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Conservation Assessment for Lake Chub, Mountain Sucker, and Finescale Dace in the Black Hills National Forest, South Dakota and Wyoming

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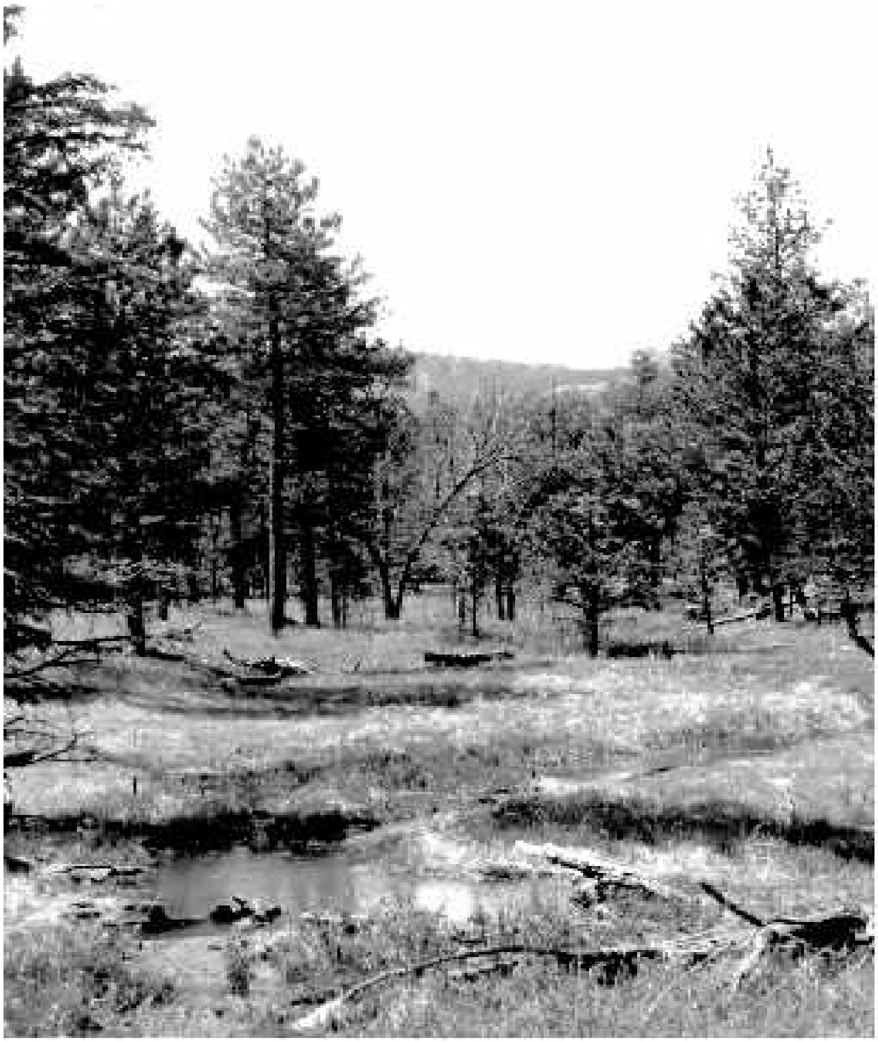
Charles R. Berry, Jr.



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EXECUTIVE SUMMARY

This report assesses the biology and overall conservation status of lake chub (LCB; *Couesius plumbeus*), mountain sucker (MTS; *Catostomus platyrhynchus*), and finescale dace (FSD; *Phoxinus neogaeus*) in the Black Hills National Forest (BHNF). Historic accounts suggest LCB occurred in streams across most of the Black Hills, but this species now occupies only a fraction of its former range. Recent surveys have documented the presence of LCB in the Belle Fourche River to the immediate north of the BHNF, but the only remaining population on BHNF land occurs in Deerfield Reservoir, where the population appears to have declined in abundance since 1994. Because LCB are concentrated in Deerfield Reservoir, stochastic environmental events related to extended droughts or severe floods may pose considerable threats to this population. More immediate risks to the LCB population in Deerfield Reservoir include the ongoing establishment of an exotic predator, rock bass (*Ampbloplites rupestris*), in littoral areas and the possibility of a chemical renovation of Deerfield Reservoir in 2003. If any of these factors were to result in the extirpation of the Deerfield Reservoir population, the probability that LCB from adjacent areas would recolonize waters in the BHNF is small given the distances to most potential source populations and the fragmentation of the intervening stream networks.

Early survey work indicates that MTS were widely distributed across the Black Hills. Recent stream surveys suggest that this species occurs in much, if not all, of its historic range. The only exception may be a contraction at the southern periphery of the MTS range in portions of the upper Cheyenne River. However, additional sampling is needed to confirm this possibility because limited sampling has occurred in this area. No trends in population densities were observed for four stream sites that were repeatedly sampled in the 1990s, suggesting stability within the core of the MTS range. The wide distribution and high abundance of MTS at many sites in the Black Hills, even after more than a century of intensive land use, suggests that current risks for this species are minimal. As a result, land uses and the attendant impacts to stream habitats would have to deviate strongly and on a forest-wide scale from historic and current norms before the existence of MTS populations in the Black Hills would be jeopardized.

Survey data suggest the historic distribution of FSD was limited to streams and spring-fed lakes and bog holes in the Redwater Creek drainage at the northern extent of the Black Hills. One population of FSD has been reported from Geis Reservoir immediately north of the Redwater Creek drainage, but it is not clear whether this occurrence is natural or the result of an introduction. Finescale dace have disappeared from four of the six sites where previous collections have been made in South Dakota and the trend for the species in this portion of its range is one of decline. In Wyoming, FSD have been sampled from four locales in lakes and reservoirs in recent years (1980s and 1990s), but uncertainty exists regarding the overall population trend in this portion of the range because little is known about the historic distribution. Although FSD have previously been collected from several streams on BHNF lands, these sites have not been resampled since the 1960s and current population status is unknown. As a result, the only known extant populations on the BHNF occur in Hemler Reservoir and upstream portions of Geis Reservoir. The greatest risk factor for FSD is the introduction of exotic predators—primarily members of the sunfish family—and such introductions are probably responsible for extirpations of two populations. Similar results would be expected for most remaining FSD populations because they occur in lentic habitats that are suitable for sunfish invasions. Hydrologic factors pose additional threats for FSD because large floods could breech

the small earthen dams that form Geis and Hemler reservoirs or lengthy droughts could dry these habitats. Additionally, populations in natural lakes and bog holes are dependent on spring flows, which are ultimately determined by aquifer levels and land uses that affect aquifer recharge. As a result, the general decrease in aquifer levels across the northern Black Hills may begin to negatively impact FSD populations at some future point. If populations of FSD continue to disappear, seasonal dewatering and other factors that fragment the Redwater Creek network will increasingly exacerbate population declines by inhibiting the dispersal of individuals that are needed to refound populations.

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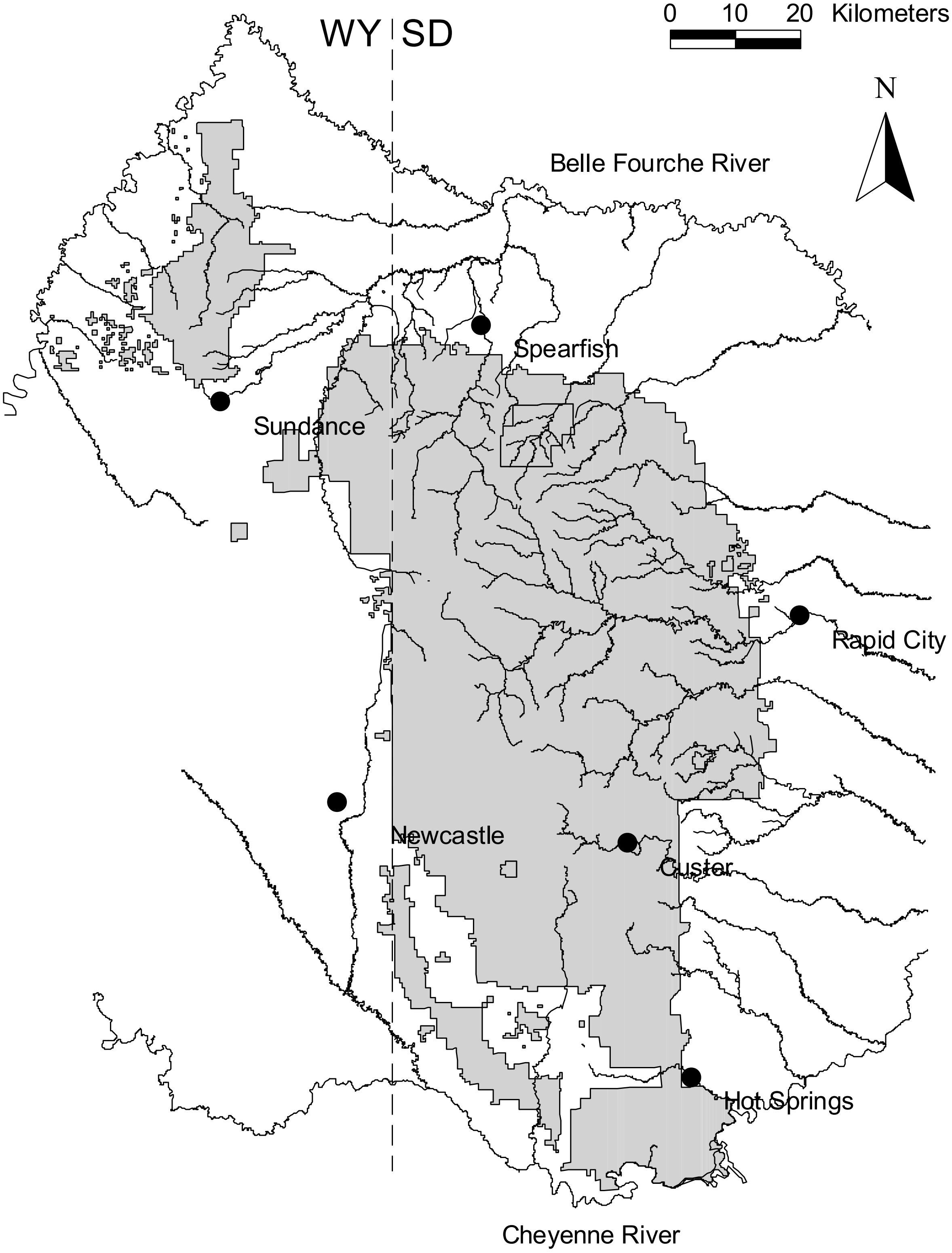
INTRODUCTION

This report assesses the biology and overall conservation status of lake chub (LCB; *Couesius plumbeus*), mountain sucker (MTS; *Catostomus platyrhynchus*), and finescale dace (FSD; *Phoxinus neogaeus*) in the Black Hills National Forest (BHNF; Figure 1). The purpose of the report is to provide managers and the public with an objective and thorough assessment of the status of these species. This assessment focuses on the biology and ecology of these fishes in the Black Hills, but discusses this information within the context of the species range and the base of knowledge that exists for each species. Factors that might limit each species are identified and a judgement is made regarding whether existing information is adequate to fully understand the biology and conservation status of the species on the BHNF. Lastly, this assessment assumes that the primary goal of management is the continued viability of the species discussed herein.

To the extent possible, the information in this report is based on peer-reviewed primary literature. However, few articles have been written that deal with LCB, MTS, or FSD in the Black Hills and knowledge regarding these species in the primary literature is limited. As a result, information sources subject to less rigorous review, such as master's theses, doctoral dissertations, and unpublished agency reports are sometimes used to provide basic biological information. Additionally, inference regarding the ecology of the fish species of concern is often extrapolated from other geographical areas—an approach that assumes the basic biology of the species is similar among areas.

The data used to assess the current distribution and population trends of LCB, MTS, and FSD on the BHNF are South Dakota Department of Game, Fish and Parks (SDGFP) stream data from 1964 (Stewart and Thilenious 1964), 1984 – 85 (Ford 1988), and 1992 – 98 (Meester 1993 – 1999), as well as lake data from 1994 – 1999 (Meester 2000) and Wyoming Game and Fish Department (WGFD) survey records contained in their Lakes and Streams Database. Earlier data sources—primarily Evermann and Cox (1896) and Bailey and Allum (1962)—are used to provide insight to the historic distribution of LCB, MTS, and FSD.

Figure 1. Stream drainage network of the Black Hills National Forest in South Dakota and Wyoming.



CURRENT MANAGEMENT SITUATION

Management Status

Finescale dace, MTS, and LCB are assigned a ranking of G5 by the Nature Conservancy—suggesting that the continued existence of each species across the entirety of their respective ranges is not at risk. However, heritage program designations vary among states and provinces within the ranges (Table 1). No special status is accorded FSD, MTS, or LCB by the U.S. Forest Service, U.S. Fish and Wildlife Service, or the Canadian government (Campbell 1997). The American Fisheries Society did not list any of these species as threatened, endangered, or of special concern in 1989 (Williams et al. 1989). On a less formal basis, the North Dakota Game and Fish Department (1994) compiled a fish species checklist for the Dakotas that was based on the expert opinion of local fisheries biologists. The status of species in the checklist differs from those in Table 1 in that MTS and LCB are listed as peripheral species in South Dakota and LCB are also considered a species of special concern. Similarly, the state of Washington considers MTS to be a candidate species for state listing (Mongillo and Hallock 1999).

Existing Management Plans, Assessments, Or Conservation Strategies

Species assessments, management plans, or conservation strategies written for FSD, LCB, or MTS are rare, but two documents of this type have been written for MTS.

Campbell, R.E. 1992. Status of the mountain sucker, Catostomus platyrhynchus, in Canada. The Canadian Field-Naturalist 106:27-35.

Decker, L.M. 1989. Coexistence of two species of sucker, *Catostomus*, in Sagehen Creek, California, and notes on their status in the western Lahontan Basin. Great Basin Naturalist 49:540-551.

A manuscript currently in press also discusses the status of MTS throughout North America.

Wydoski, R.G., and R.S. Wydoski. 2002. Age, growth, and reproduction of mountain sucker in Lost Creek Reservoir, Utah. Transactions of the American Fisheries Society 131:000-000.

Table 1. Heritage program rankings for LCB, MTS, and FSD across their respective ranges.

Items in parentheses describe special status designated by individual states. Information for heritage rankings was obtained from Stukel and Backlund (1997) and the Wyoming Natural Diversity Database (Fertig and Beauvais 1999). State fisheries biologists were contacted in the region surrounding the BHNF to determine state status.

	Region						
Species	**SD**	**WY**	**MT**	**ND**	**NE**	**Other states and provinces across species range**	**Overall**
LCB	S1[a]	S5	S5	S3	S1	Alaska-S4S5, Colorado-S1, Idaho-S1S2, Illinois-SU, Indiana-S2, Iowa-SX[b], Maine-S4, Massachusetts-S1, Michigan-S4, Minnesota-S?, New Hampshire-S4, New York-S3, Vermont-S4, Washington-SU, Wisconsin-S4; Alberta-S5, British Columbia-S5, Labrador-S3S4, Manitoba-S5, New Brunswick-S5, Northwest Territories-S?, Nova Scotia-S5, Nunavut-S?, Ontario-S5, Quebec-S5, Saskatchewan-S5, Yukon Territory-S?	G5[c]
MTS	S3	S5	S5	-	S1	California-S?, Colorado-S2?, Idaho-S5, Nevada-S?, Oregon-S4, Utah-S4, Washington-S3; Alberta-S4, British Columbia-S3, Saskatchewan-S3	G5
FSD	S1 (Endangered)	S1?[d]	-[e]	SU[d] (P/SC1)[f]	S2 (Threatened)	Maine-S4, Michigan-S5, Minnesota-S?, New Hampshire-S2, New York-S3, Vermont-S2?, Wisconsin-S4; Alberta-S2S3, British Columbia-SU, Manitoba-S5, New Brunswick-S5, Northwest Territories-S?, Ontario-S5, Quebec-S5, Saskatchewan-S5	G5

[a] S# = Nature Heritage Network state ranking. A 1 designates highest conservation concern, whereas 5 designates lowest conservation concern.
[b] G# = Nature Heritage Network global ranking. A 1 designates highest conservation concern, whereas 5 designates lowest conservation concern.
[c] Extirpated from the state.
[d] Status uncertain due to lack of information.
[e] Pure populations of finescale dace do not exist within Montana, but finescale dace and northern redbelly dace (*Phoxinus eos*) hybrids are listed as a fish of special concern.
[f] P = species at the periphery of its range; SC1 = species whose status is questioned due to suspected problems with abundance or distribution.

LAKE CHUB

REVIEW OF TECHNICAL KNOWLEDGE

Systematics

Lake chubs were originally described by Agassiz (1850) as *Gobio plumbeus* from a type specimen collected in Lake Superior. Evermann (1893a) later described the same species as *Couesius dissimilis* based on a specimen collected from Rapid Creek near Rapid City, South Dakota. The range of the lake chub was subsequently split into three parts and LCB in the Black Hills and an area extending from the upper peninsula of Michigan west to the Rocky Mountains and south of the Canadian border were considered the subspecies *C. plumbeus dissimilis*. The *Couesius* genus was later merged with the *Hybopsis* genus (Bailey 1951); at which time the subspecies designation became unavailable because it was already occupied within *Hybopsis*. The *Couesius* genus has subsequently been restored (McPhail and Lindsey 1970) and *C. plumbeus* is the contemporary designation for LCB. The status of possible subspecies has not been subject to detailed study, but subspecific designations are not currently used (Lindsey 1956).

Distribution And Abundance

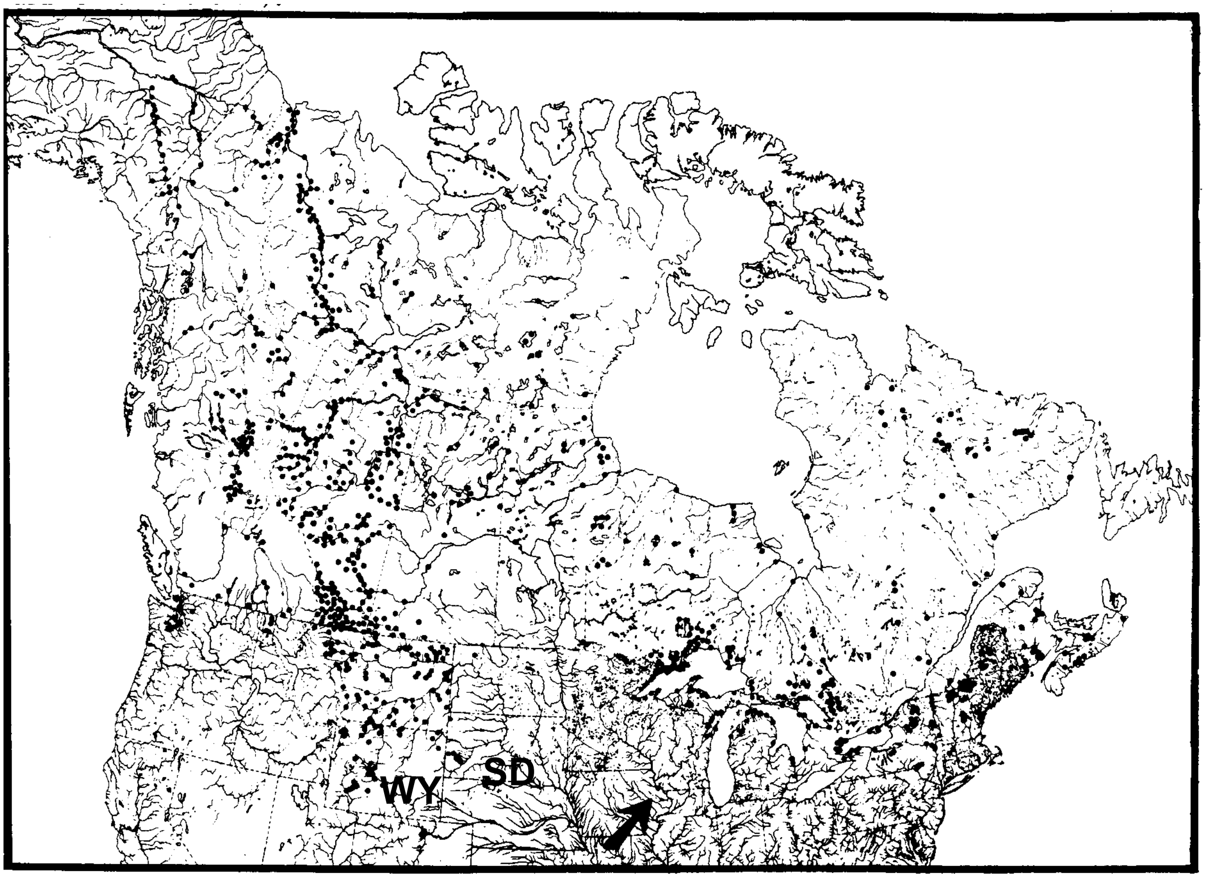
Distribution Of LCB Recognized In The Primary Literature.

The LCB is widely distributed across Canada and northern portions of the U.S. (Figure 2). Lake chubs occur in upper portions of the Columbia, Yukon, and Missouri river drainages, as well as the Peace and Fraser rivers in the western portion of the range (Scott and Crossman 1973; Wells 1980). In the eastern portion of the range, LCB occur throughout the Mackenzie and Hudson Bay drainages, the Great Lakes region, and northern Atlantic slope drainages south to the upper Delaware River (Wells 1980).

Disjunct populations of LCB occur at the southern periphery of the range; where preglacial drainage patterns allowed LCB to colonize these areas (Lemke et al. 1965). The southern portion of the range served as a refugium during the peak of Pleistocene glaciation (Briggs 1986), but glaciation also rearranged drainage networks and stranded relict populations of LCB in Iowa (Bailey 1956), northern Nebraska (Stasiak 1986), northern Colorado (Ellis 1914; Bestgen et al. 1991), and the Black Hills (Bailey and Allum 1962).

At a regional scale (i.e., states adjacent to the Black Hills), LCB occurred in the Little Missouri drainage of southwest North Dakota (Reigh and Owen 1978) and throughout tributaries to the Yellowstone River in southeast Montana as late as the 1970's (Rehwinkel 1978; Elser et al. 1980). In recent surveys of Wyoming, LCB were found across the Bighorn and Tongue drainages in the north-central part of the state (Patton 1997) and the Green River drainage in southwest Wyoming (Wheeler 1997). To the south and east of the Black Hills, populations appear to be small and limited to isolated collection sites (Bailey 1956; Stasiak 1986; Bestgen et al. 1991).

Figure 2. Distribution of LCB across North America. Modified from Wells (1980).



Historic And Present Distribution In The Black Hills

Historic accounts suggest LCB were widely distributed across the Black Hills. Early surveys by Evermann and Cox (1896) documented the presence of LCB in Rapid Creek, Spring Creek, French Creek, and Beaver Creek (Figure 3). Collection activities summarized in Bailey and Allum (1962) confirmed the presence of LCB in many of these streams into the late 1920s and early 1930s and also documented LCB in Grace Coolidge Creek and the South Fork of Castle Creek. In Wyoming, LCB have been reported from the Belle Fourche River, Blacktail Reservoir, and Inyan Kara Creek (Eisermann 1966; Patton 1997; WGFD Lakes and Streams Database).

Figure 3. Recent (squares) and historic (circles) collection sites for LCB in the Black Hills. Dates indicate the most recent year that LCB were collected from a water body. Collection site information was derived from Evermann and Cox (1896), Bailey and Allum (1962), Eisermann (1966), Patton (1997), Meester (2000), and WGFD Lake and Stream Database.

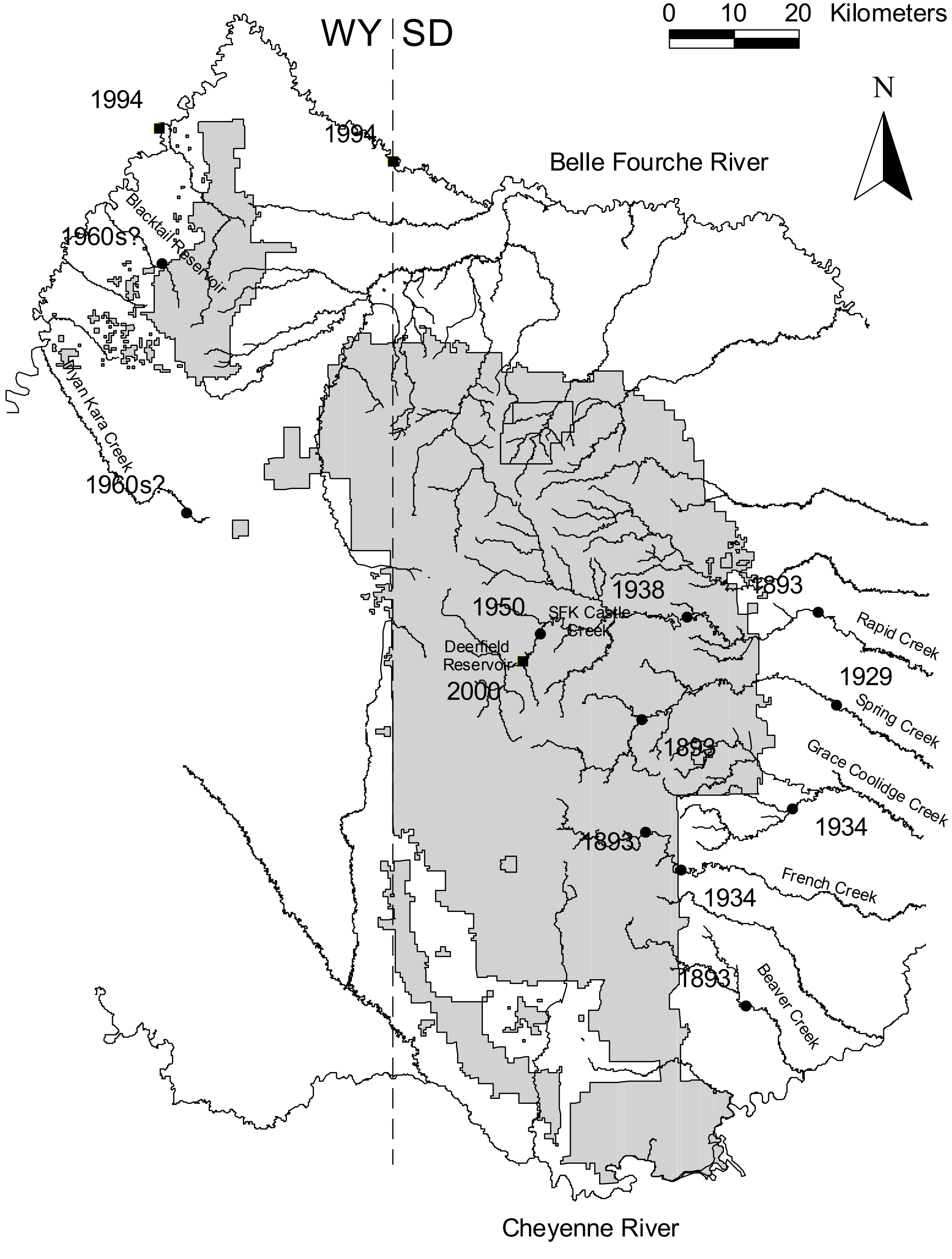
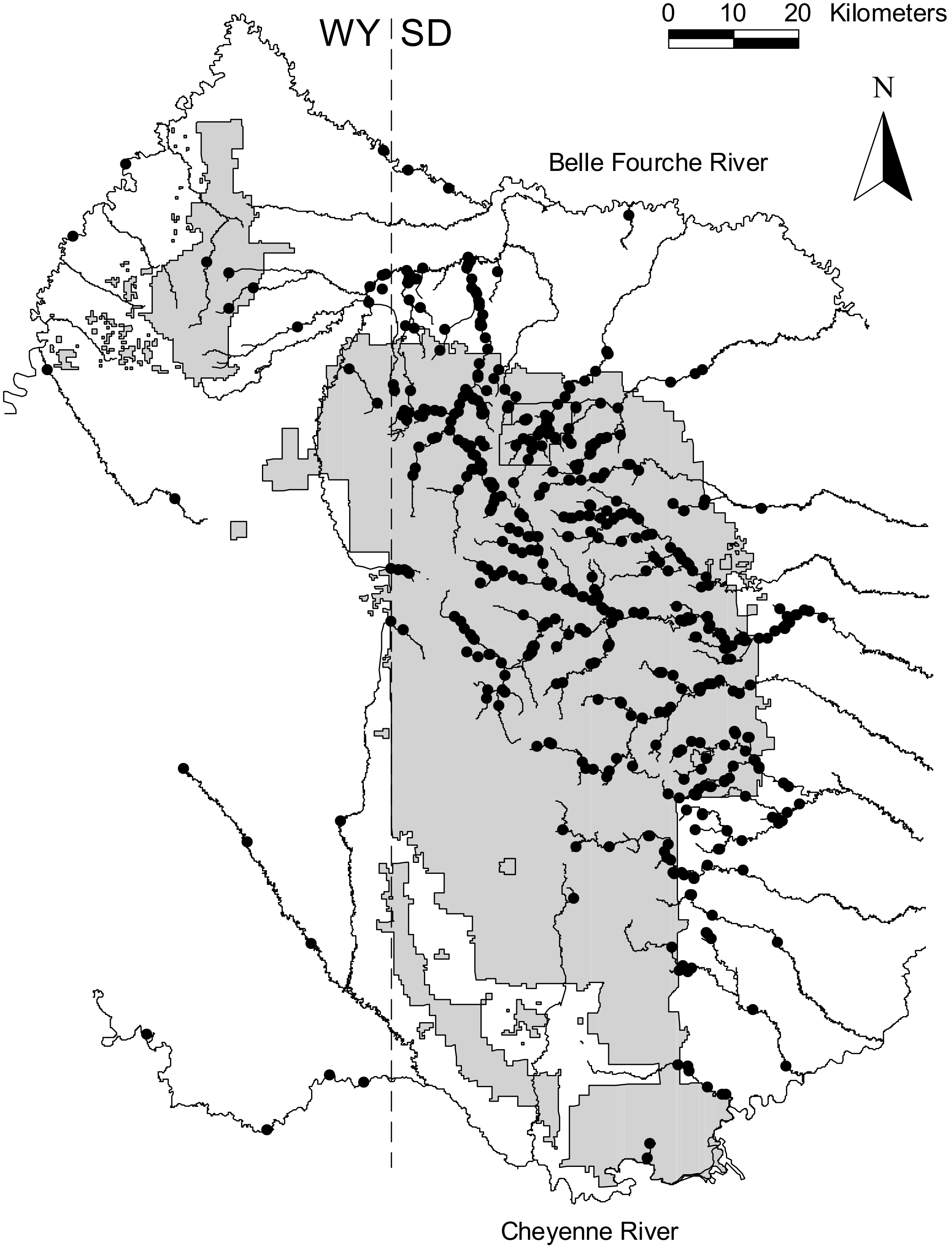


Figure 4. Stream sites sampled in the Black Hills of South Dakota and Wyoming in the 1980s and 1990s. Most sites were sampled by SDGFP personnel (Ford 1988; Meester 1993 – 1999), but sites sampled by Patton (1997), Olson (1997, 1998), Doorenbos (1998), Newman et al. (1999), and the WGFD are also included.

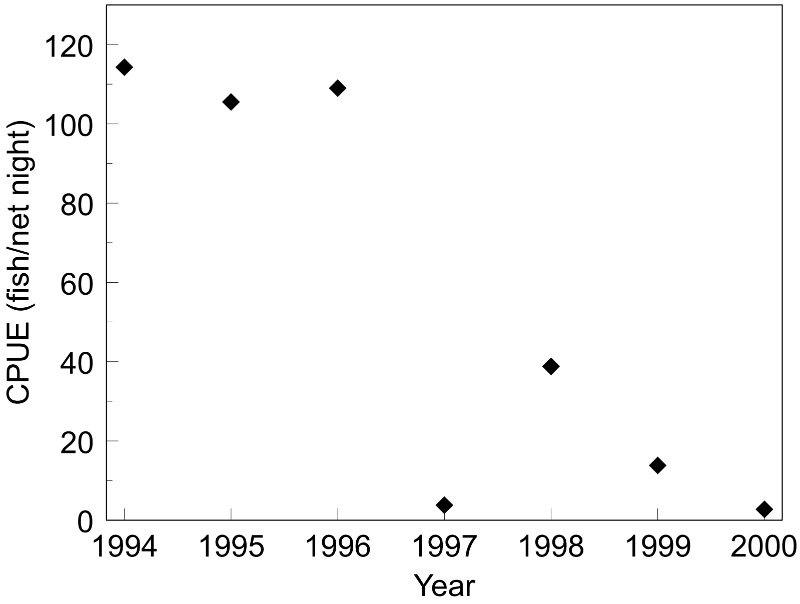


The contemporary distribution of LCB on and near BHNF lands appears to be much reduced. In the last two decades, the fish has been sampled only from three locations (Figure 3), despite extensive surveys (Figure 4). Additionally, the only remaining sizeable population may be in Deerfield Reservoir—where the fish is commonly collected during yearly surveys (Meester 2000). Occurrences of LCB in the Belle Fourche River consisted of small numbers of individuals (T. Patton, Southeast Oklahoma State University, personal communication), which may have emigrated from an upstream reservoir. Blacktail Reservoir is the most likely candidate because LCB have been collected previously from this locale (WGFD Lakes and Streams Database) and Keyhole Reservoir—also located upstream on the Belle Fourche River—does not contain a population (B. Bradshaw, WGFD, personal communication).

Estimates Of Local Abundance

Abundance estimates derived from gill net Catch Per Unit Effort (CPUE) data exist for LCB in Deerfield Reservoir (Meester 2000). These data have been collected since 1994 using a standardized sampling methodology (four overnight gill net sets using 50-m experimental nets with mesh sizes ranging from 12.7 to 50.8 mm) and provide a yearly index of relative population size. Lake chub CPUE for the seven year period from 1994 through 2000 has ranged from 2.8 fish/net night to 114.3 fish/net night, with an average of 55.4 fish/net night and a median of 38.8 fish/net night (Figure 5). It is unknown whether these numbers represent high or low population densities because comparable abundance estimates are not available from other LCB populations.

Figure 5. Catch per unit effort data for the LCB population in Deerfield Reservoir. Data were obtained by SDGFP personnel using standardized gill-net sets (Meester 2000).



Population Trend

Although no probabilistic statement can be made regarding the apparent absence of LCB from much of its historic range, the extensive sampling conducted in South Dakota in recent decades makes it unlikely that a LCB population could have escaped detection. As a result, it is probably safe to conclude that LCB are extirpated from four of the five drainages in South Dakota where they previously occurred. In the remaining drainage—Castle Creek—the LCB no longer occurs in the stream, but is limited to Deerfield Reservoir. In the Wyoming portion of the Black Hills, sampling has not been as intense and population status is less clear. However, even if LCB remain at all the previous collection sites in Wyoming, their absence from most of the South Dakota range strongly suggests that LCB have declined in abundance since European settlement of the Black Hills. Similarly, and in an area adjacent to the Black Hills, Patton (1997) also concluded that LCB had declined in abundance since surveys conducted earlier in the 20th century in eastern Wyoming.

Within Deerfield Reservoir, the population trend is also one of decline because LCB catch rates from 1997 through 2000 were approximately one-sixth the catch rates from 1994 through 1996 (Figure 5). Additionally, the slope of the best-fit regression line through these data was negative and significantly different from zero ($B_1 = -21.0$; $p = 0.01$). The factors responsible for the decline of LCB in Deerfield Reservoir are unknown (J. Erickson, SDGFP, personal communication), but two facts regarding this pattern are noteworthy. First, the decline does not appear to be ongoing, but instead may consist of two distinct levels, within which population trends are stable. Second, the current population size, even if reduced from 1994 – 1996 populations, is probably greater than that which occurred immediately after Deerfield Reservoir was chemically renovated to reduce the white sucker population (*Catostomus commersoni*) in 1982 (Vanderbush 1999).

Movements

No studies have been conducted on the dispersal/colonization movements of LCB. The only information that suggests LCB occasionally make substantial movements is derived from observations of spawning migrations—typically made by lake populations into streams (Brown et al. 1970; Bruce and Parsons 1976), but sometimes by river populations into smaller streams (Reebs et al. 1995). Of these studies, only Brown et al. (1970) marked fish in successive years and was able to provide crude estimates of distance traveled. Most LCB were recaptured near the original capture site, but some moved 4 km through a lake to spawn in a tributary stream. However, this estimate of distance traveled is likely to be conservative, given that LCB were marked in only a small part of the lake near a primary spawning stream.

Even if spawning migrations demonstrate some potential for dispersal, it is unlikely that Black Hills populations of LCB are linked to other populations. The nearest populations occur in central Montana and Wyoming, the little Missouri River, northwest Nebraska, and central South Dakota. In the latter two cases, it is unknown whether these populations continue to persist. In all cases, connection to other populations is precluded by the length and fragmentation of the intervening stream network.

Habitat Associations

Habitat preferences of LCB vary across the range. Lake chubs live in large rivers at the northern extent of their range (Scott and Crossman 1973), but are often found in lake habitat when it is available. In the southern portion of the range, LCB are found less often in lakes, but whether this is due to a difference in habitat preference or a decreased abundance of lakes is not known. As examples, Simpson (1941) remarked that LCB occurring in Wyoming are found most abundantly in streams, whereas Brown (1971) suggested that LCB in Montana preferred small streams at lower elevations and rarely occurred in larger rivers. Wherever LCB occur, the common denominator is generally clear, cool water with clean cobble or gravel substrates (Bruce and Parsons 1976; Becker 1983; Patton 1997), although the fish have occasionally been collected in turbid waters over fine substrates (R. Bramblett, Montana State University, personal communication). Unfortunately, no studies have been conducted to determine the microhabitat preferences of LCB in streams and more specific statements regarding habitat use are impossible to make.

In lake environments during the day, Emery (1973) has observed LCB actively schooling near the bottom, although cover was not used. During twilight and night periods, LCB were inactive, schooling behavior broke down, and fish rested on the lake bottom. Scott and Crossman (1973) cited evidence from gill netting catch rates to suggest that LCB move from shallow water to deep water in summer.

Food Habits

Davis and Miller (1967) classified LCB as obligatory sight feeders based on the absence of external taste buds and well developed optic lobes in the brain. Simpson (1941) considered LCB to be carnivorous based on a relatively short gut. Dietary analyses have supported these positions (Simpson 1941; McPhail and Lindsey 1970) and suggest a diet composed of mobile aquatic and terrestrial insects, as well as several types of zooplankton (e.g., chironomids, cladocerans, and copepods), although large individuals have also been know to eat small fish (McPhail and Lindsey 1970). In the most detailed analysis of LCB diets available, Simpson (1941) examined the gut contents of 11 fish from Wyoming and found 97.5% animal matter by volume. The remainder of the stomach contents was composed of organic debris, sand, and gravel—materials that were probably consumed inadvertently while ingesting prey items. Five common orders of aquatic insects composed the diet of these LCB—Odonata, Plecoptera, Trichoptera, Diptera, and Coleoptera—with Plecopterans and Trichopterans making up almost 70% of the diet. The fish that Simpson (1941) examined were collected from the Sweetwater River and are reflective of a lotic environment. It could be expected that diets of lake-dwelling LCB would differ substantially and have relatively small proportions of aquatic insects and larger proportions of zooplankton.

Demography

Lake chubs are spring spawners and usually breed in streams (Scott and Crossman 1973), although instances of spawning along lake shores or over shallow shoals have been documented (Geen 1955; Brown et al. 1970). Brown et al. (1970) noted LCB staging in a Saskatchewan lake near a major spawning tributary shortly after ice melt in early May. After two weeks, LCB moved into the tributary and commenced spawning in water temperatures that were in the midteens (°C). Brown et al. (1970) also observed that spawning did not occur in the lake until several weeks later when surface water temperatures reached 10°C. Other populations of LCB have been reported to spawn as early as April in the Great Lakes region (Scott 1954) and as late as August in British Columbia and the Northwest Territories (Dymond et al. 1929; McPhail and Lindsey 1970). The wide variation in spawning time probably results from geographic differences in thermal regimes because water temperature appears to be the most important factor affecting the developmental rates of reproductive tissues (Ahsan 1966).

Spawning involves the movement of LCB into shallow areas along the margins of a stream or a lake shore. Onshore movements begin in the morning, peak in the afternoon, and taper off in the evening (Brown et al. 1970). Individuals do not build nests or defend territories. Instead, several males chase individual females and attempt to pin them against rocks. The males then vibrate vigorously against the females—causing the release of a small number of eggs, which are then fertilized (Brown et al. 1970). This act lasts less than a second, but is repeated several times in short succession. After release, the nonadhesive eggs sink to the substrate and receive no parental care (Fujiman and Baker 1981). To the contrary, both sexes have been observed eating freshly deposited eggs (Brown et al. 1970). Richardson (1935) observed large numbers of dead LCB after spawning, which suggests that postspawning mortality may be common.

Males remain on the spawning grounds for a longer period of time than females, which accounts for the skewed sex ratios that have been observed (Brown et al. 1970; Bruce and Parsons 1976). Most spawning females are age three (67%), but four- and five-year-old fish also participate (Brown et al. 1970; Bruce and Parsons 1976). Fecundity of females from a Labrador lake averaged 914 eggs, with a range of 214 – 1,540 (Bruce and Parsons 1976). Fecundity levels are positively correlated with size, however, as indicated by the higher range (650 – 2,500) that Brown et al. (1970) observed in larger fish sampled from Saskatchewan. Eggs and oocytes have been observed in the same female, which suggests that spawning in consecutive years is a possibility.

Lake chubs hatch 10 days after fertilization when held at water temperatures between 8° and 19° C (Brown et al. 1970). Newly hatched fry range in length from 5.8 to 6.4 mm and begin feeding after 7 days when the yolk is fully absorbed (Fujiman and Baker 1981). Females grow faster and live longer than males—sometimes reaching age four (Scott and Crossman 1973). Most LCB become sexually mature at age three, although some males mature at age two (Bruce and Parsons 1976) and some females do not mature until the fourth year (Brown et al. 1970). Several authors have provided data that suggest lake populations grow more slowly than stream populations (Rawson and Elsey 1949; Brown 1971; Bruce and Parsons 1976), although the largest recorded length for a LCB is 22.7 cm for a fish from a lake in Quebec (Scott and Crossman 1973).

Sexual dimorphism peaks during the breeding season and males develop reddish-orange coloration at the bases of pectoral fins, above and behind the operculum, and at the corners of the mouth. Larger males may also develop coloration at the bases of the pelvic fins (Brown et al. 1970; Baxter and Stone 1995). Males form nuptial tubercles on all portions of the head, pectoral fin rays, and the back from the nape to the origin of the dorsal fin, but less often on pelvic fin rays (Scott and Crossman 1973). Females also exhibit coloration and become tuberculate, but these developments are less pronounced than in males.

Community Ecology

The role of LCB within a community context can only be speculated because interactions with other species have not been the subject of detailed study. However, predation by northern pike (*Esox lucius*) and various bird species on LCB has been observed and this fish is probably an important prey item where abundant (White 1953; White 1957; Scott and Crossman 1973). Reebs et al. (1995) suggested that LCB were more active at night and speculated that this behavior occurred as an adaptation to avoid predation by birds.

Risk Factors

Given the restricted distribution of LCB in the Black Hills, stochastic events currently pose the greatest threat to the continued persistence of this species. Because LCB are concentrated in Deerfield Reservoir, an unusually large flood that might compromise Deerfield Dam or a severe and extended drought could effectively extirpate this population. Similarly, a planned chemical renovation of Deerfield Reservoir in 2003 to reduce the white sucker population (Vanderbush 1999) could extirpate most of the LCB remaining in the Black Hills. The negative effects of such events, if they occur, would be magnified by the absence of nearby populations that could otherwise provide some resiliency by providing sources for recolonization.

A factor of secondary, but immediate concern, is the presence of non-native fish predators in Deerfield Reservoir. Several species of salmonids are regularly stocked in this body of water (Meester 2000) and frequently reach sizes at which piscivory becomes a dietary option (e.g., > 250 mm; East and Magnan 1991). Salmonids and LCB, however, have coexisted in Deerfield Reservoir for some time and the recent establishment of rock bass (*Ampbloplites rupestris*) may be a greater cause for concern. Although an earlier invasion by another littoral predator, largemouth bass (*Micropterus salmoides*), appears to have failed, available data suggest that the rock bass population is increasing (Meester 2000). If rock bass become numerous and prove to be a significant predator of LCB, a serious threat could be posed to the LCB population.

Water quality issues may also be of concern in Deerfield Reservoir. Ongoing studies have documented low oxygen levels in late summer near the bottom of the reservoir (S. Chipps, South Dakota State University, personal communication). This condition results from the decomposition of organic materials in the lake substrates and effectively reduces the amount of habitat available to LCB. As Deerfield Reservoir ages, continued organic loading can be expected to exacerbate this trend. Finally, if the remaining population of LCB is small enough (e.g., less than 500 individuals), inbreeding depression could begin to have deleterious effects (Soule 1980; Nelson and Soule 1987). Without estimates of population size, however, it is difficult to know how important genetic considerations are at the present time.

Response To Habitat Changes

Relationships between LCB and land management activities or natural disturbance factors have not been studied. As a result, the discussion that follows is by necessity general and based on indirect evidence. Further, this discussion addresses issues of habitat change relative to the LCB population in Deerfield Reservoir as this population probably comprises the vast majority of LCB found on or near BHNF lands. For a summary of how habitat change could affect stream fish populations, see the section on MTS (pages 45 – 52).

Management Activities

Timber Harvest, Grazing, And Prescribed Fire

Timber harvest, grazing, and prescribed fires are likely to have similar effects on LCB in Deerfield Reservoir. The main effect is expected to be accelerated rates at which sediment is transported to and through the stream network that drains into the reservoir. In the case of timber harvest, greater erosion rates stem mainly from the system of roads that are often built to facilitate the removal of timber because sediments are derived from the road surface and road cuts adjacent to the road (Furniss et al. 1991). Livestock grazing can increase sediment delivery to streams when animals collapse banks into streams or remove riparian vegetation that maintains structural integrity. Grazing also makes sediments in upslope areas more susceptible to erosion through the removal of near-ground vegetation (Platts 1991; Belsky et al. 1999). It is this latter mechanism that is responsible for the increased erosion rates associated with fires, although fires may exacerbate the erosive response by removing canopy cover that would otherwise reduce the force of raindrops striking the ground (Gresswell 1999).

Sedimentation rates stemming from grazing, timber harvest, and fires can be minimized using best management practices, but are still expected to increase as the proportion of a watershed subject to these activities increases. Over time, sediments from any or all of these management activities in the Deerfield Reservoir watershed will increase deposition within the reservoir. This deposition will gradually reduce the amount of habitat available for LCB and, at a long time scale (e.g., decades to centuries), could begin to threaten population persistence if steps are not taken to decrease infilling of the reservoir (e.g., dredging).

Recreation

The most important effects of recreation on LCB result from recreational angling and are twofold. First, this activity creates a demand for the introduction, maintenance, and supplementation of non-native salmonid sportfish populations within Deerfield Reservoir. Many of these fishes reach sizes at which piscivorous diets are adopted and thereby become potential predators of LCB. Second, recreational anglers were probably responsible for the illegal introductions of two non-native littoral predators to Deerfield Reservoir. Largemouth bass were introduced in the early 1990s, but a self-perpetuating population did not become established. More recently, rock bass were introduced and appear to have become established (Meester 2000). Data are lacking to determine what effects that rock bass or salmonids may be having on LCB, but we can envision no scenario in which such effects would be positive.

Mining

Mining currently has little relevance for the LCB population in Deerfield Reservoir as it does not constitute a major land-use activity in the upstream watershed (BHNF 1996, III 441-443).

Water Impoundment And Diversion

Water development in the Black Hills has probably had both positive and negative effects on LCB. In the middle of the 20th century, the construction of many dams and diversion structures and continued removal of beaver dams coincided with the disappearance of LCB from streams in the Black Hills (Parrish et al. 1996). The removal of beaver dams probably greatly decreased the diversity of physical habitats found in BHNF streams (Olson and Hubert 1994), whereas the artificial dams and diversion structures probably fragmented existing populations and prevented the types of movements that many stream fishes make in association with spawning, overwintering, or refounding extirpated populations (Gowan et al. 1994). Additionally, hydrograph changes downstream from large reservoirs may have modified environmental cues that stimulate spawning or altered critical habitat features (reviewed in Poff et al. 1997).

Given the current status of LCB in the Black Hills, the effect of water impoundment should not be viewed entirely negatively because without the population in Deerfield Reservoir, LCB would not exist on BHNF lands. Further, given the apparent inability of BHNF streams to support LCB populations, reservoirs with similar characteristics to Deerfield Reservoir may provide the best opportunities for expanding the range of LCB in the short term.

Natural Disturbances

Floods And Droughts

Under natural conditions, hydrologic variability related either to floods or drought would pose little threat to LCB. However, the restricted distribution of LCB in the Black Hills makes this species vulnerable to extreme hydrologic events. An exceptionally large flood could breech Deerfield Dam or an extended drought could empty the reservoir. Although either event is unlikely to occur in any one year, probabilities increase over longer time frames. Additionally, land management activities of the types (e.g., timber harvest/road building, grazing) that occur in the watershed upstream from Deerfield Reservoir often tend to exacerbate hydrologic extremes—resulting in higher peak flows and lower summer baseflows (Chamberlin et al. 1991; Furniss et al. 1991; Jones and Grant 1996).

Wildfire

For the sake of discussion, we assume wildfire to mean a large, uncontrolled burn that covers all or a significant portion of the watershed above Deerfield Reservoir. The effect of such an event on the LCB population could vary widely and would depend on events that occurred after the fire. If a major precipitation event happened shortly thereafter, significant amounts of sediments could move into Deerfield Reservoir (Swanston 1991). The flood peak associated with such an event would also be large and could threaten Deerfield Dam. Depending on the amount of sediment moving into the reservoir and the number of tributaries affected, water quality could deteriorate rapidly and cause fish mortalities. If a major precipitation event did not follow a wildfire, vegetative regrowth would gradually stabilize the watershed, but larger water and sediment yields could be expected during the ensuing years (Gresswell 1999).

SUMMARY

The basic ecology of LCB has not been studied on the BHNF and is poorly understood in general. Details regarding habitat requirements are limited to crude considerations such as the occurrence of this species in lakes and streams that usually have cool waters and clean gravel or cobble substrates (Bruce and Parsons 1976; Becker 1983; Patton 1997). Lake chubs are site-feeders that subsist on a diet of small aquatic animals—mainly zooplankton in lentic environments and aquatic insects in lotic environments (Simpson 1941; McPhail and Lindsey 1970). Breeding occurs in the spring and lake populations may make short migrations into tributaries for this purpose, although lake spawning has also been documented (Brown et al. 1970; Scott and Crossman 1973). Spawning occurs in shallow waters over cobble substrates and eggs are given no parental care (Brown et al. 1970). Fry hatch approximately 10 days after fertilization and most LCB mature sexually at age three (Bruce and Parsons 1976). Almost nothing is known regarding the role of LCB within a community context, but predation by various fish and bird species has been observed and LCB may be an important prey item where it occurs in abundance (White 1957; Scott and Crossman 1973).

Lake chubs are widely distributed across Canada and portions of the U.S.—especially the New England states, Montana, and Wyoming. Disjunct populations, including those in the Black Hills, are common near the southern extent of the range and are believed to have become isolated after the last glaciation (Bailey and Allum 1962). Although little is known regarding the dispersal ability of LCB, it is unlikely that Black Hills populations interact with other populations given the spatial arrangement of intervening stream networks and the fragmentation of these networks by dams and diversions.

Pre-settlement accounts suggest that LCB were widely distributed in streams across the Black Hills (Evermann and Cox 1896) and this remained true into the 1920s and 1930s (Bailey and Allum 1962). Recently, LCB have disappeared from most of the Black Hills and can now be found only in Deerfield Reservoir and portions of the Belle Fourche River in Wyoming. The reasons for the decline are unknown, but given the relatively pristine habitats often preferred by LCB, degraded stream habitats are probably a factor. Construction of dams and water diversions on many Black Hills streams during the mid-20th century may have exacerbated declines in LCB populations, but Deerfield Reservoir now supports the only population remaining on the BHNF.

Trend data collected by SDGFP since 1994 suggest the population in Deerfield Reservoir has declined in recent years, but two facts are noteworthy. First, the decline does not appear to be ongoing, and the population may have stabilized at a lower level. Second, the current population size, even if reduced, is probably greater than that which occurred immediately after Deerfield Reservoir was chemically renovated in 1982 to reduce the white sucker population (Vanderbush 1999). Regardless, the currently restricted distribution of LCB in the Black Hills places this species in a precarious position. A stochastic event such as a dam failure or the planned chemical renovation of Deerfield Reservoir in 2003 (Vanderbush 1999) could be catastrophic and would threaten the continued existence of LCB on the BHNF. Another cause for concern is the recent establishment of rock bass in Deerfield Reservoir. This population appears to be increasing (Meester 2000) and may come to pose a significant threat to the LCB population. Lastly, the deposition of sediments in Deerfield Reservoir is an ongoing process that will gradually decrease the amount of LCB habitat provided by the reservoir.

REVIEW OF CONSERVATION PRACTICES

Management Practices

We are not aware of any management practices that have been implemented specifically for LCB in the Black Hills or elsewhere in North America.

Models

No models have been developed for LCB or LCB habitat.

Survey And Inventory Approaches (Presence/Absence)

Developing a sampling protocol that provides a probabilistic statement regarding the presence or absence of LCB in a lentic water body would be difficult due to the relative homogeneity of habitats and a dearth of information on LCB habitat. As a result, a survey protocol for LCB should be robust and include several techniques so that both near-shore, littoral waters and pelagic waters are sampled effectively. Unfortunately, the peer-reviewed literature provides little guidance to suggest which sampling techniques would be most effective for LCB. However, data collection activities summarized in SDGFP reports (Meester 2000) suggest that LCB can be sampled in littoral habitats using beach seines and boat electrofishing. Passive capture techniques using fyke nets or trap nets could also prove effective if LCB make extensive near-shore movements (Hubert 1996), but these approaches have not been attempted to the best of our knowledge. In pelagic waters, overnight gill-net sets have proven to be effective (Meester 2000), but this technique has the disadvantage of killing most of the captured fish. In lakes and reservoirs of sufficient size, mortalities could be minimized by sampling with trawls or purse seines (Hayes et al. 1996), but these techniques, as well as boat electrofishing, have the disadvantages of requiring highly trained personnel and significant capital investments in specialized boats and other equipment.

Monitoring Approaches (Population Trend, Habitat)

Population estimates for LCB could be obtained by using any of the techniques described in the previous section with a mark-recapture estimator (Ricker 1975). In reality, however, estimates of population size for lakes are rare except when the water body is small and capture efficiencies are high. Most often, population trends in lentic waters are monitored using an index based on CPUE data. To develop a reliable index, any technique that has proven effective at sampling LCB could be used, as long as the sampling design is standardized so that data are comparable among samples. Typically, standardization involves sampling the same sites at the same time of year. The major assumption inherent to this approach is that catch rates are strongly correlated with population size. The validity of this assumption varies, but increases as the intensity and effectiveness with which a target population is sampled increases.

For future efforts to monitor LCB populations, continuation of a monitoring protocol identical to the annual standardized gill-net sets that the SDGFP uses is advised. This approach will ensure the comparability of new data and existing data that is necessary to provide credible trend assessments. Ultimately, however, it will be desirable to have a monitoring protocol that is not a source of mortality for the Deerfield LCB population. Therefore, we suggest that a standardized nonlethal monitoring protocol that relies on either (or both) boat electrofishing or beach seining be implemented in the near term and that these efforts be conducted coincidently with standardized gill-net sets for an interim period of 4 – 6 years. Data from this interim period can then be used to determine how well the nonlethal monitoring protocol tracks the Deerfield LCB population (by comparison to gill-net data). Once confidence exists in the nonlethal monitoring protocol, the use of gill-nets in Deerfield Reservoir should be discontinued or altered so as to no longer cause LCB mortality.

Monitoring of LCB habitat is difficult without a better knowledge of what constitutes LCB habitat. Biological attributes of the species, however, suggest two possibilities that may provide insight to population trends. Many studies of LCB suggest that reproduction often occurs in small streams (Brown et al. 1970; Bruce and Parsons 1976). If these areas could be identified, stream sampling protocols (e.g., Platts et al. 1983; Simonson et al. 1994) could be implemented to monitor trends in habitat condition. Important variables would likely be related to substrate conditions or dissolved oxygen levels, which are both important determinants of reproductive success in other stream dwelling fishes (Berkman and Rabeni 1987; Chapman 1988). More detailed stream habitat assessment procedures are discussed in association with monitoring MTS populations.

Within lake physicochemical measures such as chlorophyll a, water clarity, or the morphoedaphic index (measure of lake productivity based on ratio of total dissolved solids to mean lake depth) could provide useful insights about the productivity of Deerfield Reservoir (McMahon et al. 1996). A more direct approach might focus on the food resources of LCB, which would probably be zooplankton in a lentic environment. Trends in the lake variables deemed relevant to LCB populations could be coordinated with a LCB sampling protocol to reduce additional field effort, but most samples would require laboratory processing after collection.

ADDITIONAL INFORMATION NEEDS

Additional data needed for a comprehensive LCB conservation strategy in the Black Hills are summarized in Table 2. One of the most basic needs is comprehensive determination of all extant populations of LCB. Data from recent stream surveys in South Dakota are probably sufficient to conclude that Deerfield Reservoir hosts the only remaining population. In Wyoming, however, stream sampling has not been as intense and less is known about LCB populations. Better information regarding Wyoming populations (e.g., population locations, fluvial or adfluvial life histories) would allow the focus of conservation efforts to be allocated accordingly. Additionally, if it were determined that LCB in Wyoming were fluvial fish, these populations would be a valuable source for possible reintroduction efforts across the Black Hills where populations historically had fluvial life histories.

Table 2. Additional information needed to support the development of a comprehensive conservation plan for LCB in the Black Hills.

Data	Priority	Justification	Cost[*]
Distribution in Wyoming portion of Black Hills	High	Determine the size, extent, and location of additional LCB populations	Low
Potential reintroduction sites	High	Provide a basis for future expansion of the LCB distribution in the Black Hills	Low
Trend monitoring	High	Allow assessment of population status and inference regarding factors that affect abundance	Low - intermediate
Predator food habits	High	Determine the effect that predators are having on LCB	Low
Habitat associations	Intermediate	Provide an understanding of the habitat attributes that promote the existence of LCB populations	Intermediate

[*] Low: estimated cost = $1,000 – 10,000; intermediate: estimated cost = $10,000 – 100,000; high: estimated cost > $100,000

Given the precarious state of LCB in the Black Hills, any conservation strategy must begin to identify suitable reintroduction sites. Detailed information about the habitat attributes required by LCB would be valuable in this regard, but would take considerable time and effort to collect. As a more rapid alternative, biologists familiar with the Black Hills could develop a list of potential reintroduction sites based on expert opinion. This information would serve as a valuable first step toward the conservation of this species and might initially be composed of both lake and reservoir sites as these areas appear to have some capacity for supporting populations of LCB. The list of criteria that might be used to screen and rank potential reintroduction sites could include the intensity of management activities within a basin (present and future), presence/absence of non-native predators or competitors, habitat quality, habitat size, and location relative to other LCB populations. If at some future point, multiple lentic populations were established and LCB are less at risk, a similar screening and ranking process could be repeated to identify stream reintroduction sites that could ultimately be used to restore LCB to what appears to have constituted their native habitat in the Black Hills.

Information regarding trends in the LCB population in Deerfield Reservoir will continue to be useful and the population should be monitored in the near term using a standardized gill-net protocol similar to the one that SDGFP has used since 1994. Nonlethal sampling techniques such as beach seining or boat electrofishing should also be implemented over an interim period until confidence exists as to their adequacy, at which point the use of gill nets should be discontinued or altered so as to no longer cause LCB mortality. The food habits of rock bass and large salmonids within Deerfield Reservoir also need to be examined to determine whether these predators are impacting the LCB population. If significant impacts are detected, this information could be used as a rationale for predator control or to refine a list of potential reintroduction sites.

MOUNTAIN SUCKER

REVIEW OF TECHNICAL KNOWLEDGE

Systematics

Mountain suckers were first described from a specimen obtained in the Green River drainage of Wyoming as *Minomus delphinus* by Cope (1872), before eventually being reclassified as *Pantosteus platyrhynchus* (Cope and Yarrow 1875). A similar species of sucker, *P. jordani*, was originally described by Evermann (1893b) for the area east of the continental divide that includes the Black Hills. The *P. jordani* designation remained until Miller (1958) relegated *P. jordani* to synonymity with *P. platyrhynchus*. Shortly thereafter, Smith (1966) reduced *Pantosteus* to a subgenus under *Catostomus*. More recent studies incorporating genetic techniques in association with morphometrics (Smith and Koehn 1971; Smith 1992) have provided additional evidence in support of the systematic interpretation rendered by Smith, and mountain suckers are now recognized as *C. platyrhynchus*. Where distributional overlap occurs, this species can hybridize with the closely related *C. catostomus* (longnose sucker), *C. commersoni* (white sucker), *C. ardens* (Utah sucker), and *C. tahoensis* (Tahoe sucker; Hubbs et al. 1943; Smith 1966).

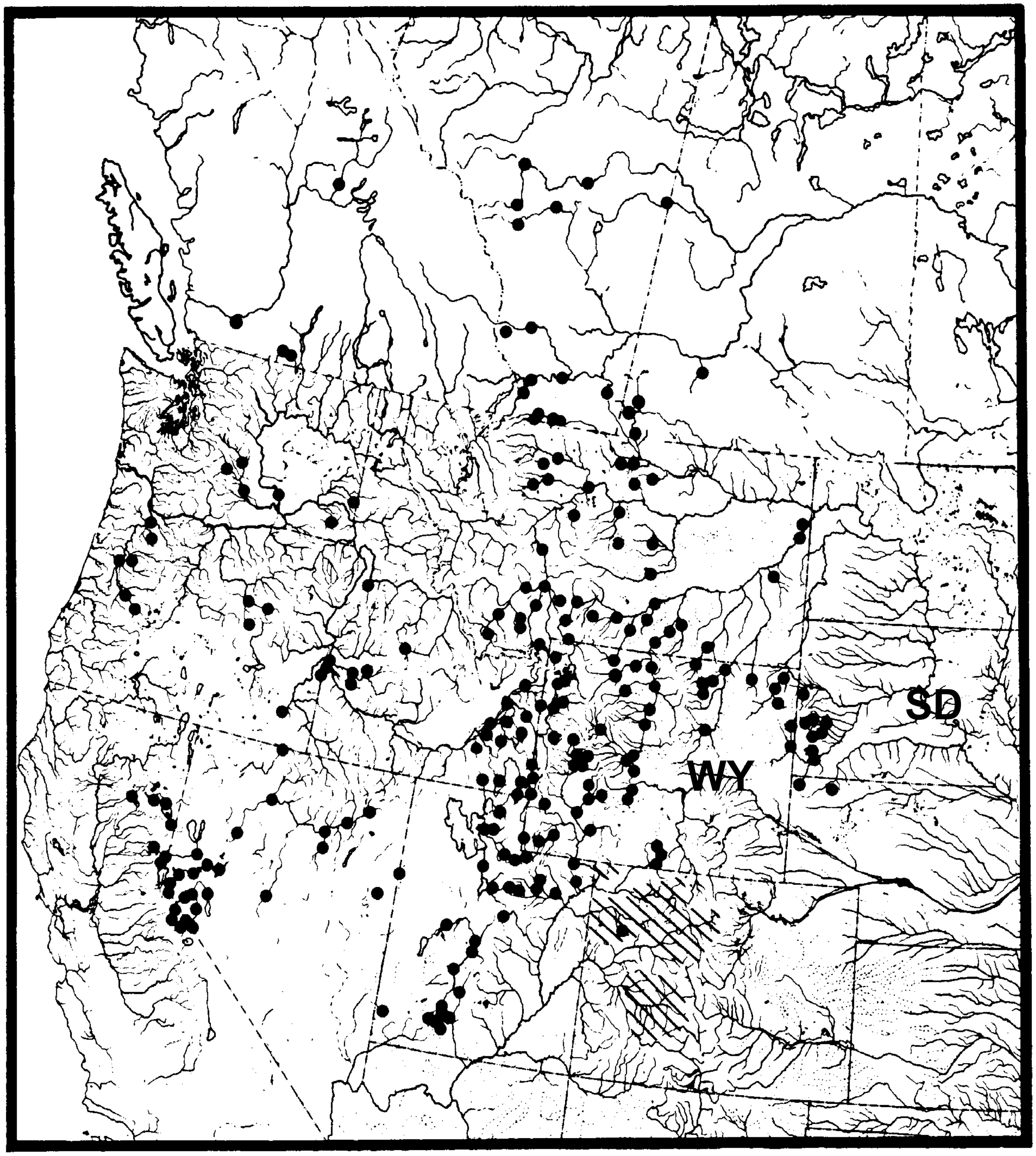
Distribution And Abundance

Distribution Of MTS Recognized In The Primary Literature.

Mountain suckers occur in much of western North America from Nevada, Utah, and eastern California and north to British Columbia, Alberta, and Saskatchewan (Figure 6). Mountain sucker populations in the Black Hills are the eastern-most extension of the species (Smith 1966; Scott and Crossman 1973; Wallace 1980). Bailey and Allum (1962) consider Black Hills populations to be glacial relicts, but they also discuss a stream piracy event wherein the headwaters of the Belle Fourche River captured the headwaters of the Little Missouri River. Because the primary range of MTS lies to the west, a piracy event seems to be an equally plausible explanation for the occurrence of MTS in the Black Hills.

At a regional scale, populations of mountain suckers are rare or nonexistent in North Dakota, Nebraska, and South Dakota outside of the Black Hills. This fish is common, however, in streams that drain mountainous areas of central and western Wyoming and Montana (Elser et al. 1980; Patton 1997; Wheeler 1997). The nearest populations outside the Black Hills (i.e., Cheyenne and Belle Fourche river drainages) that have been sampled in recent years occur in the Powder River drainage of Wyoming.

Figure 6. Distribution of MTS across North America. Modified from Smith (1966).



Historic And Present Distribution In The Black Hills

Early surveys indicate that MTS were widely distributed in streams across the Black Hills (Figure 7); Evermann and Cox 1896). During surveys conducted in 1892 and 1893, Evermann and Cox collected MTS from three different Beaver creeks, French Creek, Redwater Creek, Chicken Creek, Crow Creek, Spearfish Creek, Spring Creek, Rapid Creek, Whitewood Creek, and the Cheyenne River. Collection activities that occurred from the 1930s to the early 1950s added Castle Creek, Fall River, Iron Creek, and Grace Coolidge Creek to this list (Bailey and Allum 1962). In the most comprehensive survey of the time, Stewart and Thilenious (1964) observed MTS in most of the previous streams and in many new streams that included: Elk Creek, Grizzly Creek, Kirk Creek, Cascade Creek, Bear Butte Creek, Thompson Ditch, Slate Creek, Hot Brook, and several tributaries to Castle Creek.

Recent stream surveys (Ford 1988; Patton 1997; Olson 1997, 1998; Doorenbos 1998; Newman et al. 1999; Meester 1993 – 1999; WGFD Stream and Lake Database) suggest that MTS occur in much of the historic Black Hills range (Figure 8). Recent surveys have also found MTS in Battle Creek, Bogus Jim Creek, Boxelder Creek, Crow Creek, Deer Creek, Flynn Creek, Horse Creek, Jim Creek, Meadow Creek, Slate Creek, Swede Gulch Creek, Buskala Creek, False Bottom Creek, Foster Creek, Newton Fork, and Tilson Creek.

Estimates Of Local Abundance

Density estimates derived with a closed-population, removal-estimator methodology (Zippin 1958) exist for MTS in the South Dakota portion of the Black Hills. This information has been routinely collected by SDGFP personnel during stream surveys (Stewart and Thilenious 1964; Ford 1988; Meester 1993 – 1999). Based on the most recent set of surveys (Meester 1993 – 1999), MTS densities ranged from 7 to 13,399 fish/ha, with an average of 1,262 fish/ha and a median of 265 fish/ha for 59 sites where this species occurred. Because some stream sites were sampled multiple times during the 1990s, only the most recent data from a site were used in these calculations.

Comparable information from other portions of the MTS range is nonexistent except for Moyle and Vondracek's (1985) density estimates from nine reaches on an eastern California stream. The average and median densities that these authors observed were a third or less of those from Black Hills streams (average density = 428 fish/ha; median density = 50 fish/ha). Based on this limited comparison, MTS populations in the Black Hills appear to have relatively high densities. It should be cautioned, however, that Moyle and Vondracek's (1985) data may provide a poor baseline for comparison as these data were collected from a short segment of stream that was downstream from a dam and contained a highly altered fish community.

Figure 7. Pre-1965 collection sites for MTS in the Black Hills. Collection site information was derived from Evermann and Cox (1896), Bailey and Allum (1962), Stewart and Thilenious (1964), and the WGFD Lake and Stream Database.

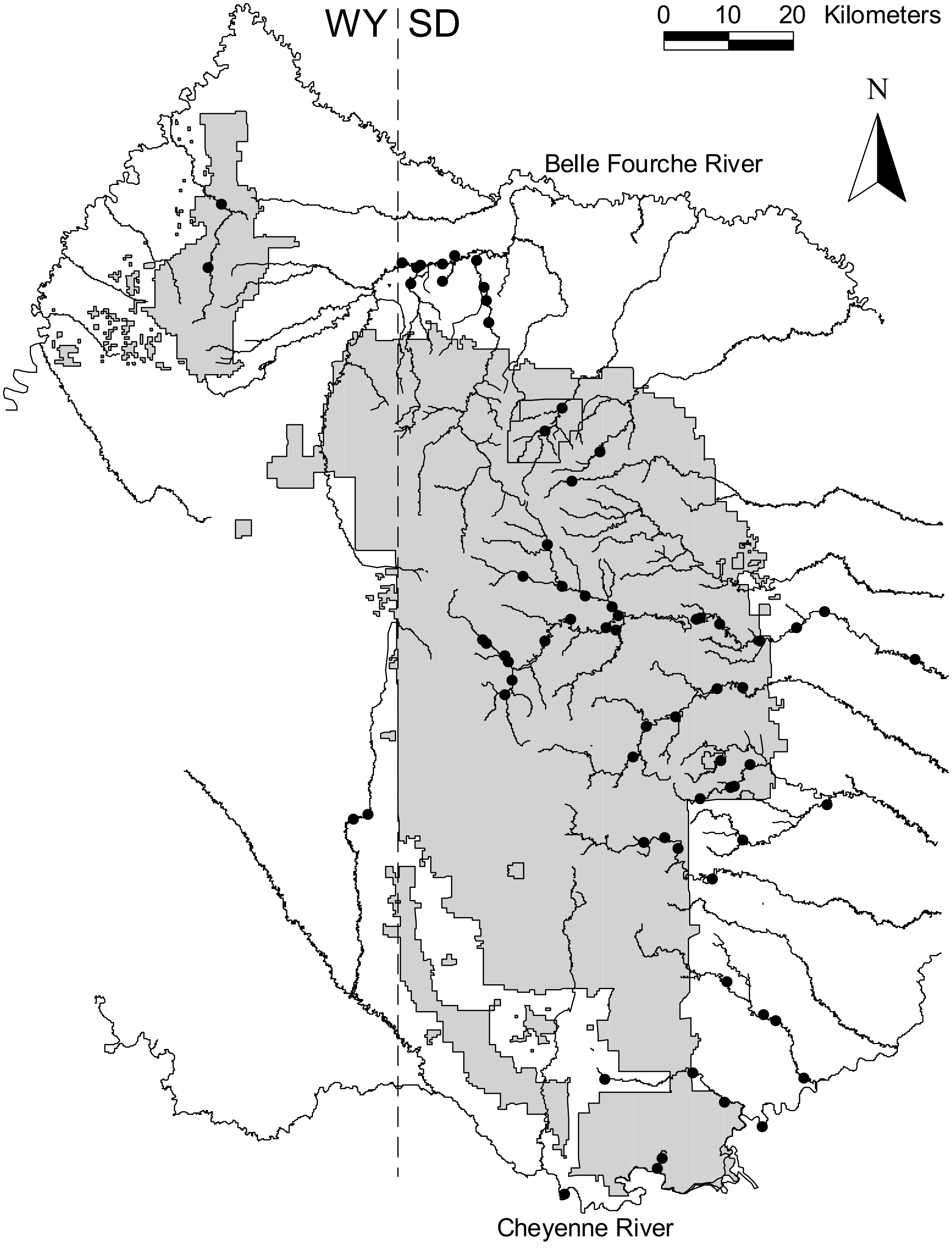
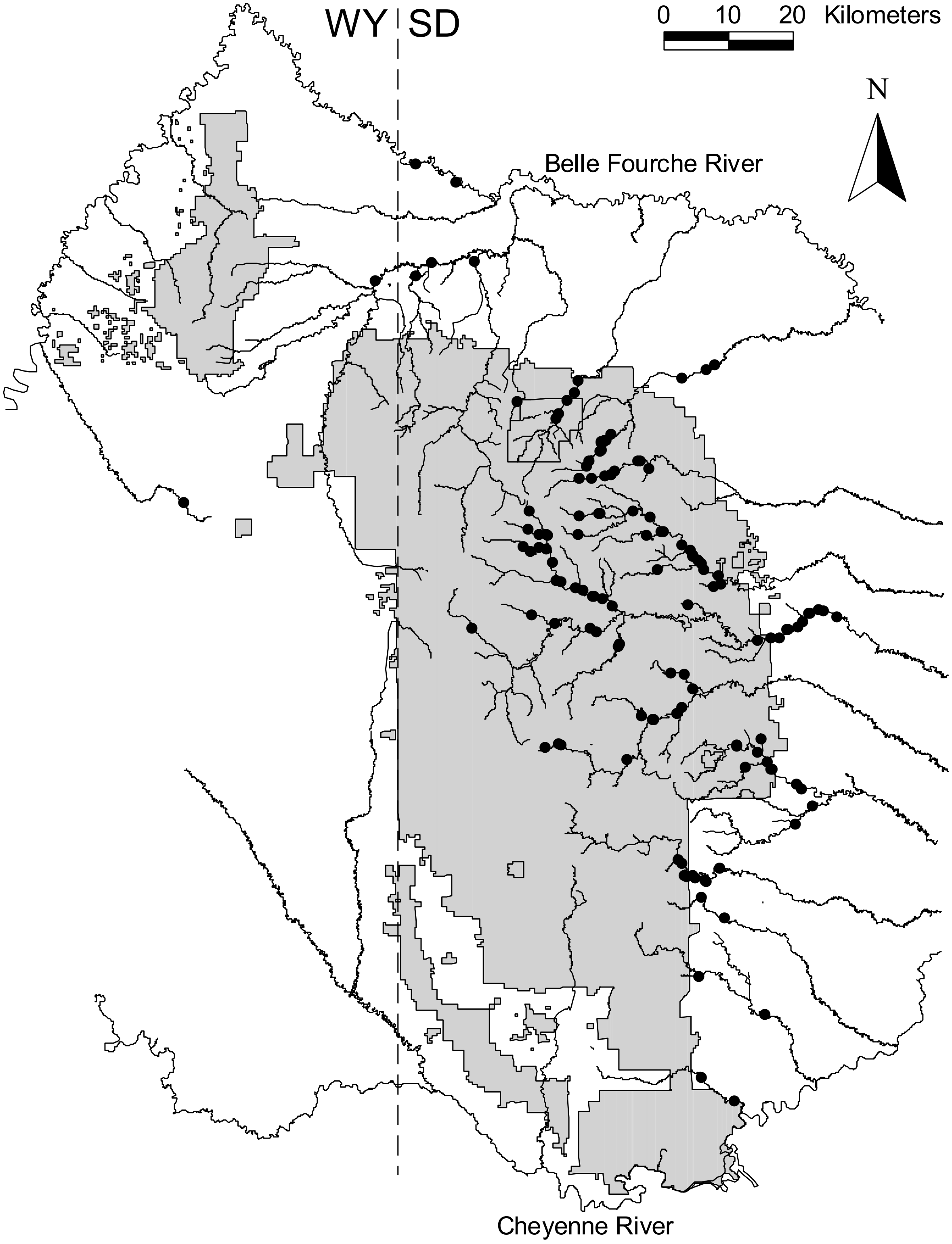


Figure 8. Recent (1980s and 1990s) collection sites for MTS in the Black Hills. Collection site information was derived primarily from SDGFP stream survey reports (Ford 1988; Meester 1993 – 1999) and supplemented with data from Patton (1997), Olson (1997, 1998), Doorenbos (1998), Newman et al. (1999), and the WGFD Stream and Lake Database.



Population Trend

Comparison of the early distribution of MTS in the Black Hills (Figure 7) with the recent distribution (Figure 8) suggests that changes over the last century have been minimal. However, one area of potential concern is the apparent absence of MTS from the upper Cheyenne River. This absence may reflect a range contraction or it could be an artifact of sampling infrequency in this drainage. In contrast to the southern Black Hills, MTS have recently been sampled more frequently in the northern Black Hills—often where the species had not previously been recorded. Rather than indicating range expansion, however, these latter changes are probably due to increased sampling intensity during the last two decades (Figure 4).

An assessment of MTS population trend is also possible using abundance data from stream sites that have been repeatedly sampled. When selecting the stream sites for this analysis, only MTS sites that were sampled at least three times from 1992 through 1998 during standardized surveys conducted by SDGFP personnel were included to ensure comparability among years. Four sites on three streams met these selection criteria (Figure 9). Trends in population density at each of the four sites were determined using simple-linear regression with year as the predictor variable and MTS density as the response variable (Figure 10). If the parameter estimate for year (i.e., the slope of the regression line) differed from zero at an alpha level of 0.05, a trend in population density was inferred.

Results from the regression analyses gave no indication of trends in MTS density at the four stream sites (Table 3). Two of the slope parameters were positive and two were negative, but none differed significantly from zero (p = 0.12 – 0.55). It should be cautioned, however, that data suitable for this trend analysis were limited and statistical power to detect trends was probably weak.

Table 3. Summary of slope parameters from simple linear regressions used to assess trends in MTS populations at four stream sites in the Black Hills.

Sample site	Years of data (n)	Slope parameter	Standard error	Confidence interval	P-value
French Creek #1	4	338	132	-228, 905	0.12
Rapid Creek #7	5	-13.9	8.30	-40.3, 12.5	0.19
Whitewood Creek #1	7	-28.4	44.6	-143, 86.3	0.55
Whitewood Creek #8	6	30.2	22.5	-32.3, 92.7	0.25

Figure 9. Sites used in a trend analysis of MTS populations in Black Hills streams. Only sites sampled at least three times between 1992 and 1998 during standardized surveys conducted by SDGFP personnel were included

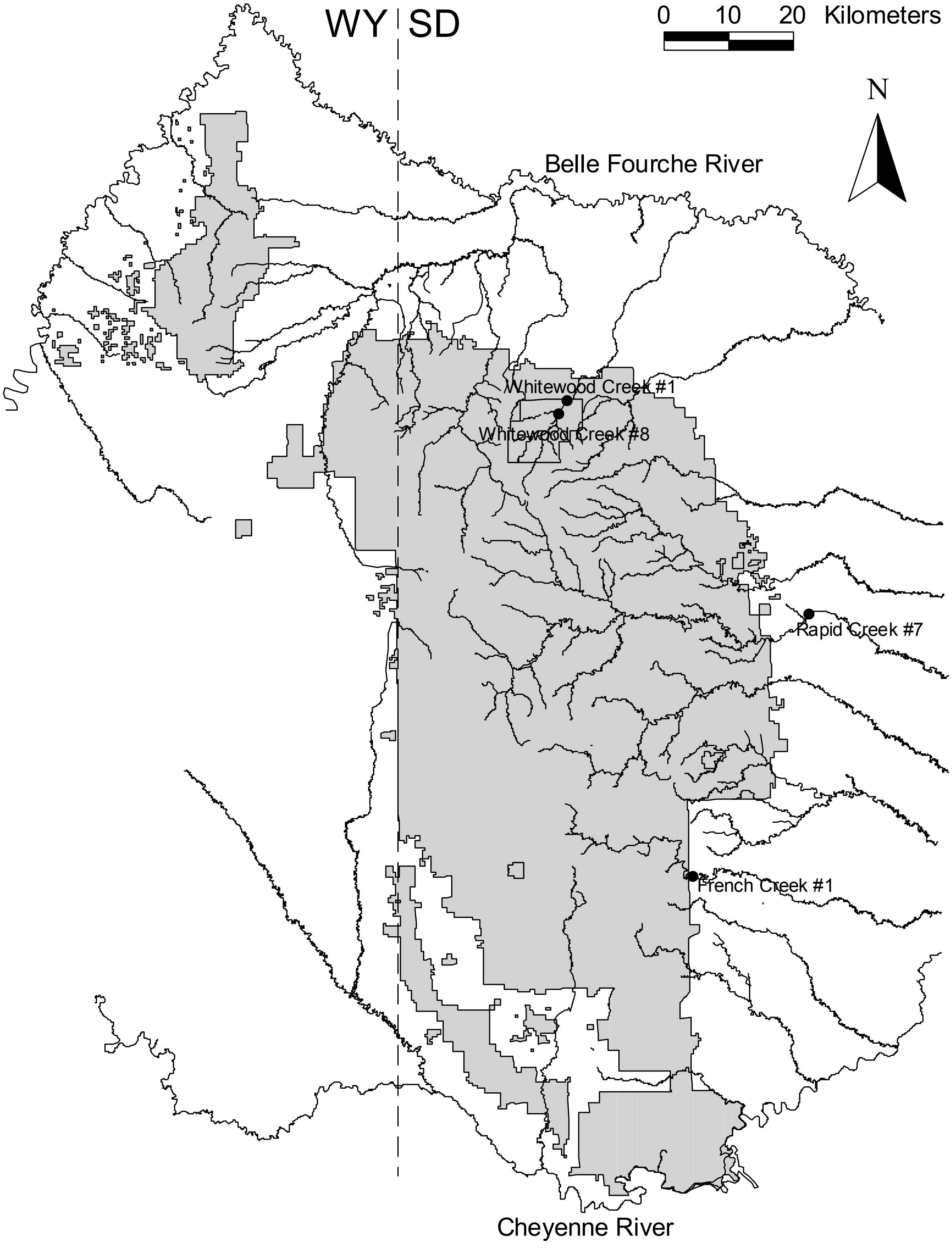
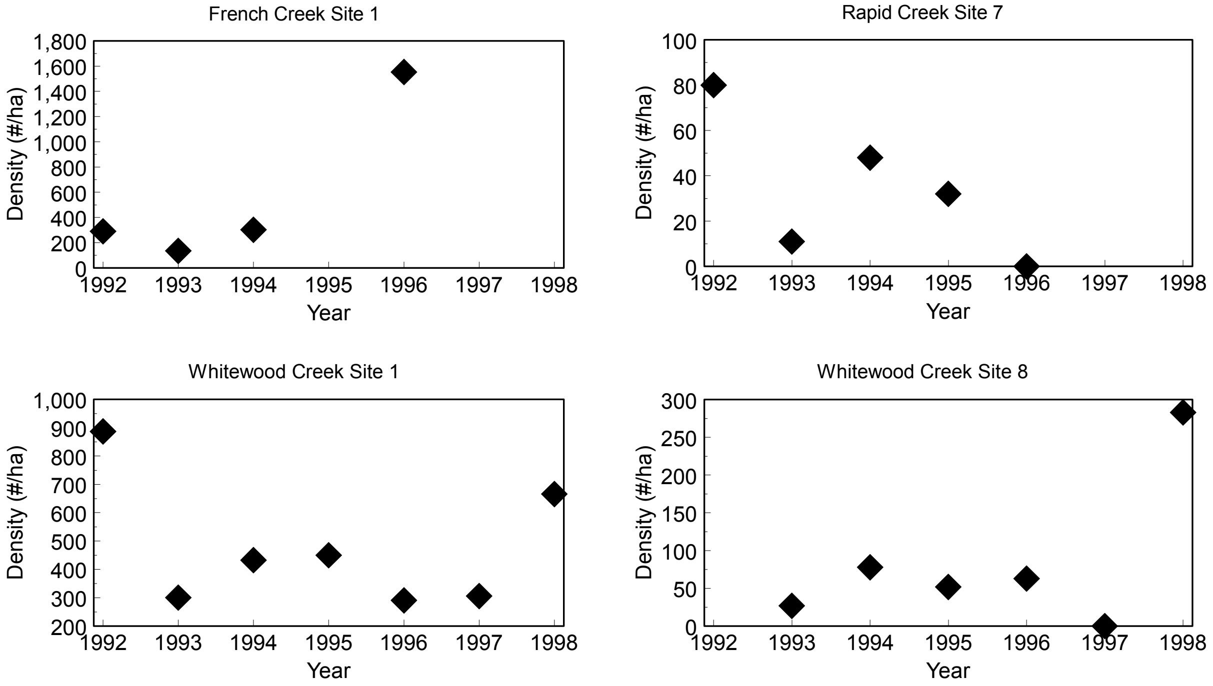


Figure 10. Densities at four sites used in a trend analysis of MTS populations in Black Hills streams. For a site to be included in the analysis, at least three density estimates obtained at the same time of year from 1992 through 1998 were required. Data were obtained from stream electrofishing surveys conducted by SDGFP personnel (Meester 1993 – 1999).



Movements

Broadscale movement patterns of MTS have not been the focus of detailed study. The only information pertaining to movements of this species is provided by authors studying spawning migrations from reservoirs into tributary streams (Decker 1989; Wydoski and Wydoski 2002). Decker (1989) observed mountain suckers moving into the downstream section of a California stream in late July and movements back into the reservoir less than a month later. Similarly, Wydoski and Wydoski (2002) recorded spawning-related movements from a Utah reservoir into a tributary stream in late May and June. In both studies, movements into tributary streams were approximately 1 km, although longer movements were precluded by beaver dams in the Utah study.

The limited amount of information on the movements of MTS provides little insight regarding the ability of Black Hills populations to interact with nearby populations. However, the fact that Black Hills populations are at the easternmost extension of the species' range means that potential linkages to other populations are limited. Populations of MTS occur in drainages adjacent to the Belle Fourche River in the Little Missouri and Powder rivers (Patton 1997), but watershed divides preclude the local exchange of individuals between these populations. Instead, a MTS would have to travel to the Missouri River and pass mainstem dams before it could successfully disperse between populations. As a practical matter, therefore, Black Hills populations of MTS are isolated from other populations.

Habitat Associations

Mountain suckers have been observed in large rivers, lakes, and reservoirs (Moyle 1976; Baxter and Stone 1995; Wydoski and Wydoski 2002) and in small prairie streams (R. Bramblett, Montana State University, personal communication), but most often occur in cool, clear mountain streams that are 3 – 12 m wide (Smith 1966; Decker and Erman 1992). These fish prefer temperatures between 13° and 23°C and moderately swift water velocities (Smith 1966; Sigler and Sigler 1996). In a study of the fish distributions within a stream, Gard and Flittner (1974) noted that MTS occurred only in a downstream section where channel gradients were lowest and temperatures were warmer than upstream areas suitable only for trout. Underwater observations made by Decker (1989) revealed that MTS were always found on the stream bottom, usually occurred in small groups, and were closely associated with cover (e.g., exposed willow or tree root masses, undercut banks, log jams, and boulders). Average depths at these locations were 0.61 m and average water velocities were 0.2 m/s (Decker 1989). Substrate composition associated with MTS habitat varies widely and ranges from mud to sand, gravel, and boulders, although cobbles are most common (Smith 1966; Decker 1989). In terms of channel morphometric units, MTS occur most often near the transitions between pools and runs (Hauser 1969; Decker 1989). Riffle habitats are rarely used, except for spawning (Hauser 1969; Wydoski and Wydoski 2002).

Food Habits

Mountain suckers are benthic feeders—as evidenced by adaptations that include a subterminal mouth and cartilaginous mouth edges that are used to scrape organic matter from rocks (Smith 1966; Baxter and Simon 1970). The diet consists mainly of simple plants like diatoms, green algae, and blue-green algae, but small invertebrate animals are also ingested. Inorganic material (e.g., silt, mud) composes a significant portion of the gut contents, but is probably ingested inadvertently (Simpson 1941; Hauser 1969). To facilitate digestion of the primarily herbivorous diet, MTS have evolved a lengthy gut that is 5 – 6 times the body length (Simpson 1941; Smith 1966).

Demographics

Mountain suckers are considered spring spawners, but the exact timing varies across the geographic range—probably in response to local variations in water temperature. Hauser (1969) documented spawning in two Montana streams during June and early July when water temperatures were between 17° and 19° C. However, Smith (1966) recorded ripe females as early as late May in a Utah stream and Decker (1989) believed MTS did not spawn until mid-August in a California stream. In northwest Wyoming, spawn timing differed by more than a month between a mountain stream and an alpine lake that was higher in elevation (Baxter and

Stone 1995). Wydoski and Wydoski (2002) recorded spawning in Utah by an adfluvial population of MTS from late May until late June, with peak spawning in mid-June at water temperatures between 9° and 11° C.

Decker and Erman (1992) speculated that MTS made a short migration into a stream from a reservoir before spawning—an activity that Wydoski and Wydoski (2002) also observed. It is unknown whether this behavior occurs in stream-resident populations that may have better access to suitable spawning habitat. The act of spawning has not been observed in MTS, but males in breeding coloration have been observed aggressively pursuing females—an activity that often precedes the release and fertilization of eggs (Baxter and Stone 1995). Hauser (1969) and Wydoski and Wydoski (2002) have suggested that spawning occurs in riffles, based on the locations of electrofishing captures and visual observations during the breeding season. Wydoski and Wydoski (2002) suggest that 75% of the MTS they observed spawning occupied water depths between 11 and 30 cm with velocities of 6 – 20 cm/s.

Scott and Crossman (1973) speculate that the construction of nests by MTS is unlikely and that eggs are probably broadcast over the substrate. Mountain sucker eggs are demersal and range in diameter from 1.5 to 2.2 mm (Hauser 1969). Fecundity of individual females ranges between 990 and 3,710 eggs (Hauser 1969) and has been directly related to female total length (eggs = -2893 + 31.2 (length), $r^2 = 0.87$; Wydoski and Wydoski 2002). Oocytes have been observed in the ovaries of ripe females, which suggests that spawning occurs in consecutive years.

Male MTS mature between age two and four, live up to six years, and reach a maximum total length of 19.6 cm (Smith 1966; Wydoski and Wydoski 2002). Females in the same population generally take an additional year to mature (Hauser 1969), but grow larger (up to 22.0 cm) and live longer than males (Hauser 1969; Wydoski and Wydoski 2002). Fastest growth occurs during the first year and decreases until sexual maturity is reached—at which point growth stabilizes (Hauser 1969).

Males in breeding condition have an orange stripe along each side above a dark greenish stripe that extends from the tip of the snout to the caudal fin (Brown 1971). Large, cone-shaped nuptial tubercles appear on the lower caudal fin and the caudal peduncle adjacent to the anal fin. Small tubercles cover the remainder of the body and all fins except the dorsal fin (Hauser 1969). Female coloration remains nondescript during breeding and tubercle development is less extensive. The female develops medium-size tubercles on the caudal peduncle and small tubercles on the dorsal and lateral parts of the head and body (Hauser 1969).

Community Ecology

Most of what is known regarding the community ecology of MTS has been obtained from Sagehen Creek in eastern California where Olson (1988) frequently observed juvenile MTS forming schools with Tahoe suckers and Lahontan speckled dace (*Rhinichthys osculus robustus*). Despite close similarities between the two sucker species, Decker (1989) never observed agonistic encounters. Evidence for competition does exist, however, between MTS and another closely related species, the bluehead sucker (Catostomus *discobolus*). Where these species are sympatric, Dunham et al. (1979) documented differences in gill-raker counts that were large relative to the differences between allopatric populations. These authors interpreted this result as indicating that competition for food resources was occurring in sympatric populations and leading to divergent gill raker morphologies. No studies have been done on interactions between MTS and white suckers, which also commonly occur in the Black Hills.

Data regarding predator-prey relationships are limited. Some authors have speculated that MTS may be an important prey item where abundances are high (Scott and Crossman 1973; Sigler and Sigler 1996; Wydoski and Wydoski 2002). However, the only empirical evidence comes from Decker and Erman (1992), who noted an inverse relationship between brown trout (*Salmo trutta*) and MTS abundance in Sagehen Creek. These authors suggest that MTS were either avoiding reaches with high numbers of brown trout or that brown trout were consuming MTS where distributions overlapped. Unfortunately, brown trout stomachs were not examined to determine which hypothesis was more viable.

Risk Factors

The wide distribution and abundance of MTS in the Black Hills, even after more than a century of intensive land use and the introduction of several non-native predators, suggests the current risks for this species are minimal. As a result, land uses and the attendant impacts to stream habitats would have to deviate strongly and on a forest-wide scale from historic and current norms before the existence of MTS populations in the Black Hills would be jeopardized. Unfortunately, external factors related to global climate change may begin to pose an increasing threat to MTS populations over the next century. Over the next 100 years, projections are for air temperatures to increase from 1.4 to 5.8 °C (Albritton et al. 2001). Because the temperatures of small mountain streams are affected most strongly by air temperatures (Isaak and Hubert 2001a), projected global warming could have a profound effect on the distribution of many aquatic species. Keleher and Rahel (1996) ran simulations for warming scenarios of 1 – 5 °C and estimate that the Rocky Mountain region would lose 16 – 68% of the summer stream habitat that is currently thermally suitable for trout. Although these results are not directly transferable to MTS, the general pattern is likely to be similar. Mountain sucker distributions will shift as fish are forced to occupy thermally suitable areas. Most often, this shift will entail decreased abundances in downstream habitats that have become too warm and increased abundances in cooler upstream areas—a trend that will increasingly fragment MTS populations and place them in more variable headwater habitats. Extirpations may occur where suitable upstream habitats either do not exist or are made inaccessible by anthropogenic obstructions.

Response To Habitat Changes

With the exception of water development (Gard and Flittner 1974; Decker 1989), the effects of management activities and natural disturbances on MTS have not been studied. By necessity, therefore, the discussion that follows is based largely on circumstantial inference related to the biology of MTS and the habitat requirements of stream-dwelling fish in general.

Management Activities

Timber Harvest And Grazing.

The effects of timber harvest and livestock grazing on MTS populations will often be similar because both management activities frequently have similar effects on stream parameters that are relevant to MTS. Both management activities often increase peak flows and flood intensity while decreasing baseflows (Jones et al. 2000; Isaak and Hubert 2001b), except for a short period of years immediately after timber harvest when evapotranspirative demands are decreased (Bosch and Hewlett 1982; Troendle and King 1985). Hydrograph alterations occur because these land uses tend to exacerbate runoff and decrease aquifer recharge by compacting soil layers, synchronizing snowmelt, and routing runoff to streams along road and trail systems (Chamberlin et al. 1991). As a result, these management activities, combined with increased human uses and evapotranspirative demands from an afforested landscape, have probably contributed to the apparent decrease in the total length of perennial streams that has occurred over the last century in the Black Hills (BHNF 1996, III-38, 40).

Timber harvest and grazing can also accelerate the rate at which sediment is transported to and through the stream network, although best management practices can help minimize sedimentation. In the case of timber harvest, greater erosion rates stem mainly from the system of roads that are built to facilitate the removal of timber. Sediments are derived from the road surface, road cuts adjacent to the road, and in extreme cases, from hillslope failures caused by improperly placed or poorly constructed roads (Furniss et al. 1991). Poorly managed livestock grazing can increase sediment delivery to streams when animals collapse banks into streams or remove riparian vegetation that maintains structural integrity. Grazing also makes sediments in upslope areas more susceptible to erosion through the removal of near-ground vegetation (Platts 1991; Belsky 1999).

The end result of improperly managed livestock grazing and timber harvest is that instream habitats can become structurally less diverse while physicochemical parameters such as temperatures, dissolved oxygen, or discharges may be made more variable (Chamberlin et al. 1991; Platts 1991). Perennial streams may become intermittent and intermittent streams may become ephemeral. Increased amounts of sediments may infiltrate and degrade high quality spawning substrates or decrease the volume of pool habitats (Chamberlin et al. 1991). Most of these habitat modifications are predicted to negatively affect MTS populations, but positive effects may also accrue. Because MTS consume simple aquatic plants, food resources may increase when animal wastes are washed into streams or the removal of riparian vegetation stimulates autochthonous production. If food were a strongly limiting factor, therefore, near-stream timber harvest or livestock grazing could be beneficial to MTS populations.

Prescribed Fire

Controlled burns could positively affect MTS populations if significant portions of a watershed were treated and fires were allowed to burn through heavily timbered (and presumably afforested) riparian areas. Removal of riparian vegetation would facilitate increases in stream productivity by allowing more solar insolation, stimulating autochthonous production, and increasing the food resources available to MTS. By killing trees in riparian areas, fires also facilitate the recruitment of large woody debris to streams (Young 1994), which typically produce structurally complex channels that provide a diversity of fish habitats (Keller and Swanson 1979).

If large portions of a watershed were treated with fires over a short time interval, decreased evapotranspirative losses could result in greater summer baseflows and increase the amount of stream habitat until tree regrowth occurred. Negative effects of prescribed fires on MTS populations may include short-term fish mortalities (Minshall and Brock 1991) or increased sediment delivery to streams (Gresswell 1999), although sedimentation could be ameliorated with proactive measures to reduce erosion.

Recreation

The most important negative effect of recreation on MTS populations can be attributed to recreational angling, which has created a demand for the introduction and maintenance of brook trout (*Salvelinus fontinalis*) and brown trout populations in Black Hills streams. Both trout species reach sizes at which piscivorous diets are often adopted (e.g., > 250 mm; East and Magnan 1991) and the brown trout especially is known to be a voracious piscivore (Alexander 1977; Ault and White 1994). Although evidence of brown trout predation on MTS is limited to the inverse correlations in abundance that Decker and Erman (1992) observed, we believe predation is a plausible mechanism by which brown trout may be impacting MTS. To the extent that brook trout populations are composed of large piscivorous individuals and their distribution overlaps that of MTS populations, brook trout may also be affecting MTS populations.

Mining

The impacts of mining on MTS populations are likely to be negative, but will be limited to areas where this land use overlaps the distribution of MTS, primarily portions of the northern Black Hills and the Harney Peak area (BHNF 1996, III-442). Mining operations that remove vegetative cover from significant portions of a watershed can result in hydrologic alterations or increased sediment delivery to streams. Additionally, many mining activities have the potential to produce acid or toxic leachate runoff that can adversely affect water qualities (Nelson et al. 1991). If runoff from mine sites is not properly controlled, direct effects to MTS populations could occur when water quality parameters exceed physiologic tolerances and cause mortalities. Indirect effects could be more subtle and may manifest through effects on food resources—whether these resources were simply eliminated or diminished in quality such that MTS consumers were affected.

The length of stream potentially affected by runoff from a poorly managed mine site would depend on the specific set of leachates, as well as temporally variable factors related to stream discharge, precipitation events, and the hydrologic interaction between the stream and mine site. In general terms, however, the strongest negative effect on a MTS population would be expected immediately downstream from a mine site and would decrease with distance downstream.

Water Impoundment And Diversion

Dams and other water development structures, when built without fish passage facilities, fragment and isolate populations of stream fishes. Not only are movements to critical spawning or overwintering areas sometimes impeded, but the recolonization of former habitats is often impossible after local extirpations have occurred. Downstream of dams, alterations to the hydrograph can change the quality of physical habitats and modify environmental cues that are sometimes needed to stimulate spawning (reviewed in Poff et al. 1997).

Case histories that document the effects of water development on MTS populations are rare and provide mixed results. Decker (1989) resampled fish communities on seven eastern California streams that were sampled in the 1930s and 1940s before the construction of dams. On the four streams where dams had been built, MTS abundance had decreased dramatically—in contrast to the stable abundances that were observed on free flowing control streams (Decker 1989). The mechanisms associated with the declines are unclear, but may have been related either to the dam and reservoir or increased competition from an increase in the populations of a closely related, sympatric sucker species. In contrast to Decker's (1989) study, Wydoski and Wydoski (2002) documented a seemingly abundant MTS population associated with a Utah reservoir. In that study, however, the fish community contained no close competitor and pre-dam data were not available to determine the effect of dam construction on MTS abundance.

Natural Disturbances

Floods And Drought

Mountain sucker populations have evolved with periodic disturbances related to natural flooding and drought cycles. Although these events can cause decreased abundance of stream fishes in the short term, recovery to predisturbance levels is usually rapid and long-term consequences are not apparent (Gard and Flittner 1974; Griswold et al. 1982). Within managed landscapes, however, negative effects to aquatic biotas may occur if natural hydrologic patterns become sufficiently modified (Poff et al. 1997; Jones et al. 2000). As an example pertinent to the Black Hills, MTS populations have probably lost habitat in association with the apparent decrease in the total length of perennial streams that has occurred over the last century (BHNF 1996, III-38, 40). Further, many suitable habitats may no longer be found in sections of stream that have become intermittent or ephemeral. It is impossible to speculate which factors are most responsible for these hydrologic alterations, but timber harvest, increased human use, road building, livestock grazing, afforestation, and beaver removal could all be playing roles.

Hydrologic alterations may also be negatively affecting MTS populations downstream of reservoirs, where seasonal patterns in hydrologic variability are often muted as water from peak runoff periods is used to fill the reservoir and then gradually released during the remainder of the year. This type of hydrograph alteration may change the physical processes that generate and maintain productive stream habitats to which MTS are accustomed or it may tip the competitive balance in favor of a potential competitor like the white sucker.

Wildfire

A wildfire is assumed to be a large, uncontrolled burn that covers all or a significant portion of a watershed upstream from a section of stream inhabited by MTS. Although no studies have specifically addressed the response of MTS populations to such a situation, it is possible to predict a generalized sequence of events from numerous fire-related studies that have been conducted (recently summarized in Gresswell 1999). During a fire, water temperatures or water chemistries may be altered to the point that fish mortalities occur (Minshall and Brock 1991). If a major precipitation event occurs shortly thereafter, hillslope failures or debris torrents could cause significant amounts of sediment to move rapidly into the stream (Swanston 1991). The result could be additional fish mortalities due to decreased water quality or the smothering of fish eggs.

Once vegetative regrowth began to stabilize the burned watershed, MTS populations would be expected to approximate or possibly exceed prefire abundances in a relatively short amount of time (e.g., 1 – 3 years). A similar phenomenon has been observed on several occasions for other

Rocky Mountain stream fishes (Novak and White 1990; Rieman and Clayton 1997) and may be attributed to several factors. First, removal of riparian vegetation often facilitates increased stream productivity by allowing more solar insolation into the stream and stimulating primary production (Wilzbach and Hall 1985). Second, the volume of stream habitat during stressful summer baseflow periods usually increases after a fire because evapotranspirative losses from trees decrease (Bosch and Hewlett 1982). Third, wildfires that burn through riparian areas facilitate the recruitment of large woody debris to streams (Young 1994), which typically results in stable and structurally complex channels that provide a diversity of fish habitats (Keller and Swanson 1979). All these factors would be expected to positively affect MTS populations in the short to mid-term (e.g., 5 – 30 years), but these benefits would begin to decrease as riparian canopies regrew and afforestation of the watershed progressed.

SUMMARY

Mountain suckers occur most often in cool, clear mountain streams and less often in lakes, reservoirs, or large rivers (Smith 1966; Baxter and Stone 1995). These fish prefer moderate water velocities, associate with a variety of substrates, occur in small groups, and are usually found near cover (Decker 1989). Underwater observations show that MTS are always closely associated with the bottom, where the subterminal mouth is used to scrape organic matter from rocks and other substrate features (Smith 1966). The primarily herbivorous diet consists of simple plants like diatoms and green algae, although small invertebrates are also consumed (Simpson 1941; Hauser 1969). Spawning occurs in spring and a short migration may be made to spawning areas (Decker and Erman 1992; Wydoski and Wydoski 2002). Fertilized eggs are probably broadcast over the substrate and left unattended (Scott and Crossman 1973). After hatching, males reach maturity in 2 – 4 years and females require one additional year (Hauser 1969). Evidence for competition with closely related sucker species exists (Dunham et al. 1979), although agonistic interactions have not been observed (Decker 1989). Some authors speculate that MTS are an important prey item where abundances are high (Scott and Crossman 1973; Sigler and Sigler 1996), but the only empirical evidence of predator-prey interactions is circumstantial and based on an inverse relationship between MTS and brown trout abundance (Decker and Erman 1992).

Mountain suckers occur in much of the western U.S.—from Utah, Nevada, and eastern California; north into British Columbia and Alberta (Smith 1966). The population in the Black Hills is the eastern-most extension of the species and is either a glacial relict or the result of a stream piracy event (Bailey and Allum 1962). Currently, MTS on the BHNF are probably isolated from other MTS populations by the spatial extent and fragmentation of the intervening stream network.

Early surveys suggest that MTS were ubiquitous to streams throughout the Black Hills (Evermann and Cox 1896; Bailey and Allum 1962). Recent surveys suggest the species remains widely distributed (Patton 1997; Meester 1993 – 1999), although range contraction may have occurred in the southern Black Hills. Within the current range, data from four stream reaches that were repeatedly sampled during the 1990s give no indication of declines in abundance. Additionally, the limited comparisons of MTS abundances that can be made between the Black Hills and other areas suggest that Black Hills abundances are high.

The wide distribution and abundance of MTS in the Black Hills suggests the current risks for this species are minimal. As a result, land uses and the attendant impacts to stream habitats would have to deviate strongly and on a forest-wide scale from historic and current norms before the existence of MTS in the Black Hills would be jeopardized. In the future, however, global climate change could pose a significant threat. In streams where distributional adjustments are not impeded by dams or diversion structures, MTS distributions are predicted to shift upstream as fish are forced to move into thermally suitable areas. Extirpations could occur where suitable upstream habitats either do not exist or are inaccessible.

REVIEW OF CONSERVATION PRACTICES

Management Practices

We are not aware of any management practices that have been implemented specifically for MTS in the Black Hills or elsewhere in North America.

Models

No models have been developed for MTS or MTS habitat.

Survey And Inventory Approaches (Presence/Absence)

The methods discussed in this section will focus on those that are applicable to a short segment of stream—often referred to as a reach—because this sampling scale is most conducive to surveying and monitoring protocols for stream fishes. Underwater observation and electrofishing with a backpack unit are the two methods that have been used to sample MTS in streams (Moyle and Vondracek 1985; Decker 1989; Meester 1993 – 1999). Both methods require similar resources in terms of time and personnel, but electrofishing may harm fish and requires a greater capital investment than snorkeling (approximately $5,000 vs. $500). However, electrofishing can also provide a suite of data (e.g., weights, ages, gut contents, etc.) that are not available from underwater observation.

Either sampling method could be used in a survey protocol to provide a probabilistic statement regarding the presence or absence of MTS in a stream reach. For example, given a desired level of confidence in detecting MTS and the probability of detection within a habitat unit, the number of habitat units that need to be sampled can be calculated (Green and Young 1993). As a simpler alternative, the length of stream that needs to be sampled to detect a predetermined percentage of the species assemblage could be calculated based on species-area relationships (Angermeir and Smogor 1995; Patton et al. 2000).

Monitoring Approaches (Population Trend, Habitat)

For the methods discussed in this section to be of utility in assessing and monitoring MTS populations and stream habitats, they must be used with a sampling design that encompasses multiple sites across the range of MTS. Additionally, these sites must be repeatedly sampled at the same time of year using the same sampling gear. Baseflow periods during late summer and early fall typically provide the best opportunity to sample fish in mountain streams.

A single electrofishing or snorkeling pass through a reach, during which fish are either captured or counted, is often strongly correlated with population size (Kruse et al. 1998; Mullner et al. 1998) and both methods can be used to provide an index to population size. Indexes can be obtained with a minimum of effort, but do not provide information about actual population sizes or the amount of variation associated with population estimates. For a small amount of additional effort, these population parameters can be obtained using closed-population, mark-recapture (Ricker 1975) or removal techniques (Zippin 1958; White et al. 1982). Mark-recapture population estimates are usually more accurate, but less precise than estimates obtained using removal methods (Peterson and Cederholm 1984). Additionally, mark-recapture requires the use of two methods for capturing fish, the marking of individual fish, several assumptions regarding tag retention and identification, and extended time periods to complete the estimate (Ricker 1975). Removal estimators, although less accurate, have a consistent bias (Peterson and Cederholm 1984; Riley et al. 1993) and usually provide the most practical means of deriving population size estimates for stream fishes.

A dearth of information regarding the habitat preferences of MTS makes it impossible to discuss a habitat monitoring protocol in anything but general terms. Literally hundreds of stream variables and measurement methods exist (for recent summaries see Overton et al. 1997 or Bain and Stevenson 1999), but transect methodologies (e.g., Platts et al. 1983; Simonson et al. 1994) offer the most robust and flexible approach currently available. Transect methodologies rely on a series of transects that are run perpendicular to the direction of flow and are spaced at even intervals along a stream reach. A host of stream variables can be measured along each transect that could include substrate characteristics, fish cover, riparian characteristics, stream geometry, flow velocity, etc. Additionally, the statistical properties of most variables are well described (Platts et al. 1983; Simonson et al. 1994) and it is possible to estimate the amount of effort that is needed to obtain a desired level of precision.

ADDITIONAL INFORMATION NEEDS

Additional information needed for the scientifically sound, forest-scale conservation of MTS on the BHNF are summarized in Table 4. Currently, little is known about the stream habitats or the associated landscape and watershed conditions that benefit MTS populations. Given the broad distribution of MTS in the Black Hills and their wide range in abundance, it would be possible to develop statistical models that identify critical stream habitats and landscape configurations.

Table 4. Additional information needed to support the development of a comprehensive forest-scale conservation plan for MTS in the Black Hills.

Data	Priority	Justification	Cost[*]
Habitat association	Intermediate	Provide an understanding of the habitat attributes that promote the existence of MTS populations	Intermediate
Landscape association	Intermediate	Provide an understanding of the landscape and watershed factors that generate MTS habitat	Intermediate
Trend monitoring	Intermediate	Allows assessment of population status and inference regarding factors that affect abundance	Low

[*] Low: estimated cost = $1,000 – 10,000; intermediate: estimated cost = $10,000 – 100,000; high: estimated cost > $100,000

Trends in MTS populations other than those inferred from distributional shifts are difficult to make. Although density estimates are available at four stream reaches that have been sampled multiple times since 1992, these reaches exist on only three streams. A better monitoring protocol would establish index reaches on a wide range of streams across the Black Hills and require that reaches be sampled at the same time of year. A monitoring design that incorportated these features would provide more detailed trend data than are currently available and give insight to population declines before MTS disappeared from a portion of their range. To maximize the use of existing data, index reaches could be established at SDFGP stream sites for which the most years of data exist and the same methods used to estimate population size. Given that MTS populations appear to be stable in the BHNF, index reaches would not need to be sampled each year, but possibly at 2- or 3-year intervals.

FINESCALE DACE

REVIEW OF TECHNICAL KNOWLEDGE

Systematics

Finescale dace were first described as *Phoxinus neogaeus* by Cope (1869) from a specimen collected in Michigan. Evermann and Cox (1896) described a specimen of the same species from Cox Lake northwest of Spearfish, South Dakota as *Leuciscus neogaeus*. Jordan (1924) moved *neogaeus* into the genus *Pfrille*, which was subsequently merged with the *Chrosomus* genus by Bailey (1951). Banarescu (1964) proposed moving *neogaeus* back into its original genus—*Phoxinus*—in a move that was subsequently supported by McPhail and Lindsey (1970) and Mahy (1972) and now provides the species' nomenclature.

Finescale dace hybridize extensively with northern redbelly dace (*Phoxinus eos*) and some populations near the Black Hills in Nebraska (Johnson 1942; Joswiak et al. 1982) and Montana (Stasiak 1978) consist entirely of hybrids. Black Hills populations are believed to be pure FSD (R. Bailey, University of Michigan, personal communication) and are generally recognized as such (Bailey and Allum 1962; Stasiak 1980). However, genetic analyses have not been conducted on these populations and a morphometric study of dace hybrids by New (1962) confuses their status. For New's work, R. Bailey provided dace from sites in Michigan and the Black Hills. New (1962) subsequently concluded that these samples frequently contained hybrids, but from the data reported in the manuscript, it is impossible to determine whether the hybrids occurred exclusively in Michigan samples or whether some hybrids originated in South Dakota. As a result, the purity of FSD populations in the Black Hills is questionable and will remain so until definitive genetic analyses are conducted.

Distribution And Abundance

Distribution Of FSD Recognized In The Primary Literature.

Finescale dace are distributed along the U.S. – Canada border from Maine to Minnesota and throughout the Mackenzie River drainage from the Arctic Circle south (Figure 11). The distribution of FSD is similar to LCB in that disjunct populations—also caused by earlier glacial events—occur at the southern edge of the FSD range (Johnson 1942; Bailey and Allum 1962). Different from the LCB, however, strong populations of FSD do not occur in states that border the Black Hills and the isolated populations that do exist are often hybridized with northern redbelly dace (Bailey and Allum 1962; Joswiak et al. 1982). The only genetically pure population of FSD that has been sampled in recent years near the Black Hills occurs in a short section of the Niobrara River in Wyoming (Joswiak et al. 1982; Patton 1997). Downstream sections of this river where it flows through northern Nebraska contain hybrid dace (Joswiak et al. 1982).

Historic And Present Distribution In The Black Hills

The information contained in this section is based on fish surveys summarized in Evermann and Cox (1896), Bailey and Allum (1962), Olson (1998), and the WGFD Lakes and Streams

Database. Records contained in these sources suggest the distribution of FSD was historically and is currently limited to a small area in the northern Black Hills (Figure 12). With the exception of Geis Reservoir on the Middle Fork of Hay Creek, all the collection sites of FSD are in the Redwater Creek drainage (Figure 13). Finescale dace have been collected from 15 sites in this area, which are split about evenly between lentic waters (eight sites) and lotic waters (seven sites). Recent observations, however, have occurred exclusively in the lentic waters of reservoirs, lakes, and spring holes.

Figure 11. Distribution of FSD across North America. Open circles designate hybrid populations. Modified from Stasiak (1980).

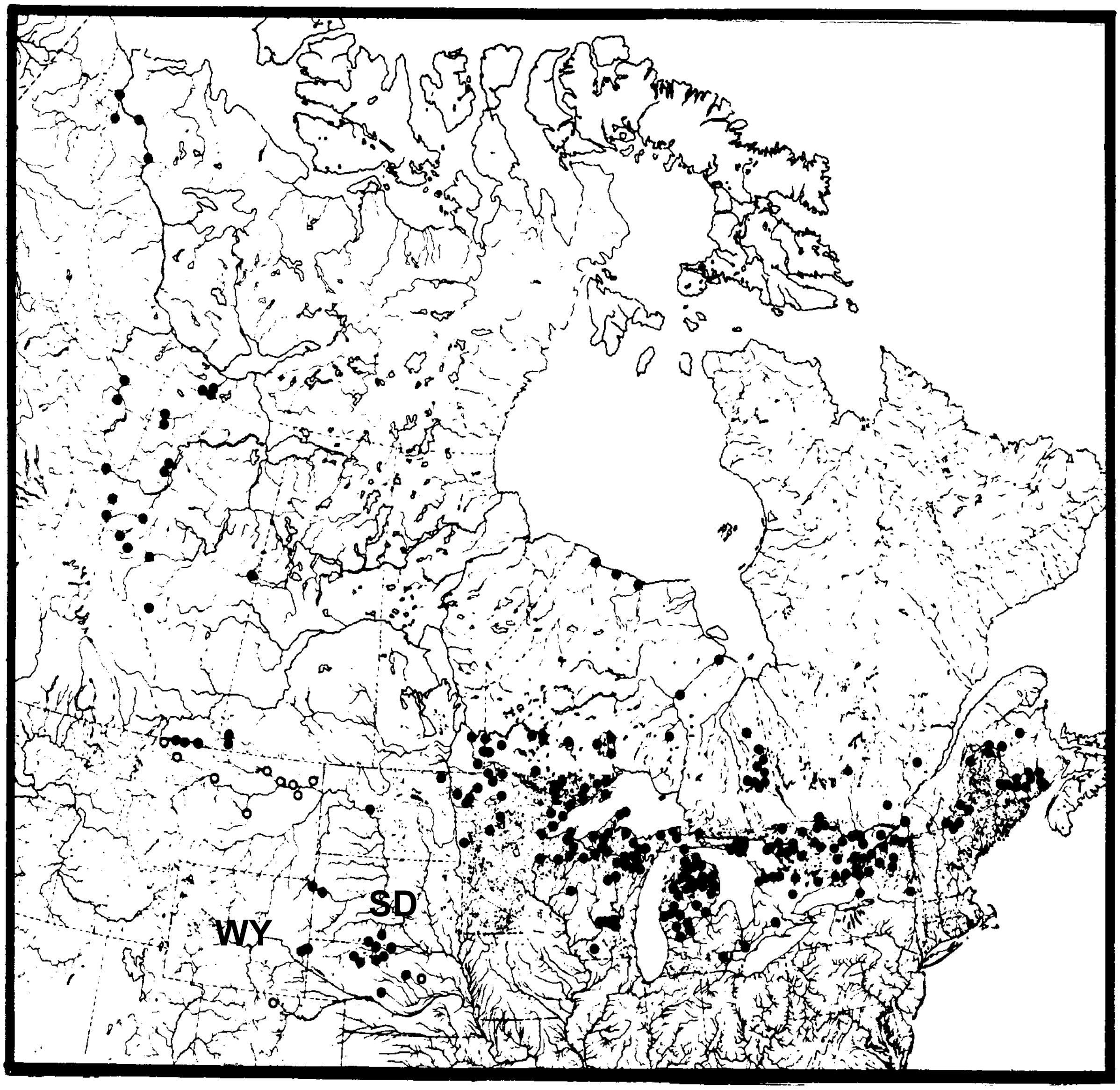
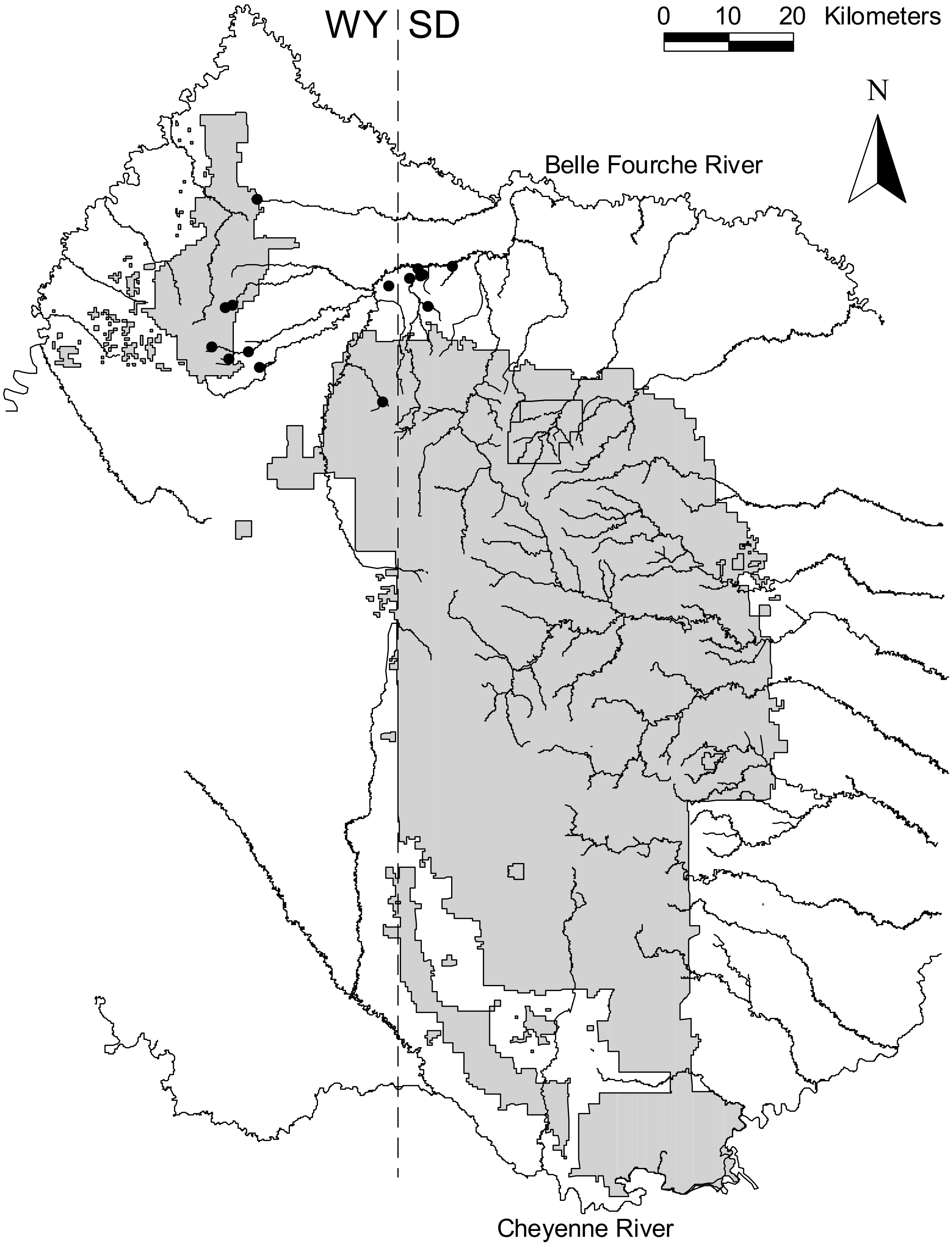


Figure 12. Collection sites for FSD in the Black Hills. Information was derived from Evermann and Cox (1893), Bailey and Allum (1962), Olsen (1998), and the WGFD Lake and Stream Database.



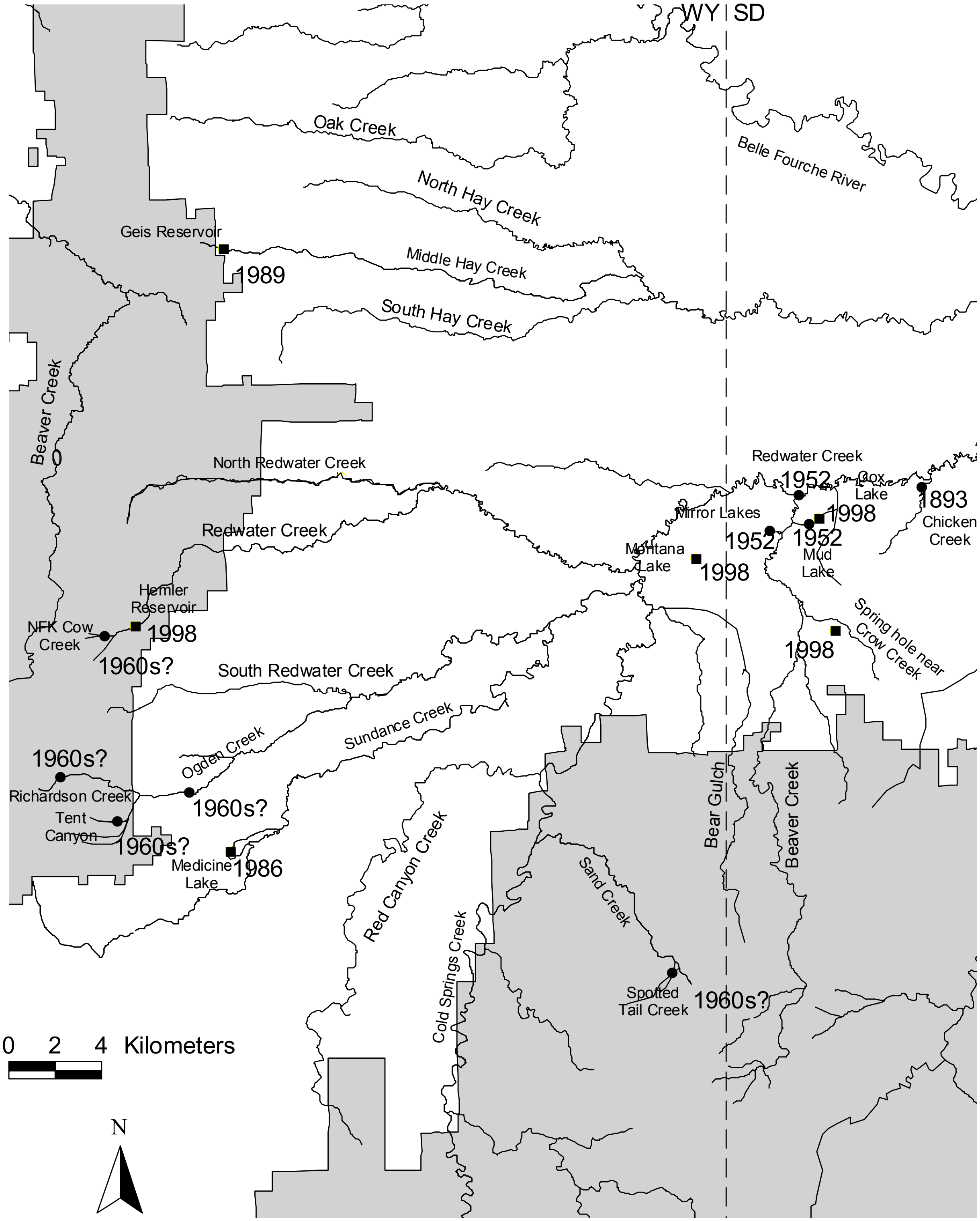


Figure 13. Closeup of the FSD distribution in the northern Black Hills. Dates indicate the most recent year that FSD were collected from a water body. Older collection sites are represented by circles; recent collection sites are represented by squares. Exact dates are not available for many of the sites in Wyoming, but these areas were probably sampled in the 1960s (B. Bradshaw, WGFD, personal communication).

In the South Dakota portion of the FSD range, no populations have been documented on BHNF lands, but populations do occur immediately to the north in Cox Lake and a small spring hole near Crow Creek (Olson 1998). Finescale dace no longer occur at four sites where previous collections have been made. Two of these sites, located on Chicken Creek and Redwater Creek, were last sampled successfully in 1893 and 1952, respectively (Evermann and Cox 1896; Bailey and Allum 1962). Finescale dace at the Redwater Creek site may have disappeared after populations of FSD were eliminated from the Mirror lakes and Mud Lake immediately upstream. Finescale dace were last collected from these lakes in 1952 (Bailey and Allum 1962), before the illegal introduction of a littoral predator, green sunfish (*Lepomis cyanellus*), appears to have extirpated these populations.

In the Wyoming portion of the FSD range, recent evidence suggests populations exist in Hemler Reservoir (BHNF land) and Montana Lake (Olson 1998). Evidence from the late 1980s suggests populations may also exist in Medicine Lake and Geis Reservoir (upstream portions of reservoir located on BHNF land). Finescale dace have been collected at sites on five streams (four sites are on BHNF land) in Wyoming (WGFD Lakes and Streams Database), but these areas have not been sampled recently. Exact collection dates for these sites are unknown, but these areas were probably sampled in the 1960s (B. Bradshaw, WGFD, personal communication).

Estimates Of Local Abundance

Quantitative estimates of density or population size do no exist for FSD in the Black Hills, nor are such data available from other areas. The only information regarding the abundance of FSD are anecdotal observations made by collectors during sampling activities, but it should be cautioned that this information allows only the crudest of qualitative inference. For example, Evermann and Cox (1896, pages 334 – 335) noted that FSD were "very abundant" during the original collection from Cox Lake, and in nearby Montana Lake, this fish "swarmed by the thousands around the mouth of a little spring…" In a recent trip to Montana Lake, however, Olson (1998) had difficulty capturing even a small number of FSD and he believed the population was small—due in part to a drought in the late 1980s that almost dried the lake. Conversely, Olson (1998) found FSD to be abundant and easily captured at Hemler Reservoir. The only other comment that exists regarding FSD abundance is associated with the Medicine Lake population—where dace were observed in large numbers during periodic lake renovation projects that occurred into the 1970s (B. Bradshaw, WGFD, personal communication).

Population Trend

Finescale dace have disappeared from four of the six sites where previous collections have been made in the South Dakota portion of the Black Hills range and the trend for the species in this portion of its range is one of decline. Greater uncertainty exists regarding population trends in Wyoming, where dace have previously been located on BHNF lands, but the historic distribution of FSD is largely unknown. Many Wyoming FSD populations were not located until the 1960s and no efforts have been made to sample these populations in the intervening years. Without better knowledge of the contemporary distribution, it is impossible to make an assessment of trend. The only statement that can be made with confidence is that the number of confirmed extirpations is less in Wyoming than in South Dakota.

Movements

No studies have been conducted regarding broadscale, dispersal/colonization movements of FSD and even anecdotal observations about these types of movements are lacking. However, even if it were assumed that FSD were relatively vagile—which is improbable given their small body size—it is unlikely that interactions with populations outside the Black Hills could be achieved. The nearest population occurs in a Wyoming tributary to the Niobrara River that is southwest of Hot Springs, South Dakota (Patton 1997). This population is separated from the northern Black Hills by an extensive stream network and a series of water developments that are likely to be impassable. Other proximal dace populations occur in the upper Missouri drainage (Stasiak 1980) and north-central Nebraska (Johnson 1942; Joswiak et al. 1982), but these populations are hybridized with northern redbelly dace.

Habitat Associations

Knowledge of the habitat requirements of FSD is crude and limited to anecdotal accounts provided by researchers studying FSD for other reasons. Taxonomists have noted that FSD often occur in cool, boggy, spring waters that are often associated with complexes of beaver (*Castor canadensis*) dams or small lakes (Bailey and Allum 1962; Baxter and Stone 1995). Scott and Crossman (1973) remarked that waters in these habitats have slightly acid pHs and are stained due to the decomposition of abundant organic matter. In a study of a FSD population that occurred in a Minnesota beaver pond, Stasiak (1978) noted that fine silt covered the bottom of the pond, large woody debris from the surrounding forest ringed the periphery of the pond, and that water temperatures reached 24°C in the summer. Tyler (1966) studied the temperature tolerance of FSD in a laboratory setting and documented significant mortality rates above 25°C.

Populations of FSD x northern redbelly dace hybrids in small Canadian lakes exhibit diel shifts in habitat use (Gauthier and Boisclair 1996). Littoral areas were used during the day and pelagic areas were used at night. Definitive answers regarding the mechanisms responsible for this behavior are not known, but speculations are that these movements may be an optimal foraging strategy designed to avoid predators and take advantage of diel movements in zooplankton (Gauthier and Boisclair 1996). Lake populations of northern redbelly dace have been observed making similar shifts in habitat use (East and Magnan 1991), but the possibility of this phenomenon has not been studied in pure FSD populations.

Food Habits

Litvak and Hansell (1990) conducted the only food habits study for FSD. In this study, the guts of 111 FSD from a small lake in Ontario were examined and found to contain a wide variety of items that included zooplankton (Amphipods, Copepods, Chironomids, and Cladocerans), aquatic insects, and plant detritus derived from the benthos and water column. Zooplankton, insects, and detritus each made up a third of stomach contents by number. Litvak and Hansell (1990) also noted that FSD appeared to have the most flexible food habits of the three fish species that were studied—the others being fathead minnows (*Pimephales promelas*) and northern redbelly dace.

Additional insights regarding the feeding habits of FSD are possible if extrapolations are made from studies of FSD x northern redbelly dace hybrids. Trudel and Boisclair (1994) noted that hybrids in a small Ontario lake shifted from a diet composed primarily of aquatic insects (Odonata and Ephemeroptera) in July to a diet of zooplankton (Cladocerans and Copepodes) in August. Conversely, Gauthier and Boisclair (1996) observed no dietary shift in a strictly zooplanktivorous hybrid population that occurred in a 5-ha lake in Quebec. These researchers also documented dace migrating offshore at dusk and found that 70% of food consumption occurred at night in pelagic waters.

Demography

The timing of FSD spawning varies by geographic location and is probably strongly linked to water temperatures. In northern portions of the range, spawning activities have been reported in June and July (Dymond 1926; Das and Nelson 1990), whereas in southern areas spawning occurs in April and May (Stasiak 1977; Becker 1983). During breeding periods, males are easily recognized by the bright yellow and red colors on their ventral surface. The fins of the male often develop a light amber or yellow tint and the first four or five rays of the pectoral fins become darkly pigmented, thickened, and strengthened—often appearing somewhat deformed (Scott and Crossman 1973). Nuptial breeding tubercles also form on the male's breast and flanking the base of the anal fin posterior to the caudal fin.

During spawning, large schools of FSD form that are composed primarily of two- and three-year-old dace, although individuals up to age four are sometimes present (Stasiak 1978). Females occasionally leave these schools and are aggressively pursued by several males as the female swims into depressions under logs or other cover. The female releases 20 – 30 eggs, which the males immediately fertilize. The eggs sink quickly to the substrate and are abandoned (Stasiak 1978). Female fecundity ranges from 400 – 3,000 eggs and is strongly related to female length (eggs = -1764 + 62.6 (length), $r^2 = 0.85$, Stasiak 1978; log eggs = 2.62 + 0.067 log (length), Das and Nelson 1990).

The only information regarding the life history of FSD comes from the work by Stasiak (1972, 1978). Studying laboratory populations, he documented an egg incubation period of 6 days at 20°C. Newly hatched fry were 4.2 mm in length and began feeding 7 days after hatching. Female FSD grew faster, lived longer, and obtained larger sizes than males. Members of both sexes mature during the second year of life (Stasiak 1972).

Community Ecology

Litvak and Hansell (1990) used dietary preferences to make inferences regarding community structure among sympatric populations of FSD, northern redbelly dace, and fathead minnows in a small Ontario lake. Of the three species studied, it was concluded that FSD occupied the largest niche space based on their broad diet. Litvak and Hansell (1990) also contended that competition was occurring among these fishes, but provided little empirical support for their position. Anecdotal observations provide most of the remaining information regarding the community ecology of FSD. Scott and Crossman (1973) note that FSD often occur with northern redbelly dace, brook stickleback (*Culaea inconstans*), and pearl dace (*Margariscus margarita*) across much of the range in Canada. Bailey and Allum (1962) also mentioned the occurrence of FSD with pearl dace and northern redbelly dace, whereas Baxter and Stone (1995) noted an association with brassy minnows (*Hybognathus hankinsoni*), fathead minnows, and pearl dace in Wyoming.

Interactions between FSD and their predators have not been the subject of detailed study, but Litvak and Hansell (1990) remarked that the introduction of largemouth bass (*Micropterus salmoides*) virtually eliminated the minnow community, including FSD, from a lake in Ontario. Predation by large brook trout on the closely related northern redbelly dace has also been documented, although it is unclear if this predation had population level effects (East and Magnan 1991; Naud and Magnan 1988).

Risk Factors

Risk factors vary dependent on whether a FSD population inhabits a stream, lake, or reservoir. For stream populations, the most important factors are probably watershed afforestation, land uses that decrease aquifer recharge by exacerbating runoff, human water uses, and beaver populations that appear to be much reduced from historic abundances (Ludlow 1875; Dodge 1965; BHNF 1996, III-253, 254; Parrish et al. 1996). Combined, these factors have probably lowered water tables along streams and decreased the amount of boggy, pond-like habitat preferred by FSD. As a result, many streams in the Redwater Creek drainage now are seasonally intermittent streams (Olson 1998). Continuation of this trend will have increasing negative effects on any stream-dwelling FSD populations that remain and could ultimately lead to their extirpation.

Hydrologic factors also have the potential to affect the fate of FSD populations that exist in reservoirs and natural lakes. Large floods could breech the small earthen dams that form Geis and Hemler reservoirs or lengthy droughts could dry these habitats. The complete loss of these lentic habitats for extended periods would almost certainly eliminate the associated FSD populations as these fish appear incapable of a strictly lotic existence and beaver activity in both areas is minimal (D. Isaak, personal observation). As an aside, Hemler Reservoir is often drafted for irrigation purposes, but a small pond remains near the lake outlet that prevents total desiccation and appears to serve as a refuge for FSD (M. Surber, BHNF, personal communication).

The natural lake habitats that support several FSD populations are dependent on inflows from springs. If these inflows were to decline in association with the general decrease in aquifer levels across the northern Black Hills (Kyllonen and Peter 1987), FSD populations could be negatively affected. At the present time, however, declining spring flows do not present a serious problem (J. Erickson, SDGFP, personal communication).

For most FSD populations in lentic waters, the greatest risk factor appears to be the introduction of exotic predators—primarily members of the sunfish family. Such introductions are probably responsible for the extirpation of FSD from the Mirror lakes and Mud Lake and similar results would be expected in the lakes and reservoirs that currently contain FSD. As populations disappear, seasonal dewatering and other factors that fragment the Redwater Creek stream network would exacerbate population declines by inhibiting the dispersal of individuals that are needed to refound populations.

Response To Habitat Changes

Of the three fish species addressed in this conservation assessment, FSD occupy the greatest diversity of aquatic habitats within the Black Hills. However, so little is known about the microhabitat requirements of FSD or their ecology that it is difficult to elaborate on previous discussions regarding the hypothesized effects of habitat changes on LCB in reservoirs (pages 19 – 23) or MTS in streams (pages 45 – 52). To avoid redundancy, therefore, the following discussion focuses primarily on populations that occur in habitats unique to FSD—spring-fed natural lakes and beaver pond complexes.

Management Activities

Timber Harvest And Grazing

The most important effects of livestock grazing on FSD populations in spring-fed natural lakes are likely to be indirect and negative. This land use, even when properly managed, tends to compact soil layers and decrease the infiltration of precipitation that is required for aquifer recharge (Chamberlin et al. 1991; Isaak and Hubert 2001b). The same indirect negative effect is generally expected from timber harvest because the roads built in conjunction with this activity route precipitation to streams, increase peak flows, and minimize subsurface aquifer recharge (Jones et al. 2000). An exception to this general pattern may occur for a short period immediately after tree removal and before regrowth when evapotranspirative water losses are minimized, but such increases are ephemeral and decrease as afforestation of the watershed progresses (Bosch and Hewlett 1982; Troendle and King 1985). As a result, grazing and timber harvest have probably contributed, along with increased human uses and evapotranspirative demand from an afforestated landscape, to the general decrease in aquifer levels that has occurred across the northern Black Hills (Kyllonen and Peter 1987). If aquifer levels continue to drop, the springs that support several FSD populations in spring-fed natural lakes could become jeopardized. For similar hydrological reasons, populations of FSD that occur in streams or are associated with beaver ponds that are dependent on late-summer aquifer discharges for maintenance of suitable baseflows may also be negatively impacted by grazing and timber harvest.

Livestock grazing and timber harvest may also indirectly affect FSD populations by affecting beaver populations. Heavy grazing by large herbivores, especially when it occurs near streams, suppresses the growth of aspen (*Populus tremuloides*), willows (*Salix* spp.), and cottonwoods (*Populus* spp.) that beavers need for food and dam construction (Novak 1987). This mechanism is believed to be partially responsible for the decline of beaver populations in Yellowstone National Park and other areas in the American west (Kay 1998). Although other processes, including trapping and landowner removal, have contributed to the decline of beaver in the BHNF (Parrish et al. 1996), livestock grazing—to the extent that it occurs near streams—can be expected to have a negative effect on the growth of willows, cottonwoods, and aspens and by extension—beaver and FSD populations. Selective harvest of pines that have invaded riparian areas, by contrast, could conceivably promote the regrowth of these riparian vegetation types and ultimately benefit FSD.

Recreation

The illegal introductions of sunfishes by recreational anglers appears to have a strong negative effect on populations of FSD in natural lakes and is probably responsible for the extirpation of populations from the Mirror lakes and Mud Lake (Olson 1998). Recreational interests also result in the repeated stocking of non-native trout species into FSD habitats (Meester 1997b), but these predators appear less likely to threaten the viability of FSD populations, given the continued coexistence of FSD with rainbow trout in Cox Lake. It is unknown, however, whether FSD would fare differently in association with a more piscivorous species of trout such as brown trout.

Mining

Mining currently has little relevance for FSD populations in the Black Hills as it does not constitute a major land-use activity in the range of this species (BHNF 1996, III 441-443).

Prescribed Fire

Controlled burns could positively affect FSD if fires were allowed to burn through riparian areas—thereby removing invasive pines and facilitating the regrowth of aspens and willows needed by beavers. Prescribed fires could also have short-term positive effects on stream and lake populations of FSD by decreasing evapotranspirative losses from trees and increasing the recharge of aquifers that provide outflows to spring-fed lakes and streams (Bosch and Hewlett 1982). However, water-related benefits would be ephemeral and decrease as afforestation of the watershed progressed (Bosch and Hewlett 1982; Troendle and King 1985).

Water Impoundment And Diversion

Given the current status of FSD populations in the Black Hills, water impoundment must be viewed favorably. Although dams presumably had initially negative effects by fragmenting populations, small reservoirs now contain two of the remaining FSD populations and the population in Cox Lake is benefited by headgates that maintain high water levels (J. Erickson, SDGFP, personal communication). Conversely, water diversion by private landowners is probably having a negative effect on FSD populations because this activity often results in impassable stream barriers and contributes to seasonal dewatering of streams. Both stream alterations serve to isolate remaining populations of FSD by hindering the exchange of individuals. When local extirpations occur, otherwise suitable habitats may not be refounded and the range of FSD may become permanently truncated. Another adverse effect of stream diversion occurs when flows are diverted away from sites where aquifer recharge occurs. Kyllonen and Peter (1987) speculated that the municipal diversion of Spearfish Creek around outcrops of the Minnelusa and Madison formations early in the 20th century had decreased water levels in these aquifers. If these decreases were large enough, spring outflows may have been affected.

Natural Disturbances

Floods And Drought

During pre-settlement times, floods and droughts probably had no long-term effects on FSD populations. However, given current landscape conditions and the probable isolation of remaining populations, the effects of hydrologic events could be exacerbated and negative consequences for FSD populations may materialize. Increases to peak flows that often stem from the types of land management activities that have occurred upstream from Geis and Hemler reservoirs (e.g., timber harvest, road building, livestock grazing; Troendle and King 1985; Jones and Grant 1996) or that could occur after a fire or removal of beaver dams may pose added threat to the small dams associated with these reservoirs. If these dams were breeched, it is likely that the associated FSD populations would be extirpated. Increased flooding may also pose some threat to populations that are associated with beaver complexes, but the probability of losing an entire population would seem to be decreased by a series of contiguous ponds—some of which would probably survive most floods. Droughts exacerbated by more rapid runoff and stream diversions for agricultural purposes now appear to dewater much of the stream network in the Redwater Creek drainage on a seasonal basis (Olsen 1998). Events of such high frequency strongly fragment and isolate remaining FSD populations and make it unlikely that extirpated populations will be refounded by dispersal from existing populations.

Wildfire

A large wildfire that burned riparian areas and most of the watershed upstream from a FSD population could have significant short-term benefits similar to those of prescribed fire. However, the removal of large expanses of vegetative cover would also increase the risk of catastrophic flooding and mass soil movements in the event of a large precipitation event (Swanston 1991). Such events would provide significant risks to the populations of FSD that occur in reservoirs as the dams could be breeched or the reservoirs filled with sediment.

SUMMARY

Finescale dace occur in small lakes and in cool, boggy environments that are often associated with springs or beaver dams (Bailey and Allum 1962; Baxter and Stone 1995). Feeding habits are flexible and zooplankton, aquatic insects, and plant materials are consumed from the benthos and water column (Litvak and Hansell 1990). Spawning occurs in spring and breeding males are easily recognized by bright yellow and red colors on their ventral surface (Scott and Crossman 1973; Stasiak 1977). Eggs are immediately fertilized after release by the females and are deposited in depressions under logs or other sources of cover (Stasiak 1978). Fry hatch after 6 days at 20°C and begin feeding a week later when the yolk is absorbed. Sexual maturity is reached at age two for both sexes, although females grow faster, live longer, and reach larger sizes than males (Stasiak 1978). Finescale dace are often found in characteristic fish communities that include northern redbelly dace, brook stickleback, and pearl dace in northern portions of the range (Scott and Crossman 1973) and brassy minnows, fathead minnows, and pearl dace in southern portions of the range (Baxter and Stone 1995). Although FSD have evolved with fish predators, contact with members of the sunfish family appears to have strongly negative effects and may lead to local extirpations of populations (Litvak and Hansell 1990).

The distribution of FSD coincides with the U.S. – Canada border from New England through Minnesota. Additional disjunct populations—including the Black Hills population—are scattered across the Great Plains (Stasiak 1980). The nearest populations of FSD to the Black Hills occur to the south in the Niobrara River drainage (Patton 1997), but interactions between this population and the Black Hills population are unlikely given the length and fragmentation of the intervening stream network.

Surveys suggest the distribution of FSD was historically and is currently limited to a small area in the northern Black Hills (Evermann and Cox 1896; Bailey and Allum 1962; Olson 1998; WGFD Lake and Streams Database). With the exception of one site, all areas where FSD have been sampled occur in the Redwater Creek drainage. In the South Dakota portion of the range, FSD no longer occur at four of the six sites where previous collections have been made and the population trend has been one of decline since European settlement. A trend assessment is impossible to make for the Wyoming portion of the range—where all the known locations of FSD on BHNF lands have occurred—because the historic distribution is largely unknown. Additionally, many FSD populations were not located until the 1960s and recent efforts have not been made to confirm their existence. However, at least four populations—two of which occur on BHNF lands—probably continue to exist within Wyoming.

Risk factors vary based on the type of habitat occupied. In small reservoirs, large floods could breech the small earthen dams that form Geis and Hemler reservoirs or lengthy droughts could dry these habitats. Either type of event would eliminate the lentic habitats that currently support two populations of FSD. In natural lakes and reservoirs, the greatest risk factor is the introduction of sunfishes. Such introductions are probably directly responsible for most of the extirpations that have occurred in South Dakota. For populations of FSD associated with beaver dam complexes, many factors—including land uses that exacerbate runoff, suppression of aspen and willow regrowth by livestock grazing, and beaver populations that appear to be much reduced from historic abundances (Ludlow 1875; Dodge 1965; BHNF 1996, III-253, 254)—may be combining to lower water tables and decreased the amount of boggy, pond-like habitat preferred by FSD. This problem is exacerbated by water diversions on private land and many streams in the Redwater Creek drainage are now seasonally intermittent (Olson 1998). Fragmentation will have increasingly negative effects on FSD as populations disappear because dispersal of individuals needed to refound extirpated populations will be inhibited.

REVIEW OF CONSERVATION PRACTICES

Management Practices

We are not aware of any management practices that have been implemented specifically for FSD in the Black Hills or elsewhere in North America.

Models

No models have been developed for FSD or FSD habitat.

Survey And Inventory Approaches (Presence/Absence)

The presence or absence of FSD in streams can be determined by underwater observation or backpack electrofishing as was discussed previously for MTS. Similarly, most techniques that are suitable for surveying lentic populations of LCB will be suitable for lentic populations of FSD. However, the small size of lakes and reservoirs inhabited by FSD makes gill-netting the only practical alternative for pelagic waters. Minnow traps have also been used by several investigators to collect FSD (Stasiak 1977; Das and Nelson 1990, Litvak and Hansell 1990) and may provide a cheap sampling method. Litvak and Hansell (1990) noted that catch rates were highest when traps were baited with dog food.

Monitoring Approaches (Population Trend, Habitat)

Trends in FSD populations could be monitored as discussed in sections on MTS (pages 54 – 56) and LCB (pages 25 – 28) using catch rate data in lentic waters and either catch rates or population estimates in lotic waters. In streams where beaver activity is high and electrofishing or snorkeling assessments may be difficult, the minnow trap may provide a useful sampling device. These traps could be deployed in a standardized sampling protocol to produce reliable catch rate data, or in some instances, to obtain population estimates using a removal estimator methodology (Bryant 2000).

Little is known about the specific habitat factors that are important to FSD. However, several remaining populations occur in small lakes that are fed by springs. Because the continued existence of these habitats is dependent on spring inflows, monitoring their discharges may provide some indication of future effects on FSD populations. In stream environments, we suspect that FSD habitat is strongly linked to the presence of beavers as their dam building activities produce the cool and boggy types of environments that are preferred by FSD (Scott and Crossman 1973; Stasiak 1977). Therefore, a quantitative assessment of FSD habitat might equate to determining the extent of beaver activity in streams across the range of FSD. Beaver dams and complexes could be mapped based on visual observations made from roads or during foot surveys in more remote areas. To assess trends, the survey could be repeated at various time intervals.

ADDITIONAL INFORMATION NEEDS

The most important information currently needed for the scientifically sound conservation of FSD on the BHNF is a comprehensive survey of streams within the Wyoming portion of the dace range (Table 5). With the exception of Hemler Reservoir, the sites where FSD have been reported from Wyoming have not been sampled recently, including four stream sites on BHNF lands. Given the length of time that has elapsed since many of these populations were sampled, their current existence is uncertain and must be resolved as a precursor to other issues. Once extant populations have been delineated, information regarding the status of individual populations would be useful. Currently, the only inference regarding trend that can be made occurs at a gross level, when individual populations are extirpated, and the distribution of the species becomes more restricted.

Where stream populations of FSD exist, a better understanding of the stream habitat factors that benefit FSD as well as the landscape characteristics that generate FSD habitat would be desirable. Additionally, because stream populations appear to be linked to beaver activity, knowledge of the historical abundance of beaver, their current status, and likely future trends would be informative. Such information could prove valuable in helping to restore the functionality of streams that no longer support FSD. For populations of FSD that inhabit spring-fed lakes and ponds, a better understanding of how current land management activities may be affecting the water table in the northern Black Hills could help identify and alleviate future threats.

Table 5. Additional information needed to support the development of a comprehensive conservation plan for FSD in the Black Hills.

Data	Priority	Justification	Cost[*]
Distribution in Wyoming portion of Black Hills	High	Determine current extent of FSD on Forest Service land	Low
Trend monitoring	High	Allow detection of population declines before population is extirpated	Low
Stream habitat / landscape associations	Intermediate	Provide an understanding of the landscape and habitat attributes that promote the existence of FSD populations	Intermediate
Hydrologic function	Intermediate	Provide an understanding of how land management activities may affect FSD populations in springs	Intermediate
Genetic	Intermediate	Definitively establish purity of Black Hills populations of FSD	Low

[*] Low: estimated cost = $1,000 – 10,000; intermediate: estimated cost = $10,000 – 100,000; high: estimated cost > $100,000

Lastly, the study by New (1962) of FSD hybrids brings the purity of Black Hills populations into question. This matter could be easily resolved by genetic analysis, but it is unclear to us whether a different taxonomic status would change the impetus for the Forest Service to conserve this species. However, if the status of FSD declines to the point that translocations are needed from areas outside the Black Hills, genetic information will then become a high priority item in the identification of suitable donor populations.

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DEFINITIONS

Autochthonous – derived from within a system, such as organic matter in a stream resulting from photosynthesis.

Adfluvial – residing in a lake or reservoir but migrating to spawn in rivers or streams.

Afforested – areas where forest regrowth has occurred.

Agonistic – combative or aggressive.

Allopatric – not co-occurring in the same area.

Anthropogenic – of or related to humans; human caused.

Benthos – animals and plants living on or within the substrate of a water body.

Ephemeral stream – stream that flows briefly and only in direct response to local precipitation, and whose channel is always above the water table.

Demersal – negatively buoyant.

Diel – involving a 24-hour period that usually includes a day and the adjoining night.

Dimorphism – the existence of two different forms (as of color or size) of a species.

Fecundity – the reproductive capacity of an individual, usually measured as the number of eggs produced by a female in a specified period of time.

Fluvial – pertaining to streams or rivers, or migrating between main rivers and tributaries.

Hydrograph – chart that depicts stream discharge rate versus time.

Intermittent stream – stream that flows continuously in wet years.

Lentic – of or in still water such as a lake or reservoir.

Life history – the series of living phenomena exhibited by a fish in the course of its development from conception to death.

Littoral – aquatic zone extending from the shoreline of lakes out to depths where light is insufficient for growth of rooted macrophytes.

Lotic – of or in running water such as a stream or river.

Morphometric – measure that describes the structure or form of an object.

Oocycte – an egg before maturation.

Operculum – bony plate that covers the gills of a fish.

Pelagic – of or in the open water column of lakes.

Perennial stream – stream that flows continuously throughout the year.

Piscivorous – fish-eating.

Reach – section of a stream between two specified points that has a consistent slope and complement of habitat units.

Stochasticity – involving chance or random events.

Stream piracy – geologic event wherein the stream drainage from one watershed is captured or otherwise redirected into an adjacent watershed.

Sympatric – co-occurring in the same area.

Vagile – adept at moving long distances.

Zooplankton – small aquatic animals suspended or weakly swimming in water.