers were required to harvest all brook trout caught and were initially only allowed to harvest brook trout from the upper reach of Quirk Creek on supervised outings. However, beginning in 2000, anglers also harvested fish from the lower reach to assess brook trout immigration and, starting in 2001, some of the more skilled anglers harvested fish on unsupervised outings. Anglers only fished from June to October, could not use bait, and were required to release all bull trout and cutthroat trout after recording the length of each fish in 5-cm size classes.

All harvested brook trout were delivered whole to the volunteer coordinator at the end of each outing for measuring (fork length, nearest 1 mm) and weighing (nearest 1 g) and then returned to the angler. Anglers on unsupervised outings recorded fork lengths (nearest 1 mm) of all brook trout caught and filled in creel cards.

Electrofishing

Removal-method estimates of the fish population in Quirk Creek were obtained by electrofishing sections of both reaches between mid-August and early September (Paul 2004). With the exception of 1987, when the mark-recapture method was used, attempts were made to capture all fish, including age-0 brook trout (< 100 mm) and cutthroat trout (< 70 mm).

To assess immigration of large (> 150 mm) brook trout into the upper reach from the lower reach, the upper 2.5 km of the lower reach was electrofished on 6 May 2000 and 2 June 2001 to capture, mark, and release 750 and 92 large (> 150 mm) brook trout, respectively. All bull trout and cutthroat trout, and marked brook trout, were measured before release. A mark-recapture estimate of the population of large brook trout present in the lower reach on 6 May 2000 was obtained by applying the Petersen estimate, corrected for size, to the marked brook trout recaptured by anglers in the lower reach in 2000.

Ageing and Maturity

Fish were aged by otoliths collected from a subsample of the fish captured by electrofishing in 1987, 1995 and 2000, and from any cutthroat trout and bull trout mortalities encountered during the study (Stelfox et. al. 2001a). Maturity was determined for all fish from which otoliths were collected.

Results

Fish Identification Education

Of 376 people who had never before taken the test, 52% failed on their first attempt. However, of those who failed their first attempt, 76% passed their second attempt, after shown the key-identifying features for each species. Mean scores on the first and second attempt were 90% and 97%, respectively. Most (≈ 75%) of the people who took the test were experienced anglers, reporting that they had fished for more than 10 years.

During the 1998–2000 periods, 54 individual anglers took the test in more than one year. Although 33% of these anglers failed the test on their very first attempt, the failure rate in subsequent years on the first attempt was only 9% and none failed their second attempt.

Angling

Average annual catch rate for brook trout in the upper reach remained high (2.2–2.5 fish/h) during the first three years of the study, but declined to 1.0 fish/h by 2002 (Table 1). In contrast, catch rates for brook trout in the lower reach changed little, ranging from 1.3 to 1.8 fish/h. Aggregate catch rates in both reaches were generally about 1.0 fish/h higher than for brook trout alone (Table 1).

Fishing effort peaked at 397 h/ha in the upper reach in 1999 and 549 h/ha in the lower reach in 2000 (Figure 1). Since then, fishing effort has been consistently higher in the lower reach, but has declined substantially in both reaches.

Table 1. Angling data summary for Quirk Creek, 1998-2003. All brook trout were harvested.

Year	Number of anglers	Number of fish caught: Bull trout	Cutthroat trout	Brook trout	Total	Number of hours fished	Catch rate (fish/h)	(brook trout/h)	Percentage of catch: Bull trout	Cutthroat trout	Brook trout
					Upper reach						
1998	97	63	349	1076	1488	436.0	3.4	2.5	4.2	23.5	72.3
1999	146	161	735	1412	2308	655.5	3.5	2.2	7.0	31.8	61.2
2000	111	68	522	1128	1718	477.3	3.6	2.4	4.0	30.4	65.7
2001	70	19	276	511	806	271.3	3.0	1.9	2.4	34.2	63.4
2002	26	1	71	83	155	82.5	1.9	1.0	0.6	45.8	53.5
2003	15	1	45	57	103	55.5	1.9	1.0	1.0	43.7	55.3
Upper Total	**465**	**313**	**1998**	**4267**	**6578**	**1978.0**	**3.3**	**2.2**	**4.8**	**30.4**	**64.9**
					Lower reach						
2000	204	115	807	1644	2566	988.8	2.6	1.7	4.5	31.4	64.1
2001	142	39	544	1101	1684	619.0	2.7	1.8	2.3	32.3	65.4
2002	119	12	287	555	854	432.5	2.0	1.3	1.4	33.6	65.0
2003	56	12	211	373	596	206.0	2.9	1.8	2.0	35.4	62.6
Lower total	**521**	**178**	**1849**	**3673**	**5700**	**2246.3**	**2.5**	**1.6**	**3.1**	**32.4**	**64.4**
Grand total	**986**	**491**	**3847**	**7940**	**12278**	**4224.3**	**2.9**	**1.9**	**4.0**	**31.3**	**64.7**

The number of hours fished on supervised outings has declined substantially since initiation of unsupervised outings in 2001 (Figure 2). This decline, in conjunction with the higher catch rates of anglers on unsupervised outings (Figure 3), has resulted in an increase in the relative importance of unsupervised outings for brook trout harvest (Figure 4). By 2003, about 2/3 of all brook trout harvested were taken by anglers on unsupervised outings.

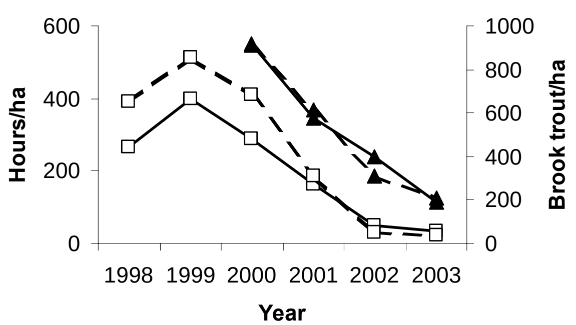

Figure 1. Fishing effort (solid lines) and brook trout harvest rates (dashed lines) in the upper (squares) and lower (triangles) reaches of Quirk Creek.

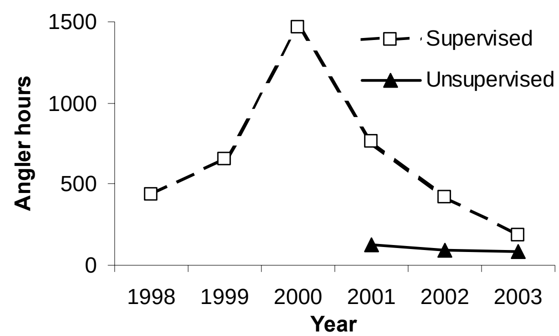

Figure 2. Number of hours fished by anglers on supervised and unsupervised outings on Quirk Creek.

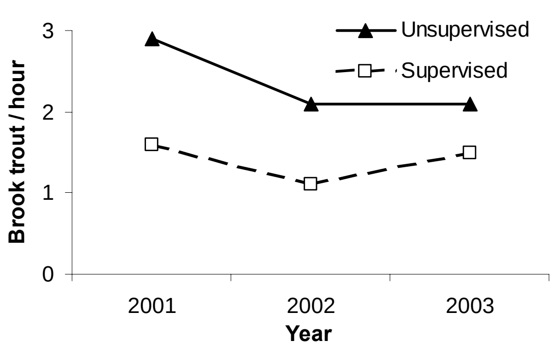

Figure 3. Brook trout catch rates for anglers on supervised and unsupervised outings on Quirk Creek.

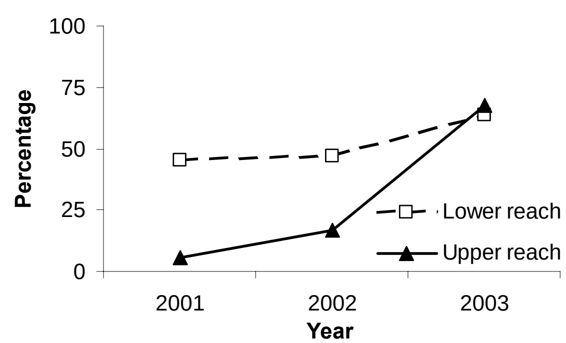

Figure 4. Percentage of brook trout harvested by anglers on unsupervised outings on Quirk Creek.

Harvest rates peaked at 913 brook trout/ha (94.4 kg/ha) in the lower reach in 2000 (Table 2). Since then, harvest rates have declined substantially in both reaches to a low of 35 brook trout/ha in the upper reach in 2003, but have been consistently greater in the lower reach (Figure 1). Mean length of harvested brook trout has changed relatively little over the study, ranging from 173 to 203 mm (Figure 5).

Table 2. Fish population estimates for, and brook trout harvested from, Quirk Creek. With the exception of the mark-recapture estimate in 1987, all population estimates were obtained by the removal method.

		Trout population estimates					Brook
					Brook		trout
Reach	Year	Total	Bull	Cutthroat	All	>150 mm	harvested
				Number per hectare			
Upper	1998	2285	64	264	1958	958	652
	1999	1652	33	167	1452	639	856
	2000	3491	39	715	2736	773	684
	2001						310
	2002	1082	[b]	73	1009	161	50
	2003	1709	79	476	1155	176	35
Lower	1987	778 [a]	50 [a]	508 [a]	219 [a]	114	
	1995	233	[b]	22	211	44	
	1996	431	[b]	28	403	197	
	1997	1456	22	361	1072	475	
	1998	2008	42	269	1697	650	
	1999	1428	[b]	175	1253	525	
	2000	2975	31	444	2500	608	913
	2001						612
	2002	1083	[b]	150	933	428	308
	2003	1217	56	775	386	139	207
				Kilograms per hectare			
Upper	1998	111.5	7.0	13.0	91.5	73.3	56.3
	1999	106.7	2.7	12.1	82.7	59.1	81.9
	2000	114.5	4.2	17.9	92.4	73.0	60.7
	2001						20.9
	2002	26.1	[b]	1.5	24.5	11.2	3.3
	2003	34.8	1.8	4.5	28.5	13.3	3.4
Lower	1987	65.6	4.2	31.7	29.7	11.6	
	1995	6.9	[b]	0.6	6.4	2.5	
	1996	20.6	[b]	2.5	18.1	13.9	
	1997	56.9	0.3	8.6	46.9	37.2	
	1998	88.9	1.9	8.6	78.3	63.1	
	1999	85.0	[b]	15.3	69.7	58.3	
	2000	98.9	3.9	17.2	77.8	60.6	94.4
	2001						41.3
	2002	46.1	[b]	8.3	37.8	27.5	23.4
	2003	25.6	1.4	9.7	14.4	12.2	18.0

[a] *Does not include age-0 fish.*
[b] *Too few bull trout were captured to obtain an estimate.*

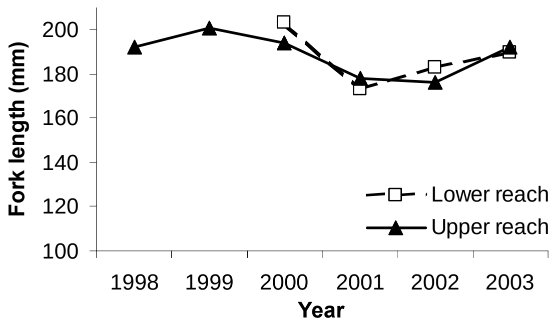

Figure 5. Mean lengths of brook trout harvested by anglers from the upper and lower reaches of Quirk Creek.

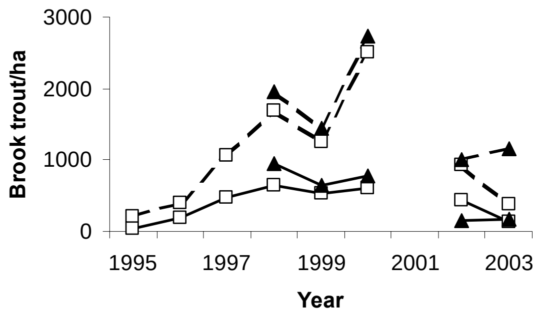

Figure 6. Densities of large (> 150 mm) brook trout (solid lines) and all brook trout (dashed lines) in the upper (triangles) and lower (squares) reaches of Quirk Creek.

Only 4% of the angler-caught brook trout in the upper reach were longer than 250 mm, compared to 32% of the bull trout and 23% of the cutthroat trout. The relationship was similar in the lower reach, where only 6% of the angler-caught brook trout were longer than 250 mm, compared to 22% of the bull trout and 25% of the cutthroat trout.

While the percentage of brook trout in the angler catch in the upper reach declined from 72% in 1998 to 54% in 2002, it remained virtually unchanged (63–65%) in the lower reach (Table 1).

Since inception of the project, anglers have harvested 7955 fish, of which only 15 (0.2%) were not brook trout. All of the misidentified fish were cutthroat trout.

Electrofishing

During the 1987–2003 period, the aggregate trout population in the lower reach declined from 778 fish/ha in 1987 to 233 fish/ha in 1995, and then increased to 2975 fish/ha in 2000 (Table 2).

In 1978, bull trout comprised 54% of the fish population in the uppermost 7 km of the creek and 8% in the lowermost 3 km (Table 3). However, the proportion of bull trout in the fish population of both reaches plummeted to only 1% by 2000. Since 1987, bull trout have not exceeded 80 fish/ha in either reach and numbers of bull trout captured have often been too low to obtain valid population estimates (Table 2).

From 1987 to 1995, cutthroat trout declined from 64% to 5% of the fish population in the lower reach (Table 3), and from 508 to 22 fish/ha, respectively (Table 2). Since then, cutthroat trout have comprised less than 25% of the fish population in the lower reach, until 2003, when they increased to 63% of the fish population and a high of 775 fish/ha. In the upper reach, density of cutthroat trout also increased in 2003, to 476 fish/ha, and the percentage of cutthroat trout in the fish population increased to 25%, up from 6% the previous year, but well below the 46% recorded in 1978 (Table 3). Although the density of cutthroat trout in the lower reach in 2003 was higher than in 1987, the biomass of cutthroat trout (9.7 kg/ha) was only about 1/3 as great as in 1987 (Table 2). Similarly, the biomass of cutthroat trout in the upper reach in 2003 (4.5 kg/ha) was much lower than in most of the previous years.

In 1978, brook trout comprised 35% of the fish population in the lowermost 3 km of the creek, and were not found in any of the four sites electrofished in the uppermost 7 km (Table 3). During the 1995–2002 period, when brook trout comprised 74–92% of the fish population in both reaches, density of large (> 150 mm) brook trout peaked at 958 fish/ha (Table 2). In 2003, the proportion of brook trout in the fish population declined to 32% in the lower reach and 70% in the upper reach — the lowest levels recorded since 1998.

The harvest of 652 and 856 brook trout/ha in the upper reach in 1998 and 1999, respectively, appeared to have very little impact on the density of large (> 150 mm) brook trout in the upper reach relative to the lower reach, which served as a control section until 2000 (Table 2; Figure 6). Subsequent to initiation of brook trout harvest in the lower reach in 2000, the density of large (> 150 mm) brook trout in the lower reach declined by 77% to 139 fish/ha in 2003 — the lowest level recorded since 1998 (Table 2; Figure 6). However, the density of large (> 150 mm) brook trout in the upper reach also declined by 77% to 176 fish/ha, even though fishing effort and brook trout harvest in the upper reach was usually less than half as great as in the lower reach during the 2000–03 period (Table 2; Figure 1).

A comparison of the length-frequency distributions of brook trout caught by angling and electrofishing in 1999 indicates that vulnerability to angling declined below about 210 mm (Figure 7). Anglers were very ineffective at catching brook trout smaller than 150 mm. Brook trout < 150 mm comprised 50–70% of the electrofishing catch, but only 3–11% of the brook trout harvest in the upper reach during the 1998–2000 period. Of 32 brook trout collected for ageing on 26 August 2000, the smallest mature female was 180 mm and none of the mature females were younger than age 3. Age-1, -2, and -3 brook trout averaged 117, 170 and 206 mm, respectively.

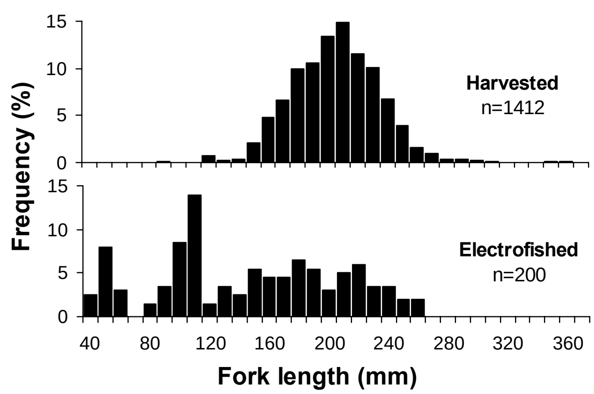

Figure 7. Size distribution of brook trout harvested in 1999 and electrofished on 16 August 1999 from the upper reach of Quirk Creek.

Of the 750 large (> 150 mm) brook trout marked in 2000, anglers subsequently harvested 391 (52%) — 349 (46%) in 2000 and 42 (6%) in 2001. Only eight (2%) of these marked fish were taken from the upper reach — four in 2000 and four in 2001. Of the 92 large brook trout marked in 2001, anglers subsequently harvested 33 (36%) in 2001. None were taken from the upper reach.

Based on recapture in the lower reach of 345 of the 750 brook trout marked on 6 May 2000, and by adjusting for growth over the course of the 2000 fishing season, we estimated that there were 2532 ± 164 (SD) large (> 150 mm) brook trout present in the lower reach on 6 May 2000, or 1407 large brook trout/ha. Using the mean weight (106 g) of the 750 brook trout that were marked on 6 May, the estimated biomass of large brook trout was 149 kg/ha.

By adding each removal-method estimate to the number of brook trout harvested prior to the electrofishing date, we extrapolated the number of large (> 150 mm) brook trout present at the start of each fishing season. Using this approach, we estimated that anglers harvested 45, 68, 57, 27 and 18% of the population of large brook trout in the upper reach in 1998, 1999, 2000, 2002 and 2003, respectively, and 64, 43 and 61% of the population of large brook trout in the lower reach in 2000, 2002 and 2003, respectively.

Table 3. Species composition of the fish population in Quirk Creek.

			Percent composition[a]		
Reach	Year	Sample size	Bull trout	Cutthroat trout	Brook trout
Upper	1978[b,c]	132	54	46	0
	1998	278	3	12	85
	1999	200	2	11	87
	2000	416	1	21	78
	2002	122	3	6	91
	2003	178	5	25	70
Lower	1978[b,d]	208	8	57	35
	1987	187	7	64	29
	1995	79	3	5	92
	1996	72	1	7	92
	1997	255	2	24	74
	1998	280	2	14	84
	1999	195	1	13	86
	2000	355	1	16	83
	2002	186	2	12	86
	2003	205	5	63	32

[a] *Determined from the number of fish in the electrofishing catch.*
[b] *Calculated from data in Volume II (Appendices) of Tripp et al. (1979).*
[c] *The section electrofished in 1978 was the uppermost 7 km of the creek.*
[d] *The section electrofished in 1978 was the lowermost 3 km of the creek.*

Discussion

The fish identification key proved to be effective in teaching anglers how to identify fish, considering that only 15 of the 7955 fish harvested by anglers participating in this project were not brook trout. Additionally, long-term retention of the key-identifying features by anglers was encouraging, given that only 9% of the anglers failed the test on their first attempt in subsequent years, even though the failure rate on their very first attempt was 33%.

Angler harvest of more than 650 brook trout/ha (55 kg/ha) from the upper reach in 1998 and 1999 appeared to have little impact on brook trout catch rates, the mean length of brook trout caught or the density of large (> 150 mm) brook trout in the upper reach relative to the lower reach. In contrast, average annual angler harvest of 25 kg/ha of trout over a 10-year period from Sagehen Creek, California, which equated to 66% of the average standing crop of trout, had a relatively large effect, given that the average total number and weight of all trout nearly doubled and the number of trout ≥ 200 mm increased 14-fold in a portion of the creek subsequently closed to angling for a six-year period (Gard and Seegrist 1972). Immigration could have reduced the effects of brook trout harvest in the upper reach of Quirk Creek. Gowan and Fausch (1996) found that movement was relatively common, with brook trout usually moving in the upstream direction during and just after runoff, and before spawning. However, in our study, upstream movement did not appear to be sufficient to mask the effects of brook trout harvest in the upper reach, since only 2% of the recaptured brook trout had immigrated into the upper reach.

The apparent lack of impact therefore suggests that angler harvest of 45–68% of large (> 150 mm) brook trout during the 1998–1999 period was insufficient to collapse the population. Although proportions harvested are based on population estimates extrapolated to the start of the fishing season, we feel these extrapolations are reasonable, given the similarity between the independent mark-recapture estimate of large (> 150 mm) brook trout present in the lower reach at the start of the 2000 fishing season (1407 fish/ha) and the extrapolated estimate (1429 fish/ha). However, our extrapolations should still be used with caution, as they do not account for all brook trout mortality or growth that occurred during the approximately two-month angling period prior to the electrofishing dates.

The population estimates suggest that a 1:100-year flood that occurred in June 1995 had a major impact on the fish population. Within five years of the flood, the aggregate population estimate for trout increased numerically by 13-fold and in biomass by 14-fold in the lower reach. Hanson and Waters (1974) documented similar effects following a flood in a Minnesota stream, with a 20-fold increase in brook trout numbers and a 6-fold increase in biomass within four years.

While densities of large (> 150 mm) brook trout have declined in both reaches since 2000, there has been surprisingly little change in the proportion of brook trout in the angler catch, although fishing effort and brook trout harvest has declined substantially in both reaches, especially in the upper reach. However, the electrofishing data suggests that a change may soon occur, given that cutthroat trout densities increased substantially in both reaches due to an influx of age-0 and age-1 cutthroat trout in 2003 (Paul 2004). A 5-fold increase in cutthroat trout density in the lower reach, in conjunction with a decline in brook trout density, resulted in cutthroat trout comprising 63% of the fish population in the lower reach in 2003, up from 12% in 2002. Although there was a larger (6.5-fold) increase in cutthroat trout density in the upper reach, the effect was diminished by a slight increase in brook trout density, resulting in cutthroat trout comprising 25% of the fish population in 2003, up from 6% in 2002.

Whether the increase in juvenile cutthroat trout density in 2003 translates into an increase in catchable-sized cutthroat trout in the future will depend on survival rates. Survival rates apparently varied greatly between the strong year-classes of cutthroat trout in 1996 and 2000 (Stelfox et al. 2001). Although age-0 cutthroat trout were absent from the 1996 electrofishing catch because sampling was conducted two weeks earlier than in 2000 (Paul 2004), survival of the 1996 year-class appears to have been relatively good based on the size distribution (Stelfox et al. 2001) and relatively high densities of cutthroat trout in the following two years. In contrast, survival of the 2000 year-class appears to have been relatively poor, since the density of cutthroat trout in 2002 was lower in both reaches than in any year since 1996.

It is possible that density of large brook trout affects the survival of cutthroat trout. Larson and Moore (1985), in a study of stream populations of brook trout and rainbow trout, found that abundance of age-0 fish of either species was greatly reduced in the presence of 300 or more adults/ha of the other species. A comparable relationship may exist between brook trout and cutthroat trout in Quirk Creek, given that relatively good survival of the 1996 cutthroat trout year-class occurred when there were less than 200 large (> 150 mm) brook trout/ha, whereas relatively poor survival of the 2000 cutthroat trout year-class occurred when there were more than 600 large brook trout/ha.

If density of large brook trout is a major factor in the survival of cutthroat trout fry, then recovery of the cutthroat trout population will be contingent upon preventing the adult brook trout population from increasing to previous high levels. However, this may be difficult to accomplish on the upper reach, where only 18% of the extrapolated population of large (> 150 mm) brook trout were harvested in 2003 compared to 61% in the more accessible lower reach, largely due to a reduction in the number of supervised outings.

Bull trout and cutthroat trout have the potential to provide a better quality fishery in Quirk Creek, based on their larger size and higher catchability. Paul et al. (2003) determined that the catchability of similar-sized bull trout and cutthroat trout was 2.5-fold greater than for brook trout. This higher catchability, however, could prevent a recovery of the native trout population. Paul et al. (2003), using a model developed with data from the Quirk Creek brook trout suppression project, calculated that bull trout and cutthroat trout populations in the upper reach would be extinct within five years at a hooking mortality rate of 10% and an angler effort of 656 angler-hours/year — equivalent to the angler effort in 1999. At hooking mortality rates of 2.5 and 5%, they could still decline.

Although the brook trout population has declined since 2000 and the cutthroat trout population increased in 2003, we cannot yet conclude that angling is an effective means of suppressing non-native trout populations, since the control section was lost when harvest began in the lower reach in 2000 to assess brook trout immigration. However, the project has demonstrated that misidentification of trout is a problem among anglers, but one that can be readily overcome by showing anglers key-identifying features for each trout species. It has also made anglers more aware of the differences between native and non-native trout. Finally, the project has demonstrated that brook trout in Quirk Creek are highly resilient to overexploitation.

Acknowledgements

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Regaining Public Trust … and Keeping It!

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ABSTRACT—"Ethics are a kind of community instinct in the making." Aldo Leopold, *A Sand County Almanac, 1949.* A definable ethic that contributes to respect, that in turn leads to trust, may well be the "mode of guidance" suggested by Leopold for effectively meeting the social dimension of natural resource decision-making of the future. The authors contend that *fair, open, and honest* are the necessary elements of our management behavior that comprise the core of an ethical approach for conducting agency programs that are increasingly under intense public scrutiny. Simply gaining the elusive public trust is not enough, however, as the public continues to respond to the authenticity exhibited by agencies and their respective professionals, maintaining credibility over time by carrying through on what our agencies say we are going to do, is critical to maintaining public trust. The authors will define how to "make sure actions on the ground match the words on the page." Drawing on their extensive public process experience, the authors contend that if natural resource agency professionals, as a community, embrace the *fair, open,* and *honest* philosophy as the cornerstone of public process, then Leopold's "mode of guidance" will have been defined for the coming century.

Introduction

"An ethic may be regarded as a mode of guidance for meeting ecological situations so new or intricate... that the path of social expediency is not discernible to the average individual... Ethics are a kind of community instinct in the making."

Aldo Leopold, A Sand County Almanac, 1949

Fifty years after Leopold (1949) penned those words, the human component of natural resource science is "so new and intricate" that the path of social expediency is, indeed, "not discernible."

As biologists, foresters, and environmental educators, we have become more than sources of information and data. We've also become professional facilitators embroiled in high stakes, natural resource issues and decisions. We've seen everything from wildly successful public and agency partnerships to dismal failures where litigation seems to be the only solution. We've pondered, time and time again, why some public interactions succeed and others fail; why some proposals move forward and others go to court.

We've analyzed various public involvement models, techniques and processes, such as focus groups, comment periods, public meetings, even charettes. Employing different models or processes doesn't seem to make a significant difference; effective, positive interactions are possible regardless of the model used. We've come to the conclusion that success is not model-dependent; the question then remains as to what factors make or break a public and resource interaction.

Our success is dependent on **processes** that bring together **people** and **information** in a way that promotes, encourages, and supports interactions based on trust. We contend that ethical principles are the framework for establishing trust. These ethics are the driving force for successful collaboration among diverse internal and external publics. They are also the driving force in how information is gathered and shared. In this paper, we advance the premise that fair, open and honest are fundamental principles that comprise the ethics required for successful resource decisions. Fair, open, and honest—the basic components of ethical behavior…that establish credibility... that can lead to trust.

We suggest that, as Leopold stated, a definable ethic is our "mode of guidance" for natural resource decision-making of the future, and second, that a fair, open, and honest ethic is that mode. Further, we contend that this ethic is an action-oriented component, not one in which we simply reflect upon past actions, but one that we use every day to make critical natural resource decisions.

Even today authors continue to support the notion that ethics are key to successful leadership...and that's what we are talking about here: Providing leadership based in ethics. Authors Kouzes and Posner surveyed thousands of businesses and government executives over the course of two decades, asking the question: What values (personal traits or characteristics) do you look for and admire in your leader? They received over 225 different traits and characteristics. After a series of analysis, the list was reduced to 20 characteristics with synonyms for clarification, onto a questionnaire that was then distributed to over seventy-five thousand people around the globe. The results? Consistently over time and across continents, honesty ranks the highest, emerging as the single most important characteristic of a leader. Attributes of integrity and character, were consistently among the top rated. Constituents, whether internal or external want their leaders to be ethical. They expect to be included in processes that recognize and honor the diversity of their contributions. We all want to make progress in our management efforts and decision-making. We can only do that by practicing these fundamental principles in our interactions.

Let's be clear. If your process is not fair, open, and honest, it will not succeed. If you are not ethical how can you sustain credibility and trust among your constituents? It's not that we are purposely or fundamentally unethical. Our science is intense, dynamic, and complex. Practicing ethical behavior means paying close attention to all aspects of what we are doing. Ethics provides the compass that guides our actions through some of our toughest interactions and management decision.

Principle Ethics in Building Trust

Fair

Being fair means several things. For example:

- Providing realistic opportunities for people to participate.

This means providing times and locations that meets the needs of your diverse audiences. We might have to acknowledge that sometimes, the high school playoffs are more important than your public meeting.

- Providing everyone the same information at the same time.
- Providing a safe physical and intellectual environment for the exchange of ideas.

- Making sure the people who are affected by your group's decisions, help make those decisions.

Does everyone have the same opportunity to reflect and respond?

A few self-directed questions are the litmus test for this component of our ethic. How fair is it for biologists to spend two years in obscurity writing a species recovery plan, then say to the public, "You have thirty days to review and comment on this 3-lb document, and, by the way, the clock started ticking last Thursday when the notice was printed in the Federal Register?"

Does everyone get the information and do they get it at the same time?

How fair is it when we provide information to some and not to others? When the others suddenly "find out" what's going on, agency credibility is in jeopardy. Everyone who cares about the issue needs to be involved in the process, not just the supporters or the locals.

For example, one Resource Manager had his predator control program suddenly "blow up" when animal rights advocates found out about it at the very end of the public comment period. When asked why he didn't let national groups know of the process sooner, the answer was, "Well, everyone around here knew about it and thought it was OK."

We need to ask ourselves a fundamental question, “Would we consider this fair if this happened to us?”

Open

The conditions for being open include:

- The process is understandable
- All input is welcome (*really welcome*)
- All pertinent information is shared

The essence of open is the questions:

Are you *really* listening?

Supreme Court Judge Stephen Breyer in his confirmation hearing responded to the question, "What is the role of the Supreme Court?" He stated eloquently, "To listen...listening gives dignity to the person being listened to."

In many ways our actions regarding public process have actually trained our constituents to be skeptical of our public involvement strategies. They have become wary of agency “input opportunities” as agencies routinely seek input from the public when a decision has essentially already been made.

Is your process designed to receive information from a diverse audience?

In most cases, natural resource professionals represent public agencies. The public has a fundamental right to provide input on issues that affect them. We need to give them a variety of ways to talk to us-public forums, solicited and unsolicited surveys and assessments, letters, phone calls, whatever is the outreach mode of the moment ...and then we need to *really* listen to their comments and factor them into public decision processes.

Is there a process for dealing with the information?

We often tend to seek validation or acceptance of our plan or strategy, rather than seeking legitimate public input within a truly collaborative process. Margaret Wheatley in her book *Leadership and the New Science* says, “No one is successful if they merely present a plan in finished form to others. It doesn’t matter how brilliant or correct the plan is – it simply doesn’t work to sign on when they haven’t been involved in the plan.”

Honest

Honesty, the heart of integrity and subsequently a key element of personal and agency credibility, is a step along the path to that elusive trust we seek as Agencies. We are responsible for processes that bring people and information together in a way that’s clear.

The conditions for being honest are:

- Letting people know what you can and can not do
- Sharing what kind of information and science we have . . . and how good it is!
- Explaining how information and science will be applied to the problem
- The timeline … some things take awhile, don’t be afraid to say so
- How we will use their input

Fundamental questions we should ask ourselves are:

Is all the information on the table? Is the information understandable? Clear to everyone, not just scientists? (would your neighbor understand it?) Have we shared the alternative and consequences?

At Stake: Credibility

The characteristics of fair, open and honest often overlap as this example illustrates. In one painfully memorable public meeting, the author asked the Assistant Director of the agency, five minutes before the meeting began, "What do we tell them about how their input will be used?" The Assistant Director shrugged and replied, "It doesn't matter… we cut a deal with all the key players at three o'clock yesterday." The public input meeting was held anyway, but had they known the truth, how would those 38 participants have felt about the fairness, openness, and honesty of that public agency and its process? How much dignity was afforded to that audience on that day? More than that - why is it considered acceptable to treat our constituents in that manner?

The examples shared above illustrate a breach of agency credibility. Credibility is at stake when there is a disconnect between our words and actions. It is not enough to espouse to these principles as important: we must give voice to our commitment to them and then set the example with our actions. It is only through consistent words and actions that we are seen as authentic and thus credible, in our management efforts. When our actions do not match our words, our future words become suspect and labeled insincere, ineffective, untrustworthy, or untruthful. When we are consistent in our words and actions people are willing to engage with us in future ventures. They say things like, “I may not agree with the action, but I was treated respectfully”, or “they practice what they preach”, “it was a tough decision, but at least they were fair about it”.

We all know how quickly news about our interactions travels throughout our networks. Margaret Wheatley says, “The capacity of a network to communicate with itself is truly awe inspiring; its transmission capability far surpasses any other mode of communication. But a living network will transmit only *what it decides is meaningful*". We want that "meaningful information" traveling through our agency networks to be that we are fair, open and honest.

If we have no credibility, how can we ever hope to regain the public trust? When our words and actions don't match up… when we are not authentic … people become less willing to engage in any future productive interactions. After all, in the absence of trust everything we do is perceived manipulation.

We contend that our personal and agency credibility, are on the line every time we interact with our constituents. We simply cannot afford to be unethical in our actions.

Trust … A Two-Way Street

You've often heard trust described as a two-way street, something that has to be mutual. The public doesn't trust government these days for a lot of very valid reasons! By the same token, Agencies often don't trust the public ... again, for a lot of very valid reasons.

Air Force General Chuck Horner, General Schwarzkopf's Deputy Commander in the Gulf War, had some interesting comments about one of our mutual "publics" - the media! When asked, "why the military had such a distrust of the media?" He could have been speaking for natural resource agencies as well when he responded:

Fear of the media seems to go with the job description of soldier, sailor, or airman [we can easily include biologist]. Why? God only knows. When you think about it, if you can trust the press and the TV commentator to tell the truth, and I do, then it's not the media we fear but the American people ... a sad commentary on our military mid-set.

Sometimes you...we...all of us do asinine things. If you are doing something stupid, pursuing a poor policy, or wasting taxpayers' dollars, and the press or television paints you in an embarrassing light that is probably a good thing. In the long run, the exposure, no matter how painful, is good for the military and the nation. If, on the other hand, you are getting the job done skillfully, pursuing a noble cause, or managing a military operation with efficiency (how rare that is!), then you have much to gain from media exposure. The American people are quite capable of judging good and bad for themselves. I guess the bottom line is we have little to fear if we trust the judgment of the folks who pay the bills.

Individually or as agencies, we may or may not trust our many and varied publics but we're pretty sure these days it's safe to say, the public doesn't trust us! This mistrust is borne not from an intentional, faulty process, or procedure, but often of actions that have inadvertently been exhibited by agencies and individuals that have preceded us (myself included). If we are perceived by our publics as being unethical we can only dispel that perception by being, from this point on - fair, open and honest.

However, in discussing this topic with colleagues, we often hear the complaint, "why should we be ethical in dealing with the public - they aren't dealing ethically with us!" Our response is simply "who's the professional here? Who should be the first to break the cycle of mistrust ... in order to craft a *new* cycle of trust?"

We need public support more than ever to do our jobs, yet in many cases, the public doesn't trust us as partners and are suspicious of our motives. This suspicion destroys our credibility and erodes our capability to manage our natural resources.

It is clear then, that we need a new approach in natural resource decision-making, one based on mutual trust between the public and public agencies. We advocate a new approach that learns from the past, recognizes the complexities of our current social and biological interactions, and applies a fundamentally ethical approach to managing our natural resources in the future.

Conclusion

We must begin engaging people in a process that is fair, open, and honest. This means

- we include everyone in a proactive process
- we are sincerely listening
- we honor the diversity of ideas
- we engage in truthful dialogue.

Only then, can we can be credible in the eyes of our publics and begin to regain the trust critical to the health and sustainability of our natural resources.

It only takes one person to make a difference. Several years ago, we witnessed one courageous agency individual take a stand when these ethical principles were breached in one small community. The Fish and Wildlife Agency was in the middle of its angling regulation process. There was a proposed change in possession limit that would have severely affected the recreation, and associated business, in a small rural community. The agency had scheduled routine public meetings in the same large towns they always held them in, and advertised in the same publications they used every year.

However, one agency staff member, working in the office serving this community, realized the local folks had not been informed of the proposed regulation change! He took it upon himself to organize an agency public meeting in the community by quickly faxing information about the change and the meeting to community businesses, newspapers, and the radio. He pulled together biologists to plan and conduct a meeting that would provide a forum for sharing information and for hearing community members concerns. As you would imagine the public outcry was swift and loud: “Trying to hide something? Too little, too late? Our input doesn’t matter? You don’t care about us?” Yes, the agency had some explaining to do, but they could (and did). At least this state agency had taken the first step toward handling an issue in an ethical manner!

So what’s next?

We are not expecting you to keep a three-ring binder full of process, procedure, and policy statements in your head to guide your every natural resource decision. What we are saying is that there are simply three fundamental principles that can be tested with some simple questions: Is what we are doing fair, open, and honest? Is what we are doing perceived by others as fair, open, and honest?

These are the questions that will keep you grounded in ethical natural resource management. Will they save us when the issues get hot? Who knows? We only know what happens when we aren’t ethical in our actions. – our credibility, and therefore, our trust is destroyed. Ethics is a choice we make and a trust we keep with those around us.

General Chuck Horner has one other telling point regarding that elusive quality we call trust. He says simply*, "trust takes time, but when you have it you have a wonderful gift."*

This is one gift we can give to ourselves … we should make it so.

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Long-term Results of Mitigating Stream Acidification Using Limestone Sand in St. Mary's River, Virginia

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ABSTRACT—The Virginia Department of Game and Inland Fisheries has been studying the impacts of acid deposition on the biota of the St. Mary's River in Augusta County, Virginia for the past 30 years. During the period preceding 1999, invertebrate diversity decreased by over 50% and the number of fish species dropped from 12 to 4 with only native brook trout still present in significant numbers. From 1994 through 1997, the stream experienced reproductive failure of brook trout for three of the four years. The Department, along with the U.S. Forest Service who administers the area, agreed that the cause of the loss of aquatic life was atmospheric acid deposition and that water quality manipulation was needed to protect the remaining aquatic species as well as restore species that had been extirpated. In a project designed by James Madison University (JMU), limestone sand treatment was proposed for mitigation of the acidity. After much environmental analysis (EA), public debate, and careful consideration, the project was approved and implemented in March of 1999 with 140 tons of limestone sand introduced to six stream locations within the drainage using a helicopter. Improvements to water quality occurred immediately, aquatic invertebrate response was noted within three months and upstream recolonization of some fish species was observed within six months. Water chemistry data have been collected and analyzed quarterly by JMU from 22 sampling locations within the wilderness and weekly samples have been collected at the wilderness boundary. In addition, aquatic invertebrate and fish populations have been surveyed annually. The pH, ANC, calcium concentrations, and calcium/hydronium ratios have all increased as a result of the limestone treatment and have remained at acceptable levels during the 5-year study period. Aquatic invertebrate diversity recovered to levels not seen in 30 years and brook trout numbers initially exploded then settled to levels about 50% higher than long-term pre-treatment averages. In September 2003, Hurricane Isabel dumped up to 51 cm of rain in the drainage and significantly disturbed stream channels and riparian vegetation. Despite the catastrophic flood event, the limestone beds remained intact and continued to provide suitable water quality. The study clearly demonstrates that this treatment method can provide long-term benefits to aquatic resources.

Introduction

Acid deposition has been impacting aquatic resources in the mid-Atlantic and southeastern United States for at least the past two decades (Herlihy, et al., 1993; Webb, et al., 1994). The pH of pre-industrial precipitation in Virginia has been estimated to be in the 5.3 to 5.6 range (Webb, 1987) while recent readings in the Shenandoah National Park averaged 4.4 (U.S. EPA, 1998). This represents a tenfold increase in precipitation acidity since the beginning of the 20th century.

Acid deposition is not necessarily harmful to aquatic life. A watershed's ability to buffer acid deposition determines whether the system suffers long-term biological degradation. In western Virginia, most of the larger stream systems are well buffered due to underlying limestone geology, but most of the wild trout resource occurs on mountain slopes composed of sandstone, quartzite and shale. These slopes provide limited buffering capacity and are subject to acidification. In 1987, a synoptic survey of water quality parameters in 350 of Virginia's 450 wild trout streams was funded by the Department of Game and Inland Fisheries. The result of that investigation indicated that 78% of the sampled waters had ANC (acid neutralizing capacity) of less than 100 ueq/L, meaning they were sensitive to acidification. Of these acid sensitive streams, 11% were already acidified (ANC< 0). One of these acidified streams was the St. Marys River, once considered one of the state's premier wild trout fisheries.

Study Area

St. Marys River is a third order coldwater stream that drains the west slope of the central Blue Ridge Mountains in southeastern Augusta County, Virginia. Its 27 km^2 watershed is the centerpiece of the 4000 hectare St. Mary's River Wilderness Area. St. Mary's River originates at 951 m above sea level and descends at a gradient of 39 m/km to its confluence with Spy Run, 11.4 km downstream. The stream is very scenic with numerous falls, cascades, large boulders and deep clear pools. The watershed includes five major tributaries. St. Mary River's low ANC levels can be traced to the geologic formations that underlie the upper watershed. Antietam quartzite is the primary rock formation while formations of Hampton quartzite underlie the upper watersheds of Sugartree Branch, Mine Bank Creek, Bear Branch, Chimney Branch, and lower reaches of St. Mary's (Werner, 1966). Both formations are known to have low solubility, thus providing few base cations and carbonate to neutralize acidic input (Downey, 1994).

The St. Mary's River has long been recognized as one of Virginia's premier wild trout fisheries. In 1935 (Surber, 1951), it was reported to support a good population of wild rainbow trout. By 1948, the lower portions of the stream began receiving stocked trout as part of the federal/state effort to expand trout fishing opportunity. The floods of 1969 and 1972 eliminated access for stocking and the stream reverted to wild trout management. At that time, St. Mary's River was one of the few streams in the state that contained reproducing populations of brook, brown and rainbow trout. It became one of the state's earliest special regulation streams when the Department so designated it in 1974 after study and recommendations by Trout Unlimited. The drainage was later proposed as a federally designated wilderness and in 1984 became one of Virginia's first wilderness areas. The primary feature of the area that drew support for wilderness designation was the wild trout fishery and the scenic qualities of the St. Mary's River.

Biological Surveys

Surber (1951) provided the earliest data on biological communities in the St. Mary's River. He collected detailed aquatic macro-invertebrate data from a number of sites in both 1936 and 1937. These data provide a valuable baseline which precedes likely impacts due to industrial based acidification. The Department of Game and Inland Fisheries collected extensive fisheries and invertebrate data as part of a statewide trout stream inventory in 1976 (Mohn and Bugas, 1980). With the designation of St. Mary's River as an acidified trout stream by Webb (1987), the Department began a program of intensive fisheries and invertebrate data collection on a biennial basis from 1986 through 1998. Since the liming operation in 1999, fish and invertebrate data have been collected annually.

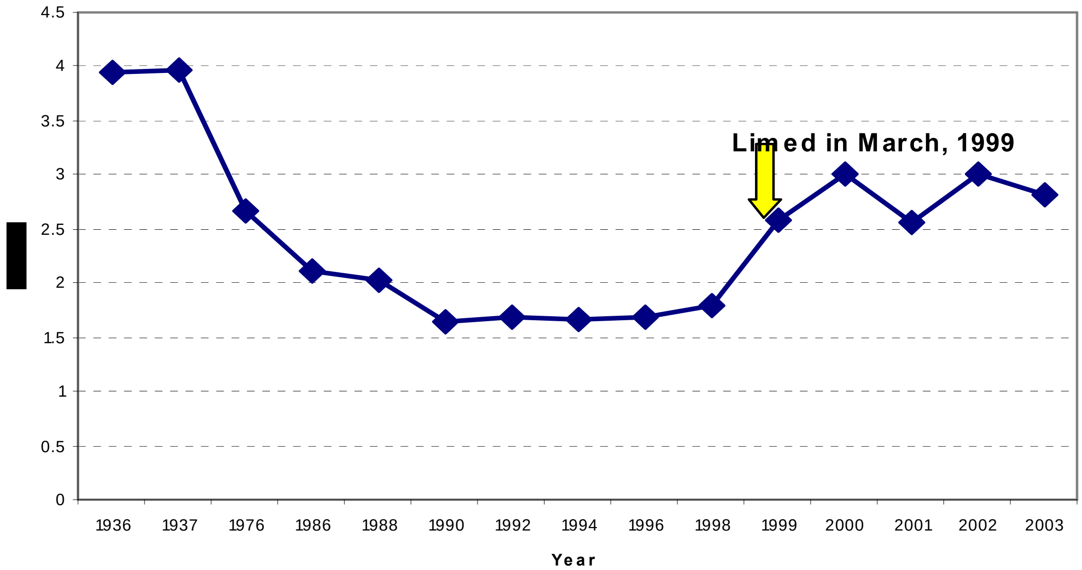
The 1976 survey by the Department of Game and Inland Fisheries provided the first recorded fisheries survey of the St. Mary's River. Six sample stations were established on the mainstem. These stations were established at approximately equal intervals along the mainstem from the lower wilderness boundary to the headwaters (Mohn, et al., 2000). Stations varied in length from 76 to 171 m and included at least three riffle, pool, and run sequences. Block nets were placed at each end of the sample stations and three-run depletions were used to estimate fish abundance and biomass. In addition, a Carle sampler (Carle, 1976) was used to collect three 0.26 m^2 invertebrate samples from riffle areas at each site. This collection technique and the sample locations compared favorably with methods used by Surber in 1936/37.

Fourteen species of fish have been collected from the St. Mary's River since 1976 but several are considered transient. The most species collected in any one-survey year was 12 in 1976. During the pre-treatment survey period 1976 – 1998, the number of fish species steadily declined from 12 to 4. In addition, several species, which were found throughout large portions of the drainage in 1976, such as blacknose dace, fantail darter, and mottled sculpin, had their ranges and numbers severely reduced. Rainbow trout, for which the St. Mary's River was best known, were extirpated from the drainage by 1994. Due to its greater acid tolerance, the native brook trout remained abundant through 1994. However, the 1996 survey indicated year class failures in two of the previous three years and a sharp drop in brook trout population numbers. The magnitude of this drop in population prompted the Department to immediately begin discussions with the USFS on acid mitigation.

The aquatic invertebrate data have shown a more gradual but no less significant reduction in both species numbers and diversity (Kauffman, et. al, 1999). Many genera of stonefly, mayfly and caddisfly were extirpated from the drainage by the mid-1980s while populations of acidophobic taxa such as the Plecoptera, *Leuctra/Alloperla* and Chironomidae showed significant increases. The invertebrate diversity as measured by the Shannon Diversity Index showed a significant decline throughout the pre-treatment study period (Figure 1).

Figure 1

Diversity (Shannon) Index



Acid Mitigation Methodology

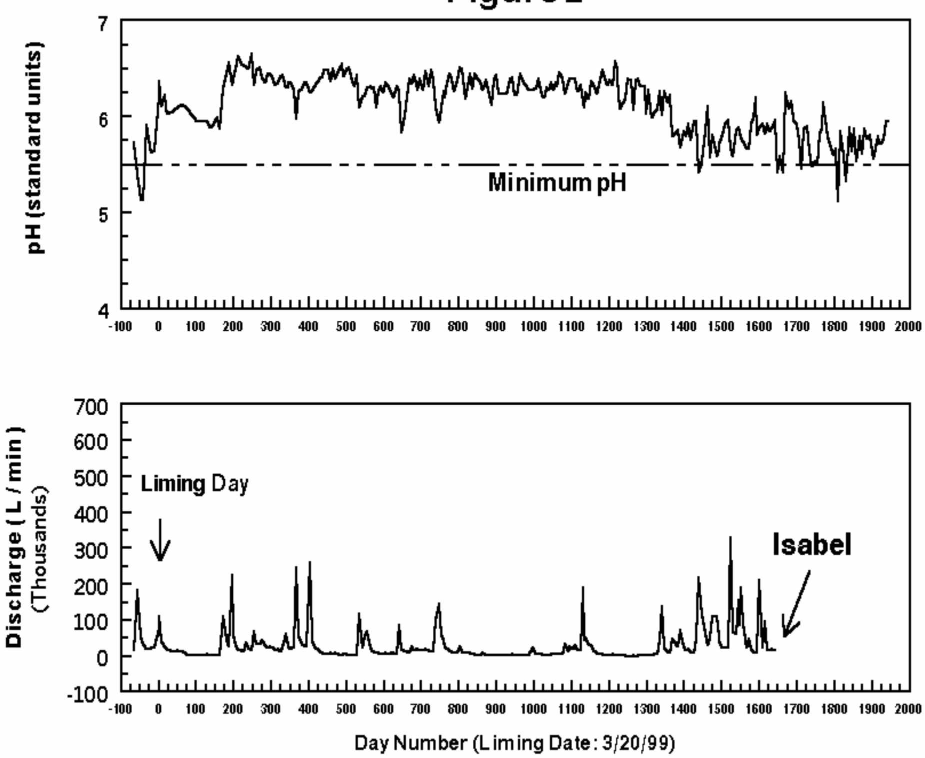
The USFS, Chemistry Department at James Madison University and Virginia Department of Game and Inland Fisheries have developed a low cost methodology for treating stream acidification using limestone sand introduced directly into the stream (Downey, et al., 1994, Hudy, et al., 2000). This methodology was utilized on March, 1999 when 140 tons of limestone were placed at six sites within the St. Mary's River Wilderness Area using a helicopter (Mohn, et al., 2000). It was estimated that this treatment would effectively mitigate the impacts of acid deposition for a period of five years. Although the use of limestone sand has become a commonly used treatment method in this region of the country, the St. Mary's River project was unique in that it would occur within a federally designated wilderness area. In this instance, there are not only biological and chemical aspects to limestone mitigation, but social, political, economic, and legal aspects as well. The process for dealing with the issues and concerns of treating a wilderness stream are described in Mohn, et al. (2000).

The limestone sand mitigation method is based on placing enough limestone to treat the receiving water for a specified period. In this study, it was estimated that the treatment would be effective for a period of five years. This calculation is based on the consumption rate of the limestone at the average annual rainfall for the drainage. Flow rates for this study period were far from normal. At time of treatment, the area was in the first year of a severe 4-year drought. That period was followed in 2002 with one of the wettest years on record and finished in the fall of 2003 with one of most devastating floods on record. In September 2003, the St. Mary's River drainage took a direct hit from Hurricane Isabel. Rain gauges at the head of the drainage recorded as much as 51 cm of rain within a 18 hour period, far more than fell anywhere else in the storm's path. This discharge resulted in major streambed alteration including establishment of new channels and severe downcutting of the channel bed. This event caused concern for the continued stability and function of the limestone sand beds.

Water Quality Response

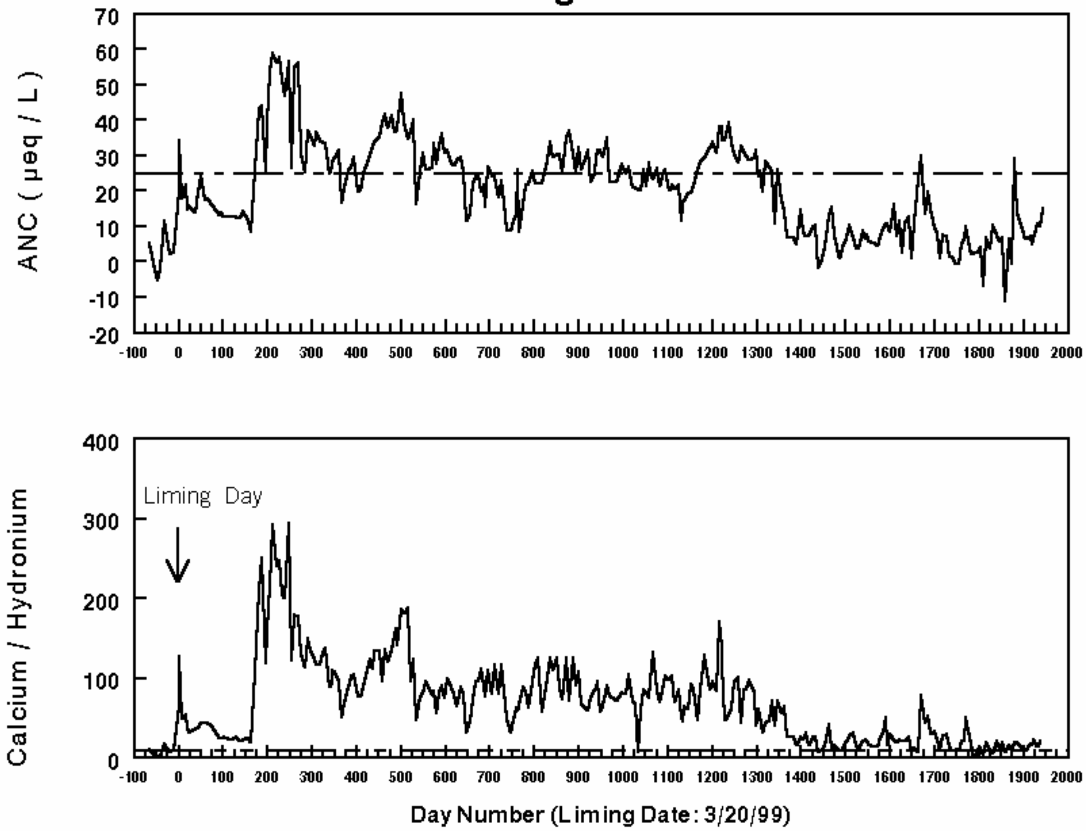
Water chemistry monitoring of the St. Mary's River began in January 1999, three months before the date of the liming treatment. A sampling site was located at the lower boundary where the stream exits the Wilderness Area. A staff gauge was installed here for recording stream discharge on sampling days. Samples have been collected no less frequently than once a week since the date of liming. The top graph in Figure 2 provides the observed pH for the 67 months since the project started. The data points are connected for clarity. A value of pH 5.5 was chosen as a minimum for protection of certain aquatic insects and fishes that were native to the St. Mary's drainage. Figure 2 reveals that the pH values were often less than the minimum acceptable value at the sampling site prior to the introduction of limestone. The average value for this period was pH 5.53 $\pm$ 0.26. In the 64 months that have elapsed since the liming, the average has been pH 6.14 $\pm$ 0.30. The bottom graph in Figure 2 shows the peaks and valleys in measured discharge that accompanied wet and dry periods. The graph ends on Day 1645 when the flood after Hurricane Isabel destroyed the gauging site. Storm events generally caused short-term decreases in pH as shown by the graphs, but even the decreases were significantly mitigated compared to the pre-liming conditions. The years 2002, 2003 and 2004 have been wet with above average discharge and it is evident from the data that pH has dropped during that time period. It is interesting to note that pH remained stable after the Hurricane Isabel flood, indicating that the limestone sand beds are effect even under catastrophic conditions. The pH drop, however, does signal a need for reliming.

Figure 2



Another water quality parameter of interest is the acid neutralizing capacity (ANC). Figure 3 provides the weekly ANC data on the top graph. The bottom graph in Figure 3 shows the calculated parameter of calcium to hydronium ion (Ca/H) ratio versus time. These are included in the same figure because both parameters are important for assessing the impact of acidity on aquatic life. The ANC values were quite low for the St. Mary's River prior to liming, often showing negative values. The pre-liming ANC average was 2.1 $\pm$ 5.0 µeq/L. The low values are the result of a lack of carbonate bearing mineral in the Antietam formation of quartzite rock that makes up most of the St. Mary's wilderness watershed. Thus little natural buffer is available to mitigate acidic inputs. The post-liming ANC values have increased due to the slow dissolution of the introduced limestone sand to an average 21.3 $\pm$ 12.7 µeq/L.. Recently the ANC values have fallen below the target also indicating that reliming will be necessary soon.

Figure 3

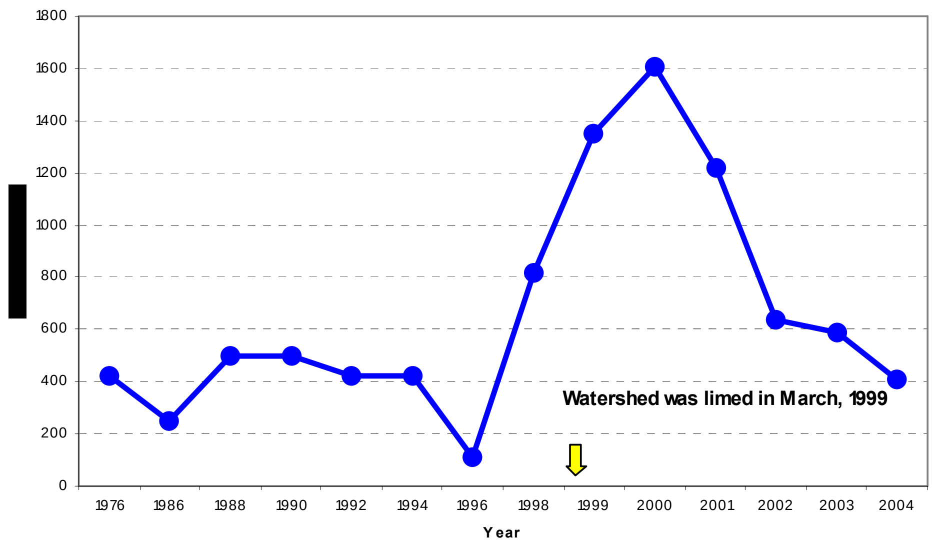


Biological Response

Post treatment trout biomass and number estimates show a dramatic response (Figure 4). However, all of this response cannot be attributed to the limestone treatment as populations began recovery in 1998. Virginia experienced a prolonged drought period that resulted in stable, low flow, mild winters from 1997 through early 2001. These conditions generally produce exceptional year-classes of brook trout. In the case of St. Mary's River and other acidified streams, the low flows not only produced good flow conditions for reproduction and recruitment but the lack of significant rainfall resulted in winter pH values higher than normal. However, brook trout numbers leveled off in 2002 and 2003, both high low years, at about 600/km which is about 50% higher than the pre-treatment average. The sharp decline in 2004 is attributed to the severe impact of Hurricane Isabel.

Figure 4

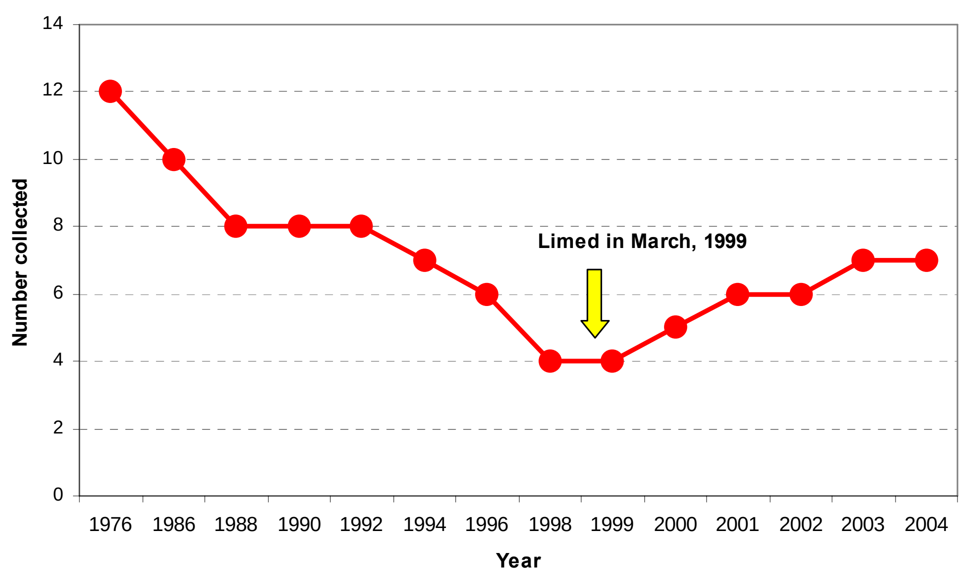
St. Marys Brook Trout Density



Non-game fish species have also shown a recovery. Prior to treatment, St. Mary's contained only 4 species of fish with only brook trout present in significant numbers. The number of species has now increased to 7 (Figure 5) with most species now present in good numbers at lower sampling sites.

Figure 5

St. Marys River - Fish Species Collected



The aquatic invertebrate populations, as measured by the Shannon diversity index, has been our most reliable indicator of stream decline over the history of our studies of the St. Mary's River. It is interesting to note that the index rebounded to 1976 levels within only 3 months of treatment (Figure 1) and has remained fairly consistence throughout the study period.

Conclusion

The use of limestone sand has proven to be an effective and cost efficient method of treating stream acidification. Stream discharge has varied significantly during the study period, yet the treatment remained effective at mitigating acidification. Despite a catastrophic flow event late in the study, the limestone sand beds remained intact and continued to be effective. The original methodology (Mohn, et.al., 2000) used to estimate the quantity of limestone needed to cover a minimum five-year treatment period appears to have been appropriate. The data indicate the need to add additional lime in the near future.

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Wild Trout in an English Chalk Stream: Modeling Habitat Juxtaposition as an Aid to Watershed Rehabilitation

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ABSTRACT—Wild brown trout (*Salmo trutta*, Linnaeus, 1758) populations in southern England are subject to both habitat degradation and overstocking, even in the internationally famous streams where dry fly-fishing began. Habitat rehabilitation within such degraded watersheds can be improved by better understanding the integration of habitat and ecological processes operating simultaneously at a range of scales. We quantify the influence of local meso-habitat juxtaposition upon wild brown trout population dynamics in two contrasting sectors of the River Piddle, Dorset, UK. Sectors examined represent 'typical' semi-natural chalk-stream conditions in the Piddle/Frome Watershed. PHABSIM was used to model meso-scale habitat composition (WUA) and habitat durations, which were tested for correlation against age-specific trout densities, obtained from eight years quantitative electrofishing data. Analyses indicate; **(1)** availability and location of marginal meso-habitats with abundant cover is critical to adult over-winter survival and **(2)** appropriate juxtaposition of spawning and rearing meso-habitats strongly influence juvenile brown trout recruitment. In the light of these data we examined the potential for integrating meso-habitat juxtaposition into initial design stages of river rehabilitation schemes. We argue that such an approach should form an integral component of watershed restoration strategies as it offers effective manipulation of natural mechanisms regulating brown trout populations at a multi-scalar level.

Introduction

The brown trout (*Salmo trutta* Linnaeus 1758.) is a polymorphic species indigenous to British rivers but there is considerable evidence of widespread and on-going decline in the status of wild stocks (Giles, 1989; Crisp, 1989). Anthropogenic influences destructive to river channel structure and ecosystem function cause widespread and severe loss of salmonid habitats (Crisp, 1989; White, 2002). In the UK, rivers have been so modified and engineered for purposes such as flood defence, land drainage and navigation that few can be regarded as in a "pristine" condition (Brookes and Shields, 1996). In recent decades, increasing development of floodplains has increased the need for river engineering to improve flood defence. Population growth, particularly in the south of England, has increased pressure on groundwater resources. In addition, on-going degradation of logic environments is largely due to agricultural land use practises associated with intensification under the EU Common Agricultural Policy (CAP). Large scale dredging programmes in the 1950's and 1960's aided wetland drainage in order to bring fertile floodplain land into production. Soil erosion from ploughed arable land increases sediment supply to rivers and exacerbates problems of eutrophication caused by intensive application of fertilisers. Destruction of river banks by sheep and dairy cattle in over-grazed riparian zones has become a major source of habitat loss everywhere from southern chalk streams to Scottish mountain burns. The recent "Salmon and Freshwater Fisheries Review" (Warren, 2000) recognised the need to place habitat enhancement at the core of wild stock conservation and recommended urgent research into factors affecting long-term sustainability and effectiveness of habitat restoration as a fisheries management tool. In this context the present study addresses the response of a wild brown trout population to temporal and spatial variation in stream habitat and, in particular the influence of local meso-habitat juxtaposition on population structure in a small chalk-stream.

The Study Area

The River Piddle is a third order stream draining a catchment of Upper Cretaceous Chalk approximately 183 km^2 in area and flows approximately 40 km south and east to form a common estuary with the River Frome, before discharging into the English Channel via Poole Harbour. Land use on the floodplain is predominantly permanent pasture and arable land. The Piddle is a typical "chalk stream" characterised by low mean gradient (2.18m/km) and base-rich alkaline waters ($CaCO3 > 200$ mg/l). Groundwater rises at a relatively constant 9-10^{o}C throughout the year maintaining stable seasonal and diel temperature regimes. The buffering effect of the aquifer produces a stable flow regime with an absence of extreme low flows and sudden spates. Winter high flows rarely exceed bankfull stage and summer base flows are maintained by groundwater (Mann et al.,1989). Long-term mean monthly flows at Tolpuddle (1965-2000) range from 0.18 m^3/s in August/September to 2.4 m^3/s in February. Median flow (Q50) over the period is 0.54 m^3/s. Primary production is dominated by large aquatic macrophytes, mainly *Ranunculus* spp. which supports high macroinvertebrate productivity forming the basis of trout diet (Maitland and Campbell, 1992). Dominant fish species are resident and anadromous brown trout and Atlantic salmon, with minnow, bullhead, stone loach, pike, and eel common (Strevens, 1999).

The morphology of low gradient chalk streams tends to produce more habitat features, such as undercut banks, trench pools and low width-depth ratios, in comparison to moderate gradient reaches, and these features are positively correlated with a high mean standing stock of trout (Kozel et al., 1989). However, the Piddle has suffered many problems common to intensively farmed lowland catchments. Physical habitat degradation from overgrazing resulting in loss of riparian vegetation has led to widespread channel over-widening and increased sedimentation. Habitat diversity has been lost due to historical anthropogenic manipulations particularly associated with milling, irrigation and land drainage. Run-off from agricultural land has caused siltation problems detrimental to salmonid spawning (Crisp, 1989) and elevated nitrate levels have resulted in widespread algal colonization of substrates. The catchment is heavily abstracted for a variety of water uses which has exacerbated low flow problems since the mid-1980s. This has had significant ecological impacts including severe reduction in juvenile trout habitats over a 10-km length of the middle river (Strevens, 1999).

A programme of physical restoration was initiated at Tolpuddle in 1994 primarily to restore channel diversity and improve spawning habitat and refugia for larger wild trout (Summers et al., 1996; Summers et al., 1997). Fencing and substrate re-distribution using current deflectors and weirs were the most commonly used techniques (Langford et al., 2001). The fishery has been managed for over twenty years on a "catch and release" basis and retains a significant self-sustaining population of native resident and anadromous brown trout, which is not subject to angler harvest, or stocking of hatchery trout. Pike (*Esox lucius*) which were present prior to 1993 were removed and subsequently controlled to alleviate the effects of piscivorous predation and physical habitat was assumed to be the most important population-limiting factor. The study site comprised two main stem river sectors approximately 2 km apart and 0.5 km in length both divided into 4 electrofishing sections. The trout population was monitored annually by electrofishing in early autumn over the period 1993-2001.

Methods

Two representative reaches of contrasting habitat characteristics and population dynamics were selected for both sectors for application of the Physical Habitat Simulation Model (PHABSIM). Both reaches were coincident with electrofishing sections. Sector 1 was of higher habitat diversity flowing through fenced open pasture with a trout population dominated by adults. Sector 2 was more uniform, partly over-shaded by riparian trees and consisted principally of age 0+ and 1+ trout.

Hydraulically linked transects were used to characterize hydraulic and physical habitat attributes of each study reach in accordance with PHABSIM requirements (Bovee et.al., 1998). Transects were placed to represent mesohabitat types present in approximate proportion to the contribution of each habitat type to the total make-up of the river sector. Approximate cell boundaries were determined from habitat mapping and located at intervals ranging from 5 – 23m depending on microhabitat complexity. All mesohabitats present were represented by at least one transect to accurately represent habitat availability and continuity thus ensuring that habitat juxtaposition was accurately sampled. Field measurements of depth (cm), and mean column velocity (m/s) at 0.6 depth at each transect were taken for a minimum of three discharges over one hydrological cycle as outlined by Bovee et al. Substrate and cover were measured twice (summer and winter) at corresponding intervals between 0.3 – 0.6 m along transects to define a series of cells around measurement points. Hydraulic models were calibrated using standard procedures described by Elliott et al, (1996). The stage discharge and water surface profile models were used to simulate hydraulic characteristics for each cell at specified discharges, the latter being more reliable for simulating flows above the highest calibration flow. Category 2 Habitat Suitability Criteria developed for brown trout on the River Piddle (*Bird et al, 1995*) were used in the HABTAE programme to calculate composite suitability indices for cells that were aggregated to derive total Weighted Useable Area (WUA) and mesohabitat WUA for each reach.

Time series were derived from habitat (WUA) – flow functional relationships in order to show duration and extent of habitat availability. Monthly time steps were aggregated into seasonal time steps as follows; (1) winter habitat durations (Nov – Mar) for spawning/incubation and adult life stages (2) summer growing season habitat (June – Oct) for fry and adult life stages. Habitat specific time series were also generated for meso-habitat types using the same procedure. Indices of habitat availability were developed representing different perspectives of the time series for each trout life stage. For example, habitat shortages during a particular season were evaluated using minimum habitat to represent acute low habitat events and the mean of the lowest 50% of values was used to depict longer-term effects of habitat minima. Habitat metrics were also developed for "near-shore" zones within 2 metres of the bank to evaluate importance of marginal habitats and for specific meso-habitat types. Length-frequency histograms of trout numbers allowed three age cohorts to be identified corresponding to fry (age 0+), juvenile trout (age 1+), and adult trout (age >1+). Trout abundance (N) in each age class was expressed in terms of density (N/m^2) to take account of variations in area between reaches and was used to assess annual changes in population structure in response to temporal habitat variations.

Linear regression analyses and Pearson correlation coefficients *(r)* were used to test for association between age specific trout densities and habitat (WUA). Where appropriate habitat data were natural log transformed in cases where variances exceeded mean values in order to stabilise variance and approximate a normal distribution, as in Nehring and Anderson (1993). The nature of the variables was such that associations could be assumed to be uni-directional and thus one-tailed tests of significance were employed. Relationships where $p < 0.05$ were considered to be significant.

Results

Adult Trout

Winter habitat durations (Nov – Mar) demonstrated effects of low mean monthly flows (MMF) in winter in depressing adult habitat availability. Moderate to high winter flows ($MMF>1.0m^3s^{-1}$) made little difference. Time series for streamside marginal habitats indicated these were a critical resource for adults in winter (fig. 1). Seasonal variations in marginal habitat showed that adult habitat availability was greater in winter than in summer, winter habitat exceeding summer habitat 80% of the time.

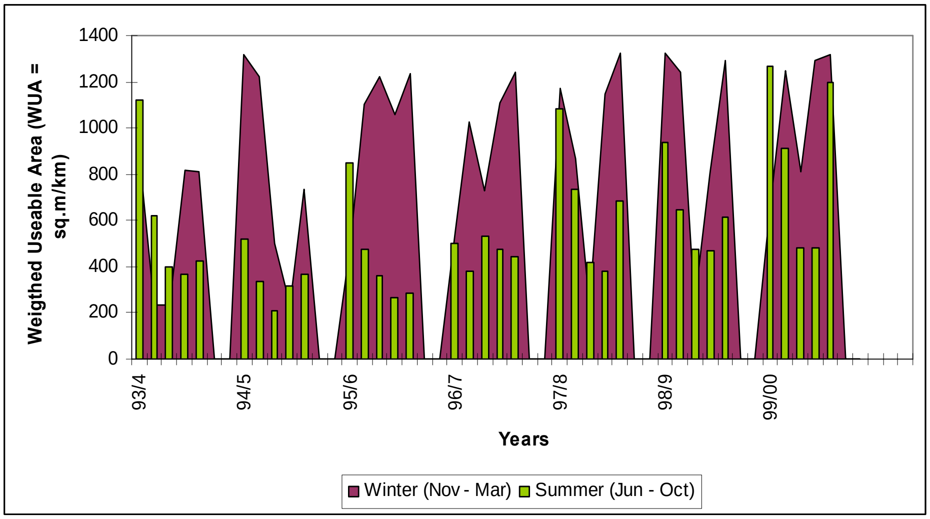

Fig 1. Time series comparison of mean monthly marginal habitat durations for adult trout in summer and winter

Adult densities were strongly correlated with winter habitat metrics (fig. 2) but no significant relationships with summer habitat were present in any study reaches. Availability of high quality marginal habitats in winter accounted for 91% of variation in adult densities in the upper sector but there were no associations with mean winter habitat suggesting overall habitat availability was relatively unimportant for adults compared to marginal habitats.

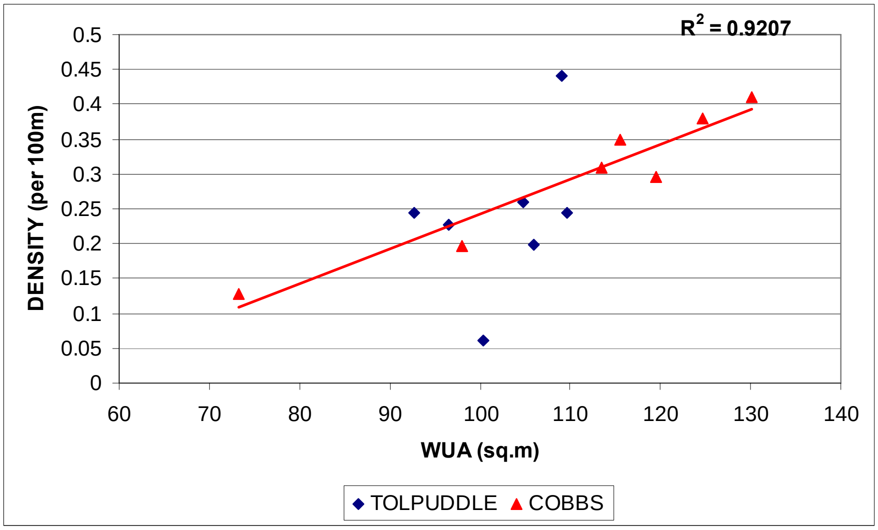

Fig 2. Relationship between annual adult density (1994 – 2000) and mean marginal habitat winter (WUA) for adults

Juveniles Trout: Spawning

Riffle zones provided better quality spawning than glides and pools over most of the simulated flow range except at very low winter flows (below 0.2 m^3/s) when glides provided the best areas (fig. 3). Most spawning habitat was consistently available when winter flows fluctuated between approximately 0.5 – 2.0 m^3/s showing the importance of a stable flow regime over the egg deposition to hatching period. Moderate flows during incubation, hatching and swim-up/dispersal of fry (Jan – Mar) resulted in the strongest 0+ year classes.

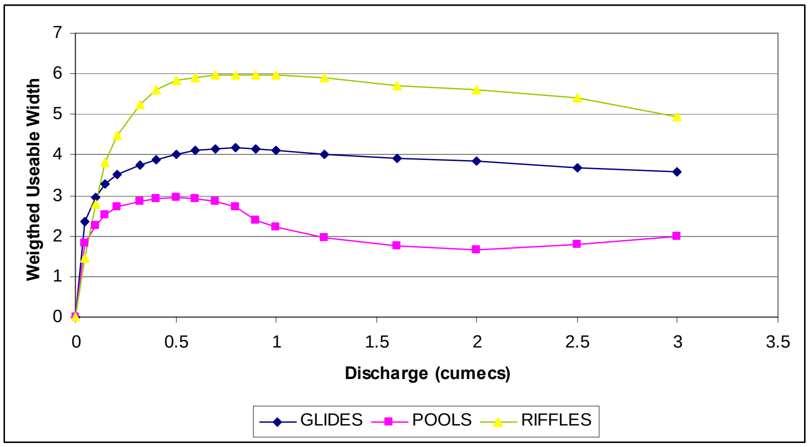

Fig 3. Habitat - discharge relationships for selected meso habitats

During the critical (hatching) period in February/March, mean spawning habitat availability in riffles was significantly correlated with densities of 0+ trout accounting for 65% of variability (*F=9.33; p=0.028*) (fig. 4). Mean riffle habitat was more highly correlated with fry density than total spawning WUA during the same period ($r^2 = 0.65$; p = 0.014 and $r^2 = 0.59$; p=0.025 respectively). There were no relationships with spawning metrics for other time periods. Densities of trout age 1+ the following year were significantly correlated with riffles and glides but showed a stronger association with glides. Mean glide hatching period ($r^2 = 0.74$, p=0.014) was virtually as good a predictor of juvenile density as total mean hatching period in the upper sector ($r^2 = 0.82$, p=0.006).

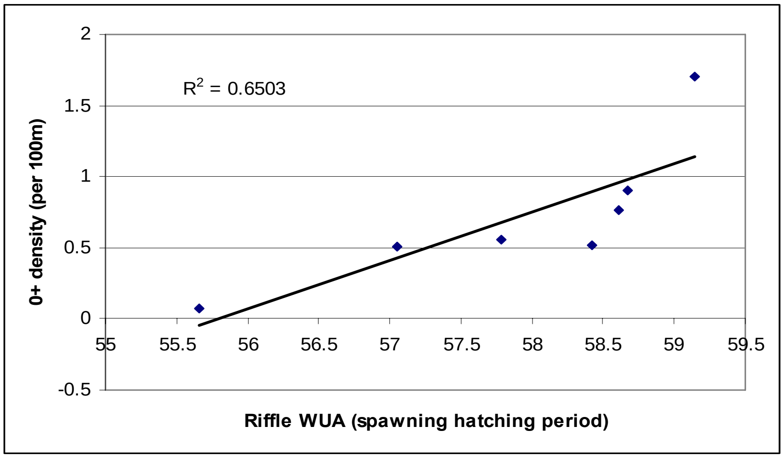

Fig.4. Relationship between spawning habitat in riffles during incubation/hatching period (February/March) and 0+ densities the following September

Juveniles Trout: Summer growing season

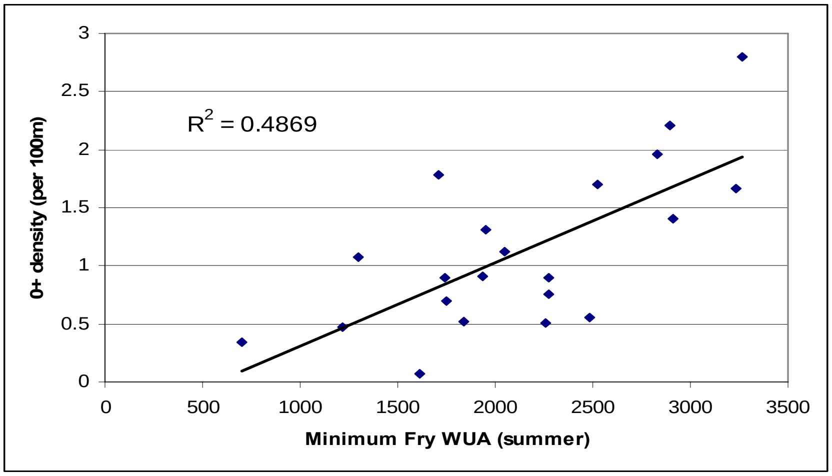

Fig. 5. Relationship between minimum fry rearing habitat during the first summer and 0+ density in September

Young-of-year habitat during the first growing season was positively correlated with 0+ fry densities in both reaches. Average summer habitat (July – September), summer minima and near shore habitat metrics were the best predictors of fry densities. Habitat predictions for young-of-year growing season combined for all reaches produced significant correlations with 0+ densities (r^2 = 0.49; p=0.001). Minimum monthly habitat availability was the best predictor of fry densities, accounting for 49% of variance (F= 18.1; p<0.01) (fig.5).

In the lower sector all meso-habitat metrics (except maxima) were significantly positively correlated with 0+ density for the three meso-habitat types present (riffles, glides, flats) but no one habitat type was found to be more important. Time series analysis of meso-habitat durations indicated that spawning habitat was in the order of 50% greater in riffles relative to glides but glides were more important as summer rearing habitats (fig. 6).

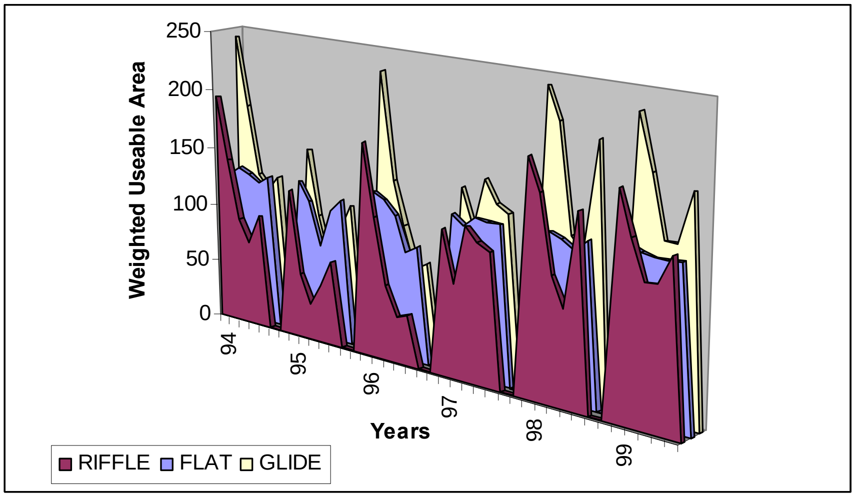

Fig 6. Time series (1994 – 1999) showing variations in summer habitat (WUA) for 0+ trout during the first growing season (June – October) in selected meso habitat types.

Metrics developed for a glide-riffle-glide habitat assemblage, which represented the best juxtaposition of rearing habitats significantly explained between 30 – 44% of variance in 0+ and 1+ densities. The best overall predictor of 0+ densities across all reaches was a combination of spawning WUA during the critical hatching period and minimum rearing habitat availability in summer. A multiple regression model indicated these two metrics explained 68% of variation in 0+ densities (F=19.28; p<0.01).

Discussion

Analysis of population data suggested population size was primarily regulated by year on year variations in recruitment of 0+, which showed an increasing trend with time and increased in relation to spawning stock size. Substantial increases in the ratio of adults to young-of-year in the Lower sector indicated pike predation was probably a major population-limiting factor at the commencement of the study period. Trout biomass becomes asymptotic in later years indicating that, even in high productivity chalk streams, density can have a major effect on growth, and that biological productivity may have become a more important limiting factor than physical habitat in the Lower River.

However, availability of spawning and rearing areas played a fundamental role in limiting 0+ recruitment, especially where local recruitment was evident. Our findings indicated that juxtaposition of spawning and rearing meso-habitats strongly influenced juvenile brown trout recruitment. The importance of riffles and glides was evident and the significance of longer-term summer habitat minima associated with low flows in the first growing season was marked, especially in the lower sector where 0+ densities were highest. This is in line with the findings of Elliott (1994) who demonstrated that spawning success in a Cumbrian stream had no effect on densities of surviving fry, which were regulated by density-dependent mortality in response to low amounts of nursery habitat. Time series of meso-habitat durations for young-of-year showed significant increases in contributions of riffles and glides to summer habitat in later years (Fig. 6). This reflects the role of flow augmentation in reducing critical low flow periods and consequent habitat depletion that caused widespread reductions in juvenile stocks throughout the middle catchment up to the mid 1990's (Strevens, 1999). These habitat increases corresponded with ongoing upward trends in 0+ densities throughout the study area over the survey period. The high densities of 0+ and 1+ trout in the lower reach demonstrated competitive segregation with juveniles dominant in the upper part and fry in the lower part. 1+ trout tend to be dominant and expel 0+ fry to shallow riffles and low velocity river margins where they are most commonly found (Bohlin, 1977; Cunjak and Power, 1986). In the upper sector where 0+ recruitment was low, the importance of spawning riffles to year class strength was more apparent, possibly due in part to lower availability of early rearing habitat and intra-specific competition between fry and parr.

Adult brown trout normally maintain station close to a shelter (Boussu, 1954; Heggenes, 1988b) and availability and diversity of cover have a significant effect on population density by increasing the numbers of territories and hence stream carrying capacity. In the Upper sector where the relative proportion of adults was higher, winter availability of meso-habitats associated with abundant marginal cover were critical to over-winter survival. Brown trout tend to have a strong preference for positions beneath overhead cover (Lewis, 1969), either above stream cover (<1m) or in-stream submerged cover. In summer, overhead cover provides shading that is important to adult trout, which become increasingly negatively phototropic as they develop progressively stronger shelter seeking behaviour with age (Bachman, 1984; Bagliniere and Maisse, 1999). In chalk streams, expansive tresses of ranunculus providing both velocity shelter and submerged overhead cover are abundant throughout the channel in summer. This abundance together with the relatively lower importance of "edge" habitats probably explains why no relationships with adult population size were observed. In winter, overhead cover can become a critically limiting resource often restricted to the stream margins where die-back of lush emergent marginal plants creates long tangled rafts of weed which snag around obstructions and woody debris creating complex cover zones of overhead and obstacle cover. When combined with sufficient depth these "features" provide excellent winter refugia for larger trout. Cunjak and Power (1986) demonstrated that association to cover was significantly greater in winter than summer for brown trout and that submerged cover was utilised more frequently than above water cover (Cunjak and Power, 1987). Low water temperatures may encourage adults to seek out habitats characterised by slower velocities than preferred in summer in response to reduced nutritional requirements, drift availability and the need to conserve energy (Cunjak and Power, 1986). Territorial behaviour also tends to decrease as temperatures fall and feeding ceases with the onset of winter (Mason and Chapman, 1965; Cunjak and Power, 1986). The lack of variation in adult densities and the strength of the association with marginal cover (r^2= 0.92) suggest that available winter habitat is the primary factor limiting adult carrying capacity at the reach scale. Our findings support the view that submerged cover is an important factor effecting winter survival of salmonids and that in small lowland streams marginal cover is a critical determinant of carrying capacity.

A juxtaposition of micro-habitats comprising a variety of specific stations used at different times makes up a trout “home range” (Shirvell and Dungey, 1983). The limited movement of resident brown trout in chalk streams (Soloman and Templeton, 1976) suggests habitat selection that enables trout to complete their life cycles within a relatively “local” area (Bachman, 1984). Thus, habitat juxtaposition is important in mitigating life history strategies of brown trout populations. Greater habitat diversity increases the likelihood that a self-sustaining population will be maintained by “local” adult stock. Furthermore, better understanding of the importance of different meso-habitat combinations for different life stages offers a means of enabling river rehabilitation schemes to more effectively manipulate natural population regulating mechanisms.

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Volunteers as an Integral Component of the Fisheries Program in Yellowstone National Park

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ABSTRACT—During the past decade, integrity of Yellowstone National Park aquatic resources has been threatened by a convergence of nonindigenous species. Priorities for research and monitoring have shifted and a majority of funding for fisheries in Yellowstone is now directed at the new emerging crises. At the same time, there are many basic questions regarding park fisheries that require immediate attention. To address this issue, a new program was established that brings dedicated volunteers, mostly from the angling community, to Yellowstone where they can participate as a member of a team directed at projects using fly-fishing as a collection technique. In 2002 and 2003, 114 fly-fishing volunteers from throughout the United States assisted with several specific fisheries projects, directed at genetic status, life history patterns, and species composition. The fly-fishing volunteer program has been successful at educating the public about fisheries issues in Yellowstone while providing a useful database of information for park biologists, garnered through stream and lake sampling with rod and reel. Future efforts will include a study where fish population information as measured by electrofishing is compared to that collected by fly-fishing. If similarities exist, angling could potentially be used to estimate other important fish population metrics.

Population Trends and an Assessment of Extinction Risk for Westslope Cutthroat Trout in Select Idaho Waters

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ABSTRACT—Despite westslope cutthroat trout (*Oncorhynchus clarki lewisi*) being petitioned for listing under the Endangered Species Act, formal evaluations of extinction risk for the sub-species have been quite limited. In this study, we summarize existing population trend data for westslope cutthroat trout, use the trend data to estimate population growth rates, and combine these with various likely initial population sizes to assess generalized extinction risk for westslope cutthroat trout within select Idaho drainages. Population data consisted of over 30 years of snorkel trend counts for westslope cutthroat trout across a broad geographic area in Idaho. Results of trend analysis including both inspection of graphs, and calculation of infinitesimal growth rates, indicate that westslope cutthroat trout have maintained or increased their population abundance over a large area within the state of Idaho during the past 15-34 years. Total estimates of westslope cutthroat trout numbers within various Geographic Management Units (GMU's) conservatively range from 6,500 to 341,000 fish, with a combined estimate of approximately 1.2 million fish for the GMUs considered in this study. Mean sub-basin population size ranged from about 400 to 13,000 fish. Population persistence for 100 years ranged from high to low for various individual local populations. However, the study results suggest that numerous sub-populations within most GMU's, available to interact within a classic or less traditional metapopulation framework, would result in a high ($\geq$ 95%) probability of westslope cutthroat trout persistence over 100 years.

Introduction

The westslope cutthroat trout (*Oncorhynchus clarki lewisi*) is one of two recognized sub-species of cutthroat trout residing in the Columbia and upper Missouri river basins. Although westslope cutthroat trout have been the subject of numerous localized investigations, relatively few authors have focused on general sub-species status on a broad geographic scale. McIntyre and Rieman (1995) noted that range declines have occurred across historic westslope cutthroat trout range. Causes of declines include predation by, and competition with exotic native species, overharvest, genetic introgression, habitat degradation and fragmentation (Liknes and Graham 1988; Rieman and Apperson 1989; Thurow et al 1997).

Westslope cutthroat trout were petitioned for listing under the Endangered Species Act in 1997. This petition was initially found to be unwarranted by the United States Fish and Wildlife Service (USFWS) which concluded that a large number of westslope cutthroat trout populations exist across the sub-species range (Federal Register 65Fed.reg.20120). However, a subsequent legal decision required the USFWS to reevaluate the status of westslope cutthroat trout. Results of the second evaluation completed in 2003 also concluded that the sub-species does not need ESA protection (Federal Register 68.reg.46989).

Despite two formal ESA reviews, quantitative evaluations of extinction risk for westslope cutthroat trout populations using Population Viability Analysis (PVA) have been quite limited. McIntyre and Rieman (1995) summarized data for 6 westslope cutthroat trout populations in Idaho and Montana, and calculated variances of infinitesimal rate of growth. Using the modeling approach of Dennis et al. (1991), the authors concluded that stochastic extinction risk will increase sharply for populations that drop to fewer than 2000 individuals. They assumed their study populations varied around an equilibrium with no long-term trend in population number. Thus, their results represent risk associated with random and not deterministic factors. Using a complex Bayesian modeling approach, Shepard et al. (1997) estimated extinction probabilities for 144 westslope cutthroat trout populations in the upper Missouri River basin in Montana. Ninety percent of the populations evaluated had a high or very high probability of going extinct during 100 years based on model projections.

The studies of McIntyre and Rieman (1995) and Shepard et al. (1997) suggest that some westslope cutthroat trout populations could be in jeopardy of extinction but the applicability of those findings to the entire sub-species range is unknown. For example, Shepard et al. (1997) noted that most of the populations considered in their study were small and resided in isolated headwater stream segments less than 10 km long. In Idaho, the presence of fluvial populations in large river systems within the Federal Wilderness system with histories of restrictive fishing regulations dating back 30 years or more may provide increased population resiliency.

In this study, we 1) summarize existing population trend data for westslope cutthroat trout to provide perspective on their current status in Idaho, and 2) use these trend data to estimate population growth rates and combine these with rough approximations of population sizes to assess generalized extinction risk for westslope cutthroat trout within select Idaho drainages.

Methods

Population Trends

Historical snorkel counts

With assistance from IDFG personnel, we summarized snorkeling data collected over three decades from mainstem river sites in four westslope cutthroat trout streams including the St. Joe, Coeur d'Alene, Selway, and Middle Fork Salmon rivers. Snorkeling techniques used on these rivers are similar and described in detail in Rankel (1971), Corley (1972), Lindland (1974), and Johnson and Bjornn (1978). Briefly, one or two divers float downstream counting all westslope cutthroat trout observed, either in the entire stream channel or within prescribed counting lanes.

Snorkel counts were begun on 27 sites on the mainstem St. Joe River in 1969, 29 mainstem and tributary sites on the Coeur d'Alene River in 1973, 27 sites on the Selway River in 1973, and 12 sites on the Middle Fork Salmon (Figure 1). Snorkel site lengths and/or counting lane widths were measured periodically during the sampling periods to ensure the same reaches were being sampled and to enable calculation of fish densities (fish/100m^2).

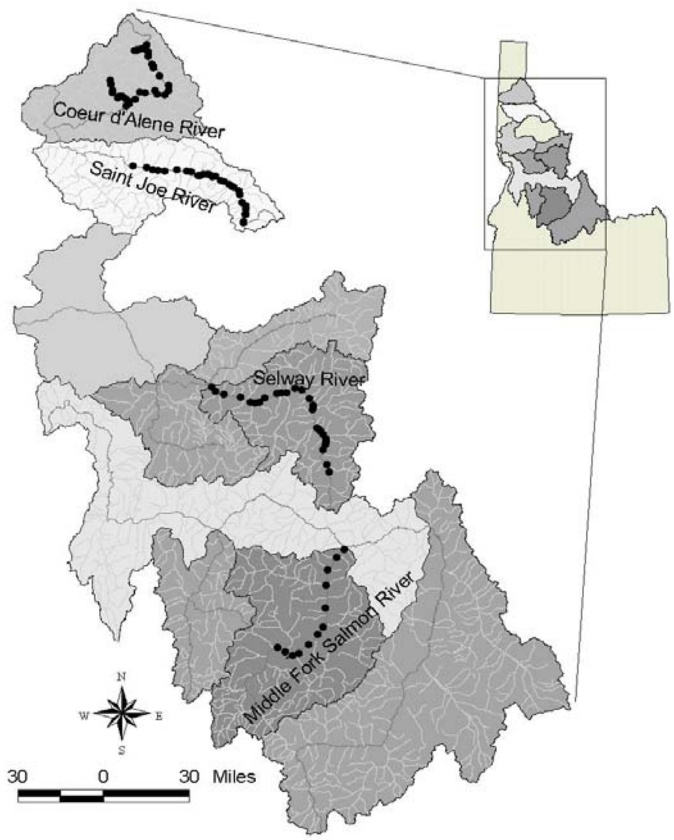

Figure 1. Location of historical snorkel count sites used to monitor westslope cutthroat trout abundance in Idaho, 1969 to present.

General Parr Monitoring counts

In addition to the above historical counts, a sizeable number of additional snorkel count sites have been established for a shorter time across many waters within westslope cutthroat trout range in Idaho. Since 1985, these trend counts have been conducted by IDFG personnel funded via several Bonneville Power Administration-funded research projects as part of what has been termed General Parr Monitoring (GPM). Although originally designed to track trends for anadromous species, observations on all resident fish present have been recorded as well. The dataset contains cutthroat trout density estimates for a few mainstem river sites, but the bulk are conducted in smaller tributary streams typically snorkeled by crawling upstream. Petrosky and Holubetz (1986) provide a more detailed description, including snorkeling techniques, physical parameter measurements, and conversion of raw fish counts to densities (fish/100m^2).

To evaluate westslope cutthroat trend using the above data, we first subdivided the area containing snorkel counts into Geographic Management Units or GMU's (Figure 2) (Lentsch et al 1997). GMU's were large segments of major drainages likely to contain metapopulations (Hanski 1991) based on expert opinion and on extensive westslope cutthroat movement studies conducted in the past (Bjornn and Mallet 1964; Hunt and Bjornn (1991). For each GMU, we subsequently queried the General Parr Monitoring database (J. Griswold IDFG, unpublished dataset) for those snorkel sites where 1) counts were conducted in 10 or more years since 1985 and 2) where 1 or more westslope cutthroat trout were observed during the entire counting period (Figure 2). These individual monitoring sites average about 100 m in length and ranged in number from 1 to 10 on individual streams. Mean density (fish/100m^2) of westslope cutthroat trout observed from 1985 to present was calculated for all such monitoring sites within each GMU. We subjectively considered five individual snorkel sites as the minimum necessary to derive mean trend values for a GMU.

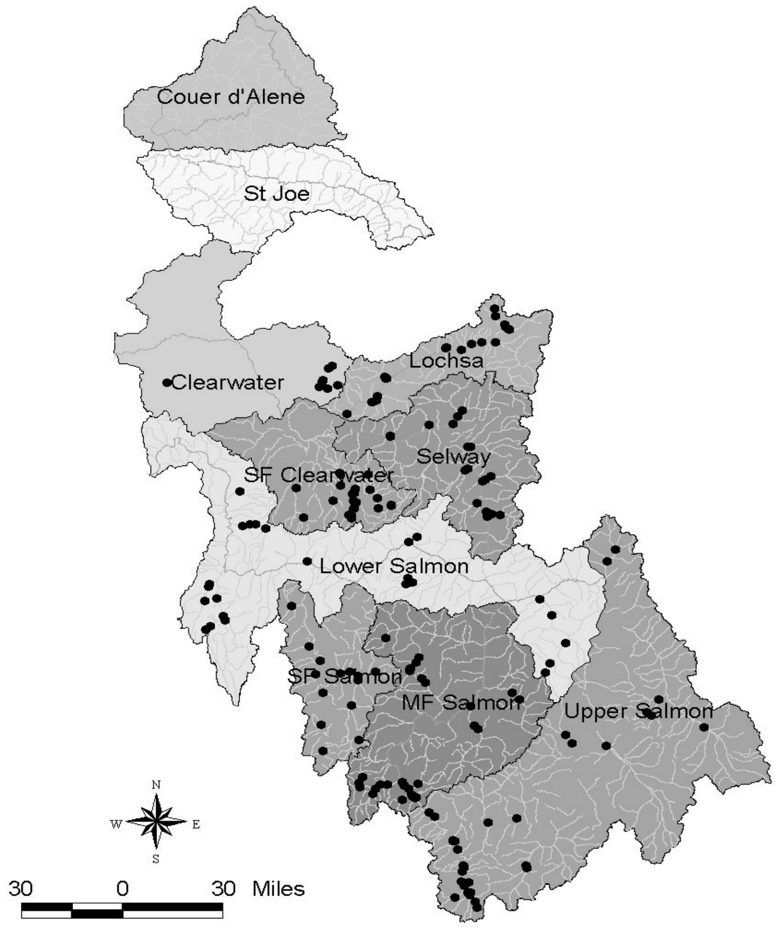

Figure 2. Streams in Idaho westslope cutthroat range within 10 Geographical Management Units (GMUs) and location of GPM snorkel sites (n=206) with ten or more years of data and where WCT were observed in at least one count year.

Estimation of GMU Population Sizes

To approximate the number of westslope cutthroat trout residing in select GMUs we began by summarizing stream lengths inhabited by westslope cutthroat trout in those basins. We relied on the most recent IDFG estimate of inhabited stream kilometers in Idaho within each GMU, derived from existing population data and professional judgment as summarized in a 2002 multi-state status review (Shepard et al. 2003).

In addition to those sites monitored over many years for trend information, the GPM database contains a large number of snorkel sites where one-time population estimates are conducted. The entire GPM database consists of 6,300 counts of trout and salmon abundance conducted at 2300 different sampling locations scattered across anadromous fish-bearing portions of the state since 1985 (IDFG, unpublished data). We considered all GPM snorkel estimates conducted during the period 1996-2000, including those with and without westslope cutthroat trout present, useful in approximating current cutthroat trout population size for a GMU. For all such snorkel counts, the number of westslope cutthroat trout observed was divided by the length of stream snorkeled to obtain a linear estimate of fish density (fish/km). We subsequently calculated mean linear densities of cutthroat observed for each GMU.

To estimate total GMU-wide population sizes, the estimates of stream km occupied by westslope cutthroat trout within each GMU described above were multiplied by the mean estimate of linear density (fish/km) for the sub-species in the same GMU. Because the GMU boundaries we originally selected for the Upper Salmon and Lower Salmon GMU's did not coincide with geographic subdivisions used in mapping present westslope cutthroat distribution (Shepard et al. 2003), we did not attempt to estimate total population sizes for these two areas.

The extinction risk modeling effort below evaluates the effect of multiple subpopulations on persistence probabilities for westslope cutthroat within an entire GMU. Stream basins from third to fourth order in size are thought to mark the boundary between local sub-populations of westslope cutthroat trout (B. Rieman, USFS, pers. communication). To provide a rough approximation of average sub-population size in the various study waters, we used ArcView GIS software to calculate the proportion of stream kilometers within third and fourth order basins for each GMU. This value was multiplied by the total GMU-wide population estimate above. We subsequently divided these estimates by the number of third or fourth order drainages within a GMU to yield approximate mean sub-basin population sizes.

Extinction Risk Modeling

We analyzed westslope cutthroat trend data from both the historical and GPM snorkel counts above using the stochastic exponential growth model of Dennis et al. (1991). The mean instantaneous rate of population change (μ) and the variance in rate of change (σ^2) were calculated for trend datasets within each GMU using STOCHMVP, a software program developed to facilitate use of the Dennis model (E.O. Garton, Dept of Fish and Wildlife Resources, University of Idaho). For those GMU's where two sources of long-term data were available (Middle Fork Salmon and Selway rivers), the longer of the available datasets was used to estimate μ and σ^2.

We utilized estimates of these two parameters and a range of population sizes to estimate probability of single populations persisting for 100 years within the various GMU's, again with the aid of the program STOCHMVP. As in Reiman and McIntyre (1993), the sensitivity of model results to a persistence threshold was evaluated by comparing results for two arbitrarily selected thresholds; in the present study 10 and 100 fish.

Estimates of μ are often imprecise and a given value can have major impacts on extinction risk using the Dennis model (Goodman 2002). Accordingly, we estimated the probability of persistence for individual populations of westslope cutthroat trout in various GMU's using two estimates of instantaneous growth rate. These values included a calculated growth rate from observed data, along with the associated variance estimate, and an assumed μ of 0.0 reflecting a population at equilibrium (Rieman and McIntyre 1993). The equilibrium growth rate was assumed to have a variance identical to the observed value for a given GMU.

Many of the large, relatively pristine drainages in Idaho that are protected by wilderness designation likely harbor numerous local populations in a metapopulation structure (Hanski 1991). Accordingly, the probability of persistence (100 years) for a single large population composed of multiple sub-populations was also estimated as $1-(P_1 \cdot P_2 \cdot \ldots . P_i)$ where P_i = the probability of falling below the threshold in each of the *i* sub-populations (Rieman and McIntyre 1993). This process was repeated for three paired μ and σ^2 values likely to encompass the range for Idaho populations based on the calculated values for individual GMU's above. We selected an extinction threshold of 10 fish and assumed no re-founding or temporal correlations in population size among sub-populations in this final modeling effort.

Results and Discussion

Population Trends

Historical snorkel counts

Trend counts in the historical mainstem St. Joe, Middle Fork Salmon, and Selway River snorkel sites all increased markedly during the mid- to late-1970s during a period following establishment of special regulations on much or all of their length (Figure 3). The sharp rise in the St. Joe River cutthroat trout population during this period was studied intensively and attributed to reductions in angler exploitation from a trophy fish regulation adopted in 1972 (Johnson and Bjornn 1978). The increase in cutthroat abundance on the Selway and Middle Fork Salmon Rivers is less well understood, but Ortmann (IDFG, unpublished data) observed that population increases on the latter water were less likely related to fishing regulation change than to natural factors.

Following three to five-fold increases in population numbers during the first 15 years of trend monitoring, populations in the two waters containing anadromous fish (Selway and Middle Fork Salmon Rivers) declined substantially during the late-1980's and early-1990's (Figure 3). In contrast, the St. Joe River population appeared to peak in size in 1995 and then declined.

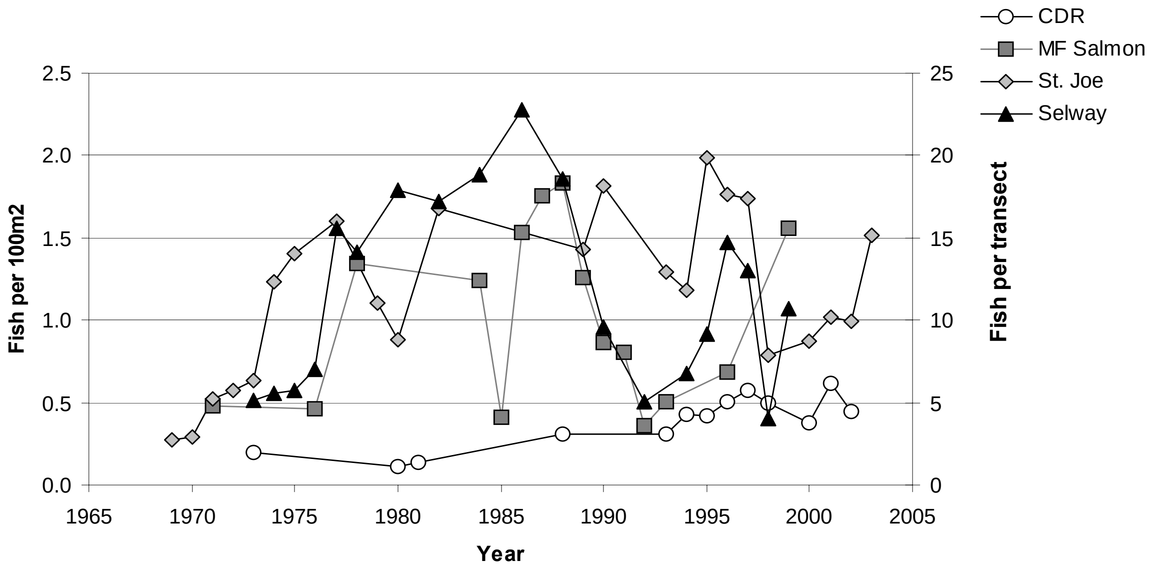

Figure 3. Trends in westslope cutthroat trout abundance (fish/100m2) determined by snorkeling in the St. Joe, Middle Fork Salmon, and Coeur d'Alene rivers, 1969-2000. Selway River data are available only in fish/transect (right scale).

Declines in all three of these streams occurred despite a continuation of mainstem special regulations and expansion of catch-and-release to most or all tributaries. The Selway and Middle Fork Salmon populations appear to have declined in near synchrony. All three populations appear to have increased sharply in the mid-1990s. Although data are more limited, cutthroat abundance in the Coeur d'Alene River also improved markedly during the late-1990s. An evaluation of possible reasons for the similarity of trends in these populations is outside the scope of this paper. However, it is worth noting that in general, drought conditions prevailed across Idaho from 1987 to 1994 (except 1993) with improved water conditions in subsequent years.

General Parr Monitoring counts

Examination of the GPM snorkel data suggests that most westslope cutthroat trout populations monitored within waters supporting anadromous species are either stable or increasing. Mean density (fish/100m2) in four of eight GMU's being monitored, including the Lochsa River, Lower Salmon River, South Fork Clearwater River, and South Fork Salmon River appeared to be flat, or nearly so (Figure 4). Mean density in the Middle Fork Salmon GPM sites appear to have declined since 1985, although data collected from the historical trend sites over a longer period do not demonstrate the same results. Conversely, westslope cutthroat populations in the Clearwater, Selway, and Upper Salmon rivers appear to have increased during the period from 1985 to 2000, as characterized by relatively steep trend line increases (Figure 4).

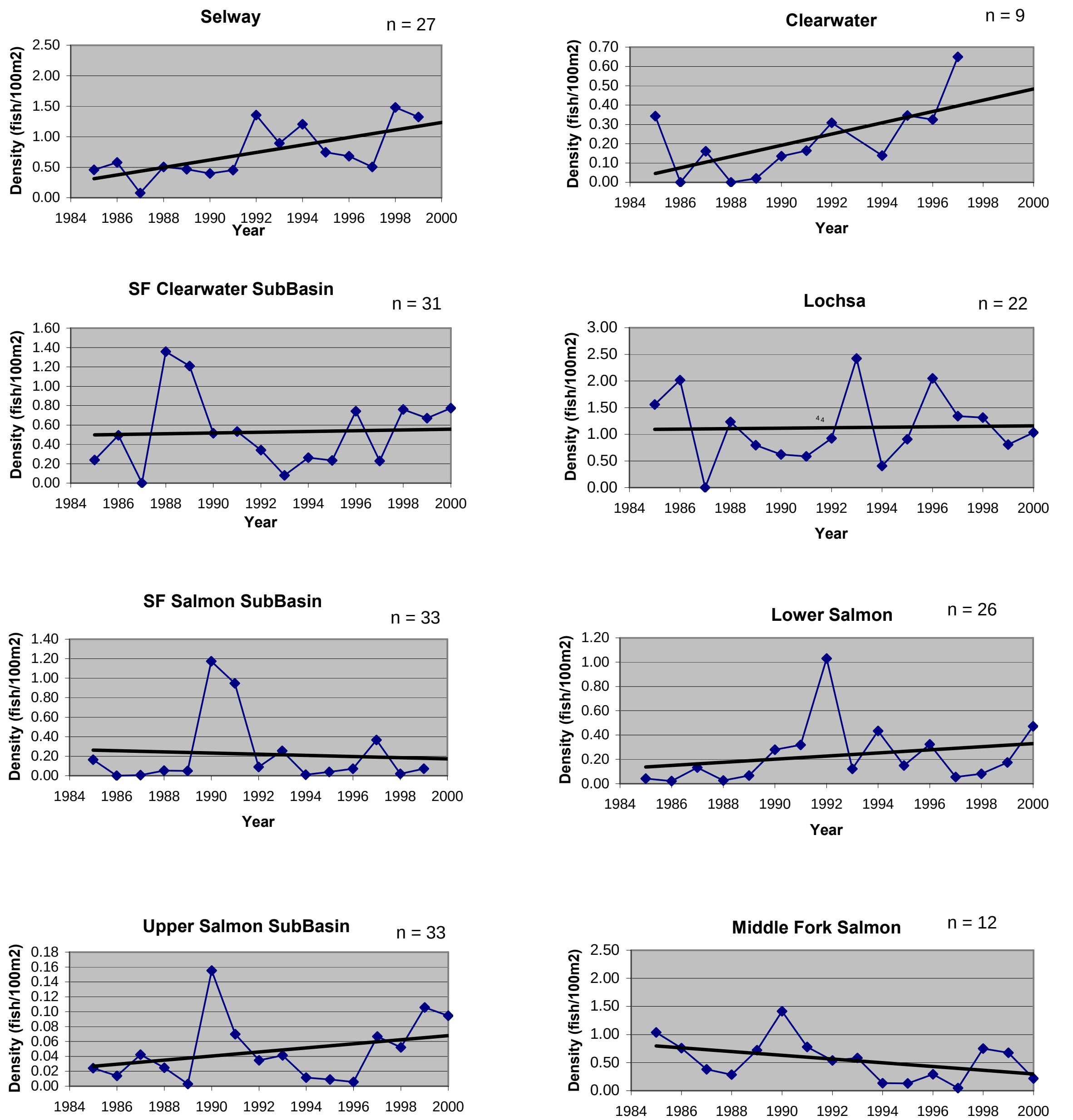

Figure 4. Trends in westslope cutthroat abundance (fish/100m2) determined by snorkeling at General Parr Monitoring (GMU) locations having 10 or more years of data during 1985-2000; n = numbers of individual count sites within the GMU.

Estimation of GMU Population Sizes

For the GMU's considered in this study, estimates of stream length occupied by westslope cutthroat trout range from 1,057 km for the South Fork Clearwater River to 3,351 km for the Middle Fork Salmon River (Table 1). Estimated mean linear density ranged from 5 to 149 fish/km on the South Fork Salmon and Selway rivers, respectively. With the exception of the mainstem historical snorkel counts, electronic data pertaining to fish densities on the St. Joe River were unavailable. However, historically available estimates of density on the St. Joe River have typically been high relative to other streams (Figure 3). We approximated what is likely a minimum number of fish present in the St. Joe River drainage assuming an average value of westslope abundance of 68 fish/km derived from all other drainages in this study (Table1). Estimates of westslope cutthroat trout numbers within the various GMU's ranged from 6,568 to 341,767 fish (Table 1) with a total combined estimate of approximately 1.2 million fish for the GMU's considered. Although space limitations preclude a detailed discussion of possible positive or negative biases in the above estimates, it is likely that total GMU population estimates for westslope cutthroat trout in this study (Table 1) were underestimated rather than overestimated.

Table 1. Approximate numbers of westslope cutthroat trout (WCT) present in geographic management areas (GMU's) based on linear "density" estimates (WCT/km) and number of km with WCT present.

Basin	Stream length[1] w/WCT (km)	Mean WCT/km	Total estimated WCT	No. of 3rd-4th order sub-basins	% GMU stream km in 3rd-4th order sub-basins	Estimated WCT in 3rd-4th order sub-basins	Mean populations size/sub-basin
Middle Fork Salmon River	3351	88	294,878	28	75%	221,158	7,899
St. Joe River	2185	68[2]	148,565	20	74%	109,938	5,497
Coeur d'Alene River	1446	15	21,058	13	74%	15,583	1,199
Selway River	2294	149	341,767	21	78%	266,578	12,694
Clearwater River	3507	40	139,504	33	79%	110,208	3,340
Lochsa River	1340	134	179,547	17	67%	120,297	7,076
South Fork Clearwater	1057	25	26,415	15	74%	19,547	1,303
South Fork Salmon	1314	5	6,568	13	84%	5,517	424

[1] As summarized by IDFG fishery management 2002 (Corsi unpublished data)
[2] Minimum estimate assuming average WCT/km across all drainages.

Assuming the total GMU-wide population estimates are within the realm of true abundance, the approximate mean sub-population sizes within the various GMU's ranged from about 400 to nearly 13,000 fish (Table 1). This estimate is admittedly a crude approximation. However, despite some limitations, these estimates provide a general range of sub-population sizes for Idaho westslope cutthroat trout populations for use when considering results from the extinction risk modeling below.

Extinction Risk Modeling

Estimated instantaneous rates of change (μ) for westslope cutthroat trout populations in 10 Idaho streams ranged from –0.059 to 0.1643 (Table 2). Based on the historical and GMP trend datasets, eight of 10 estimates were positive over the monitoring period implying increased population growth. Only two trend datasets including the Lochsa River and South Fork Salmon River counts, produced estimates of negative population growth.

Estimates of variance of instantaneous rates of change (σ^2) differed markedly among the four historical datasets and the shorter-term GPM trend sites (Table 2). Within the historical snorkeling datasets, estimates of σ^2 ranged from 0.07 to 0.37; estimates from the GPM database ranged from 0.50 to 4.03. With the exception of the highest estimate (4.03), the present estimates of variance around μ were similar to the range (0.07-1.02) reported by McIntyre and Rieman (1995) for seven westslope cutthroat trout populations in Idaho and Montana.

Table 2. Estimated mean (μ) and variance (σ^2) for instantaneous rates of change in westslope cutthroat trout populations calculated from snorkel counts in Idaho streams, 1969-2002.[1]

Basin	Dataset period	Years obs.	Sites counted	μ	σ^2
MFk Salmon River	1971-1999	15	12	0.0421	0.37
St. Joe River	1969-2002	20	27	0.0155	0.12
Coeur d'Alene River	1973-2002	13	29	0.0272	0.07
Selway River	1973-1999	19	27	0.0284	0.23
Clearwater River	1986-1998	10	9	0.054	1.05
Lower Salmon River	1985-2000	15	26	0.1643	1.28
Lochsa River	1985-2000	15	22	-0.0277	0.50
SFk Clearwater River	1985-2000	15	31	0.0777	0.74
SFk Salmon River	1986-2000	14	17	-0.059	4.03
Upper Salmon River	1985-2000	14	33	0.1004	0.89

[1] Calculated after Dennis et al. 1991

Predicted probabilities of persistence for 100 years in single local populations of westslope cutthroat trout in Idaho were strongly influenced by both the estimate of μ employed (observed versus an assumed equilibrium value of 0.0), and by the selection of an extinction threshold (Table 3). Not surprisingly, populations with low estimates of σ^2 (e.g., Coeur d'Alene River = 0.07 from Table 2) had a high probability of 100-year persistence, while populations with high variance (e.g. South Fork Salmon River = 4.03) had low probabilities of persistence regardless of the selected value for μ (Table 3). Persistence probabilities were relatively insensitive to changes in initial population size over the three population sizes modeled when holding all other factors constant. These results are similar to those reported by Rieman and McIntyre (1993) in their assessment of bull trout extinction risk.

Table 3. Estimated probabilities of persistence for single populations of Idaho westslope cutthroat trout given three different initial sizes. I alternately assumed extinction thresholds of 10 or 100 fish and also alternated estimates of μ and σ^2 and from existing trend data or an equilibrium value of μ (0.0) with observed σ^2.

	Pop size = 2500				Pop size = 5000				Pop size = 10,000			
	$\mu = 0.00$		$\mu =$ observed		$\mu = 0.00$		$\mu =$ observed		$\mu = 0.00$		$\mu =$ observed	
	Threshold		Threshold		Threshold		Threshold		Threshold		Threshold	
Stream	10	100	10	100	10	100	10	100	10	100	10	100
MF Salmon	.637	.403	.827	.617	.694	.480	.866	.695	.746	.551	.898	.760
St. Joe	.899	.648	.951	.782	.947	.743	.971	.855	.991	.820	.984	.907
CDA	1.000	.779	.997	.952	1.000	.868	.999	.978	1.000	.935	1.000	.990
Selway	.752	.498	.886	.683	.808	.586	.918	.762	.856	.664	.942	.824
Clearwater	.410	.247	.576	.380	.456	.297	.624	.445	.500	.347	.668	.505
Lower Salmon	.376	.225	.774	.576	.419	.272	.813	.649	.460	.317	.846	.709
Lochsa	.565	.351	.428	.241	.621	.420	.483	.297	.672	.485	.535	.354
SF Clearwater	.479	.292	.751	.543	.530	.351	.794	.618	.578	.408	.830	.681
SF Salmon	.217	.127	.158	.090	.243	.155	.179	.110	.269	.181	.200	.131
Upper Salmon	.442	.267	.753	.550	.490	.322	.796	.623	.536	.375	.831	.686

If large numbers of local populations of westslope cutthroat trout in Idaho were believed to function in complete isolation within the range of three population sizes modeled (2,500 to 10,000 fish), then many would be assumed to be at risk of extinction within 100 years based on the results of Table 3. Isolated populations may indeed exist in some instances (e.g. South Fork Salmon) where the approximated sub-population size averaged less than one thousand individuals (Table 1) and westslope cutthroat trout are not widely distributed within the GMU (Thurow 1985). In addition, other large stream systems with less prominent westslope cutthroat populations than those considered in the present study (e.g., The St Maries River) may be areas for concern. However, given the widespread distribution of westslope cutthroat trout in many of the relatively pristine watersheds in central Idaho, and extensive movement patterns documented for fluvial populations (Bjornn and Mallet 1964; Hunt and Bjornn 1991), it is likely that many local sub-populations within the study area function as a classic metapopulation (Levins 1970) or a less traditional form often observed (Harrison 1991). In either event, an increased probability of overall persistence would be expected compared to the above estimates of persistence for single local populations (Harrison 1991; Doak and Mills 1994).

An assessment of the number of westslope cutthroat trout populations necessary to ensure a high probability of persistence in the absence of dispersal provides some perspective on extinction risk for such meta-populations. For populations ranging in initial size from 500 to 20,000 individuals, the number of sub-populations needed to maintain a 95% probability of at least one population persisting 100 years ranged from two to seven populations for the two simulations involving modest growth and equilibrium growth (Figure 5). In the case of a declining population experiencing high variance, the number of populations needed for 95% persistence ranged from eight to 18 populations, across the same initial population size range.

A major limitation of the PVA modeling approach used in this study is that in calculating extinction risk, the Dennis et al. (1991) model assumes that no density-dependence occurs. McFadden (1977) argued for the widespread reality of density-dependent processes in fish populations noting that the existence of such processes comprise the very core of fishery science. Assuming density-dependence actually occurs, Goodman (2002) observed that most combinations of reasonable parameter values in the Dennis et al. (1993) model will result in projections either trending to unrealistically high levels or to short-term extinction. Use of a positive growth value in the model will likely result in optimistic estimates of persistence, although results could not be biased past the predictions observed for equilibrium growth (Figure 5). Conversely, use of a negative growth value in the model will result in unduly pessimistic persistence predictions (Goodman 2002). Because the majority of available trend data for Idaho westslope cutthroat trout suggest a either positive population growth or equilibrium growth, the curve in Figure 5 based on equilibrium growth ($\mu = 0.0$) is probably the best point estimate. This curve suggests that, under the range of population sizes modeled (250-30,000), only three to nine sub-populations would be needed to ensure population persistence. Results of Table 1 suggest that the number of sub-populations present (3rd-4th order drainages) in many Idaho GMU's exceed these levels and they should therefore be adequate for persistence.

A very conservative approach to assessing westslope cutthroat extinction risk would be to consider the area *between* the curves for equilibrium and negative population growth as the guideline for assessing risk (Figure 5). As an example, for multiple sub-populations with initial sizes of 10,000, the number of sub-populations necessary to ensure 100 year persistence would be 4 to 9. For populations with initial sizes of 1,000 the number of populations needed would range from 7 to 15. Again, based on the rough estimates of sub-basin population sizes for various GMU's (Table 1), a sufficient number of populations appear available in most cases.

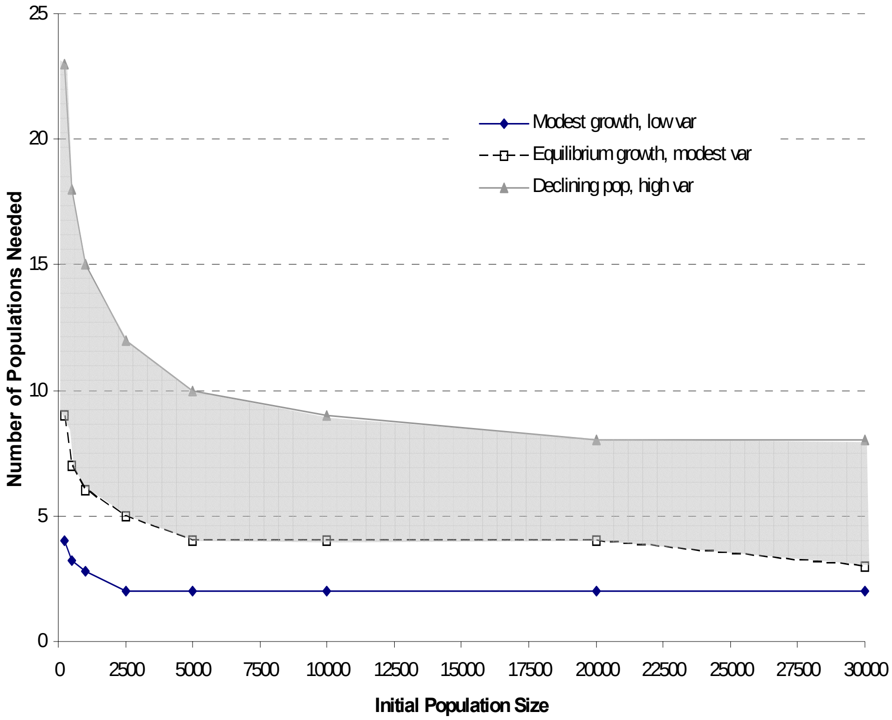

Figure 5. Estimated number of westslope cutthroat trout populations necessary to ensure a 0.95 probability of persistence for at least one population, given a range of likely sizes. Modest growth values from Middle Fork Salmon River (μ = 0.042, σ^2 = 0.37), equilibrium μ assumed to be 0.0 with moderate variances of 0.75, declining population μ from the SFk Salmon River (μ = -0.059) with high variance (1.28) from the Lower Salmon River. All populations assumed to be completely independent. Equilibrium growth value the best point estimate for most Idaho populations based on available trend data; shaded area a conservative estimate due to inability of Dennis et al (1993) model to consider density dependence (see text).

Although generalizations on the number of sub-populations needed for persistence in this study should only be viewed as approximations, it seems likely that they are quite conservative. The GMU population sizes developed to put the various extinction estimates into general perspective are likely underestimated given the use of snorkel counts and location of GPM sample sites. Perhaps the most compelling reason to suspect the persistence estimates produced by this study are conservative relates to sampling error likely involved in our estimation of population trends. The variance (σ^2) associated with point estimates of infinitesimal population growth rate (Table 2) assume no measurement error in the population trend data; i.e., 1) all variation in snorkel counts is due solely to population changes and 2) counts within the snorkel trend count zones reflect the stream-wide population trend perfectly. In reality, differences in personnel snorkeling skills, possible annual differences in fish movement, and a host of other factors are reflected in the snorkel count data. In past extinction assessments, sampling error associated with population trend data has turned out to be important, creating considerable negative bias in the persistence probabilities derived via the Dennis model (Rieman and McIntyre 1993; B. Rieman, USFS, personal communication).

Applying the model projections of Table 3 and Figure 5 directly to specific estimates of sub-population size is impossible; the requisite data are not available. Even if available, it is questionable whether such absolute extinction risk estimates would be rigorous enough (Ralls et al 2002). Instead, we have opted to develop some simple, generalized risk models that compare the relative extinction risk for various populations with a range of population size and growth rates characterized by available data. The use of such generalized PVA models to evaluate relative extinction risk can be quite useful when viewed as thought experiments (Ralls et al. 2002). In fact, it has been argued that use of such a simplistic modeling approach is preferable to more "realistic" spatial models that are often too poorly parameterized to be of much use (Doak and Mills 1994).

Conclusions

The dataset developed for assessing trend in this study is comprised of 301 individual sites where westslope cutthroat trout trends counts have been conducted via snorkeling over a 10 to 34 year period. Given the extensive monitoring period and the relatively broad dispersion of the snorkel monitoring sites used in this study, these data likely comprise the most extensive monitoring effort for a resident trout species ever conducted in America. Taken collectively, the data do not suggest that westslope cutthroat trout are declining in abundance within Idaho. Rather, the broad distribution of sites involved in both the historical sites (Figure 1) and the GPM dataset (Figure 2) and results of the trend analysis (Figures 3 and 4; Table 2) demonstrates that westslope cutthroat trout have maintained or increased their population abundance over a very large area within the state of Idaho during the past several decades.

Total estimates of westslope cutthroat trout numbers within the various GMU's ranged from 6,568 to 341,767 fish with a total estimate of approximately 1.2 million fish for all GMU's combined. Although estimates of precision for these estimates are not presented due to non-random sampling, consideration of possible sources of bias indicates that the above estimates are likely to be conservative.

Although estimates of population persistence for 100 years ranged from high to low for various individual local populations, the above study results suggest that numerous large sub-populations within most GMU's, available to interact within in a classic or less traditional metapopulation framework, would result a high ($\geq$95%) probability of persistence over 100 years in many instances.

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