Response of the Green River Fish Community to Changes in Flow and Temperature Regimes from Flaming Gorge Dam since 1996 based on sampling conducted from 2002 to 2004

By

Kevin R. Bestgen, Koreen A. Zelasko, and Robert I. Compton
Larval Fish Laboratory
Department of Fishery and Wildlife Biology
Colorado State University
Fort Collins, Colorado 80523

and

Tom Chart
U.S. Fish and Wildlife Service
2367 W. Oaten Circle, Suite 50
West Valley City, Utah 84119

Final Report
Colorado River Recovery Implementation Program Project Number 115
Larval Fish Laboratory Contribution 144

March 2006

TABLE OF CONTENTS

EXECUTIVE SUMMARY

The Green River in Utah and Colorado, downstream of Flaming Gorge Dam, offers an opportunity to evaluate changes in the fish community in relation to several different dam operation regimes. A primary purpose of this study was to determine the cumulative effect of flow and temperature regime changes on physical habitat and native and non-native fishes in the Green River downstream of Flaming Gorge Dam, including Browns Park, Lodore and Whirlpool canyons, and Island-Rainbow Park. We place emphasis on comparisons between the 1994 to 1996 and 2002 to 2004 periods. To provide a more comprehensive historical context, we also compare these recent data to findings from the pre-impoundment period (before 1962), and two other post-impoundment periods, 1963 to 1978, and 1978 to 1992. We also characterize the distribution of the humpback chub in Whirlpool Canyon. Recommendations for monitoring effects of Flaming Gorge flow and temperature regimes on the fish community on the study reach are also provided.

Four major dam-related events affected physical habitat and the fish community of the upper Green River: dam installation in 1962, penstock modification in 1978, flow re-regulation beginning in 1992, and higher peak flows, and lower base flows, and drought events since 1997. The impacts of construction and operation of Flaming Gorge Dam on physical habitat in the highly regulated reach of the Green River from the dam downstream to the Yampa River and in Whirlpool Canyon downstream of the Yampa River were partially remediated by thermal modifications implemented in 1978, discharge re-regulation in 1992, and 2002 to 2004 drought-period changes to baseflow levels and patterns. High releases in 1997 and 1999 and drought-induced high water temperatures created conditions that more closely resembled pre-dam conditions. Peak flows were relatively high in 1997 and 1999; peak flows in most other years were at the relatively low power plant capacity including those from 2002 to 2004. Flows in the

2002 to 2004 study period also had a more stable, albeit slightly higher than historical, summer-winter base flow pattern because power generation fluctuations at Flaming Gorge Dam were minimized. In Lodore Canyon, drought in 2002 to 2004 resulted in low base flows, and thermal regimes in the study area were more similar to those in the pre-dam era than during any other post-impoundment period. This ensured a close match of Green and Yampa River water temperatures during drought years when Colorado pikeminnow larvae were drifting downstream from the Yampa River in summer.

Green River peak flows in Whirlpool Canyon in the pre-impoundment period showed a typical dome-shaped hydrograph, with low late-summer, autumn, winter, and early-spring base flows, rising to a peak in late spring and early summer. Post-impoundment Green River peak flows in Whirlpool Canyon were lower and arrived earlier than historically, an effect of Flaming Gorge Dam. Whirlpool Canyon flow peaks during 2002 and 2004 were lower than the historical average while the 2003 peak flow exceeded the average historical condition (1946 to 1962). Base flows in Whirlpool Canyon in the 2002 to 2004 period matched the historical average pattern reasonably closely. Thermal regimes in 2002 to 2004 were similar to historical ones because dam-related cooling was probably overwhelmed by enhanced warming during low flows.

We used raft-based electrofishing gear, seines, drift nets, and trammel nets to sample Green River fishes from Browns Park downstream to Island-Rainbow Park. When data obtained with all gears were combined, native fishes in the 2002 to 2004 period comprised only 10.3% of total catch and non-natives were 89.3%; the remaining 0.4% were hybrids. Fish species richness was lowest in upstream reaches and cold-water or cool-water tolerant species were more abundant in electrofishing samples. Predominant taxa included brown trout, white sucker, fathead minnow, and mountain whitefish. Species richness was higher in downstream reaches and species that tolerated warm water, such as sand shiner, red shiner, channel catfish, and smallmouth bass, were more abundant. Flannelmouth sucker, bluehead sucker, and common carp were widespread.

The Green River upstream of the Yampa River supported eight native fishes (nine if stocked bonytail is included) in 2002 to 2004, and only Colorado pikeminnow did not reproduce. Native fishes in Lodore Canyon in 2002 to 2004 declined in abundance in electrofishing, seine, and drift net samples compared to those collected in 1994 to 1996. Ten native fishes were collected in the Green River downstream of the Yampa River. Native fishes were numerically dominant in trammel net samples in Whirlpool Canyon; no comparative 1994 to 1996 data were available for that reach.

Abundance of non-native fishes in Browns Park and Lodore Canyon in 2002 to 2004 increased since 1994 to 1996 sampling. Increases were throughout the study area and were particularly evident for small-bodied cyprinids, channel catfish, and smallmouth bass in the upper portions of Lodore Canyon. Smallmouth bass reproduction, which was not observed in Lodore Canyon prior to this study, increased through the 2002 to 2004 period and was widespread. Salmonids were temporarily reduced in abundance in 2002, but increased in 2003 and 2004, and were similar in abundance to that observed in 1994 to 1996. Abundance of predaceous fishes in the Green River study area has increased.

The distribution and abundance of several warm-water fishes in upper Lodore Canyon may have been restricted by relatively cool water temperatures in upstream reaches in most years. Previously, this limitation was observed in lower Lodore Canyon but moved upstream because of increased summer temperatures in 2002 to 2004. Abundance of several combinations of hybrid suckers was high and increased since 1980 and again since the 1994 to 1996 period. Particularly common were hybrids that had white sucker as one parental type. Occurrence of white sucker and hybrids declined in a downstream direction in Lodore Canyon, presumably in response to warmer water temperatures. Possible occurrence of Utah sucker is being investigated with comparisons of museum specimens.

Trammel net sampling in Whirlpool Canyon detected a small population of humpback chub and a relatively large population of roundtail chub. Hatchery-stocked bonytail were also captured. Drift net sampling failed to detect reproduction by Colorado pikeminnow. Drift net sampling revealed that fish, including early life stages of smallmouth bass, were displaced in response to turbidity and higher flow events.

Colorado pikeminnow continued to use Lodore Canyon heavily in summer, based on captures we made and those in a concurrent companion study. Ripe male Colorado pikeminnow were detected there in 2001 and 2003, indicating suitable conditions for reproduction and perhaps, attempted spawning.

Based on our 2002 to 2004 sampling, the net effect of flow and temperature regimes on the native fish community, was mixed. During this study, we were able to obtain reliable information on the response of the fish community to flow and temperature effects, but only at the lower end of the flow spectrum and the high end of the temperature spectrum. Flow and temperature conditions observed in 2002 to 2004 fell within the bounds for recommendations that would be implemented in years with low or moderately low water availability, which happen in only 30% of flow years. Average, moderately high, and high flow years, which occur the other 70% of the time, were not evaluated because those conditions were not realized during this study period. Additional years of sampling when those flow conditions are available are needed to fully assess the effects of pending flow and temperature recommendations for Flaming Gorge Dam on the fish community of the Green River.

We recommend additional monitoring of water temperatures, flows, and the fish community of the Green River in Browns Park, Lodore and Whirlpool canyons, and Island-Rainbow Park. Comparisons of fish community response to different flow and temperature regimes from Flaming Gorge Dam may enhance understanding of factors that may limit invasive species, particularly predators. We also recommend PIT-tagging and scanning all chubs captured, including roundtail chubs, and removal of predaceous and other non-native fishes from the study area.

LIST OF TABLES

LIST OF FIGURES

Page

KEY WORDS

Colorado pikeminnow, roundtail chub, humpback chub, razorback sucker, river regulation, restoration, endangered fishes, longitudinal abundance patterns, drift net sampling, fish community change, water temperature

INTRODUCTION

Altered flow, sediment, and temperature regimes in regulated rivers have been implicated as factors responsible for reduced distribution and abundance of native aquatic biota (Petts 1984, Ward 1989, Ward and Stanford 1995, Stanford et al. 1996, Poff et al. 1997). In snow-melt dominated streams in western USA, dams reduce spring discharge maxima, which disrupts channel-flood plain interactions and channel forming processes. Deep-releases from reservoirs reduce spring and summer water temperature downstream from dams and cause shifts in composition of fish and invertebrate communities from warm stenothermic native species to non-indigenous cold-tolerant eurytherms (Holden 1979, Ward and Stanford 1979, Stanford et al. 1996). Remnant populations of native fishes may persist as long-lived adults, but reproduction is often reduced or eliminated by cold spring and summer water temperatures and other effects of regulation. Non-indigenous species that spread from reservoirs or tailwaters are another mechanism for reduction of native species. Such reductions may manifest through negative effects of predation, competition, or introduction of diseases and parasites and may be especially problematic in western USA systems because indigenous communities have naturally low diversity and may be susceptible to invasion (Minckley and Deacon 1968, Stanford and Ward 1986, Moyle et al. 1986, Carlson and Muth 1989, Minckley and Deacon 1991, Olden et al. In press).

Protocols for river restoration promote re-establishment of natural physical processes in streams with hypothesized benefits for native biota, including fishes (Stanford et al. 1996, Poff et al. 1997). Re-establishment of annual flow maxima to maintain channel geomorphology and flood plain connectedness, reduction of baseflow fluctuations to enhance stability and food web functioning of low velocity nearshore areas, and restoration of more natural seasonal temperature patterns are examples of key processes that may facilitate recovery of native biota (Stanford et al. 1996). However, tests of river restoration hypotheses are few (but see Propst and Gido 2004).

A main challenge of testing river restoration hypotheses is linking population measurements such as fish abundance to a driving variable or set of variables with reasonable certainty. For example, it is difficult to isolate effects of flow or temperature regimes when distribution and abundance of invasive species are expanding, perhaps though interactions with variable environmental conditions (Bestgen et al. In press). Because cause and effect experiments are not possible in this setting, correlations and weight of evidence approaches are the prevailing technique to accomplish this (Beyers 1998). Populations that are composed of species whose biology is poorly understood and which are relatively long-lived make these assessments more difficult, because responses to changes are slow and assignment of causative factors is difficult.

The reach of the Green River, Utah and Colorado, downstream of Flaming Gorge Dam offers an opportunity to evaluate effectiveness of actions to re-establish native fishes. This is an ideal study area because more natural hydrologic and temperature regimes designed to benefit native fishes have been implemented over a series of years. The Green River also has a source pool of native fishes downstream, including the mostly unregulated and tributary Yampa River and regulation-attenuated downstream reaches of the Green River, to re-colonize upstream reaches. Furthermore, descriptions of the fish community are available from immediate pre-dam (pre-1962) and post-dam (1964-1966) periods, as well as before and after installation of a temperature control device (1978-1980). Additional sampling was conducted from 1994 to 1996 that further evaluated changes in the fish community following implementation of flow recommendations contained in the 1992 Biological Opinion on operation of Flaming Gorge Dam (U. S. Fish and Wildlife Service 1992, Bestgen and Crist 2000).

Since 1996, a new set of flow and temperature recommendations for the Green River was developed (Muth et al. 2000). Those recommendations will not be fully implemented until Endangered Species Act and National Environmental Policy Act compliance activities are complete, which is expected in 2006. Annual peak and base flow recommendations were developed for each of five hydrologic conditions that ranged from dry to wet, based on exceedance probabilities: dry (90 to 100% exceedance), moderately dry (70 to 90% exceedance), average (30 to 70% exceedance), moderately wet (10-30% exceedance), and wet (0-10% exceedance). Flaming Gorge Dam spring peak release recommendations range from 130 m^3/s (full power-plant capacity) in dry years to > 244 m^3/s in wet years. Base flow (August to February) release recommendations were also scaled to hydrologic conditions and ranged from 23 to 28 m^3/s in dry years to 79 to 85 m^3/s in wet years. Warmer releases (up to 15°C) and lower summer base flows were also recommended to improve downstream water temperature conditions for native fishes in the Green River. Specifically, water temperatures of 18 to 20°C were targeted for two to five weeks in summer in Lodore Canyon. Another goal of warmer summer water temperatures was to ensure that the Green River was no more than 5°C cooler than the Yampa River, to reduce the possibility of cold shock to Colorado pikeminnow *Ptychocheilus lucius* larvae as they drift downstream from the Yampa River. The range of flow and temperature recommendations were expected to benefit all life stages of native, endangered fishes and further their recovery. Muth et al. (2000) recognized "uncertainties" regarding effects of recommended flow and temperature regimes. A primary uncertainty was the extent to which native or endangered fishes might benefit from recommended flow and temperature regimes and whether negative effects of hypothesized increased distribution and abundance of certain non-native fishes would offset benefits.

In the years following 1996, some reservoir operations that matched recommendations in Muth et al. (2000) occurred. For example, flows in excess of 226.6 m^3/sec (8,000 cfs) were released in spring 1997, and flows in excess of 283.3 m^3/sec (10,000 cfs) were released in spring 1999. The magnitude of those spring releases, which were not exceeded since 1986, fell within the 1992 Biological Opinion recommendations, but were timed to be consistent with the Muth et al. (2000) recommendations. In addition, low summer base flow releases in 2000 and 2001 were patterned to follow the hydrologic water year (e.g., Muth et al. 2000). Also consistent with Muth et al. (2000) recommendations, main channel temperatures in excess of 20°C were maintained in Lodore Canyon in July and August 2000.

Because portions of the new flow and temperature recommendations occurred since 1996, we thought it timely to reinitiate sampling of the fish community downstream of Flaming

Gorge Dam in 2002. Information gathered will be used to evaluate whether flow and temperature regimes based on Flaming Gorge Dam recommendations are benefitting endangered or native fishes in the Green River without increasing distribution and abundance of non-native fishes.

Study goals and specific objectives were:

Goal. Determine if changes in Green River flow and thermal regimes since 1996 are associated with changes in distribution and abundance patterns of native and non-native fishes in Browns Park and Lodore Canyon. Information will be used to evaluate if flow and temperature regimes from Flaming Gorge Dam are benefiting endangered fishes without causing detrimental increases in abundance of non-native fishes.

Objective 1. Determine if shifts in distribution and abundance of large-bodied fishes occurred in Browns Park and Lodore Canyon by comparing results of shoreline electrofishing and trammel net surveys with results of previous studies, particularly Bestgen and Crist (2000).

Objective 2. Determine if shifts in distribution and abundance of small-bodied fishes occurred in Browns Park and Lodore Canyon by comparing results of seining low-velocity, nearshore areas with results of previous studies, particularly Bestgen and Crist (2000).

Objective 3. Determine if Colorado pikeminnow spawn in the Green River upstream of the Yampa River confluence by establishing a larval drift station in lower Lodore Canyon, and by summer sampling to determine presence of ripe adults.

Objective 4. Collect data from the hydrological records for two time periods (WY 1992-1996 vs. 1997-2004) as recorded by the USGS at their near Greendale, Utah gaging station (09234500).

Objective 5. Collect main channel temperatures in Browns Park and Lodore Canyon in coordination with other ongoing efforts.

Objective 6. Compare and contrast the summer thermal regime of the Green River through Browns Park and Lodore Canyon for two time periods (WY 1991 to 1996 vs. 1997 to 2003). Temperature models will be used to generate thermal regimes, where empirical data are lacking.

Goal: Develop a program to monitor the effect of future Flaming Gorge operations on the fish community in Lodore and Whirlpool canyons.

Objective 7. Conduct fish community sampling into Whirlpool Canyon.

Objective 8. Collect flow data in Whirlpool Canyon USGS stations at the Deerlodge Park gage on the Yampa River, and the Greendale, Utah and Jensen, Utah gages on the Green River.

Objective 9. Collect water temperature data in Whirlpool Canyon.

Objective 10. Based on the conclusions of these investigations (Obj. 1-9), recommend a process to monitor future effects of operations at Flaming Gorge Dam on the downstream fish community.

Goal. Characterize the humpback chub *Gila cypha* population in Whirlpool Canyon.

Objective 11. Gather preliminary information to describe the distribution, relative abundance, and size structure of *Gila spp*. in Whirlpool Canyon through the fish community sampling methodologies.

Objective 12. Characterize morphology of the *Gila spp*. collected in Whirlpool Canyon (Douglas et al. (1989; 1998).

Objective 13. Develop program to monitor the humpback chub population in Whirlpool Canyon.

STUDY AREA

The Green River Basin drains mountainous and high desert portions of southern Wyoming, eastern Utah, and northwestern Colorado (Fig. 1) and is the largest tributary of the Colorado River. The study area encompassed the Green and Yampa rivers in and above Dinosaur National Monument and included highly regulated and regulation-attenuated river reaches. The Green River below Flaming Gorge Dam downstream to near Jensen, Utah, was the primary focus of the study. The river below Flaming Gorge Dam flows for 22 km through Red Canyon, enters the 48-km-long low gradient Browns Park valley, and then flows for 32 km through high-gradient Lodore Canyon before entering Echo Park at the Yampa River confluence. From Echo Park, the Green River flows 18 km mostly through Whirlpool Canyon, another 11 km through Island-Rainbow Park, and another 13 km through Split Mountain Canyon before emerging into the alluvial Uintah Basin valley reach near Jensen.

Extent of river meandering (river plan form) and habitat types in the major reaches of river are controlled by local geology. In Lodore, Whirlpool, and Split Mountain canyons the

Green River is generally confined to a single, relatively narrow channel. Canyon river reaches constricted by debris fans form riffles and rapids with cobble and boulder substrate. These river reaches generally have higher current velocities and deeper runs and pools than lower gradient areas without debris fans. In lower gradient alluvial reaches such as Browns and Island-Rainbow parks, river plan form is characterized by restricted meanders (Grams 1997), and the channel is relatively wide, shallow, and sometimes interspersed with islands. Lower velocity runs with sand and cobble substrate predominate.

Flaming Gorge Dam in northeastern Utah impounds the mainstem Green River at the upstream end of the study area. With a storage capacity of 3.74 million acre-ft (4,624 million m^3), it is the largest reservoir in the Green River Basin. Upstream Fontenelle Reservoir, the only other mainstem Green River reservoir, does not appreciably affect releases from Flaming Gorge Dam. Flaming Gorge Dam was closed in 1962 as part of the Colorado River Storage Project (CRSP) to store water so that states of the Upper and Lower Colorado River basins (Wyoming, Colorado, Utah, New Mexico, Nevada, Arizona, California) could develop their respective water entitlements. The dam and reservoir also produce hydropower and provide fish and wildlife benefits. Mean annual discharge in the Green River upstream of the Yampa River has not changed appreciably as a result of dam closure (59 m^3/s post-dam versus 56 m^3/s pre-dam). However, seasonal flow variability has been considerably reduced. Since Flaming Gorge Dam was closed in 1962, June mean maximum flow of the Green River has been reduced from 381 m^3/s to 139 m^3/s (Tyus and Karp 1991, Muth et al. 2000). Flows at all other times of the year have increased. In addition, releases for peaking power production were made up to twice per day, resulting in short-term daily fluctuations much greater than historically occurred. Annual volume of sediment transported by the Green River below Flaming Gorge Dam has decreased by 54% as a result of deposition in the upstream reservoir (Andrews 1986).

The relatively unregulated Yampa River, a major tributary of the Green River, exhibited greater seasonal variability and more stable daily flows than the Green River below the dam. During the period 1963 to 2004, annual flow maxima in the Yampa River occasionally reached

566 m^3/s, but sometimes declined to < 2 m^3/s in late summer (U. S. Geological Survey et seq.; Maybell Gauge, 09251000).

Methods

Collections of small-bodied (< 150 mm TL, most were 25 to 75-mm TL) and large-bodied (150- mm TL or greater) fishes were made from 2002 to 2004 to describe current distribution and abundance patterns of fishes in the Green River from Swinging Bridge in Browns Park to the downstream end of Island-Rainbow Park. Water temperature and discharge data were used to understand the relationship of physical factors to longitudinal distribution and abundance patterns of fishes in the regulated portion of the Green River. Discharge and temperature regimes associated with specific dam operations were compared to changes in fish distribution and abundance patterns.

Collection of small-bodied fish.—Low-velocity channel margins in Browns Park, Lodore and Whirlpool canyons, and Island-Rainbow Park were sampled with seines (1.3 and 4.6 m length, 1.6 and 4.7-mm mesh). Seine sampling was conducted in spring, summer, and autumn in each year from 2002 to 2004. Habitat types sampled were mostly backwaters, but channel margin eddies, low velocity runs and pools, and a few riffles were also sampled. More effort was expended in larger habitat areas and less in smaller ones so that the proportion of the habitat sampled was approximately equal across different-sized areas. Riffles were occasionally "kick-seined", whereby substrate was vigorously disturbed and dislodged fish were captured in a stationary downstream seine. Some vouchers were preserved in 10% formalin and identified at the Larval Fish Laboratory, Colorado State University. We scrutinized all young chubs captured and preserved in 2002 to 2004 by counting dorsal and anal fin rays, and characterizing shape characteristics of the snout and mouth (mouth terminal in roundtail chub *Gila robusta,* slightly subterminal in humpback chub), body depth, fin lengths, and the line of the angle of the anal fin base relative to the upper lobe of the caudal fin, which assisted with identifications (Muth 1990, Douglas et al. 1989).

Drift net samples collected in the Green River just upstream of the Yampa River in 2002 to 2004 were used to describe species composition and abundance patterns of fish larvae transported downstream and to determine if upstream spawning of Colorado pikeminnow had occurred. Samples were collected daily in the morning (ca. 0700 to 0900 h) with conical drift nets (0.15 m^2 mouth diameter, 4 m long, 560 μm mesh) set near shore in water 30 to 50-cm deep. Three nets were set on each sampling occasion for up to 2 h, but sampling ceased if debris load exceeded 3.8 L/sample. Water depth at which a white object disappeared from sight was recorded as a measure of water turbidity. General Oceanics flow meters (model 2030) suspended in each net mouth recorded water velocity. Samples were fixed immediately in 100% ethanol and fish were removed from debris within 4 h and preserved in 100% ethanol. Ethanol was used to ensure that fish otoliths or other tissues were useful for later analysis, if desired.

Collection of large-bodied fish.—Most collections of large-bodied fishes were by electrofishing and (mostly in Whirlpool Canyon) trammel nets; angling was used on a limited basis. Two electrofishing sampling trips were made each year from 2002 to 2004, one in mid to late July and one in mid to late September. Inflatable raft-based Smith-Root electrofishers were employed. Water conductivity was 300 to 700 microsiemens and electrofishing units usually produced 3-6 amperes with about 350 volts. Generally, 1.5 to 3 km reaches were electrofished before fish were identified, and lengths and weights taken on about every other sample. Electrofishing effort was continuous throughout the reach and concentrated along river banks, in deep pools, riffles, eddies, or near cover. We attempted to capture all fishes that were stunned. Electrofishing samples collected in 2002 in summer (both rafts) and autumn (one raft) were potentially biased downward compared to samples collected in 2003 and 2004 because the anode and cathode plugs were switched. This caused the relatively small surface area dropper sphere in the front of the raft to become the cathode and the relatively large surface area cables positioned mid-raft to become the anodes. The result was fewer fish were attracted to the anode and sometimes came up at the side of the raft rather than the front, which resulted in less efficient sampling. However, mean capture rate of all species combined over the study period was only slightly higher when the 2002 data was excluded (2.65 fish/h average per all species, 95% CI =

2.43 to 2.89, compared to 2.42 fish/h, 95% CI = 2.24 to 2.61 with 2002 data included). Also, of nine species most often captured by electrofishing, five increased in mean abundance when 2002 data were excluded and four decreased. Since inclusion of that data appeared to have little effect on overall abundance values, we decided to retain 2002 data in all analyses.

Trammel net sampling was mostly confined to Whirlpool Canyon, because that reach contained deep eddy habitat most likely to support humpback chub. A few trammel net samples were collected in Mitten Park and Lodore Canyon. Multi-filament trammel nets (23 m long x 1.8 m deep; 25-cm outer mesh, 2.5-cm inner mesh) were set in late summer or autumn when water temperatures were relatively low so that fish mortality would be minimized. In autumn 2003 we conducted three consecutive sampling trips (passes) in an attempt to obtain an abundance estimate for chubs in Whirlpool Canyon. The main sampling sites for trammel nets were in upper Whirlpool Canyon just upstream and downstream of Rain Camp (river kilometer (RK) 550.8), at a large eddy pool about 1 km upstream of Jones Hole Creek (RK 546.4), and at Compromise Camp (RK 544). Trammel nets were usually fished during crepuscular and nighttime hours, and lesser amounts in day time, and were set in low velocity habitats and in the separation zone of the main channel current and eddy pools (eddy lines). The number of nets set (up to six) was contingent on habitat availability and accessibility. Nets were cleared of fish every 2 h. Fish captured by electrofishing and trammel nets were measured (total length [TL]) and sex and reproductive condition was noted, if discernable. Large samples of small-bodied fish captured by electrofishing were counted. In addition, endangered fishes and roundtail chubs were individually weighed and scanned for a passive interrogated transponder (PIT) tag. If none was detected, a PIT tag was implanted and the fish was released. Special care was taken to identify potential catostomid hybrids. We identified these based on morphological characteristics of the mouth and body that were intermediate between putative parental types and on intermediate squamation patterns and scale counts in the lateral series (Hubbs and Miller 1953). Vouchers of some specimens, including catostomid hybrids, were preserved in 10% formalin and are housed at the Larval Fish Laboratory, Colorado State University.

Stream flow and water temperature data collection and presentation.—Most temperature data were from lower Lodore Canyon and Mitten Park (pers. comm., G. Smith, U. S. Fish and Wildlife Service, Denver, Colorado). Flow data were from U. S. Geological Survey gauges. First, we describe the major changes in flow and temperature regimes in Lodore and Whirlpool canyons as a result of construction and operation of Flaming Gorge Dam. We characterized flow differences in the historical period (Lodore Canyon, Greendale gauge, # 09234500, 1951 to 1962; Whirlpool Canyon, Jensen gauge # 09261000, 1946 to 1962), a post-impoundment period (1962 to 1991), a post-Biological Opinion period (1992-1996), and a recent (1997 to 2004) period which was thought to represent conditions that affected the fish communities during this investigation. We used the Jensen, Utah gauge to characterize Whirlpool Canyon and Island-Rainbow Park flows because the few small tributaries present between Whirlpool Canyon and the downstream gauge will not affect flow estimates appreciably.

Fish sample data analysis and presentation.—Composition of the fish community in the study area (Fig. 1) was calculated for samples collected in the 2002 to 2004 period for each gear type. Longitudinal and temporal shifts in community composition were characterized by partitioning samples into reaches both for small-bodied fishes captured primarily by seining and for large-bodied fishes captured primarily by electrofishing in the periods 1994 to 1996 and 2002 to 2004. Samples were from eight contiguous Green River reaches: a Browns Park reach, four 8-RK Lodore Canyon reaches, upper and lower Whirlpool Canyon reaches (9 and 8 RK) and the Island-Rainbow Park reach (10 RK); electrofishing samples were collected only in Lodore and Whirlpool canyons.

Change in numerically dominant native and non-native fishes in reaches was compared among study periods 1994 to 1996 and 2002 to 2004 and reaches to facilitate detection of longitudinal shifts in community composition. We defined relative abundance as the percent composition that a particular species represented in a sample. For seine samples, catch-per-unit-effort (CPUE) was number of fish of a particular species divided by area sampled per seine haul (density). For electrofishing samples, CPUE was number of fish captured per amount of time sampled. For trammel net samples, CPUE was fish captured per net hr. We did not measure seine sample area during the 1994 to 1996 period so density comparisons were not possible between the two time periods. Statistical analyses were possible only for electrofishing CPUE data in Lodore Canyon between the two periods because there were no 1994 to 1996 data for Whirlpool Canyon. We plotted changes in individual species abundance in each Lodore Canyon reach between the two periods, but did not conduct statistical analyses of reach or annual differences within studies; we were mainly interested in broader scale abundance patterns between the two periods. We also compared size-structures of selected taxa sampled in the two periods in each of the four Lodore Canyon reaches for native fishes and a few potentially problematic non-native species.

We then analyzed effects of season, reach, year, and turbidity on electrofishing and seine data to document study area scale variation in species richness and fish abundance. We did not include flow level as an analysis variable because it varied little during our sampling in the base flow period. Water temperature was not included because these differences should be reflected as the season class variable, warm in summer and cool in autumn. Statistical analyses were conducted with a general linear model (GLM) approach (Proc Genmod, in SAS), using a negative binomial data distribution for counts of fish in a sample after adjustment for effort (fish captured per unit effort), and a log link. A main advantage of this approach over traditional analysis of variance (ANOVA) applications that assume normal distributions is that the negative binomial data distribution (number of fish captured per sampling effort) allows for a large number of zero observations in the data set without transformations (or back-transformations). Normality assumptions using transformations for zero-rich fishery data are rarely met in traditional ANOVA applications and, as a result of inflated variance estimates, significance tests among groups are usually conservative (pers. comm., G. White, Colorado State University). This approach also allowed inclusion of categorical variables such as turbidity in analyses to examine catch rate variability. Significance tests for this approach used a chi-square test rather than the more familiar *F*-test used in most ANOVA applications. Differences among capture rates were tested with a least squares means procedure. We then plotted density data for native or abundant non-native species captured in seine samples to delineate variation at reach, season, and year scales for 2002 to 2004 data. Capture rates of common species in trammel net samples were calculated as fish per net h, where a net h is one 22.9 m-long trammel net set for one h. We also compared day-time (nets set between 0800 and 1700 h) and night-time (nets set after 1700 h or before 0800 h the following day) capture rates of chubs. Species composition and abundance in drift samples were compared across years. Influence of disturbance events was determined by examining drift rates associated with storm-induced turbidity or flow events. Species accounts for rare or endangered native fishes and selected non-native fishes present details that were not previously described.

Historical changes in fish communities were also assessed using data for the regulated reach of the Green River upstream of the Yampa River. We compared composition and reproductive status of each species for time periods corresponding to just prior to Flaming Gorge Dam closure (historical), just after reservoir filling and commencement of normal operations in 1967, after installation of a temperature control device via penstock modifications in 1978, after discharge re-regulation in 1992 (1994 to 1996 sampling), and for the 2002 to 2004 period (this study). Data gathered from historical sources (Bosley 1960, Gaufin et al. 1960, Banks 1964, Smith 1966, Vanicek et al. 1970, Holden and Crist 1981, Bestgen and Crist 2000) were used to assess reproductive status in the reach based on presence of young fish or by assumptions about the probability of reproduction given water temperature regimes in the reach and presence of adults. Reproductive status of fishes from 1967 to 1978 was inferred from data gathered in 1964 and 1966 by Vanicek et al. (1970) and data collected in 1978 (Holden and Crist 1981) prior to temperature modifications. Data describing the reproductive status of fishes in 1965 was not used because flows from Flaming Gorge Dam were low and warm during reservoir-filling and were not typical of conditions from 1967 to 1978 (Vanicek et al. 1970).

Historical changes in species composition and relative abundance of fishes specific to Lodore Canyon were assessed by comparing data collected in three periods: 1978 to1980 (Holden and Crist 1981), 1994 to 1996 (Bestgen and Crist 2000), and 2002 to 2004 (this study). Holden and Crist (1981) sampled fishes at fixed sites (not continuous as in the latter two periods) including a 1.9-km-long upper Lodore Canyon site near Wade and Curtis Campground (RK

581), and a 1.6-km-long lower Lodore Canyon site near Alcove Brook (RK 561). To make fish composition data from their fixed sites comparable to more recent data (Bestgen and Crist 2000, this study), we used a subset of our samples collected in a 4.8 RK reach encompassing their fixed stations. The more recent samples were used only if the sampling segment ended within the reach; samples started within the reach that ended outside of it were not used. We combined seine and electrofishing data in each of the 1994 to 1996 and the 2002 to 2004 periods for comparisons to be consistent with Holden and Crist (1981). Changes in species composition among the three periods (1978 to 1980, 1994 to 1996, 2002 to 2004) were assessed by comparing the number and relative abundance of native and introduced species captured in each period. Differences in habitat availability, collecting gear, and sampling effort may affect assessments of faunal differences among the three periods. Therefore, our qualitative assessments of temporal change for those reaches were conservative, but supported by concurrently considering broader spatial scale patterns of changes in fish distribution and abundance throughout Lodore Canyon.

RESULTS AND DISCUSSION

Effects of Flaming Gorge Dam on Green River discharge and temperature patterns, Lodore Canyon.—Four major dam-related events affected physical habitat and the fish community of the upper Green River: dam installation in 1962, penstock modification in 1978, flow re-regulation beginning in 1992, and higher peak flows, lower base flows, and drought events since 1997. Prior to construction of Flaming Gorge Dam, discharge patterns of the Green River exhibited high spring peaks followed by lower, stable base flows in other seasons, particularly winter (Fig. 2). Water temperature ranged from a low of 0°C in winter to over 22°C in summer (Fig. 3) and turbidity increased seasonally from inputs of fine inorganic silts (Vanicek et al. 1970). Immediate dam effects included inundation of riverine habitat in the reservoir zone. Downstream, naturally variable and relatively high spring discharge maxima were reduced (Fig. 4). Power plant operations increased base flow levels and daily discharge fluctuations.

Naturally turbid water was cleared when virtually all sediment was trapped behind the dam. Water drawn from the hypolimnion of the reservoir reduced temperature of summer releases to about 6°C after reservoir filling was completed in 1967. Downstream warming occurred but mean monthly summer water temperatures reached a maximum of about 14°C in the post-dam period in Lodore Canyon compared with > 20°C in the pre-impoundment period (Fig. 3 in Bestgen and Crist 2000). Post-dam winter water temperatures in the tailwater were increased compared to pre-impoundment conditions from near 0°C to about 4°C.

Cold tailwater releases begun after 1967 dramatically reduced trout growth in the fishery that established after dam closure, which prompted the second major Green River change due to Flaming Gorge Dam (Holden and Crist 1981). A multi-level penstock was installed and became operational in June 1978 to allow water to be drawn from warmer upper layers of the reservoir in summer. Temperature of multi-level penstock releases during summer operations were targeted at about 13°C, a level thought to maximize growth of rainbow trout *Oncorhynchus mykiss* and maximize downstream extent of the tailwater fishery. Effects of penstock modifications on native fishes in reaches further downstream were assessed from 1978 to 1980 by Holden and Crist (1981). Penstock modifications raised water temperature in the Green River upstream of the Yampa River, but summer maxima rarely exceeded 17°C in the period 1978 to 1991 (Fig. 3 in Bestgen and Crist 2000).

The third major dam-related operation event occurred in 1992 in response to a Biological Opinion on operation of Flaming Gorge Dam (Tyus and Karp 1991, U. S. Fish and Wildlife Service 1992). Spring release levels from the dam were increased in an attempt to simulate a more natural runoff pattern and enhance floodplain inundation downstream of Jensen, Utah (Fig. 2). The spring through autumn flows implemented in 1992 were designed to enhance physical habitat in the Green River downstream of the Yampa River confluence; habitat enhancement in the regulated Green River reach upstream of the Yampa River was not a priority because few endangered fishes were thought to occur there. Releases up to maximum power plant levels (130 m^3/sec, Fig. 4) were made for up to six weeks in spring around the time when unregulated

Yampa River discharge peaked. During summer, release levels were designed to maximize nursery habitat for Colorado pikeminnow near and downstream of Jensen, Utah (Tyus and Karp 1991). To achieve the desired stable base flow of 51 m^3/sec ($\pm$ 12.5%, Pucherelli et al. 1990, Tyus and Karp 1991, U. S. Fish and Wildlife Service 1992), releases were usually decreased to as low as 22.3 m^3/sec soon after prescribed high spring flows ended because Yampa River discharge was usually high. Dam releases were increased later in summer after Yampa River discharge declined and when Green River discharge at Jensen fell below target levels.

Water temperatures in the Green River upstream of the Yampa River in the period 1992 to 1996 showed a slightly warmer summer pattern than for the previous 1978 to 1991 period as summer maxima typically peaked at about 18°C or less for a short period (Bestgen and Crist 2000). Summer discharge manipulations altered thermal regimes in the regulated reach because flow level and water temperature were inversely correlated in the Green River downstream of Flaming Gorge Dam (Vanicek et al. 1970, Bestgen and Crist 2000). This is because reservoir releases that are cold relative to air temperature progressively warm via solar insolation downstream from the dam, but warm faster in upstream reaches if flows are lower and slower. Thus, under low drought flows, relatively warm water can be expected well upstream into Browns Park if air temperatures are warm. Dynamics of flow and water temperatures and regression models to predict downstream warming were discussed in detail in Bestgen and Crist (2000) and Muth et al (2000). Because flows typically increased from Flaming Gorge Dam later in summer during 1992 to 1996, maximum temperatures were sometimes achieved in late June but often declined by mid July.

Flow and temperature regimes in the Green River downstream of Flaming Gorge Dam since 1997 followed elements of both the 1992 Biological Opinion and newer proposed recommendations. Higher spring releases were realized in 1997 and 1999 (Fig. 5), but spring peaks during study years 2002 to 2004 were at or below power plant capacity (about 130 m^3/sec, Fig. 4). Compared to earlier post-regulation periods, in 1997 to 2004, and particularly for 2002 to 2004 study period (and in 2000 and 2001), base flows (August to February) were lower, stable, and more closely approximated the historical hydrograph (Fig. 6, Table 1). Flows that resulted from the 1992 Opinion recommendations were more variable. For example, summer base flows more than doubled from about 23 m^3/sec in June to nearly 50 m^3/sec in August in the low flow year 1994.

In all periods, regulated flows showed large differences in hydrograph shape. Historical flows had a longer duration peak, increased to peak and declined from peak more gradually, and base flows were lower. Additionally, timing of the peak from the regulated Green River upstream of the Yampa River exhibited peak flows several weeks earlier than was historically seen, peaking now in late May rather than mid or late June (Fig. 2), as dictated by the 1992 Biological Opinion.

As a result of relatively low baseflows in 1997 to 2004, summer water temperatures in Lodore Canyon increased to levels likely not observed since Flaming Gorge Reservoir filled in 1967 (Fig. 3). Summer water temperatures were particularly warm in the Green River upstream of the Yampa River in the 2002 to 2004 study period and matched the summer maxima observed in the limited 1957 to 1962 historical period (Fig. 7). Water temperatures were particularly warm during summer 2002, when daytime highs reached 25°C in late July. Only on four days in the 2002 to 2004 period were water temperatures in the Green River more than 5°C different than the Yampa River, and those were all in 2002 in late August (Table 2).

Effects of Flaming Gorge Dam on Green River discharge and temperature patterns, Whirlpool Canyon.—Annual peak flows in the pre-impoundment period (1946-1962) showed a typical dome-shaped hydrograph, with low late summer, autumn, winter, and early spring base flows, rising to a peak in later spring and early summer (Fig. 8). Post-Flaming Gorge peak flows in the 1992 to 1996 and 1997 to 2004 periods were lower than historical, from a standpoint of mean daily annual flow, peak flow, and maximum May-June average flows (Table 1). Base flows in the period August to February were typically lower in the pre-impoundment period than post. Mean July-August summer base flows were lower in 1992 to 1996 than historical flows, and mean July-August base flows in the 1997 to 2004 period were the lowest of all three periods.

Peak flows for the Green River in Whirlpool Canyon during study years 2002 and 2004 showed a lower and earlier peak than the historical average condition, while the 2003 peak flow exceeded the average historical condition (1946 to 1962) and preceded timing of the peak by just a few days (Fig. 9). Base flows in the 2002 to 2004 period matched the historical average pattern reasonably closely.

Temperatures in Whirlpool Canyon (Mitten Park station, just upstream of Whirlpool Canyon), in the 1992 to 1996 and 1997 to 2004 periods, were relatively similar (Fig. 10). The primary difference was a period from mid-May to mid-July, when water temperatures were up to 4°C cooler in the 1992 to 1996 period compared to 1997 to 2004. This was due to relatively high and cold flows observed in 1993 and 1995. Water temperatures during study years 2002 to 2004 were similar to, or higher than, average conditions observed for the 1997 to 2004 period (Fig. 11).

Thus, three actions subsequent to dam closure in 1962 have incrementally modified flows and temperatures in the Green River upstream and downstream of the Yampa River to a state that more closely resembles historical, unregulated conditions. The flow and thermal regimes observed in 2002 to 2004 likely will reflect those that would be realized in low or moderately-low flow years, under new Flaming Gorge Dam flow and temperature recommendations (Muth et al. 2000). Therefore, studies conducted in the three years from 2002 to 2004 offered an opportunity to assess fish community response to flows that were at the lower end of the recommendations spectrum. A more complete assessment of fish community response will be possible only with additional years of sampling conducted in average, moderately-high, or high flow years.

During 2002 to 2004, warmer temperatures in Lodore Canyon resulted in minimal water temperature differences in the Green River upstream of the Yampa River and the Yampa River, which was a goal of flow and temperature recommendations of Muth et al. (2000). Minimizing temperature differences may reduce the likelihood of temperature shock when Colorado pikeminnow larvae are carried from warm Yampa River flows into the colder Green River (Berry 1988, Tyus 1991). This problem may be more pronounced when Yampa River flows are very low and warm and flows in the regulated Green River are higher and colder.

Composition of the fish community in Browns Park, Lodore Canyon, Whirlpool Canyon, and Island-Rainbow Park reaches 2002-2004.—A total of 209,466 specimens representing 9 native and 19 non-native species and 7 hybrid combinations were captured by all sampling gears during 2002 to 2004 (Table 3). Flannelmouth sucker *Catostomus latipinnis* was the most common native fish captured by all sampling gears, followed by bluehead sucker *Catostomus discobolus*, mountain whitefish *Prosopium williamsoni*, speckled dace *Rhinichthys osculus*, roundtail chub, mottled sculpin *Cottus bairdi*, Colorado pikeminnow, bonytail *Gila elegans*, and humpback chub. No razorback suckers or mountain suckers were captured during this study; both taxa were previously collected (Holden and Crist 1981). Hybrids of razorback sucker and other sucker species were captured and will be discussed later in this report.

The most abundant non-native fish species captured by all gears was red shiner *Cyprinella lutrensis*, followed by sand shiner *Notropis stramineus*, fathead minnow *Pimephales promelas*, and redside shiner *Richardsonius balteatus*, all small-bodied fishes. The most abundant large-bodied non-native fishes included white sucker *Catostomus commersoni* (mostly small-bodied specimens captured), common carp *Cyprinus carpio*, channel catfish *Ictalurus punctatus*, brown trout *Salmo trutta*, and smallmouth bass *Micropterus dolomieu*. Of taxa considered predaceous or potentially so, channel catfish, brown trout, smallmouth bass, and green sunfish *Lepomis cyanellus* were most abundant.

Active sampling gears (seining and electrofishing) captured the most fish. Drift net samples were dominated by larvae or small juveniles (usually 5 to 25 mm TL), seine samples by small-bodied fishes (usually 20 to 75 mm TL), and trammel nets and electrofishing captured mostly large-bodied fishes (usually 150 mm TL or greater).

Seining yielded eight native and 17 non-native species, and four hybrids. Flannelmouth sucker, bluehead sucker, mountain whitefish, and speckled dace, in descending order of abundance, were the most common native fishes collected. Red shiner, sand shiner, fathead minnow, and redside shiner, in descending order of abundance, were the most abundant non-native taxa collected. Seining also captured the highest proportion of non-native fishes at 92.4%, reflecting the dominance of small-bodied nonnative fishes in shallow, low-velocity near-shore environments.

Electrofishing captured nine native fishes, 16 non-natives, and seven hybrids. The two most abundant native fish species were flannelmouth sucker and bluehead sucker, and the most abundant non-native taxa were brown trout, common carp, and channel catfish. The percent of native and non-native fishes captured by electrofishing was about equal, and electrofishing captured a higher percentage of hybrids than any other gear. Trammel net samples yielded six native and eight non-native species, and three hybrids, and captured the highest proportion of native fish (about 2/3) of any sampling gear. The most common native fishes captured were flannelmouth sucker and roundtail chub, while the overwhelming majority of non-native fish was channel catfish. Five native fishes, eight non-natives, and one hybrid were captured in drift nets. The most abundant native fish in drift net samples were flannelmouth sucker and bluehead sucker, and non-native fishes were dominated by red shiner and sand shiner.

When all sampling gears were combined, native fishes comprised only 10.3% of total catch and non-natives were 89.3%; the remaining 0.4% were hybrids. High non-native fish abundance in seine samples reflected the dominance of small-bodied red shiner, sand shiner, and fathead minnow in the low-velocity habitats where most samples were collected. Those taxa dominated the fish community in low-velocity habitats in other areas of the Colorado River Basin (Propst and Gido 2004). For example, in backwaters of the middle and lower Green River, Haines and Tyus (1990) found that those species comprised about 90% or more of the fish community. Drift net samples were also dominated by non-native fishes but to a lesser extent than in seine samples.

Although all gear types were not used with equal effort in all reaches or years, the differences in community composition that each sampling method yielded were striking. This points to the need to employ a wide variety of gear types when attempting to understand community composition and abundance of fishes in the Green River.

Comparison of fish community composition across the longitudinal gradient in Lodore and Whirlpool canyons between 1994 to 1996 and 2002 to 2004 periods, electrofishing samples.—Electrofishing of mostly large-bodied fishes in Lodore and Whirlpool canyons in 2002 to 2004 captured relatively fewer species upstream than downstream, taxa in upstream reaches were cold-water or cool-water tolerant, and downstream were tolerant of cool or warm water (Tables 4 and 5). A total of 25 fishes were detected by electrofishing; species richness in electrofishing samples in the LD1 to LD4 reaches in Lodore Canyon and the WH1 and WH2 reaches in Whirlpool Canyon was 15, 13, 17, 19, 21, and 20, respectively. Relatively low species richness in reach LD2 was due to absence of roundtail chub and rare non-native fishes that were present downstream.

In the upstream LD1 reach in 1994 to 1996, seven native and eight introduced species and four hybrids were detected by electrofishing. In decreasing order of abundance, flannelmouth sucker, brown trout, common carp, bluehead sucker, and redside shiner were the five most common species collected, and represented 72% of the total assemblage. Native species were 51% of the total. In LD1 in the 2002 to 2004 period, seven native and nine introduced species and four hybrid combinations were detected by electrofishing. Brown trout, flannelmouth sucker, white sucker, common carp, and channel catfish were the five most common species, and made up 72% of the fish community. Native fishes were 32% of the total. Increased abundance of channel catfish in the 2002 to 2004 was notable.

In reach LD2 in 1994 to 1996, seven native and seven introduced species and four hybrids were captured by electrofishing. Flannelmouth sucker, brown trout, bluehead sucker, mountain whitefish, and common carp were the most common species collected, and represented 87% of all fish captured. Native species represented 62% of the fish community by number. In LD2 in the 2002 to 2004 period, six native and seven introduced species and three hybrids were captured by electrofishing. Brown trout, flannelmouth sucker, bluehead sucker, white sucker, and common carp were the five most common species, and made up 86% of the fish community. Native fishes were 42% of the total.

In LD3 in 1994 to 1996, seven native and eight introduced species and four hybrid suckers were captured by electrofishing. Flannelmouth sucker, brown trout, bluehead sucker, common carp, and mountain whitefish were the most common species collected, and represented 88% of the total assemblage. Native species represented 56% of all fish captured, by number. The most upstream captures of red shiner in Lodore Canyon were in this reach. In LD3 in the 2002 to 2004 period, seven native and ten introduced species and three hybrids were captured by electrofishing. Brown trout, flannelmouth sucker, bluehead sucker, common carp, and channel catfish were the five most common species, and made up 81% of the fish community. Native fishes were 39% of the total, by number.

In LD4 in 1994 to 1996, eight native and 11 introduced species and five hybrids were captured by electrofishing. Flannelmouth sucker, bluehead sucker, common carp, brown trout, and red shiner were the most numerically abundant species collected, and represented 87% of the total assemblage. Native species represented 63% of all fishes captured, by number. The only razorback sucker captured in the 1994 to 1996 survey came from LD4. First captures of northern pike *Esox lucius* and higher abundance of channel catfish were also notable in the 1994 to 1996 period. In LD4 in the 2002 to 2004 period, seven native and twelve introduced species and six hybrids were captured by electrofishing. Flannelmouth sucker, common carp, channel catfish, bluehead sucker were the four most common species, smallmouth bass and brown trout were tied for the fifth most common species, and together made up 82% of the fish community. Capture of relatively large numbers of smallmouth bass and the relatively low numbers of brown trout were notable. Native fishes were 42% of the total, by number.

In reach WH1 in 2002 to 2004, nine native and 12 introduced species and five hybrids were captured by electrofishing. Flannelmouth sucker, bluehead sucker, channel catfish, smallmouth bass, and common carp were the five most common species collected and represented 86% of the total assemblage. Native species represented 57% of all fishes captured.

In reach WH2 in 2002 to 2004, nine native and 12 introduced species and five hybrids were captured by electrofishing. Bluehead sucker, flannelmouth sucker, smallmouth bass, channel catfish, and common carp were the five most common species collected in descending order of abundance, and represented 86% of the total assemblage. Native species represented 55% of all fishes captured, by number.

The longitudinal pattern of increasing downstream species richness in electrofishing samples was due mostly to faster addition rate of warm-water tolerant species in a downstream direction as cold-water or cool-water species dropped out. Over all Lodore Canyon reaches, percent native fish in electrofishing samples declined from 63% in the 1994 to 1996 period to 39% in the 2002 to 2004 period. Native fish composition in Whirlpool Canyon reaches in 2002 to 2004 was slightly higher at about 56%; no comparative 1994 to 1996 data are available. Major differences in Lodore Canyon fish community composition in 2002 to 2004 electrofishing samples compared to 1994 to 1996 were increased distribution and abundance of channel catfish and smallmouth bass, and brown trout supplanted flannelmouth sucker as the most abundant species in reaches LD1 to LD3.

Comparison of fish community composition across the longitudinal gradient in Browns Park, Lodore and Whirlpool canyons, and Island-Rainbow Park between the 1994 to 1996 and 2002 to 2004 periods, seine samples.—Similar to electrofishing sample data, the fish community of the Green River, as measured by seine sampling suggested that upstream reaches supported relatively fewer species, and most were cold- or cool-water tolerant. In comparison, downstream reaches supported a community with higher species richness that was composed mostly of cool- or warm-water tolerant taxa (Tables 6 and 7). A total of 25 fish species were detected by seine sampling; species richness in seine samples in the Browns Park, LD1 to LD4, WH1 and WH2, and Island-Rainbow Park reaches was 14, 15, 15, 18, 20, 20, 14, and 18, respectively. The absence of six species in WH2 was due to absence of rare taxa: three native and three non-native.

In the upstream Browns Park reach in 1994 to 1996, a total of five native and six introduced species and three hybrid suckers was captured. In descending order of abundance, fathead minnow, bluehead sucker, white sucker, flannelmouth sucker, and redside shiner made up 96% of the fish community. Native species represented 33% of the fish community, by number. Mountain whitefish and brown trout were also captured, as was a single juvenile sand shiner. In Browns Park in 2002 to 2004, a total of six native and eight introduced species and three sucker hybrids was detected. Fathead minnow, redside shiner, white sucker, mountain whitefish, and flannelmouth sucker made up 94% of the fish community. Native species represented 21% of the fish community. Notable captures in this reach included a near-dead hatchery-origin bonytail that was stocked in autumn 2002 six days before sampling took place and 0.4 RK upstream. We also documented upstream dispersal of red shiner in the Green River between April and September 2003 sampling periods, which is discussed more fully in a later section.

In the 1994 to 1996 period, the fish assemblage of Lodore Canyon differed from that found in upstream Browns Park and also varied along the longitudinal gradient of reaches LD1 to LD4. In LD1, four native and five introduced species and three hybrids were captured by seining. Flannelmouth sucker, bluehead sucker, white sucker, redside shiner and fathead minnow were the most common species collected, representing 94% of the total assemblage structure and native fishes were 70.9% of those captured. Notable in reach LD1 and downstream reaches in the 2002 to 2004 period were severe reductions in the abundance of native fishes and increased diversity and abundance of non-native fishes, particularly red shiner and sand shiner. In LD1 in the 2002 to 2004 period, six native and ten introduced species and two hybrids were detected by seining. Fathead minnow, redside shiner, flannelmouth sucker, white sucker, and red shiner were the most common species collected, representing 94% of the total assemblage and numerically, native fishes were 19.4% of those captured.

In LD2 in 1994 to 1996, seining captured five native and 11 introduced species, and two hybrids. Speckled dace, flannelmouth sucker, bluehead sucker, white sucker, and redside shiner were the most common species collected, representing 98% of the total fish community and native species represented 94% of all fish captured. Notable finds were roundtail chub and smallmouth bass. In LD2 in the 2002 to 2004 period, six native and ten introduced species and two hybrids were detected by seining. The fish community of LD2 showed a distinct “break” or transition from an upstream cool water assemblage to a more warm water one downstream, as red shiner were present in relatively large numbers. There was no corresponding abrupt shift in water temperature. In 1994 to 1996 this break was in downstream reach LD4. Fathead minnow, white sucker, red shiner, flannelmouth sucker, and green sunfish were the most common species collected, representing 80% of the total assemblage structure and numerically, native fishes were 21% of those captured.

In LD3 in 1994 to 1996, four native and three introduced species were captured by seining. Flannelmouth sucker, speckled dace, bluehead sucker, white sucker, and mottled sculpin were the most common species collected, representing 98.5% of the fish community and native species represented 94% of fish captured. In LD3 in the 2002 to 2004 period, six native and 13 introduced species and two hybrids were captured by seining. Red shiner, sand shiner, fathead minnow, speckled dace, and flannelmouth sucker were the most common species collected, representing 91% of the total assemblage; native fishes were 11% of those captured, by number.

In LD4 in 1994 to 1996, five native and eight introduced species and two hybrids were captured by seining. The fish community in LD4 during the 1994 to 1996 time period shifted from one dominated mostly by cold-water to cool-water tolerant species upstream to one dominated by mostly warm-water tolerant species downstream. Red shiner, fathead minnow, redside shiner, speckled dace, and flannelmouth sucker were the most common species collected, representing 88% of the total fish community and only 16.8% of all fishes captured were native. The presence and abundance of red shiner and sand shiner was notable. In LD4 in the 2002 to 2004 period, six native and 15 introduced species and two hybrids were collected by seining. Red shiner, sand shiner, fathead minnow, white sucker, and flannelmouth sucker were the most common species collected, representing 96% of the assemblage; numerically, native fishes were 3% of those captured.

In reach 1 of Whirlpool Canyon (WH1) in the 2002 to 2004 period, seven native and 15 introduced species and three hybrids were captured by seining. Red shiner, sand shiner, redside shiner, fathead minnow, and flannelmouth sucker were the most common species collected, representing 96% of the assemblage; native fishes were 4% of those captured. In reach WH2 in the 2002 to 2004 period, four native and 11 introduced species and two hybrids were captured by seining. Red shiner, sand shiner, fathead minnow, flannelmouth sucker, and white sucker were the most common species collected, representing 98% of the assemblage; numerically, native fishes were 3% of those captured.

In the Island-Rainbow Park (IRP) reach of the Green River in the period 1994 to 1996, seven native and 11 introduced species and two hybrids were captured by seining. Non-native taxa represented 98.2% of all fishes captured; most were tolerant of warm water. Red shiner was the numerically dominant species and, when combined with fathead minnow, sand shiner, redside shiner, and flannelmouth sucker, comprised 97 % of the fish community. Relatively unusual captures included a single juvenile Colorado pikeminnow, about 150 mm TL, a juvenile northern pike (112-mm TL), largemouth bass *Micropterus salmoides*, and green sunfish. In the IRP reach of the Green River in the 2002 to 2004 period, seven native and 12 introduced species and two hybrids were captured by seining. Red shiner, sand shiner, fathead minnow, flannelmouth sucker, and white sucker were the most common species collected, representing 96% of the assemblage; numerically, native fishes were 4% of those captured. Both age-0 and age 1+ Colorado pikeminnow were captured in backwaters.

Similar to electrofishing samples, the longitudinal pattern of increasing downstream species richness in seine samples was due mostly to addition of more warm-water tolerant species in a downstream direction as a few cold-water or cool-water species dropped out. Dramatic declines in native fish abundance were noted in nearly every reach between the periods 1994 to 1996 and 2002 to 2004. In Browns Park, native fish composition declined from 33% to 21% and over all Lodore Canyon reaches, percent native fish in seine samples declined from 69% in the 1994 to 1996 period to 14% in the 2002 to 2004 period. Increased abundance of red shiner, sand shiner, and fathead minnow and reduced abundance of speckled dace and native suckers were mainly responsible for changes in fish community composition. Proportion of native fish in samples downstream of Lodore Canyon was low.

Spatial and temporal patterns of distribution, abundance, and size-structure of selected species, comparison of 1994 to 1996 and 2002 to 2004 periods.—This section presents overall patterns of distribution, abundance, and size structure for selected species and examines changes between study periods 1994 to 1996 and 2002 to 2004. We present data for native fishes and selected, potentially problematic, non-native species.

Native fishes.—Mountain whitefish was found throughout the study area but was most common in upstream Browns Park, was moderately abundant in Lodore Canyon reaches, and was comparatively uncommon in Whirlpool Canyon and Island-Rainbow Park. Relative abundance and CPUE of mountain whitefish in Lodore Canyon electrofishing samples in the 2002 to 2004 period were dramatically lower than in the 1994 to 1996 period (Fig. 12). Mountain whitefish now comprises 2% or less of the fish community in all reaches and analysis of CPUE data indicated a significant decline (Table 8). Abundance of mountain whitefish in seine samples was greater in nearly all reaches in the recent period than in 1994 to 1996, and particularly so in Browns Park and LD3 reaches (Fig. 12). This may be due to collection of more seine samples in spring when young mountain whitefish were more abundant. Relative abundance and CPUE data for mountain whitefish and other species generally tracked each other reasonably closely. This suggested that each metric was a surrogate for the other and that changes in relative abundance between the 1994 to 1996 and 2002 to 2004 periods may reflect actual changes in density. The abundance of small mountain whitefish in Browns Park seine samples and their low abundance elsewhere suggested that most fish in downstream reaches were derived from upstream.

Size-structure of mountain whitefish from electrofishing data showed that the majority of fish captured were in the 101 to 200-mm TL size class in each of the 1994 to 1996 and 2002 to 2004 periods (Fig. 13). More large fish were found in the 2002 to 2004 period than earlier, especially in the upper portion of Lodore Canyon. We captured only small mountain whitefish in Island-Rainbow Park. Most of those were captured in spring, with few or none in summer or autumn. Based on size of fish captured in seine samples and downstream absence of adults, autumn-spawning mountain whitefish hatched prior to spring runoff and may be dispersed downstream by higher spring flows.

Adult Colorado pikeminnow were found in all Lodore Canyon reaches in both time periods and Whirlpool Canyon and Island-Rainbow Park in 2002 to 2004. Relative abundance and CPUE for Colorado pikeminnow in electrofishing samples declined in 2002 to 2004 compared with 1994 to 1996, particularly in reaches LD3 and 4 (Fig. 14); CPUE data were not significantly different. In the recent period, Colorado pikeminnow relative abundance was highest in lower Whirlpool Canyon, but was similar to or less than Colorado pikeminnow abundance in the lower reaches of Lodore Canyon where it was most common in 1994 to 1996.

Colorado pikeminnow in Lodore Canyon in each study period were a mix of sizes including a single fish in each period in the 351 to 400-mm TL size class (Fig. 15). Modal size in the 1994 to 1996 period was larger than in the 2002 to 2004 period. Modal size of pikeminnow in Whirlpool Canyon was similar to that in Lodore Canyon in the 2002 to 2004 period.

Roundtail chub was found in every reach of the study area in 2002 to 2004 except Browns Park and showed increased downstream abundance in both seine and electrofishing samples in a downstream direction. However, roundtail chub abundance in electrofishing samples declined in 2002 to 2004 compared to the 1994 to 1996 period (Fig. 16); catch rate was reduced by more than 50%, a significant decline (Table 8). Roundtail chub abundance was higher in upper Whirlpool Canyon compared to other reaches but overall, represented a relatively small portion of the fish community there. Roundtail chub abundance in seine samples was low in all Lodore Canyon reaches in 2002 to 2004 and declined relative to samples collected in 1994 to 1996. Similar to adult chub captured by electrofishing, small roundtail chub captured in seine samples were most abundant in upper Whirlpool Canyon and were present in lower Whirlpool Canyon and Island-Rainbow Park, but overall represented a relatively small proportion of samples.

In general, only large roundtail chub (usually > 300 mm TL) were present in Lodore Canyon. A mix of sizes was present in Whirlpool Canyon samples (Fig. 17) and included nearly all adult roundtail chubs < 300 mm TL captured in this study.

Speckled dace was distributed throughout the study area but was generally uncommon in the 2002 to 2004 period in most reaches (Fig. 18). Speckled dace abundance in electrofishing samples declined from the 1994 to 1996 period to the 2002 to 2004 period; CPUE data showed a significant decline (Table 8). Speckled dace relative abundance in seine samples was low and similar in the two study periods in Browns Park and LD1 reaches, but declined rather dramatically in Lodore Canyon reaches LD2 to LD4 in 2002 to 2004. Speckled dace was moderately abundant in Whirlpool Canyon and Island-Rainbow Park reaches in 2002 to 2004, but was uncommon in Island-Rainbow Park in 1994 to 1996.

Flannelmouth sucker was found in all study reaches. Abundance of adults increased slightly in a downstream direction while young fish in seine samples declined in abundance downstream (Fig. 19). Flannelmouth sucker relative abundance and CPUE in electrofishing samples declined by nearly 40% in all Lodore Canyon reaches in 2002 to 2004, compared to the 1994 to 1996 period; CPUE data demonstrated a significant decline. Flannelmouth sucker typically represented 20 to 25% of electrofishing samples in all Lodore and Whirlpool Canyon reaches. Relative abundance of flannelmouth sucker in seine samples was dramatically reduced in Browns Park and Lodore Canyon reaches 1 to 3 in the 2002 to 2004 period. Relative abundance in LD4 and Island-Rainbow Park was similar and relatively low in the two periods.

Flannelmouth sucker in the 451- to 500-mm TL size class from electrofishing samples dominated collections in all study reaches and in both study periods (Fig. 20). Fewer small (< 400 mm TL) flannelmouth suckers were found in samples in Lodore Canyon in the recent period compared to 1994 to 1996, particularly in reaches LD1 to LD3. Relatively large numbers of small flannelmouth sucker were present in the upper portions of Whirlpool Canyon, with many found in Echo and Mitten parks.

Bluehead sucker was also distributed throughout the study area, and similar to flannelmouth sucker, large juveniles and adults in electrofishing samples increased in abundance downstream and relative abundance of small individuals in seine samples declined downstream (Fig. 21). Bluehead sucker relative abundance and, particularly, CPUE in electrofishing samples declined in all Lodore Canyon reaches in 2002 to 2004 compared to 1994 to 1996; the decline in CPUE for bluehead sucker was significant. Bluehead sucker abundance ranged from 5% to 10% of the electrofishing sample in upper Lodore Canyon in both sampling periods and represented 25% of the electrofishing sample in lower Whirlpool Canyon in 2002 to 2004. Relative abundance of bluehead sucker in seine samples declined dramatically in Browns Park and all Lodore Canyon reaches and constituted < 5% of the fish community in 2002 to 2004. Bluehead sucker relative abundance was similar in the two periods in Island-Rainbow Park. Density of bluehead sucker in seine samples increased sharply in WH1 in 2002 to 2004, compared to upstream and downstream reaches.

Bluehead suckers in the 301 to 400-mm TL size class in electrofishing samples dominated in all study reaches. Frequency distributions appeared relatively stable in both study periods in Lodore Canyon except that smaller fish (250 mm TL or less) were less common (Fig. 22). Small bluehead suckers were relatively common in upper Whirlpool Canyon reach WH1, particularly in Echo and Mitten parks.

Mottled sculpin was found throughout the study area and generally represented a small proportion of fishes captured with both seine and electrofishing gears. Mottled sculpin was the only native fish to increase in abundance in Lodore Canyon electrofishing samples between the 1994 to 1996 and 2002 to 2004 periods, particularly in LD3 and LD4 (Fig. 23); the difference in CPUE was significant. Mottled sculpin relative abundance in seine samples was generally higher in the 1994 to 1996 period, particularly in reaches LD2 to LD4 compared to the 2002 to 2004 period. Sculpin abundance in seine samples in 2002 to 2004 was relatively low in all reaches except LD3.

Thus, abundance of all native fish declined in electrofishing samples collected from Lodore Canyon between the periods 1994 to 1996 and 2002 to 2004, with the exception of mottled sculpin. Declines of all taxa were statistically significant except for Colorado pikeminnow, which showed a moderately substantial decline that was statistically not significant. Declines were severe for speckled dace, roundtail chub, and mountain whitefish. Abundance declines of even common taxa were substantial; CPUE for bluehead sucker and flannelmouth sucker declined 39 and 37%, respectively, over the period.

In seine samples, relative abundance of all but one native fish declined between the 1994 to 1996 and 2002 to 2004 periods. Declines were severe for flannelmouth sucker, bluehead sucker, and speckled dace; relative abundance of those taxa in Lodore Canyon seine samples declined 81, 83, and 85%, respectively, between the periods. Mountain whitefish was the only native fish to increase in abundance between the periods and that difference was slight.

Size structure changes for native fish in Lodore Canyon showed that small individuals (300 mm TL or less) were less common in 2002 to 2004 compared to 1994 to 1996, which may reflect diminished recruitment for those species. Reduced recruitment may also partially explain lower abundance of adults in 2002 to 2004 period. Differences in roundtail chub length frequency in the Green River upstream of the Yampa River (all large) and downstream (mixed sizes) may reflect differences in the predator fish community in each reach; this is discussed below in more detail.

Reduced absolute abundance of native fishes and lower recruitment in Lodore Canyon and elsewhere may be a function of increased abundance of non-native fishes. For example, increased abundance of red shiner may cause increased levels of predation on early life stages of all native fishes, including catostomids (Ruppert et al. 1993, Bestgen et al. 1997, Bestgen et al. in press). Fathead minnow is also an efficient predator on catostomid fish larvae, and their increased abundance may be associated with reduced survival of early life stages of native fishes (Dunsmoor 1993). Effects of small-bodied predators are likely most severe when age-1 and 2 fish prey on larvae of native fishes. Larger larvae, juveniles, and small-bodied adults of some native fishes that may or may not occupy backwaters may be affected by other predators as well. These include expanding populations of smallmouth bass and channel catfish in the study area, particularly Lodore Canyon, and brown trout which remained abundant in Lodore Canyon in 2002 to 2004.

Non-native fishes.—All trouts were combined to illustrate the general abundance pattern for salmonids (excluding whitefish) because those cold-water taxa responded similarly to environmental conditions. Brown trout constituted about 85% of all trouts captured, with rainbow trout and rainbow x cutthroat trout hybrids (plus a few cutthroat trout, Snake River subspecies) constituting the other 15%. Brown trout was found in all study reaches except Island-Rainbow Park. In both time periods, salmonid abundance patterns were similar: they were most common in upper Lodore Canyon (LD2) and were abundant (Fig. 24); CPUE for the two periods was not significant. Trout abundance declined rather rapidly downstream and by reach LD4 trout were < 10% of the fish community. Salmonid abundance increased slightly in WH2 because of fish moving from the abundant Jones Hole Creek population.

Brown trout population size structure in Lodore Canyon differed among the two periods. In 1994 to 1996, a wide variety of sizes were present and the most abundant size classes were usually the intermediate 251 to 400-mm TL size classes (Fig. 25).   In 2002 to 2004, we also captured a wide variety of sizes of brown trout, but two size groups dominated samples. The smaller size group was composed of size-classes that were 151 to 250 mm TL, and the second was composed of fish in the 351 to 450-mm TL size classes; fish of intermediate size were present but less common. Even the relatively small samples from reach WH2 showed a similar bimodal pattern.

Red shiner expanded their distribution and were more abundant in the upper portions of the study area in the 2002 to 2004 sampling period compared to 1994 to 1996 (Fig. 26). In 1994 to 1996, red shiner was absent from Browns Park and Lodore Canyon reaches LD1 to LD3, but were common in LD4. Red shiners were not present in the upper extent of the study area in 2002, but invaded Browns Park in 2003, and increased in abundance in all Lodore Canyon reaches. Relative abundance and density of red shiner was high in Whirlpool Canyon reaches and in Island-Rainbow Park in 2002 to 2004.

In spring and summer 2003, red shiner was abundant in lower Lodore Canyon, rare or not present in upper Lodore Canyon, and in low abundance in newly invaded Browns Parl, with two specimens captured in spring and four in summer 2003. By autumn 2003, we documented red shiner throughout Browns Park from just upstream of Swinging Bridge (RK 612.8) downstream through Lodore Canyon. Red shiner was often the most abundant species in seine samples in Browns Park in autumn 2003, particularly in backwater habitat. Adult and juvenile size classes were present, which suggested successful reproduction by that species in that reach. In 2004, we did not observe red shiners upstream of Lodore Canyon in spring, summer, or autumn, a pattern similar to that observed in 2002 and from 1994 to 1996.

Sand shiner was found in all study reaches and increased in abundance downstream (Fig. 27). Similar to red shiner, sand shiner distribution and abundance in Lodore Canyon increased in 2002 to 2004 compared to 1994 to 1996. Relative abundance of sand shiner in seine samples increased most in LD3 and LD4 in 2002 to 2004 and it was more common in the IRP reach in 2002 to 2004 than the earlier period. Sand shiner was also abundant in Whirlpool Canyon. Four sand shiner were found in lower Browns Park in 2002, but were not captured there in 2003 or 2004.

Redside shiner was found in all study area reaches and showed a pattern of decreasing abundance downstream (Fig. 28). Redside shiner abundance showed a particularly large increase in Browns Park and reaches LD1 and LD2 in Lodore Canyon in the 2002 to 2004 period.

Fathead minnow was distributed throughout all study reaches and its abundance generally declined in a downstream direction in the 2002 to 2004 period (Fig. 29). Fathead minnow relative abundance in Browns Park was high and similar among the two study periods, increased in 2002 to 2004 in reaches LD1 to LD3, and declined in LD4 compared to 1994 to 1996. Fathead minnow abundance was relatively low in the Island-Rainbow Park reach in 2002 to 2004 compared to the 1994 to 1996 period.

White sucker was found throughout the study area and its abundance in both electrofishing and seine samples declined downstream (Fig. 30). White sucker abundance in electrofishing samples was highest in upper Lodore Canyon; CPUE data showed a significant increase in white sucker abundance in the 2002 to 2004 period (Table 8). Small-bodied white suckers captured in seine samples comprised a substantial portion of the fish community in Browns Park and upper Lodore Canyon samples, but declined in abundance in reach LD3 and downstream. White sucker abundance showed a small increase in the Island-Rainbow Park reach in the recent period compared to 1994 to 1996 samples. Identity of white suckers and possible occurrence of Utah sucker is ongoing using comparisons of known-identity specimens.

Size structure of white sucker in electrofishing samples showed a preponderance of relatively small fish that were 250 mm TL or less in upper Lodore Canyon, and were more abundant in the 2002 to 2004 period than before (Fig. 31). Relatively larger fish were present in reaches LD1 to LD3, and small fish predominated in samples in LD4 and downstream. Large white suckers were uncommon in Whirlpool Canyon.

Hybrid suckers were present in most reaches of the study area and comprised 3.9% of all fishes captured in the 1994 to 1996 sampling period and were 7% of all suckers captured by electrofishing in Lodore Canyon. Hybrids, particularly those involving white sucker as a putative parent, were most abundant upstream and represented 9.0, 8.8, and 8.6% of all suckers captured in Lodore reaches 1 through 3, respectively. Numerical abundance of hybrids declined in LD4 to 4.2% of all catostomids captured.

In 2002 to 2004 sampling, hybrid suckers comprised 4.3% of all fishes captured and were 8.5% of all suckers captured by electrofishing in Lodore Canyon. Hybrid suckers comprised 2.4% of all fishes captured in Whirlpool Canyon samples and were 4.2% of all suckers captured there by electrofishing in the 2002 to 2004 period. Flannelmouth x white sucker was the most abundant hybrid captured, followed by flannelmouth x bluehead sucker and bluehead x white sucker. Hybrids, particularly those involving white sucker as a putative parent, were most abundant upstream and represented 13.2 and 9.0% of all suckers captured in Lodore Canyon in LD1 and LD2, respectively. Hybrid sucker abundance declined in lower Lodore reaches LD3 and LD4 to 5.3 and 6.4%, respectively, of all catostomids captured, but overall, relative abundance of hybrid suckers increased in 2002 to 2004 in all reaches of Lodore Canyon. In each of the 1994 to 1996 and 2002 to 2004 periods, relative abundance of hybrids declined downstream consistent with a downstream decline in white sucker abundance.

Channel catfish showed increased distribution and abundance in Lodore Canyon in the recent sampling period compared to 1994 to 1996 (Fig. 32). In 1994 to 1996, channel catfish was nearly absent from reaches LD1 and LD2 and was found in low abundance in LD3 and LD4. Relative abundance and CPUE data showed that channel catfish was more abundant in upper Lodore Canyon in 2002 to 2004 and more than doubled their abundance in LD4. Catch rates increased from 1.3 fish/h in 1994 to 1996 to 6.1 fish/h in the 2002 to 2004 period, a significant increase. Channel catfish abundance in LD4 in 2002 to 2004 was similar to that in Whirlpool Canyon in the same period.

Size structure of channel catfish in electrofishing samples in Lodore Canyon changed between the two periods, mainly because of increased abundance in 2002 to 2004 (Fig. 33). In general, during both periods larger fish (> 300 mm TL) were more common in the upper reaches of Lodore Canyon and smaller fish were more common in lower Lodore Canyon; the pattern is more obvious in the recent period. Similar to lower Lodore Canyon, channel catfish in Whirlpool Canyon in the 251 to 300 mm TL size class dominated populations, although a few very large fish were also present. No small channel catfish were sampled in Lodore Canyon, perhaps indicating absence of reproduction by that species there. Small channel catfish were present in downstream Whirlpool Canyon.

Smallmouth bass distribution and abundance in Lodore Canyon increased dramatically in the 2002 to 2004 period compared to 1994 to 1996 (Fig. 34); CPUE data showed a significant increase (Table 8). Only a single smallmouth bass was collected from 1994 to 1996 by electrofishing and was found in reach LD3. Now smallmouth bass occupy all Lodore Canyon reaches and increased in abundance in a downstream fashion, where they now constitute > 6% of the fish community in LD 4 and > 10% of the fish community in Whirlpool Canyon. Seine samples showed a similar pattern, as smallmouth bass reproduction was first noted in lower Lodore Canyon in 2002 (N = 4, 13 to 85 mm TL) and was more progressively widespread in 2003 and 2004. Smallmouth bass were not found in Browns Park.

Size structure of smallmouth bass captured in 2002 to 2004 suggested a population invading from downstream, as the few fish present upstream were relatively large. Based on the low numbers of small fish upstream, reproduction there was presumed low or absent (Fig. 35). In reaches LD3 and LD4, modal size was smaller at 201 to 250 mm TL and a wide variety of sizes of smallmouth bass were present, including small and very large fish. In Whirlpool Canyon, modal size continued to decline as fish in the 151 to 200-mm TL size class were most abundant.

In contrast to native fishes, non-native fishes generally remained abundant in the 2002 to 2004 period or increased in distribution and abundance in the study area since 1994 to 1996. Of the five non-native species for which we compared electroshing CPUE data in Lodore Canyon between the 1994 to 1996 and 2002 to 2004 periods, brown trout increased slightly and northern pike remained similar in abundance between the two periods. Similarity in brown trout abundance was somewhat surprising given the relatively warm conditions present during summer in 2002 to 2004. During summer 2002, when water temperatures were warm, brown trout were in poor condition and uncommon relative to 2003 and 2004. Brown trout apparently quickly recolonized the reach from upstream and returned to former abundance levels. White sucker, channel catfish, and smallmouth bass all increased in abundance in 2002 to 2004 compared to the 1994 to 1996 period; the latter two taxa expanded their upstream distribution as well.

Size distributions of those same five non-native fish captured by electrofishing all changed in 2002 to 2004. Reasons for the bi-modal size distribution of brown trout in 2002 to 2004 were unknown, but may reflect reduced recruitment and abundance in 2002, a very warm year. More large smallmouth bass and channel catfish were found upstream, perhaps suggesting invasion from downstream rather than escapement from upstream Flaming Gorge Reservoir. Age-0 smallmouth bass were detected only in lowermost Lodore Canyon in 2002 in low numbers, the first documented reproduction there. More were noted in 2003 and 2004 and each year moved progressively upstream. Reproduction was verified by collection of very small smallmouth bass in drift net samples in 2003 and 2004, but not 2002, perhaps reflecting very low reproduction in summer 2002. Smallmouth bass reproduction continues to expand in Lodore Canyon, as an age-0 specimen was captured in autumn 2005 in uppermost Lodore Canyon. Small white suckers were also more abundant upstream, possibly indicating higher recruitment or higher reproduction there.

In Lodore Canyon, red shiner and sand shiner invaded upstream reaches in 2002 to 2004 and were relatively more abundant there than in 1994 to 1996. Over all Lodore Canyon reaches, relative abundance of those two taxa increased 925 and 352% respectively, and fathead minnow increased 243% between the two sampling periods. Currently, smallmouth bass and green sunfish are more common in backwaters where few or none previously existed; their abundance has increased by several orders of magnitude.

Increased hybrid sucker numbers in Lodore Canyon between 1994 to1996 and 2002 to 2004 periods may be due in large part to relatively recent expansion of white suckers into the Green River upstream of the Yampa River. A single specimen of that species was detected in a rotenone sample taken in September 1957 in the Green River in Browns Park (Nolting 1957). White sucker was not detected in samples collected just after impoundment of Flaming Gorge Reservoir and was rare in collections made by Holden and Crist (1981) in 1978 to 1980. However, white sucker was relatively common from 1994 to 1996 and 2002 to 2004, particularly in upstream Browns Park where water was colder. Razorback x flannelmouth sucker hybrids were detected as long ago as 1889 in the Upper Colorado River Basin and in 1950 in the upper Green River near Hideout Canyon (Hubbs and Miller 1953), so some level of hybridization of native taxa should be expected even under relatively pristine conditions. However, the variety of hybrid combinations and their abundance in the Green River appear to be recent phenomena, as Holden and Crist (1981) reported only a few hybrids.

Some hybrid specimens appeared to have traits of white, bluehead, and flannelmouth sucker, which may indicate some level of back-crossing among hybrids. The relatively high and apparently increasing number of hybrids noted in this study and in other areas (Holden and Stalnaker 1975a; 1975b), and the large number of hybrid combinations, suggest that this phenomenon deserves further study. Detection of hybrids in field samples will be critical to accurately determine the extent of hybridization and whether the proportion of hybrids among all catostomids present is increasing over time. A better understanding of hybridization mechanisms and hybrid fertility, especially in river reaches strongly affected by regulation, may also aid restoration of rare catostomids such as razorback suckers.

Effects of season, year, and turbidity on capture success, electrofishing samples.—The overall effect of season on the average catch rate of all fish species in samples from 1994 to 1996 was significant ($\chi^2 = 68.17$, 2 *df*, $p < 0.0001$), with catch rates higher in autumn (3.14 fish/h, 95% CI = 2.85 to 3.49) than in spring (1.88 fish/h, 95% CI = 1.54 to 2.30) or summer (1.73 fish/h, 95% CI = 1.55 to 1.93). Spring and summer catch rates were not different ($\chi^2 = 0.54$, 1 *df*, $p = 0.46$).

Mean catch rates of all fish species in samples between autumn (3.05 fish/h, 95% CI = 2.73 to 3.41) and summer (1.72 fish/h, 95% CI = 1.53 to 1.93) in 2002 to 2004 were also different ($\chi^2 = 48.19$, 1 *df*, $p < 0.0001$). Comparison of summer and autumn catch rates for abundant species, such as brown trout, bluehead sucker, flannelmouth sucker, white sucker, channel catfish, and smallmouth bass showed that capture rates were higher in autumn. Colorado pikeminnow capture rates were not different between the two periods ($p = 0.32$). Species richness in electrofishing samples collected from 2002 to 2004 was also higher in autumn (N = 31, including hybrids) than in summer (N = 24).

The CPUE data for electrofishing samples in the 2002 to 2004 period showed differences among years ($\chi^2 = 11.77$, 2 *df*, $p = 0.0028$). Mean catch rates in 2002 were lowest (1.95 fish/h, 95% CI = 1.69 to 2.25), and different from 2003 ($p = 0.0007$) and 2004 ($p = 0.015$). Catch rates were higher in 2003 (2.72 fish/h, 95% CI = 2.39 to 3.10) and 2004 (2.54 fish/h, 95% CI = 2.17 to 2.96); the catch rate difference among those years was not significant ($p = 0.49$). Species richness in electrofishing samples collected in the 2002 to 2004 period was highest in 2002 and 2004 (N = 26 each year, including hybrid combinations) and lowest in 2003 (N = 23).

Capture rates of fish in samples when water was turbid were similar to or higher than those when water was clear. For example, in the 1994 to 1996 period, average capture rates of all fish species in turbid water (3.01 fish/h, 95% CI = 2.40 to 3.78) were higher ($\chi^2 = 3.59$, 1 *df*, $p = 0.0581$) than when water was clear (2.3 fish/h, 95% CI = 1.97 to 2.70). In the 2002 to 2004 period the same pattern prevailed because capture rates of fish in turbid water (2.51 fish/h, 95% CI = 2.26 to 2.78) were higher ($\chi^2 = 1.28$, 1 *df*, $p = 0.26$), but not significantly so, than those when water was clear (2.28 fish/h, 95% CI = 1.99 to 2.60). Catch rates for individual species such as brown trout, bluehead sucker, flannelmouth sucker, white sucker, Colorado pikeminnow, and smallmouth bass in turbid and clear water were also similar. Channel catfish capture rates were slightly lower in turbid (5.0 fish/h) than clear water (6.9 fish/h, $p = 0.06$).

Abundance reductions for many native fishes between the 1994 to 1996 and 2002 to 2004 sampling periods led us to examine if seasonal differences in catch rates or differences in environmental factors such as turbidity influenced capture rates. Changes in seasonal capture rates among the two sampling periods were postulated because two of the four sampling trips in the 1994 to 1996 period were in spring and one in each of summer and autumn. Conversely, in the 2002 to 2004 period only summer and autumn electrofishing sampling (one per year, total of three summer and three autumn samples) were conducted. Thus, fifty percent of samples were taken during spring in the 1994 to 1996 period, a time of relatively low capture rates, and 50% of samples in the 2002 to 2004 period were taken in the autumn season, a time of relatively high catch rates. Because of these differences, we believe that relatively low catch rates in the recent period were not a function of differences in sampling seasons among the studies. Differences in the two periods may even be conservative because more samples in the 1994 to 1996 period were collected in spring when capture rates were relatively low. We also do not think catch rate differences were due to crew or gear differences, because similar personnel, gear, and methods were used in each of the two time periods.

Comparison of seasonal differences in catch rates from the 2002 to 2004 period suggested CPUE was substantially higher in autumn than in summer for most species. Higher capture rates in autumn suggested that monitoring sampling for most species may be best conducted at that time. A negative aspect of autumn sampling is that detection of ripe Colorado pikeminnow would not be possible. Higher species richness in autumn electrofishing samples from 2002 to 2004 is additional support to conduct adult fish monitoring in autumn.

We expected capture rates of fish to decline when water was turbid compared to when water was clear, because fish presumably would be more difficult to see and capture. When all available data in both 1994 to 1996 and 2002 to 2004 periods was analyzed, we found this was not true because capture rates were similar in turbid and clear water. Comparison of capture rates for some of the more common species (brown trout, flannelmouth sucker, and bluehead sucker) and ones of particular interest (Colorado pikeminnow, smallmouth bass) were also similar in turbid and clear water. This suggested that monitoring sampling should not be impeded by turbid water, if there is a choice to be made. We speculate that high turbid-water catch rates may be due to greater susceptibility overall and higher abundance of fish in shallow riffles. Clear and shallow riffles sometimes support few fish, perhaps because fish lack the cover needed to occupy such places or because they detect the approaching raft and retreat to other habitats. Speas et al. (2004) found that capture rates of rainbow trout increased in turbid compared to clear water and attributed the increase to inability of fish to see the approaching boat, and possibly, reduced ability to escape due to low water temperatures. Channel catfish capture rates were slightly lower in turbid than clear water. This is reasonable because many channel catfish never undergo taxis to the raft anode where they are visible and available for capture. Instead, channel catfish are often stunned directly over the substrate and roll along the bottom in the current, such that their capture would be less likely in turbid than in clear water.

Although fish capture rates were affected by a number of factors including turbidity and sampling season, we could not attribute any differences to factors that existed solely in one or the other periods, 1994 to 1996 or 2002 to 2004. In the absence of artifacts caused by environmental conditions or differences in sampling efficiency, we consider reductions in capture rates in the recent period as unbiased and reflect real reductions in native fish abundance.

Effects of reach, season, and year, on fish density, 2002 to 2004 seine samples.—Seine samples showed significant differences in abundance of all taxa captured among years, seasons, and reaches and all first order interactions among those effects were also significant. Across all years from 2002 to 2004, average abundance of each fish taxa in seine samples was lowest in Browns Park (mean = 1.15 fish/10m^2, 95% CL = 0.805 to 1.50) and the Lodore Canyon reaches LD1 and LD2 (LD1 mean = 0.95 fish/10m^2, 95% CL = 0.61 to 1.30; LD2 mean = 0.71 fish/10m^2, 95% CL = 0.42 to 1.30), slightly higher in LD3 and LD4 (LD3 mean = 1.49 fish/10m^2, 95% CL = 1.02 to 1.97; LD4 mean = 1.36 fish/10m^2, 95% CL = 0.88 to 1.84), increased substantially in Whirlpool Canyon (WH1 mean = 4.14 fish/10m^2, 95% CL = 3.04 to 5.23; WH2 mean = 3.13 fish/10m^2, 95% CL = 2.17 to 4.10), and then declined in Island-Rainbow Park (IRP mean = 1.44 fish/10m^2, 95% CL = 1.09 to 1.78. The GLM analysis of abundance as a function of reaches was significant (χ^2 = 91.69, 7 *df*, p < 0.0001).

Seine samples also showed significant differences in fish abundance across seasons (χ^2 = 19.08, 2 *df*, $p < 0.0001$). Abundance was highest in autumn (2.47 fish/10 m^2, 95% CI = 2.12 to 2.90), intermediate in summer (1.83 fish/10 m^2, 95% CI = 1.44 to 2.28), and lowest in spring (1.25 fish/10 m^2, 95% CI = 0.92 to 1.70). Species richness in seine samples collected from 2002 to 2004 was high in summer (N = 25) and autumn (N = 26), and lower in spring (N = 23).

Seine samples also showed significant differences in fish abundance across years (χ^2 = 8.05, 2 *df*, $p = 0.0178$). Abundance was highest in 2003 (2.17 fish/10 m^2, 95% CI = 1.82 to 2.57), intermediate in 2002 (1.88 fish/10 m^2, 95% CI = 1.54 to 2.30), and lowest in 2004 (1.31 fish/10 m^2, 95% CI = 0.93 to 1.86). Species richness (including hybrid combinations) in seine samples collected in the 2002 to 2004 period was highest in 2003 (N = 28), intermediate in 2002 (N = 25), and lowest in 2004 (N = 23).

Individual species patterns.—High variation and interactions noted in statistical analyses and the large influence of a few abundant non-native fishes prompted us to examine patterns of fish abundance at annual, seasonal, and reach scales for each native and several common non-native species in seine samples collected during 2002 to 2004. Abundance of common native fishes was highest in 2002, and in most years, highest in summer. For example, bluehead and flannelmouth sucker and speckled dace abundance was highest in 2002 in all reaches, and comparatively low in 2003 and 2004 (Fig. 36 and 37). Abundance in all years was relatively low in spring, high in summer, and low again by autumn, except for speckled dace in Browns Park, which was rare in all seasons. Each species was relatively common in Browns Park, increased in abundance in Lodore and Whirlpool canyons, and declined in abundance in Island-Rainbow Park. Roundtail chub was most abundant in 2002 but was absent in Browns Park, rare in Lodore Canyon, most common in Whirlpool Canyon, and low in Island-Rainbow Park. Most specimens were captured in summer, and in two cases, showed highest relative abundance then (Fig. 37). Mottled sculpin was most abundant in Lodore Canyon, less so in Browns Park, and uncommon in downstream reaches (Fig. 38). Seasonal abundance of mottled sculpin was highest in summer in Lodore Canyon, where we often found small individuals in shallow near shore runs over sand substrate or in macrophytes. Mountain whitefish showed a distinctly different distribution pattern than mottled sculpin, being most common in Browns Park, and were abundant there in spring of each year (Fig. 38). Mountain whitefish was relatively abundant in spring 2003 seine samples in Lodore Canyon, but rare in all other years, reaches, and seasons. Mountain whitefish was most abundant in 2003, and we found a limited number as far downstream as Island-Rainbow Park in spring, but few otherwise. Too few Colorado pikeminnow were captured in seine samples to graph results; we discuss young pikeminnow captures in the species accounts section. Only one bonytail, and no humpback chub or razorback sucker was captured in seine samples.

Abundance patterns of the most common non-native fishes, red shiner and sand shiner, generally increased in a downstream direction, being rare or absent in Browns Park, and abundant downstream (Fig. 26, 27, 39). Fathead minnow was most common in upstream Browns Park and declined in abundance downstream (Fig. 29, 40). Seasonal abundance patterns of red shiner, sand shiner, and fathead minnow were highly variable, but generally increased from spring to summer to autumn. Annual abundance was highest in 2002 and 2003 but lower in 2004. Higher spring abundance was not always associated with higher autumn abundance the previous year. Green sunfish abundance was highest in Lodore Canyon and generally low elsewhere (Fig. 40). In Lodore Canyon, spring and summer abundance was relatively low and autumn abundance higher. Patterns elsewhere were variable. Redside shiner and white sucker showed some consistencies in abundance patterns, being more common in upstream Browns Park than in downstream reaches (Fig. 28, 30, 41). White sucker showed seasonal abundance patterns similar to native suckers, relatively low in spring, relatively high in summer, and lower in autumn, and was most abundant in 2002 over all seasons, especially in Browns Park. Redside shiner abundance was relatively high in spring and summer, but lower in autumn except in Browns Park and Lodore Canyon in 2003, where autumn abundance was quite high. Smallmouth bass were not found in Browns Park, but were more common in downstream reaches (Fig. 42). Spring abundance of smallmouth bass was low, relatively high in summer, and lower in autumn. Abundance was lowest in 2002 and higher in 2003 and 2004.

Seasonal differences in pooled fish abundance (lowest in spring, intermediate in summer, and relatively high in autumn) were what we expected. Spring abundance was lowest presumably because of high overwinter mortality or few fish occupying relatively cold backwaters, summer abundance increased because of successful reproduction, and autumn abundance was perhaps highest because most fish were susceptible to the capture gear used. Several abundant non-native species are very small at hatching and are difficult to capture in summer because they are not susceptible to the mesh size of the seine, but grow sufficiently by autumn so that capture efficiency is high. Highest fish abundance in 2003 seine samples was likely due to presence of a few very large samples of sand shiner, red shiner, and reside shiner in some reaches or seasons.

Highest species richness in seine samples in summer and autumn is likely due to increased presence and abundance of age-0 life stages of rare species that survived to that point in the year. Highest species richness in 2003 was due to presence of a few rare species that were absent in other years and to the large number of fish captured, which increased our chances of detecting rare species.

Reach and seasonal abundance patterns of each fish species depended, in large part, upon individual thermal tolerances and timing of spawning, and differed among native and non-native fishes. Relatively high abundance of most native taxa in summer (bluehead sucker, flannelmouth sucker, and speckled dace) was likely because most were late-spring or early-summer spawning species. Thus, age-0 fish were not present during spring seine sampling, but were relatively abundant and large enough in summer to be captured in seines. Mortality resulted in lower autumn abundance. Additional reasons for lower autumn abundance could be that fish in some taxa grew sufficiently to move from backwaters to main channel habitat or were fast enough to avoid capture by seines. Overwinter mortality and habitat shifts likely resulted in additional reductions in native fish abundance in the subsequent spring.

Seasonal abundance patterns were similar in the Green River in the mostly regulated reach upstream of the Yampa River and in the partially regulated reach downstream of the Yampa River. Highest abundance of native taxa in Lodore and Whirlpool canyons is likely a product of the relatively high abundance of progeny produced in those reaches rather than ingress of larvae from upstream. Cool-water tolerant mountain whitefish was most abundant in upstream reaches, likely because abundance of adults was highest in cool reaches upstream of there. Fishermen accounts support the idea of large upstream populations of adult mountain whitefish (pers. comm. J. Bundy). Highest relative abundance of mountain whitefish in spring samples was due to hatching of autumn-spawned eggs. Lower abundance in summer and autumn was likely due to mortality and, perhaps, intolerance of warm water in downstream reaches.

Native fish relative abundance and density in summer was highest in 2002 in the Green River by a considerable margin. The pattern was supported in both the mostly regulated reach upstream of the Yampa River and in the regulation-attenuated reach downstream of the Yampa River. Spring runoff and summer base flow levels were similar among all years and summer thermal regimes were warm in each year. Thus, flow or thermal regimes in each year did not explain differing summer abundance of native fishes in the Green River. Higher summer abundance of native bluehead sucker, flannelmouth sucker, and speckled dace in 2002 did not result in higher autumn density, compared to other years. Autumn density of native suckers was similar to the low densities of flannelmouth sucker and bluehead sucker observed by Haines and Tyus (1990) in Green River backwaters. Additional years of sampling in more varied environmental conditions may be needed to determine factors that influence seasonal and annual differences in native fish abundance (e.g., Propst and Gido 2004).

Presence and severity of turbidity events in the Green River may be an explanation for the relatively high variability in summer season abundance of native fishes observed in seine samples collected from 2002 to 2004. We observed relatively high native fish abundance in seine samples in summer 2002 when no severe turbidity events were observed. Alternatively, in 2003 and 2004 we observed low native fish abundance in summer and two severe turbidity events occurred in each year. Turbidity events may cause fish to lose orientation in turbid water and be transported downstream. This hypothesis was supported by high catch rates of fish in drift net samples in the Green River in 2003 and 2004 after flood events (Fig. 43). This also occurred in the Yampa River, when Colorado pikeminnow (and other taxa) abundance in drift samples typically increased when turbidity events occurred (Bestgen et al. 1998). Because such events in the Green River happened after native fish reproduction occurred, no additional larvae were available for re-colonization and their abundance remained low through autumn. Non-native fish were likely similarly reduced by turbidity events (e.g., red shiner and sand shiner in Whirlpool Canyon, summer 2003) but, because many of those taxa spawn into late-summer, populations could recover by autumn to achieve high population levels we observed. Fish transport may be particularly high from canyon reaches, because few shallow, channel margin backwaters occurred where fish could find refuge.

Abundance patterns for red shiner and sand shiner increased downstream and, although highly variable and in contrast to patterns for several native fishes, often increased from spring through summer and autumn. Reasons for this pattern were discussed above for pooled fish sample statistical analyses. Relatively low summer abundance of some small-bodied non-native cyprinids may be related to turbidity events; for example, sand shiner and red shiner in Whirlpool Canyon in 2003.

Green sunfish abundance that was highest in Lodore Canyon was likely a result of consistent sampling of a few permanent backwaters that usually held green sunfish. Spring smallmouth bass abundance was low, relatively high in summer, and lower in autumn. Age-0 smallmouth bass were typically observed in low-velocity backwaters or slow, sandy main channel margins, but age-1 bass moved to rocky main channel areas that had higher current velocity and cobble substrate. Abundance was lowest in 2002 and higher in 2003 and 2004, reflecting an expanding population.

Trammel net sampling.—A total of 535.5 net h (one 23 m-long trammel net set for one h = one net h) of trammel net sampling was conducted during 2002 to 2004. Trammel nets were effective at capturing native suckers and chubs. The most common native species captured was flannelmouth sucker (0.54 fish/h), followed by roundtail chub (0.51 fish/h), and bluehead sucker (0.23 fish/h; Table 3). Humpback chub capture rate was 0.016 fish/h, or about one fish for every 60 h of trammel net sets. The few net sets in autumn 2004 (n = 16 net h) captured 0.313

bonytail/h. Channel catfish was the only common non-native fish captured in trammel nets; its capture rate was 0.50 fish/h. The few trammel net sets made in Lodore Canyon captured few fish; little habitat was available for effective trammel netting there.

We examined differences in capture rates of chubs with nets set in day compared to night. Day and night sets captured roundtail chubs at about the same rate, 0.52 (SE = 0.089, N = 108 fish) and 0.51 (SE = 0.055, N = 182 fish) fish/h, respectively. Nighttime capture rate for humpback chub was 0.022 fish/h (SE = 0.008, N = 7) and higher than the 0.005 fish/h capture rate for daytime (SE = 0.005, N = 1), but numbers of fish captured was low so no statistical tests were performed. Day and night sets captured bonytail at about the same rate, 0.37 (SE = 0.37, N = 2 fish) and 0.29 (SE = 0.11, N = 7 fish) fish/h, respectively; number of fish captured was also low for bonytail.

Equivalent day and night time capture rates for roundtail chub were surprising, given that most chub sampling programs target crepuscular dusk and dawn times. Average capture rate of roundtail chub in Whirlpool Canyon trammel net samples was higher than average capture rate for roundtail chub in the Black Rocks, Colorado River, 1998 to 2000 (McAda 2002). Humpback chub was captured mostly at night, but capture rates for it and bonytail were small so veracity of day:night differences in capture rates was difficult to assess. Channel catfish was sometimes very abundant in trammel net samples, reaching catch rates of 8 to 15 fish/h compared to a maximum of 4 fish/h for roundtail chub. Large catches of channel catfish sometimes inhibited additional trammel-net sampling for chub due to the long time required to clear nets of fish.

Selectivity and relative inefficiency of electrofishing gear compared to trammel nets to sample chub in deep pools of Whirlpool Canyon was evident. For example, during several electrofishing trips a large pool near Stateline Campground was sampled, but only one or no chub were collected. Sampling the same pool with six trammel nets on three separate nights in October 2003 yielded 50, 48, and 28 roundtail chub. Similar results were obtained in the relatively deep upper portion of Whirlpool Canyon, where few or no roundtail chub were captured with electrofishing gear on several occasions and subsequent sampling with up to six trammel nets yielded up to 40 roundtail chub in a single night. Although electrofishing is a reasonable means to supplement distributional data and increase sample sizes, effective chub monitoring requires trammel netting.

Drift net sampling.—Drift net sampling was conducted in the Green River just upstream of the Yampa River in each year of 2002 to 2004 period, as well as in 1994 and 1995. No early life stages of Colorado pikeminnow were captured in any year that drift net sampling was conducted. We detected two putative razorback sucker x white sucker hybrids, one in each of 2003 and 2004; these individuals were 33 and 30 mm TL, respectively.

Based on spawning ecology of Colorado pikeminnow and water temperatures that were present in the Green River in Lodore Canyon, pikeminnow should have been able to spawn there in all years of this study (Nesler et al. 1988, Bestgen et al. 1998). Drift net sampling conducted in the adjacent lower Yampa River in each of those years detected reproduction by Colorado pikeminnow, based on capture of larvae. Some unknown habitat differences or low numbers of ripe adults may explain the apparent absence of spawning by Colorado pikeminnow in the Green River in Lodore Canyon.

Putative razorback sucker x white sucker hybrids that we captured in drift nets set in lower Lodore Canyon in 2003 and 2004 were identified with traditional morphological analysis (Snyder and Muth 2004). We plan to verify the taxonomic identity of those putative hybrids more fully with genetic analysis. If that identification is verified, it would represent the first documented hybridization between those sucker species. Regardless, expanding populations of white sucker may pose an additional threat to razorback sucker recovery efforts. White suckers in the San Juan River system have been established in the Animas River for many years (Platania et al. 1987; 1988) but remain rare in the downstream main stem of the San Juan River (pers. comm., D. Propst, New Mexico Dept of Game and Fish).

Few fish species and individuals were captured in drift net samples in 1994 to 1995, but species diversity and number of fish captured increased substantially in the period 2002 to 2004 (Table 9, Fig. 43). Sampling effort also increased in the 2002 to 2004 period. In 1994 to 1995, six species were captured and > 90% of those individuals were native. Up to 16 species were captured in a single year (2003) in the 2002 to 2004 period, when substantially more non-native taxa were captured than in previous years.

The proportion of native fishes was 50% or less in samples in each year in the 2002 to 2004 period. Native taxa numbers in all years were dominated by flannelmouth sucker and bluehead sucker, particularly in 2003, and fewer numbers of speckled dace and very few (n = 3) early life stage roundtail chub were collected, and only in 2003. In all samples, mountain whitefish specimens were juveniles > 50 mm TL.

Large increases in non-native fish in 2002 to 2004 drift net samples were mostly because of increased abundance of non-native cyprinids and white suckers. In 2002, the large increase in the number of fish collected was due to red shiner, and those were mostly large juveniles and adults. In 2003 and 2004, sand shiner, red shiner, and white sucker were the most common non-native taxa. We also collected young (N = 6, 10 to 24 mm TL) smallmouth bass in drift samples in 2003, and greater numbers in 2004 (N = 93, 8 to 28 mm TL), but none in 2002. Those small smallmouth bass represent the first documented reproduction in Lodore Canyon.

In 2003, most fish (60%) were captured the day of and the day following two turbidity events, one on 22 July and one on 6 August. The 22 July 2003 event was especially severe, as water clarity was reduced to zero and silt loads were extremely high from runoff in the Vermillion Creek drainage basin. Most bluehead sucker (53%) and flannelmouth sucker (66%) captured that year were taken during these events, as were most smallmouth bass (4 of 6) and sand shiner (74%). About 34% of red shiner captured that year were taken during these two events. We noted relatively high mortality of those fish in drift net samples collected on 22 July; usually fish collected in drift nets are alive upon capture and larger suckers present in mid- to late-July are noticeable. Thus, loss of visibility and outright mortality from sediment loads may be factors associated with increased transport rates.

A similar high turbidity and increased flow event occurred on 17 July 2004, when a total of 532 fish of all species (29% of all captured in 2004) were captured. All 2004 smallmouth bass specimens were captured in the period 17 to 25 July, 87 of 93 (94%) on 17 July. Flow stage dropped 7 to 8 cm the next day, but turbidity remained high for several days. On 25 July, a similar turbidity event occurred, and only 5 fish were captured (three flannelmouth sucker, one white sucker, and one smallmouth bass). A similar situation occurred in 1994: most fish captured that year (N = 223, 72.6%) were taken on 7 July when water turbidity increased dramatically from thunderstorm runoff.

Large captures of fish, including smallmouth bass, in drift net samples in the Green River upstream of the Yampa River in 2003 and 2004 associated with turbidity events suggested susceptibility to displacement. The literature also suggested negative effects of flow increases on early life stages of smallmouth bass (Lukas and Orth 1995, Peterson and Kwak 1999, references therein); Winemiller and Taylor (1982) documented negative effects of turbidity and high flows on disruption of smallmouth bass spawning, nest destruction, and displacement of young. Increased flows levels via dam releases or turbidity events may be potential controlling mechanisms for early life stages of smallmouth bass in this portion of the river.

Species accounts

Colorado pikeminnow.—Large juvenile or adult (399 mm TL or larger) Colorado pikeminnow were captured throughout the study area in all years in the period 2002 to 2004 and in all reaches except Browns Park. A total of 41 Colorado pikeminnow captures were made; an additional 10 fish were observed but not netted. One individual was recaptured twice during this study and three individuals were recaptured once each. Thus, we captured a total of 37 individuals. We captured one Colorado pikeminnow by angling, 35 by electrofishing, one with a seine, and four in trammel nets. Colorado pikeminnow averaged 573 mm TL (399 - 765 mm TL; Fig. 15). Thirteen captures were made in 2002, 19 in 2003, and 9 in 2004. Nineteen of the Colorado pikeminnow were found in the Green River upstream of the Yampa River and 22 were found downstream. Mean length (589 mm TL) and weight (1,774 g) of Colorado pikeminnow in the Green River captured upstream of the Yampa River were greater than mean length (553 mm TL) and weight (1,463 g) for Colorado pikeminnow downstream of the Yampa River; *t*-tests demonstrated marginally significant differences in length ($p = 0.13$) and weight ($p = 0.07$). Captures and observations were made throughout Lodore and Whirlpool canyons; four captures were from reach LD1, three from LD2, four from LD3, eight from LD4, six and 14 were from

Whirlpool Canyon reaches 1 and 2, respectively, and two were from Island-Rainbow Park. Capture rates by electrofishing were 0.39 fish/h for reaches LD1 through LD4. The length-weight relationship calculated for large juvenile and adult Colorado pikeminnow captured in the Green River, Lodore Canyon, 1994 to 1996 was:

$$\textit{Weight}(\text{g}) = 39.04 * e^{0.0065 * TL\ (\text{mm})},\ r^2 = 0.98.$$

The length-weight relationship calculated for large juvenile and adult Colorado pikeminnow captured in the Green River, Lodore Canyon, 2002 to 2004 was:

$$\textit{Weight}\ (\text{g}) = 64.09 * e^{0.0054 * TL\ (\text{mm})},\ r^2 = 0.94.$$

Length-weight relationships showed Colorado pikeminnow from Lodore Canyon were, on average, heavier at length in the 1994 to 1996 period than fish from the 2002 to 2004 period (Fig. 44). The shape of the curve for the 1994 to 1996 period was not affected by the largest fish.

We detected PIT tags in 22 of the 41 Colorado pikeminnow we captured compared to only 2 out of 17 in the 1994 to 1996 period. Three of the four individuals captured and recaptured during this study were at large sufficiently long (captured in different sampling trips, one fish was captured in the same trip) to observe changes in length or weight. The three fish were at large about an average of 2 months of the summer growing season and despite negligible changes in TL, gained 4, 12, and 46% in weight.

Young Colorado pikeminnow were captured in seine samples from the upstream Whirlpool Canyon reach in Echo Park (N = 5, 28 to 50 mm TL) and in the lower end of the Island-Rainbow Park reach (N = 15, 33 to 89 mm TL). Twelve of the 20 young pikeminnow were collected in 2002. Seven of these 12 were collected in spring 2002, which means they were produced in 2001. One young Colorado pikeminnow was collected in 2003 and seven were captured in 2004.

Comparisons of our data to historical captures suggested that Colorado pikeminnow were more abundant in Lodore Canyon in 1994-1996 than at any time since 1980 (Holden and Crist 1981). Using slightly different electrofishing gear and sampling fixed sites, Holden and Crist (1981) captured two Colorado pikeminnow for a capture rate of 0.08 fish/h of electrofishing. In 1987-1988, Karp and Tyus (1990) sampled Lodore Canyon with an electrofishing raft and captured three Colorado pikeminnow (mean = 0.27 fish/h). Capture rates for Colorado pikeminnow were relatively high in 1994 to 1996 at 0.6 fish/h (we modified our estimates of pikeminnow abundance reported in Bestgen and Crist 2000 using the same methods used in this study) but apparently declined in 2002 to 2004 to about 0.4 fish/h.

Higher recapture rates in the 2002 to 2004 period was likely the result of intensive basin-wide sampling for Colorado pikeminnow in 2000 to 2003 and capture and tagging of a relatively large number of pikeminnow in Lodore Canyon in a separate study (Kitcheyan and Montagne 2006). Lodore Canyon remains an important seasonal-use area for Colorado pikeminnow. Kitcheyan and Montagne (2006) found that Colorado pikeminnow used Lodore Canyon seasonally, mostly in summer and autumn, and particularly in low-flow years. Presence of a ripe and tuberculate male Colorado pikeminnow in Lodore Canyon in summer in each of 2001 and 2003 indicates possible spawning in Lodore Canyon.

Roundtail chub.—Chubs were absent in Browns Park, rare in Lodore Canyon, more common in Whirlpool Canyon, and less so in IRP in 2002 to 2004 (Fig. 16). Electrofishing captured only nine adult roundtail chub in three years of sampling from 2002 to 2004 in Lodore Canyon in reaches LD1, LD3, and LD4; seining captured only seven small individuals (12 to 125 mm TL) and they were from the same reaches. In comparison, 46 roundtail chub adults were captured with electrofishing in the Whirlpool Canyon reach and 272 small chubs (13 to 117 mm TL) were captured by seining. A total of 72 small chubs (18 to 86 mm TL) were captured in the Island-Rainbow Park reach with seining from 2002 to 2004. All small chubs captured in 2002 to 2004 were identified as roundtail chub, based on morphological and meristic characters, although the longer fin lengths were often more typical of humpback chub (Muth 1990).

Chubs were more common in Lodore Canyon in 1994 to 1996 than in 2002 to 2004. Fourteen adult roundtail chubs were captured in the earlier period, with at least one from each Lodore Canyon reach. Roundtail chubs were in Lodore Canyon seine samples in 1994 to 1996 in reaches 2 (N = 2) and 4 (N = 28); 20 of the LD4 reach fish were taken in a single sample in a backwater just upstream of the Yampa River. Roundtail chub was only slightly more abundant in the Green River in Island-Rainbow Park seine samples (n = 39). Adult chubs captured in 1994 to 1996 in Lodore Canyon were generally large; mean TL was 374 mm (255 - 451 mm TL).

Limited trammel net sampling in Lodore Canyon did not detect roundtail chub. Trammel net sampling in Whirlpool Canyon captured 290 adult roundtail chub. Dorsal/anal fin ray counts of large juvenile and adult roundtail chub from all reaches and captured by all gears were as follows: 7/9 (N = 2), 8/8 (N = 1), 8/9 (N = 8), 8/10 (N = 5), 9/6 (N = 1, damaged anal fin), 9/8 (N = 12), 9/9 = (N = 232), 9/10 (N = 27).

Adult roundtail chub captured in electrofishing and trammel net samples ranged in size from 220 to 439-mm TL, modal size class was 251 to 275-mm TL. Chubs smaller than 200-mm TL were either rare or not susceptible to electrofishing and trammel nets. Most adult roundtail chub captured in 2002 to 2004 from Lodore Canyon were large; mean TL was 367 mm (range 236 to 439).

In September and October 2003 sampling of Whirlpool Canyon, the three sampling passes yielded 84, 88, and 96 roundtail chubs of which 12 were recaptures. Using model M_o in program MARK, abundance of roundtail chubs was estimated at 1956 (*SE* = 529, 1214 to 3546, 95% profile likelihood interval, *CV* = 27%, probability of capture = 0.046). Of the 12 recaptures made during the sampling passes, eight chubs remained in place, two moved upstream about 1.5 RK from just downstream of Rain Camp (RK 357.1), and two fish moved from upstream of Rain Camp downstream to State Line, a distance of about 5.3 RK. One roundtail chub, first captured and tagged at Stateline in October 2003, moved 2.5 km downstream to Compromise Camp and was recaptured in September 2004.

We do not know to what degree roundtail chub abundance estimate assumptions were met. We do not expect recruitment or tag loss during the period. Some fish were moving within the study in the relatively short time period encompassed by the three-pass sampling, but we do not know how this might have affected estimates. Broader spatial sampling was needed to determine if movement of tagged fish out of the study reach and movement of untagged fish into the study reach biased abundance estimates. Most fish we released were in relatively good condition, and the high recapture rate for the few humpback chub captured (3 of 8, see below) suggested high survival of marked animals during our sampling. Longer-term mortality from tagging effects may merit more attention, given other investigators' concerns over handling mortality (pers. comm., C. McAda, U. S. Fish and Wildlife Service, Grand Junction). Nevertheless, estimated abundance of roundtail chubs in Whirlpool Canyon may offer some benchmark for future comparisons.

The wider range of roundtail chub size classes present in Whirlpool Canyon compared to Lodore indicated a more stable population with some recruitment. The Whirlpool Canyon reach doubtless gets some addition of young from upstream reaches because young chubs are present in drift samples collected in summer in the lower Yampa River (Bestgen et al. 1998). Near absence of certain predators such as abundant brown trout may contribute to the relative success of roundtail chubs in Whirlpool Canyon. Salmonids are significant predators on humpback chubs in the Colorado River, Grand Canyon (Marsh and Douglas 1997). Smallmouth bass and channel catfish are also potential predators in both Lodore and Whirlpool canyons. Continued population monitoring may be needed to reveal whether the Whirlpool Canyon population of roundtail chub is stable.

Humpback chub.—A total of eight humpback chub were collected during sampling conducted from 2002 to 2004. Humpback chub in Whirlpool Canyon were not the classic deep-bodied form found in Colorado River in places such as Black Rocks, Westwater, and Grand canyons. Rather, they conformed to the humpback chub morphotype found in the rest of the Green River Basin in Desolation-Gray canyons of the Green River, Yampa Canyon of the Yampa River, and the Little Snake River (Fig. 45). They were typified by a relatively finefeatured, small, and delicate head and mouth, a slightly overhung snout and subterminal mouth, a small nuchal hump, and a slender body and caudal peduncle. The line extending backwards from the angle of the anal fin passed just over or in front of the upper lobe of the caudal peduncle. Body coloration was typically silvery. Dorsal/anal fin ray counts were usually 9/9 (N = 5), but 8/10 (N = 1), 9/10 (N = 1), and 10/10 (N = 1) were also observed.

In comparison, roundtail chub had stouter overall features, including a larger head, a terminal mouth, no nuchal hump, and a more robust body and caudal peduncle. A few roundtail chub > 350 mm TL possessed slightly raised nuchal humps, but the other features just discussed were used to differentiate humpback from roundtail chub. The line extending backwards from the angle of the anal fin passed through the upper lobe of the caudal peduncle. Body coloration was silvery but often darker, especially dorsally.

All humpback chubs collected were from the steep-walled upper portion of Whirlpool Canyon in a 1.8-km-long reach beginning at RK 551.8. One additional humpback chub was collected by angling in 2003 in the same reach; it was released before it was tagged. All humpback chub captured were from deep pools or recirculating eddies associated with an upstream riffle, and were typically captured in nets set downstream of large boulders that produced areas with relatively low current velocity. Humpback chub was nearly always captured with roundtail chub and often in the same net set. In autumn 2004, we captured humpback chub, roundtail chub, and bonytail from the same eddy in upper Whirlpool Canyon, which is likely the first such record in many years.

We captured humpback chub individuals in each year of the study: two in 2002, four in 2003, and two in 2004. One fish first marked on 6 October 2003 at RK 551.8 was recaptured there on 13 October 2003, and then again on 28 July 2004. That fish increased in length from 263 mm TL in 2003 to 297 mm TL in 2004 (34 mm change) and gained 76 grams (136 in 2003 to 212 in 2004) over the capture interval. Another humpback chub captured on 7 October 2003 was recaptured by the Utah Division of Wildlife Resources (S. Finney, pers. comm.) on 29 October 2003 in the same location. Another humpback chub first marked on 23 September 2004 at RK 550.5 was recaptured 0.4 RK upstream in Whirlpool Canyon on 14 September 2005. That fish increased in length from 332 mm TL in 2004 to 352 mm TL in 2005 (20 mm change) and gained 68 grams (252 g in 2004 to 320 g in 2005) over the interval. One humpback chub we captured on 28 July 2004 at RK550.4 was a recapture (375 mm TL) that was first captured and tagged in Desolation Canyon at RK 338.9 on 14 April 2003. It moved upstream 212 RK and increased in length 114 mm over the recapture interval. That same recaptured 2004 individual was captured by electrofishing, the remainder of humpback chubs were captured in trammel nets. Humpback chubs ranged in length from 232 to 375-mm TL, the modal size class was 251 to 275 mm-TL (Fig. 17).

The low number of humpback chubs collected over three years of study and the low proportion of humpback chub to roundtail chub (4/256) in samples collected in three consecutive sampling passes conducted in autumn 2003 in Whirlpool Canyon suggested that the humpback chub population in Whirlpool Canyon is relatively small at this time. This is supported by a companion study, where few humpback chubs were captured in the lower Yampa and in the Green River in Whirlpool and Split Mountain canyons (Finney 2006 draft). Extensive analysis of young chubs captured in Whirlpool Canyon and downstream did not reveal reproduction by humpback chub in the 2002 to 2004 period.

The Black Rocks area of the Colorado River supports a substantial humpback chub population in a reach smaller than Whirlpool Canyon, so extent of habitat does not appear limiting. However, Black Rocks is much deeper and may contain more of the habitat preferred by humpback chub. Soundings conducted throughout Whirlpool Canyon near steep-walled areas revealed that few areas were deeper than 5 m, and most areas where humpback chubs were collected were 2 to 3 meters deep during summer low-flow periods. Deep pools with sufficiently low current velocity in which to set nets were also relatively uncommon in Whirlpool Canyon at flows > about 85 m^3/s. Presence of all three chub species supports the idea that Whirlpool Canyon may be an important conservation area for chubs in the Green River Basin.

Bonytail.—Our autumn 2004 Green River sampling occurred less than two weeks (22-23 Sept.) following two separate releases of hatchery-produced bonytail at the Echo Park boat ramp (RK 554.6, 12 Sept. 2004, n = 2,587, mean TL = 219 mm; 16 Sept. 2004, n = 2,988, mean TL =

239 mm). We detected bonytail by raft electrofishing from about 100 m upstream of the boat ramp downstream to just below Jones Hole Creek (RK 544.5), a dispersal distance of 9.7 RK in a maximum time of 11 d. The largest concentration of fish was in pools and eddies within about 200 m downstream of the boat ramp. We captured several individuals and let 60 or more go unnetted, and quit sampling after that until we were about 800 m below the boat ramp. We did not net many individuals and stopped sampling in that reach to reduce stress on stocked fish, because electrofished individuals displayed flared gill covers and a rigid body.

We also sampled bonytail with trammel nets in Whirlpool Canyon that same night and captured nine individuals, along with 15 roundtail chubs and a single humpback chub. Bonytail were captured mostly in eddies and pools 2 to 3 m-deep, but we also captured them in swifter runs and riffles with current velocity of 0.7 m/sec or greater. Individuals from riffles were apparently feeding, as several plump individuals excreted algae and insect parts upon handling after capture. In general, bonytail appeared in good condition. Dorsal/anal fin ray counts of individuals examined were all 10/10 (N = 13) or 10/11 (N = 2). Bonytail ranged in size from 201 to 285-mm TL; the mode of the size distribution was the 226 to 250-mm TL size class (Fig. 17).

We also found a bonytail that had been consumed by a 330-mm TL smallmouth bass. The bonytail, about 225-mm TL (68% of bass length), was noted because the caudal fin was protruding from the esophagus into the mouth of the bass. We documented a similar instance of predation by a smallmouth bass on a stocked bonytail in summer 2005 (unpublished data).

We did not detect bonytail in Lodore or Whirlpool canyons in 2002 or 2003. Bonytail were stocked in the Green River in Browns Park (4 October 2002, RK 597, n = 8,600, mean TL = 224 mm) and Echo Park in 2002 (10 October 2002, RK 554.6, n = 5,000, no mean TL reported, assume 224 mm TL) in autumn 2002. Our autumn 2002 seine sampling on 10 October occurred six d after stocking in Browns Park, when we found the single near-dead individual. Our autumn 2002 trammel net sampling occurred the day of stocking at Echo Park. Since we were downstream several km we would not expect to capture any of those bonytail that same season. Bonytail were stocked in the Green River at Echo Park in 2003 (7 October 2003, RK 554.6, n =

1,592, mean TL = 210 mm) in autumn 2003. Our autumn 2003 trammel net sampling occurred six to nine d after stocking at Echo Park, but we did not detect any of those individuals.

We documented that hatchery-reared bonytail survived for short periods in the wild, but they do not appear to survive over winter because we captured no fish that were at large more than a few weeks. We did capture early June-stocked bonytail downstream of Echo Park in late September 2005 (unpublished data). Lack of longer-term recaptures was in spite of stocking in all three of our sampling years, 2002 to 2004, in areas upstream of our sampling. We feel confident to detect their presence because we used an array of gears in a wide variety of habitat types in all seasons except winter. Even though bonytail captured in autumn 2004 were healthy shortly after stocking, observations of fish captured in autumn 2005 that were stocked the previous June revealed that most fish were in poor condition. Most fish had multiple *Lernea* infections, some had fungus, and their general appearance and color were poor. In summer 2005, we also netted a weakly swimming bonytail in Split Mountain Canyon with an aquarium net. The specimen had a fungus infection, *Lernea* scars, missing scales, and was near death. Means to increase survival of stocked bonytail in this portion of the Green River should be investigated.

Razorback sucker.—We did not capture any pure adult razorback suckers in the study area in the three years of sampling from 2002 to 2004; one was detected in Lodore Canyon in the 1994 to 1996 period. We captured two razorback sucker X flannelmouth sucker hybrids, identified on the basis of characteristics that were intermediate between the presumptive parental species. Both individuals were from lower Whirlpool Canyon and one recaptured individual was in the Recovery Program database as a pure razorback sucker even though it was clearly a hybrid and exhibited nearly no nuchal hump. We already discussed the two putative razorback sucker X white sucker hybrids, one in each of years 2003 and 2004, captured in drift nets set in the Green River upstream of the Yampa River.

Non-native predators.—Eighteen northern pike were captured by electrofishing, two by seining, and one in a trammel net and all were found in the lowermost 16 km of Lodore Canyon and Whirlpool Canyon. Three were taken in 2002, 10 in 2003, and eight in 2004. Five pike were captured in Lodore Canyon reach LD3, 11 in LD4, three in WH1 and two WH2. Most LD4 fish were captured within 1 km of the confluence of the Yampa and Green rivers. Northern pike captured averaged 644 mm TL (191 to 825 mm TL), and all were healthy and fat. Most were captured from deep eddies and pools or runs. There was evidence of reproduction by northern pike in Lodore Canyon as a 191 mm TL fish, presumably age-0, was captured in summer 2003 in lower Lodore Canyon. There was also evidence that northern pike may be reproducing upstream of Lodore Canyon, based on finding age-0 specimens in Browns Park in autumn 2005 (n = 10).

Number of northern pike (13 in 1994 to 1996, 21 in 2002 to 2004) and CPUE in each period (0.2 fish/h in each study period) was similar, as was the size distribution, with the exception of the small individual. Thus, northern pike abundance in the study area appeared stable between the 1994 to 1996 and 2002 to 2004 study periods. We captured four tagged pike in 2004; all originated in the Yampa River between Craig and Lily Park and were tagged in 2003. Three of the recaptured fish were from the Green River just upstream of the Yampa River and one was captured just below the confluence.

Channel catfish distribution and abundance in Lodore Canyon in the 2002 to 2004 sampling period increased upstream relative to that found in 1994 to 1996; abundance increases were evident across Lodore Canyon and channel catfish was most abundant in reach LD4. Abundance of channel catfish among the 2002 to 2004 sample years in Lodore Canyon was highest in 2002 and 2003 but similar to smallmouth bass, declined in 2004, particularly in reaches LD1 to LD3. Channel catfish abundance was similar in the Green River in each Whirlpool Canyon reach across all sample years in the 2002 to 2004 period. There was no evidence of reproduction by channel catfish in the Lodore Canyon reach of the Green River because we did not collect any channel catfish < 180 mm TL in seine or drift net samples.

Smallmouth bass increased in abundance in Lodore Canyon throughout the study period, and were much more abundant in 2002 to 2004 than in the 1994 to 1996 period, when only a single smallmouth bass was captured by electrofishing and two were captured by seining. Smallmouth bass abundance increased most between 2002 and 2003, particularly in the lower two Lodore Canyon reaches and in Whirlpool Canyon (Fig. 46). Abundance of smallmouth bass in

2003 and 2004 was similar in most reaches but increased slightly in LD2 in 2004 and declined in WH2 in the same year.

We also recaptured tagged smallmouth bass during 2004. One individual captured in lower Lodore Canyon originated in the downstream Jensen-Ouray reach (pers. comm., R. Brunson, Utah Division of Wildlife Resources). We also recaptured three tagged bass (240, 271, 340 mm TL) in the Whirlpool Canyon reach, fish that were marked in conjunction with an abundance estimation and removal study in that area (M. Fuller, pers. comm, U. S. Fish and Wildlife Service).

It appears from seine and drift net sampling data that smallmouth bass first reproduced in Lodore Canyon in 2002 in limited numbers. The upstream range expansion of smallmouth bass in Lodore Canyon continued through 2005 when small age-0 bass were noted nearly to the head of Lodore Canyon in autumn (unpublished data). It is possible that Lodore Canyon was colonized by fish that escaped over the spillway from Flaming Gorge Reservoir during releases in 1997 and 1999. However, if that were the case, we would expect higher abundance of, and more reproduction by, smallmouth bass in upper Lodore Canyon and Browns Park. The opposite pattern seems to prevail; more bass and most reproduction occur in downstream reaches of Lodore Canyon. Low levels of smallmouth bass were present in Lodore Canyon since 1994 to 1996 sampling. It is possible that recent increases were the result of a slowly expanding population between 1994 to 1996 and2002 to 2004.

Reasons for the recent large increase in abundance of channel catfish and smallmouth bass in Lodore Canyon are not precisely known but are likely related to low flows and warm water conditions present in the Green River beginning about 2001. Prior to then, higher base flows prevailed and temperatures were cooler. Perhaps those warm water conditions promoted successful smallmouth bass reproduction (but not channel catfish) beginning in 2002 in Lodore Canyon. Prior to that, reproduction and recruitment may have been limited by cooler summer temperatures which limited reproduction directly, and delayed hatching or limited growth of smallmouth bass so that their overwinter survival was poor. It is also possible that sufficient numbers of smallmouth bass had not yet invaded Lodore Canyon prior to 2002 to permit successful reproduction. Anderson (2005) also found increased abundance of young smallmouth bass in the Yampa River in 2001, 2003, and 2004 when summer water temperatures were elevated (no sampling conducted in 2002).

The literature supports the idea that cooler, higher flows and flow fluctuations may limit growth and reproductive success by smallmouth bass (Lucas and Orth 1995, Peterson and Kwak 1999). Effects of flow and temperature regimes on smallmouth bass in Lodore Canyon and downstream will require additional monitoring in years when higher flows and cooler water temperatures prevail. High flows and turbidity events are known to displace young smallmouth bass from nests (Winemiller and Taylor 1982, Lukas and Orth 1995, Peterson and Kwak 1999). Use of flow manipulations from Flaming Gorge Dam to disadvantage smallmouth bass will require additional information so that effects can be optimized if implemented. Minimal information needs are onset and duration of the spawning season of smallmouth bass in relation to water temperature and other important environmental variables, and flow levels needed to cause the desired negative effect. Flow management activities to disadvantage selected species need to consider potential effects on other fishes in the system.

Slightly increased abundance of brown trout between the study periods 1994 to 1996 and 2002 to 2004 was not expected because of warm summer temperatures in the latter period. Apparently longer periods of warm water or higher temperatures are required to impact brown trout populations in Lodore canyon.

Composition and reproductive status of the fish community of the Green River upstream of the Yampa River, 1962-2004.—Limited sampling during pre-impoundment investigations in the present-day reservoir basin area documented that eleven native and six introduced fishes, and one catostomid hybrid inhabited that reach prior to 1960 (Table 10). All native fishes, including endangered Colorado pikeminnow, bonytail, humpback chub, and razorback sucker inhabited the reach and likely reproduced there.

During reservoir filling from 1963 to 1967, sampling yielded seven native and eight introduced taxa in the study area. The limited reproduction noted (Vanicek and Kramer 1969, Vanicek et al. 1970) for only a few species occurred in 1965 when reservoir discharge was relatively low and warm and only in downstream reaches. After normal dam operations commenced in 1967, sampling noted near absence of reproduction by native fishes in the regulated reach and their replacement by salmonids. Following penstock modifications that warmed reservoir releases in June 1978, sampling detected a total of nine native and 13 introduced taxa and two sucker hybrids. With the exception of rare native or incidental non-native taxa, most fishes reproduced in the regulated reach. Colorado pikeminnow (n = 2) and razorback sucker were captured (n = 2) but reproduction was not noted or suspected. Natives mountain whitefish, mountain sucker, and mottled sculpin re-appeared in collections, and first documented collections of introduced red shiner, sand shiner, and white sucker were made (Holden and Crist 1981).

Sampling during 1994 to 1996 detected a total of eight native fishes, 14 introduced ones, and five sucker hybrids. Notable was presence of a moderately-abundant population of Colorado pikeminnow (N = 17 individuals captured). Roundtail chub was rare, and mountain sucker was absent. Similar to previous studies, bonytail and humpback chub were not detected in the study area, and razorback sucker was rare (N = 1). Reproduction by Colorado pikeminnow and razorback sucker was not detected. Reproduction by six other native taxa, including roundtail chub, was detected. Sampling also detected the presence of three additional hybrid sucker combinations and three additional piscivores: northern pike, green sunfish, and smallmouth bass. Non-native Utah chub *Gila atraria*, and creek chub *Semotilus atromaculatus*, which were rare in 1978-1980, were not detected during sampling in 1994-1996.

Sampling during 2002-2004 captured a total of 24 species, eight native and 16 non-native, and seven hybrids. A single bonytail was the additional native species collected in the 2002 to 2004 period (autumn 2002) and was captured in Browns Park just after stocking. All other native fishes collected from 1994 to 1996 were also collected during 2002 to 2004, with the exception of razorback sucker. Abundance of Colorado pikeminnow was slightly reduced compared to 1994 to 1996 based on electrofishing sampling, but angling conducted in a separate study showed high abundance of Colorado pikeminnow in Lodore Canyon (Kitcheyan and Montagne 2006). Potential reproduction by Colorado pikeminnow and razorback sucker in Lodore Canyon has already been discussed. Roundtail chub were rare in this reach and reproduction, while detected, was limited.

Additional non-native fishes collected in the Green River upstream of the Yampa River in 2002 to 2004 included brook stickleback *Culea inconstans*, bluegill *Lepomis macrochirus*, and black bullhead *Ameiurus melas*. None of these taxa was collected during 1994 to 1996 (black bullhead was captured in 1978-1980). We also noted reproduction by northern pike and smallmouth bass during this time period.

Thus, the historical fish community of the Green River upstream of the Yampa River has undergone dramatic shifts in composition, abundance, and reproductive capability at least three times since 1962 in response to dam construction and river regulation. A pre-impoundment fish removal program also likely reduced distribution and abundance of native fishes in the regulated reach; subsequent re-invasion likely did not occur because of cold releases (Holden 1991). Post-penstock re-invasion of the regulated reach and subsequent reproduction by native and non-native fishes demonstrated the powerful role that water temperature played in regulating distribution and community composition.

Discharge modifications implemented in 1992, and the thermal enhancement that occurred as a result of it, also appeared to further shift the fish community and habitat in the Lodore Canyon reach of the Green River toward pre-dam conditions, based on 1994 to 1996 sampling (Bestgen and Crist 2000). It should be noted that fish community shifts detected during sampling from 1994 to1996 may be due to a combination of delayed response of the fish community to thermal modification in 1978 and changes in flow and temperature regimes since 1992 (Bestgen and Crist 2000). Several native taxa including bluehead sucker, flannelmouth sucker, speckled dace, and mountain whitefish remained abundant. Colorado pikeminnow also increased in distribution and abundance in the 1994 to1996 period compared to earlier ones, which may have been attributable to improved habitat conditions or to increased dispersal of individuals from expanding populations in other portions of the basin in the mid-1990's (Bestgen et al. 2005). Distribution and abundance of cold water rainbow and cutthroat trout have declined since 1980; brown trout, which are more tolerant of warm water, were relatively more abundant. Salmonids of any kind were quite rare in lowermost Lodore Canyon in 1994-1996, whereas introduced warm water species such as red shiner, sand shiner, fathead minnow, and channel catfish remained common or increased in abundance between 1980 and 1994.

Sampling in 2002 to 2004, a period defined mostly by low spring peak flows and low, warm summer base flows, showed that nearly every native fish declined in abundance in Lodore Canyon since 1994 to 1996. More non-native fishes were detected, and many pre-existing ones increased their distribution and abundance. Range and abundance expansions were particularly notable for smallmouth bass, channel catfish, and most small-bodied non-native cyprinids, and reproduction was noted for smallmouth bass, bluegill, black bullhead, creek chub, and brook stickleback for the first time. Abundance of salmonids apparently did not decline appreciably or permanently during the period when relatively warm base flows were present.

Changes in the Lodore Canyon fish community from sampling conducted in three periods, 1978 to 1980, 1994 to 1996, and 2002 to 2004.—Fish community comparisons from collections made in the 1978 to1980, 1994 to 1996, and 2002 to 2004 periods showed establishment of abundant non-native fish species over time and subsequent reductions in the once dominant native fish community (Table 11). At the upper Lodore Canyon site, collections in 1978-1980 revealed the presence of six native fishes, 10 introduced taxa, and one hybrid. Collections made in 1994 to1996 documented the presence of six native fishes, nine introduced taxa, and three hybrids. Only four native taxa, mountain whitefish, speckled dace, flannelmouth sucker, and bluehead sucker, were found in both periods. Small numbers of roundtail chub and mountain sucker were present in early collections, but were absent in later ones. Colorado pikeminnow, mottled sculpin, and bluehead X flannelmouth sucker were collected in the later period but not in the earlier one. Seven introduced taxa and one hybrid sucker were common to each period. The 1978 to 1980 collections documented the presence of Utah chub and red shiner, which were not collected in 1994-1996. The 2002 to 2004 sampling documented presence of introduced green sunfish and one additional hybrid, taxa not collected in the 1978-1980 period.

Sampling in 2002 to 2004 at the upper Lodore Canyon site, revealed the presence of seven native fishes (added roundtail chub compared to 1994 to 1996), 12 introduced fishes (added red shiner, sand shiner, and smallmouth bass), and four hybrid suckers. Early collections documented the presence of Utah chub and mountain sucker, which were not collected in the two more recent periods.

Overall, relative abundance of native species in the upper Lodore Canyon site declined from 92 to 65 to 13% of the fish community over the three study periods. Only a single native taxon, mountain whitefish, increased in relative abundance, but only by 0.1%. Bluehead sucker and speckled dace declined dramatically, while flannelmouth sucker were severely reduced over the last two periods. Colorado pikeminnow, roundtail chub, and mottled sculpin remained but were rare. Nearly all non-native taxa increased in relative abundance in 2002 to 2004 compared to prior periods. Largest increases in non-native fishes between the first two periods were by cool-water tolerant fishes such as brown trout, redside shiner, and white sucker. Largest increases in non-native fishes between the 1994 to 1996 period and 2002 to 2004 was by redside shiner and fathead minnow, and warm-water tolerant red and sand shiner and channel catfish. Abundance of hybrid suckers increased between the two early periods, but declined in the 2002 to 2004 period.

At the lower Lodore Canyon site at Alcove Brook, collections in 1978 to 1980 documented the presence of eight native fishes, 13 introduced ones, and one hybrid sucker. Sampling in 1994 to1996 captured seven native fishes, 13 introduced ones, and five hybrid sucker. Sampling in 2002 to 2004 captured seven native fishes, 14 introduced ones (added black bullhead and creek chub, lost cutthroat trout *Oncorhynchus clarki*, the non-native Snake River subspecies), four hybrid suckers and one hybrid centrarchid . Six native taxa (speckled dace, Colorado pikeminnow, roundtail chub, flannelmouth sucker, bluehead sucker, and mottled sculpin) were common to all three periods. Razorback sucker and mountain sucker were captured at this site in 1978 to 1980 but not later, and mountain whitefish was detected in the two later periods but not earlier. Collections made in 1994-1996 documented presence of introduced northern pike, green sunfish, and smallmouth bass, all of which may have only recently colonized from the Yampa River; their presence was verified in the 2002 to 2004 period.

Overall, relative abundance of native species in lower Lodore Canyon declined from 32 to 21 to 6% of the fish community over the three sampling periods. Native taxa declined in abundance or remained rare over the three study periods; speckled dace and roundtail chub showed particularly large declines. Ten introduced taxa increased in abundance, rainbow trout declined in abundance, and five others remained similar in abundance between the two earliest periods. Largest increases were by fathead minnow, sand shiner, white sucker, and hybrid suckers. Between the 1994 to 1996 and 2002 to 2004 periods, fathead minnow, redside shiner and all salmonids declined, and sand shiner and red shiner increased dramatically.

Species composition shifts at both sites suggested a progressive shift from a cold water fish community to one dominated by cool-water or warm-water taxa, with non-native warm water fish progressively invading and establishing upstream. The number of fish in samples in each of the upper and lower Lodore Canyon sites in the two earlier periods was similar, but more fish were sampled in the later 2002 to 2004 period. Although the likely reason for the increase is due to a larger number of seine samples in the last period, we feel as though the overall trends are reflective of the entire fish community and not due just to increased sampling in backwaters.

This limited data set of electrofishing and seine captured fish at these two Lodore Canyon sites showed an opposite trend of declining incidence of hybrid suckers compared to canyon-wide data collected in the same period. We believe this is an artifact of combining data using different gear types, because increased abundance of seine-collected small-bodied fish in the recent sample reduced the percent composition of hybrids, which are most commonly detected among the fewer relatively large-bodied fish.

SUMMARY

A main tenet of dam re-regulation to create more natural flow and temperature regimes is that native fishes will benefit (Stanford et al. 1996, Poff et al. 1997, Muth et al. 2000). The data sets presented here were used to evaluate that hypothesis.

Flow and temperature effects on the Green River fish community between 1994 to 1996 and 2002 to 2004.—Evaluating biotic response to flow and temperature recommendations is a considerable challenge given the multitude of factors that can affect the distribution and abundance of natural populations. A first challenge is obtaining reliable measurements of the population parameters of interest so that inferences can be made with confidence. We think we did a reasonable assessment of the fish community because our sampling was spatially and temporally intensive and we used a variety of gears to sample nearly all life stages of fish present in the system. Linking population measurements such as abundance levels to a driving variable or set of variables with certainty is another major challenge.

These difficulties not withstanding, we believe reasons for some observed changes in the fish community in the period 2002 to 2004, as compared to previous periods, were associated with changes in flow and temperature regimes. Expanded distribution and abundance of red shiner, sand shiner, channel catfish, and smallmouth bass were almost certainly a product of low flows and, particularly, warm temperatures. Water temperature in mid-summer in Browns Park and upper Lodore Canyon may be 2 to 3°C cooler than in lower Lodore Canyon (Bestgen and Crist 2000). However, upstream observations of 22°C or more were recorded in our study, conditions which likely promoted upstream expansion by warm water species. Distribution and abundance of most other non-native fishes also increased.

Warmer summer base flows may also have benefitted Colorado pikeminnow. Although our electrofishing data from 2002 to 2004 indicated a slight decline in Colorado pikeminnow, Kitcheyan and Montagne (2006) detected large numbers of Colorado pikeminnow in Lodore Canyon in 2002 and 2003, when flows were low in the rest of the basin. We did not detect reproduction based on capture of larvae in drift nets, but we collected ripe Colorado pikeminnow in Lodore Canyon in 2003 and there was a similar report in 2001 (T. Modde, pers comm., U. S. Fish and Wildlife Service, Vernal, Utah). Further, Kitcheyan and Montagne (2006) also reported small aggregations of Colorado pikeminnow during summer at two locations in Lodore Canyon, which may represent spawning groups. These data support the contention that lower and warmer summer base flows in Lodore Canyon may be a benefit to Colorado pikeminnow.

We did not anticipate reduced abundance of most other native fish of all life stages in the study area during 2002 to 2004. Declines in relative abundance of young native fishes may be a function of the influx of large numbers of non-native cyprinids that were captured in seine samples. The subsequent effect may be that they comprise a smaller portion of a larger sample, but do not actually decline in abundance. However, density and relative abundance trends for 2002 to 2004 data often tracked each other closely, suggesting that relative abundance measures often reflect density as well, and that reductions were real. Continued density difference comparisons will allow us to track such changes more closely.

Evaluation of the high flows in 1997 and 1999 was difficult because those events were relatively far removed, on a temporal scale, from our sampling. Those flows no doubt had positive benefit for physical habitat, including channel scouring and reworking, and associated removal of invasive tree species in Lodore Canyon and downstream reaches (Muth et al. 2000). Benefits to biota would be best evaluated in a time frame closely associated with the event. Potential measures of effects of high flows would be sampling of fish in low-velocity areas in summer and autumn such as was done in this study. With such data collected over a relatively long time frame, differences in reproductive success and abundance of native and non-native fishes could be evaluated. Escapement of smallmouth bass from Flaming Gorge Reservoir during spills is a potential downside of high flows, especially if release elevation levels coincide with vertical distribution of fish in the reservoir.

The net effect of flow and temperature regimes on the native fish community in this river reach, based on our 2002 to 2004 sampling, was mixed. During this study, we were able to obtain information on the response of the fish community to flow and temperature effects, but only at the lower end of the flow spectrum. Flow and temperature conditions observed in 2002 to 2004 fell within the bounds for recommendations that would be implemented in years with low or moderately low water availability, which happen in only 30% of years. Average, moderately high, and high flow years, which occur the other 70% of the time, were not evaluated because those conditions were not realized during this study period. Additional years of sampling when those flow conditions return are needed to fully assess the effects of pending flow and temperature recommendations for Flaming Gorge Dam on the fish community of the Green River. Because components of the fish community may be changing regardless of changes in flow and temperature conditions, monitoring should be on an annual basis. This would allow for documentation of fish community changes, such as new or ongoing species invasions that may otherwise be attributed to flow or temperature changes. Once additional years of sampling data are accumulated under different flow and temperature regimes, managers can make informed decisions about the effects of Flaming Gorge recommendations on the native fish community, including endangered fishes.

Recommendations for temperature and fish community monitoring in the Green River downstream of Flaming Gorge Dam.

Temperature monitoring.—We recommend continued collection of water temperature data and the addition of real-time temperature monitoring stations at Echo Park in the lower Yampa River and in the Green River just upstream of the Yampa River. An alternative to real-time monitoring is to model flow-specific temperature differences in each river. This will allow for continuous monitoring of the summer water temperature differences between the two systems when Colorado pikeminnow are drifting into the Green River from the Yampa River.

Large-bodied fish sampling recommendations.—We recommend two electrofishing sampling passes, one in summer and one in autumn. Samples collected during those times had the highest species richness and fish density. A main advantage of summer sampling may be to demonstrate use of Lodore Canyon by spawning Colorado pikeminnow, when water temperatures are warm. Monitoring of potential spawning by Colorado pikeminnow in Lodore Canyon may also be accomplished by drift net sampling in lower Lodore Canyon. Autumn electrofishing sampling is useful because capture rates were high. Using both summer and autumn sampling increases the likelihood that at least one sampling pass will have conditions conducive to high sampling efficiency.

Whirlpool Canyon chub sampling recommendations.—Confirmation of a small population of humpback chub in Whirlpool Canyon and capture of stocked bonytail and substantial numbers of roundtail chub suggest that additional monitoring should occur to document status of those populations and response to operational changes at Flaming Gorge Dam. One sampling pass of three to four days with trammel nets under low flow conditions in late summer or autumn sampling is recommended because water temperatures are relatively low and may reduce sampling mortality. More intensive capture-recapture sampling to document population abundance of chub species would be useful on a less than annual basis. All chubs should be PIT tagged to better understand abundance, survival, and movement of those species.

Small-bodied fishes.—We recommend two seine sampling passes, one in each of summer and autumn. Native fish abundance and species richness was usually high in summer, so sampling then is needed to document annual reproductive success. Autumn sampling for monitoring is also advantageous because non-native fish abundance and species richness is high and fish are larger and easier to identify.

Drift net sampling is one means to detect reproduction by adult Colorado pikeminnow in the Green River upstream of the Yampa River. This sampling is relatively easily accomplished when associated drift net sampling occurs in the lower Yampa River and may be a viable substitute for electrofishing sampling conducted in summer to detect whether ripe Colorado pikeminnow are using Lodore Canyon.

Sampling of any kind is not recommended in spring. Fish activity may be reduced when water temperatures are cold, and higher flows may limit sampling efficiency. Higher flows in spring due to elevated runoff from the Yampa River especially limits effectiveness of sampling in Whirlpool Canyon and downstream. This is because eddies that are sampled with trammel nets and backwaters that are sampled by seines are mostly washed out at higher flows.

CONCLUSIONS

- ♦ The impacts of construction and operation of Flaming Gorge Dam on physical habitat in the highly regulated reach of the Green River from the dam downstream to the Yampa River and in Whirlpool Canyon downstream of the Yampa River were partially remediated by thermal modifications implemented in 1978, discharge re-regulation in 1992, and 2002 to 2004 drought-period changes to baseflow levels and patterns.
- ♦ Thermal enhancement of the regulated reach via penstock modification in 1978 had a large restorative effect because reproduction by most native fishes in the Green River

upstream of the Yampa River was restored. Low-flow conditions in drought years 2002 to 2004 increased water temperatures in the Green River to levels that likely mimicked pre-dam conditions. This ensured a close match of Green and Yampa River water temperatures during drought years when Colorado pikeminnow larvae were drifting downstream from the Yampa River in summer.

- ♦ The Green River upstream of the Yampa River supported eight native fishes (nine if bonytail is included) in 2002 to 2004, and only Colorado pikeminnow and bonytail did not reproduce. Native fishes in Lodore Canyon declined in abundance in electrofishing, seine, and drift net samples compared to those collected in 1994 to 1996. Ten native fishes were collected in the Green River downstream of the Yampa River. Native fishes were numerically dominant in electrofishing and trammel net samples; seine samples were dominated by non-native taxa.
- ♦ Abundance of non-native fishes in Browns Park and Lodore Canyon increased in 2002 to 2004 compared to 1994 to 1996. Increases were throughout the study area and largest in the upper portions of Lodore Canyon for small-bodied cyprinids, channel catfish, and smallmouth bass. Smallmouth bass reproduction, which was not previously observed in Lodore Canyon, was widespread and northern pike reproduction was also observed. Salmonids were temporarily reduced in abundance in 2002, but increased in 2003 and 2004, and remained similar in abundance to that observed in 1994 to 1996. The overall predator load in the Green River study area has increased.
- ♦ Diversity and abundance of other non-native fishes increased since 1994 to 1996. For example, species not detected in 1994 to 1996 but found in 2002 to 2004 included creek chub, brook stickleback, bluegill, and black crappie. Abundance of green sunfish and black bullhead increased in 2002 to 2004 compared to the 1994 to 1996 sampling period.
- ♦ A strong water temperature gradient played a role in controlling distribution and abundance of fishes in Lodore Canyon. Further upstream expansion of several warm-water fishes was restricted to upper Lodore Canyon because of relatively cool upstream

water temperatures in most years. Previously, this limitation was observed in lower Lodore Canyon but moved upstream because of elevated temperatures in 2002 to 2004.

- Abundance of several hybrid combinations of suckers was high and increased since 1980 and again since the 1994 to 1996 period. Particularly common were hybrids that had white sucker as one parental type. Occurrence of cool water white suckers and hybrids declined in a downstream direction in Lodore Canyon, presumably in response to warmer water temperatures.
- Trammel net sampling detected a small population of humpback chub in Whirlpool Canyon and a relatively large population of roundtail chub. Hatchery-stocked bonytail were also captured. Trammel net sampling was a useful technique to sample chubs.
- Stocked bonytail were susceptible to predation and infections and survival rates were low based on absence of recaptures of fish more than a few weeks after stocking.
- Drift net sampling failed to detect reproduction by Colorado pikeminnow. Drift sampling revealed important patterns of increased downstream transport rates of fish in response to turbidity events.
- Colorado pikeminnow continued to use Lodore Canyon heavily in summer, based on captures we made and those in a companion study. Ripe male Colorado pikeminnow were detected there in 2001 and 2003, indicating suitable conditions for reproduction.
- More complete evaluation of the response of the fish community to flow and temperature regimes from Flaming Gorge Dam will require additional years of sampling in conditions other than the low and warm summer flows we observed.

RECOMMENDATIONS

- Continue removal of predaceous and other non-native fishes from Lodore Canyon.
- Continue to monitor water temperature, flows, and the fish community of the Green River in Browns Park, Lodore and Whirlpool canyons, and Island-Rainbow Park, as outlined.

- Compare fish community response to different flow and temperature regimes from Flaming Gorge Dam to better understand factors that may limit invasive species, particularly predators.
- Begin PIT-tagging and scanning all chubs captured, including roundtail chubs.
- Investigate reasons for low survival of stocked bonytail in the Green River.

ACKNOWLEDGMENTS

This study was funded by the Recovery Implementation Program for Endangered Fish Species in the Upper Colorado River Basin. The Recovery Program is a joint effort of the U. S. Fish and Wildlife Service (USFWS), U. S. Bureau of Reclamation (USBR), Western Area Power Administration, states of Colorado, Utah, and Wyoming, Upper Basin water users, environmental organizations, and the Colorado River Energy Distributors Association. Funding for this research was administered by the USBR under Cooperative Agreements with Colorado State University and the Larval Fish Laboratory (LFL). Project administration was facilitated by L. Crist, T. Chart, D. Speas, J. Nusbaum, and C. Morales. Laboratory, field, or administrative assistance was provided by D. Irving, F. Pfeifer, L. Crist, D. Speas, T. Modde, M. Caldwell, D. Beers, R. Remington, C. Huffacker, C. Smith, M. Trammel, C. Kitcheyan, M. Fuller, S. Finney, C. Walford, T. Sorensen; we are sure there are others that have escaped memory who also deserve thanks. C. McAda, U. S. Fish and Wildlife Service, Grand Junction provided an electrofishing raft for one trip. Boating and collecting permits from the states of Colorado and Utah, the U. S. Fish and Wildlife Service, the National Park Service/Dinosaur National Monument, and Browns Park National Wildlife Refuge were appreciated. Reviews by L. Crist., D. Propst, and D. Speas much improved this report.

References

Anderson, R. M. 1997. An evaluation of fish community structure and habitat potential for Colorado squawfish and razorback sucker in the unoccupied reach (Palisade to Rifle) of the Colorado River, 1993-1995. Colorado Division of Wildlife, Fort Collins, Colorado.

Andrews, E. D. 1986. Downstream effects of Flaming Gorge Reservoir on the Green River, Colorado and Utah. Geological Society of America Bulletin 9:1012-1023.

Banks, J. L. 1964. Fish species distribution in Dinosaur National Monument during 1961-1962. Masters thesis. Colorado State University, Ft. Collins.

Berry, C. R., Jr. 1988. Effects of cold shock on Colorado squawfish larvae. Southwestern Naturalist 33:193-197.

Bestgen, K. R., and L. Crist. 2000. Response of the Green River fish community and habitat to construction and re-regulation of Flaming Gorge Dam, 1962-1996. Unpublished report to the Recovery Implementation Program for Endangered Fishes in the Upper Colorado River Basin. Larval Fish Laboratory Contribution 107.

Bestgen, K. R., D. W. Beyers, J. A. Rice, and G. B. Haines. 1997. Recruitment models for Colorado squawfish: tools for evaluating relative importance of natural and managed processes. Unpublished report to the U. S. National Park Service and the National Biological Survey. Larval Fish Laboratory Contribution 95. 55 pp.

Bestgen, K. R., R. T. Muth, and M. A. Trammell. 1998. Downstream transport of Colorado squawfish larvae in the Green River drainage: temporal and spatial variation in abundance and relationships with juvenile recruitment. Colorado River Recovery Implementation Program Project Number 32. Larval Fish Laboratory Contribution 97. 63 pp.

Bestgen, K. R., D. W. Beyers, J. A. Rice, and G. B. Haines. In press. Factors affecting recruitment of young Colorado pikeminnow: synthesis of predation experiments, individual-based modeling, and field evidence. Transactions of the American Fisheries Society.

Beyers, D. W. 1998. Causal inference in environmental impact studies. Journal of the North American Benthological Society 17:367-373.

Bosley, C. E. 1960. Pre-impoundment study of the Flaming Gorge Reservoir. Wyoming Game and Fish Commission, Fisheries Technical Report No. 9.

Burdick, B. D. 1995. Ichthyofaunal studies of the Gunnison River, Colorado, 1992-1994. U. S. Fish and Wildlife Service, unpublished Final Report. Aspinall Studies Recovery Program Project No. 42.

Carlson, C. A., and R. T. Muth. 1989. The Colorado River: lifeline of the American Southwest. Pages 220-239 *in* D. P. Dodge, editor. Proceedings of the International Large River Symposium. Canadian Special Publication of Fisheries and Aquatic Sciences 106, Ottawa.

Douglas, M. E., W. L. Minckley, and H. M. Tyus. 1989. Qualitative characters, identification of Colorado River chubs (Cyprinidae: Genus *Gila*) and the "Art of Seeing Well". Copeia 1989:653–662.

Douglas, M. E., R. R. Miller, and W. L Minckley. 1998. Multivariate discrimination of Colorado plateau *Gila* spp.: “The art of seeing well” revisited. Transactions of the American Fisheries Society 127:163–173.

Dunsmoor, L. 1993. Laboratory studies of fathead minnow predation on Catostomid larvae. Draft Klamath Tribes Research Report: KT-93-01. 16p.

Finney, S. In press. Colorado pikeminnow (*Ptychocheilus lucius*) upstream of critical habitat in the Yampa River, Colorado. Southwestern Naturalist.

Gaufin, A. R., G. R. Smith, and P. Dotson. 1960. Aquatic survey of the Green River and tributaries within the Flaming Gorge Reservoir basin, Appendix A. Pages 139-162 *in* A. M. Woodbury, editor. Ecological studies of flora and fauna of Flaming Gorge Reservoir basin, Utah and Wyoming. University of Utah Anthropological Papers 48.

Grams, P. E. 1997. Geomorphology of the Green River in Dinosaur National Monument. Masters thesis. Utah State University, Logan.

Haines, G. B., and H. M. Tyus. 1990. Fish associations and environmental variables in age–0 Colorado squawfish habitats, Green River, Utah. Journal of Freshwater Ecology 5:427–436.

Holden, P. B. 1979. Ecology of riverine fishes in regulated stream systems with emphasis on the Colorado River. Pages 57-74 *in* J. V. Ward and J. A. Stanford, editors. The ecology of regulated streams. Plenum, New York, New York.

Holden, P. B. 1991. Ghosts of the Green River: Impacts of Green River poisoning on management of native fishes. Pages 43-54 *in* W. L. Minckley and J. E. Deacon, editors. Battle against extinction: Native fish management in the American Southwest. University of Arizona Press, Tucson.

Holden, P. B., and C. B. Stalnaker. 1975a. Distribution and abundance of fishes in the middle and upper Colorado River basins, 1967-1973. Transactions of the American Fisheries Society 104:217-231.

Holden, P. B., and C. B. Stalnaker. 1975b. Distribution of fishes in the Dolores and Yampa River systems of the upper Colorado River basin. The Southwestern Naturalist 19:403-412.

Holden, P. B., and L. W. Crist. 1981. Documentation of changes in the macroinvertebrate and fish populations in the Green River due to inlet modification of the Flaming Gorge Dam. Final report to U.S. Fish and Wildlife Service, Salt Lake City, Utah. BIO/West Contract No. 0-07-40-S1357.

Hubbs, C. L., and R. R. Miller. 1953. Hybridization in nature between the fish genera *Catostomus* and *Xyrauchen*. Papers of the Michigan Academy of Science, Arts, and Letters 38:207-233.

Karp, C. A., and H. M. Tyus. 1990. Humpback chub (*Gila cypha*) in the Yampa and Green rivers with observation on other sympatric fishes. Great Basin Naturalist 50:257-264.

Kitcheyan, D. C., and M. Montagne. 2006. Movement, migration, and habitat use by Colorado pikeminnow (*Ptychocheilus lucius*) in a regulated river below Flaming Gorge Dam, Utah. Prepared for Dinosaur National Monument DINO-N-465.006 and Central Utah Project Completion Act Office, U. S. Department of Interior. Prepared by U. S. Fish and Wildlife Service, Colorado River Fish Project, Vernal, UT.

Lukas, J. A., and D. J. Orth. 1995. Factors affecting nesting success of smallmouth bass in a regulated Virginia stream. Transactions of the American Fisheries Society 124:726-735.

Marsh, P. C.,, and M. E. Douglas. 1997. Predation by introduced fishes on endangered humpback chub and other native species in the Little Colorado River, Arizona. Transactions of the American Fisheries Society 126:343-346.

Minckley, W. L., and J. E. Deacon. 1968. Southwestern fishes and the enigma of "endangered species". Science 159:1424-1433.

Minckley, W. L., and J. E. Deacon, editors. 1991. Battle against extinction: Native fish management in the American West. University of Arizona Press, Tucson.

Moyle, P. B., H. W. Li, and B. A. Barton. 1986. The Frankenstein effect: Impact of introduced fishes on native fishes in North America. Pages 415-426 *in* R. H. Stroud, editor. Fish Culture in Fisheries Management. American Fisheries Society, Bethesda, Maryland.

Muth, R. T. 1990. Ontogeny and taxonomy of humpback chub, bonytail, and roundtail chub larvae and early juveniles. Doctoral Dissertation. Colorado State University, Fort Collins.

Muth, R. T., L. W. Crist, K. E. LaGory, J. W. Hayse, K. R. Bestgen, T. P. Ryan, J. K. Lyons, and R. A. Valdez. 2000. Flow and temperature recommendations for endangered fishes in the Green River downstream of Flaming Gorge Dam. Final Report FG-53 to the Upper Colorado River Endangered Fish Recovery Program.

Nesler, T. P., R. T. Muth, and A. F. Wasowicz. 1988. Evidence for baseline flow spikes as spawning cues for Colorado squawfish in the Yampa River, Colorado. American Fisheries Society Symposium 5:68-79.

Nolting, D. H. 1957. Miscellaneous survey and management work. Project number 13A, State of Colorado. 4 pp.

Olden, J. D., N. L. Poff, and K. R. Bestgen. In press. Life-history strategies predict fish invasions and extirpations in the Colorado River Basin. Ecological Monographs.

Peterson, J. T., and T. J. Kwak. 1999. Modeling the effects of land use and climate change on riverine smallmouth bass. Ecological Applications 9:1391-1404.

Petts, G. E. 1984. Impounded rivers: perspectives for ecological management. John Wiley and Sons, New York, New York.

Poff, N. L., and others. 1997. The natural flow regime. BioScience 47:769-784.

Propst, D. L., and K. B. Gido. 2004. Responses of native and nonnative fishes to natural flow regime mimicry in the San Juan River. Transactions of the American Fisheries Society 133:922-931.

Pucherelli, M., R. Clark, and R. D. Williams. 1990. Mapping backwater habitat on the Green River as related to the operation of Flaming Gorge Dam using remote sensing and GIS. U. S. Bureau of Reclamation 90 (18):1-11.

Ruppert, J. B., R. T. Muth, and T. P. Nesler. 1993. Predation on fish larvae by adult red shiner, Yampa and Green rivers, Colorado. Southwestern Naturalist 38:397-399.

Smith, G. R. 1966. Distribution and evolution of the North American catostomid fishes of the subgenus *Pantosteus*, genus *Catostomus*. Miscellaneous Publication of the Museum of Zoology, University of Michigan 129:1-132.

Snyder, D. E., and R. T. Muth. 2004. Catostomid fish larvae and early juveniles of the Upper Colorado River Basin-Morphological descriptions, comparisons, and computer-interactive key. Colorado Division of Wildlife, Technical Publication 42, Fort Collins. Contribution 139 of the Larval Fish Laboratory, Colorado State University, Fort Collins.

Speas, D. W., C. J. Walters, D. L. Ward, and R. S. Rogers. 2004. Effects of intraspecific density and environmental variables on electrofishing catchability of brown and rainbow trout in the Colorado River. North American Journal of Fisheries Management 24:586-596.

Stanford, J. A., and J. V. Ward. 1986. Fish of the Colorado system. Pages 353-374 *in* B. R. Davies and K. F. Walker, editors. The ecology of river systems. Dr. W. Junk Publishers, Dordrecht, The Netherlands.

Stanford, J. A., and others. 1996. A general protocol for restoration of regulated rivers. Regulated rivers: research and management, 12:391-413.

Tyus, H. M. 1991. Ecology and management of Colorado squawfish. Pages 379-402 *in* W. L. Minckley and J. E. Deacon, editors. Battle against extinction: Native fish management in the American Southwest. University of Arizona Press, Tucson.

Tyus, H. M., and C. A. Karp. 1991. Habitat use and streamflow needs of rare and endangered fishes, Green River, Utah. U. S. Fish and Wildlife Service, Vernal, Utah.

U. S. Fish and Wildlife Service. 1992. Final biological opinion on the operation of Flaming Gorge Dam. Fish and Wildlife Service, Mountain-Prairie Region, Denver, CO.

Vanicek, C. D., and R. H. Kramer. 1969. Life history of the Colorado squawfish, *Ptychocheilus lucius*, and the Colorado chub, *Gila robusta*, in the Green River in Dinosaur National Monument 1964-1966. Transactions of the American Fisheries Society. 98:193-208.

Vanicek, C. D., R. H. Kramer, and D. R. Franklin. 1970. Distribution of Green River fishes in Utah and Colorado following closure of Flaming Gorge Dam. Southwestern Naturalist 14:297-315.

Ward, J. V. 1989. Riverine-wetland interactions. Pages 385-400 *In* R.R. Sharitz and J.W. Gibbons, editors. Freshwater wetlands and Wildlife. D.O.E. Symposium Series # 61, Oak Ridge, TN.

Ward, J. V., and J. A. Stanford editors 1979. The ecology of regulated streams. Plenum Press, New York.

Ward, J. V., and J. A. Stanford. 1995. Ecological connectivity in alluvial river ecosystems and its disruption by flow regulation. Regulated Rivers: Research and Management 11:105-119.

Table 1. Flow characteristics (mean daily flow in m^3/sec)for the Green River upstream of the Yampa River (Greendale, gauge # 09234500), and the Green River downstream of the Yampa River (Jensen gauge # 09261000). Mean August-February flow is considered the base flow period for the Green River.

Location	Period	Mean daily	Annual low	Annual high	Mean May-June	Mean July-August	Mean Aug-Feb
Greendale	2002	26.9	22.4	112.5	40.5	23.4	25.2
	2003	28.8	21.3	130.0	47.2	22.9	25.9
	2004	30.7	22.5	128.9	43.2	32.6	28.7
	1992-1996	51.4	35.0	110.4	81.0	41.5	47.1
	1997-2004	52.3	35.7	115.8	83.7	43.6	41.8
	1951-1962	59.7	15.0	220.4	160.5	71.0	24.4
Jensen	2002	46.8	22.8	201.7	96.3	24.4	31.9
	2003	80.0	27.2	538.2	253.0	38.4	33.1
	2004	67.3	28.3	322.9	151.8	44.0	44.7
	1992-1996	110.1	47.0	400.6	298.6	79.7	59.5
	1997-2004	107.7	43.5	389.9	279.9	66.1	56.1
	1946-1962	124.5	24.3	469.6	391.9	111.6	39.4

Table 2. Average daily summer (1 June to 30 September, *only 1 June to 14 July in 2004) water temperature (maximum) of the Green and Yampa rivers, Echo Park, Dinosaur National Park, Colorado, 2000 to 2004. The number of days when Green River water temperature was 5°C or more cooler than the Yampa River is also shown. None of those days were in the period when Colorado pikeminnow larvae were drifting from the Yampa River.

	Mean summer temperature°C (maximum)		Number of days difference
Year	Green River	Yampa River	exceeded 5°C
2000	17.0 (20.7)	19.7 (24.1)	2
2001	19.0 (23.4)	20.5 (25.6)	1
2002	18.5 (24.5)	20.4 (25.3)	4
2003	18.7 (23.2)	20.1 (25.9)	0
2004*	19.3 (23.6)	18.1 (21.6)	0

Table 3. Fish community composition (% relative abundance) from electrofishing, seining, trammel and drift nets in the Green River, 2002 – 2004. Electrofishing was conducted in Lodore and Whirlpool canyons. Seining was conducted in Browns Park, Lodore and Whirlpool canyons, Island and Rainbow parks. Most trammel net samples were from Whirlpool Canyon. Drift net samples were from lower Lodore Canyon.

	% Relative Abundance				
Species	Electrofishing	Seine	Trammel	Drift	Total
mountain whitefish	1.0	1.7	0	0.2	1.7
humpback chub	<0.1	0	0.6	0	<0.1
bonytail	0.7	<0.1	0.7	0	<0.1
roundtail chub	0.5	0.2	24.0	0.1	0.3
Colorado pikeminnow	0.3	<0.1	0.3	0	<0.1
speckled dace	0.2	1.0	0	1.9	1.0
bluehead sucker	18.2	1.3	12.9	14.4	2.5
flannelmouth sucker	24.9	3.1	27.4	16.8	4.7
mottled sculpin	0.9	0.1	0	0	0.1
cutthroat trout	<0.1	0	0	0	<0.1
rainbow trout	2.1	<0.1	0.3	0	0.1
cutthroat X rainbow trout	<0.1	0	0	0	<0.1
brown trout	12.6	<0.1	0	0	0.7
red shiner	0.7	43.6	0	42.0	41.1
common carp	11.3	0.3	0.7	0.5	0.8
fathead minnow	0	10.4	0	0.4	9.6
sand shiner	<0.1	26.2	0	12.5	24.4
redside shiner	<0.1	6.7	0	<0.1	6.2
creek chub	0	<0.1	0	0	<0.1
non-native cyprinids	0	0	0	1.7	<0.1
unidentified minnow	0	<0.1	0	0	<0.1
white sucker	5.4	4.4	1.9	6.3	4.5
bluehead X white sucker	0.4	<0.1	0.2	0	<0.1
flannelmouth X bluehead sucker	0.9	<0.1	0.1	0	0.1
flannelmouth X white sucker	1.9	0.1	1.2	0	0.2
flannelmouth X bluehead X white sucker	0.1	0	0	0	<0.1
razorback X flannelmouth sucker	<0.1	0	0	0	<0.1
razorback sucker hybrid	0	<0.1	0	<0.1	<0.1
unidentified sucker	0	0.1	0	0.7	0.1
black bullhead	<0.1	<0.1	0	0	<0.1
channel catfish	10.2	<0.1	27.9	0	0.7
northern pike	0.1	<0.1	0.1	0	<0.1
plains killifish	0	<0.1	0	0	<0.1
brook stickleback	0	<0.1	0	0	<0.1
green sunfish	0.2	0.6	0	0.2	0.5
bluegill	<0.1	<0.1	0	0	<0.1
green sunfish X bluegill	<0.1	0	0	0	<0.1
smallmouth bass	7.0	0.2	0.8	2.3	0.6
black crappie	0.1	<0.1	0.7	0	<0.1
walleye	<0.1	0	0.1	0	<0.1
Total fish	10725	193264	1206	4269	209464
Effort (hours or samples)	151.9	800	535.5	420	
# native species	9	8	6	5	9
% native species	46.8	7.4	66	33.3	10.3
# non-native species	16	18	8	8	20
% non-native species	49.8	92.4	32.5	65.9	89.3
# hybrid combinations	7	4	3	1	9
% hybrid combinations	3.4	0.2	1.5	<0.1	0.3

Table 4. Relative abundance (percent composition) of fishes in six reaches of the Green River in Lodore (reaches LD1 to LD4) and Whirlpool (WH1 and 2) canyons, captured by electrofishing, 2002-2004.

	Lodore Canyon				Whirlpool Canyon	
	LD1	LD2	LD3	LD4	WH1	WH2
RK=	585.9-579.0	578.9-571.0	570.9-563.0	562.9-555.3	555.2-546.6	546.5-538.1
Species	(N=1019)	(N=1799)	(N=1219)	(N=1900)	(N=2116)	(N=2672)
mountain whitefish	1.3	1.4	1.9	2.1	0.4	0.1
roundtail chub	0.1	0	0.3	0.2	1.7	0.4
bonytail	0	0	0	0	3.4	0.2
humpback chub	0	0	0	0	<0.1	0.0
Colorado pikeminnow	0.4	0.2	0.3	0.4	0.1	0.5
speckled dace	0.8	0.2	0.2	0.3	0.1	0.1
bluehead sucker (BH)	7.7	17.2	12.5	10.9	21.7	28.0
flannelmouth sucker (FM)	20.4	22.1	22.8	26.3	29.4	24.8
mottled sculpin	1.1	0.6	1.4	1.8	0.2	0.6
brown trout	22.9	32.7	27.9	6.8	1.0	1.5
rainbow trout	2.8	1.6	0.8	2.1	0.9	3.8
cutthroat X rainbow trout	0.1	0	0	0.1	0	0.1
northern pike	0	0	0.1	0.5	0.1	0.1
walleye	0	0	0	0	0.1	0.1
red shiner	0	0	0	0.6	0.1	2.2
common carp	11.1	5.4	11.6	18.1	10.6	10.8
sand shiner	0	0	0	0	<0.1	0
redside shiner	0.3	0	0	0.1	0	0
white sucker (WS)	15.8	8.1	6.3	6.5	2.3	1.0
BH X WS	0.6	0.3	0.1	0.3	0.7	0.2
FM X BH	1.7	1.8	0.4	0.4	0.6	1.0
FM X WS	4.3	2.6	2.3	2.2	1.4	0.6
FM X BH X WS	0.1	0	0	0.1	0.2	0
RZ X FM	0	0	0	0	<0.1	0.1
black bullhead	0	0	0.1	0.1	0	<0.1
channel catfish	8.3	4.9	6.4	12.9	13.8	11.5
green sunfish	0.1	0.1	0.1	0.3	0.2	0.1
bluegill	0	0	0.1	0	0	0
green sunfish X bluegill	0	0	0	0.1	0	0
black crappie	0	0	0	0.2	0.3	<0.1
smallmouth bass	0.2	1.0	4.5	6.8	10.6	11.9

Table 5. Comparison of Green River fish community species composition and rank order of abundance for fishes captured by electrofishing in periods 1994 to 1996 and 2002 to 2004 in four reaches of Lodore Canyon (LD1 to LD4), and for Whirlpool Canyon (WH1 and WH2) in 2002 to 2004. FM = flannelmouth sucker, BT = brown trout, CP = common carp, BH = bluehead, RDS = redside shiner, WS = white sucker, CC = channel catfish, MWF = mountain whitefish, RS = red shiner, and SMB = smallmouth bass.

	Attribute					
	Species richness					
Reach and year	Native fishes	Non-native fishes	Hybrid combinations	Five most common taxa, descending order	% of five most common	% native taxa
LD1, 1994-1996	7	8	4	FM, BT, CP, BH, RDS	72	51
LD1, 2002-2004	7	9	4	BT, FM, WS, CP, CC	79	32
LD2, 1994-1996	7	7	4	FM, BT, BH, MWF, CP	87	62
LD2, 2002-2004	6	7	3	BT, FM, BH, WS, CP	86	42
LD3, 1994-1996	7	8	4	FM, BT, BH, CP, MWF	88	56
LD3, 2002-2004	7	10	3	BT, FM, BH, CP, CC	81	39
LD4, 1994-1996	8	11	5	FM, BH, CP, BT, RS	87	63
LD4, 2002-2004	7	12	6	FM, CP, CC, BH, SMB, BT*	82	42
WH1, 2002-2004	9	12	5	FM, BH, CC, SMB, CP	86	57
WH2, 2002-2004	9	12	5	BH, FM, SMB, CC, CP	87	55

Species abbreviations:
*, SMB and BT tied

Table 6. Composition of the Green River fish community in eight reaches, 2002-2004, from seine sampling.

Species	RK=	Browns Park 613.0-586.0 (N=25178)	Lodore Canyon LD1 585.9-579.0 (N=7896)	Lodore Canyon LD2 578.9-571.0 (N=5114)	Lodore Canyon LD3 570.9-563.0 (N=18181)	Lodore Canyon LD4 562.9-555.3 (N=22951)	Whirlpool Canyon WH1 555.2-546.6 (N=52558)	Whirlpool Canyon WH2 546.5-538.1 (N=34763)	Island and Rainbow Parks 538.0-528.0 (N=26623)
mountain whitefish		10.5	1.3	0.4	2.8	0.2	<0.1	0	0.1
roundtail chub		0	<0.1	0	<0.1	<0.1	0.4	0.2	0.3
bonytail		<0.1	0	0	0	0	0	0	0
Colorado pikeminnow		0	0	0	0	0	<0.1	0	0.1
speckled dace		0.7	3.4	6.5	3.3	0.2	0.5	0.2	0.6
bluehead sucker (BH)		3.2	2.2	1.6	0.9	0.8	1.1	0.6	1.1
flannelmouth sucker (FM)		6.9	12.3	12.2	3.3	1.5	1.3	1.7	2.1
razorback sucker (RZ)		0	0	0	0	0	0	0	0
mottled sculpin		0.1	0.1	0.1	0.3	<0.1	<0.1	0	<0.1
brown trout		<0.1	<0.1	<0.1	<0.1	0	0	0	0
northern pike		0	0	0	<0.1	<0.1	0	0	0
red shiner		0.7	4.7	16.0	57.0	68.6	50.1	54.4	43.5
common carp		0.6	2.0	2.8	0.2	0.1	<0.1	<0.1	<0.1
sand shiner		<0.1	0.8	0.5	14.4	17.6	39.7	34.1	41.8
fathead minnow		30.4	31.3	24.6	12.7	6.2	1.6	5.9	7.8
redside shiner		29.2	29.9	7.1	1.4	0.6	3.7	0.9	1.0
creek chub		0	0	0	<0.1	<0.1	0	0	0
unidentified minnow		<0.1	0	0	0	0	<0.1	0	0
white sucker (WS)		17.0	10.8	16.1	2.8	2.3	1.2	1.6	1.2
BH X WS		0.1	<0.1	<0.1	<0.1	<0.1	<0.1	0	0
FM X BH		<0.1	0	0	0	0	<0.1	<0.1	<0.1
FM X WS		0.3	0.7	0.9	0.1	0.1	0.1	<0.1	<0.1
RZ X FM		0	0	0	0	0	0	0	0
unidentified sucker		0.3	0.1	<0.1	0.1	0.1	<0.1	<0.1	<0.1
plains killifish		0	0	0	0	0	<0.1	0	0
black bullhead		0	0	0.1	<0.1	0.1	<0.1	<0.1	<0.1
channel catfish		0	0	0	0	<0.1	<0.1	<0.1	<0.1
brook stickleback		0	0	0	0	<0.1	<0.1	0	<0.1
green sunfish		<0.1	<0.1	11.0	0.6	1.2	0.1	0.1	0.3
black crappie		0	0	0	0	0	<0.1	0	0
bluegill		0	0	0	0	<0.1	0	0	0
smallmouth bass		0	0.1	0.2	0.1	0.3	0.2	0.3	0.3

Table 7. Comparison of Green River fish community species composition and rank order of abundance for fishes captured by seining in periods 1994 to 1996 and 2002 to 2004 in Browns Park (BP), four reaches of Lodore Canyon (LD1 to LD4), Island-Rainbow Park (IRP), and Whirlpool Canyon (WH1 and WH2) in 2002 to 2004. FM = flannelmouth sucker, BT = brown trout, CP = common carp, BH = bluehead, RDS = redside shiner, WS = white sucker, CC = channel catfish, MWF = mountain whitefish, RS = red shiner, and SMB = smallmouth bass.

	Attribute					
		Species richness				
Reach and year	Native fishes	Non-native fishes	Hybrid combinations	Five most common taxa, descending order	% of five most common	% native taxa
BP1, 1994-1996	5	6	3	FH, BH, WS, FM, RDS	96	33
BP1, 2002-2004	6	8	3	FH, RDS, WS, MWF, FM	94	21
LD1, 1994-1996	4	5	3	FM, BH, WS, RDS, FH	94	71
LD1, 2002-2004	6	10	2	FH, RDS, FM, WS, RS	94	19
LD2, 1994-1996	5	5	0	SD, FM, BH, WS, RDS	98	94
LD2, 2002-2004	5	11	2	FH, WS, RS, FM, GS	80	21
LD3, 1994-1996	4	3	0	FM, SD, BH, WS, MS	99	94
LD3, 2002-2004	6	13	2	RS, SS, FH, SD, FM	91	11
LD4, 1994-1996	5	8	2	RS, FH, RDS, SD, FM	88	17
LD4, 2002-2004	6	15	2	RS, SS, FH, WS, FM	96	3
WH1, 2002-2004	7	15	3	RS, SS, RDS, FH, FM	96	4
WH2, 2002-2004	4	11	2	RS, SS, FH, FM, WS	98	3
IRP, 1994-1996	7	11	2	RS, FH, SS, RDS, FM	97	2
IRP, 2002-2004	7	12	2	RS, SS, FH, FM, WS	96	4

Species abbreviations:

*, SMB and BT tied

Table 8. Mean catch-per-unit-effort abundance (fish captured per h of electrofishing, 95% confidence limit below) of native and selected non-native fishes captured in the Green River, Lodore Canyon, in 1994 to 1996 and 2002 to 2004 periods. The chi-square and *p*-values test the hypothesis that the means are statistically significantly different between the two periods.

Species	1994 to 1996 mean 95%CI	2002 to 2004 mean 95%CI	*Chi-square*	*p-value*
Native fishes				
mountain whitefish	4.2 3.10 to 5.33	1.1 0.70 to 1.57	28.12	< 0.0001
Colorado pikeminnow	0.62 0.25 to 0.99	0.39 0.08 to 0.69	0.97	0.325
roundtail chub	0.3 0.15 to 0.52	0.1 0.05 to 0.23	9.32	0.0023
speckled dace	1.4 0.73 to 2.07	0.3 0.12 to 0.39	36.54	< 0.0001
bluehead sucker	16.6 12.69 to 20.58	10.1 8.02 to 12.23	6.88	0.0087
flannelmouth sucker	28.5 22.99 to 34.02	17.9 15.41 to 20.35	12.27	0.0005
mottled sculpin	0.5 0.22 to 0.75	0.9 0.39 to 1.49	5.65	0.0174
Nonnative fishes				
brown trout	15.0 11.55 to 18.48	16.4 13.45 to 19.25	0.29	0.5926
northern pike	0.2 0.02 to 0.32	0.2 0.07 to 0.25	0.11	0.7453
white sucker	2.2 1.54 to 2.87	5.6 4.30 to 6.9	19.27	< 0.0001
channel catfish	1.3 0.61 to 1.95	6.1 4.71 to 7.51	50.98	< 0.0001
smallmouth bass	0.0 -0.03 to 0.10	2.4 1.76 to 3.13	80.34	< 0.0001

Table 9. Composition of fish sampled in drift net collections from the 1994 to 1995 period compared with that from 2002 to 2004 in the Green River just upstream of the Yampa River. N = the number of samples collected each year.

Species	1994 6/28 to 8/5 (n = 99)	1995 8/8 to 8/15 (n = 24)	2002 6/18 to 8/4 (n = 168)	2003 7/9 to 8/18 (n = 150)	2004 7/11 to 8/11 (n = 102)
mountain whitefish				2	6
roundtail chub	1			3	
speckled dace	72	1	41	27	11
bluehead sucker	148	22	58	493	63
flannelmouth sucker	81	3	75	455	189
razorback sucker hybrid				1	1
red shiner			1457	193	142
common carp				6	14
fathead minnow				8	7
sand shiner	1		41	385	109
redside shiner				2	
nonative cyprinids			4	63	5
white sucker	1	2	14	167	89
unidentified suckers	3		2	15	12
green sunfish				10	
smallmouth bass				6	93
total fish captured	307	28	1692	1836	741
% composition, native fishes	98.4	92.9	10.3	53.4	36.4

2003 RZB hybrid LFL# 87325
2004 RZB hybrid LFL# 91501

Table 10.--Composition and status of the fish community of the Green River upstream of the Yampa River in pre-dam and four post-impoundment sampling periods (Gaufin et al. 1960, Banks 1964, Smith 1966, Vanicek et al. 1970, Holden and Crist 1981). An X = present, R = reproduction, ? indicates uncertainty. N = native, I = introduced.

		Pre-impoundment	Post-impoundment			
	Status	1959-1960	1967-1978	1978-1980	1994-1996	2002-2004
Mountain whitefish	N	R		R	R	R
Humpback chub	N	R				
Bonytail	N	R?				X
Roundtail chub	N	R	X	R	R	R
Colorado pikeminnow	N	R	X	X	X	R?
Speckled dace	N	R	X	R	R	R
Bluehead sucker	N	R	X	R	R	R
Flannelmouth sucker	N	R	X	R	R	R
Mountain sucker	N	R		R		
Razorback sucker	N	R?	X	X	X	
Mottled sculpin	N	R		R	R	R
Cutthroat trout	I			X	X	X
Cutthroat x rainbow	I				X	X
Rainbow trout	I		X	X	X	X
Brown trout	I		X	R	R	R
Northern pike	I				X	R
Red shiner	I			R	R	R
Common carp	I	R	R	R	R	R
Utah chub	I		X	X		
Fathead minnow	I	R	R	R	R	R
Sand shiner	I			R	R	R
Redside shiner	I	R	R	R	R	R
Creek chub	I	X		X		R
White sucker	I	X?		R	R	R
WS x FM				X	X	X
FM x BH					X	X
WS x BH					X	X
RZB x FM		X		X	X	X
WS x FM x BH					X	X
Black bullhead	I	R	X	X	X	R
Channel catfish	I	R?	X	X	X	X
Brook stickleback	I					R
Black crappie	I					X
Bluegill	I					X
Green sunfish	I				R?	R
Green sunfish x bluegill	I					X
Smallmouth bass	I				X	R

Table 11.—Comparison of relative abundance (percent composition) of the fish community during 1978-1980 (Holden and Crist 1981), 1994-1996 (Bestgen and Crist 2000), and 2002-2004 (this study) in two reaches of the Green River in Lodore Canyon. Fishes were captured by seining and electrofishing.

	Upper Lodore Canyon			Lower Lodore Canyon		
Species	1978-80	1994-96	2002-04	1978-80	1994-96	2002-04
mountain whitefish	<0.1	1.5	1.6	0	0.6	0.5
roundtail chub	<0.1	0	0.1	3.5	0.1	<0.1
Colorado pikeminnow	0	<0.1	<0.1	<0.1	0.1	<0.1
speckled dace	24.5	4.9	2.6	18.8	5.7	0.4
bluehead sucker (BH)	20.8	13.1	2.1	3.1	4.6	1.6
flannelmouth sucker (FM)	45.9	43.3	11.9	5.4	9.6	3.4
mountain sucker	1.0	0	0	<0.1	0	0
razorback sucker (RZ)	0	0	0	<0.1	0	0
mottled sculpin	0	0.3	0.2	1.1	0.3	0.2
brown trout	0.3	2.1	3.0	0.6	1.0	0.5
cutthroat trout	0.3	0.1	<0.1	0.3	0.1	0
rainbow trout	1.8	0.2	0.3	1.2	0.2	<0.1
cutthroat X rainbow trout	0	0	0	0	0	0
northern pike	0	0	0	0	<0.1	<0.1
red shiner	0.1	0	5.1	38.0	27.4	61.5
common carp	1.3	1.3	3.4	3.2	1.9	1.1
Utah chub	0.2	0	0	0	0	0
sand shiner	0	0	0.9	0.2	0.9	17.0
fathead minnow	3.2	5.8	29.9	3.0	29.0	7.3
redside shiner	0.3	12.2	26.3	20.8	13.6	0.6
creek chub	0	0	0	<0.1	0	<0.1
white sucker (WS)	0.1	12.6	9.7	0.3	3.7	2.0
BH X WS	0	0.8	0.1	0	0.1	<0.1
FM X BH	0	0.7	0.2	0	0.2	<0.1
FM X WS	<0.1	0.5	1.0	0	<0.1	0.2
FM X BH X WS	0	0	<0.1	0	<0.1	<0.1
RZ X FM	0	0	0	<0.1	<0.1	0
unidentified sucker	0	0.4	0.1	0	0.3	0.2
black bullhead	0	0	0	<0.1	0	<0.1
channel catfish	<0.1	0.1	1.2	0.3	0.6	0.5
brook stickleback	0	0	0	0	0	0
green sunfish	0	<0.1	0.1	0	<0.1	2.2
bluegill	0	0	0	0	0	0
green sunfish X bluegill	0	0	0	0	0	<0.1
black crappie	0	0	0	0	0	0
smallmouth bass	0	0	0.1	0	<0.1	0.6
Total fish captured	2296	2713	6967	4270	6592	12220
Percent native fishes	92	63	19	32	21	6

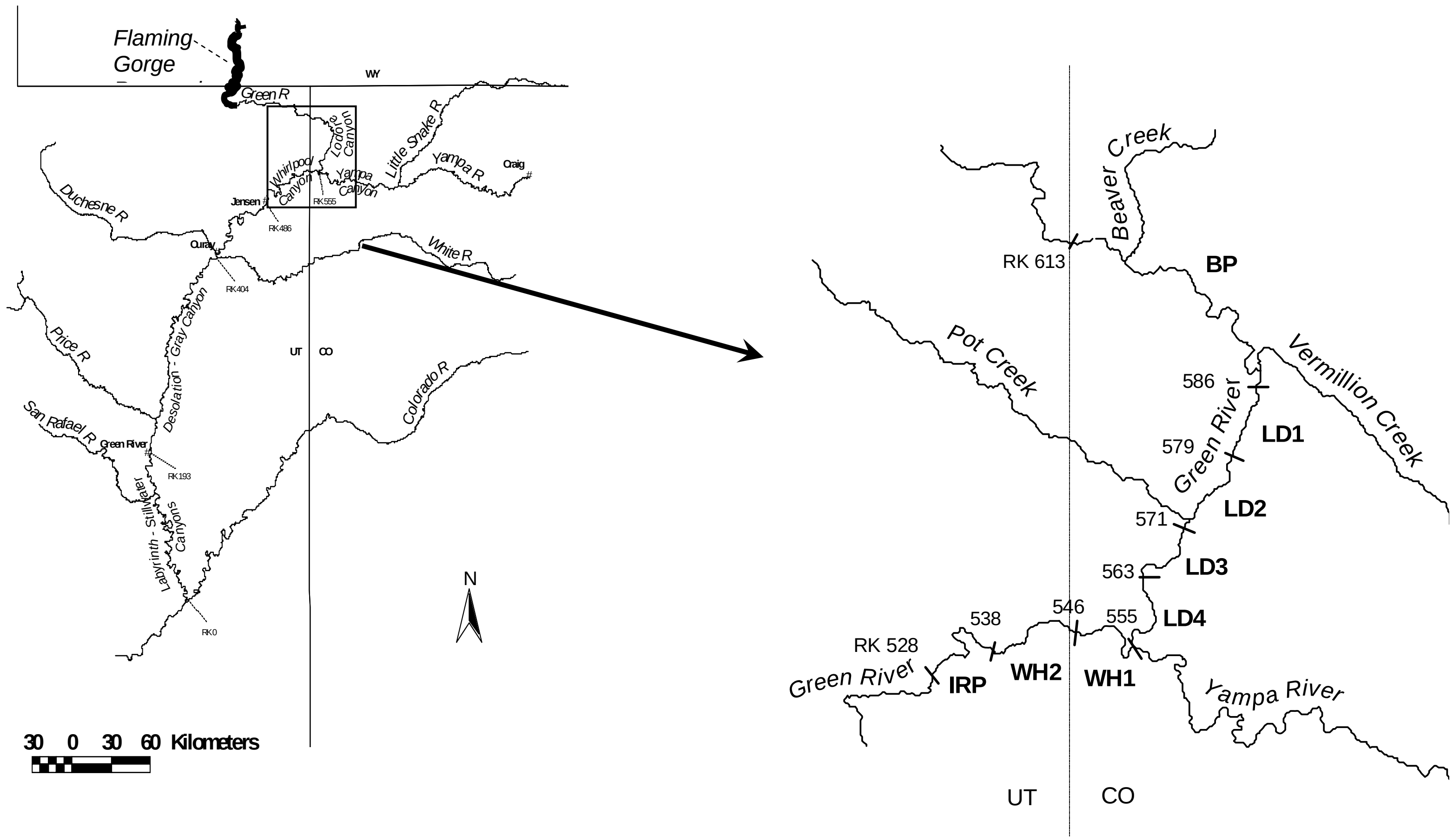


Figure 1. Map of study area. Reach BP is Browns Park, reaches LD1 to LD4 correspond to Lodore Canyon reaches 1 to 4, WH1 and WH2 are Whirlpool Canyon reaches 1 and 2, and IRP is the Island-Rainbow Park reach.

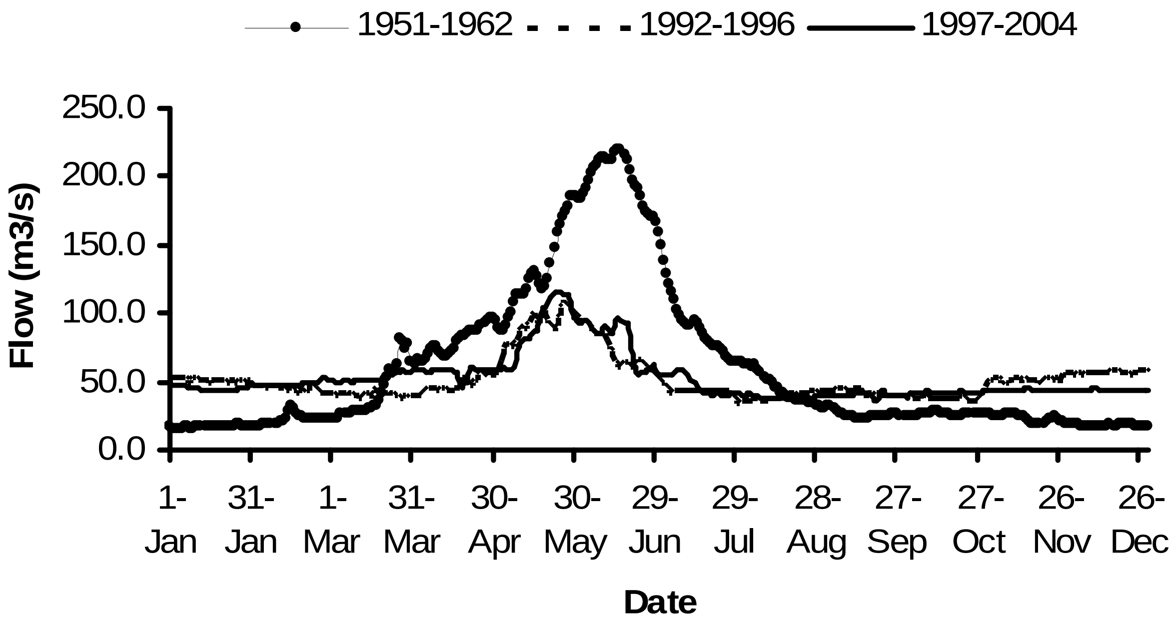


Figure 2. Mean daily flows for the Green River upstream of the Yampa River in the pre-Flaming Gorge Dam period (1951 to1962, Greendale gauge # 09234500), compared to post- regulation 1992 to 1996 and 1997 to 2004 periods.

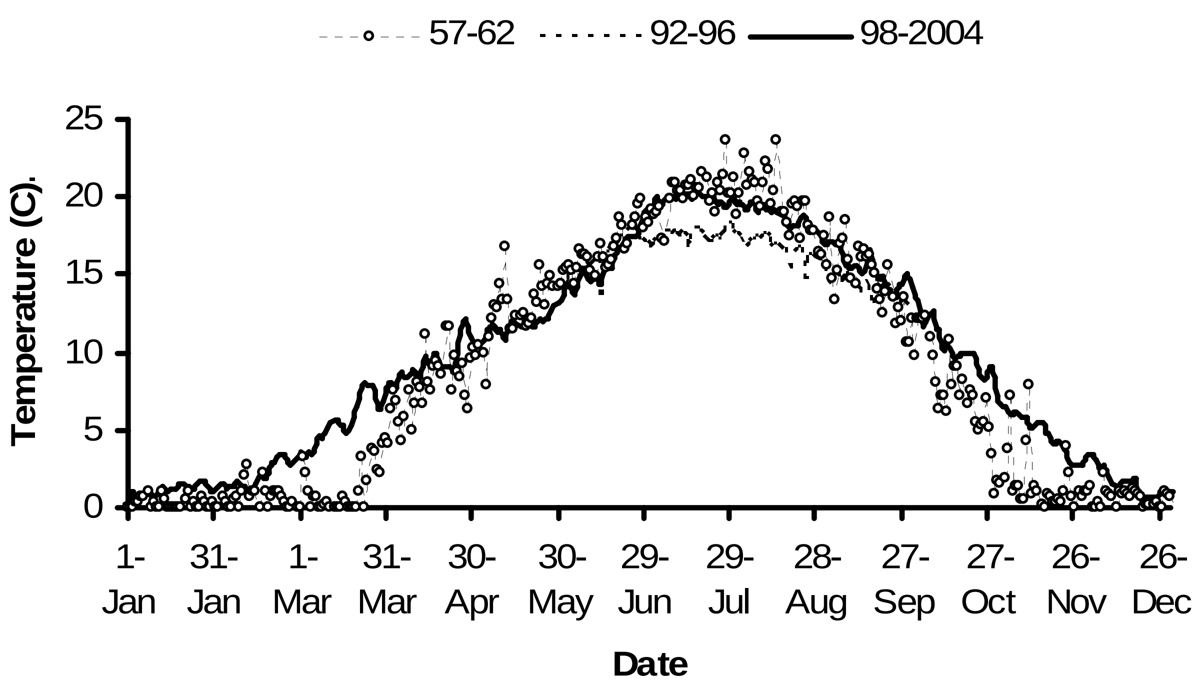


Figure 3. Mean daily water temperatures for the Green River upstream of the Yampa River in the pre-Flaming Gorge Dam period (1957 to1962, Greendale gauge # 09234500), compared to post-1992 Biological Opinion (1992-1996, only June through Sept. portrayed), and recent (1998 to 2004) periods.

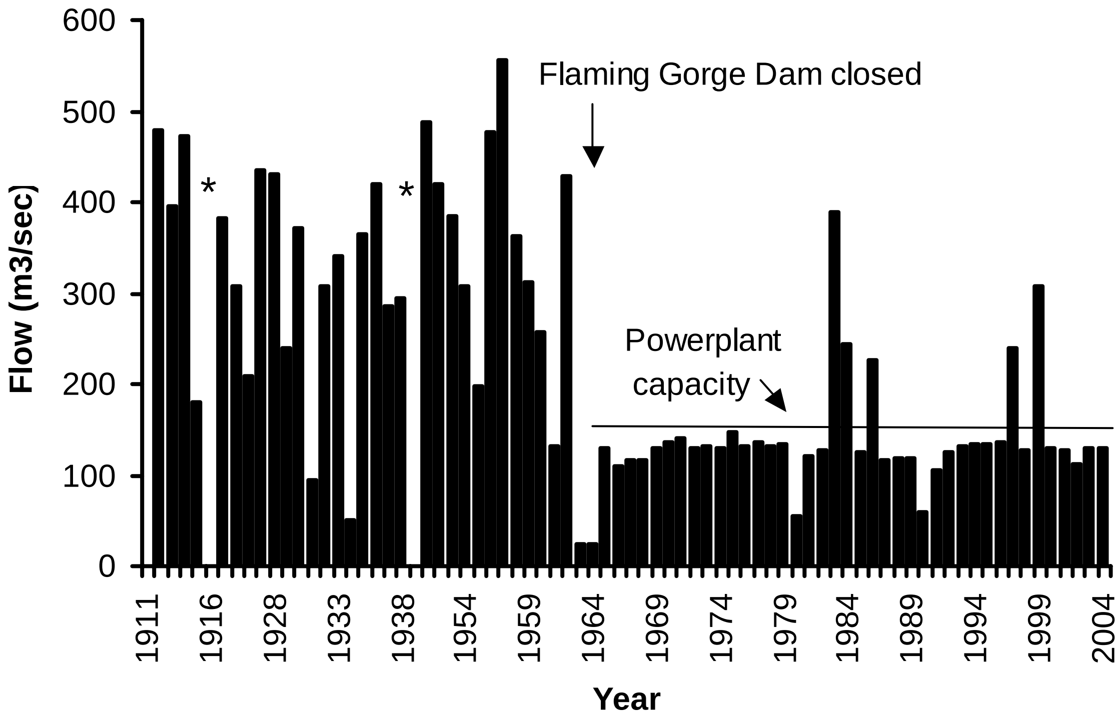


Figure 4. Variation in annual peak flow maxima (mean daily) for the Green River near present-day Flaming Gorge Dam in pre-impoundment (1911 to 1962) and post-impoundment (1963 to 2004) periods. Discharge records are from the following localities: near Bridgeport, Utah (upper Browns Park, 1912 to 1915, gauge # 09235000); near Linwood, Utah (1924 to 1938, submerged by Flaming Gorge Reservoir, gauge # 09230500); near Greendale, Utah (1951 to 2004, gauge # 09234500), asterisks denote breaks in record.

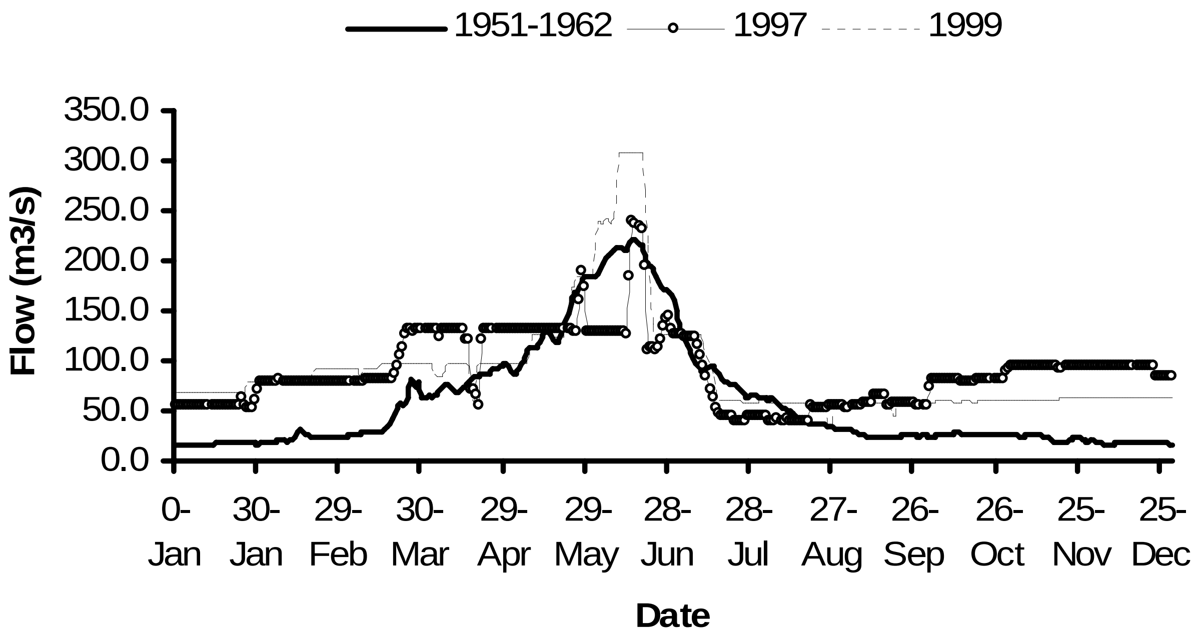


Figure 5. Mean daily flows for the Green River upstream of the Yampa River in post-Flaming Gorge Dam in years1997 and 1999 years compared to mean daily flows in the pre-regulation period (1951 to1962, Greendale gauge # 09234500).

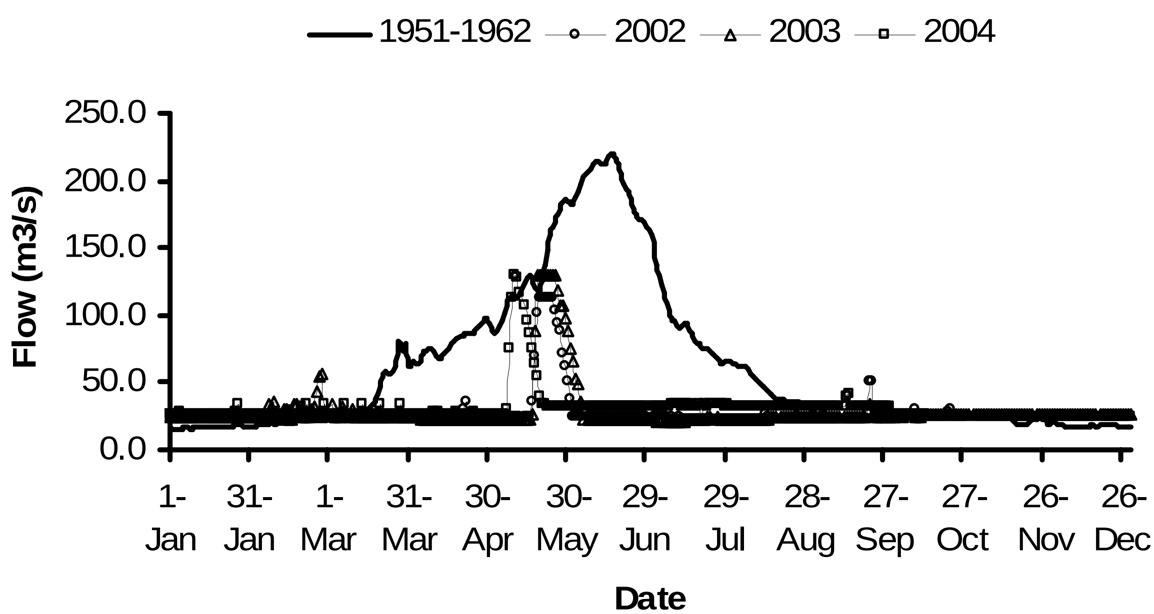


Figure 6. Mean daily flows for the Green River upstream of the Yampa River in the pre-Flaming Gorge Dam period (1951 to1962, Greendale gauge # 09234500), compared to 2002 to 2004 hydrographs.

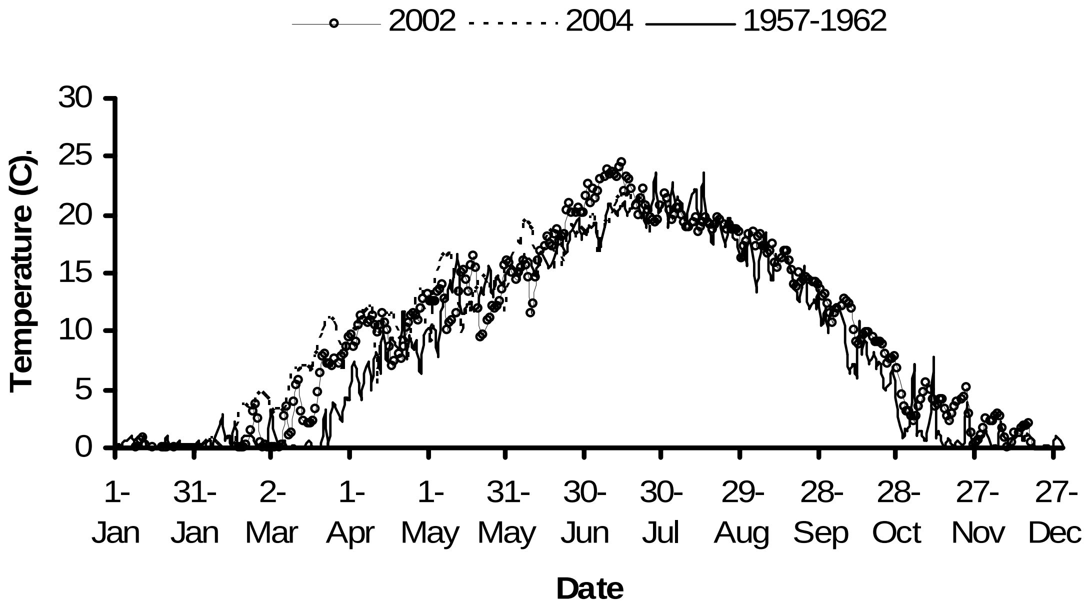


Figure 7. Mean daily temperatures for the Green River upstream of the Yampa River in the pre-Flaming Gorge Dam period (1957 to1962, Greendale gauge # 09234500), compared to study years 2002 and 2004 (the similar 2003 pattern not shown) from gauge records in lower Lodore Canyon.

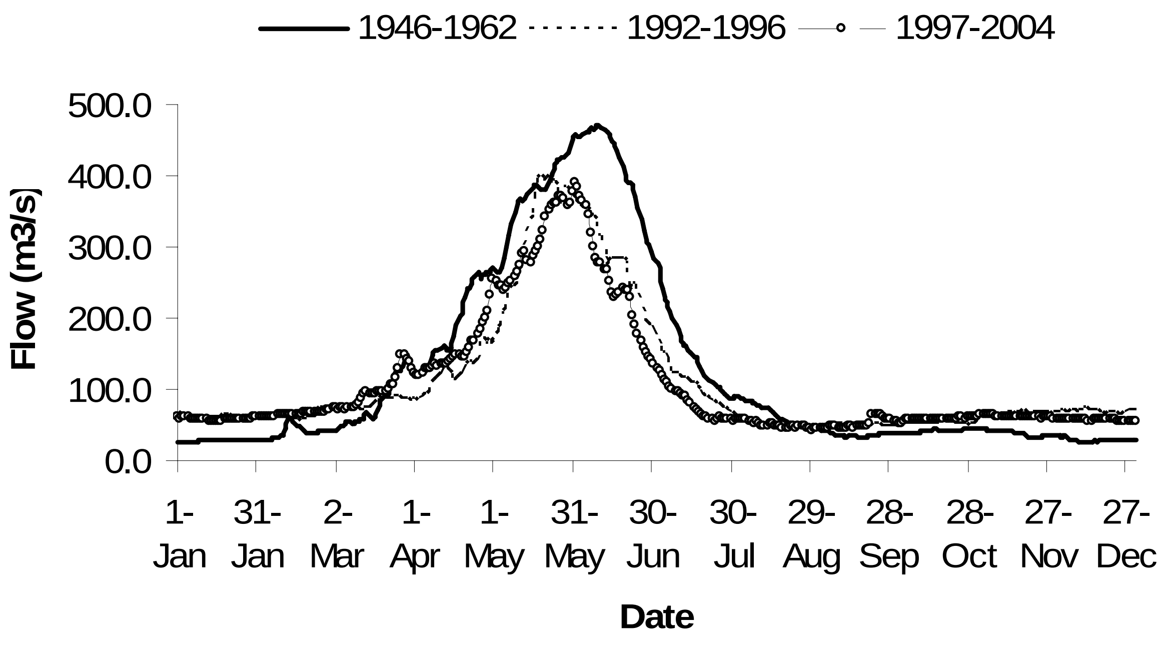


Figure 8. Mean daily flows for the Green River in Whirlpool Canyon downstream of the Yampa River in the pre-Flaming Gorge Dam period (1946 to1962, Jensen gauge # 09261000), compared to post-regulation 1992 to 1996 and 1997 to 2004 periods.

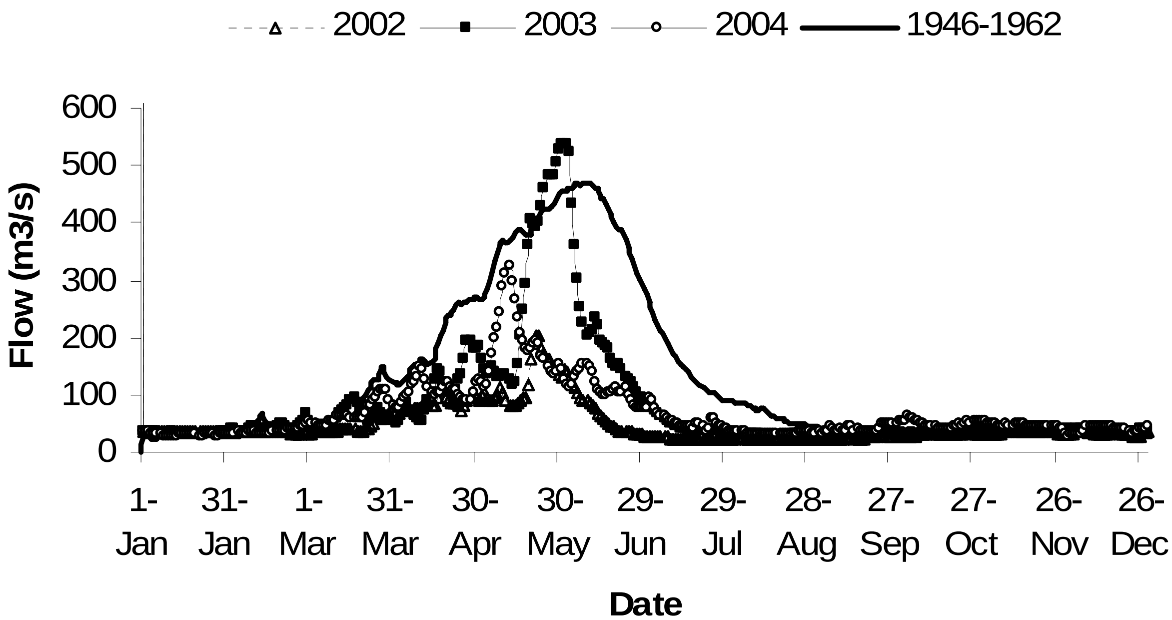


Figure 9. Mean daily flows for the Green River in Whirlpool Canyon downstream of the Yampa River in the pre-Flaming Gorge Dam period (1946 to1962, Jensen gauge # 09261000), compared to post-regulation daily flows in the 2002 to 2004 study period.

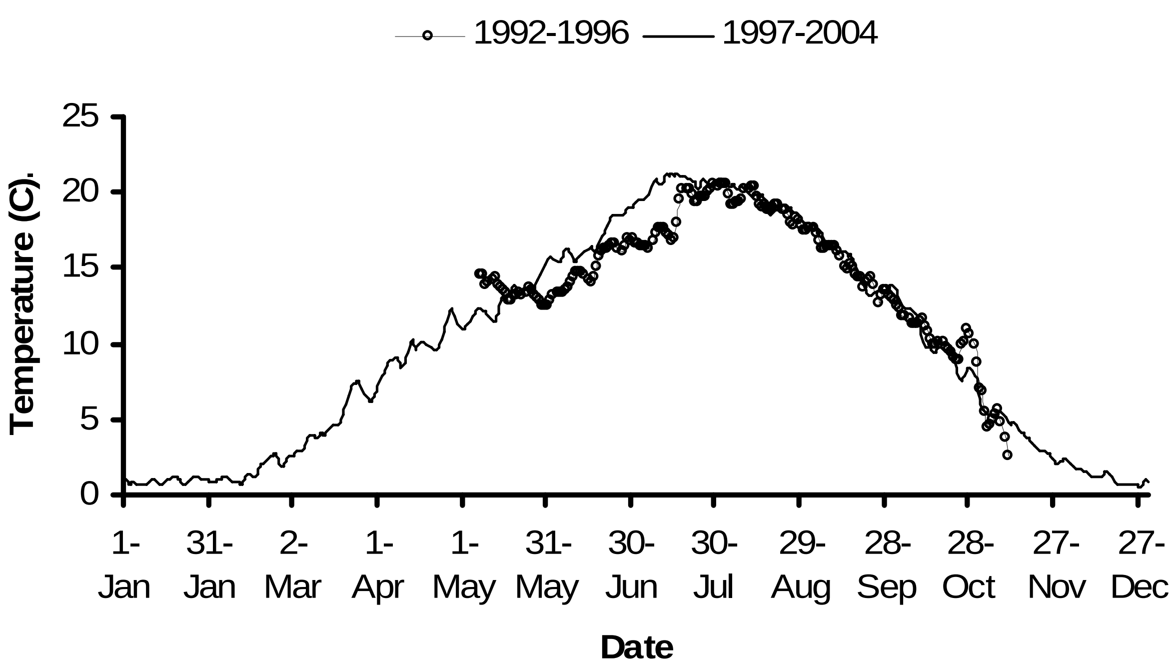


Figure 10. Mean daily water temperatures for the Green River in Whirlpool Canyon downstream of the Yampa River in the post-1992 Biological Opinion (1992-1996, partial year only available), and recent (1997 to 2004) periods.

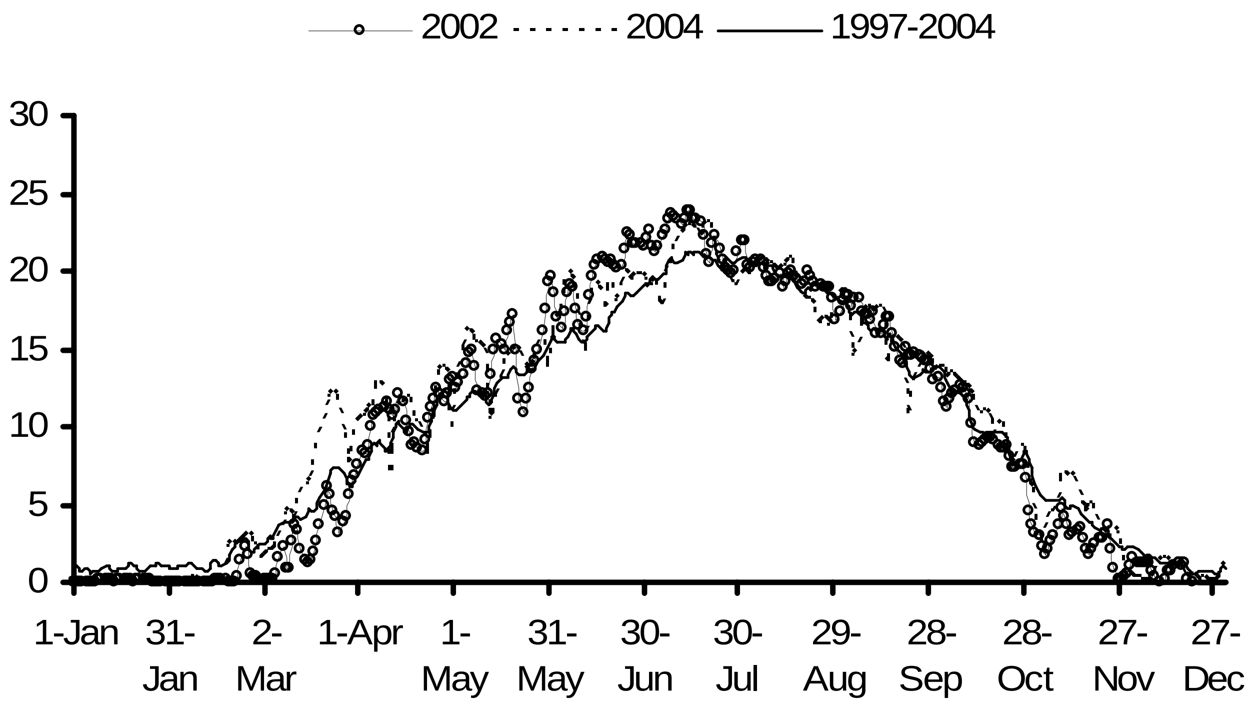
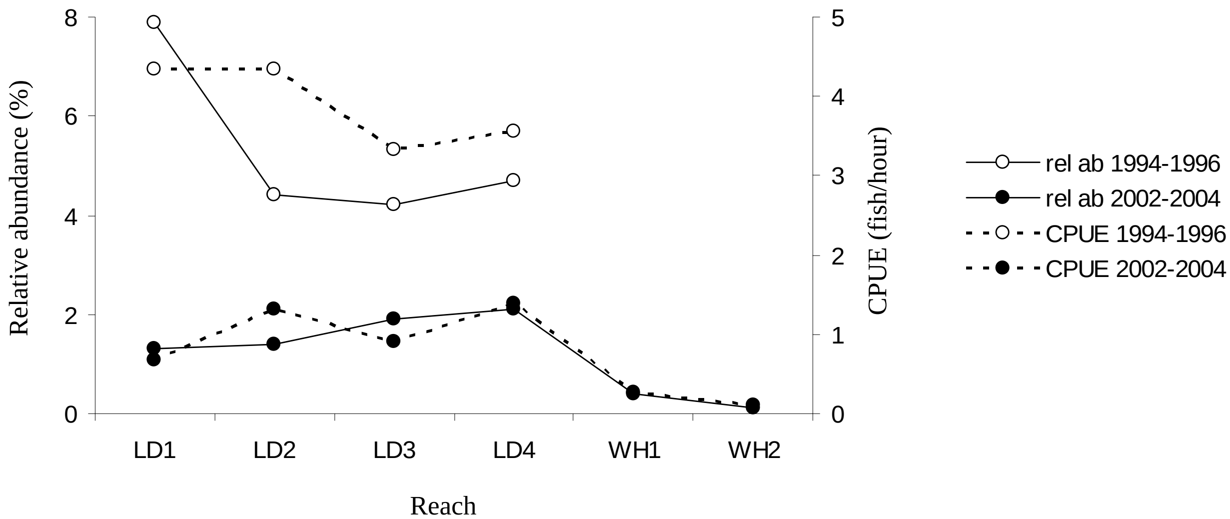


Figure 11. Mean daily water temperatures for the Green River in Whirlpool Canyon downstream of the Yampa River from 1997 to 2004 compared to daily water temperatures in study years 2002 and 2004 (similar 2003 pattern not shown for clarity).

A. electrofishing



B. seine

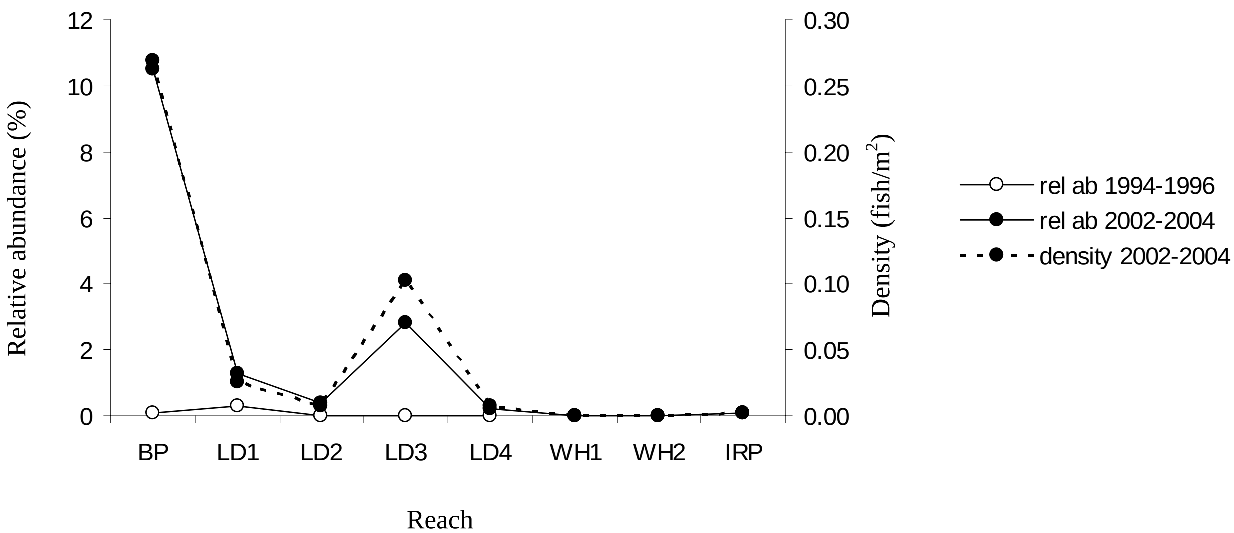


Figure 12.—Distribution and abundance of mountain whitefish in electrofishing and seine samples from the Green River downstream of Flaming Gorge Dam, Colorado and Utah, in periods 1994-1996 and 2002-2004. Only relative abundance data are available for seine samples in the 1994-1996 period; CPUE is number of fish captured per hour of electrofishing. Sampling areas include Browns Park (BP, RK 613.0-586.0), four Lodore Canyon reaches (LD1, RK 585.9 to 579.0; LD2, RK 578.9 to 571.0, LD3, RK 570.9 to 563.0; LD4, RK 562.9 to 555.3), two Whirlpool Canyon reaches (WH1, RK 555.2 to 546.6; WH2, RK 546.5 to 538.1), and Island-Rainbow Park (IRP, RK 538.0-528.0).

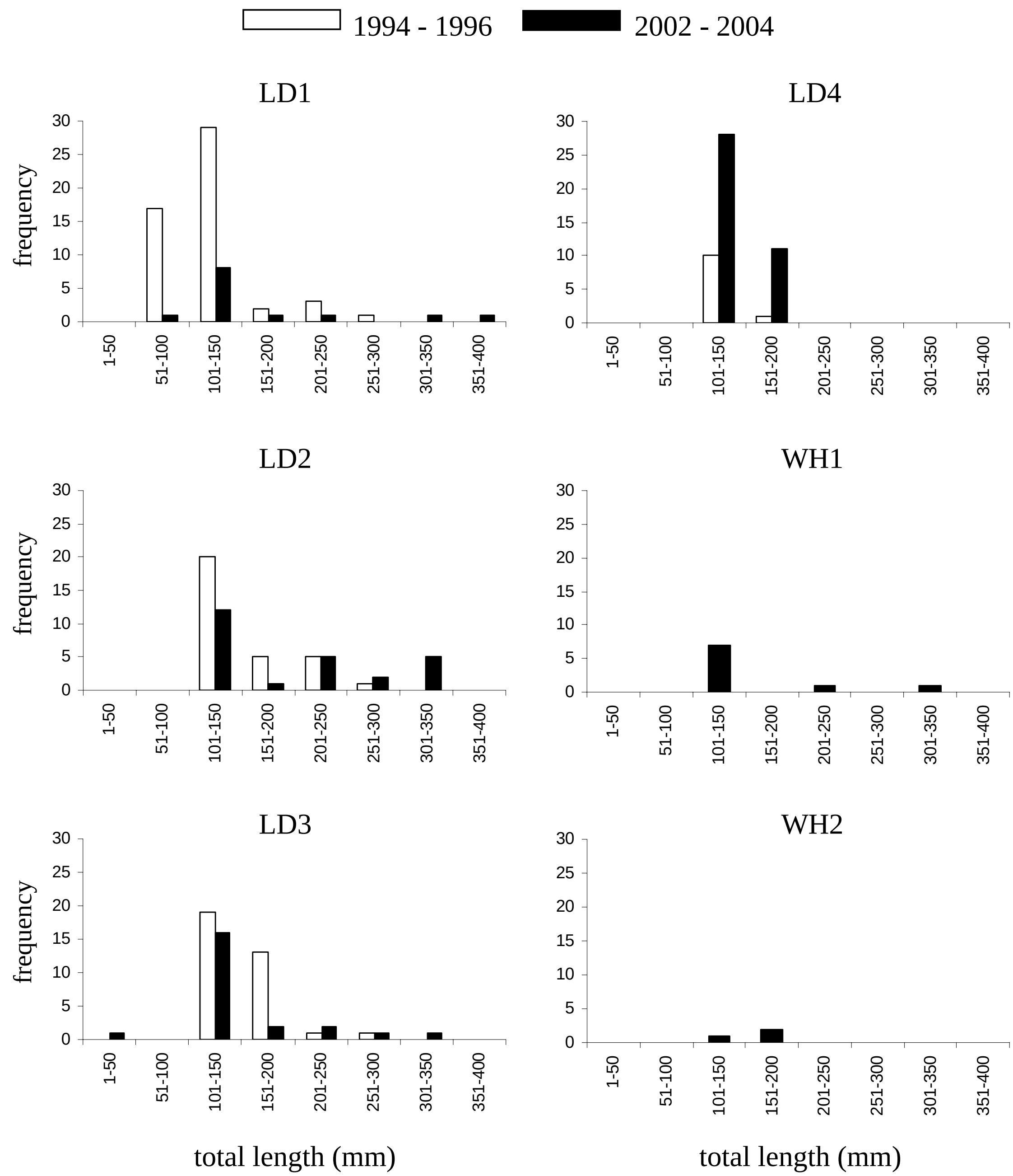


Figure 13.—Length-frequency histograms for mountain whitefish from four reaches of Lodore Canyon (LD1-LD4, 1994-1996 and 2002-2004) and two reaches of Whirlpool Canyon (WH1-WH2, 2002-2004), Green River. Reach LD1 = RK 585.9 to 579.0, LD2 = RK 578.9 to 571.0, LD3 = RK 570.9 to 563.0, LD4 = RK 562.9 to 555.3, WH1 = RK 555.2 to 546.6, WH2 = RK 546.5 to 538.1. Fish were captured by electrofishing.

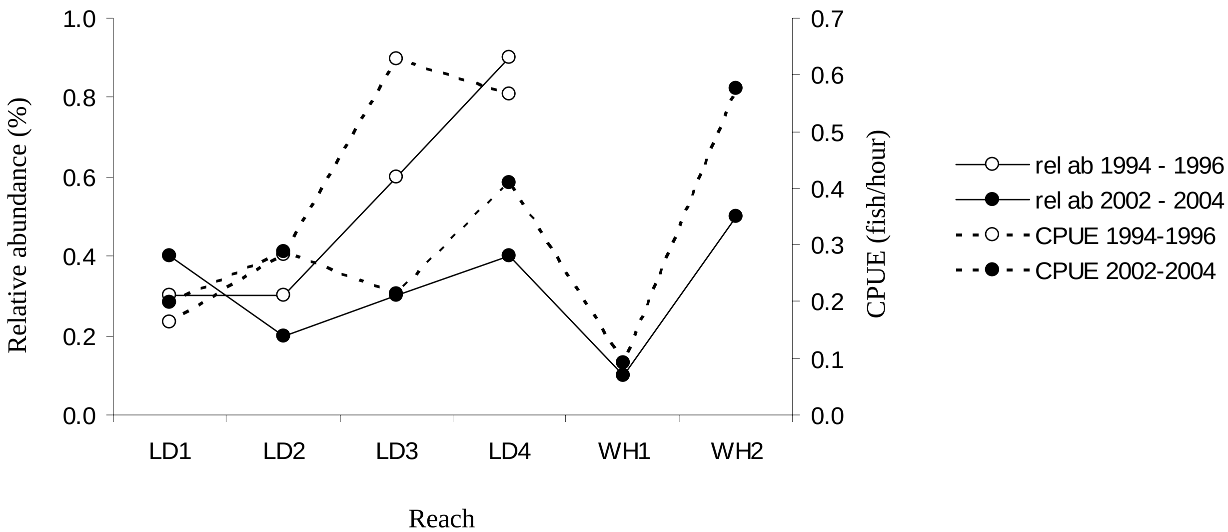


Figure 14—Distribution and abundance of Colorado pikeminnow in electrofishing samples from the Green River downstream of Flaming Gorge Dam, Colorado and Utah, in periods 1994-1996 and 2002-2004. CPUE is number of fish captured per hour of electrofishing. Sampling areas include four Lodore Canyon reaches (LD1, RK 585.9 to 579.0; LD2, RK 578.9 to 571.0, LD3, RK 570.9 to 563.0; LD4, RK 562.9 to 555.3) and two Whirlpool Canyon reaches (WH1, RK 555.2 to 546.6; WH2, RK 546.5 to 538.1).

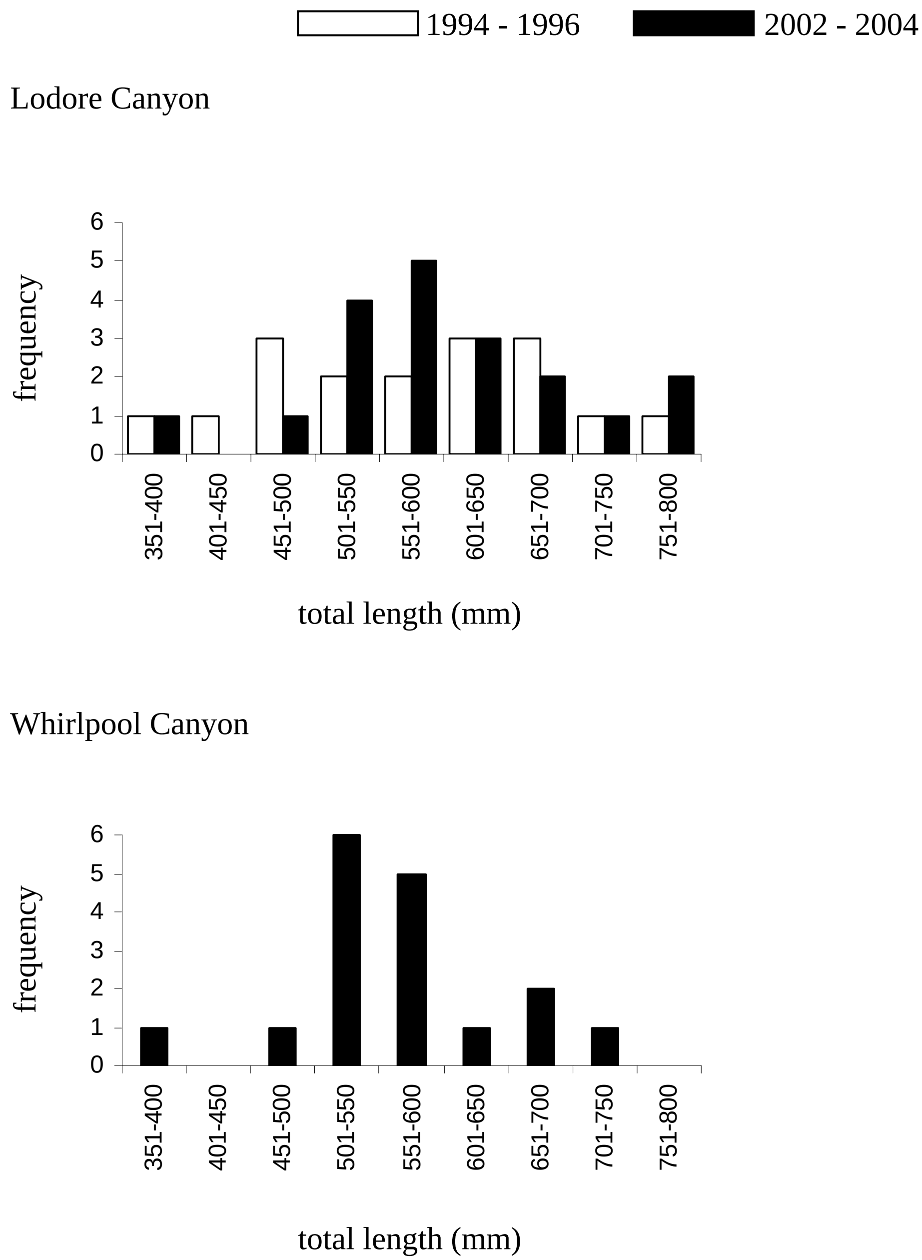
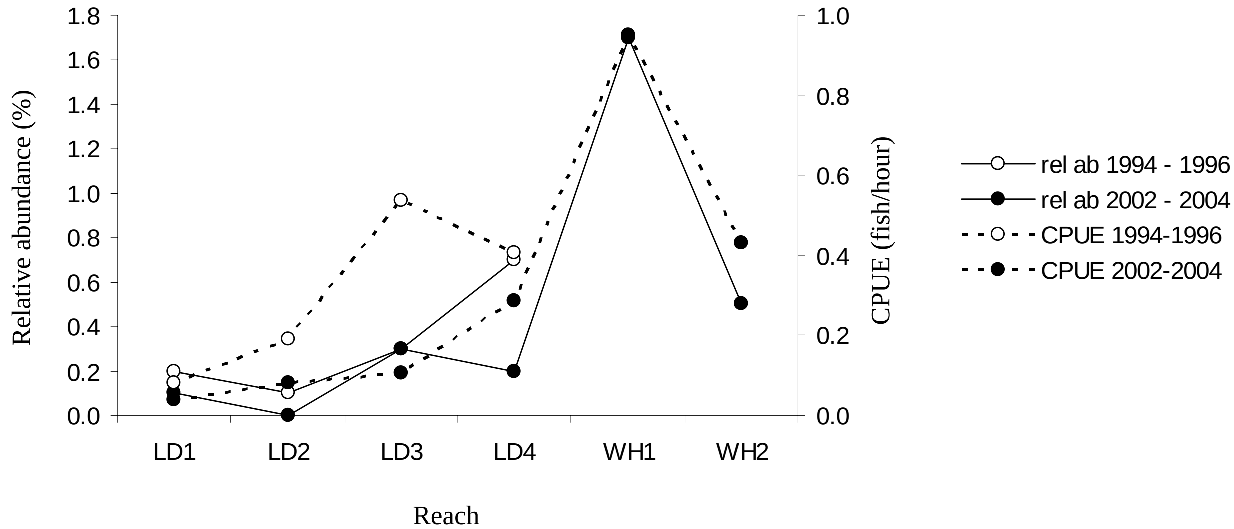


Figure 15.—Length-frequency histograms for Colorado pikeminnow from Lodore Canyon (1994-1996 and 2002-2004), and Whirlpool Canyon (2002-2004), Green River. Lodore Canyon = RK 585.9 to 555.3 and Whirlpool Canyon = RK 555.2 to 538.1. Fish were captured by electrofishing.

A. electrofishing



B. seine

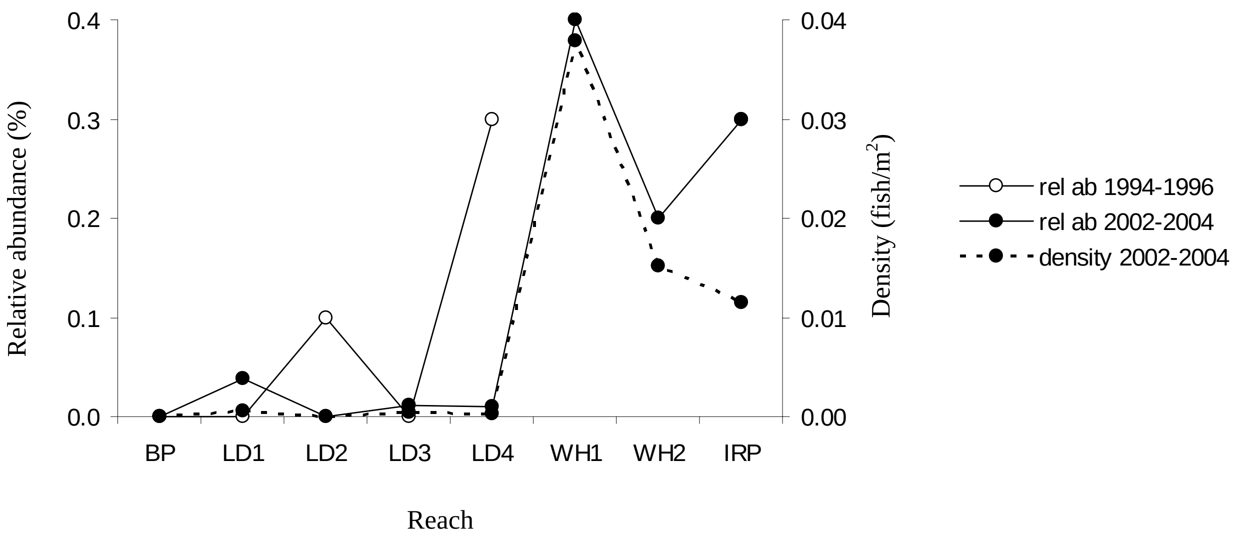


Figure 16.— Distribution and abundance of roundtail chub in electrofishing and seine samples from the Green River downstream of Flaming Gorge Dam, Colorado and Utah, in periods 1994-1996 and 2002-2004. Only relative abundance data are available for seine samples in the 1994-1996 period; CPUE is number of fish captured per hour of electrofishing. Sampling areas include Browns Park (BP, RK 613.0-586.0), four Lodore Canyon reaches (LD1, RK 585.9 to 579.0; LD2, RK 578.9 to 571.0, LD3, RK 570.9 to 563.0; LD4, RK 562.9 to 555.3), two Whirlpool Canyon reaches (WH1, RK 555.2 to 546.6; WH2, RK 546.5 to 538.1), and Island-Rainbow Park (IRP, RK 538.0-528.0).

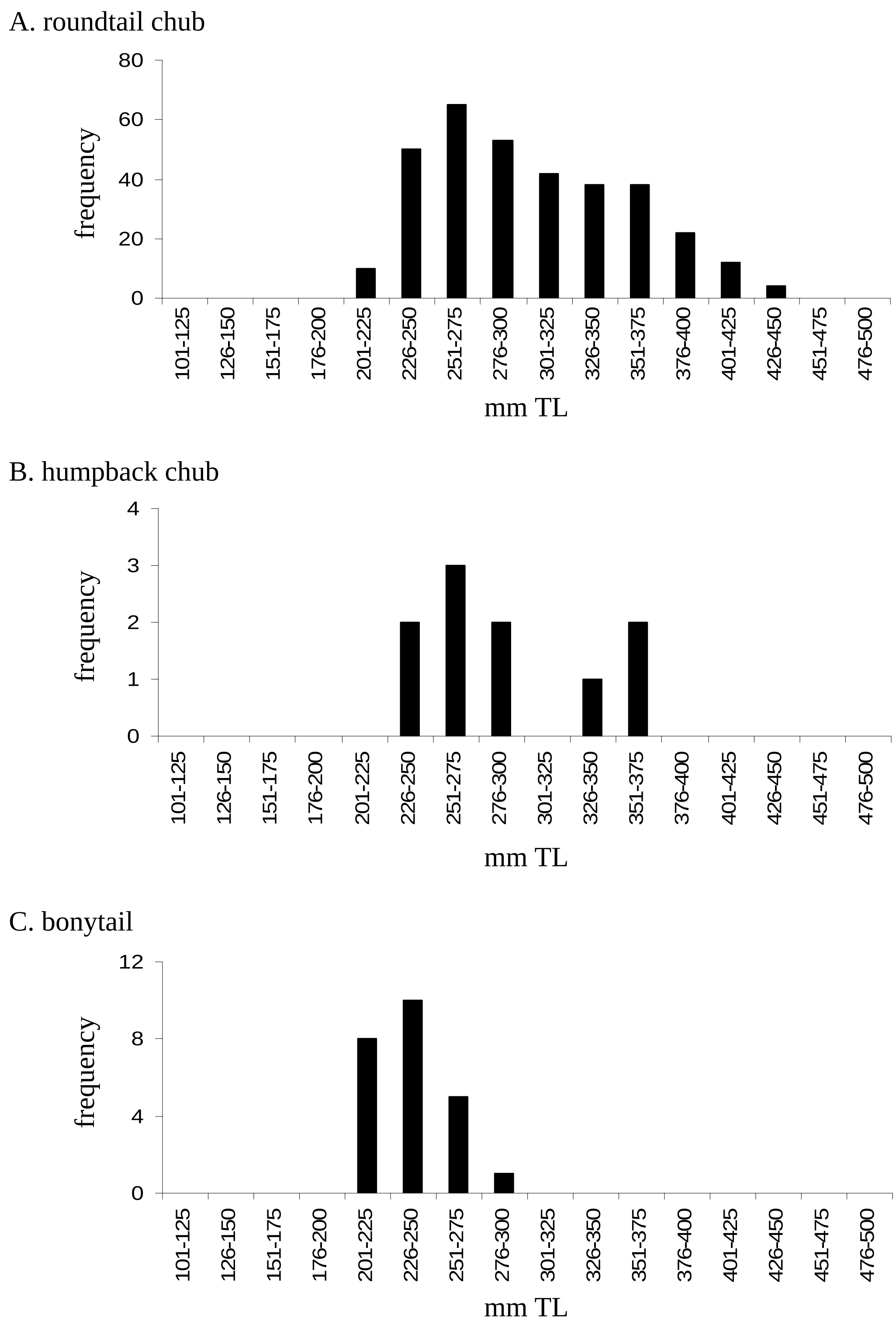


Figure 17.—Length-frequency histograms for roundtail chub (A), humpback chub (B), and bonytail (C) from Lodore and Whirlpool canyons, Green River, 2002-2004. Lodore Canyon = RK 585.9 to 555.3 and Whirlpool Canyon = RK 555.2 to 538.1. Fish were captured by electrofishing and with trammel nets.

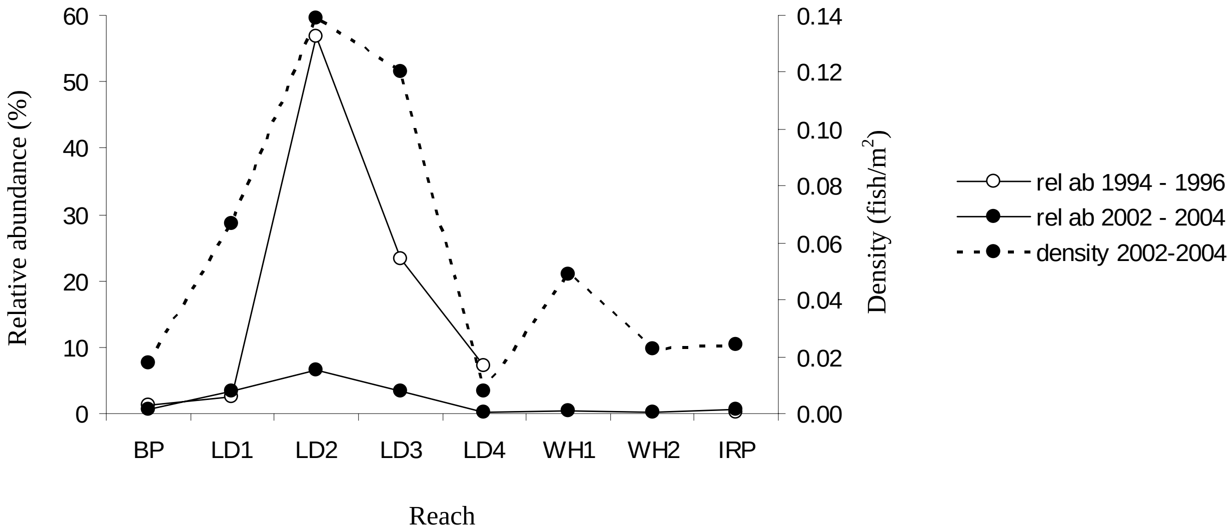
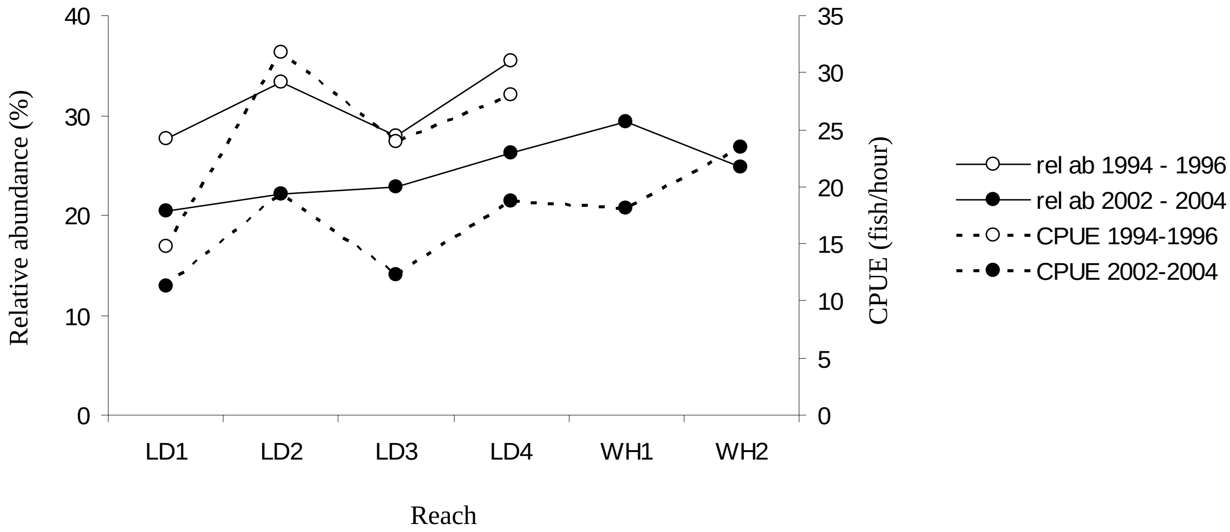


Figure 18.— Distribution and abundance of speckled dace in seine samples from the Green River downstream of Flaming Gorge Dam, Colorado and Utah, in periods 1994-1996 and 2002-2004. Only relative abundance data are available for seine samples in the 1994-1996 period. Sampling areas include Browns Park (BP, RK 613.0-586.0), four Lodore Canyon reaches (LD1, RK 585.9 to 579.0; LD2, RK 578.9 to 571.0, LD3, RK 570.9 to 563.0; LD4, RK 562.9 to 555.3), two Whirlpool Canyon reaches (WH1, RK 555.2 to 546.6; WH2, RK 546.5 to 538.1), and Island-Rainbow Park (IRP, RK 538.0-528.0).

A. electrofishing



B. seine

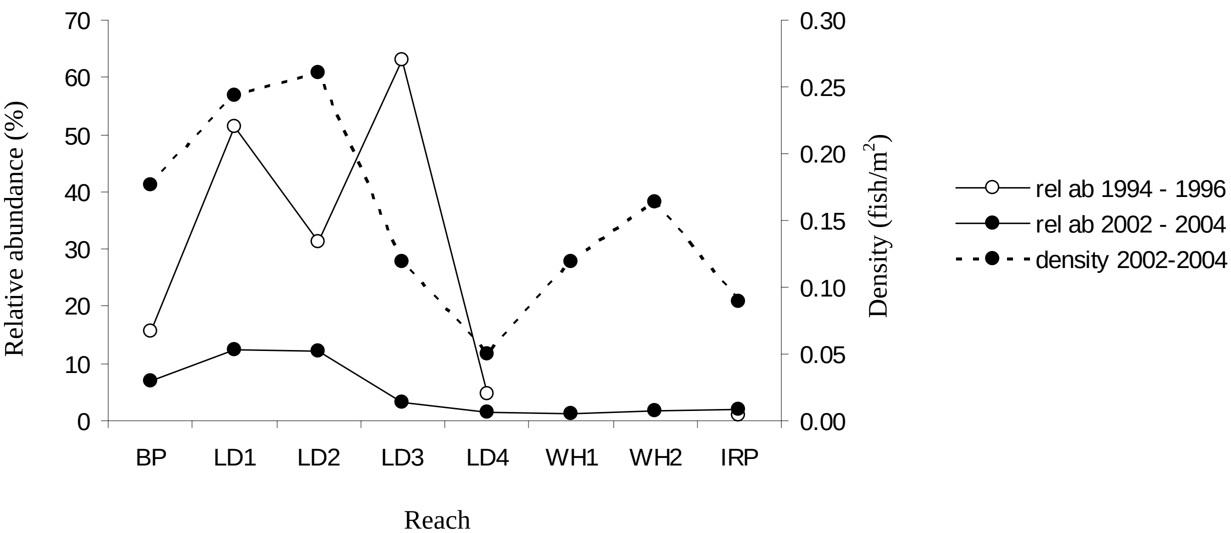


Figure 19.— Distribution and abundance of flannelmouth sucker in electrofishing and seine samples from the Green River downstream of Flaming Gorge Dam, Colorado and Utah, in periods 1994-1996 and 2002-2004. Only relative abundance data are available for seine samples in the 1994-1996 period; CPUE is number of fish captured per hour of electrofishing. Sampling areas include Browns Park (BP, RK 613.0-586.0), four Lodore Canyon reaches (LD1, RK 585.9 to 579.0; LD2, RK 578.9 to 571.0, LD3, RK 570.9 to 563.0; LD4, RK 562.9 to 555.3), two Whirlpool Canyon reaches (WH1, RK 555.2 to 546.6; WH2, RK 546.5 to 538.1), and Island-Rainbow Park (IRP, RK 538.0-528.0).

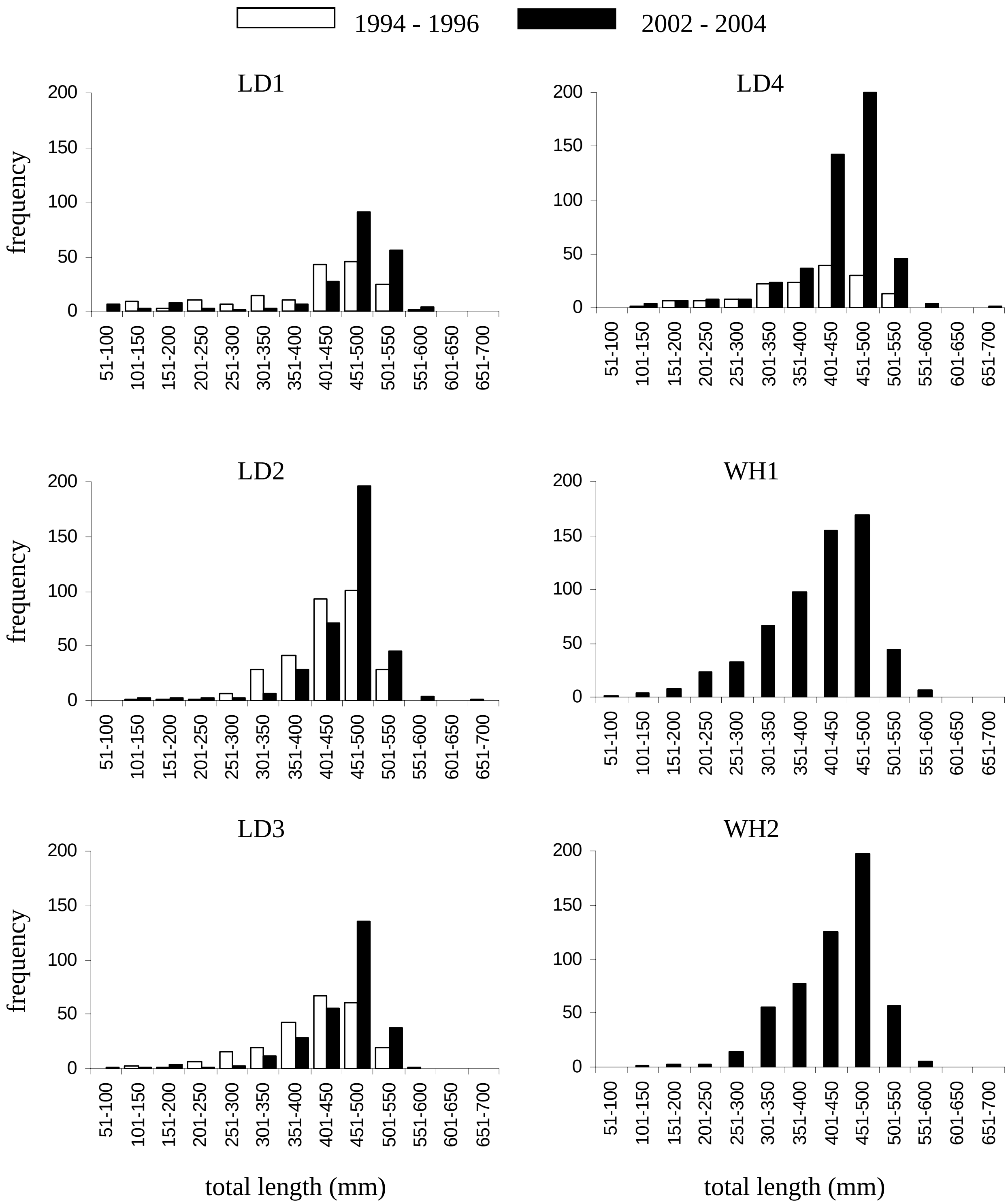
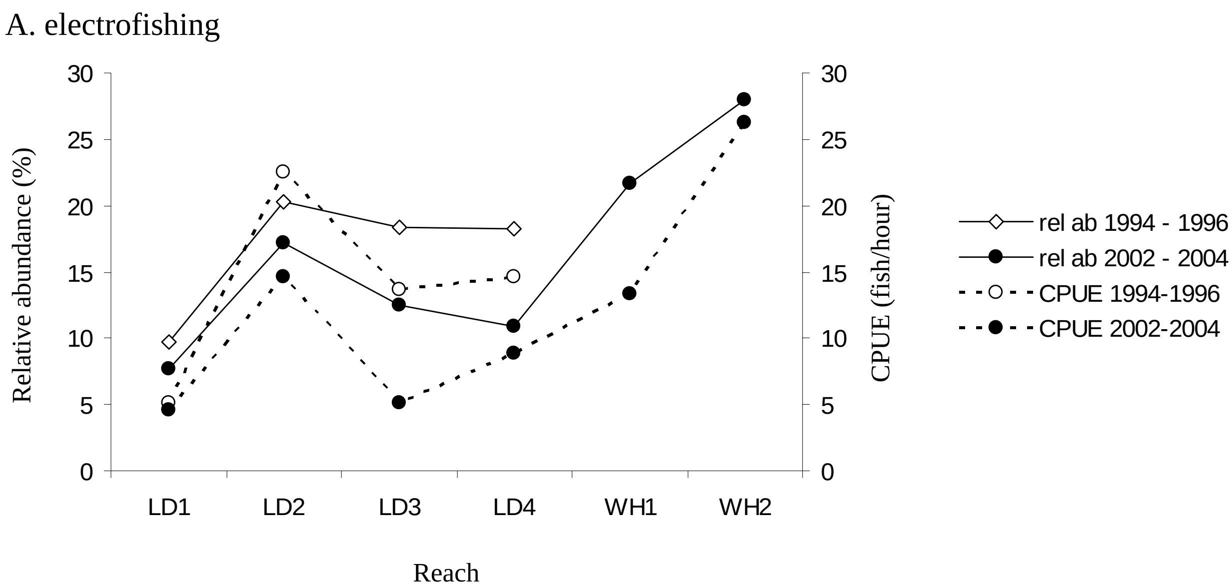


Figure 20.—Length-frequency histograms for flannelmouth sucker from four reaches of Lodore Canyon (LD1-LD4, 1994-1996 and 2002-2004) and two reaches of Whirlpool Canyon (WH1-WH2, 2002-2004), Green River. Reach LD1 = RK 585.9 to 579.0, LD2 = RK 578.9 to 571.0, LD3 = RK 570.9 to 563.0, LD4 = RK 562.9 to 555.3, WH1 = RK 555.2 to 546.6, WH2 = RK 546.5 to 538.1. Fish were captured by electrofishing.



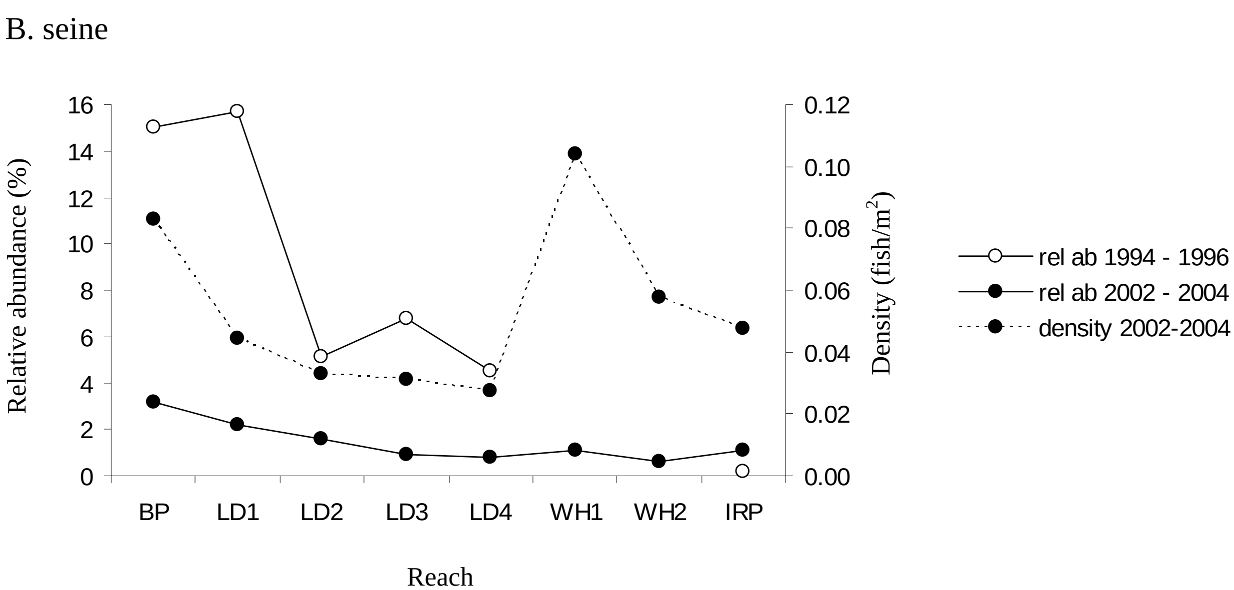


Figure 21.— Distribution and abundance of bluehead sucker in electrofishing and seine samples from the Green River downstream of Flaming Gorge Dam, Colorado and Utah, in periods 1994-1996 and 2002-2004. Only relative abundance data are available for seine samples in the 1994-1996 period; CPUE is number of fish captured per hour of electrofishing. Sampling areas include Browns Park (BP, RK 613.0-586.0), four Lodore Canyon reaches (LD1, RK 585.9 to 579.0; LD2, RK 578.9 to 571.0, LD3, RK 570.9 to 563.0; LD4, RK 562.9 to 555.3), two Whirlpool Canyon reaches (WH1, RK 555.2 to 546.6; WH2, RK 546.5 to 538.1), and Island-Rainbow Park (IRP, RK 538.0-528.0).

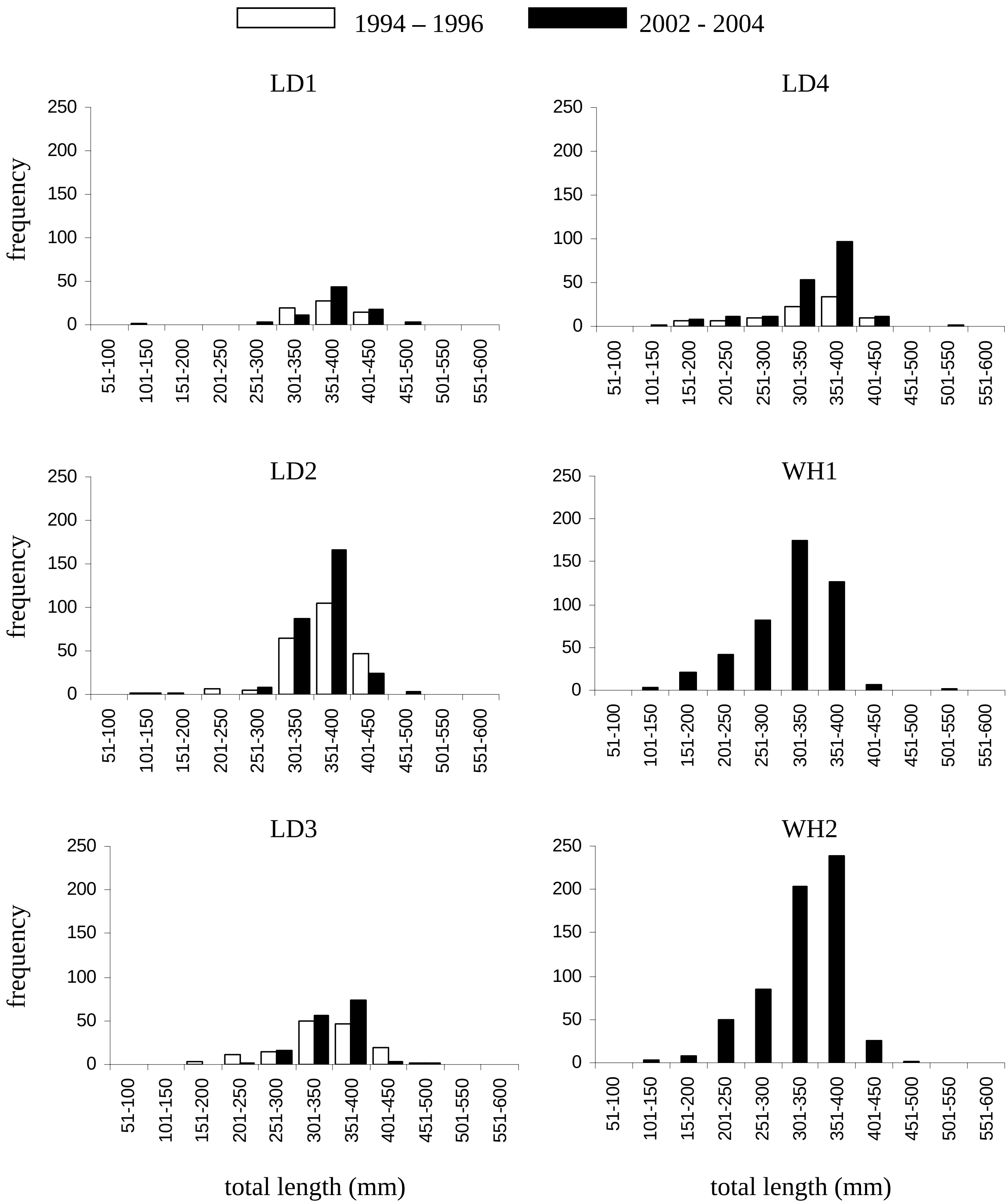
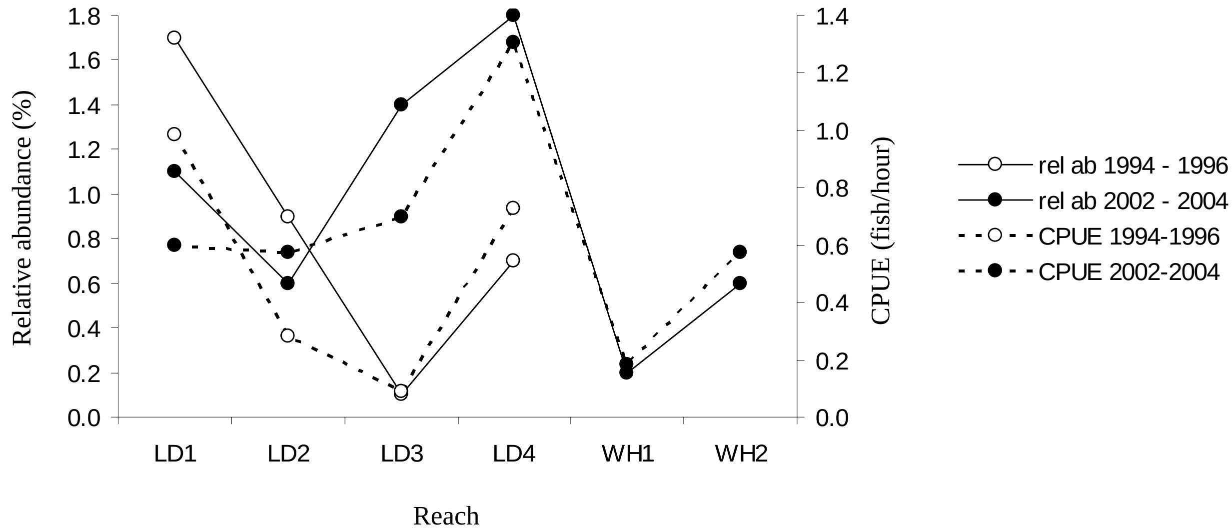


Figure 22.—Length-frequency histograms for bluehead sucker from four reaches of Lodore Canyon (LD1-LD4, 1994-1996 and 2002-2004) and two reaches of Whirlpool Canyon (WH1-WH2, 2002-2004), Green River. Reach LD1 = RK 585.9 to 579.0, LD2 = RK 578.9 to 571.0, LD3 = RK 570.9 to 563.0, LD4 = RK 562.9 to 555.3, WH1 = RK 555.2 to 546.6, WH2 = RK 546.5 to 538.1. Fish were captured by electrofishing.

A. electrofishing



B. seine

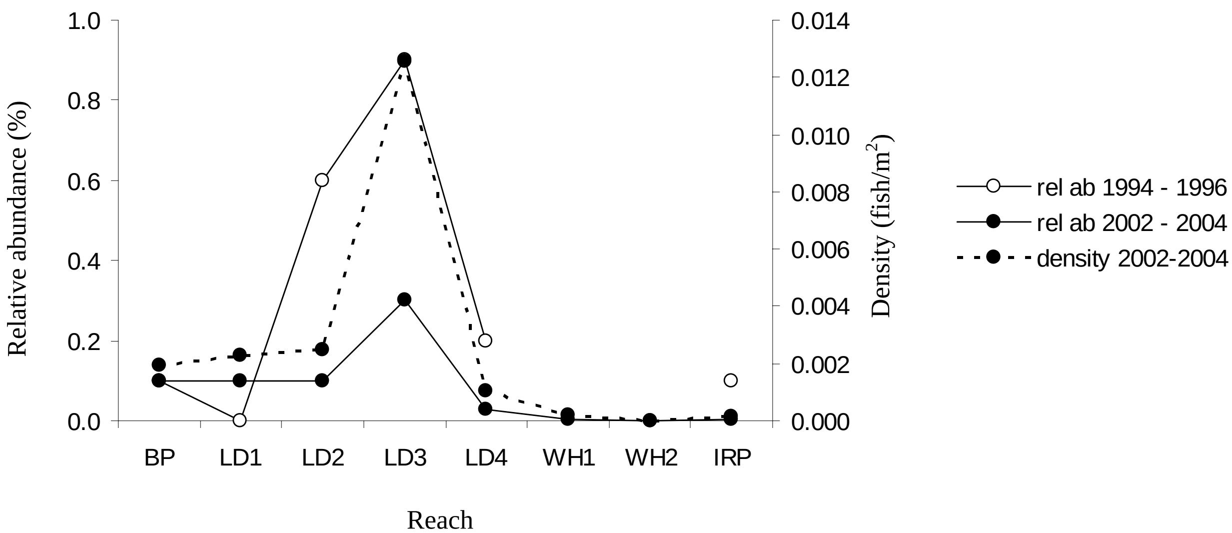


Figure 23.— Distribution and abundance of mottled sculpin in electrofishing and seine samples from the Green River downstream of Flaming Gorge Dam, Colorado and Utah, in periods 1994-1996 and 2002-2004. Only relative abundance data are available for seine samples in the 1994-1996 period; CPUE is number of fish captured per hour of electrofishing. Sampling areas include Browns Park (BP, RK 613.0-586.0), four Lodore Canyon reaches (LD1, RK 585.9 to 579.0; LD2, RK 578.9 to 571.0, LD3, RK 570.9 to 563.0; LD4, RK 562.9 to 555.3), two Whirlpool Canyon reaches (WH1, RK 555.2 to 546.6; WH2, RK 546.5 to 538.1), and Island-Rainbow Park (IRP, RK 538.0-528.0).

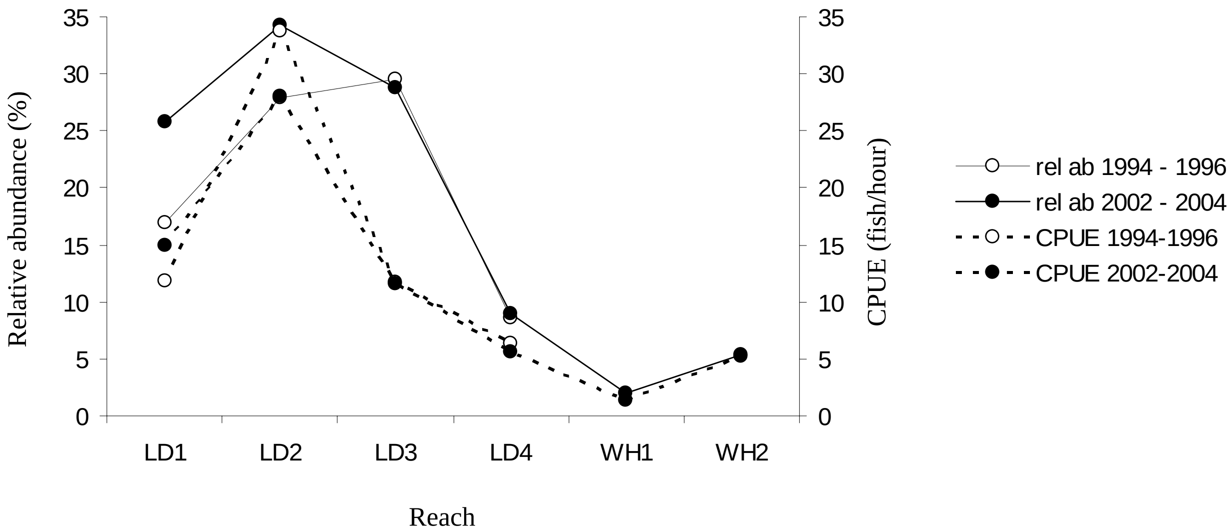


Figure 24.— Distribution and abundance of salmonids in electrofishing samples from the Green River downstream of Flaming Gorge Dam, Colorado and Utah, in periods 1994-1996 and 2002-2004. CPUE is number of fish captured per hour of electrofishing. Sampling areas include four Lodore Canyon reaches (LD1, RK 585.9 to 579.0; LD2, RK 578.9 to 571.0, LD3, RK 570.9 to 563.0; LD4, RK 562.9 to 555.3) and two Whirlpool Canyon reaches (WH1, RK 555.2 to 546.6; WH2, RK 546.5 to 538.1).

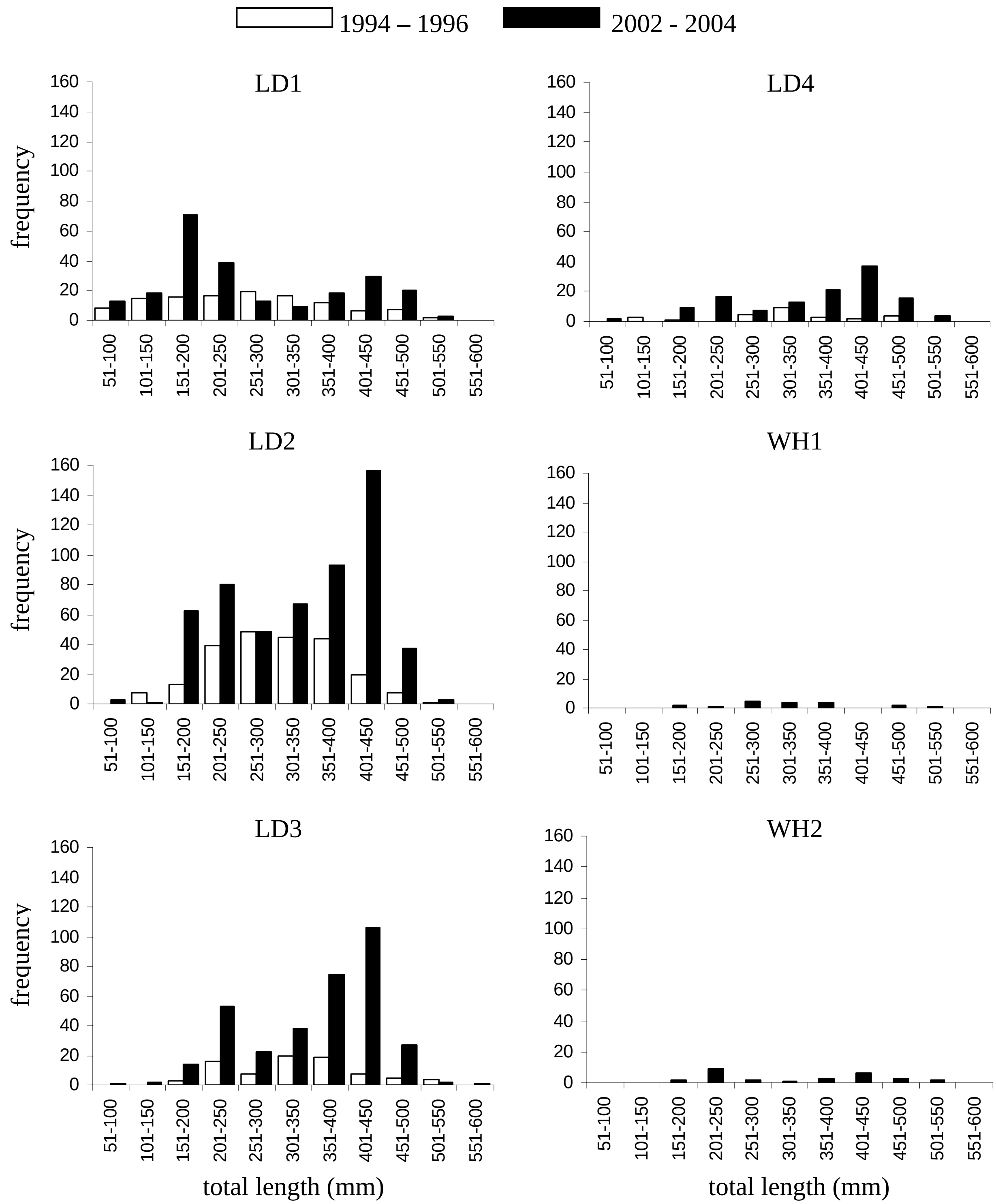


Figure 25.—Length-frequency histograms for brown trout from four reaches of Lodore Canyon (LD1-LD4, 1994-1996 and 2002-2004) and two reaches of Whirlpool Canyon (WH1-WH2, 2002-2004), Green River. Reach LD1 = RK 585.9 to 579.0, LD2 = RK 578.9 to 571.0, LD3 = RK 570.9 to 563.0, LD4 = RK 562.9 to 555.3, WH1 = RK 555.2 to 546.6, WH2 = RK 546.5 to 538.1. Fish were captured by electrofishing.

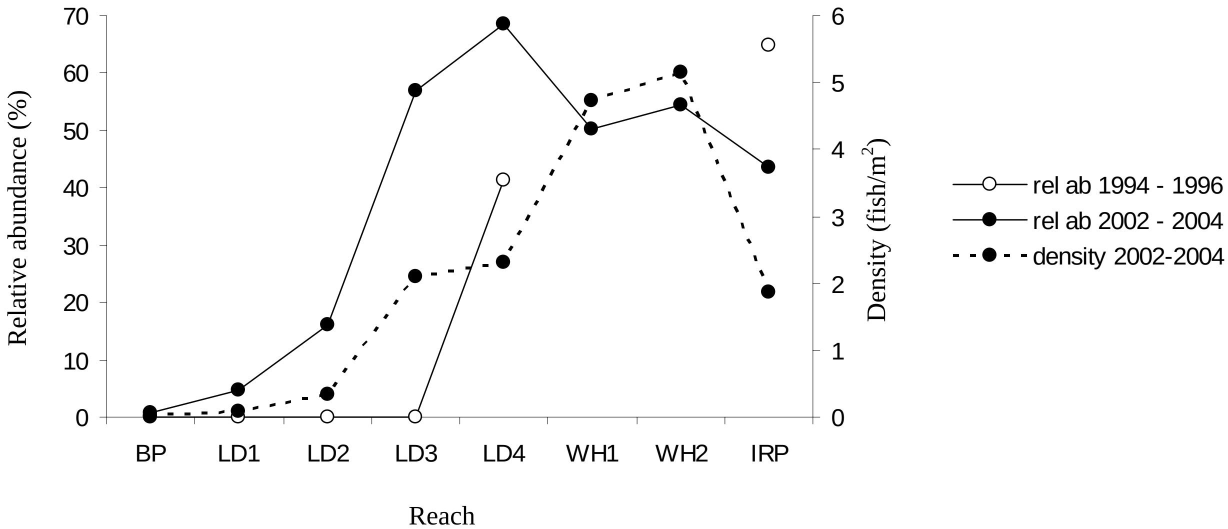


Figure 26.— Distribution and abundance of red shiner in seine samples from the Green River downstream of Flaming Gorge Dam, Colorado and Utah, in periods 1994-1996 and 2002-2004. Only relative abundance data are available for seine samples in the 1994-1996 period. Sampling areas include Browns Park (BP, RK 613.0-586.0), four Lodore Canyon reaches (LD1, RK 585.9 to 579.0; LD2, RK 578.9 to 571.0, LD3, RK 570.9 to 563.0; LD4, RK 562.9 to 555.3), two Whirlpool Canyon reaches (WH1, RK 555.2 to 546.6; WH2, RK 546.5 to 538.1), and Island-Rainbow Park (IRP, RK 538.0-528.0).

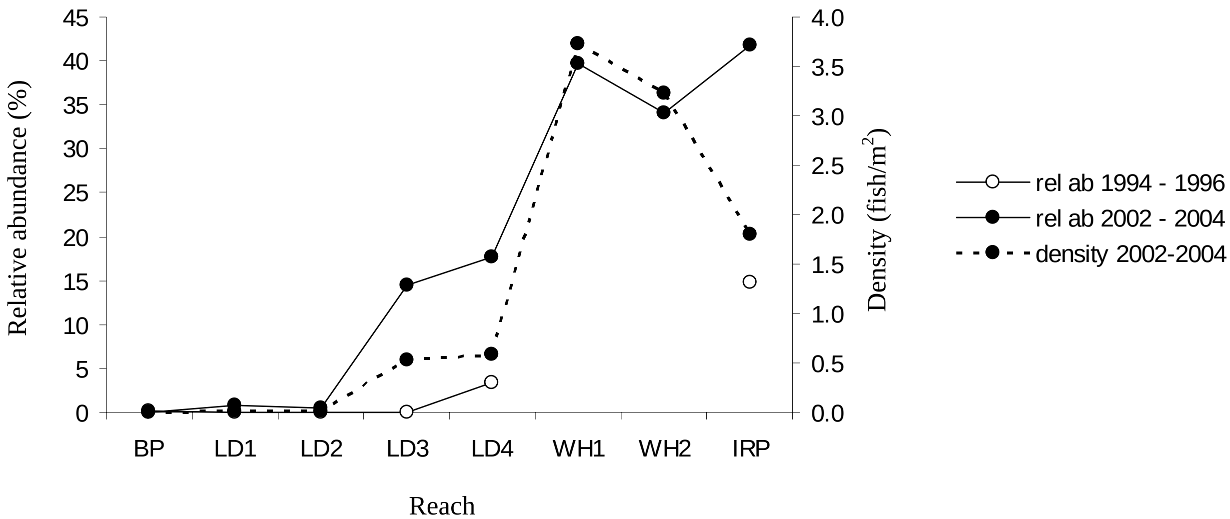


Figure 27.— Distribution and abundance of sand shiner in seine samples from the Green River downstream of Flaming Gorge Dam, Colorado and Utah, in periods 1994-1996 and 2002-2004. Only relative abundance data are available for seine samples in the 1994-1996 period. Sampling areas include Browns Park (BP, RK 613.0-586.0), four Lodore Canyon reaches (LD1, RK 585.9 to 579.0; LD2, RK 578.9 to 571.0, LD3, RK 570.9 to 563.0; LD4, RK 562.9 to 555.3), two Whirlpool Canyon reaches (WH1, RK 555.2 to 546.6; WH2, RK 546.5 to 538.1), and Island-Rainbow Park (IRP, RK 538.0-528.0).

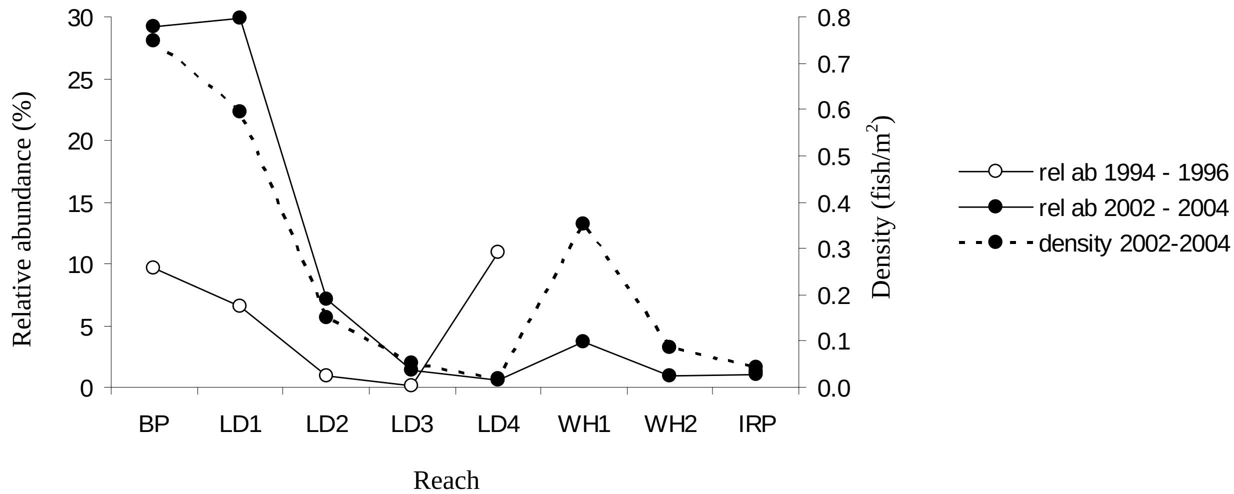


Figure 28.— Distribution and abundance of redside shiner in seine samples from the Green River downstream of Flaming Gorge Dam, Colorado and Utah, in periods 1994-1996 and 2002-2004. Only relative abundance data are available for seine samples in the 1994-1996 period. Sampling areas include Browns Park (BP, RK 613.0-586.0), four Lodore Canyon reaches (LD1, RK 585.9 to 579.0; LD2, RK 578.9 to 571.0, LD3, RK 570.9 to 563.0; LD4, RK 562.9 to 555.3), two Whirlpool Canyon reaches (WH1, RK 555.2 to 546.6; WH2, RK 546.5 to 538.1), and Island-Rainbow Park (IRP, RK 538.0-528.0).

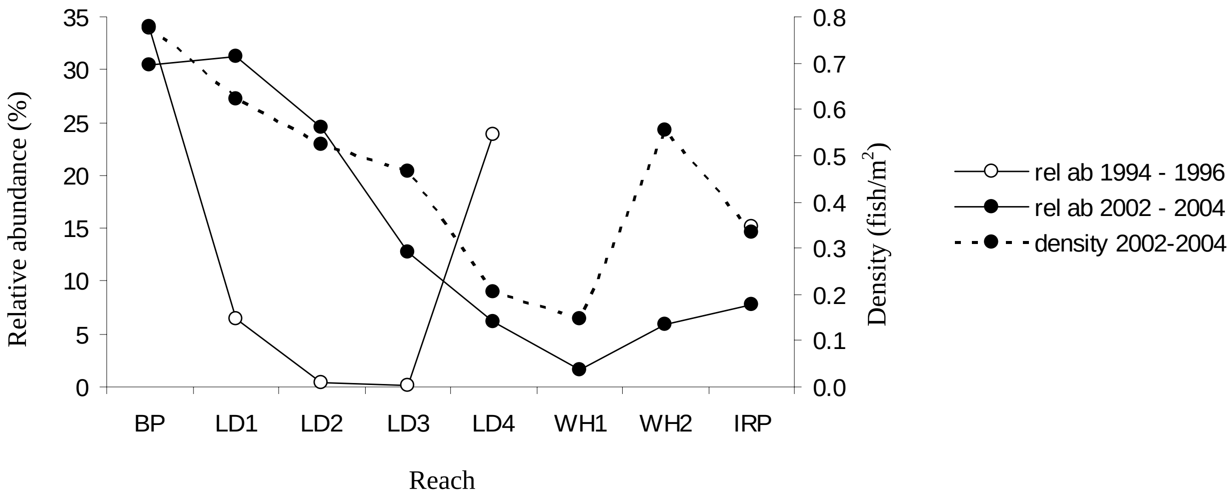
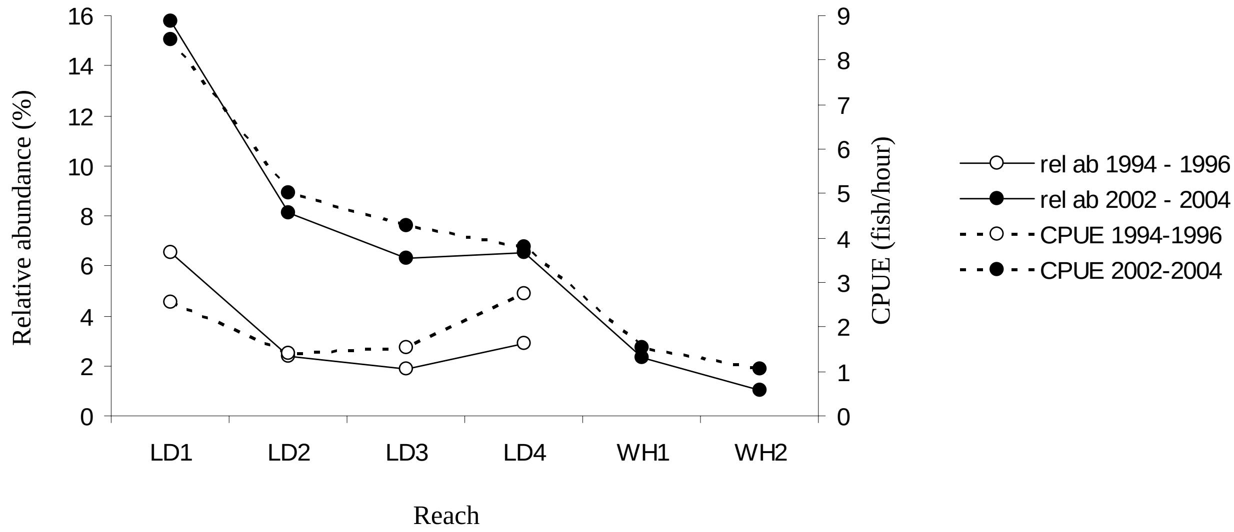


Figure 29.— Distribution and abundance of fathead minnow in seine samples from the Green River downstream of Flaming Gorge Dam, Colorado and Utah, in periods 1994-1996 and 2002-2004. Only relative abundance data are available for seine samples in the 1994-1996 period. Sampling areas include Browns Park (BP, RK 613.0-586.0), four Lodore Canyon reaches (LD1, RK 585.9 to 579.0; LD2, RK 578.9 to 571.0, LD3, RK 570.9 to 563.0; LD4, RK 562.9 to 555.3), two Whirlpool Canyon reaches (WH1, RK 555.2 to 546.6; WH2, RK 546.5 to 538.1), and Island-Rainbow Park (IRP, RK 538.0-528.0).

A. electrofishing



B. seine

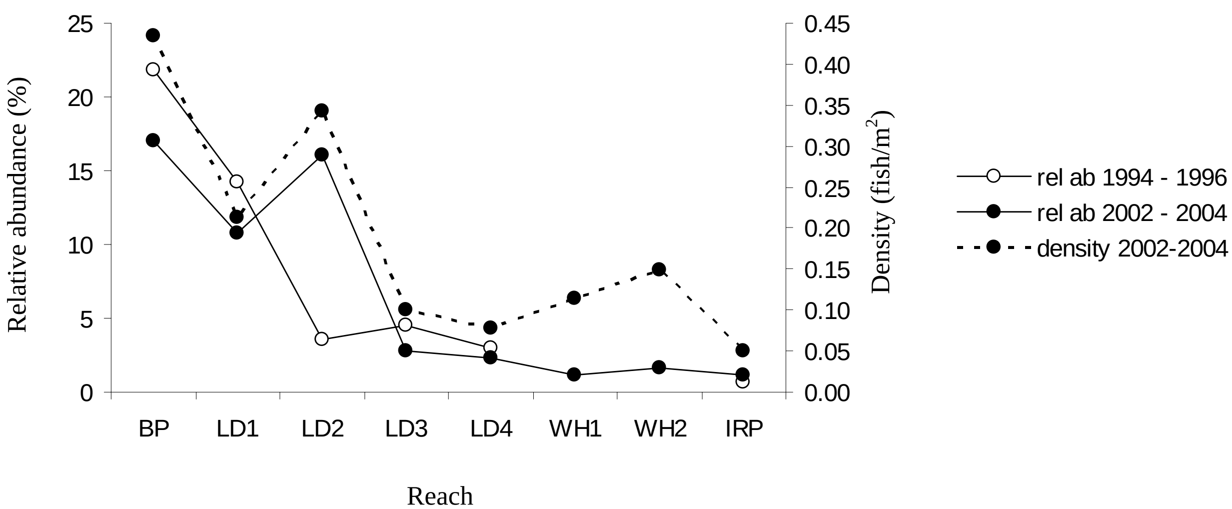


Figure 30.— Distribution and abundance of white sucker in electrofishing and seine samples from the Green River downstream of Flaming Gorge Dam, Colorado and Utah, in periods 1994-1996 and 2002-2004. Only relative abundance data are available for seine samples in the 1994-1996 period; CPUE is number of fish captured per hour of electrofishing. Sampling areas include Browns Park (BP, RK 613.0-586.0), four Lodore Canyon reaches (LD1, RK 585.9 to 579.0; LD2, RK 578.9 to 571.0, LD3, RK 570.9 to 563.0; LD4, RK 562.9 to 555.3), two Whirlpool Canyon reaches (WH1, RK 555.2 to 546.6; WH2, RK 546.5 to 538.1), and Island-Rainbow Park (IRP, RK 538.0-528.0).

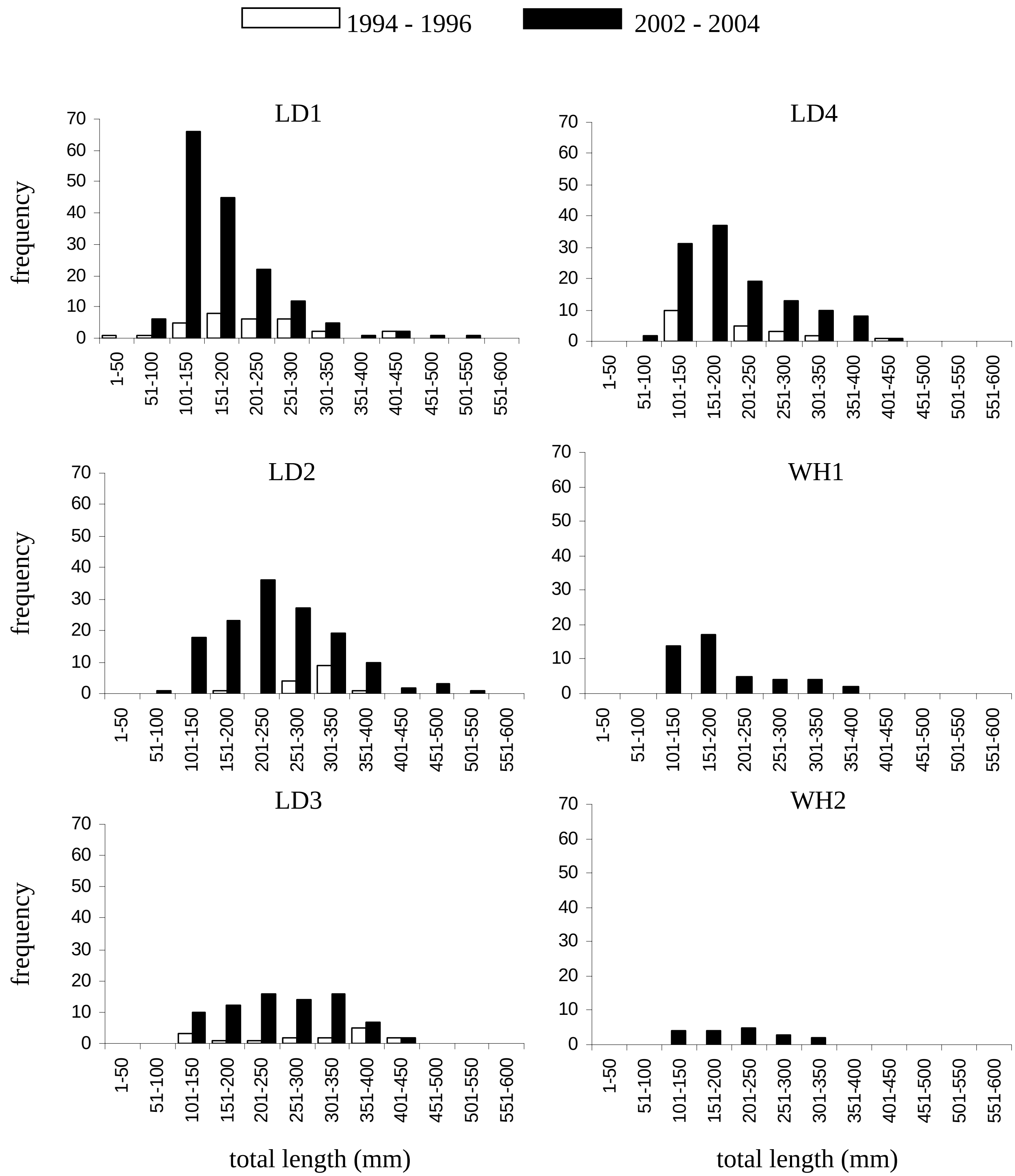


Figure 31.—Length-frequency histograms for white sucker from four reaches of Lodore Canyon (LD1-LD4, 1994-1996 and 2002-2004) and two reaches of Whirlpool Canyon (WH1-WH2, 2002-2004), Green River. Reach LD1 = RK 585.9 to 579.0, LD2 = RK 578.9 to 571.0, LD3 = RK 570.9 to 563.0, LD4 = RK 562.9 to 555.3, WH1 = RK 555.2 to 546.6, WH2 = RK 546.5 to 538.1. Fish were captured by electrofishing.

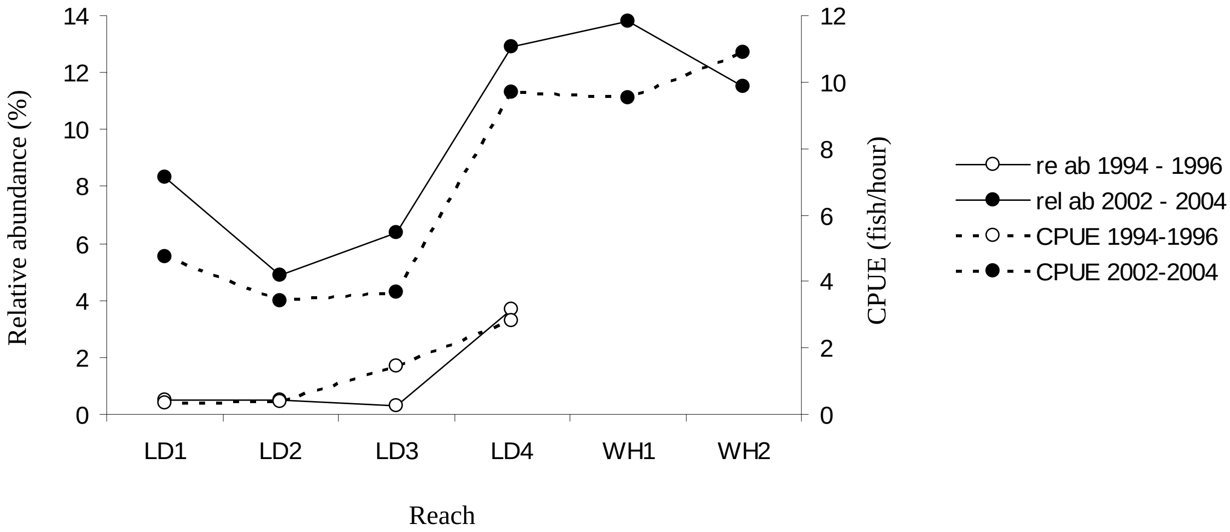


Figure 32.— Distribution and abundance of channel catfish in electrofishing samples from the Green River downstream of Flaming Gorge Dam, Colorado and Utah, in periods 1994-1996 and 2002-2004. CPUE is number of fish captured per hour of electrofishing. Sampling areas include four Lodore Canyon reaches (LD1, RK 585.9 to 579.0; LD2, RK 578.9 to 571.0, LD3, RK 570.9 to 563.0; LD4, RK 562.9 to 555.3) and two Whirlpool Canyon reaches (WH1, RK 555.2 to 546.6; WH2, RK 546.5 to 538.1).

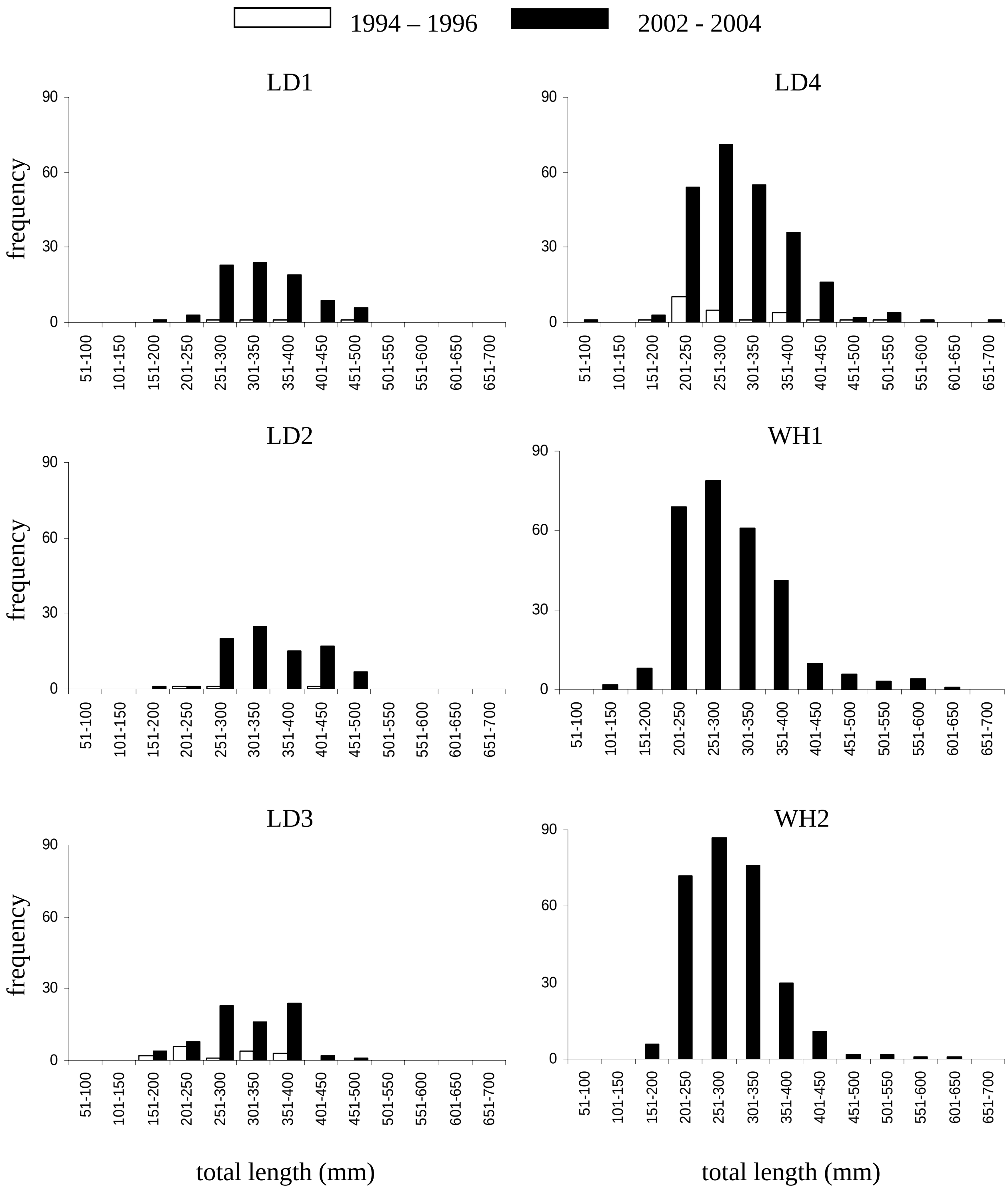
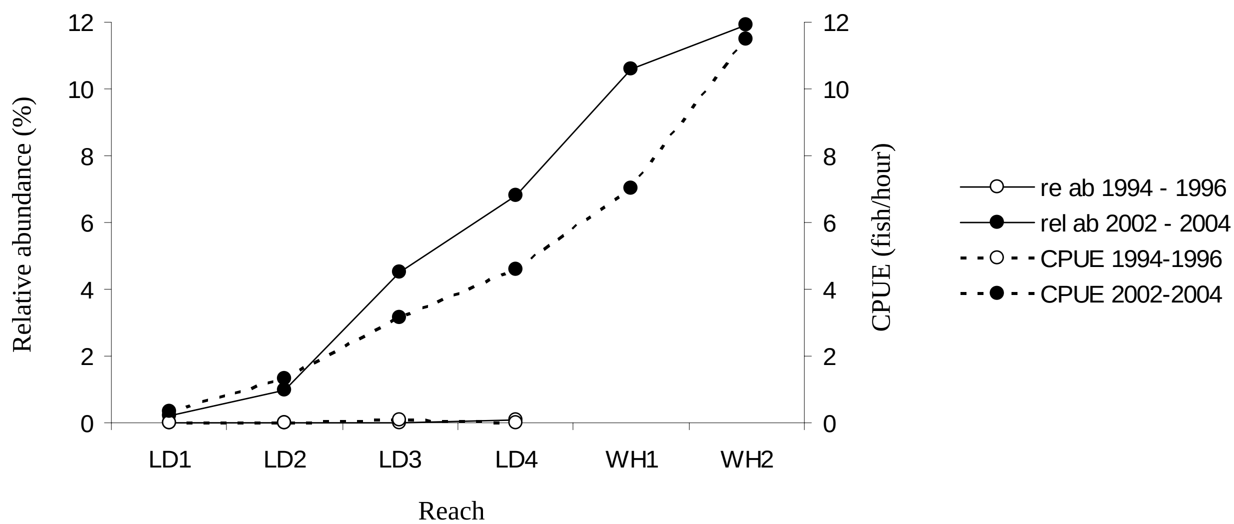


Figure 33.—Length-frequency histograms for channel catfish from four reaches of Lodore Canyon (LD1-LD4, 1994-1996 and 2002-2004) and two reaches of Whirlpool Canyon (WH1-WH2, 2002-2004), Green River. Reach LD1 = RK 585.9 to 579.0, LD2 = RK 578.9 to 571.0, LD3 = RK 570.9 to 563.0, LD4 = RK 562.9 to 555.3, WH1 = RK 555.2 to 546.6, WH2 = RK 546.5 to 538.1. Fish were captured by electrofishing.

A. electrofishing



B. seine

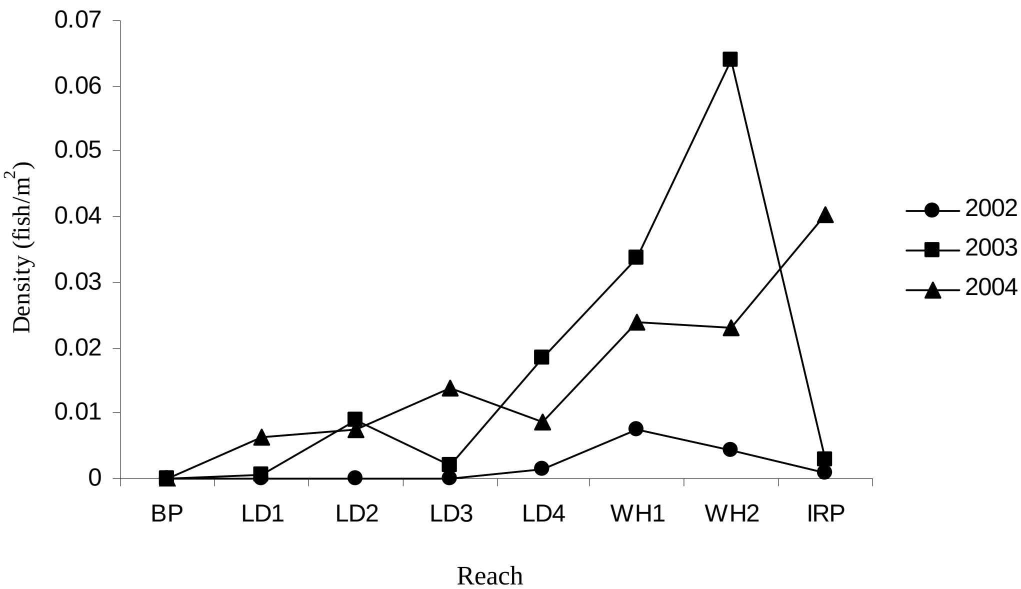


Figure 34.— Distribution and abundance of smallmouth bass in electrofishing and seine samples from the Green River downstream of Flaming Gorge Dam, Colorado and Utah, in periods 1994-1996 and 2002-2004. Only CPUE data are portrayed for seine samples in the 2002-2004 period because only two smallmouth bass were captured in the 1994-1996 period; CPUE is number of fish captured per hour of electrofishing or number of fish seined per m^2. Sampling areas include Browns Park (BP, RK 613.0-586.0), four Lodore Canyon reaches (LD1, RK 585.9 to 579.0; LD2, RK 578.9 to 571.0, LD3, RK 570.9 to 563.0; LD4, RK 562.9 to 555.3), two Whirlpool Canyon reaches (WH1, RK 555.2 to 546.6; WH2, RK 546.5 to 538.1), and Island-Rainbow Park (IRP, RK 538.0-528.0).

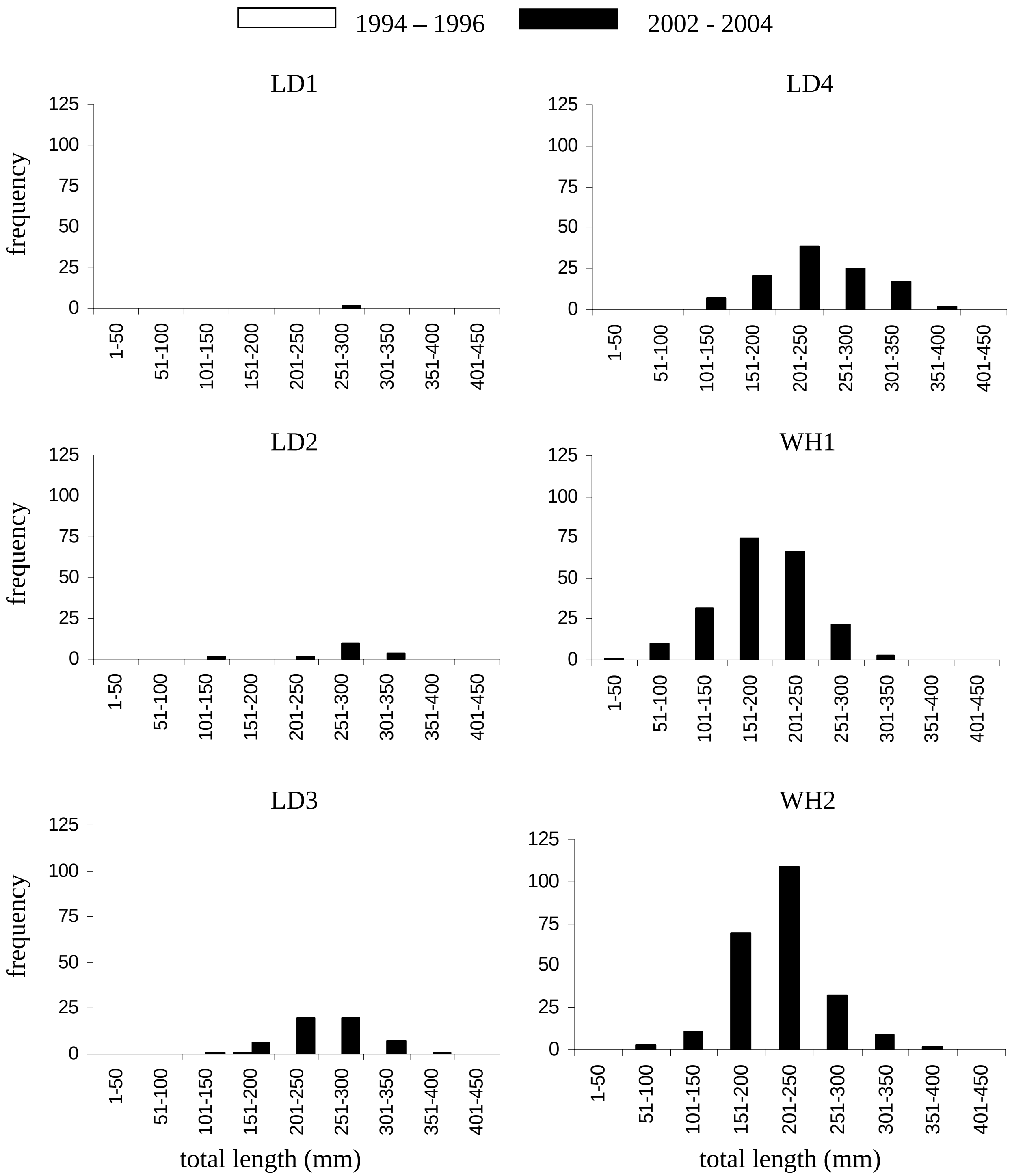


Figure 35.—Length-frequency histograms for smallmouth bass from four reaches of Lodore Canyon (LD1-LD4, 1994-1996 and 2002-2004) and two reaches of Whirlpool Canyon (WH1-WH2, 2002-2004), Green River. Reach LD1 = RK 585.9 to 579.0, LD2 = RK 578.9 to 571.0, LD3 = RK 570.9 to 563.0, LD4 = RK 562.9 to 555.3, WH1 = RK 555.2 to 546.6, WH2 = RK 546.5 to 538.1. Fish were captured by electrofishing.

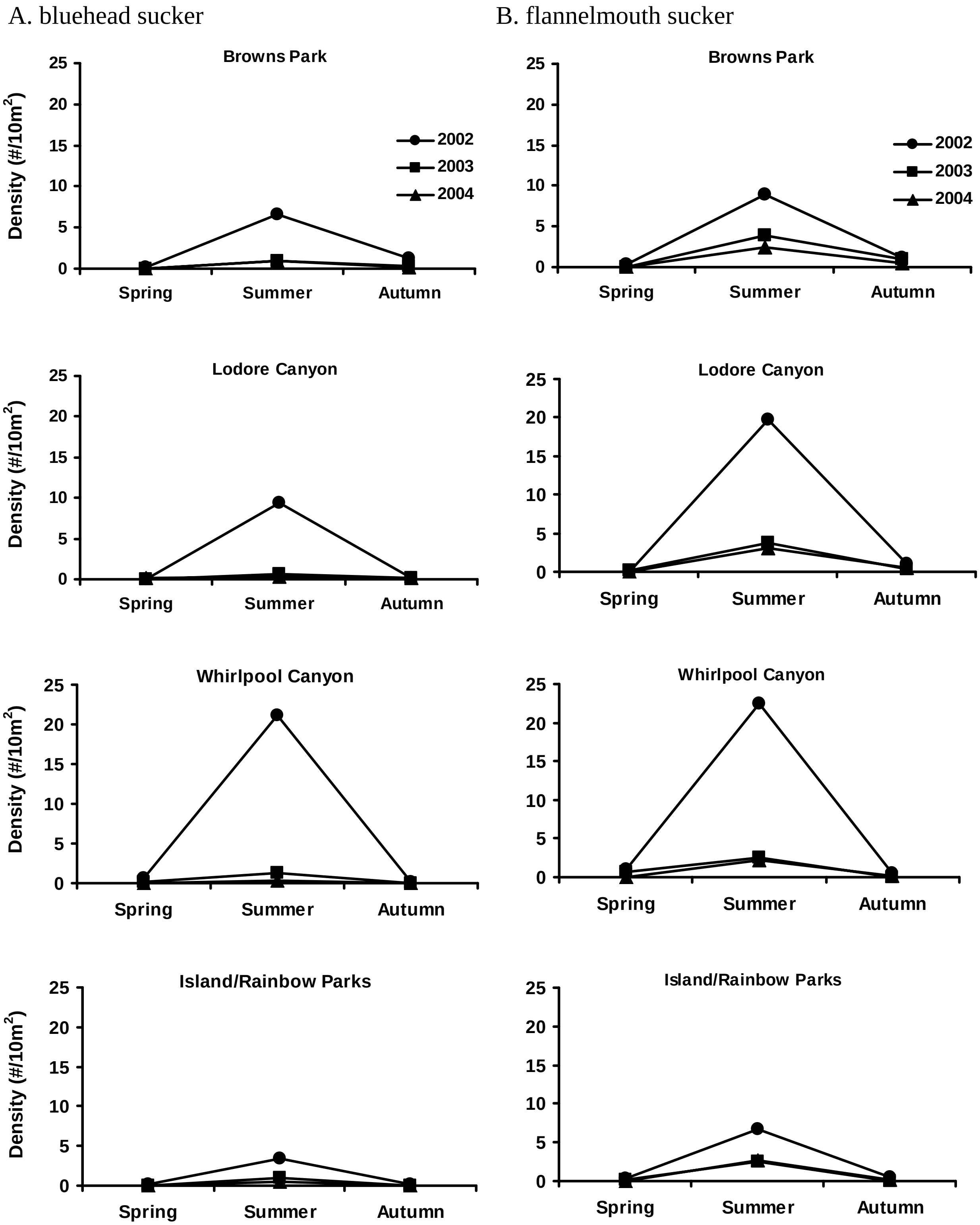


Figure 36.—Mean density of bluehead sucker (A) and flannelmouth sucker (B) in seine samples in spring, summer, and autumn in the Green River, 2002-2004. Browns Park = RK 613.0-586.0, Lodore Canyon = RK 585.9-555.3, Whirlpool Canyon = RK 555.2-538.1, Island-Rainbow Park = RK 538.0-528.0.

A. speckled dace B. roundtail chub

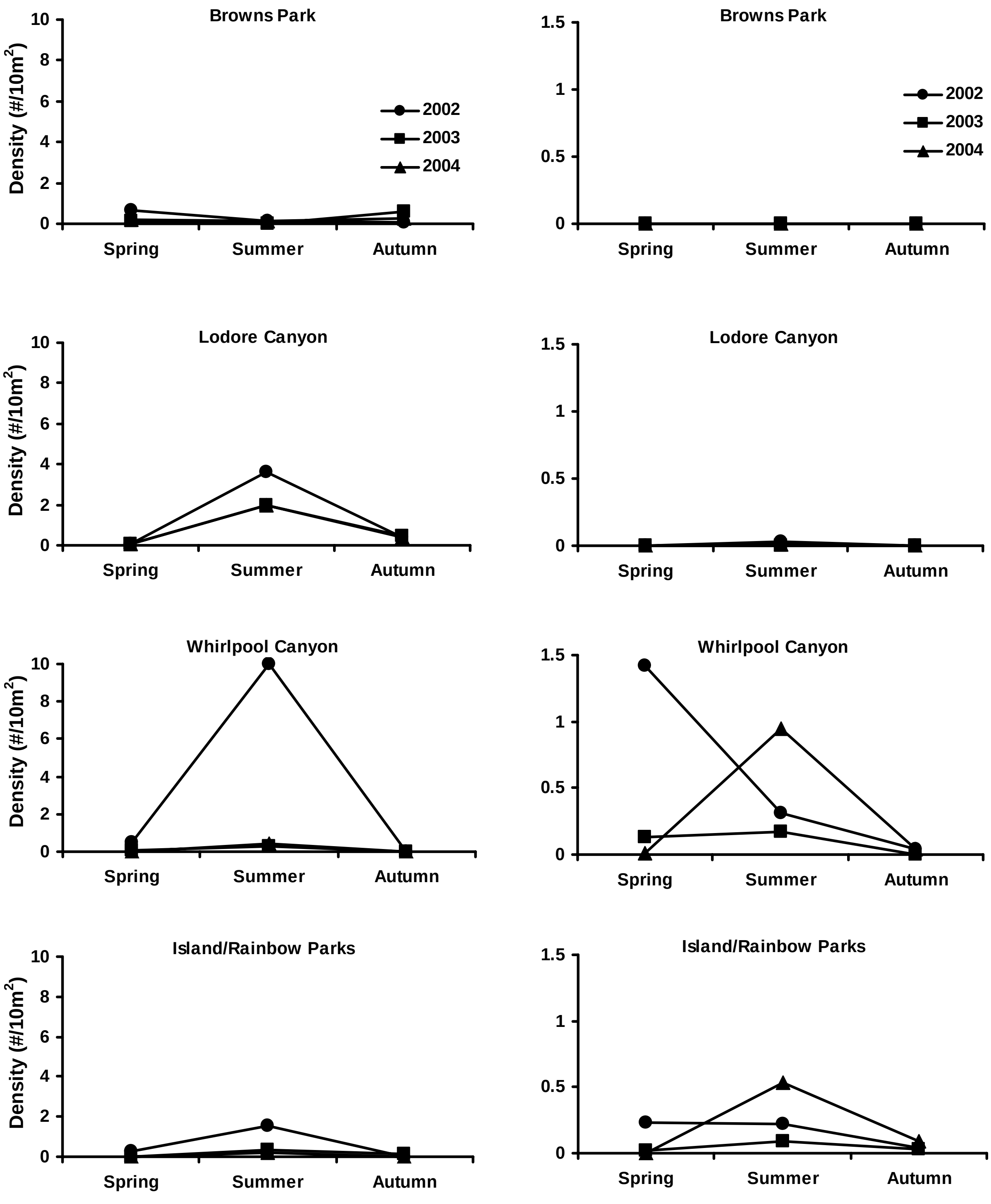


Figure 37.—Mean density of speckled dace (A) and roundtail chub (B) in seine samples in spring, summer, and autumn in the Green River, 2002-2004. Browns Park = RK 613.0-586.0, Lodore Canyon = RK 585.9-555.3, Whirlpool Canyon = RK 555.2-538.1, Island/Rainbow Parks = RK 538.0-528.0. No roundtail chubs were captured in Browns Park.

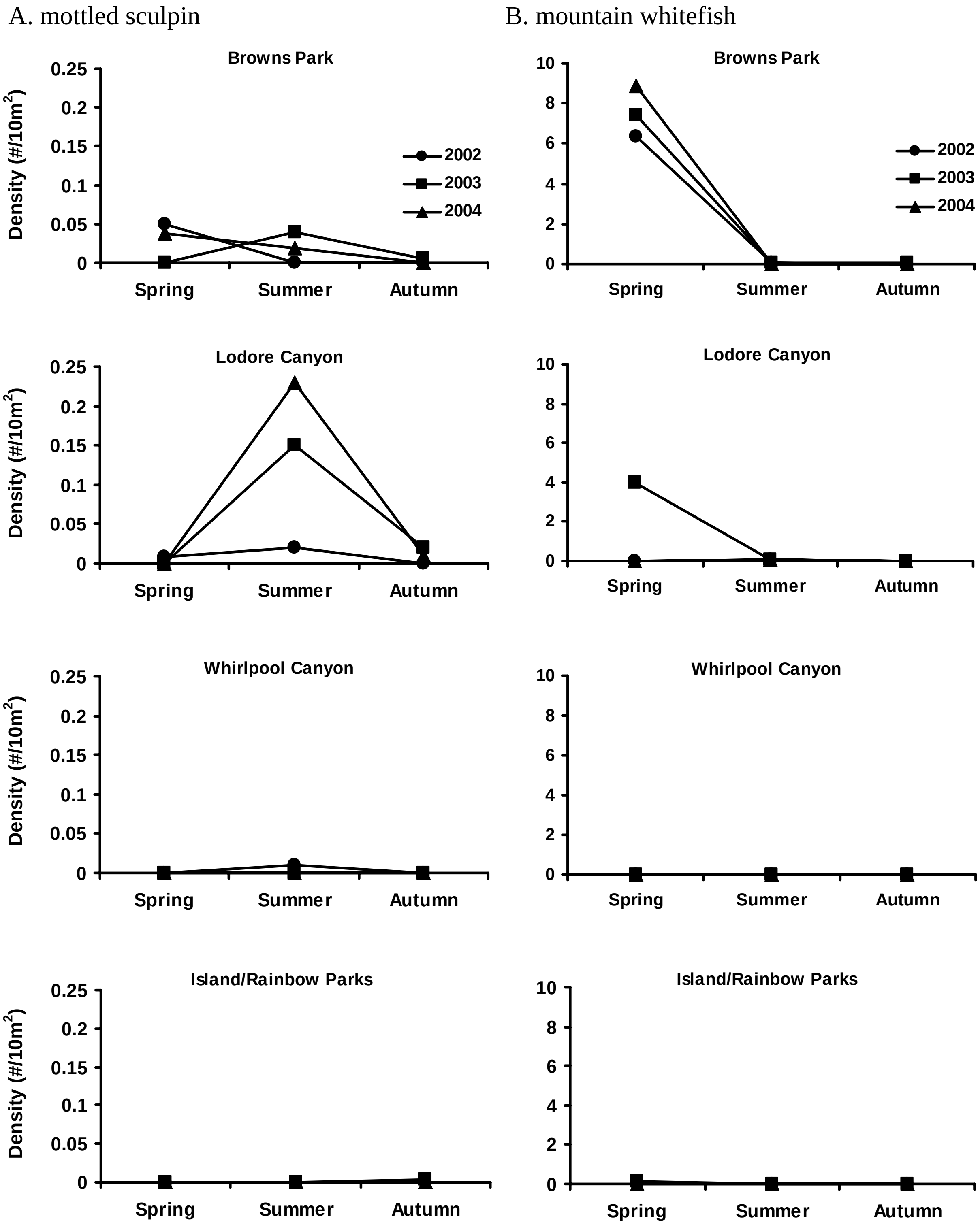


Figure 38.—Mean density of mottled sculpin (A) and mountain whitefish (B) in seine samples in spring, summer, and autumn in the Green River, 2002-2004. Browns Park = RK 613.0-586.0, Lodore Canyon = RK 585.9-555.3, Whirlpool Canyon = RK 555.2-538.1, Island/Rainbow Parks = RK 538.0-528.0.

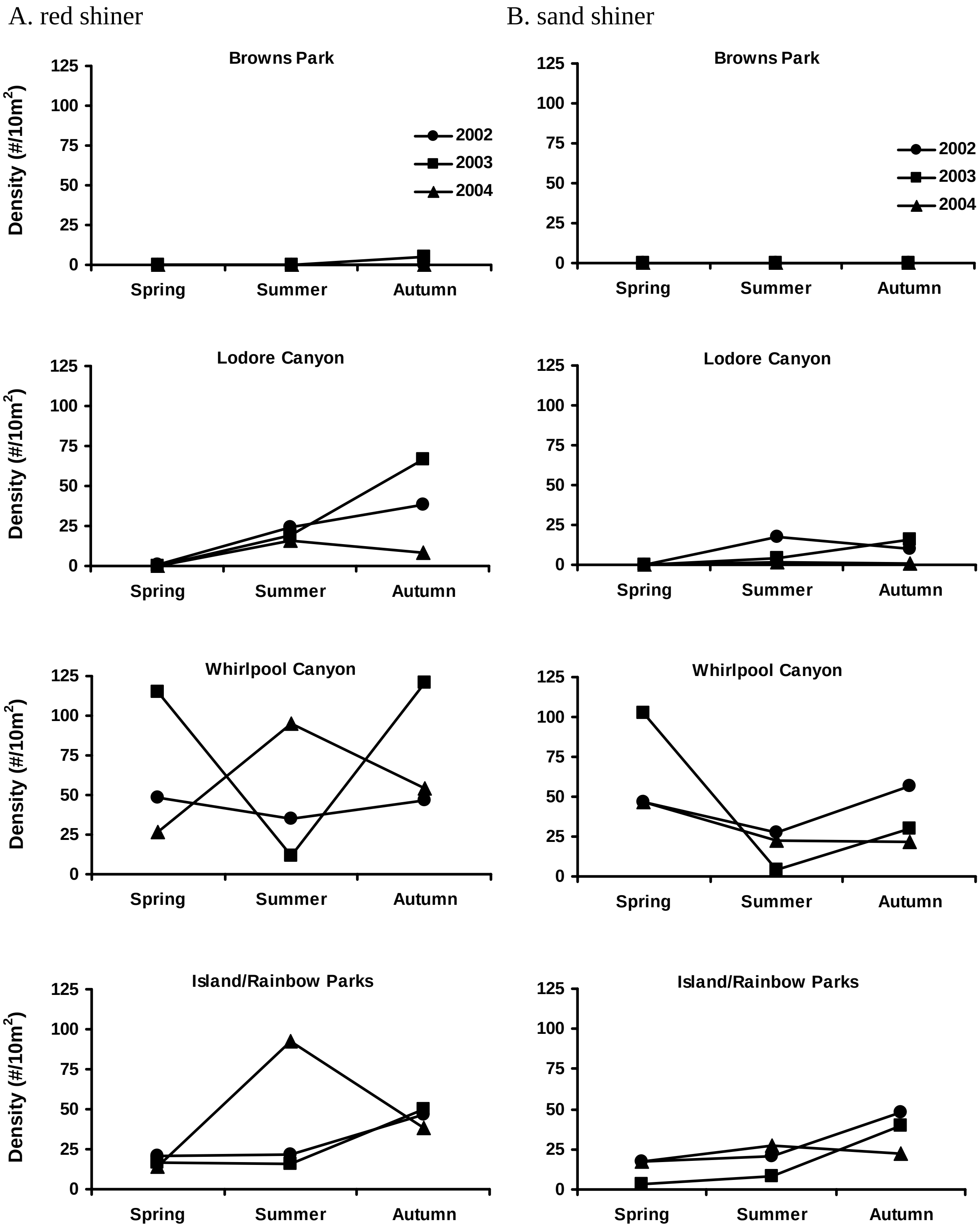


Figure 39.—Mean density of red shiner (A) and sand shiner (B) in seine samples in spring, summer, and autumn in the Green River, 2002-2004. Browns Park = RK 613.0-586.0, Lodore Canyon = RK 585.9-555.3, Whirlpool Canyon = RK 555.2-538.1, Island/Rainbow Parks = RK 538.0-528.0.

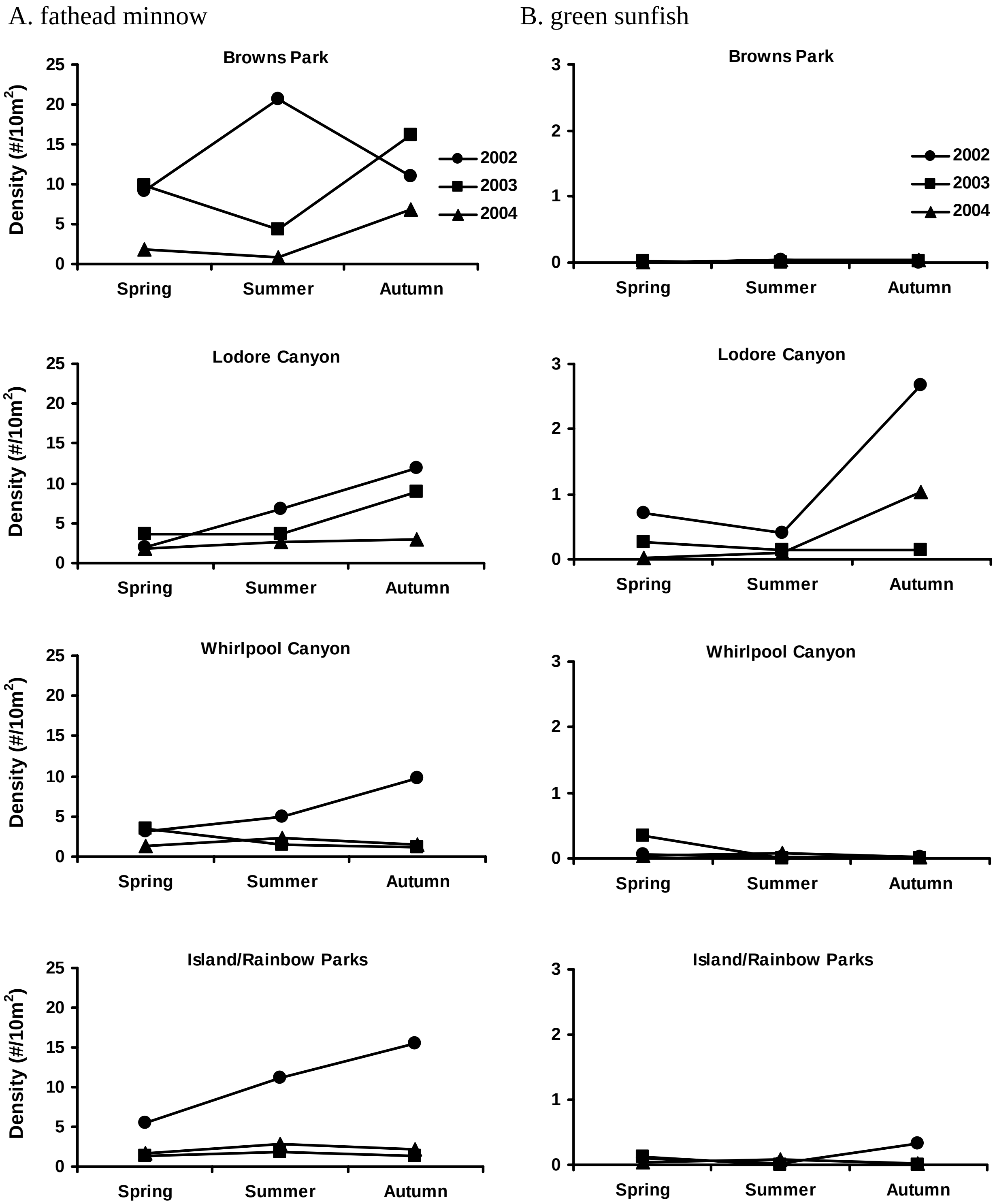


Figure 40.—Mean density of fathead minnow (A) and green sunfish (B) in seine samples in spring, summer, and autumn in the Green River, 2002-2004. Browns Park = RK 613.0-586.0, Lodore Canyon = RK 585.9-555.3, Whirlpool Canyon = RK 555.2-538.1, Island/Rainbow Parks = RK 538.0-528.0.

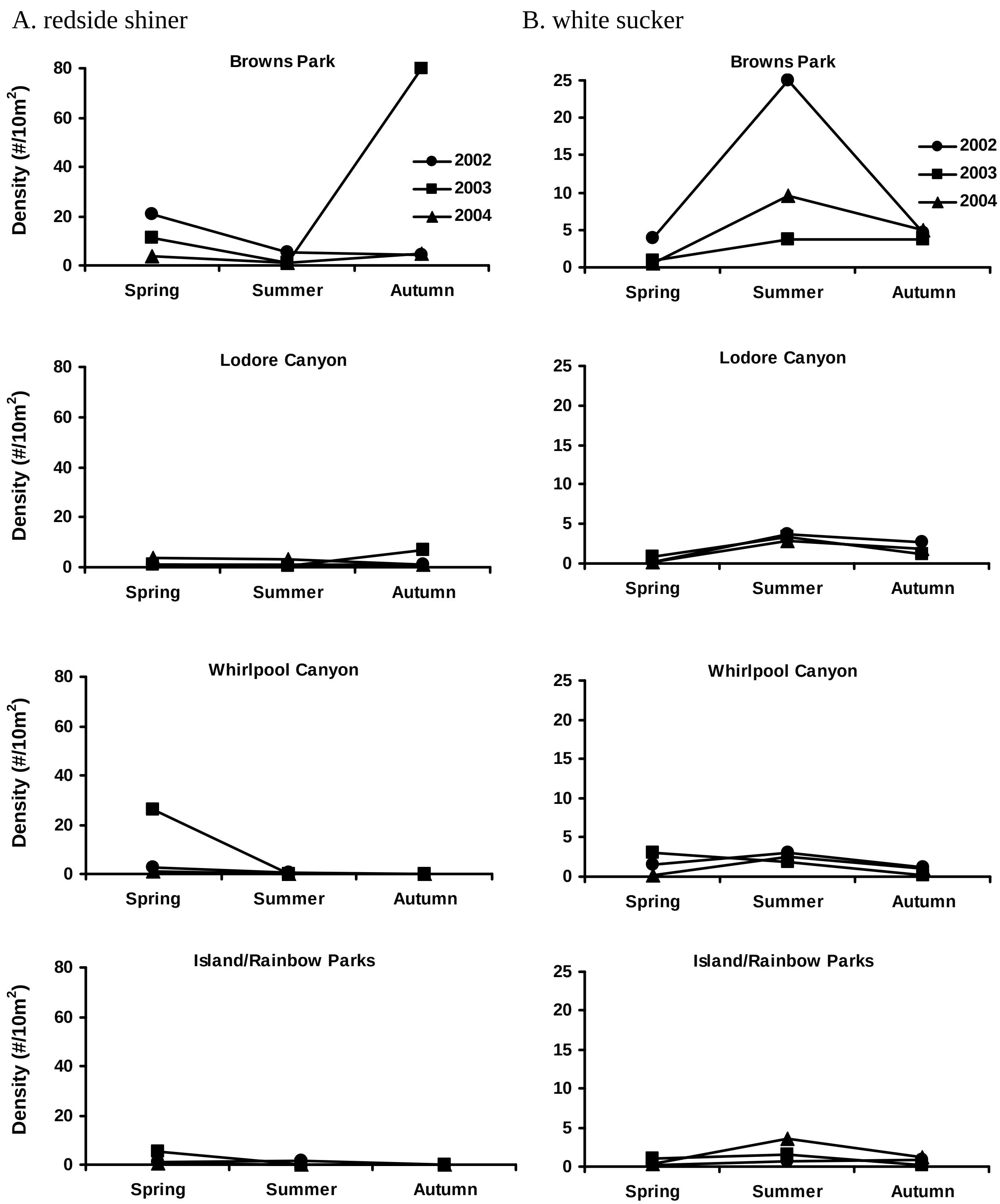


Figure 41.—Mean density of redside shiner (A) and white sucker (B) in seine samples in spring, summer, and autumn in the Green River, 2002-2004. Browns Park = RK 613.0-586.0, Lodore Canyon = RK 585.9-555.3, Whirlpool Canyon = RK 555.2-538.1, Island/Rainbow Parks = RK 538.0-528.0.

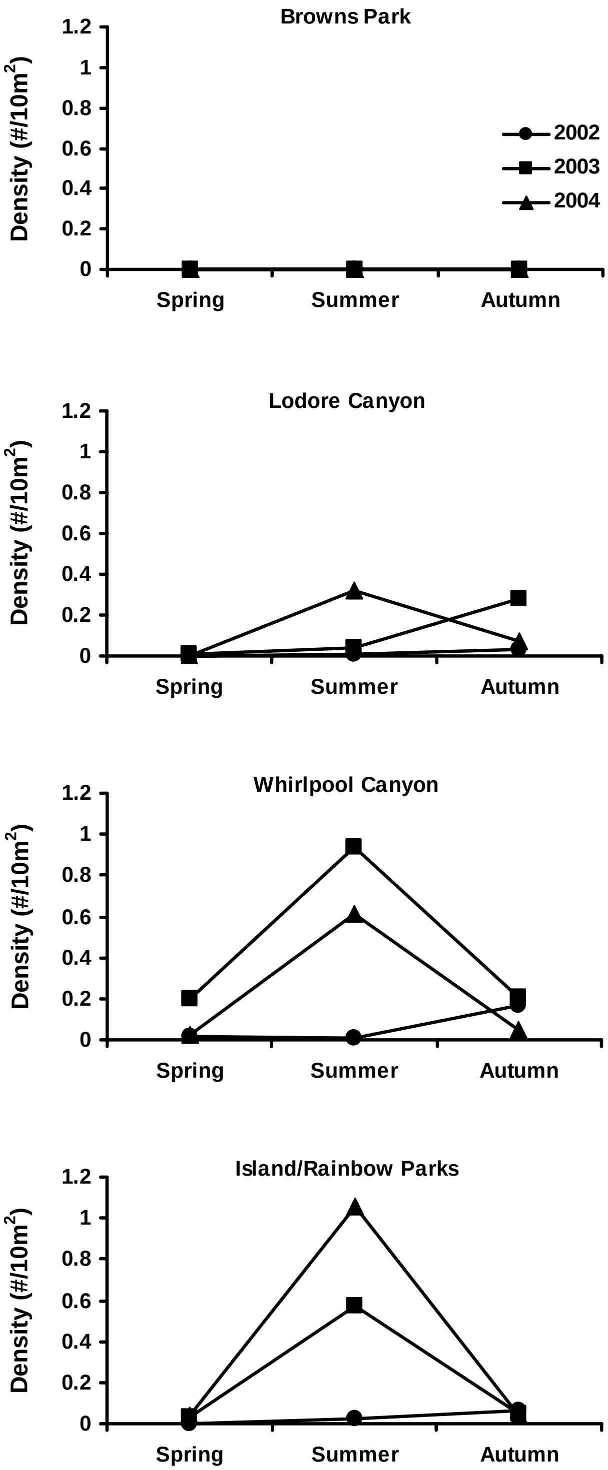


Figure 42.—Mean density of smallmouth bass in seine samples in spring, summer, and autumn in the Green River, 2002-2004. Browns Park = RK 613.0-586.0, Lodore Canyon = RK 585.9-555.3, Whirlpool Canyon = RK 555.2-538.1, Island/Rainbow Parks = RK 538.0-528.0. No smallmouth bass were captured in Browns Park.

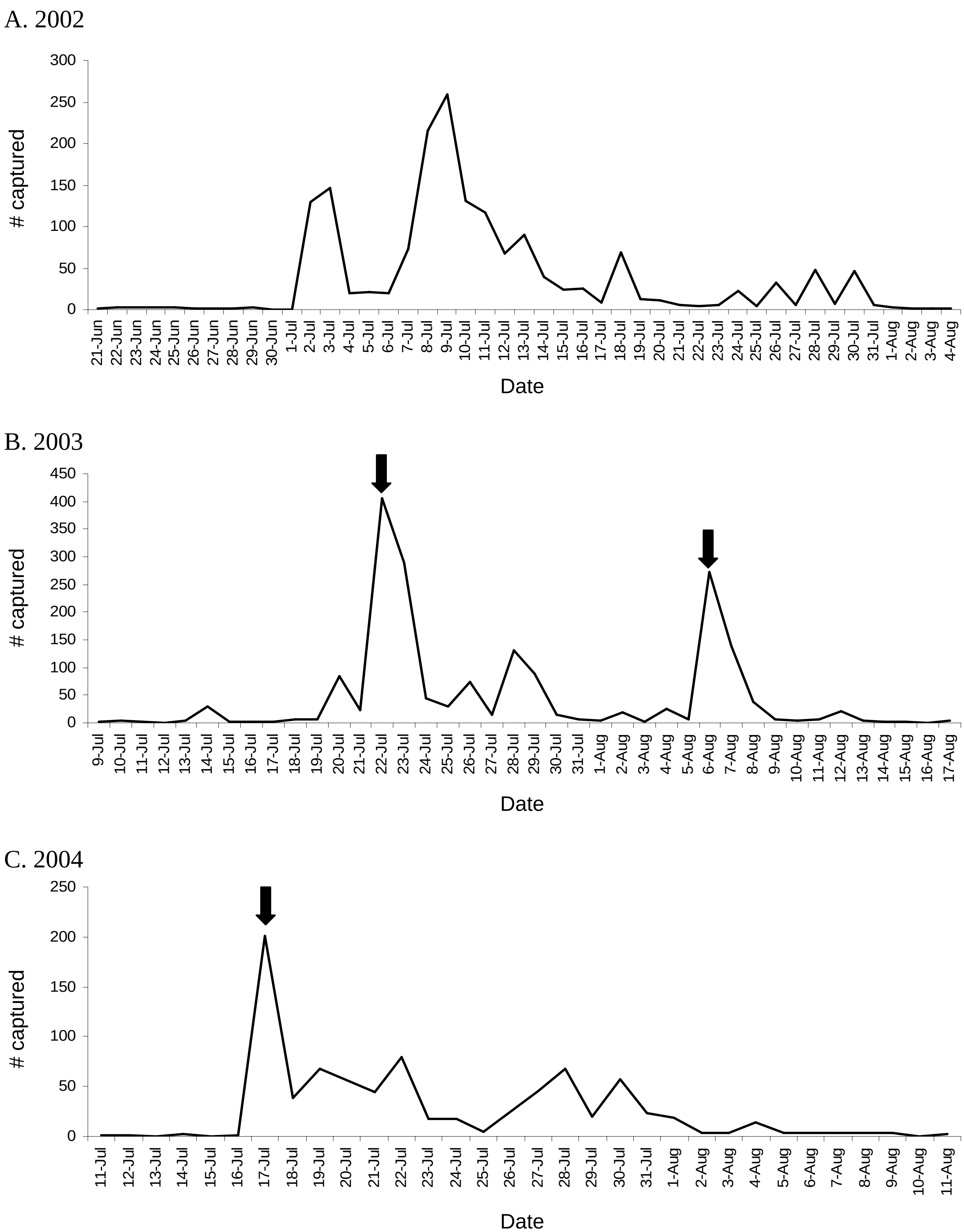


Figure 43.—Total fish captured per day in drift nets in lower Lodore Canyon, Green River (RK 556.0), in 2002 (A), 2003 (B), and 2004 (C). Arrows indicate occurrence of high flow and turbidity events caused by thunderstorm runoff.

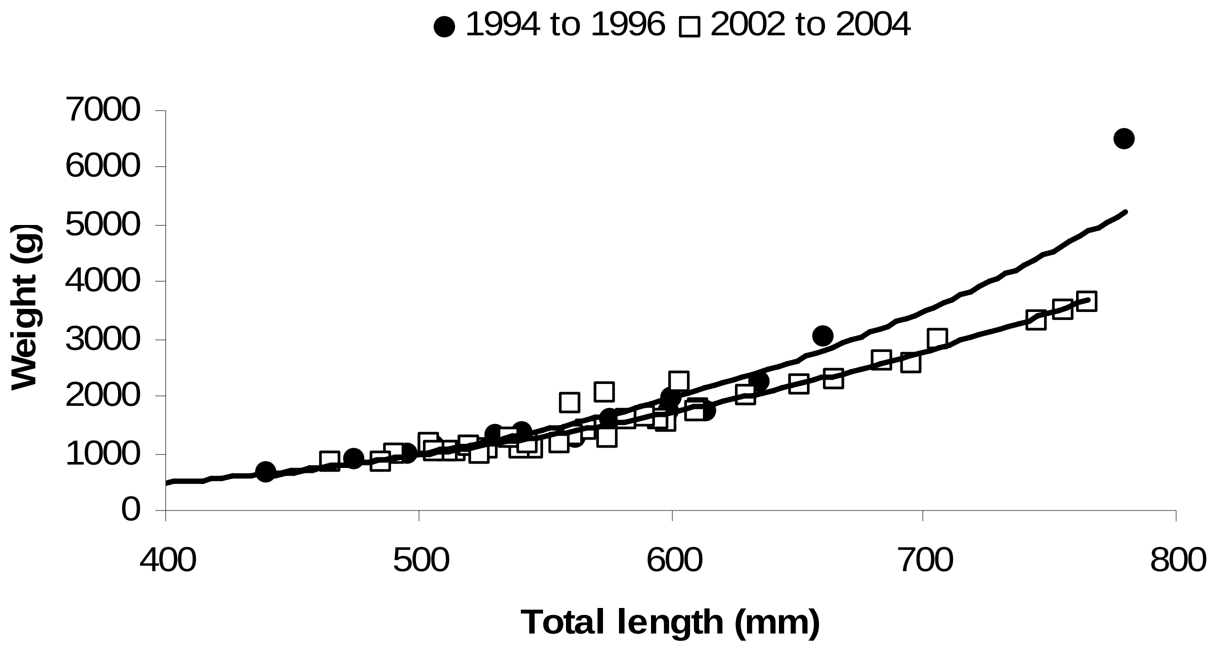


Figure 44.—Length-weight relationships for Colorado pikeminnow captured in Lodore Canyon in the periods 1994 to 1996 and 2002 to 2004.

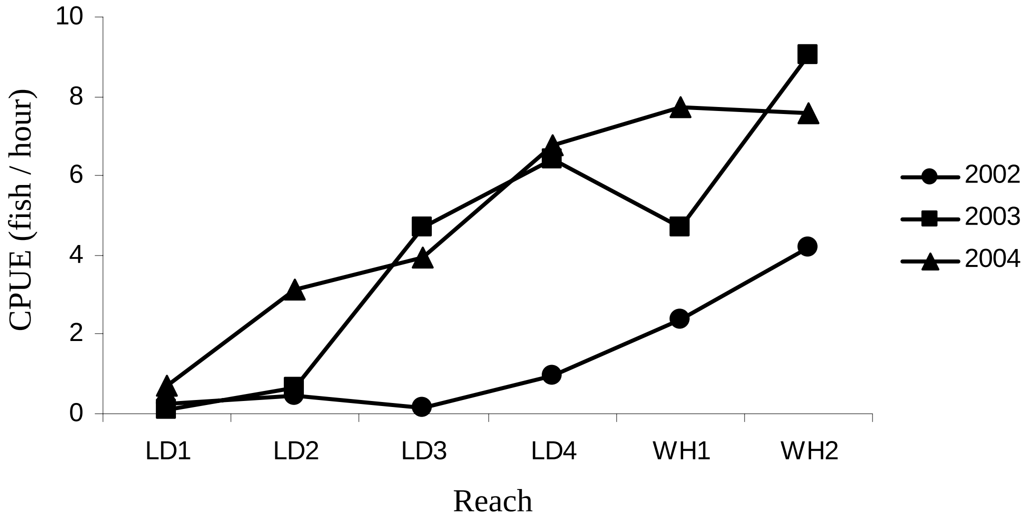
A.

B.

C.

Figure 45.—Humpback chub (288 mm TL) captured 8 October 2002 in Whirlpool Canyon, Green River (A); humpback chub (360 mm TL) captured in Grand Canyon, Colorado River (B); roundtail chub (393 mm TL) captured 14 September 2005 in Whirlpool Canyon, Green River (C).

A. smallmouth bass



B. brown trout

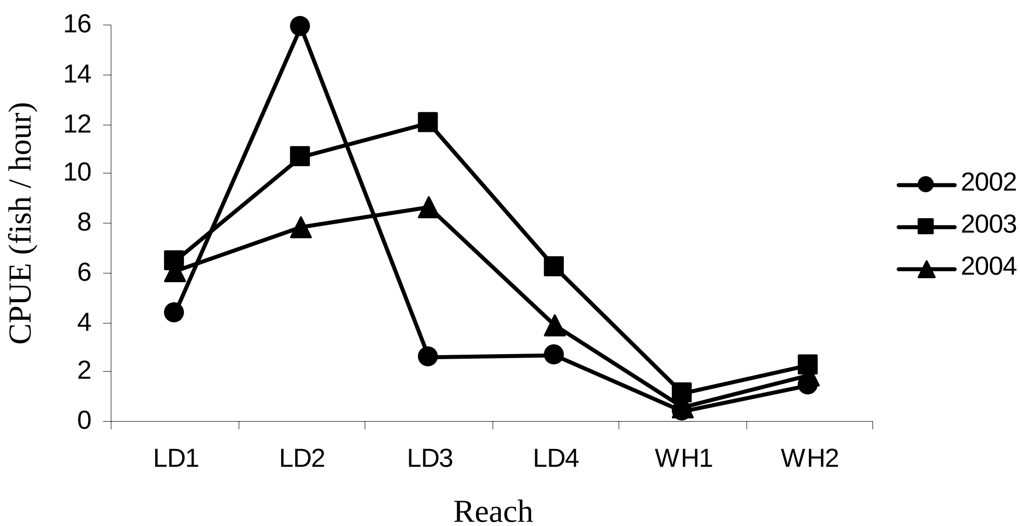


Figure 46.— Catch per unit effort (number of fish captured per hour of electrofishing) of smallmouth bass (A) and brown trout (B) in electrofishing samples from the Green River downstream of Flaming Gorge Dam, Colorado and Utah, 2002-2004. Sampling areas include four Lodore Canyon reaches (LD1, RK 585.9 to 579.0; LD2, RK 578.9 to 571.0, LD3, RK 570.9 to 563.0; LD4, RK 562.9 to 555.3) and two Whirlpool Canyon reaches (WH1, RK 555.2 to 546.6; WH2, RK 546.5 to 538.1).