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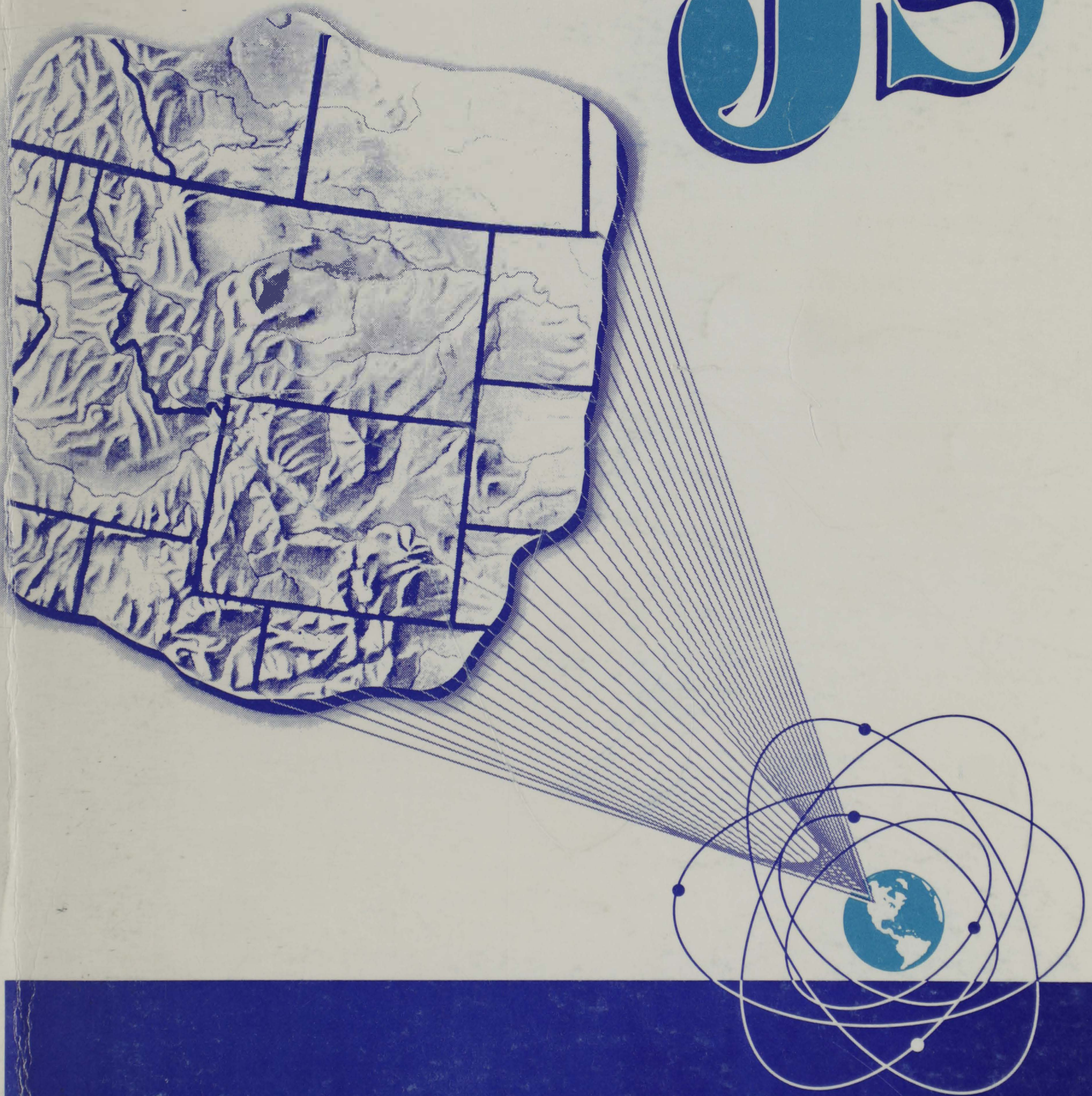

Intermountain Journal of Sciences

The Intermountain Journal of Sciences is a regional peer-reviewed journal that encourages scientists, educators and students to submit their research, management applications or view-points concerning the sciences applicable to the intermountain region. Original manuscripts dealing with biological, environmental engineering, mathematical, molecular-cellular, pharmaceutical, physical and social sciences are welcome.

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Referees and associate editors judge submitted manuscripts on originality, technical accuracy, interpretation and contribution to the scientific literature. Format and style generally follow the *Guidelines for Manuscripts Submitted to the Intermountain Journal of Sciences, Dusek 1995*.* Organization may vary to accommodate the content of the article, although the text is expected to elucidate application of results.

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Manuscripts are submitted to the Editor-in-Chief (EIC) for initial consideration for publication in the IJS. This review shall include, but not be limited to, appropriateness for publication in this journal correct formatting, and inclusion of a letter of submittal by the author with information about the manuscript as stated in the "Guidelines for manuscripts submitted to the *Intermountain Journal of Sciences*" (Dusek 1995). This cover letter must also include a statement by the author that this paper has not been submitted for publication or published elsewhere. The EIC notes the date of receipt of the manuscript and assigns it a reference number, IJS-xxxx. The EIC forwards a letter of manuscript receipt and the reference number to the corresponding author. The corresponding author is the author who signed the submittal letter.

Three hard copies of the submitted manuscript, with copies of the "Guidelines and checklist for IJS referees" attached are forwarded to the appropriate Associate Editor. The Associate Editor retains one coy of the manuscript and guidelines for his/her review, and submits a similar package to each of two other reviewers. A minimum of two reviewers, including the Associate Editor is required for each manuscript. The two other reviewers are instructed to return the manuscript and their comments to the Associate Editor, who completes and returns to the EIC a blue "Cover Form' and all manuscripts and reviewer comments plus a recommendation for publication with or without revisions or rejection of the manuscript. This initial review process is limited to 30 days.

The EIC reviews the recommendation and all comments. The EIC then notifies the corresponding author of the results of the review and the publication decision.

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For accepted manuscripts, each copy of the manuscript containing comments thereon and other comments are returned to the corresponding author. Revised manuscripts are to be returned to the EIC in hard copy, four copies if further review is required, or one hard copy plus the computer disk if only minor revision or formatting is necessary. The revised manuscript shall be returned to the EIC within 14 days of the notification. Review of the revised manuscript by the Associate Editor and reviewers shall be completed and returned to the EIC within 14 days. An accepted manuscript will then be forwarded to the Managing Editor (ME) for final processing.

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Each manuscript that is rejected for publication is returned by the EIC to the corresponding author along with the reasons for rejection. The author is also advised that the manuscript may be resubmitted provided all major criticisms and comments have been addressed in the new manuscript. The new manuscript may be returned to the initial review process if deemed appropriate by the EIC. If the manuscript is rejected a second time by either the EI or the Associate Editor and reviewers no further consideration will be given for publication

of the manu cript in IJS. The corre ponding author will be notified of thi deci ion.

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The identity of all reviewers shall remain anonymous to the authors, called a blind review process. All criticisms or comments by authors shall be directed to the EIC; they may be referred to the ME or the Editorial Board by the EIC for resolution.

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Each manuscript submitted by an Associate Editor shall be reviewed by the EIC and a minimum of two other reviewers with expertise in the subject being addressed. Each manuscript submitted by the EIC shall be forwarded with the necessary review materials to the Chairman of the Editorial Board of IJS, who will serve as the EIC for that manuscript.

Abstracts

Only abstracts from the annual meetings of the sponsoring organizations will be published in IJS. Other submissions of abstracts shall be considered on a case-by-case basis by the Editorial Board. Sponsoring organization shall collect abstracts, review them for subject accuracy, key or can them onto a 3.5 diskette, and ubmit the di kette and hard copy of each abstract to the EIC on or before November 1. Each abstract shall be reviewed by the EIC to assure proper grammar, compliance with IJS Guidelines for Ab tracts Only and for assignment to the appropriate discipline section. All abstracts will be published in the December issue only.

Commentary

Submissions concerning management applications or viewpoints concerning current scientific or social issues of interest to the Intermountain region will be considered for publication in the Commentary Section. This section will feature concise, well-written manuscripts limited to 1,500 words. Commentaries will be limited to one per issue.

Submissions will be peer reviewed and page charges will be calculated at the same rate as for regular articles.

Literature Cited

Dusek, Gary L. 1995. Guidelines for manuscripts ubmitted to the *Intermountain Journal of Sciences*. Int. J. Sci. 1(1):61-70.

Send all manuscripts to: Editor-in-Chief, P.O. Box 3014, Bozeman, MT 59772-3014

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Status of Sauger in Montana

Thomas E. McMahon, Ecology Department, Fish and Wildlife Program, Montana State University, Bozeman MT 59717
William M. Gardner, Montana Fish, Wildlife and Parks, P.O. Box 938, Lewistown, MT 59457

Abstract

Though abundant throughout the Yellowstone and Missouri River drainages in eastern Montana prior to the 1990s, recent survey data suggest that native sauger (*Stizostedion canadense*) have declined throughout the state. We compared historical and recent survey information to assess the extent of the decline, examine possible reasons for the decline, and suggest management options for protecting and bolstering remaining populations. Sauger declined state-wide in concert with the drought in the late 1980s in several locations including the Missouri River, Fort Peck Reservoir, and Yellowstone River. Despite improved flows since the mid-1990s, there was little rebound in sauger numbers in most locales. Of special concern are the substantial declines in spawning runs of sauger in several key spawning tributaries (Marias, Milk, and Tongue rivers). Estimated range of sauger in 1999 was 1570 km, a 53 percent decline from their historical range. Decline of this highly migratory species was more extensive in tributaries (75%) because of loss of migration routes caused by damming and dewatering. The roles of hybridization with walleye (*Stizostedion vitreum vitreum)*, interactions with abundant nonnative piscivores (walleye and smallmouth bass *Micropterus dolomieu*), and angler harvest in the sauger decline are uncertain. Better information is needed on movement patterns, spawning ecology, fish passage problems, and angler harvest to develop management actions for sauger recovery.

Key words: fishery management, Montana fishes, sauger, *Stizostedion canadense*, Missouri River, Yellowstone River.

Introduction

Sauger (*Stizostedion canadense*), a percid fish closely related to walleye (*S. vitreum vitreum*), is one of the most widely distributed of North American fishes, occupying large, turbid rivers and lakes throughout central and eastern North America from northern Canada to Alabama, and westward to the upper Missouri River drainage (Scott and Crossman 1973). The species was first described during the Lewis and Clark expedition in the early 1800s from the Missouri River near the mouth of the Marias River, Montana (Moring 1996). Historical distribution in Montana was the Missouri River and its major tributaries below Great Falls, and the Yellowstone River and its major tributaries below and including the Clarks Fork (Brown 1971, Holton and Johnson 1996). Both Missouri and Yellowstone river populations have supported popular sport fisheries.

The first intensive sampling efforts conducted in the Missouri River below Morony Dam and the Yellowstone River below the mouth of the Tongue River in the 1960s and 1970s found sauger abundant and widespread in eastern Montana. Posewitz (1963) captured large numbers of sauger spawning in the Marias River in the early 1960s, and Berg (1981) and Gardner and Berg (1982) found sauger common-to-abundant in the lower Marias, Judith, and Teton rivers and in the Missouri River between Great Falls and Fort Peck Reservoir. Large spawning congregations of sauger also were noted in the lower Tongue and Powder rivers in the Yellowstone drainage (Elser et al. 1977, Rehwinkle 1978).

Apparently widespread declines of sauger in both the Yellowstone and

Missouri drainages were reported in the early 1990s. Penkal (1992) described a decline in the spawning run in the lower Tongue River, and Stewart (1992) reported a >85 percent decline in age-0 and adult sauger in the lower Yellowstone River from the 1980s to the 1990s. Declines of similar magnitude also were observed in the Missouri (upstream from Ft. Peck Reservoir) and Marias rivers (Penkal 1990, Gardner 1998, Hill et al. 1998). Though severe drought in the late 1980s was thought to have triggered the sauger decline (Penkal 1990), an apparent lack of rebound in sauger abundance despite improved flow conditions in the mid-1990s raised concern over the status of sauger in the state (Gardner 1998, Stewart 1998). Population collapses have occurred in other parts of the native range of sauger including Nebraska (Hesse 1994), the Great Lakes (Rawson and Schell 1978), and the Tennessee River system (Pegg et al. 1996, 1997). High exploitation, water flow fluctuations, migration barriers, hybridization with walleye, and loss of spawning habitat were implicated in these declines (Hesse 1994, Pegg et al. 1997).

Concern over the status of sauger in Montana provided the impetus for this synthesis of historical and current information on its distribution and abundance. Our objectives were to summarize available information about this little-known species, compare past and present abundance and distribution data to assess the extent of a sauger decline, examine possible reasons for documented declines, and suggest management options for protecting and bolstering remaining populations of this native fish and the sport fisheries they support.

METHODS

We assessed the status of sauger in five main areas corresponding to their historical range in Montana: the Missouri River and tributaries upstream of Fort Peck Reservoir, Fort Peck Reservoir, the Missouri River from Fort Peck Dam downstream to the North Dakota border, the Yellowstone River and a major tributary, the Bighorn River, and the Yellowstone River and tributaries from the North Dakota border to the mouth of the Bighorn River (Fig. 1).

We obtained data on sauger abundance and distribution from published and unpublished fishery survey reports and from

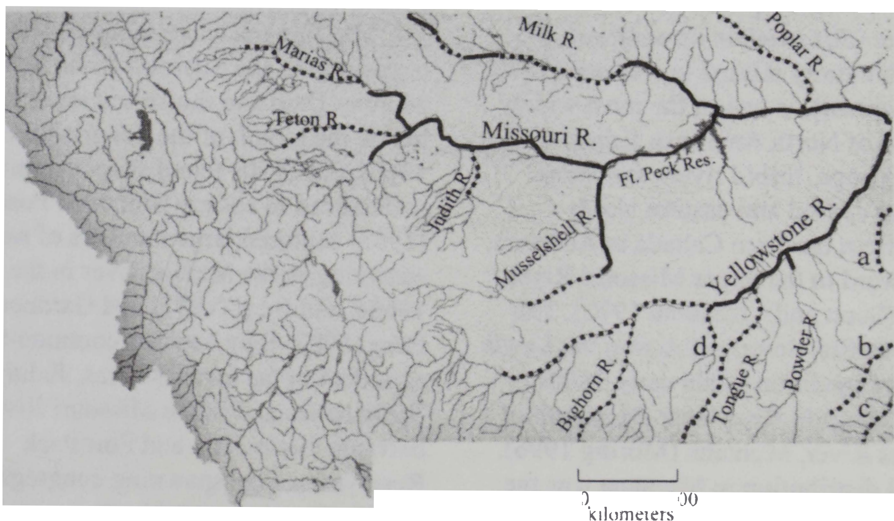

Figure 1. Estimated historical and present distrion of sauger in Montana. Solid line indicates areas where sauger are still present, and dashed line indicate areas where sauger were likely present historically but are now rare or absent. Unnamed drainages, a= Beaver Creek, b = Box Elder Creek, c = Little Missouri River, d = Rosebud Creek.

unpublished data provided by biologists contacted throughout the state. Abundance data were summarized as number caught/hr or km of electrofishing or number caught/net based on gill net or seine haul population surveys. If several areas were sampled in the same river section, data from all sections were averaged (McMahon 1999). Though such effort data do not provide actual abundance estimates, they can nevertheless provide insight into population trends when collected over time using standardized sampling schemes (Ney 1993). Much of the data on sauger abundance trends used here were collected over periods of at least 9 years using a consistent sampling protocol. We evaluated possible associations between sauger abundance and river discharge using simple linear correlation based on U.S. Geological Survey discharge records for the Missouri and Yellowstone rivers *(www.montana.usgs.gov)*.

We assessed overall status of sauger by comparing estimated historical with present distribution. Historical information was based on published range maps (Brown 1971, Holton and Johnson 1996), initial extensive surveys conducted in the 1970s (e.g., Berg 1981), and historical descriptions of habitat conditions relative to sauger habitat requirements. We excluded some streams from the analysis where sauger are now rare or absent but where historical information on their distribution was lacking (Poplar River and Beaver Creek in the Missouri drainage, Little Missouri River, Box Elder Creek, and Rosebud Creek in the Yellowstone drainage). Present-day range was based on the fish collection database by Montana Fish, Wildlife, and Parks (FWP) *(www.nris.state.mt.us)*, and on expert opinion from FWP biologists.

Past and Present Distribution in Montana

Missouri River Upstream of Fort Peck Reservoir

Mainstem.—Sauger abundance has been monitored since the 1970s for the river section between Morony Dam, near Great Falls, and the mouth of the Marias River (Fig. 1). In 1978-1980, sauger catch averaged 25.9 fish per hour of electrofishing (Fig. 2). In 1979, for example, 120 sauger were caught in 3 hours of sampling (Penkal 1990). Following a 7-

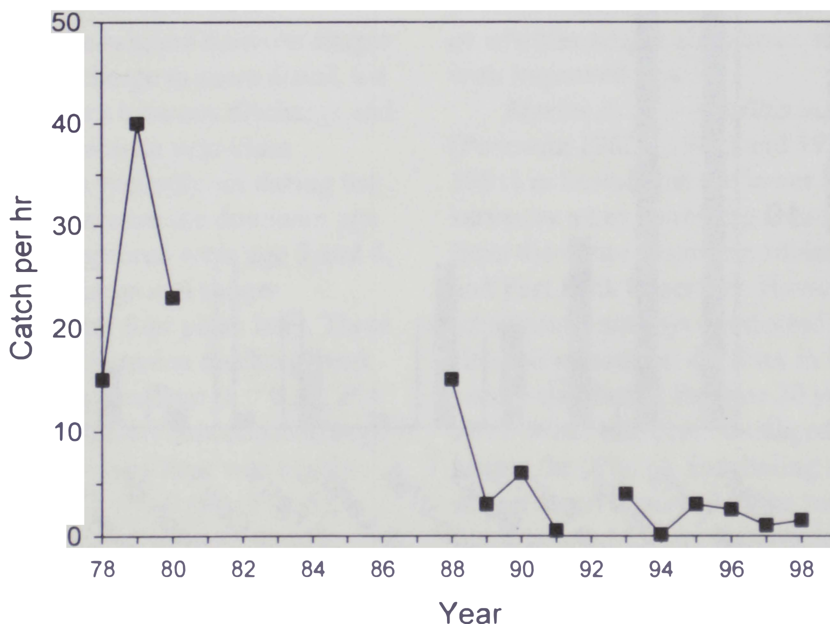

Figure 2. Electrofishing catch rate of sauger, Morony Dam to Marias River sampling section, Missouri River, 1978-1998.

yr gap in sampling, sauger catch rate was <6 fish/hr in 1989-1998. In the 1993-1998 period sauger catch averaged 2.2 fish/hr, a 90 percent decrease in average catch rate from the 1978-1981 period. In 1997 only nine sauger were caught in 8.2 hours of electrofishing (Hill et al. 1998). Abundance patterns of other sport fishes (rainbow trout *Oncorhynchus mykiss*, brown trout *Salmo trutta*, mountain whitefish *Prosopium williamsoni*, walleye, smallmouth bass *Micropterus dolomieu*) show high year-to-year variation (Hill et al. 1998), but none exhibit a comparable decline, and some (smallmouth bass, walleye) increased over this period. Sauger were the most abundant sport fish in the late 1980s (65% of total sport fish catch) but were one of the rarer sport fishes sampled (<9% of the catch) during the 1994-1997 sampling.

More extensive surveys of the mainstem Missouri between Morony Dam and the headwaters of Fort Peck Reservoir conducted in the 1970s found sauger one of the most abundant and widely distributed fish species (Berg 1981; Fig. 3). Of the 9835 fish captured by electrofishing, 2916 were sauger or 29.6 percent of the total catch. Catch rate in recent surveys (Fig. 3) has declined considerably in the upper river, averaging 70 percent fewer sauger than surveys during the 1970s (Gardner 1997, 1998, L. Bergstedt, Montana State University, personal communication). In contrast sauger abundance in the lower river was mostly higher than earlier surveys.

Previous studies have documented a positive correlation between discharge and sauger year-class strength (Nelson 1968, Fischbach 1998). Average daily discharge for the spring-summer period in the Missouri River at Virgelle from 1958 to 1998 indicated that discharge was below the 41-year average of 311 m^3/s in 7 of the last 14 years, particularly during the period of 1985-1994 when 7 of 10 years were below the long term average (Fig. 4). Discharge in 1988 and 1992 was the lowest over this 41-yr period. Summer flows in these years were below the 153 m^3/s minimum flow deemed necessary to prevent dewatering of Missouri River sidechannels, a key rearing

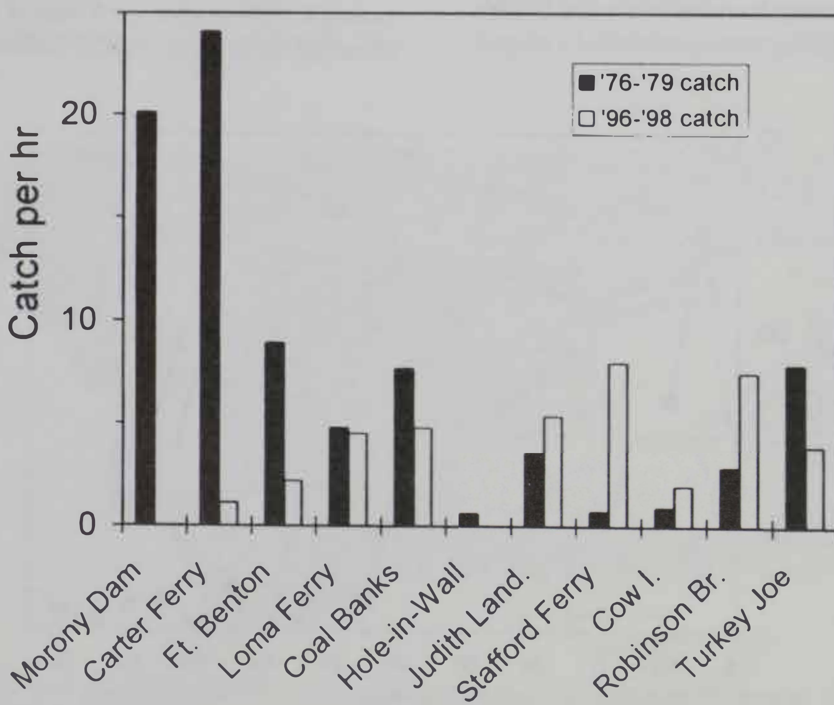

Figure 3. Comparison of mean electrofishing catch rates for sauger in sampling sections of the Missouri River from Morony Dam to the headwaters of Fort Peck Reservoir, 1976-1979, and 1996-1998.

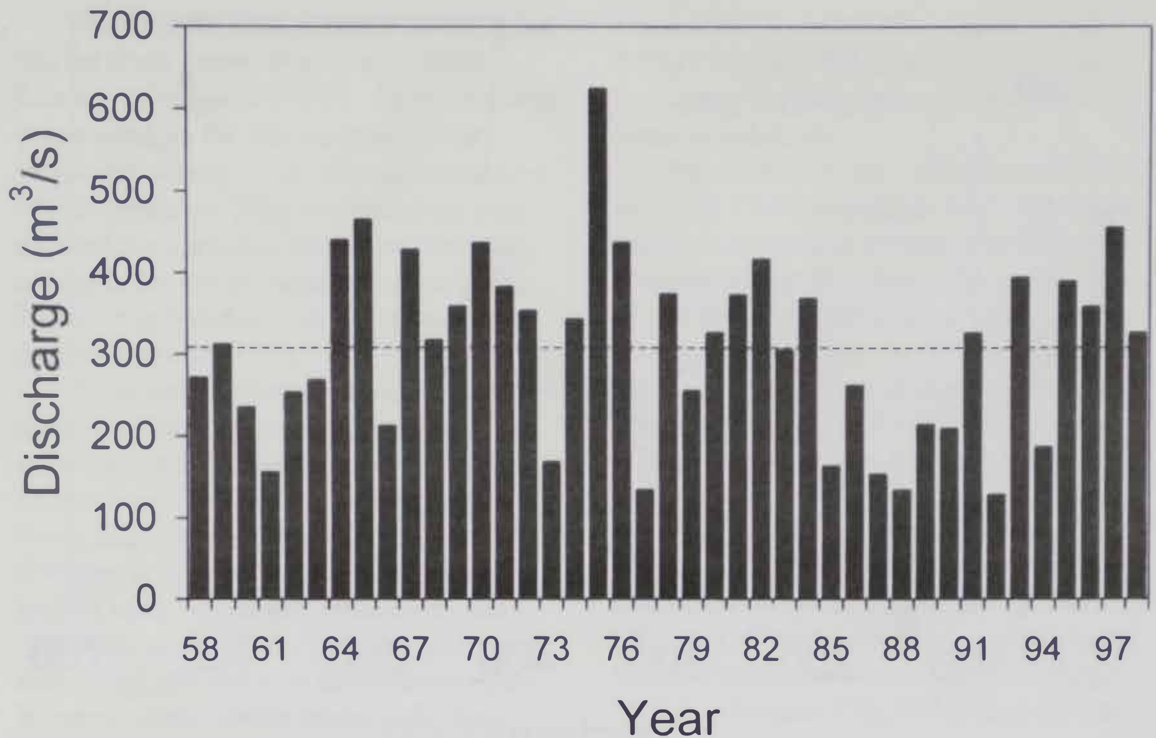

Figure 4. Mean spring-summer (1 May-30 September) discharge of the Missouri River at Virgelle, 1958-1998. Forty-one year average shown as dashed line.

habitat for age-0 sauger (Gardner and Berg 1982). The decline in sauger catch therefore coincided with the initiation of drought in the late 1980s (Penkal 1990); however, flows have been well above average since 1995 but sauger abundance has remained low.

To examine the relation between sauger abundance and discharge in more detail, we tested for a lag effect between discharge and sauger catch rate because year-class strength in fishes is typically set during the first year of life. Because the dominant age classes of sauger captured were age 3 and 4, we compared discharge and sauger recruitment three and four years later. There was no association between discharge and sauger catch three years later ($r = 0.27$, $P = 0.38$), but the correlation between discharge and sauger catch 4 years later was highly significant ($r = 0.87$, $P < 0.001$; Fig. 5), indicating that higher discharge should result in higher sauger abundance. However, sauger catch rates in recent years apparently have not responded to increased flows. Sauger catch rates in 1995 and 1997, corresponding to high flow years 1991 (mean daily discharge, 330 m^3/s) and 1993 (400 m^3/s), respectively, were 80 percent below the value predicted by the regression equation, suggesting additional factors were involved in continued low numbers. A series of flows >280 m^3/s since 1995 should provide a critical test in the next few years of whether sauger abundance will rebound with improved flows.

Marias River.—Studies in the 1960s (Posewitz 1962a, 1963) and 1970s (Berg 1981) indicated that the lower Marias River serves as a key spawning area for sauger from the entire mainstem Missouri River and Fort Peck Reservoir. However, intermittent surveys conducted since 1978 showed substantial declines in the spring sauger run during the past 20 years. In 1979-1982, catch rate averaged about 30 sauger /hr (Fig. 6). Paralleling the decline in sauger catch observed in the mainstem reach below Morony Dam, catch rate declined by about 50 percent to an average of 14.0 sauger/hr during the low water years of the late 1980s (Gardner 1998). In 1996-1999, sauger catch averaged about 3 fish/hr, a further 75 percent decline.

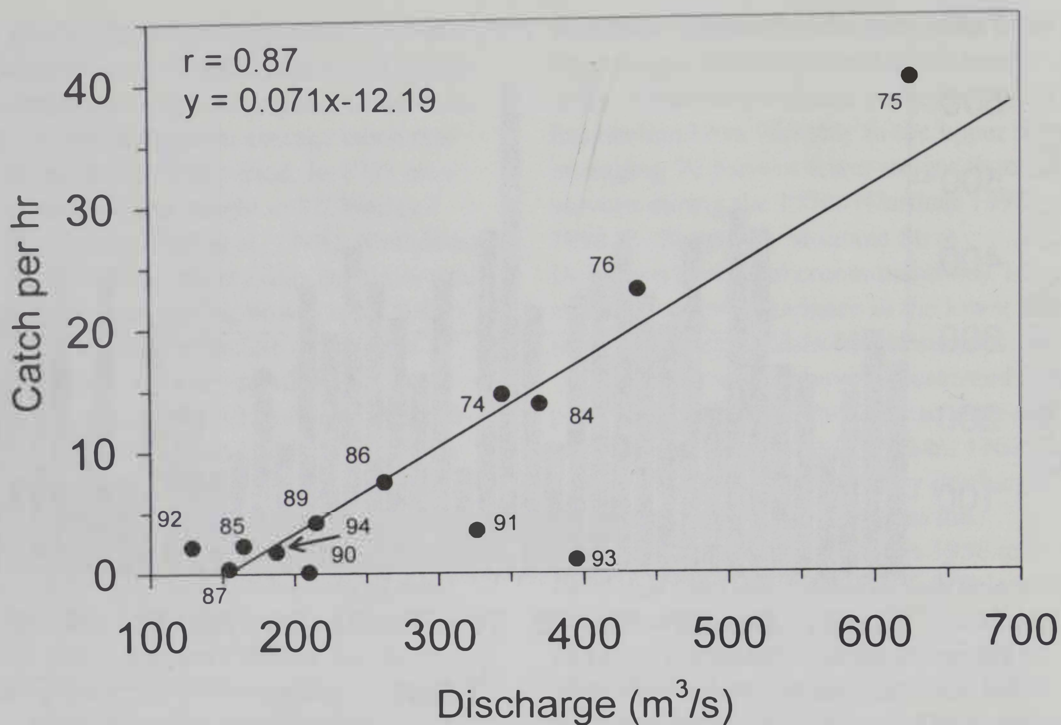

Figure 5. Relationship between mean spring-summer discharge of the Missouri River at Virgelle and sauger catch rate below Morony Dam four years later.

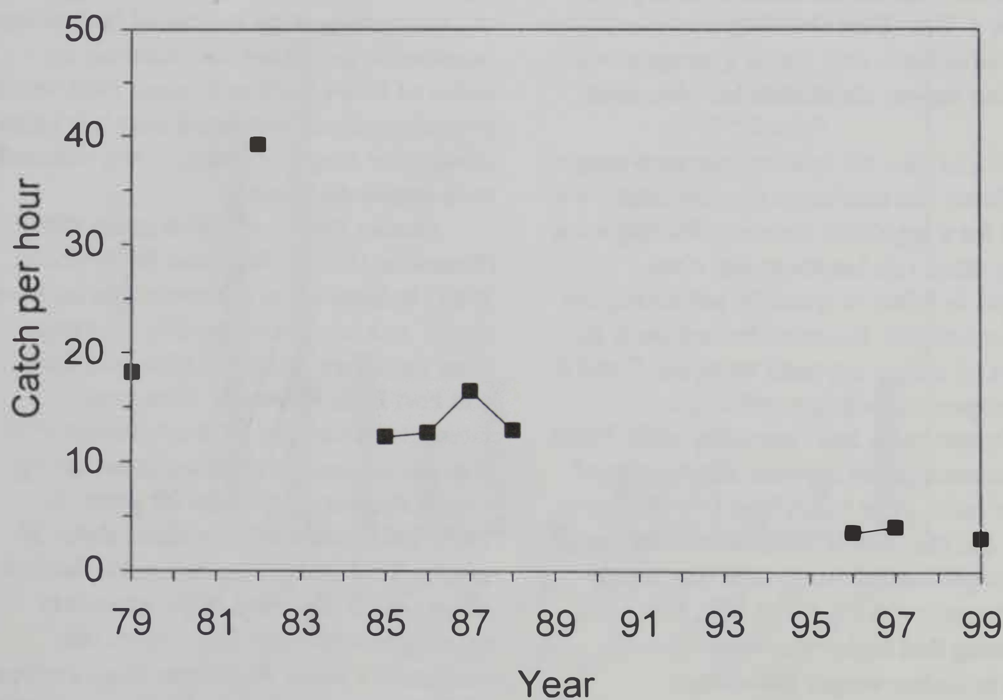

Figure 6. Electrofishing catch rate of sauger in the lower Marias River sampling section during the April-May spring spawning period, 1979-1999.

Sauger were once common in the upper Marias River above what is now Tiber Reservoir (Posewitz 1962b). Tiber Dam was constructed on the Marias River in the 1950s, and a large-scale chemical treatment of the upper river prior to closing the dam resulted in extirpation of sauger above the reservoir. No sauger were collected in this 100-km reach during early 1990s surveys (Liknes and Hill 1994).

Teton River.—Sauger were common in the lower Teton River during spring spawning surveys conducted in the late 1970s (Berg 1981) and in a survey of the lower 120 km conducted during fall 1979 (Gardner and Berg 1982). There has been limited sampling on the Teton River since 1979, but several lines of evidence suggest that sauger are now rare in the entire river. In recent years (1997), sauger have been absent from angler catches, and they were absent from the lower river section in 1997 unlike the 1979 survey which found them common (Gardner 1998). Irrigation demand severely dewaters most portions of the lower Teton River (Berg 1981, Gardner 1998).

Judith River.—Berg (1981) electroshocked a "significant number" of spawning sauger in the lower Judith River in May 1979. Sauger were common during a fish survey conducted by Gardner and Berg (1982) in fall 1979, but no surveys have been conducted since and current status is unknown.

Musselshell River.—Wiedenheft (1980) surveyed 11 4-km sections over 350 km of the river in summer 1979 and found sauger common below the town of Musselshell but absent above a local water diversion dam. No data are available on current status of the sauger population as high turbidity and low conductivity limit electrofishing effectiveness. Anglers target a sauger spawning run up the Musselshell River in the spring when they congregate below the Musselshell diversion dam, but where and how many sauger spawn in this system are unknown. Chronic dewatering of the lower Musselshell limits its suitability as sauger habitat (Montana Fish, Wildlife, and Parks 1997).

Fort Peck Reservoir

Sauger abundance in Fort Peck Reservoir has been monitored since 1980 via a nearly-annual series of extensive shoreline seine hauls and gill netting. Sauger age-0 abundance in beach seine surveys averaged 0.4 fish/seine haul from 1981 to 1986 but dropped sharply in the late 1980s coincident with low water levels (Brunsing 1998; Fig. 7). Water levels have

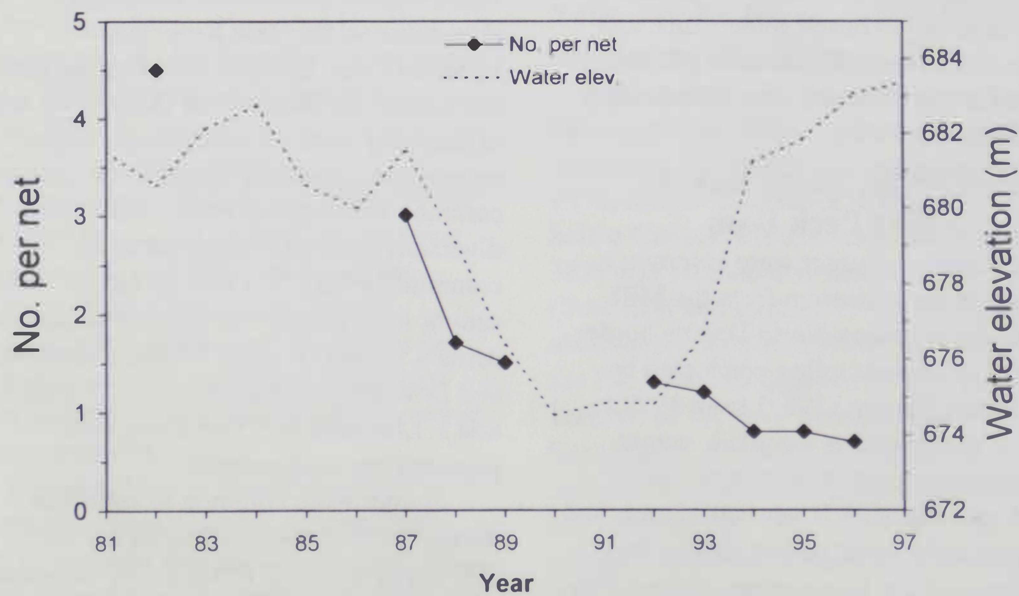

Figure 7. Number of sauger caught per gill net and water level elevation in Fort Peck Reservoir, 1981-1996.

risen steadily since the early 1990s, but abundance has remained low, averaging 0.1 fish/haul, a 75 percent average decrease from the 1980s.

Gillnet catches of sauger mirrored the decline in age-0 abundance (Fig. 7). Adult sauger abundance declined steadily from 3-4.4 fish/net in the 1980s (Needham and Gilge 1983, Wiedenheft 1989) to <1.0 fish/net since 1994. In contrast, northern pike (*Esox lucius*), walleye, and yellow perch (*Perca flavescens*) have shown marked increases in abundance over the past decade (Brunsing 1998). The number of sauger caught by anglers also decreased from 1990 to 1997. An estimated 3128 sauger were harvested in the summer 1990 creel survey, or 12 percent of the total catch. In 1997, 917 sauger were harvested, representing 2.6 percent of the total catch. The reservoir-wide angler catch rate of sauger declined by 75 percent, from 0.023 to 0.005 fish/hr. Average size of sauger harvested was similar in each year (50 cm and 1.2 kg), in contrast to other piscivores (walleye, northern pike) that have increased significantly in abundance and growth since the early 1990s following introduction of cisco (*Coregonus artedi*) (Brunsing 1998).

Sauger are most abundant in the more turbid and riverine-like Missouri Arm in the upper reservoir. Greater than 90 percent of sauger captured in beach seine hauls, and >78 percent of those captured in gill nets, occurred in the Missouri Arm (Wiedenheft 1990, Brunsing 1998).

Missouri River, North Dakota Border to Fort Peck Dam

Mainstem.—Sauger were common to abundant in the mainstem from the Milk River to the Montana-North Dakota border during extensive sampling conducted by Gardner and Stewart (1987) from 1979 to 1983. Of eight reaches sampled, sauger were most abundant in the warm, turbid section near the Milk River confluence, and rare in the cold, clear section below Fort Peck Dam. Of the five species of sport fish collected (shovelnose sturgeon *Scaphirhynchus platorynchus*, northern pike, burbot *Lota lota*, walleye, and sauger), sauger were the most abundant, comprising 69 percent or 3612 of the total 5206 sportfishes captured. Only limited sampling has been done since, but more recent surveys suggested that sauger numbers are about 50 percent lower than in the early 1980s. Electrofishing catch rates in the section from the Milk River confluence to Wolf Point were 1.5 sauger/hr in 1998 compared to an average of 5.2 /hr in 1979-1983 sampling (M. Ruggles and D. Fuller, FWP, Fort Peck, MT, personal communication). The section from Wolf Point to the Yellowstone River confluence yielded 2.4 sauger/hr in 1998, compared to 4.7/hr in 1979-1983.

Milk River.—The Milk River is a large tributary that extends from the Missouri River just below Fort Peck Dam northwestward into Canada. Its high turbidity, deep pools, and gravelly riffles characterize high quality habitat for sauger, and historically it likely supported an abundant resident population (Gardner and Stewart 1987). Seven major water diversions now occur on the Milk River beginning with the Vandalia Dam 187 km from the mouth, and all are considered migratory barriers to sauger (K. Gilge, FWP, Havre, MT, personal communication). There is little information on sauger abundance in the Milk River above Vandalia Dam. Limited sampling suggests that sauger are uncommon throughout much of this long river section, though angler reports indicated they may still be locally common between the Havre and Dodson diversion dams (K. Gilge, personal communication). In 1998, no sauger were caught from among the 652 fishes captured using a variety of gears between Vandalia and Dodson dams (M. Ruggles, D. Fuller, and J. Liebelt, FWP, Fort Peck, MT, personal communication).

Sauger were common to abundant during 1979-1984 sampling in the lower 100 km of the Milk River below Vandalia Dam (Gardner and Stewart 1987). Gill net catch averaged 2.4 sauger/net, and spring

electrofishing yielded 10.2 sauger/hr. Overall, 918 sauger or 91 percent of the total catch of 1024 sport fish, were captured in 4 years of sampling. Large numbers of spawning sauger were found near the mouth; this reach is thought to support the majority of total spawning activity of sauger from the Missouri River between Fort Peck Dam and Lake Sakakawea in North Dakota (Gardner and Stewart 1987). Sampling has been limited since the 1980s, but gill netting and electrofishing surveys conducted in 1998 suggest sauger may be less abundant than in the past. Catch rates of sauger in 1998 averaged 0.3 sauger/gill net, and no sauger were caught during electrofishing surveys (M. Ruggles, D. Fuller, and J. Liebelt, personal communication).

Yellowstone River Upstream of the Bighorn River

Mainstem.—Historically, sauger were likely common in the Yellowstone River upstream to what is now Billings (Brown 1971) and have been collected as far upstream as the town of Big Timber (Swedberg 1984). They also may have occurred historically in the turbid Clarks Fork tributary (Holton and Johnson 1996). However, they are now uncommon above the Bighorn River confluence (Fig. 1). The Huntley diversion dam near Billings is a migratory barrier (Swedberg 1985), and electrofishing surveys between Huntley diversion and the Bighorn River confluence have yielded few sauger in recent years (Poore 1990, K. Frazer and M. Vaughn, FWP, Billings, MT, personal communication). Haddix and Estes (1976) noted that the mouth of the Bighorn River was a popular and productive area for sauger angling in the 1970s. Sauger numbers in the 1990s were thought to be much lower in this reach based on many fewer anglers (Frazer and Vaughn, personal communication).

Bighorn River.—Given its turbid, warmwater characteristics, sauger likely were historically abundant in the Bighorn River. Sauger were moderately abundant in the lower Bighorn River prior to completion of the Bighorn Dam (river km 128) in 1965, averaging 5.6 fish/km and ranging from 3 to 18 percent of the total catch (Bishop 1967). Sauger were rare during the 1990s; periodic electrofishing surveys from 1991 to 1996 yielded few sauger (<0.4 fish/km) (K. Frazer and M. Vaughn, personal communication). In the past sauger concentrated in good numbers during spring in the lower 2 km of the Bighorn (Stewart 1987), but more recently, concentration in this reach was much reduced with only one sauger electrofished in spring 1996 and none in spring 1999 (Frazer and Vaughn, personal communication). Rancher diversion dam just below the mouth of the Bighorn River, and Manning diversion dam 6.5 km upstream of the mouth, likely restrict movement (Montana Fish, Wildlife and Parks 1997), and reservoir release of colder, clearer water has reduced habitat suitability for sauger throughout the lower Bighorn River.

Sauger are moderately abundant in the upper, more turbid portion of Bighorn Reservoir (Kreuger et al. 1997) and relatively rare elsewhere (Frazer and Vaughn, personal communication). Sauger migrate out of the reservoir 80 km upstream to spawn in the Bighorn River in Wyoming, the last remaining sauger population in that state (Kreuger et al. 1997).

Yellowstone River, North Dakota Border to the Bighorn River

Mainstem.—Sauger were abundant throughout the Yellowstone River during surveys in the 1970s. For example, in spring 1974, sauger abundance in the river section below Cartersville diversion dam (river km 383) was estimated at 1265 fish per km (Peterman and Haddix 1975). However, abundance has declined substantially since the late 1980s throughout all five urvey sections of the river between the Cartersville diversion dam at the town of Forsyth and the Intake diversion dam (river km 118). In the 1970s and 1980s, fall sauger abundance averaged about 12 fish/hr (Fig. 8). Sauger abundance dropped harply beginning in 1987, and since 1990 ha

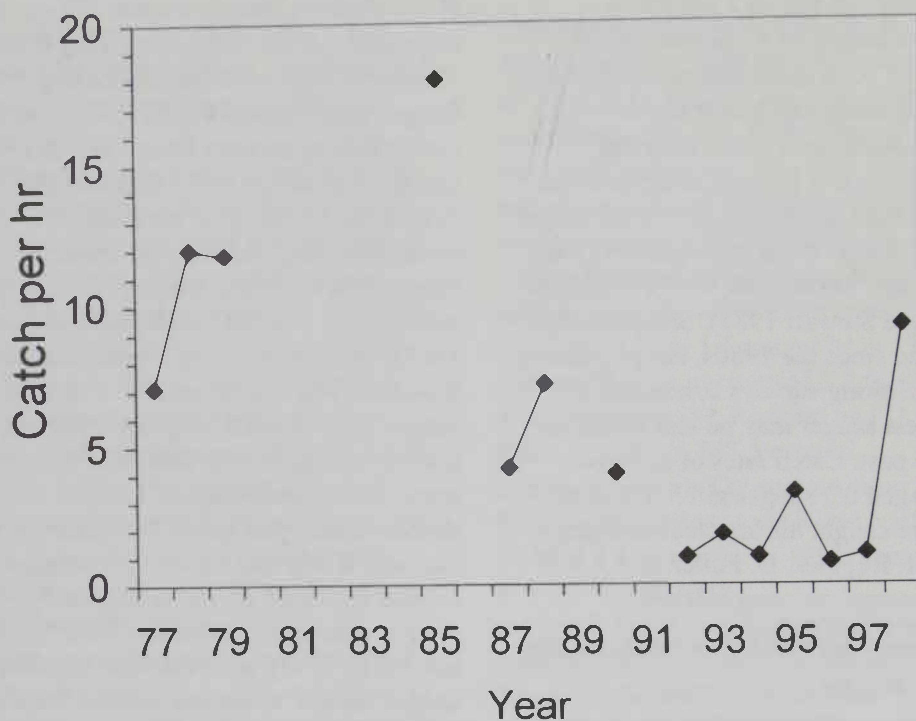

Figure 8. Mean electrofishing catch rates of sauger in the lower Yellowstone River, 1977-1998.

averaged about 2 fish/hr, an 83 percent drop in average abundance. An illustration of the decline is shown by a catch of 358 sauger in 3 days of sampling below Intake in fall 1985, and only 20 sauger in 2 days of sampling the same section in fall 1997 (Stewart 1998). An exception to this pattern is the catch rate in fall 1998, when the sauger abundance averaged 9.2 sauger/hr. Above the Cartersville Dam, sauger abundance has remained at low levels (~10% of downstream abundance) due to restricted passage (Stewart 1998).

Initiation of sharp declines in sauger abundance in the Yellowstone River coincided with low water levels. Average spring-summer discharge of the Yellowstone River at Sidney from 1987-1990 was well below the 48-year average of 506 m^3/s when the sauger decline began (Fig. 9). Sauger catch rate was positively correlated with spring-summer discharge 3 years earlier ($r = 0.52$, $P = 0.058$; Fig. 10). Sauger catch was lower than expected in the early 1990s, but there was some evidence for a positive response to higher discharge in 1995 (Fig. 10).

Spawning for the entire Yellowstone River may be confined to two tributaries, the Powder and Tongue rivers. Penkal (1992) electroshocked only two mature sauger in a 83-km section of the mainstem from Miles City to Forsyth during early May when sauger were abundant in the Tongue and Powder rivers. The only other documented spawning in the lower Yellowstone was at a few sites below Intake diversion (Penkal 1992). Walleye, in contrast, spawn in numerous locations below Intake (Penkal 1992).

Sauger larvae drift downstream from the Tongue and Powder River spawning grounds to rear in the lower mainstem and in Lake Sakakawea, and gradually move upstream in late summer and fall (Penkal 1992). Age-0 abundance in the lower Yellowstone varies widely but appears positively associated with Lake Sakakawea water levels (Stewart 1996a). However, age-0 abundance has been low since 1995 despite higher discharge and reservoir water levels (Stewart 1998). Recent evidence

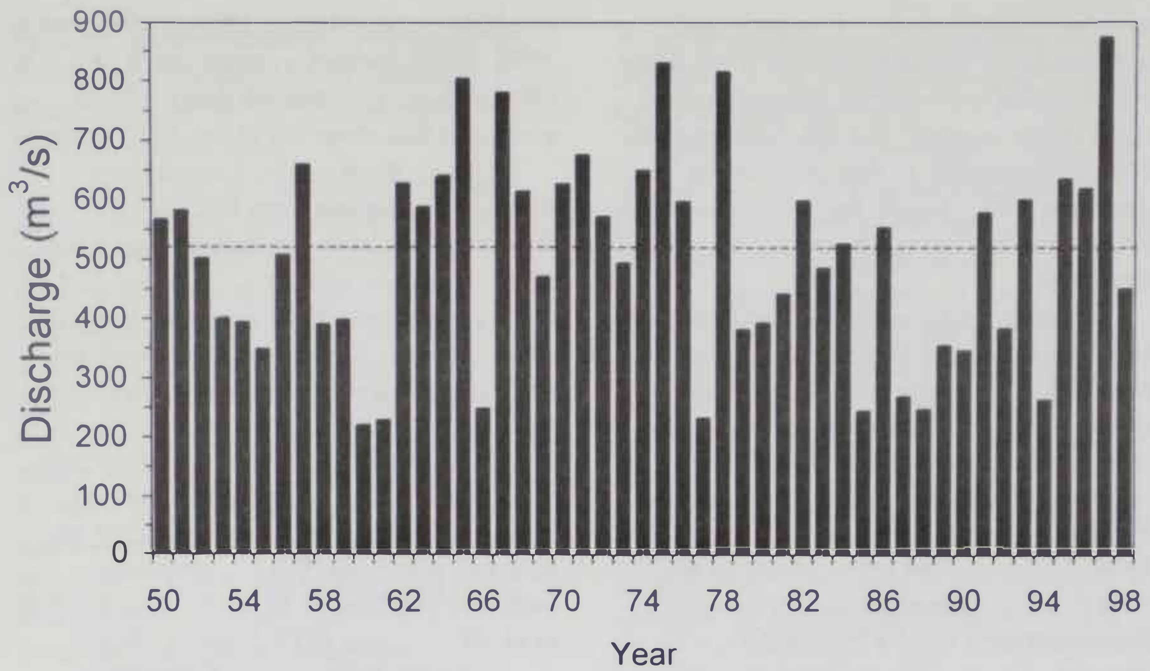

Figure 9. Mean spring-summer (1 May-30 September) discharge of the lower Yellowstone River at Sidney, 1950-1998. Thirty-nine year average shown as dashed line.

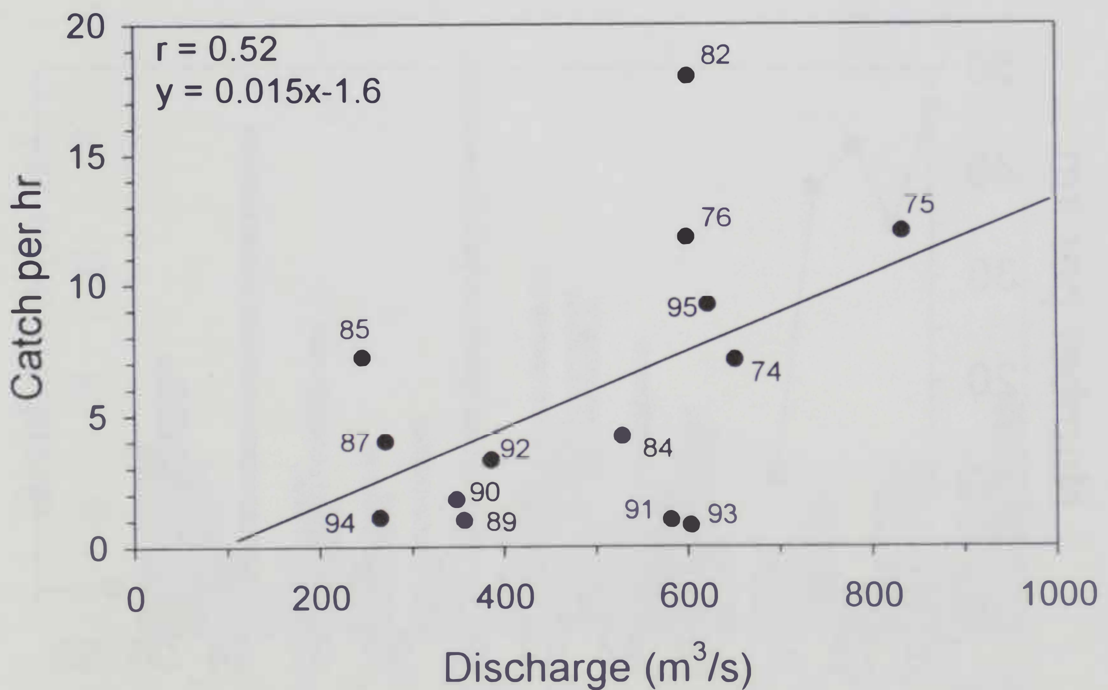

Figure 10. Relationship between mean spring-summer discharge of the Yellowstone River at Sidney and average sauger catch rate in the lower Yellowstone River three years later.

suggests that thousands of juvenile and adult sauger in the lower Yellowstone River are entrained in the Intake water diversion canal in late summer and fall (Hiebert et al. 2000), presumably as they are moving upstream. The ultimate fate of entrained fish is unknown but we would expect significant mortality.

Powder River.—The Powder River is a large prairie stream that is characterized by high turbidities, little pool development, and shifting sand substrate. Surveys conducted in the 1970s indicated sauger were historically uncommon throughout the 336-km drainage in Montana (Rehwinkel 1978). However, the lower 16 km of the river is a key spawning area for sauger. During spring 1976-1979 sampling, 620 sauger or 5.6/hr, were captured during electrofishing surveys in this section; many fish caught were mature (Rehwinkel 1978, Penkal 1992). Fish tagged during spawning were recovered throughout the lower 150 km of the lower Yellowstone mainstem indicating that long-distance movement to spawning grounds was common. No sampling occurred from 1979 to 1997, but a 1998 survey yielded a sauger catch of 6.9/hr (W. Gardner, unpublished data), a level similar to that observed in the 1970 surveys.

Tongue River.—The Tongue River flows 325 km north from the Montana-Wyoming border to the Yellowstone River near Miles City, Montana. There are five dams on the river, from the T and Y water diversion dam, 33 km from the mouth, to the Tongue River Dam near the Wyoming border. The upper river extends another 100 km above the Tongue River Reservoir into Wyoming. Sauger likely were abundant historically throughout the system but are now rare above the T and Y diversion, including the Tongue River Reservoir (Elser et al. 1977, Riggs 1978, Stewart 1996b).

The river below T and Y diversion supported a strong spawning run of sauger, averaging about 40 fish/km during spring electrofishing surveys conducted in the 1970s (Fig. 11; Elser et al. 1977, Penkal 1992). Though there is a gap in data from 1981-1991, surveys since 1991 indicate the sauger spawning run has declined markedly

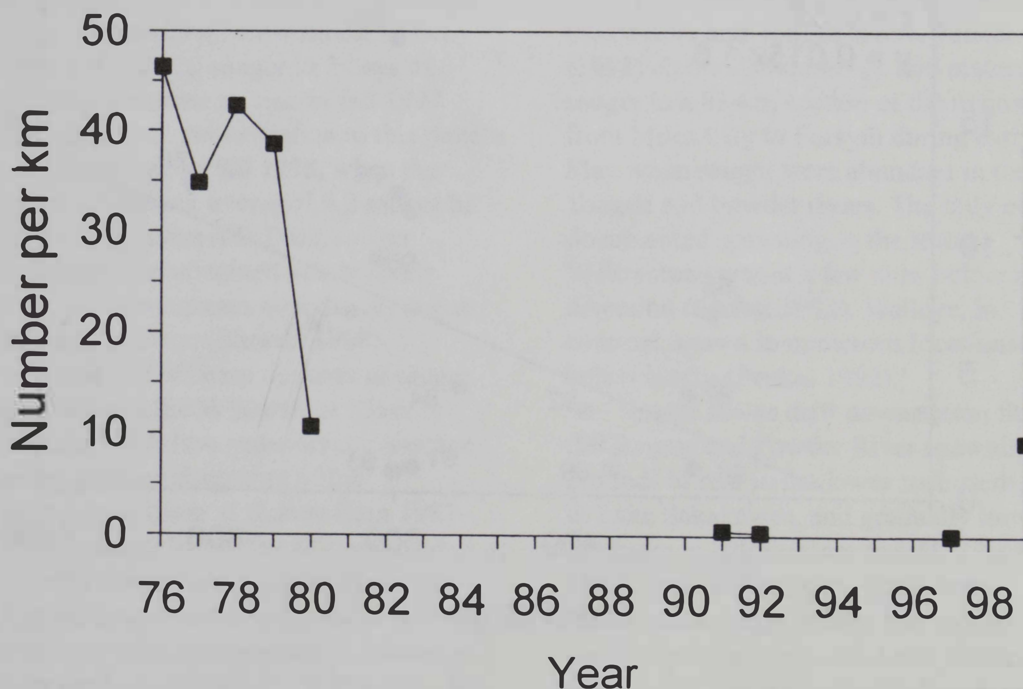

Figure 11. Spring electrofishing catch of sauger below T and Y diversion dam, lower Tongue River, 1976-1999.

from historical abundance, averaging < 5 fish/km or 90 percent fewer fish than observed in the 1970s (Stewart 1993, Gardner, unpub. data).

Chronically low spring water levels likely have had a significant effect on spawner abundance and perhaps reproductive success. Elser et al. (1977) calculated a desired passage and spawning flow level of 15 m^3/s during April for successful sauger reproduction, and Penkal (1992) observed a precipitous decline in sauger spawning when flows dropped below 8.5 m^3/s. During the high spawning run years of 1976-1979, April discharge was typically >12 m^3/s. However, since 1980 average April discharge has met or exceeded the recommended flow of 15 m^3/s in only 5 years (Fig. 12). In addition, daily flows in 9 years were <8.5 m^3/s for >15 days (McMahon 1999). Discharge levels in some recent years have approached those of the late 1970s, but a rebound in the spawning run has not been observed.

Whether other spawning areas have been colonized as the run in the Tongue River has declined is unknown.

Little Missouri Drainage.—The Little Missouri River system, located in the far southeastern corner of the state, drains northward into Lake Sakakawea in North Dakota. Montana portions of three of the larger tributaries, Box Elder, Little Beaver, and Beaver creeks, were surveyed in the late 1970s (Elser et al. 1980) and early 1990s (Barfoot 1993, Guzevich 1993). Sauger were conspicuously absent from among the 25 species found in Little Beaver and Beaver creeks, but were present in Box Elder Creek. Holton and Johnson (1996) reported the presence of sauger in Beaver Creek and Box Elder Creek. However, based on the above surveys sauger apparently are now absent from Beaver Creek. Box Elder Creek has not been sampled in recent years, so the status of sauger in that system is unknown. It is likely that these tributaries were used by

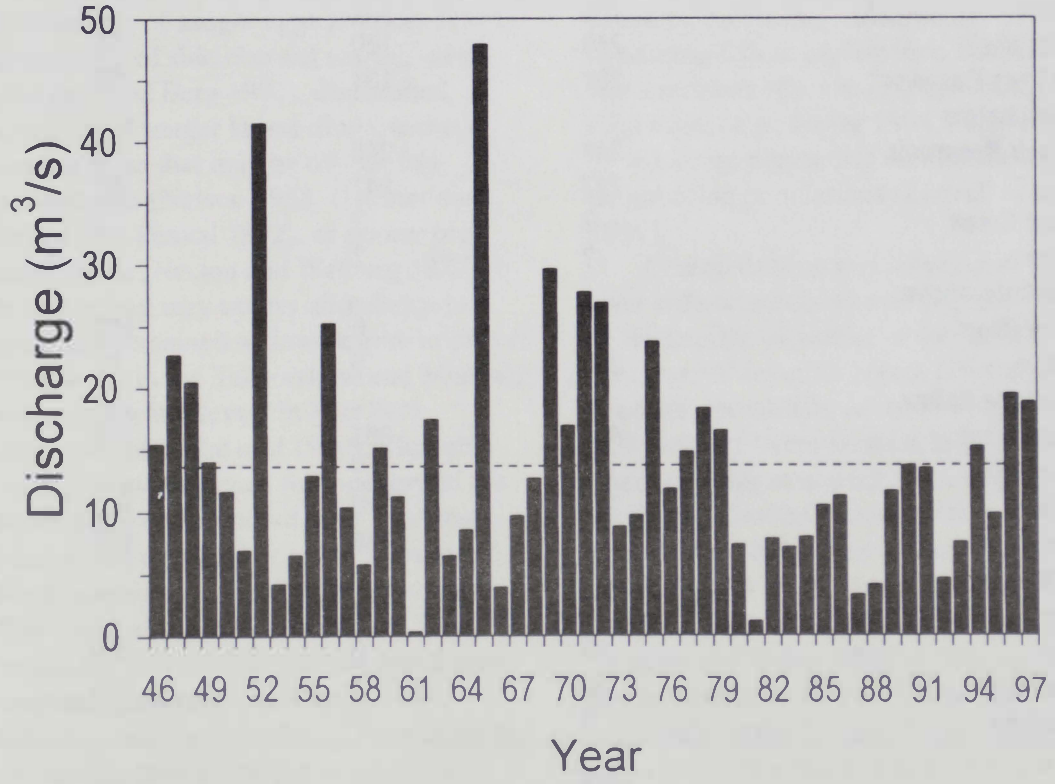

Figure 12. Mean April discharge of the Tongue River near mouth, 1946-1998. Fifty-three year average shown as dashed line.

sauger for spawning and rearing, as Guzevich (1993) and Elser et al. (1980) found walleye moderately abundant in large permanent pools in Little Beaver Creek, including age-0 fish, suggesting that habitat requirements were probably suitable for sauger in the past.

Present Status of Sauger in Montana

Historically, sauger likely occupied at least 3376 km of riverine habitat in Montana (Table 1). Present-day range is estimated at 1570 km, a 53 percent decline, though status is uncertain over some of their range. Decline has been much more extensive in tributaries; of the estimated historical occupancy of 1896 km, only 479 km are known to currently support sauger, a 75 percent reduction. In contrast, we estimate range reduction in mainstem rivers at 22 percent (1395 to 1091 km). Sauger remain common in four locales but apparently at lower than historical levels: Missouri mainstem between the Marias River and Fort Peck Reservoir (256 km); upper 25 percent of Fort Peck Reservoir (66 km); Missouri mainstem below Fort Peck Reservoir (246 km); and the lower Yellowstone mainstem below Cartersville diversion (381 km), comprising 949 km or 28 percent of the historical range.

Table 1. Historical vs. present-day range of sauger in Montana by major drainage. Range reported as kilometers of river occupied.

Drainage/River	Historical	Present (1999)	Percent loss
Missouri above Fort Peck Reservoir	333	333	0
Marias	274	96	65
Teton	80	0	100
Judith	85	?	
Musselshell	240	120	50
Fort Peck Reservoir*	261	131	50
Missouri below Fort Peck Reservoir	246	246	0
Milk	727	241	67
Beaver Creek	?	?	
Poplar	?	?	
Yellowstone above Bighorn River	80	0	100
Bighorn	206	6	97
Yellowstone below Bighorn River	475	381	20
Rosebud	?	?	
Tongue	241	0	100
Powder	16	16	100
Little Missouri			
Little Missouri	?	?	
Beaver	80	0	100
L. Beaver	32	0	100
Box Elder	?	?	
Total	3376	1570	53

*including Fort Peck Reservoir, 261 km long. We assumed that sauger typically occupy the upper half of the reservoir.

Discussion

Causes of the Decline

River Flows and Reservoir Water Levels.—A clear association occurred between low river flows and low reservoir water levels and the timing of the marked sauger decline throughout Montana in the late 1980s. All major sauger populations for which there is adequate trend data—Missouri River below Morony Dam, Marias River, Fort Peck Reservoir, and the lower Yellowstone River—declined substantially during a 3-4 year drought. This state-wide decline mirrored patterns observed in other sauger populations where abundance was positively correlated with river flows and reservoir water levels (Nelson 1968, Fischbach 1998). Populations also tended to exhibit region-wide trends in abundance (Lyons and Welke 1996). Species that exhibit high interpopulation synchrony in abundance are more susceptible to regional-scale environmental disturbance (Dunning et al. 1992), and we believe that sauger in Montana conform to such a pattern.

Reduced, or fluctuating, flow may lead to stranding of sauger eggs (Nelson 1968), dewatering of side channel rearing areas (Gardner and Berg 1982), diminished transport of sauger larvae downstream to rearing areas that may be 60-300 km downstream (Nelson 1968, Gardner and Berg 1982, Penkal 1992), or poorer prey recruitment (Nelson and Walburg 1977). It is perplexing why sauger abundance has generally remained so low despite improved river flows in the Yellowstone and Missouri rivers and water levels in Fort Peck Reservoir since the mid 1990s. Though there is some evidence for recovery of the lower Yellowstone population, there has been a lack of recovery in the Missouri River population, despite above-average flows in 6 of the last 8 years from 1991-98, and in the Fort Peck Reservoir population, despite high water levels since 1994. Concern over their declining abundance led to classification of sauger as a Montana "species of special concern" (Hunter 1994) in 2000 (Graham 2000).

Migratory barriers/habitat loss.—Dams and water diversion structures have blocked or impeded migratory access to large areas of the historical range of sauger in Montana, and have undoubtedly served as the primary cause for their significant range reduction, especially in tributaries. Their highly migratory nature, coupled with their apparent propensity to spawn in only a few areas, make them particularly susceptible. For example, the loss of an important spawning tributary to damming resulted in a decline of sauger in the entire upper Tennessee River system (St. John 1990). Similar declines following fragmentation of fish populations as a result of damming of tributaries also have been documented for other species (Luttrell et al. 1999). The role of migratory barriers in the recent decline in sauger is unclear. Most of the migratory barriers were in place during the 1970s and 1980s when sauger were abundant and widespread throughout the Yellowstone and Missouri drainages. However, these structures may impede recovery by causing direct mortality, e.g., entrainment in the Intake water diversion canal, by increasing vulnerability of remaining fish to exploitation because of a concentration effect during spawning migration, (e.g., Hesse 1994, Pegg et al. 1996), or by preventing recolonization from neighboring populations (Luttrell et al. 1999).

Channelization and subsequent loss of river side-channels was an important factor in the decline of sauger in the mainstem Missouri River in Nebraska (Hesse 1994). Channel complexity in the Missouri and Yellowstone rivers remains largely intact, so channel changes and associated habitat loss likely have not been a major factor in the recent decline. Chronic dewatering of several major tributaries in Montana where sauger were historically abundant (Teton, Tongue, and Musselshell rivers), however, has undoubtedly played a significant role, especially in the Tongue River, one of apparently only two main spawning areas for the entire Yellowstone population (Penkal 1992). In the middle Missouri,

dewatering of sidechannels used as rearing habitat for age-0 sauger as a result of power peaking operations at Morony Dam (Penkal 1990), could be an important factor affecting year-class strength, particularly during low flow years.

Hybridization with Walleye.—Walleye and sauger can readily hybridize and produce fertile offspring, but hybridization is rare under natural circumstances despite their overlapping distribution and similar spawning habitat requirements (Billington et al. 1988, White and Schell 1995). However, hybridization rates and loss of stock integrity may become significant when sauger populations fall to low levels, spawning habitat is limited, or when sauger x walleye hybrids (saugeye) are stocked (White and Schell 1995, Van Zee et al. 1996, Fiss et al. 1997).

We compiled hybridization data from Montana and nearby regions to assess if hybridization rates have increased and to compare Montana hybridization rates to other locales (Table 2). Overall, sauger hybridization rates in Fort Peck Reservoir, the middle Missouri River, and the lower Yellowstone River are similar to other Missouri River reservoirs (Lakes Sakakawea and Lewis and Clark), averaging about 10 percent. Rates of hybridization are well below that reported from waters where saugeye have been stocked (~74%; Fiss et al. 1997) and there is no indication thus far of an increase associated with the sauger decline.

Whether hybridization rates will increase in the future is uncertain. In some systems, e.g., Lewis and Clark Reservoir, South Dakota, sauger and walleye have coexisted for many years, spawning in the same limited habitat; yet hybridization levels remain about 10 percent, and substantial loss of stock integrity has not been observed. Other than stocking of saugeye (Fiss et al. 1997), predicting what factors will magnify hybridization and, thus, targeting practices that may lessen or enhance risk are difficult. Given this uncertainty, continued monitoring is necessary, as well as close genetic screening of walleye or sauger broodstock used in artificial propagation (Ward 1992, Leary and Allendorf 1997, Billington 1998).

Species interactions.—Historically, sauger were the most common top predator in the Missouri and Yellowstone Rivers and major tributaries in eastern Montana. Two nonnative piscivores, walleye and smallmouth bass, now co-occur with sauger over much of the sauger's historical range in the state (Holton and Johnson 1996). Smallmouth bass are now the dominant top predator in the Tongue and upper Missouri rivers. Smallmouth bass and walleye are now abundant in the upper Missouri River below Morony Dam (Hill et al. 1998). Decline of sauger cannot be directly related to the expansion of these two species because the most marked decrease occurred when both smallmouth bass and walleye abundances were still low. Direct

Table 2. Proportion of sauger x walleye hybrids in Montana and surrounding regions. *N* is the total number of *Stizostedion* tested.

Location	Date	*N*	No. of hybrids (%)	Source
Middle Missouri R.	1996	14	0 (0)	Billington et al. 1997
	1999	109	5 (4.5)	N. Billington unpub. data
Fort Peck L.	1997	50	3 (6.0)	Billington 1998
Fort Peck L.	1995	158	15 (9.5)	Leary and Allendorf 1997
Lower Yellowstone R.	1995	48	7 (14.6)	Leary and Allendorf 1997
Lower Missouri R.	1996	85	4 (4.7)	Leary 1998
Lewis and Clark L., SD	1995	50	5 (10)	Van Zee et al. 1996
Bighorn L. and R., WY	1995	164	0 (0)	Kreuger et al. 1997
Boysen L., WY	1995	98	0 (0)	Kreuger et al. 1997
Lake Sakakawea, ND	1991	279	28 (10)	Ward 1992

competition among sauger and smallmouth bass and walleye is unlikely given dissimilar habitat preferences (e.g., Nelson and Walburg 1977, Rawson and Scholl 1978). However, changing habitat conditions could shift the balance in favor of these other species. Increases in water clarity as a result of damming and altered spring flows favor both walleye and smallmouth bass (Nelson and Walburg 1977). We hypothesize that while species interactions have not directly caused sauger declines, expanding populations of other piscivores could impede recovery through predation on juvenile sauger (Zimmerman 1999).

Overexploitation.—Overfishing has been implicated in sauger declines in other regions (Hesse 1994, Pegg et al. 1996, Maceina et al. 1998). High exploitation rates probably were not a significant factor leading to the state-wide sauger decline in Montana, however, because of low angler density from the remoteness and large size of eastern Montana waters. Tag return data from the Missouri and Yellowstone rivers indicate low (<10%) exploitation rates (Berg 1981, Gardner and Stewart 1987, Stewart 1998). However, actual exploitation rates could be higher if harvest is concentrated when sauger are aggregated in high densities in restricted areas during winter and early spring (Hesse 1994, Pegg et al. 1996). Though creel census data are lacking from most areas, quality of the popular sauger fishery in eastern Montana has also declined considerably over the past decade based on anecdotal angler reports and documented reductions in fish size on the lower Yellowstone River (Stewart 1998). The harvest limit of sauger in the upper Missouri River upstream of the Judith River confluence was reduced from 5 fish to 1 fish in 1999 because of the low density in this reach (Fig. 5). More extensive creel data are needed to better quantify harvest rates, provide a benchmark for judging future change in fishery quality, and suggest if and what type of angling regulation changes could bolster the fishery.

Conclusions

Sauger persist in about half, and remain common in only about 28 percent, of the estimated 3376 river-kilometers of their historical range in Montana. Losses have been particularly acute in tributaries where an estimated 75 percent of the former range no longer supports sauger compared to 22 percent of mainstem waters. Although much of the range reduction likely occurred over the past 80 years as a result of construction of migratory barriers and chronic dewatering of some tributaries, state-wide drought in the late 1980s appeared to trigger a marked decline in sauger in large portions of the Missouri and Yellowstone drainages. Despite improved flows since the mid-1990s, evidence of recovery was minimal in most areas. Attributing the recent decline to habitat loss is difficult because many of the major habitat alterations that affected sauger, namely migratory barriers in the form of dams and water diversion structures and canals, were in place before the decline began. However, barriers to migration of this highly migratory species may be impeding recovery by entrainment in irrigation canals, by increasing their vulnerability to exploitation, and by reducing recolonization from neighboring populations. On the other hand, hybridization with walleye apparently is not abnormally high nor increasing at this point in time. Evidence was lacking for displacement of sauger by nonnative walleye or smallmouth bass, but over time reduced turbidity may shift the balance in favor of these species.

A general lack of basic information on sauger ecology in Montana and other portions of its range makes development of management policies difficult (Pegg et al. 1997). Tagging studies in both the Yellowstone and Missouri systems indicate that sauger are highly migratory, moving throughout a very large area that crosses jurisdictional boundaries both within and outside of Montana. How habitat conditions in one area affect the population as a whole remains unknown. Radiotracking and tag

recovery studies (e.g., Pegg et al. 1997), though logistically daunting over such a large area, would help refine important habitat types, seasonal movement patterns, and exploitation rates, and perhaps identify heretofore unknown spawning locales.

Improved fish passage at key diversions dams, improved minimum flows in the Tongue River during spawning and other historical habitats like the Teton River, and reduction of entrainment at the Intake water diversion canal would help promote sauger recovery (Montana Fish, Wildlife and Parks 1997). These actions should improve access to many kilometers of historical sauger habitat and reduce high mortality of juvenile sauger moving upstream from the lower Yellowstone River and Lake Sakakawea in North Dakota. Habitat enhancement of current spawning areas also could help offset the loss of historical spawning grounds.

Continued monitoring of sauger abundance in sites that have historical population data is key to tracking population trends. Survey of little known areas (Milk, Teton, Judith,and Musselshell rivers) also is needed to further clarify population status and define where sauger persist.

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Factors Influencing Distributions of Salmonids in the Little Bighorn River Drainage, Wyoming

Neil D. Stichert, Department of Renewable Resources, University of Wyoming, Laramie, WY 82071-3354
Wayne A. Hubert, Wyoming Cooperative Fish and Wildlife Research Unit, USGS, Department of Zoology and Physiology, University of Wyoming, Laramie, WY 82071-3166
Quentin D. Skinner, Department of Renewable Resources, University of Wyoming, Laramie, WY 82071-3354

Abstract

We assessed the influences of geomorphology and introductions of nonnative salmonid species on the distributions of salmonids in headwater streams of the Bighorn Mountains in Wyoming. We sampled geomorphic features and fish at 71 sites among streams in four watersheds near the headwaters of the Little Bighorn River in 1999. Brook trout *(Salvelinus fontinalis)*, rainbow trout *(Oncorhynchus mykiss)*, cutthroat trout *(O. clarki)*, and cutthroat trout x rainbow trout hybrids were found in the study area. Distributions were influenced by stream size, channel slope, natural barriers to upstream movement by salmonids, spatial patterns of nonnative salmonid introductions, and recent stocking of nonnative salmonid species. Salmonids were found at sites with wetted widths $\geq$ 1.1 m and channel slopes $\leq$ 16 percent, and the probability of salmonid presence increased as wetted width increased and channel slope decreased. Natural barriers to upstream movement appeared to have excluded nonnative salmonids from colonizing some small headwater streams. Two allopatric populations of genetically pure Yellowstone cutthroat trout were isolated from nonnative salmonids by natural barriers. Past introductions of brook trout and rainbow trout haves led to naturalized populations in large portions of the study area. Recent stocking of cutthroat trout and rainbow trout accounted for occurrence of these species in some headwater streams where evidence of natural reproduction was not found.

Key words: Salmonidae, trout, mountains, streams, geomorphology, native fishes, introduced fishes, Bighorn Mountains, Big Horn National Forest

Introduction

A focus of fisheries science in the Intermountain West is on the decline of native cutthroat trout (*Oncorhynchus clarki*) with substantial research being directed toward understanding their habitat requirements and the influences of anthropogenic activities (Hildebrand and Kershner 2000, Kruse et al. 2000). Cutthroat trout occupied the widest natural geographic distribution of any Salmonidae in the United States (Behnke 1992). The Yellowstone subspecies (*O. c. bouvieri*) had a wide range occupying the Yellowstone River drainage in Montana and Wyoming, including the Little Bighorn River drainage in Montana and Wyoming, and the Snake River drainage downstream to Shoshone Falls in Idaho (Varley and Gresswell 1988, Behnke 1992). However, the distribution of Yellowstone cutthroat trout has declined substantially since settlement by Europeans (Varley and Gresswell 1988, Kruse et al. 2000).

Fisheries scientists are adopting the watershed as their scale for research and management (Wesche and Isaak 1999) and the effects of spatial habitat patterns on salmonid distributions at this scale are becoming more evident (Meehan 1991). For example, Kruse et al. (1997) identified watershed-scale geomorphic features that affected the distributions of Yellowstone cutthroat trout in the Absaroka Mountains, Wyoming. They found that channel slope, elevation, stream size, and barriers to

upstream movement significantly influenced the presence and absence of Yellowstone cutthroat trout where nonnative salmonid species did not occur. Nevertheless, the effects of geomorphic features on either native cutthroat trout or nonnative species of salmonids are not well defined.

Numerous anthropogenic activities are affecting native cutthroat trout in the Intermountain West (Meehan 1991, Young 1995), but the most extensive may be the introduction of nonnative salmonid species (Varley and Gresswell 1988, Behnke 1992). Brook trout (*Salvelinus fontinalis*), rainbow trout (*Oncorhynchus mykiss*), and brown trout (*Salmo trutta*) have been widely introduced and have become naturalized in many montane stream systems, contributing to declines of native cutthroat trout through competition and hybridization (Griffith 1972, DeStaso and Rahel 1994, Henderson et al. 2000, Kruse et al. 2000, Novinger 2000).

The salmonid assemblages in streams of the Bighorn Mountains in northern Wyoming probably are representative of many montane stream systems throughout the Intermountain West. Fisheries management in the Bighorn Mountains during much of the last century focused on introductions of nonnative salmonids to expand distributions and enhance sport fisheries because most headwater streams and alpine lakes of the Bighorn Mountains had no natural salmonid populations. Consequently, distributions and species assemblages of salmonids have been substantially altered by management activities.

Our objectives were to describe the current distributions of salmonid species in montane portions of the Little Bighorn River drainage in Wyoming and to assess geomorphic and anthropogenic factors that influence distributions. Geomorphic factors included elevation, stream width, channel slope, and natural barriers to upstream movements by salmonids, and anthropogenic factors included past introductions and recent stockings of nonnative salmonids.

Study Area and Methods

The Bighorn Mountains are approximately 120 km long and 50 km wide with elevations of 1650-3950 m above mean sea level (Lageson and Spearing 1988). The mountain range is almost totally encompassed by the Big Horn National Forest. Primary land uses are recreation, livestock grazing, and some timber harvest.

The anticlinal uplift of the Bighorn Mountains consists of a granitic core covered with sedimentary limestone and dolomite. The uplift created a plateau-like geomorphology that affects the longitudinal profile of many headwater streams. Many headwater streams originate from springs or alpine lakes near the base of the granitic core, cross subalpine meadows on the plateau, and flow through steep, deeply incised canyons before emerging onto the plains at the base of the mountains. In many streams the upstream movement of fish onto the plateau is blocked by falls and cascades in the canyons.

The Little Bighorn River drainage is at the northern end of the Bighorn Mountains (Fig.1). Four headwater watersheds converge near the northern edge of the Big Horn National Forest and the Montana-Wyoming state line to form the mainstem of the Little Bighorn River that flows north into the Yellowstone River. Falls and cascades in deeply incised canyons near the downstream ends of three (West Fork Little Bighorn River, Upper Little Bighorn River, and Dry Fork Little Big Horn River) of four watersheds are believed to have prevented natural colonization of these watersheds by Yellowstone cutthroat trout.

Perennial streams identified on U.S. Geological Survey, 1:24,000-scale topographic maps were sampled during summer 1999 in the headwaters of the Little Bighorn River. Sampling sites at about 1.5-km intervals were selected for sampling. Sampling progressed upstream in small headwater streams until no fish were found. If we encountered a probable barrier to upstream movement by salmonids, a site was sampled immediately upstream. Probable barriers were defined as channel-

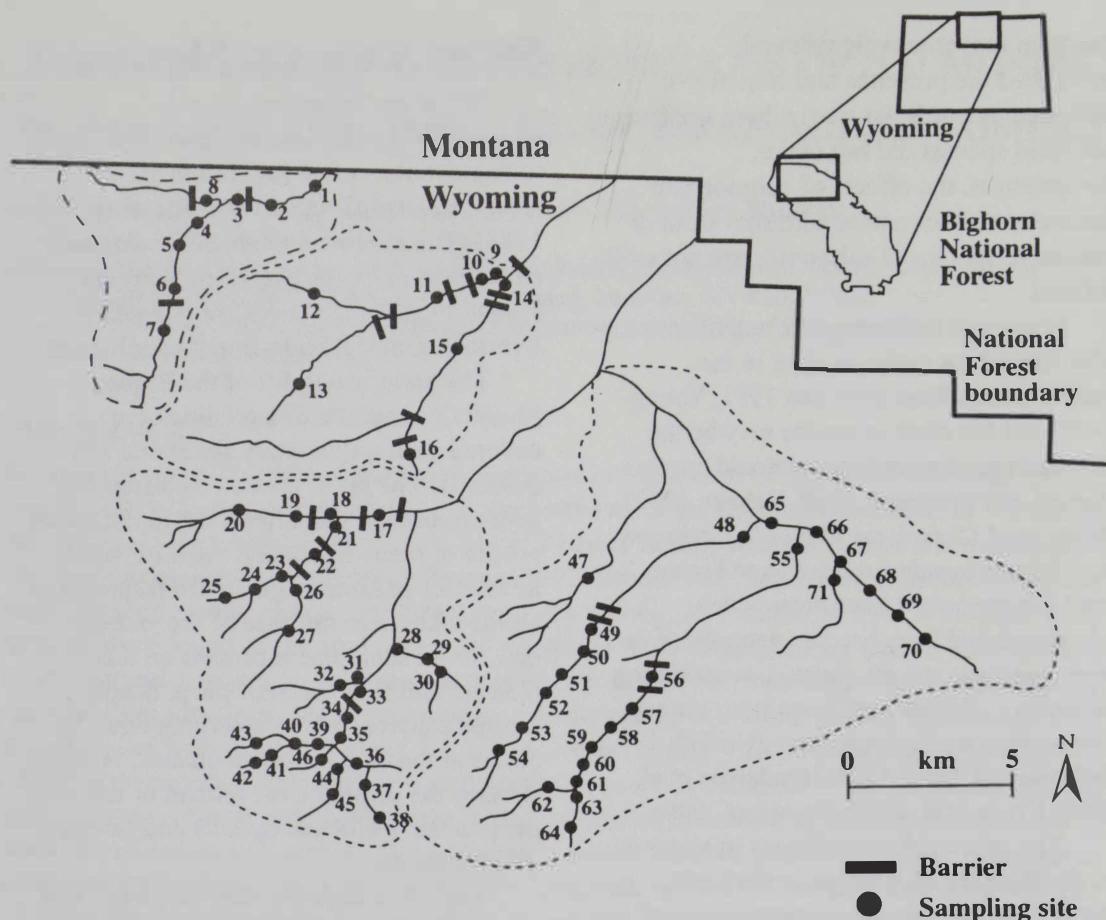

Figure 1. Map of the Little Bighorn River drainage in Wyoming showing the four watersheds within the drainage. Numbers indicate locations of sampling sites described in Table 1.

wide falls created by rock formations at least 1.5 m high (Stuber et al. 1988) or as steep (> 30% channel slope) cascades >15 m in length. Locations of barriers were identified on topographic maps. Because of the inaccessibility of some steep-walled canyons, some selected sampling sites could not be sampled.

Fish were sampled by electrofishing a 100-m reach at each accessible site. Two or three electrofishing passes were made progressing upstream and all stunned fish were collected and released. Captured fish were identified to species using morphological features (Baxter and Simon 1970). Spotting patterns, throat slashes, and white fin margins were used to identify cutthroat trout x rainbow trout hybrids in the field (Kruse 1998).

Elevation, channel slope, and wetted width were determined at each sampling site. Elevation was estimated using 1:24,000-scale topographic maps at the midpoint of the 100-m fish-sampling reach. An Abney level was used to measure channel slope of the reach (Isaak et al.1999). Individual habitat units (cascades, riffles, glides and pools) were identified over the length of each reach (Bisson et al.1982) and the length of each habitat unit was measured. Wetted width was measured perpendicular to streamflow at the midpoint of each habitat unit. We used habitat unit lengths and wetted widths to compute a weighted average wetted width for each reach.

Wyoming Game and Fish Department files in the Sheridan Regional Office were reviewed for stocking records of salmonids within the Little Bighorn River drainage.

We obtained information on species, stocking locations, and tocking dates. At sampling sites where allopatric populations of cutthroat trout were found and there were no records of recent stocking, a portion of the upper lobe of the caudal fin was removed from each cutthroat trout and preserved in ethanol. Genetic analysis of the caudal fish tissue was conducted at the Brigham Young University Fish Genetics Laboratory, Provo, Utah. Two mitochondrial DNA markers were identified through DNA restricted fragment analysis and compared to an existing library of cutthroat trout DNA fragment patterns. Two nuclear-based DNA markers were isolated from the internal transcriber spacer region of ribosomal DNA and compared to other cutthroat trout samples. In combination, these four markers are believed to provide an accurate appraisal of introgression between cutthroat trout and rainbow trout (Billington and Hebert 1991).

Evidence of natural reproduction was used as an index of naturalization of introduced species. Recently stocked salmonids exceeded 100 mm TL at the time of stocking (Wyoming Game and Fish Department, Sheridan Regional Office, file records). Consequently, the presence of fish < 100 mm TL at a site provided evidence of natural reproduction.

Multiple-logistic regression was used to assess the influence of geomorphic features on the presence/absence of salmonids (Ramsey and Schafer 1997). The form of the logistic function is

$$P = \exp^u/(1 + \exp^u)$$

where P = the estimated probability, exp = base of the natural logarithm (i.e., 2.718), and u = the linear model

$$u = A + b_1X_1 + b_2X_2 + \ldots + b_nX_m$$

where A = regression constant, b_n = regression coefficients, and X_m = independent variables. Significance of models and individual independent variables was assessed using Chi-square tests (a = 0.05). The goodness of fit of each model was evaluated using the Hosmer-Lemeshow statistic with values ≥ 0.5 considered satisfactory (Ramsey and Schafer 1997). Because sampling sites were selected to be representative of the population of possible sampling sites, different response outcomes could have occurred with different sets of sites. Consequently, probabilities cannot be estimated from such retrospective samples, but odds ratios can be estimated and used to assess the influence of variation in independent variables on the odds of presence of salmonids (Ramsey and Schafer 1997). We used Statistix for Windows (Analytical Software 1996) to compute the logistic regressions.

Results

Geomorphic Factors

Seventy-one sites were sampled across the Little Bighorn River drainage on the Big Horn National Forest (Table 1, Fig. 1). Sampled sites ranged from 1695 to 2792 m above mean sea level with wetted widths of 0.6-6.9 m and channel slopes of 1.0-25.0 percent. Brook trout, rainbow trout, cutthroat trout, or cutthroat trout x rainbow trout hybrids alone or in various combinations were found at 51 sites over the entire range of elevations sampled. No salmonids were found at sites with wetted widths < 1.1 m or channel slopes > 16 percent (14% of sampled sites).

We identified 22 barriers to upstream salmonid movement across the headwaters of the Little Bighorn River drainage (Fig. 1), three in the Lodgegrass Creek watershed, nine in the Pumpkin Creek watershed, six in the Upper Little Bighorn River watershed, and four in the Dry Fork Little Bighorn River watershed. We found no salmonids upstream from some barriers in the headwaters of Lodgegrass (Fig. 1, site 7), Pumpkin (sites 11 and 12), Cub (site 13), Mann (sites 15 and 16), and Wagonbox (sites 19 and 20) creeks. Wetted widths were 2.3-4.1 m and channel slopes were 2.5-15.5 percent among these eight sites, within the ranges of wetted widths and channel slopes occupied by salmonids at other sites in the study area. Consequently, we assumed that barriers to upstream movement had prevented these eight sites from being colonized by salmonids.

Table 1. Sites sampled in the headwaters of the Little Bighorn River watershed on the Bighorn National Forest in Wyoming during 1999. Site numbers correspond with locations on Figure 1. Species found included rainbow trout (RBT), cutthroat trout (CUT), brook trout (BKT), and cutthroat trout x rainbow trout hybrids (CUT x RBT). Underline of species acronym indicates that evidence of natural reproduction was observed.

Stream	Site	Elevation (m)	Wetted width (m)	Channel slope (%)	Species found
	Lodgegrass Creek Drainage				
Lodgegrass Creek	1	1,957	4.7	14.0	RBT, CUT x RBT
	2	2,030	4.3	15.0	RBT, CUT, CUT x RBT
	3	2,073	4.6	15.5	CUT
	4	2,256	4.2	11.5	CUT
	5	2,304	4.4	7.0	CUT
	6	2,374	4.1	10.5	CUT
	7	2,414	4.1	19.0	
Line Creek	8	2,226	4.1	25.0	
	West Fork Little Bighorn River Drainage				
Pumpkin Creek	9	1,695	4.2	7.5	CUT
	10	1,707	4.3	8.0	CUT
	11	1,774	4.1	14.5	
	12	2,060	3.6	12.0	
Cub Creek	13	2,195	3.0	15.5	
Mann Creek	14	1,695	3.8	13.0	CUT
	15	1,829	3.2	20.0	
	16	2,121	2.3	16.0	
	Upper Little Bighorn River Drainage				
Wagonbox Creek	17	2,176	3.9	7.0	BKT
	18	2,274	5.0	16.0	BKT
	19	2,390	3.1	2.5	
	20	2,536	2.3	9.5	
Duncum Creek	21	2,274	3.4	12.5	BKT
	22	2,432	4.1	4.0	BKT
	23	2,512	3.1	4.0	BKT
	24	2,579	2.9	13.0	BKT
	25	2,640	1.0	8.5	
Duncum Creek Tributary	26	2,505	3.1	4.5	BKT
	27	2,609	2.4	3.5	BKT
Dayton Gulch	28	2,536	2.2	2.4	CUT, BKT
	29	2,597	2.1	2.5	BKT
	30	2,621	1.3	2.5	
Half Ounce Creek	31	2,536	3.1	2.5	CUT, BKT
	32	2,548	1.4	2.0	BKT
Little Bighorn River	33	2,548	5.8	2.5	BKT
	34	2,576	6.1	3.5	BKT
	35	2,633	6.4	1.5	CUT, BKT
	36	2,682	2.3	3.0	CUT, BKT
	37	2,719	1.6	2.0	CUT
	38	2,758	1.0	7.0	
Little Bighorn River Tributary 1	39	2,670	3.5	11.0	CUT, BKT
	40	2,707	2.6	5.0	CUT, BKT
	41	2,792	1.5	2.5	BKT
	42	2,792	1.1	2.5	BKT

Table 1. Continued.

Stream	Site	Elevation (m)	Wetted width (m)	Channel slope (%)	Species found
Little Bighorn River Tributary 2	43	2,780	1.8	2.0	BKT
Little Bighorn River Tributary 3	44	2,688	2.0	10.5	BKT
	45	2,749	1.0	7.5	
Little Bighorn River Tributary 4	46	2,658	1.2	6.5	CUT, BKT
	Dry Fork Little Bighorn River Drainage				
Beartrap Creek	47	2,402	0.6	19.0	
Lick Creek	48	1,981	4.2	10.0	RBT, CUT, CUT x RBT
	49	2,499	3.9	16.0	CUT
	50	2,548	4.8	14.0	CUT
	51	2,597	3.8	2.5	CUT
	52	2,621	2.7	2.5	CUT
	53	2,694	2.8	1.0	CUT
	54	2,731	2.2	4.5	
Lake Creek	55	1,975	4.4	3.0	RBT
	56	2,403	3.3	11.5	
	57	2,499	4.0	2.5	RBT, BKT
	58	2,524	5.0	8.0	BKT
	59	2,597	3.1	5.0	RBT
	60	2,627	2.9	3.0	RBT
	61	2,658	1.9	2.0	RBT
	62	2,706	1.4	2.0	
Ice Creek	63	2,646	2.0	4.0	RBT
	64	2,731	1.6	4.0	
Dry Fork	65	1,878	6.9	4.0	RBT, BKT
	66	1,923	5.4	5.0	RBT, BKT
	67	1,963	4.4	6.5	BKT
	68	2,006	4.4	5.5	BKT
	69	2,060	2.1	7.0	BKT
	70	2,109	1.0	4.5	
Garland Gulch	71	2,402	2.2	4.5	

Wetted width and channel slope in separate models each accounted for presence/absence of salmonids among the 71 sites (Table 2). The wetted width equation indicated that probability of salmonid presence increased as wetted width increased. This equation had a high rate of correct classification for sites where occurrence of fish was predicted (90%), but it was not as good for sites where we predicted an absence of fish (38%). Conversely, the channel slope equation indicated that the probability of salmonid presence decreased as channel slope increased. Correct classification rates were similar to the wetted width model.

When the eight headwater sites assumed to be isolated by barriers to upstream movement by salmonids were removed from the data set, wetted width alone and a model with both wetted width and channel slope accounted for presence/absence of salmonids and the rates of correct predictions were enhanced (Table 2).

No significant logistic regression equations that fit the logistic model were found that accounted for presence/absence of individual salmonid species.

Nonnative Salmonids

Wyoming Game and Fish Department records indicated past introductions as well as recent stockings of nonnative salmonids over a large portion of the Little Bighorn River drainage. Brook trout were introduced in the Lodgegrass Creek watershed in 1953, in the Upper Little Bighorn River watershed in 1945, and in the Dry Fork Little Bighorn River watershed in 1950, 1951, 1956, and 1960. Rainbow trout were introduced in the Lodgegrass Creek watershed in 1938, 1939,

Table 2. Logistic regression models with significant geometric variables, correct classification rates, Hosmer-Lemeshow (H-L) statistics, and odds rations predicting presence or absence of salmonid fishes in montane streams of the Little Bighorn River drainage in northern Wyoming.

Model parameters	Value (SE)	P	Number of cases	Classification percent (present/absent/overall)	H-L	Odds ratio
Models with all sites						
Constant	-1.188 (0.69)	0.0857	71	90/38/75	9.62	
Wetted width	0.710 (0.24)	0.0027				2.03
Constant	2.060 (0.53)	0.0001	71	92/33/74	7.64	
Channel slope	-0.149 (0.05)	0.0043				0.86
Models without isolated headwater sites						
Constant	-1.666 (0.85)	0.0507	63	94/54/86	8.91	
Wetted width	1.187 (0.37)	0.0012				3.28
Constant	-1.157 (0.96)	0.2301	63	94/54/86	8.18	
Wetted width	1.674 (0.50)	0.0009				5.33
Channel slope	-0.251 (0.11)	0.0248				0.78

1942, and 1944, and in the Dry Fork Little Bighorn River watershed in 1937 and 1944. Between 1985 and 1995 rainbow trout were stocked annually near the headwater of Lake Creek in the Dry Fork Little Bighorn River watershed. Stocking of cutthroat trout from hatcheries took place in the Lodgegrass Creek watershed in 1945, 1956, and 1958, in the West Fork Little Bighorn River watershed in 1946, in the Upper Little Bighorn River watershed in 1946 and through the 1990s, and in the Dry Fork Little Bighorn River watershed in 1940, 1958, 1960, 1966-1969, and every 1-4 years after 1976. Between 1985 and 1999 cutthroat trout stocking in the Dry Fork Little Bighorn River watershed was exclusively within the headwater of Lick Creek.

Distribution patterns of nonnative salmonids in 1999 were related to past introductions across the Little Bighorn River drainage (Table I). Brook trout were found throughout the Upper Little Bighorn River watershed and much of the of the Dry Fork Little Bighorn River watershed, both of which received introductions of brook trout between 1945 and 1960. However, no brook trout were found in the Lodgegrass Creek watershed, despite the record of an introduction in 1953. Rainbow trout were found downstream from a barrier in Lodgegrass Creek (sites 1 and 2) and at several locations in the Dry Fork Little Bighorn River watershed. Rainbow trout had not been stocked in the Lodgegrass Creek watershed since 1944, but they had been stocked in the Dry Fork Little Bighorn River watershed as recently as 1995. Cutthroat trout were found throughout the Little Bighorn River study area, but there were records of past stocking of cutthroat trout from hatcheries in all four of the watersheds within the study area. Cutthroat trout x rainbow trout hybrids were observed downstream from a barrier in Lodgegrass Creek (sites 1 and 2) and at the most downstream site in Lick Creek (site 48), sites where both cutthroat trout and rainbow trout also were found.

Natural Reproduction

Evidence of natural reproduction by salmonid species varied across the Little Bighorn River drainage (Table 1). We found evidence of natural reproduction by cutthroat trout only in the upstream portion of Lodgegrass Creek (Fig. I, sites 3 and 4) and near the junction of Pumpkin and Mann creeks (sites 9, 10 and 14) where cutthroat trout were allopatric. Rainbow trout reproduction was evident only in the

downstream portion of Lodgegrass Creek (sites 1 and 2), Lake Creek (site 55), and the Dry Fork Little Bighorn River (sites 65 and 66). Occurrence of cutthroat trout x rainbow trout hybrids in the downstream portion of Lodgegra s Creek (sites 1 and 2) and at the most down tream site on Lick Creek (site 48) in the Dry Fork Little Bighorn River drainage indicated natural reproduction by *Oncorhynchus* in the vicinity of these stream reaches. Brook trout reproduction was indicated throughout the Upper Little Bighorn River and Dry Fork Little Bighorn River watersheds.

Cutthroat Trout

Barriers to upstream movement of salmonids on Lodgegrass (downstream from site 3) and Pumpkin-Mann (downstream from sites 9 and 14) creeks isolated allopatric, genetically pure populations of cutthroat trout. Mitochondrial and nuclear DNA analysis of 21 fish from the upstream portion of Lodgegrass Creek and 24 fish from near the confluence of Pumpkin and Mann creeks indicated that both populations were genetically pure Yellowstone cutthroat trout (Evans and Shiozawa 2000).

Where cutthroat trout were found in other portions of the Little Bighorn River drainage, they unlikely were genetically pure Yellowstone cutthroat trout endemic to the drainage. Genetic analysis was not conducted for samples of cutthroat trout from most sites where they were found because of visual evidence of hybridization with rainbow trout or knowledge of recent stocking of cutthroat trout at or near these sites. Cutthroat trout at the downstream ends of Lodgegrass Creek (sites I and 2) and Lick Creek (site 48) were found in association with reproducing rainbow trout and cutthroat trout x rainbow trout hybrids indicating that the cutthroat trout in these reaches were part of an integrated *Oncorhynchus* population. Additionally, cutthroat trout at sites in the Upper Little Bighorn River and Dry Fork Little Bighorn River drainages showed no evide,1ce of natural reproduction and their distributions were associated with locations of recent stockings of cutthroat trout by the Wyoming Game and Fish Department.

DISCUSSION

Stream size, channel slope, and barriers to upstream movement interact to limit the distributions of salmon ids (Nelson et al. 1992, Kruse et al 1997). We found no salmonids at sites <1.1 m wide or having channel slopes > 16 percent in the Little Bighorn River dramage, features characteristics of headwaters of many montane stream systems (Kruse et al. 1997, Kruse 1998). Logistic regression indicated an increasing probability of occurrence of salmonid as wetted width increased and as channel slope decreased. We found no salmonids upstream from barriers near the headwater of several streams originating in steep valleys. While the streams at the e sites were of moderate size (2.3-4.1 m wide), channel slopes tended to be relatively high (2.5-19.0 percent). Kruse et al. (1997) found sites with cutthroat trout had wetted widths >1.3 m and channel slopes of< 17 percent in the Absaroka Mountains of northwestern Wyoming, similar to our observations for salmonids in the Little Bighorn River drainage. Cutthroat trout were rarely found in streams with channel slopes >10 percent in the Absaroka Mountains (Kruse et al. 1997), but we found cutthroat trout at several sites with channel slopes of 10-16 percent.

Two allopatric, reproducing cutthroat trout populations were found in the Little Bighorn River drainage, one in the up tream portion of Lodgegra s Creek (Fig I, sites 3-6) and a second in the vic101ty of the confluence of Pumpkin and Mann creeks (ites 9, IO and 14). The population in the headwaters of Lodgegrass Creek likely was native to the dramage as we found no evidence of past stockmg, and these fish differed somewhat from hatchery stock in Yellow tone Lake widely used in Wyoming (Evans and Shiowaza 2000) However, genetic analysis suggested that the Pumpkin-Mann creek population wa similar to fish from Yellowstone Lake (Evans and Shiozawa 2000), a source of

hatchery stock used by many management agencies (Varley and Gresswell 1988) including the Wyoming Game and Fish Department. Results of this analysis suggested that this population was the result of the 1946 introduction. An endemic population of Yellowstone cutthroat trout likely did not occur in Pumpkin-Mann creeks because waterfalls and cascades downstream in Pumpkin Creek and West Fork Little Bighorn River probably prevented natural colonization. Similarly, waterfalls and cascades near the downstream ends of the Upper Little Bighorn River and Dry Fork Little Bighorn River watersheds probably prevented natural colonization from downstream by Yellowstone cutthroat trout. Salmonid species found in these two watersheds in 1999 were likely to be the products of past and recent stocking. However, given the history of stocking, it cannot be determined if natural populations of Yellowstone cutthroat trout may have occurred in any of these watersheds.

Introductions of nonnative salmonids in the Little Bighorn River drainage began in 1933 and stocking continues to affect fish distributions. These activities appear to have extended distributions of salmonid species and diversified salmonid assemblages in these streams, similar to other locations in the western United States (Krueger and May 1991, Li and Moyle 1999). Naturalized populations of brook trout now occur throughout the Upper Little Bighorn River and Dry Fork Little Bighorn River watersheds, but brook trout were not found in Lodgegrass Creek despite an introduction in I 953. Two naturalized populations of rainbow trout occurred in the drainage in the downstream portion of Lodgegrass Creek (Fig. I, sites I and 2) and in the Dry Fork Little Bighorn River and downstream portions of Lake and Lick creeks (sites 48, 55, 65 and 66). Both naturalized rainbow trout populations occurred at low elevations (<2030 m) in reaches that are relatively wide (4.3-6.9 m wetted width) compared to the array of streams in the montane portion of theLittle Bighorn River drainage. Recent stockings of rainbow trout near the headwaters of Lake Creek (Fig. 1, sites 57-61) and of cutthroat trout near the headwaters of Lick Creek (sites 49-53) appeared to be the sources of the populations of these species observed in 1999. We found no evidence of natural reproduction by rainbow trout or cutthroat trout in these headwater areas. The sites in Lake Creek were at elevations of 2499-2658 m and had wetted widths of 1.9- 4.0 m. Similarly, sites on Lick Creek were at elevations of 2499-2694 m and had wetted widths of 2.8-3.9 m. Observations by Wyoming Game and Fish Department personnel (Dey and Annear 1993) and us suggest that availability of suitable spawning gravel limits reproduction in these tributaries. Gravel is relatively rare and where it occurs it tends to be highly embedded with sediment and calcium carbonate deposits that cement the rocks together.

Geomorphic features combine with past introductions and recent stockings of nonnative salmonids to affect distributions of salmonid species in the Little Bighorn River drainage. Nonnative salmonids probably have moved upstream from low-elevation introduction sites to colonize many headwater streams where barriers to upstream movement are not present. Fish are present in these headwater sites because recolonization from downstream source populations is possible following stochastic events, such as drought, floods, or severe winters, that may periodically decimate fish populations (Kruse et al. 200 I). Several headwater streams do not have fish in them because barriers have prevented colonization from downstream. However, introductions of cutthroat trout or nonnative species upstream from barriers in small, high-gradient headwater streams, e.g., sites in the headwaters Lodgegrass, Pumpkin, Mann, and Wagonbox creeks, would unlikely result in naturalized populations because of limited space, lack of spawning and nursery habitat, and stochastic events that periodically decimate populations (Lamberti et al. 1991, Hildebrand and

Kershner 2000). Due to topography of some of the Upper Little Bighorn River and Dry Fork Little Bighorn River watersheds, sources of nonnative salmonids occur in low- to moderate-gradient headwater streams at high elevations. Recently stocked cutthroat trout or rainbow trout and naturalized brook trout in these headwaters are likely to contribute recruits to downstream areas. Some fish from these stocks probably migrate downstream over barriers, provide a degree of continuity with downstream source populations, and enable salmonids to occur in relatively high-gradient reaches where stochastic events eliminate fish relatively frequently.

Our findings tend to confirm the belief that almost all of the Little Bighorn River drainage was naturally void of native cutthroat trout. Only one small population in the headwaters of Lodgegrass Creek is likely to be native; consequently, conservation efforts should be directed toward that population. The potential exists to create naturalized, allopatric populations of genetically pure Yellowstone cutthroat trout in other portions of the drainage, as apparently happened at the juncture of Pumpkin and Mann creeks. Some headwater streams upstream from natural barriers, currently void of salmonids, may be large enough to support introduced cutthroat trout populations, but most are probably too small or lack sufficient habitat diversity to sustain populations over the long term. Most of the streams in the drainage harbor naturalized populations of exotic salmonids that would require removal prior to cutthroat trout introductions. Also, a lack suitable spawning habitat may discourage naturalization of stocked cutthroat trout in most headwater streams.

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Observations of Northern Goshawk Prey Delivery Behavior in Southcentral Wyoming

Rhett E. Good[1], Wyoming Cooperative Fish and Wildlife Research Unit, Department of Zoology, P.O. Box 3166, University of Wyoming, Laramie, WY 82071
Stanley H. Anderson, Wyoming Cooperative Fish and Wildlife Research Unit, Department of Zoology, P.O. Box 3166, University of Wyoming, Laramie, WY 82071
John R. Squires, Rocky Mountain Forest and Range Experiment Station, 222 S. 22nd Street, Laramie, WY 82070.
Greg McDaniel, Rocky Mountain Forest and Range Experiment Station, 222 S. 22nd Street, Laramie, WY 82070.

Abstract

We observed prey deliveries by northern goshawks *(Accipiter gentilis)* in southcentral Wyoming during the breeding seasons of 1996 and 1997. Female goshawks brooded and fed young, delivered prey, defended young, and maintained nest structures while males delivered prey and defended young. Females tended to be aggressive toward males after prey deliveries. Males delivered prey directly to nests during the nestling stage when females were absent. Males and females continued to deliver prey to nests during the fledgling stage, often being intercepted by the young on their way to nests. Males and females made 71 and 29 percent of deliveries to nests, respectively. Red squirrels *(Tamiasciurus hudsonicus)* were the prey item delivered most often.

Key Words: *Accipiter gentilis,* nest behavior, northern goshawk, prey delivery rates, transfers.

Introduction

The northern goshawk (*Accipiter gentilis*) is considered a sensitive species by the USDA Forest Service in the Rocky Mountain Region. Northern goshawks prefer to nest in mature forests (Reynolds et al. 1982, Speiser and Bosakowski 1987, Squires and Ruggerio 1996).

Food habits of northern goshawks have been well studied (Schnell 1958, Gryzbowski and Eaton 1976, Reynolds et al. 1982, Zachel 1985, Kennedy 1991, Boal and Mannan 1994, Bull and Hohman 1994, Doyle and Smith 1994, Reynolds et al. 1994, Watson et al. 1998). Fewer studies have documented behavior at nests. Previous studies of goshawk nest behavior have typically ended about 7 days after fledging and were conducted without the use of radio telemetry (Zirrer 1947, Schnell 1958, Allen 1978, Lee 1981, Boal 1994, Boal et al. 1994). Although fledging occurs between 36-42 days post-hatch (Boal 1994a), fledglings remain within 300 m of the nest area for approximately 25 days, during which time they are still provisioned by adults (Kenward et al. 1993). Aspects of goshawk ecology, such as male and female provisioning rates, may change after fledging occurs.

Previous work on goshawk nesting behavior assumed the male captured and delivered prey when: 1) the male delivered prey directly to the nest; and 2) vocalizations associated with prey transfers were heard away from nests. The female was assumed to capture and deliver prey only when: 1) the male was not observed or heard during the prey delivery; and 2) the observer did not believe the delivery to be a cache retrieval. In most cases observers had no knowledge of goshawk activities or locations away from nests prior to deliveries.

[1]Current address: Western EcoSystems Technology, Inc., 2003 Central Avenue, Cheyenne, WY 82001

Only Lee (1981) conducted a nest behavior study in the Central Rocky Mountains. As part of a habitat study of kill areas in southcentral Wyoming (Good 1998), we monitored male and female goshawks by telemetry and direct observation to confirm prey deliveries by adults through the nestling and fledgling stages, i.e., up to 62 days post hatch, in south-central Wyoming. We describe prey delivery rates and adult and juvenile behavior at the nest and after fledging.

Study Area

The study was conducted on the Medicine Bow National Forest in the Sierra Madre and Medicine Bow ranges in south-central Wyoming. The USDA Forest Service manages forested lands, whereas the surrounding rangelands are a mosaic of state and private land and land administered by USDI Bureau of Land Management (BLM). Elevation in the Sierra Madre and Medicine Bow ranges varies from about 1828 to 3749 m (Alexander et al. 1986). Temperatures from May through August vary from 15-20 °C during the day, and 0-10 °C at night. Mean annual precipitation varies from 38 cm at 1830 m to 64 cm at 3050 m elevation (Alexander et al. 1986).

Lodgepole pine (*Pinus contorta*) and aspen (*Populus tremuloides*) dominate lower elevation forests between 2590-3050 m, whereas sub-alpine fir (*Abies lasiocarpa*) and Englemann spruce (*Picea engelmanni*) are dominant from 2740 m to timberline. The prevalence of lodgepole pine and sub-alpine fir between 2740 to 3050 m depends upon aspect, soils and other factors (Alexander et al. 1986). Lodgepole pine and aspen stands are interspersed with natural meadows, clearcuts, and roads. We monitored goshawk nests in low-elevation lodgepole pine and aspen forests. Goshawk hunting areas also occurred in low-elevation lodgepole pine and aspen forests (Good 1998).

Methods and Materials

We conducted nest observations from June to mid-August 1996 and 1997. We fitted male and female goshawks with radio transmitters and followed them using three manually operated tracking towers during this same period.

We attached transmitters with tip switches to the adult male and female goshawk at each nest using a backpack harness (weight 25.5 g). Foraging movements of male goshawks were monitored using three manually operated null-peak antennas. Tracking towers were placed on tall knobs surrounding the nest and/or foraging areas. Eight male goshawks were monitored during the summers 1996 and 1997. At 2-week intervals during the breeding season we monitored goshawk movements and observed behavior at each nest. Male foraging movements were monitored from the time goshawk nestlings were 8-20 days to 50-62 days.

We recorded locations of male goshawks during tracking sessions at two-minute intervals until the male returned to the nest with prey. Tracking sessions lasted 24 hours, and locations were taken during daylight hours. One to three foraging bouts were recorded/session, mostly during morning and evening. Locations were determined at 10-min intervals when foraging bouts were not being recorded. Locations also were determined from females every 10 minutes. We confirmed prey deliveries to nests using an observer in a ground blind near each nest. Observers maintained radio contact with telemetry personnel to confirm the presence of adult males in nest stands.

We assessed the accuracy of kill-site locations by comparing calculated locations of test transmitters with actual locations of those transmitters differentially corrected with Global Positioning System (GPS). Technicians on the ground with a transmitter on a 7.9-m (26-ft) pole walked randomly configured transects that were straight-line, half-circle, or sigmoid in shape. Test locations were taken at two-

minute intervals. Test locations determined while the observer was stationary were an average of 100.2 m (SD = 76.25 m, n = 64) from GPS locations. We used only stationary test locations to estimate our accuracy of locating kills because goshawk kill-sites were stationary locations. Stationary telemetry locations were located an average of 155.6 m (SD = 119.6 m, n = 171) from GPS locations.

In Sweden male goshawks usually returned directly to the nest to deliver prey after making a kill during the breeding season (R. Kenward, personal communication.). We identified a kill-site by determining where a goshawk began a direct flight path toward his nest (see Fig. 1). Widén (1982) noted that radio signals of goshawks often were variable in strength while it was chasing prey and were faint when it was on the ground with prey. We observed similar signal patterns that helped us determine the exact point of kill, which preceded the goshawk's return to the nest. We assumed a goshawk's last stationary location before a direct return to the nest marked the kill site.

Goshawks are considered short duration, sit-and-wait predators (Squires and Reynolds 1997). When hunting, goshawks fly to a perch for a short period of time, scanning for prey. The goshawk will then fly to another perch. Inter-perch flight times range from 24 sec (Widén 1984) to 3.5 min (Kennedy 1991). Foraging males remain perched for a median time of 3 min, an average of 8 min, 36 sec (Kennedy 1991). Thus, we believed that we could identify hunting behavior with radio telemetry by noting when a goshawk changed stationary locations every 2–10 min.

We identified male prey deliveries by monitoring their movements at two-minute intervals (Good 1998); we determined if he had exhibited hunting behavior and visited

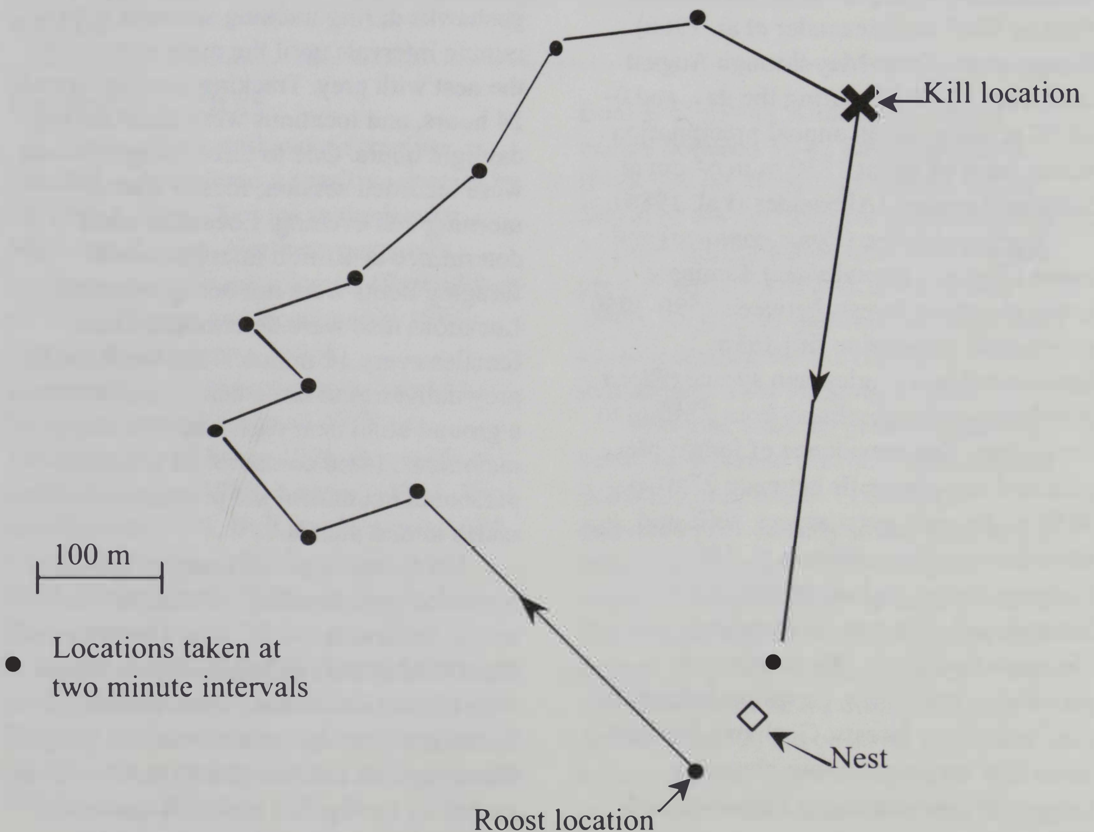

Figure 1. An example of a male goshawk foraging tract.

the nest within approximately 30-minutes of the prey delivery to the nest, or if he was heard, observed, or located by radio signal near the nest, female or fledglings around the time of a prey delivery. We attributed deliveries to females if the male did not appear to be hunting prior to a prey delivery, had not visited the nest area within approximately 30 minutes of a prey delivery, or the delivery did not appear to be a cache retrieval. When prey transfers between male and female goshawks occurred away from nests, we used telemetry to confirm that male and female goshawks were in the same area.

We observed each nest once every two weeks from before dawn until the first prey delivery by the male. If a prey delivery occurred early in the morning (0600-0900 hrs) we continued to observe nests until the male delivered prey a second time or until 1200 hrs, whichever occurred first. We used this approach to accommodate aspects of an ongoing adult habitat use study. We also conducted nest observations from 1700 hrs until the male delivered prey in the evening. To increase the number of deliveries observed we occasionally conducted observations between 1200–1700 hrs. Observation periods averaged 6 hrs, for a total of 295 observation hours (range 11.75–67 hrs/nest, $\bar{x}$ = 36.8 hrs/nest, *n* = 8 nests) over two years.

Results

Male and female prey deliveries

We observed a total of 69 deliveries during 1996 and 1997. Males made 42 deliveries (71%), females made 17 deliveries (29%), and we were unable to determine which adult made the delivery for 10 deliveries. Females delivered prey more frequently than males at two nests. At one of these two nests, the male made no deliveries, whereas the female made at least five deliveries during six days of observation (39 hrs). At the other nest, the male made two deliveries and the female made three deliveries during seven days of observation (40.5 hrs).

Prey transfers and intraspecific behavior

Females tended to be aggressive toward males during food deliveries at nests, giving dismissal calls (Schnell 1958) and alarm calls (Squires and Reynolds 1997). On one occasion a female gave an alarm call and taloned a male when he tried to land on the nest with prey.

During the nestling stage, we identified, through direct observation, the location (at nest vs. away from nest) where males delivered prey on 10 (24 %) of 42 occasions. We observed prey items being transferred from the male to the female away from the nest on three of those 10 occasions. Other deliveries by the male most likely occurred away from the nest, and thus, were not directly observed by us. The male delivered prey directly to the nest on seven occasions during the nestling stage. In all seven cases, the female was not at the nest during the delivery although they were nearby on those occasions and visited the nest within a few minutes of the prey delivery. Males left immediately after delivering prey, except one instance in which a male stood on the nest rim for 30 sec until the female returned to the nest.

Prey were generally delivered to fledglings away from nests (*n* = 8) after fledging (range, 25–400 m) between the ages 39 and 62 days, whereas four deliveries on four different days (fledgling ages 45-55 days) occurred directly at nests. Juveniles were present at the nest prior to the delivery during two of these direct deliveries to the nest.

Only adult females were observed feeding nestlings. Nestlings fed themselves as they matured. We observed the earliest self-feeding at 20 days. The female did the majority of feeding until 26-27 days after which the nestlings increasingly fed themselves. Adults provisioned juveniles up to 62 days post-hatch.

Prey delivery rates

Male and female goshawks delivered 0.23 items/hr/pair (*n* = 8 goshawk pairs, includes deliveries for which the sex of the delivering adult was not determined).

Delivery rates varied between nests. Male and female delivery rates averaged 0.14 items/hr and 0.06 items/hr, respectively.

Food habits

Thirty-eight of 69 items delivered to nests were identified to class (Table 1). Birds comprised 21.1 percent and mammals 78.9 percent of identified deliveries ($n = 38$ deliveries).

Discussion

Feeding behavior at nests was similar to the behaviors described by Schnell (1958) and Boal (1994). Our total delivery rates of 0.23 items/hour were less than 0.25 items/hour reported by Boal and Mannan (1994), and 0.31 items/hour reported by Younk and Bechard (1994). Because we often observed nests until a prey delivery was made, our determination of delivery rates probably overestimated actual delivery rates. Thus, the difference in delivery rates between our study and Boal and Mannan (1994) and Younk and Bechard (1994) probably was even greater than shown above. Younk and Bechard (1994) found goshawks in Nevada to be very productive during one year of their study (2.8 young/ breeding pair) and hypothesized that an increase in prey populations may have resulted in a large number of yearling females breeding. Although we did not measure productivity, goshawks in southcentral Wyoming may have had smaller broods than those in Nevada. Younk and Bechard (1994) also found female goshawks to remain at the nest almost continuously, which may have resulted from a large or readily available prey population (Ward and Kennedy 1996). Prey abundance or availability may have been lower in our study area in Wyoming than in Nevada, thus accounting for our higher rates of female prey deliveries.

Schnell (1958) described females as hostile to males during prey deliveries, but provided no specific examples. Few documented instances of female goshawk aggression toward males exist. Boal et al. (1994) documented males leaving the nest after delivering prey when the female approached and gave dismissal calls. However, Boal et al. (1994) also documented a male goshawk brooding young when a female was giving dismissal calls. We documented a female being physically aggressive toward a male. We know of no other published account of such aggressive behavior.

Male goshawks provided the majority of prey deliveries to the young (71%), which is consistent with other studies (Schnell 1958, Zachel 1985, Younk and Bechard 1994, Boal and Mannan 1996). However, females provided a larger

Table 1. Prey items observed during 69 deliveries to eight goshawk nests in southcentral Wyoming, 1996 - 1997.

Prey item	Number Delivered (%)	Percent of Identified Deliveries (n = 38)
Unknown Bird	5 (7.2)	13.1
American Robin (*Turdus migratorius*)	2 (2.9)	5.2
Northern Flicker (*Colaptes auratus*)	1 (1.4)	2.6
Red Squirrel (*Tamiasciurus hudsonicus*)	12 (17.3)	31.5
Ground Squirrel or Chipmunk	4 (5.8)	10.5
Least Chipmunk (*Tamias minimus*)	1 (1.4)	2.6
Lagomorph	1 (1.4)	2.6
Unknown Mammal	12 (17.3)	31.5
Unknown	31 (44.9)	N/A
Total	69 (100.0)	100.0
Total Birds	8 (11.6)	21.1
Total Mammals	30 (46.3)	78.9

proportion of prey deliveries (29%) to the young in our study than has been previously documented. Zachel (1985) reported two females providing 12.1 percent and 8.8 percent of food delivered to nests. Boal and Mannan (1996) found female goshawks to deliver 19 percent of prey. Schnell (1958) reported a female providing 15 percent of food to the nest. Younk and Bechard (1994) found females did not provide any food during the nestling stage.

Comparison of our delivery rates with other studies was difficult. We assigned deliveries to males and females based upon telemetry efforts and nest observations, whereas previous studies have relied solely upon nest observations and assumptions concerning male delivery behavior. By confirming hunting activity by male goshawks prior to a delivery through use of both motion-sensitive transmitters and direct observation, we perhaps more accurately identified sex of the adult that captured and delivered prey than did earlier researchers.

Additionally, we measured delivery rates through the majority of the fledgling stage, whereas observations were discontinued soon after fledging in earlier studies. Female goshawks may deliver prey more often after young have fledged when the risk of predation is reduced. Additionally, the amount of food needed by the young may be higher after they have fledged and are more active, yet still rely on adults for the majority of their food.

Our results indicated that females in our Wyoming study might capture and deliver prey more frequently than goshawks from other areas. Our study was the first to combine intensive telemetry efforts with nest observations. Schnell (1958), Zachel (1985), Younk and Bechard (1994), and Boal and Mannan (1996) all relied upon visual or auditory cues during nest observations to assign prey deliveries to males or females. We feel that we could more accurately attribute deliveries to either the male or the female based on a combination of telemetry data and visual and auditory cues.

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ANALYSIS OF TOTAL ORGANIC CARBON IN SEVERAL REGIONAL BOTTLED WATERS

Holly Waddell[1] Department of Biological and Physical Sciences, Montana State University-Billings
Billings MT 59101-0298

Will Wickun, Department of Biological and Physical Sciences, Montana State University-Billings
Billings MT 59101-0298

ABSTRACT

We analyzed several different brands of regional-bottled natural spring water for total organic carbon *(TOC)* content using the method of heated persulfate oxidation. Although some distinction can be drawn between the brands considered based on TOC value, all the types of water examined may be considered to be of good quality. We discuss a brief overview of industry and federal standards for bottled water.

Key Words: bottled water analysis, drinking water analysis, TOC analysis

INTRODUCTION

According to the Food and Drug Administration (FDA), bottled water, like all other foods regulated by the FDA, must be processed, packaged, shipped, and stored in a safe and sanitary manner and must be truthfully and accurately labeled. The FDA defines "spring water" as water obtained from an underground formation from which water flows naturally to the surface or would flow if it were not collected underground via a borehole. To be identified as "spring water" on the label, water must be collected at the spring or through a borehole next to the point where it emerges (Federal Register 1993).

Bottled water products also must meet specific FDA quality standards for contaminants (Federal Register 1996). These are set in response to requirements that the U. S. Environmental Protection Agency (EPA) has established for tap water. In addition to the FDA's Quality Standards, bottled water companies must adhere to Standards of Identity (Labeling Regulations) and Good Manufacturing Practices.

The Information Collection Rule - Disinfectant/Disinfection By-Products Rule (ICR-D/DBPR) was promulgated in 1996 by the EPA to determine and establish acceptable levels for occurrence in drinking water of: (1) disease-causing microorganisms, and (2) chemical by-products that form when disinfectants used for microbial control react with naturally-occurring organic compounds already present in source water. The organic material is determined as total organic carbon (TOC) in units of ppm C (mg C/L solution).

Today's ICR-D/DBPR approves three methods for TOC analysis: Standard Method 5310 B (high temperature combustion), 5310 C (persulfate ultraviolet or heated persulfate oxidation), and 5310 D (wet oxidation). To satisfy requirements of the ICR-D/DBPR, a TOC analytical method must have a detection limit of at least 0.5 ppm C and a reproducibility in terms of standard deviation of at least ± 0.1 ppm C over a range of 2-5 ppm C. EPA believes that all of these methods can achieve the precision and detection level necessary for compliance determinations required in today's rule when quality-control procedures contained in method

[1] Current address: Barry University, 11300 NE 2nd Avenue, Miami Shores FL 33161-6695

descriptions and ICR-D/DBPR are followed. Although any of the three methods may be used, EPA advises that the same method be employed for all measurements to reduce the impact of possible instrument bias.

To demonstrate how the EPA Standard Method 5310 C can be applied, we determined concentration of TOC in laboratory tap water, laboratory deionized (DI) water processed by reverse osmosis (RO), and six different bottled waters. The study was not intended to be exhaustive but rather to reveal any regional differences in the TOC of bottled water.

Some of the additional common abbreviations encountered in the EPA Standard Method 5310 C are: TC = Total Carbon, TIC = Total Inorganic Carbon (from naturally occurring carbonates), POC = Purgable Organic Carbon, and NPOC = Nonpurgable Organic Carbon. TOC generally is identical to NPOC, the carbon that can be oxidized to carbon dioxide after the inorganic and volatile organic carbon species have been removed by acidification and purging. Volatile organic species include benzene, carbon tetrachloride, vinyl chloride, and other low-molecular-weight chlorinated hydrocarbons. Although TOC in water samples should ideally include POC, most laboratories report TOC analyses of samples from which volatile organics have been previously removed. In fact, methods involving persulfate oxidation, which are widely accepted and used, each call for acidification and purging to remove inorganic carbon before oxidation of organics. This purging also removes volatile organics before oxidation. However, the results are still generally accepted as TOC. Methods to measure POC have not been standardized and, therefore, POC was not included in this study. Because natural spring waters generally contain insignificant quantities of volatile organic substances, the conditions employed are the same as those for measuring NPOC, whose value, in practice coincides with that of TOC.

Methods

The instrument design of the OI Model 700 TOC Analyzer allows for automated preacidification and purging of the sample aliquot inside the reaction vessel. In this investigation, the water sample was introduced into the digestion vessel of the instrument by means of a 5.00-mL sample loop. The sample loop injection affords greater consistency of volume than would the other option of syringe injection. The sample was then pre-acidified with 200-mL of 5 percent (v/v) phosphoric acid (Baker, Reagent-Grade). After acidification, a gas stream of ultra-pure nitrogen purged out any CO_2 formed from inorganic and volatile organic carbon species. This CO_2 was vented from the instrument. When purging TIC and POC was complete, purge gas to the reaction vessel was shut off, the reaction vessel was heated to 100 ∞C, and 1000-mL of sodium persulfate (Baker, Reagent-Grade, 100 g/L) reagent was added to the sample. Persulfate quickly reacts with organic carbon in the sample to form CO_2, which accumulates in the reaction vessel. After a pre-set reaction time (nominally 3 min), the reaction vessel was placed in-line with the molecular sieve trap and a gas stream purged out any CO_2 produced by the persulfate oxidation. This CO_2 was carried to the trap held at 25 °C where it was retained and concentrated. The trap was then placed in-line with the detector and heated to 200 °C to release the CO_2. The CO_2 was carried into the detector, which is a linearized, non-dispersive, single-beam analyzer sensitized to detect 0-50 mg carbon as CO_2. The resulting concentration of TOC in the sample was displayed/ printed.

Sample (bottle) size was 500 mL and the instrument was pre-flushed with the sample to remove any residual traces of the previous sample. The minimum detection limit (MDL) of the instrument was calculated as three times the standard deviation of detector response (in mV) to 20 external blanks consisting of ultra-pure reagent water (EM Science OmniSolv®).

We determined concentrations of TOC in the analytical samples using a two-concentration calibration curve generated with primary-standard grade potassium hydrogen phthalate (Baker) in ultra-pure reagent water and spanning a concentration range from zero to 1.0 ppm C. Ten samples each of the blanks and of the standards were analyzed in duplicate, and the responses from all 40 analyses were used to generate the curve, which was then programmed into the instrument. The calibration was routinely verified prior to the determination of each different analytical sample.

We analyzed 10 samples of each brand of natural spring water in triplicate/bottle. The samples were obtained in batches from several different retail outlets in the region. All brands selected were supplied in PET (polyethylene terephthalate) bottles. HDPE and LDPE bottles, which tend to impart the distinct taste of plastic, were avoided.

No specific preliminary preparation of the samples was carried out, but because of the sensitivity of the method, we carefully washed and rinsed all glassware with ultra-pure water.

Results And Discussion

The MDL of the instrument was determined to be 0.0155 ppm C as TOC and this compares very favorably with the TOC range of 0.004 to 10,000 ppm C specified by the manufacturer. Results from the analysis of TOC in six bottled drinking waters appear in Table 1. Concentrations in the bottled waters ranged from 0.162 to 0.573 ppm C. All results were well above the MDL for this instrument. The largest differences in TOC within one brand of water appeared in the Giant Springs brand. The data for Giant Springs had a surprisingly large standard deviation over the range 0.183-0.461 ppm. Application of

Table 1. TOC values of brands of bottled spring water analyzed

Brand of Water (Source, if available)	TOC (ppm) of 10 Samples per Brand with 3 Replicate Measurements per Sample Bottle*					Mean TOC (ppm) with Std Dev from Mean of 10 Samples
Cascade Clear	**0.168**	**0.196**	**0.148**	**0.154**	**0.158**	0.162 ± 0.018
	(0.004)	(0.065)	(0.004)	(0.014)	(0.005)	
	0.149	0.149	0.151	0.151	0.194	
	(0.003)	(0.003)	(0.005)	(0.001)	(0.069)	
EartH2O (Opal Springs)	**0.164**	**0.176**	**0.212**	**0.207**	**0.180**	0.194 ± 0.022
	(0.003)	(0.001)	(0.052)	(0.054)	(0.001)	
	0.186	0.181	0.186	0.226	0.225	
	(0.002)	(0.008)	(0.004)	(0.047)	(0.018)	
Giant Springs	**0.444**	**0.431**	**0.275**	**0.324**	**0.291**	0.320 ± 0.095
	(0.062)	(0.055)	(0.023)	(0.002)	(0.014)	
	0.461	**0.183**	**0.300**	**0.258**	**0.236**	
	(0.070)	(0.069)	(0.010)	(0.031)	(0.042)	
Montana Silver Spray	**0.456**	**0.494**	**0.519**	**0.487**	**0.573**	0.528 ± 0.054
	(0.010)	(0.059)	(0.110)	(0.071)	(0.108)	
	0.546	**0.515**	**0.628**	**0.585**	**0.476**	
	(0.086)	(0.083)	(0.038)	(0.058)	(0.051)	
Mountain Mist (artesian water)	**0.546**	**0.568**	**0.613**	**0.550**	**0.589**	0.573 ± 0.035
	(0.087)	(0.072)	(0.052)	(0.081)	(0.079)	
	0.600	**0.587**	**0.501**	**0.612**	**0.564**	
	(0.043)	(0.065)	(0.004)	(0.082)	(0.075)	
Red Lodge (Crystal Springs)	**0.250**	**0.246**	**0.247**	**0.235**	**0.230**	0.237 ± 0.013
	(0.014)	(0.018)	(0.013)	(0.008)	(0.012)	
	0.218	**0.226**	**0.227**	**0.233**	**0.259**	
	(0.019)	(0.004)	(0.007)	(0.005)	(0 016)	

(*Numbers in parentheses are respective standard deviations of three replicates.)

the Student's t-statistic indicated that this noticeably large difference in TOC is probably real at the 95 percent confidence level. Within other brands, however, there is no apparent difference in the TOC measured. Table 2 lists the typical TOC in laboratory tap water, in laboratory DI water prepared by reverse osmosis, in ultra-pure reagent water, and in a 1.000 ppm carbon commercial standard (Ultra Scientific). The laboratory tap water, which is unfiltered municipal water, may have a slightly higher than normal TOC, presumably because of drought conditions that existed in the region at the time of this study. Although these data have a large standard deviation, significance testing indicated that an apparent discrepancy was no greater than what might be found by chance simply because of natural variability. The level of TOC in laboratory DI water, 0.119 ppm C, is typical of this large-throughput water purification system. The accuracy (1.009 ppm) and the small standard deviation (± 0.033 ppm) in the TOC value for the commercial (1.000 ppm C) standard indicated high precision for this method.

According to the International Bottled Water Association (IBWA) the deteriorating taste and quality of tap water and fear of unknown contaminants have lead many Americans to believe that bottled water is of higher purity than tap water. According to a recent study performed by the IBWA, (International Bottled Water Association 2000), 63 percent of Americans are not aware that the FDA regulates bottled water as a food product. Among those with the knowledge that the FDA regulates bottled water, most people (53%) feel more confident about bottled water's purity and safety. Seventy-one percent of Americans feel that the quality of bottled water is high, 30 percent feel that it is extremely or very high, while another 41 percent feel it is somewhat high.

However, the Natural Resources Defense Council (1999) issued a report concluding that bottled water is insufficiently regulated by federal and state agencies and by the bottled water industry. Further, based on an analytical product survey conducted by NRDC, bottled water may not be as pure as we are led to believe.

The solution recommended by NRDC (1999) was to overhaul the FDA's regulatory regime for bottled water or give the program to EPA and impose additional

Table 2. Typical TOC values of laboratory tap water, laboratory DI water, ultrapure water, and 1.000 ppm external standard

Sample	TOC (ppm) of 1 Replicate per Sample					Overall MeanTOC (ppm)with Std Dev
Laboratory	1.401	1.407	1.395	1.425	1.419	1.441 ± 0.089
tap water	1.728	1.528	1.426	1.362	1.477	
(municipal)	1.390	1.386	1.558	1.466	1.515	
	1.389	1.388	1.433	1.389	1.326	
Laboratory	0.036	0.088	0.131	0.131	0.141	0.119 ± 0.023
DI water	0.131	0.120	0.148	0.128	0.122	
(by RO)	0.114	0.114	0.122	0.123	0.122	
	0.114	0.118	0.129	0.118	0.120	
Ultra-pure	0.036	0.016	0.025	0.026	0.028	0.032 ± 0.007
(OmniSolv“)	0.026	0.026	0.026	0.028	0.031	
	0.030	0.035	0.033	0.040	0.038	
	0.032	0.037	0.037	0.040	0.039	
1.000 ppm	1.066	1.041	1.049	1.005	0.989	1.009 ± 0.033
Standard	0.962	0.969	0.981	0.986	1.005	
(Ultragrade‘)	1.024	0.978	1.031	1.053	1.058	
	1.003	0.982	1.022	0.961	1.022	

disclosure requirements on the bottled water industry.

The NRDC (1999) contended that federal bottled water regulation is weaker than the tap water regulations facing public water systems because, in part, the FDA regulatory structure for setting allowable contaminant levels and associated monitoring and treatment requirements is not as stringent as that of the EPA for tap water.

Closer examination of the water quality standards for chemical contaminants revealed that the FDA bottled water quality standards are the same as EPA's tap water standards for 62 out of 71 chemical substances highlighted in the NRDC Report (1999: Chapter 4, Table 6). FDA standards for lead, copper, and fluoride are stricter than EPA's. For three of the remaining contaminants—asbestos, acrylamide, and epichlorohydrin—FDA has determined, as the law allows it to do, that establishing specific standards for bottled water is unnecessary.

States are under no legal obligation to adopt the FDA bottled water standards, and state regulations for bottled water, if any, vary widely. Although some states, such has California, Texas, and Washington, have bottled water programs that are relatively well developed, other states, such as Alaska, Utah, and orth Dakota, have no or virtually no program. Of interest in thi study is the fact that Montana's state bottled water program, which is stricter than that set by the FDA, requires that all in-state bottlers become Public Water Systems and meet EPA drinking water standards prior to start-up. In the final analysis, individual consumer's taste and perception of cleanliness apparently will determine the quality of bottled water.

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Corrigendum

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... Recharge Area

The size of the recharge area of each spring system was estimated using established values for spring discharge, local precipitation, and evapotranspiration in the following equation for a water budget in a watershed

$$P = ET + R + G + \Delta S \qquad (7)$$

where P is precipitation, ET is evapotranspiration, R is surface runoff, G is groundwater recharge, and ΔS is the change in storage (Manga 1997). Change in storage for the short time period under consideration here would make a relatively small contribution to the total water budget and this term was therefore omitted. I assumed that each spring system was fed by groundwater; thus, G was estimated by the measured discharge in the springs. Surface runoff in the study area is negligible with no evidence of perennial streams or developed stream channels other than those supplied with water from springs. Precipitation values for the proposed recharge areas were derived using regression analysis from mean precipitation values at Snotel sites in the region obtained from the NRCS and Dirks and Martner (1982) (Table 3, Fig. 5). Potential evapotranspiration values were obtained from Martner (1986) and Wyoming Water Development Commission and University of Wyoming (1990). Given mean annual precipitation and evapotranspiration rates, an estimate of the area needed to supply enough water for each spring system's discharge was calculated from equation (7). I used a 1:100,000 scale topographic map of the region and a 50-cm by 35-cm transparent grid with 1-cm squares to estimate recharge areas for each of the spring systems based on the areas calculated above and regional surface topography.

Table 3. Mean annual precipitation rates for Snotel sites in the Henry's Fork region (1961 to 1991).

Site	Elevation (m)	Precipitation (mm)
Black Bear, MT	2423	1567
Madison Plateau, MT	2362	1084
Whiskey Creek, MT	2072	929
Big Springs, ID	1981	779
Ashton, ID	1584	523
Island Park, ID	1917	767
White Elephant, ID	2350	1219
Lewis Lake Divide, WY	2392	1447
Grassy Lake, WY	2214	1422

Heat Flux

The total heat, H, discharged by each of the Henry's Fork springs was calculated using the equation

$$H = \rho C q \Delta\theta \qquad (8)$$

in which ρ and C are the density and heat capacity of water, q is the discharge from the spring, and $\Delta\theta$ is the change in temperature of the groundwater between recharge and discharge (Manga 1998, in press). It is assumed that groundwater movement in the aquifer advects all the heat horizontally. The mean heat flux entering the base of the aquifer is equal to the total heat flux divided by the surface area of the aquifer. ...

Results

Stable Isotopes

Trends evident in $\delta\,^{18}O$ and $\delta\,D$ values of spring water, snow cores, and summer precipitation in the Henry's Fork region (Table 2)* included: 1) little year-to-year or seasonal variability in the stable isotope values for the spring waters; 2) Snow Creek

*See original article Vol. 6 (3).

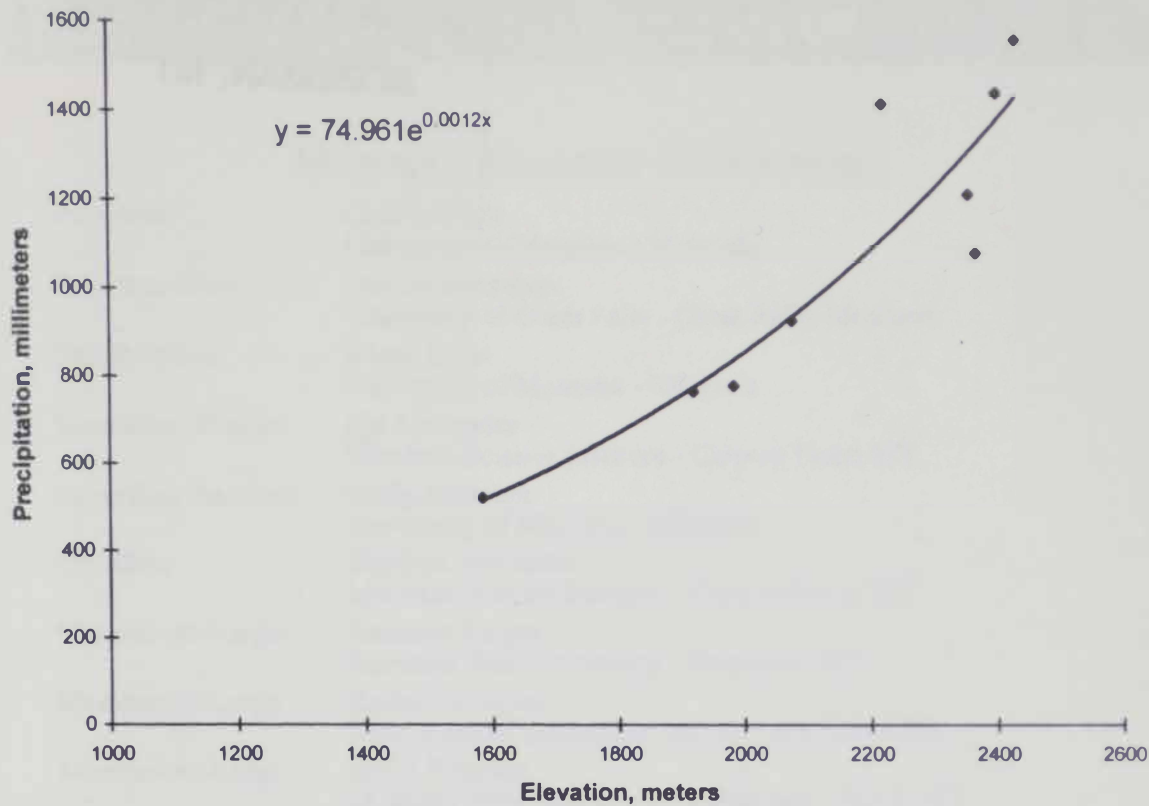

Figure 5. Mean annual precipitation as a function of altitude in western Yellowstone National Park and eastern Island Park, Idaho.

and Warm River water had the heaviest stable isotope ratios; 3) stable isotope ratios of snow cores were lighter than spring water at Big Springs and Lucky Dog, heavier than spring water at Warm River springs and the same as spring water at Snow Creek; 4) stable isotope values of all spring waters and snow cores were significantly lighter than of summer precipitation; 5) stable isotope values of summer precipitation were heaviest at Warm River and became progressively lighter to the north; and 6) the lightest stable isotope values of snow cores were those on the eastern side of the Continental Divide. ...

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