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Bower, Michael R., *Distributions and habitat associations of bluehead suckers, flannelmouth suckers, and roundtail chubs in the upper Muddy Creek watershed of southern Carbon County, Wyoming*, M.S., Department of Zoology and Physiology, August, 2005.

*Abstract*—The native fishes of the Upper Colorado River Basin have experienced dramatic declines in range and abundance over the last 100 years. Of those species not currently listed as threatened or endangered, three of the least studied are the bluehead sucker *Catostomus discobolus*, flannelmouth sucker *Catostomus latipinnis*, and roundtail chub *Gila robusta*. It was my goal to describe the distribution of each of these three species and their habitat associations within the upper Muddy Creek watershed of southern Carbon County, Wyoming. Fish and habitat were sampled from 57 randomly located 200-m reaches during the summer and fall of 2003 and 2004. Roundtail chubs were the most common of the three species and were found throughout the study area. Bluehead suckers were also common and widely distributed, but were most abundant within a confined area. Flannelmouth suckers were the least common species and were found sporadically. Introduced creek chubs *Semotilus atromaculatus* and white suckers *Catostomus commersoni* were abundant. Hybrids of the two species of native suckers and white suckers were commonly observed. The most notable habitat associations for all three species were their occurrence in pools with rock substrates. Roundtail chubs showed positive associations with reaches that had no observable surface flow and contained remnant pools whereas both bluehead suckers and flannelmouth suckers showed negative associations with these reaches. Management issues result from exotic fishes, the disruption of stream connectivity, and alteration of watershed processes that create and maintain habitat diversity.

# DISTRIBUTIONS AND HABITAT ASSOCIATIONS OF BLUEHEAD SUCKERS, FLANNELMOUTH SUCKERS, AND ROUNDTAIL CHUBS IN THE UPPER MUDDY CREEK WATERSHED OF SOUTHERN CARBON COUNTY, WYOMING

By
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# Chapter I

# Introduction

## Background

Native fishes of the Upper Colorado River Basin (UCRB) have experienced dramatic declines and several extirpations within the last 100 years (Miller 1961; Minckley and Deacon 1968; Rinne and Minckley 1991). The native fish assemblage in the UCRB within Wyoming was composed of 11 species (see Table 1.1; Baxter and Simon 1970; Wheeler 1997). Four of these species were coldwater fishes (i.e., Colorado River cutthroat trout, mountain whitefish, mountain sucker, and mottled sculpin) and the remainder were warmwater fishes. Among the seven warmwater fishes, six species could be classified as warmwater riverine fishes in that they are most frequently associated with river or relatively large tributary habitats. The six warmwater riverine fishes are bluehead sucker, flannelmouth sucker, razorback sucker, roundtail chub, bonytail, and Colorado pikeminnow.

Riverine habitat in the UCRB of Wyoming has changed since settlement by Europeans, primarily in association with water development. Additionally, stream channel degradation has occurred as a result of various land uses and development activities. Long portions of the Green River in Wyoming have been inundated by reservoirs (i.e., Fontenelle and Flaming Gorge reservoirs). Due to hypolimnetic discharge from Fontenelle Dam, the river has been converted to a coldwater stream capable of supporting salmonids downstream to Flaming Gorge Reservoir. Storage of water and sediment within Fontenelle Reservoir and smaller reservoirs in the watershed have led to moderation of the magnitude of spring floods and substantial loss of

habitat diversity associated with channel incision and elimination of frequent over bank flows. Similarly, storage and subsequent release of water from reservoirs have attenuated the hydrographs of rivers once dominated by spring runoff originating from snowmelt. Additionally, construction of small dams to divert water from tributary streams has reduced summer flows in some portions of tributary streams. The construction of numerous reservoirs and irrigation diversion dams across the basin has also created many barriers to upstream movements by fishes.

The fish fauna in the UCRB of Wyoming has changed due to the introduction of 20 exotic fishes (Table 1.1). Seven of the introduced fishes are coldwater species (i.e., rainbow trout, golden trout, kokanee, brown trout, brook trout, lake trout, and Arctic grayling). Additionally, two exotic subspecies of cutthroat trout have been introduced from the Snake River and Bear River watersheds. The other 13 fishes are generally recognized as warmwater species. These exotic warmwater fishes include piscivorous species that can prey upon native riverine species, numerous species that compete for food and habitat, and some that can hybridize with native fishes.

The cumulative effects of habitat alterations and introductions of exotic fishes have probably caused the extirpation of three of the six species of warmwater riverine fishes of the Colorado River drainage within Wyoming. Razorback suckers, bonytails, and Colorado pikeminnows appear to have been extirpated from the Green River when Flaming Gorge Reservoir was impounded and fish toxicants were applied in the upstream watershed to eradicate non-game fishes (Baxter and Simon 1970). Colorado pikeminnows were recently captured in the Little Snake River in Wyoming, but their occurrence in Wyoming is exceedingly rare. The three remaining native warmwater species (i.e., roundtail chub, flannelmouth sucker, and bluehead sucker) have been impacted by the array of changes in the UCRB in Wyoming.

Of the native fish fauna of the UCRB, three of the least studied species are the roundtail chub, flannelmouth sucker, and bluehead sucker. These three species share several morphological characteristics and life history adaptations with other big-river species that have evolved in the harsh and highly variable conditions of the UCRB including streamlined body forms, leather-like skins, large fins, and long life spans (Bezzerides and Bestgen 2002).

A recent status review by Bezzerides and Bestgen (2002) found that in the UCRB roundtail chubs currently occupy approximately 55%, flannelmouth suckers persist in approximately 50%, and bluehead suckers inhabit approximately 45% of their historical habitats. Reasons cited for the observed declines of these three species in the UCRB include the detrimental effects of dams, introduced competitors and predators, habitat degradation, and hybridization (Wiltzius 1978; Bestgen and Probst 1989; Martinez et al. 1994; Bestgen and Crist 2000).

The distribution of roundtail chubs in Wyoming is currently believed to be restricted to Muddy Creek in the Little Snake River drainage, several locations within the Blacks Fork drainage including the Hams Fork, Muddy Creek, and mainstem Blacks Fork, as well as some lakes with outlets into the upper Green River watershed (Gill et al. in press). During a 1965 survey, roundtail chubs were found at the downstream end of the Big Sandy River (Baxter and Simon 1970), but more recent surveys have failed to detect their presence (Wheeler 1997; Gill et al. in press)

Flannelmouth suckers have a somewhat wider distribution than roundtail chubs in Wyoming. Surveys by the Wyoming Game and Fish Department (WGFD) and University of Wyoming have found them in the Green River upstream and downstream from Fontenelle Reservoir and in other portions of the Green River drainage including the Big Sandy River, Little

Sandy River, Bitter Creek, Blacks Fork, Hams Fork, Smiths Fork, Muddy Creek, and Henrys Fork (Wheeler 1997; Gill et al. in press). They are also found in the Little Snake River-Muddy Creek watershed (Baxter and Simon 1970). Of these locations, Bitter Creek represents the only waterbody where flannelmouth suckers are not known to be sympatric with introduced white suckers. Recent surveys by Bureau of Land Management (BLM) and WGFD biologists indicate that their reproduction in Wyoming may be limited to sites in the Little Snake River or Muddy Creek and to two tributaries to the Green River, the Big Sandy River and Bitter Creek (Kevin Gelwicks, WGFD, personal communication).

Bluehead suckers may have the most limited distribution of the three surviving native warmwater riverine fishes within the UCRB in Wyoming. Recent surveys by the WGFD within the Green River drainage have documented the presence of bluehead suckers in a small reservoir on the Henrys Fork (Ringdahl Reservoir), the Big Sandy River, the Little Sandy River, the mainstem Green River, and the Blacks Fork (Wheeler 1997; Gill et al. in press). They are also known to occur within the Little Snake River drainage, including the Muddy Creek and Savery Creek watersheds (Baxter and Stone 1995; Wheeler 1997). The only site where bluehead suckers are not known to be sympatric with introduced white suckers is Ringdahl Reservoir (Kevin Gelwicks, WGFD, personal communication).

The Wyoming Natural Diversity Database has described the status of roundtail chubs as G2/G3/S2, flannelmouth suckers as G3/G4/S3, and bluehead suckers as G4/S2/S3, where G2 = globally imperiled or vulnerable to extinction, G3 = globally rare or locally rare throughout its range, G4 = apparently secure but rare in parts of its range, S2 = imperiled in Wyoming, and S3 = vulnerable to extinction in Wyoming. The WGFD identifies all three species as ones with on-going significant losses of habitat with populations that are greatly restricted or declining, and

extirpation in Wyoming appears possible. The BLM identifies all three species as sensitive species in Wyoming, Colorado, and Utah.

The Wyoming portion of the UCRB is composed of two watersheds, the Little Snake River and the Green River. The Little Snake River watershed is the smaller of the two (7,059 $km^2$). The watershed is bounded on the east by the Continental Divide and the primary source of water is runoff and groundwater originating in the Sierra Madre. The Little Snake River represents one of the few rivers within the UCRB where physical habitat alterations do not appear to have significantly altered hydrologic or geomorphic processes. Seasonal fluctuations in water and sediment transport within this unregulated river system can be extreme. These extremes are demonstrated by high peaks in stream discharge during runoff events, and conditions approaching zero flow at base flow (Hawkins and O'Brien 2001). During base-flow conditions, fish habitat is typified by scattered remnant pools connected by minimal surface flow. These remnant pools can experience diel temperature fluctuations of nearly 20 $^{\circ}$C (Hawkins et al. 1997). Sampling by Hawkins et al. (1997; 2001) downstream of Baggs, Wyoming during 1994 and 1995 found a paucity of nonnative fishes while roundtail chubs, flannelmouth suckers, and bluehead suckers were the three most abundant species within the Little Snake River. It was suggested that the numerical dominance of the ichthyofauna of the Little Snake River by native fishes may be due to the often extreme hydrologic conditions including its peak-to-base flow ratio, large sediment load, and extremely low base flow, as well as resulting physico-chemical characteristics involving water temperature, water quality, and sediment transport (Hawkins et al. 2001).

## Study area

As a major tributary to the Little Snake River, Muddy Creek originates in the foothills of the Sierra Madre where it is typical of coldwater foothills streams before transitioning into a warmwater stream and joining the Little Snake River near Baggs, Wyoming. The watershed includes Muddy Creek and several tributaries including McKinney Creek and Littlefield Creek. Muddy Creek exhibits seasonal fluctuations in streamflow and sediment transport (Goertler 1992) similar to the Little Snake River. The hydrograph is dominated by runoff originating from snowmelt in the spring followed by periods of low flow during summer. Precipitation events through the summer and into the fall frequently result in stormflows and measurable streamflows in late fall. During dry years, Muddy Creek could be classified as an intermittent stream for much of its length (Goertler 1992). A series of mainstem impoundments occur on Muddy Creek approximately halfway between its headwaters and mouth. Several impoundments also occur on ephemeral tributaries to Muddy Creek. It is suspected that these impoundments affect hydrologic conditions and geomorphic processes of importance to the maintenance of native fish communities (Beatty 2005).

The study area included 82 km of stream habitat: 63 km of Muddy Creek, 16 km of McKinney Creek, and 2 km of Littlefield Creek (Figure 1.1). The downstream boundary of the study area was a large headcut stabilization structure on Muddy Creek that inhibited the upstream movement of fishes. The upstream boundary on Muddy Creek was a road crossing and associated culvert that inhibited the upstream movement of fishes. McKinney Creek was sampled upstream from its mouth to a point where only brook trout were found. The study area included the lowest 2 km of Littlefield Creek below a barrier to upstream movement by fish (Figure 1.1).

Several instream structures that may function as barriers to the upstream movement of fishes were located within the study area. The effectiveness of these potential barriers in inhibiting or precluding the upstream movement of fishes is currently unknown, but likely varies as a function of their design and stream discharges. An irrigation diversion structure occurred on Muddy Creek between the downstream boundary of the study area and the mouth of McKinney Creek. During high streamflow conditions, this structure may be navigable by fishes moving upstream. A fish barrier was constructed on Muddy Creek immediately upstream of the confluence with McKinney Creek to inhibit the recolonization of nonnative fishes following removal efforts to re-establish native coldwater fishes in the system. Based strictly on design specifications, this barrier is likely to preclude upstream fish movements at nearly all discharge levels. Several "gully plugs" were also found within Muddy Creek between the confluence with McKinney Creek and the upstream boundary. The design of these "gully plugs" varied greatly and it is suspected that their effectiveness in inhibiting or precluding the upstream movement of fishes varies.

Land uses within the study area included livestock grazing and recreation. Future land uses may include development of natural gas and coal extraction. Coalbed methane development is occurring in the Muddy Creek watershed and future expansion into the study area is likely as a result of the development of the Atlantic Rim Coalbed Natural Gas Project.

## Management Issues

One management issue in the watershed is alteration of the dynamics (i.e., seasonal variability and long-term patterns) of streamflow in the mainstem of Muddy Creek and its tributaries associated with human uses of the watershed and the effects of this alteration on the

three native fishes. Much of the Muddy Creek watershed experiences intermittent surface flows during summer and fall with these areas typified by isolated remnant pool habitats. It is not known how construction of ponds for livestock water, wetlands for wildlife habitat, or diversions for irrigation affect the dynamics of flows and availability of suitable habitat for native fishes across the watershed. Additionally, it is not known how the discharge of waters produced during natural gas extraction or the alteration of groundwater aquifers by such activity, both potentially associated with future extraction of coalbed methane, may affect hydrologic processes and fish habitat.

Another management need is the definition of the habitat and flow needs of native fishes in the watershed. Some efforts have been made to define the needs of these fishes in larger rivers, but their needs in small tributary streams are unknown (Bezzerides and Bestgen 2002). Such information is needed to assess the effects and manage the development of livestock watering ponds, wetlands, water diversions for irrigation, and coalbed methane wells across the Muddy Creek watershed. Information is also needed to define the desired future condition of stream habitat in this watershed and to set stream habitat management goals and priorities.

A third management issue is the effect of existing and potential future barriers to movements by native fishes in the Muddy Creek watershed. Muddy Creek is currently fragmented into distinct segments by water diversion structures, constructed wetlands, a large headcut stabilization structure, and barriers constructed to prevent the recolonization of reintroduction areas by nonnative fishes. Several structures prevent upstream movements of fishes and have isolated and fragmented populations of the three native fishes in the upstream portion of the watershed. A reintroduction area for Colorado River cutthroat trout has been identified in the upper Muddy Creek watershed beginning immediately upstream from the mouth

of McKinney Creek. The fish barrier on Muddy Creek just upstream of the confluence with McKinney Creek was constructed to isolate this headwater portion of the watershed. Within the reintroduction area, fishes are being removed using piscicides in order to eliminate brook trout. Colorado River cutthroat trout have been reintroduced to 18 km of Littlefield Creek. The remainder of the reintroduction area will be treated in the near future. In addition to Colorado River cutthroat trout, native fishes known to occur in the reintroduction area include flannelmouth suckers, mountain suckers, and speckled dace. It is not known how the design of the reintroduction area, including the location of the fish barrier on Muddy Creek and the implementation of chemical fish removals within warmwater habitats, will affect populations of bluehead suckers, flannelmouth suckers, or roundtail chubs.

## Purpose

Goal: To determine the physical features and distributions of habitats that are required to maintain populations of bluehead suckers, flannelmouth suckers, and roundtail chubs in the upper Muddy Creek watershed.

Stream fishes often require unique habitats for spawning, feeding, rearing, and refuge. The spatial heterogeneity and connectivity of a stream system can necessitate the movement of fishes among habitats in order to complete their life cycles (Schlosser 1995). This research was designed to determine both the use and selection of habitats by the three target species in the upper Muddy Creek watershed. Reported habitat associations have focused on larger streams and rivers such as the Colorado River (Beyers et al. 2001) and San Juan River (Gido and Probst 1999); whereas, my focus was on small tributary streams. Small tributary streams may be of high conservation value by providing seasonal or permanent refuges from mainstem habitat

alterations and introduced fishes (Chart 2000; Bezzerides and Bestgen 2002). Populations of roundtail chubs, flannelmouth suckers, and bluehead suckers are currently isolated in the upper reaches of the Muddy Creek watershed by a series of artificial wetlands and a large headcut stabilization structure. These populations may be of high conservation value due to their isolation from mainstem habitats and their limited spatial extent may allow conservation at a manageable scale.

## Objectives

My research objectives were to:

(1) describe the distribution and relative abundance of the three target species (bluehead sucker, flannelmouth sucker, and roundtail chub), introduced species that may be predators or competitors (creek chub and brook trout), and species with the potential to hybridize with the target species (white sucker) at each study site, and

(2) determine the spatial distributions and physical features of habitats used and selected by juveniles and adults of the three target species during summer and fall.

To accomplish these objectives, I utilized a stratified random approach to identify sampling reaches throughout the study area. Chapter II presents the distributions of bluehead suckers, flannelmouth suckers, and roundtail chubs in the upper Muddy Creek watershed. Knowledge of these distributions is vital to the identification of biologically meaningful conservation and management strategies for the three target species. Knowledge of distributional patterns can lead to several additional hypotheses regarding the ecology of the three target fishes within the upper Muddy Creek watershed. Chapter III presents findings from

resource selection analyses for bluehead suckers, flannelmouth suckers, and roundtail chubs in the upper Muddy Creek watershed. Information pertaining to the habitat associations of the three target species is important when considering land management actions or when designing habitat management plans. Considerations for the management of the three target species and their associated habitats are presented in Chapter IV. Future research needs are presented in Chapter V.

## Chapter II

## Distributions and Relative Abundance of Bluehead Suckers, Flannelmouth Suckers, and Roundtail Chubs in the Upper Muddy Creek Watershed

*Abstract.* –My objective was to describe the distributions of bluehead suckers, flannelmouth suckers, and roundtail chubs in a small tributary stream system, the upper Muddy Creek watershed of southern Carbon County, Wyoming. Fish were sampled in 57 randomly selected 200-m reaches during the summer and fall of 2003 and 2004. Eight fish species were found within the study area. Two of these species, white suckers and creek chubs, were introduced fishes. Hybrids of native bluehead suckers and flannelmouth suckers with introduced white suckers appeared to be common. Bluehead suckers were widely distributed, and were most abundant within a relatively short length of Muddy Creek bounded by potential barriers to upstream movement by fishes. Flannelmouth suckers were also widely distributed though less abundant. Roundtail chubs were widely distributed throughout the warmwater portions of the study area prior to August, but after August 1st they were not detected within the lower portions of the study area when long reaches with no surface flows were observed. The negative effects of exotic fishes on populations of the three target species likely included hybridization and competition.

## Introduction

The occurrence of sympatric populations of bluehead suckers, flannelmouth suckers, and roundtail chubs in the upper Muddy Creek watershed provides a unique opportunity for the conservation of these native warmwater fishes within Wyoming. The watershed has been a focus of native fish and habitat restoration activities since the mid-1980's. Efforts have focused on the restoration of habitat conditions necessary to support coldwater fish communities (i.e., Colorado River cutthroat trout, mountain sucker) within the headwater reaches of Muddy and Littlefield creeks, the chemical removal of exotic fishes, and the reintroduction of the native coldwater fish community (Little Snake River Working Group 1994). Given the known presence of sympatric populations of bluehead suckers, flannelmouth suckers, and roundtail chubs within or in close proximity to identified reintroduction areas for native coldwater fishes, information pertaining to the distribution and abundance of these three species is needed in order to take a holistic approach to native fish community management within the watershed.

The lack of information related to factors affecting the persistence of populations of these three native species within the upper Muddy Creek watershed has limited the development of biologically meaningful conservation and management strategies. Factors cited as contributing to declines in the distribution of the three species in the UCRB include the detrimental effects of dams, introduced competitors and predators, habitat degradation, and hybridization (Wiltzius 1978; Bestgen and Probst 1989; Martinez et al. 1994; Bestgen and Crist 2000). Additionally, it has been speculated that the maintenance of natural flow regimes and highly variable environments typical of the Colorado River Basin may be a key factor accounting for the persistence of native fishes, particularly in the presence of introduced competitors (Quist et al. 2004a).

My objectives were to characterize the hydrologic conditions present within the study area, describe the distribution and relative abundance of populations of the three target species during summer and fall, and identify factors that may affect populations of the three species within the upper Muddy Creek watershed.

## Methods

*Hydrologic conditions*

Streamflow conditions, water temperatures, and precipitation were measured at a series of hydrologic monitoring sites (Figure 2.1; Appendix A1). Seven sites were selected to monitor streamflow within the study area based on the addition of streamflow at tributary streams and drainages. These sites were in pools upstream of hydraulic controls and consisted of a stilling well that was fitted with an AquaRod® to measure stream stage. The stilling well was slightly off channel with a 2.5-cm-diameter pipe connecting the well to water within the pool. Within 24 h of deployment, the AquaRods® were "wetted" by submersing them in water up to the vents that are just below the base of the pod for a period of 24 h (Sequoia Scientific Inc.). Aquarods were set to record stream stage every 15 minutes. Data accumulated by the AquaRods® were downloaded monthly over the period from June through November of 2003. Streamflow measurements were made using the cross-sectional area method (Gallagher and Stevenson 1999) at these sites over the course of the year in order to establish a stage-discharge relationship. Measurements were made during the summer months and during low flow conditions in the fall.

Ten sites were selected to monitor water temperatures based on the location of tributary streams and drainages with additional locations in upper McKinney Creek to characterize the upper thermal limits for warmwater fishes (Figure 2.1, Appendix A1). These sites were in the

thalweg of pools and consisted of either an Onset StowAway® or a HOBO® temperature logger attached to a post that was driven into the streambed. The loggers were set to collect readings every 15 minutes and data were downloaded monthly.

Three precipitation gauges were installed in the study area at sites 1, 2, and 5 (Figure 2.1, Appendix A1). These consisted of tipping bucket rain gauges attached to event loggers. The event loggers recorded the time of each tip corresponding to 0.1 mm of precipitation and they were downloaded monthly over the period from June through November of 2003.

*Fish sampling*

The study streams were divided into 200-m reaches and 57 of these reaches were randomly selected for sampling. Patton et al. (2000) found that sampling 200-m of stream generally captured all of the fish species present among warmwater streams in the Missouri River drainage of Wyoming. I assumed that all species present would be detected within 200-m of stream in the upper Muddy Creek watershed.

Reaches were randomly assigned to one of two time strata to allow temporal differences in fish community and physical habitat features to be assessed. An equal proportion of the reaches were assigned to each time strata. These strata consisted of early and late sampling periods. Early sampling was conducted from June 17 through July 31. Late sampling was conducted from August 1 through October 8. Within both time strata, the chronological order in which reaches were sampled was randomly generated.

Each pool or run within a selected reach was sampled using electrofishing and seining in order to account for the differential species and size selectivity of the two gears. The pool or run was isolated by placing block nets at the upstream and downstream boundaries of the channel unit. A single electrofishing pass was made and all fish collected were identified and the total

length (TL) was measured to the nearest millimeter. These fish were temporarily retained. The channel unit was then sampled by seining, where possible, with a 4.8-mm mesh bag seine, 4.6 m long by 1.2 m in height, with a 1.2 $m^3$ square bag sewn into the center of the net. All fish collected by seining were identified and TL was measured to the nearest millimeter. All fish collected during electrofishing and seining efforts were then returned to the channel unit. Though rare, it was occasionally not possible to identify young-of-year catostomids to the species level. Individuals for which a positive identification was not possible were recorded as "unknown" and excluded from subsequent analyses.

Hybrid catostomids were identified based on intermediate morphologies. The specific morphological features used to distinguish among the three sucker species were derived from Baxter and Stone (1995) and are presented in Table 2.2. Douglas and Douglas (2003) corroborated field identifications of pure and hybrid suckers within the Yampa River using nuclear intron analyses. They found that hybrids could be identified in the field based on intermediate phenotypes with less than 5% error.

Distributional patterns for the three species were presented for only those individuals thought to be adults. Given the paucity of information pertaining to length at maturity for the three species within small tributary streams, these sizes were based on field observations. These observations included tuberculation, expression of gametes, and expression of spawning colorations. The length at maturity estimates were 150 mm TL for bluehead suckers, 200 mm TL for flannelmouth suckers, and 150 mm TL for roundtail chubs. Fish equal to or greater than the length at maturity were termed adults, and fish less than the length at maturity were termed subadults.

## Results

*Hydrologic conditions*

Monitoring sites were operated from mid-June through early-October of 2003 to characterize hydrologic conditions during the first sampling season. Highly variable amounts of precipitation and streamflow, as well as highly fluctuating water temperatures over the course of the summer and fall were observed. A single monitoring site just downstream of the confluence of Muddy and McKinney creeks (Figure 2.1) was chosen to represent precipitation within the study area. Precipitation at this site was highly sporadic, beginning in August and increasing in frequency and intensity into the fall (Figure 2.2).

A single site in close proximity to the selected precipitation gauge was chosen to represent streamflow in relation to precipitation events over the summer and fall of 2003. A significant stage-dischage relationship (i.e., rating curve) was observed ($R^2 = 0.89$). Streamflow conditions showed temporal variation over the summer and fall of 2003, most often as a result of periods of drying followed by increasing streamflow after precipitation events (Figure 2.2). This response to precipitation events resulted in frequent, though brief, stormflows typified by rapid increases in streamflow. Typical baseflow conditions were encountered by early-July though streamflow continued to decline, approaching zero by early August. As precipitation increased into the fall, streamflow increased. The maximum mean daily discharge recorded was 0.17 $m^3/s$ on October 2 in association with the season's first snow storm.

There was significant spatial variation in streamflow. Long stretches of the study streams experienced no surface flows for periods of time, with water persisting in remnant pools. Much of the downstream portion of the Muddy Creek study area had no observable surface flow by the

middle of August, 2003. The majority of McKinney Creek within the study area had no observable surface flow at some time subsequent to mid-August, 2003. No-flow conditions were generally short-lived, being frequently interrupted by continuous surface flows resulting from precipitation events. No-flow conditions were more persistent in the downstream portion of the Muddy Creek study area.

Water temperatures at the nine monitoring stations showed both temporal and spatial variation (Figures 2.3-2.6). In general, maximum water temperatures occurred during baseflow conditions, decreasing into the fall as air temperatures decreased or in response to precipitation. Several sites that were chosen to record water temperature eventually experienced no observable surface flow. Even though attempts were made to locate temperature loggers in remnant pools, there were periods of time when loggers were no longer submerged. At these times, temperatures recorded by the loggers represented ambient air temperatures, not water temperatures. For this reason, describing the maximum water temperatures encountered by the three target species over the course of the study was not possible.

*Fish assemblages*

Eight fish species were sampled in the study area during the summer and fall of 2003 and 2004 (Figure 2.7). Five species (speckled dace, roundtail chub, bluehead sucker, flannelmouth sucker, and mountain sucker) were native to the UCRB and three species (creek chub, white sucker, and brook trout) have been introduced to the UCRB. Native fishes numerically dominated the fish samples, comprising 70% of the fishes collected. Introduced fishes and catostomid hybrids accounted for 30% of the fishes sampled. The two most abundant species were speckled dace (39%) and roundtail chub (24%). Bluehead suckers were abundant within portions of the study area. Flannelmouth suckers were sampled, but much less frequently than

the other native fishes. Both creek chubs and white suckers were relatively abundant. White suckers were more abundant than either bluehead suckers or flannelmouth suckers. Hybrids between the two native sucker species and the introduced white sucker appeared to be common throughout the study area. Hybrids between bluehead suckers and flannelmouth suckers were identified, though rare.

*Distribution of bluehead suckers*

Bluehead suckers were most abundant within a stretch bounded on the downstream end by an irrigation diversion and on the upstream end by the fish barrier and the confluence of Muddy and McKinney creeks (Figure 2.8). This area had an abundance of rock substrates, frequent pool-riffle sequences, and perennial streamflows. Bluehead suckers were present, but less abundant below the irrigation diversion, within the lower to mid-elevations of McKinney Creek, and above the fish barrier on Muddy Creek. A similar distributional pattern of bluehead suckers was observed for both time periods.

*Distribution of flannelmouth suckers*

Flannelmouth suckers were most abundant immediately downstream of the fish barrier and irrigation diversion on Muddy Creek (Figure 2.9). Flannelmouth suckers were present in the lower reaches of McKinney Creek and only a single subadult was collected above the fish barrier in Muddy Creek. A similar distributional pattern of flannelmouth suckers was observed for both time periods.

*Distribution of roundtail chubs*

Prior to August, adult roundtail chubs were widely distributed below the confluence of Muddy and McKinney creeks (Figure 2.10). The areas of highest abundance were immediately

downstream from the fish barrier on Muddy Creek and in a single reach near the downstream limits of the study area on Muddy Creek. Adult roundtail chubs were also present, though less abundant, above the fish barrier on Muddy Creek. After July 31st adult roundtail chubs were not found in the lower portions of the study area, but they were found in portions having perennial streamflow (Figures 2.11 and 2.12).

## Discussion

*Introduced fishes*

The introduction of exotic species is a threat to biological diversity, particularly within the Colorado River basin (Fuller et al. 1999). The presence of introduced creek chubs and white suckers, as well as hybrids between native suckers and white suckers within the study area, suggests that impacts associated with exotic fishes introduced into the UCRB is a concern within the upper Muddy Creek watershed.

Immediately downstream of Muddy Creek in the Little Snake River, Hawkins et al. (1997; 2001) found a fish assemblage dominated by native species (69% in 1994 and 72% in 1995). Though the dynamics maintaining the dominance of native fishes within the Little Snake River remain unknown, Hawkins et al. (1997; 2001) speculated that unregulated streamflows and associated physico-chemical characteristics, such as water temperature, water quality, and sediment transport, may give the native fishes a competitive advantage over exotic fishes. The upper Muddy Creek watershed showed similar extremes in discharge and physico-chemical characteristics Extreme environmental conditions may explain the persistence of a diverse native warmwater fish community in the presence of exotic fishes.

The two most common species collected by Hawkins et al. (1997; 2001) within the Little Snake River were bluehead suckers and flannelmouth suckers. White suckers were present, but hybrids between the two native suckers and white suckers were exceedingly rare. The Little Snake River is a much larger stream than those of the upper Muddy Creek watershed. This may allow for the segregation of populations of the three sucker species, particularly in regards to spawning habitats. Hubbs (1955) suggested limited spawning habitats as one explanation for the hybridization among fish species. Within the upper Muddy Creek watershed, rock substrates generally associated with the spawning habitats of bluehead suckers, flannelmouth suckers, and white suckers are limited.

Though the fish assemblage within the study area was numerically dominated by native fishes, several factors related to the adverse effects of exotic fishes should be considered. The effects of creek chubs upon the native warmwater fishes are unknown, but given high degrees of diet overlap observed in upper Muddy Creek interspecific competition is possible, particularly if food resources are found to be limiting (Quist et al. in press). The effects of white suckers upon populations of bluehead suckers and flannelmouth suckers are more tangible. Hybrids between these species were frequently sampled within the study area. The adverse effects of hybridization among native and exotic fishes include loss of reproductive capacity of the native species and, in cases of introgressive hybridization, can lead to 'extinction via hybridization' (Rhymer and Simberloff 1996).

The presence of a hybrid swarm that may result from extensive backcrossing of hybrid individuals could diminish the conservation value of populations of bluehead suckers and flannelmouth suckers within the study area and require conservation strategies such as chemical removal of the entire fish community and reintroduction from neighboring populations that have

not experienced introgression. In the Yampa River, Colorado, Douglas and Douglas (2003) did not identify the presence of backcrossing of first generation ($F_1$) hybrids between native bluehead suckers or flannelmouth suckers and white suckers.

Based strictly on morphological observations in the field, backcrossing of $F_1$ hybrids seemed likely in the case of flannelmouth sucker x white sucker crosses due to a gradation of morphologies, but less apparent in the case of bluehead sucker x white sucker crosses. Preliminary results from genetic analyses utilizing microsatellite DNA and amplified fragment length polymorphism (AFLP) markers support the morphological evidence, but additional work is needed to identify and quantify levels of introgression in bluehead sucker and flannelmouth sucker populations within the Muddy Creek watershed. The AFLP analyses supported the field identification of greater than 95% of individuals of the three target species and their hybrids, indicating that field identifications were highly accurate (Dave McDonald, University of Wyoming, personal communication).

*Potential movement barriers*

One factor potentially affecting the viability of populations of bluehead suckers, flannelmouth suckers, and roundtail chubs within the upper Muddy Creek watershed is the presence of barriers to upstream movement by fishes. Stream fishes require habitats for spawning, feeding, rearing, and refuge. The spatial heterogeneity and connectivity of the stream system can necessitate the movement of fishes among these habitats in order to complete their life cycles (Schlosser 1995). Interruption of movement among required habitats by the construction of instream structures can have demographic effects, decreasing population viability. The distributions of the three target species during the summer and fall of 2003

suggest several potential implications of habitat fragmentation in regards to access to refuge habitats and subsequent ability to recolonized adjacent reaches.

Water year 2003 was the fourth consecutive year of below average precipitation for much of Wyoming (Swanson et al. 2004). The nearest stream gage, located on the Little Snake River near Slater, Colorado (USGS 09253000), recorded an average annual streamflow for water year 2003 of 4.61 $m^3/s$ compared to a long term average annual streamflow of 6.46 $m^3/s$ based on a 53-year period of record from 1943 through 2002. As such, hydrologic conditions encountered over the course of sampling during 2003 may be thought of as unusual, though likely within the range of variability within which populations of native fishes have evolved within the upper Muddy Creek watershed.

During 2003, refuge habitats consisting of remnant pools within stretches having no observable surfaced flow were a significant feature of the upper Muddy Creek watershed. The ability of fishes to access refuge habitats during drought conditions and recolonize adjacent reaches following drought have been shown to be major factors influencing fish communities in streams experiencing extensive drying (Magoulick 2000; Lake 2003; Magoulick and Kobza 2003; Scheurer et al. 2003). Given the prevalence of stream channel drying and associated remnant pool habitats in some areas during sampling, instream structures may affect the viability of populations of bluehead suckers, flannelmouth suckers, and roundtail chubs by disrupting access to refuge habitats or subsequent recolonization of the stream system.

Both bluehead sucker and flannelmouth sucker adults were associated with areas of perennial streamflow, possibly lessening their reliance on access to pools as refuge habitats. Another explanation for the observed abundances of these two species may be that the distributions of these two species have already been influenced by movement barriers. The areas

of highest abundance of these two species may represent source habitats with sink habitats occurring below movement barriers in areas where remnant pools eventually became isolated as a result of stream drying.

Early in the summer adult roundtail chubs were distributed in areas that eventually experienced no observable surface flows. By August adult roundtail chubs were not detected in these areas, suggesting three possibilities. First, they may have moved. Second, they may have experienced mortality within drying pools. Third, the resolution of sampling during the late period may have been insufficient to detect the presence of adult roundtail chubs given the limited spatial distribution of remnant pools within the lower portions of the study area. If adult roundtail chubs exited the study area through the downstream boundary, they were displaced over the headcut stabilization structure that would prohibit their recolonization of the lower reaches of the study area. Seasonal movements of roundtail chubs between canyon reaches in the winter and valley reaches in the summer have been observed in Aravaipa Creek, Arizona (Siebert 1980).

*Conclusions*

The occurrence of sympatric populations of bluehead suckers, flannelmouth suckers, and roundtail chubs within the upper Muddy Creek watershed provides a rare opportunity for conservation of these fishes within Wyoming. Distributions of these three target fishes varied by species. Both bluehead suckers and flannelmouth suckers were most abundant within an area bounded by potential barriers to the upstream movement of fishes. This area was typified by perennial streamflow and frequent pool-riffle sequences. Roundtail chubs were widely distributed early in the summer, but absent from the lower portion of the study area after August 1$^{st}$. The role of movement barriers in fragmenting populations of the three target species remains

unknown. The presence of several exotic fishes as well as hybrids among native and exotic Catostomids suggests that the adverse effects of hybridization and competition may be of importance.

# Chapter III

## Effects of Habitat Availability and Fragmentation on Resource Selection by Bluehead Suckers, Flannelmouth Suckers, and Roundtail Chubs in the Upper Muddy Creek Watershed

*Abstract* – My objective was to describe the habitat associations of bluehead suckers, flannelmouth suckers, and roundtail chubs in a small tributary system, the upper Muddy Creek watershed of southern Carbon County, Wyoming. Fish and habitat were sampled at 57 randomly selected 200-m reaches during the summer and fall of 2003 and 2004. A series of thresholds of occurrence were identified at multiple spatial scales in order to define habitats available to each species. Within available habitats, the information-theoretic approach was used to test *a priori* hypotheses and identify the most informative models accounting for variation in an index of abundance of each of the three target species. The most consistent habitat associations among species were positive relationships with both rock substrates and pools. The presence of isolated pools within a reach appeared to play a role in determining distributional patterns of the three target species. Roundtail chubs showed positive associations with reaches having isolated pools, whereas bluehead suckers and flannelmouth suckers showed negative associations and selected reaches having perennial streamflows.

## Introduction

Bluehead suckers, flannelmouth suckers, and roundtail chubs are three of the least-studied native fishes of the upper Colorado River basin (UCRB). Previous studies have described the habitat associations of these fishes in large rivers with few studies conducted on small tributary streams (Bezzerides and Bestgen 2002).

Adult bluehead suckers are commonly associated with pools and runs having moderate to fast currents with rock substrates (Gorman et al. 1994; Beyers et al. 2001). Bluehead sucker juveniles prefer low-velocity habitats along shorelines and in backwaters (Hoffnagle et al. 1994; Robinson et al. 1998).

Flannelmouth suckers have been associated with deep pool and run habitats (Valdez et al. 1982b; Gorman et al. 1994). Flannelmouth suckers have been observed over substrates from clay to cobble, though they are most often found over rock substrates (Sigler and Miller 1963; Holden and Stalnaker 1975a). Young flannelmouth suckers have most often been observed in low-velocity habitats such as backwaters, eddies, and side channels (Childs et al. 1998; Gido and Probst 1999).

Adult roundtail chubs have been found to prefer pool-riffle habitats where they occur in low-velocity, deep pools within close proximity to cover in the form of large rocks, undercut banks, and woody debris (Bestgen 1985; Bestgen and Probst 1989; Rinne 1992; Barrett and Maughan 1995; Brouder et al. 2000). Within the Gila River basin, juvenile and adult roundtail chubs occupied habitats with fine sand to boulder substrates, but most often were found over sand-gravel substrates (Bestgen 1985; Rinne 1992). In Wet Beaver Creek, Arizona, adult roundtail chubs selected boulder and bedrock substrates (Barrett and Maughan 1995). Juvenile

roundtail chubs have generally been found to occupy shallow, low-velocity habitats (Neve 1976; Bestgen 1985; Barrett and Maughan 1995).

I focused on the habitat associations of these three native species in a small tributary system. This focus differs from previous studies that involved large streams and rivers. My objective was to describe the habitat associations of bluehead suckers, flannelmouth suckers, and roundtail chubs within the upper Muddy Creek watershed. I used a hierarchical approach that identified associations at several spatial scales. A combination of thresholds of occurrence and physical habitat models were used to describe relations to available habitats.

## Methods

### *Design*

The study area was segmented based on areas of continuous habitat. Potential movement barriers and a transition to coldwater habitats were used to define the boundaries of three stream segments (Table 3.1, Figure 3.1). Segment 1 began at the downstream boundary of the study area and continued upstream on Muddy Creek to the irrigation diversion located 43 km upstream. Segment 2 consisted of those portions of Muddy Creek from the irrigation diversion upstream to the fish barrier immediately upstream of the confluence of McKinney Creek and McKinney Creek upstream to where an exclusively coldwater fish community occurred. Segment 3 consisted of Muddy Creek above the fish barrier to below a culvert known to limit the movement of fishes upstream as well as Littlefield Creek upstream to a fish barrier.

Each segment was divided into 200-m reaches and from 7 to 30 reaches were randomly selected. The number of reaches selected in each segment was proportional to the length of the segment relative to the total length of streams in the study area. Patton et al. (2000) found that

sampling 200-m of stream generally captured all of the fish species present among warmwater streams in the Missouri River drainage of Wyoming. I assumed that sampling a similar length of stream would enable all species to be captured in the Muddy Creek watershed.

Within each segment, reaches were randomly assigned to one of two time strata to allow temporal differences in fish community and physical habitat features to be assessed. An equal proportion of the reaches were assigned to each time strata within each segment. These strata consisted of early (June 1 – July 31) and late (August 1 – October 8) sampling periods. All sampling in 2004 took place before August. Within both time strata, the chronological order in which reaches were sampled was randomly generated.

*Field methods*

Study reaches were shifted such that the upstream and downstream boundaries terminated at a transition between two channel units, when possible. In cases where a channel unit extended well beyond the upstream or downstream terminus of the reach, the reach was made 175-225 m in length in order to sample a maximum of 50 m of the unit. Each reach was divided into channel units consisting of pool, glide, run, or riffle channel-unit types classified following Flosi and Reynolds (1994).

Prior to sampling fish and habitat at each reach, a water quality sample was taken using a Hydrolab Datasonde 4 or YSI Model 55 multiprobe. Water quality parameters that were measured included specific conductance (µS), dissolved oxygen (mg/L), water temperature ($^{\circ}$C), and turbidity (NTU's). Multiprobes were calibrated prior to initial sampling and calibrated monthly thereafter.

Measurements were made at each study reach to describe the isolation of remnant pools if no surface flow was observed. Isolation of pool channel units was measured as the distance from

the remnant pool to the next closest flowing water both upstream and downstream. The proportion of the length of each reach with no observable surface flow (i.e., dry) was computed for each study reach.

Within pools and runs, physical attributes including water surface area, water depth, substrate, and cover were measured. Area and depth were measured using a surveying rod and measuring tape. The total area of the channel unit was calculated as length x average width. Depth measures included area of the channel unit that was greater than 0.5 m in depth, maximum depth within the unit, and residual depth of pools. Residual depth was the depth at which pools would become stagnant if surface flow declined to zero and was calculated as maximum pool depth minus the depth of the nearest downstream hydrologic control. The proportions of each of six substrate classes (silt, sand, gravel, cobble, boulder, and bedrock) were visually estimated in each unit using the modified Wentworth classification (Cummins 1962, Table 4). I measured areas of three types of cover: overhanging vegetation, submergent aquatic vegetation, and brush. Overhanging vegetation was measured as the area of vegetation that overhung the water surface and provided shade during a portion of the day. Submergent aquatic vegetation was measured as the area of the channel bed that had at least 50% coverage of vegetation including algal mats, mosses, or rooted aquatic plants. Submergent aquatic vegetation did not include thin films of algae on the channel substrates. The number of pieces of woody debris was recorded by diameter size class (1-5, 6-10, 11-50, >51 cm).

Within riffles, several physical attributes were measured. Area and depth were measured using a surveying rod or measuring tape. The area of riffle was measured as the length times the average width. The proportions of each of six substrate classes (silt, sand, gravel, cobble, boulder, and bedrock) were visually estimated in each riffle using the modified Wentworth

classification (Cummins 1962, Table 4). The embeddedness of gravel and cobble substrate classes was estimated as the degree that the larger particles (boulder, cobble, or gravel) were surrounded or covered by fine sediment. Embeddedness ratings of 1-5 were used based on the percentage of the surface of larger substrates that was covered by fine sediment (1 = >75%, 2 = 51-75%, 3 = 26-50%, 4 = 5-25%, 5 = <5%; (Platts et al. 1983).

Each pool or run within a selected reach was sampled using electrofishing and seining in order to account for the differential species and size selectivity of the two gears. The pool or run was isolated by placing block nets at the upstream and downstream boundaries of the channel unit. A single electrofishing pass was made and all fish collected were identified and the total length (TL) was measured to the nearest millimeter. These fish were temporarily retained. The channel unit was then sampled by seining, where possible, with a 4.8-mm mesh bag seine, 4.6 m long by 1.2 m in height, with a 1.2 $m^3$ square bag sewn into the center of the net. All fish collected by seining were identified and TL was measured to the nearest millimeter. All fish collected during electrofishing and seining efforts were then returned to the channel unit. Though rare, it was occasionally not possible to identify young-of-year catostomids to the species level. Individuals for which a positive identification was not possible were recorded as "unknown" and excluded from subsequent analyses.

The total number of fish of each target species and length class was summed and proportionally adjusted to a 100-m reach to yield an index of relative abundance (fish/100 m) for each sampled reach.

*Analysis*

Physical features of habitats selected by fishes often vary among species as well as life history stages of species (Schlosser 1991). For this reason, two length classes were identified

to allow for the identification of differential habitat associations among life history stages. Given the paucity of information pertaining to length at maturity for the three species within small tributary streams, these lengths were based on field observations. These observations included tuberculation, expression of gametes, and expression of spawning colorations. The length at maturity estimates were 150 mm TL for bluehead suckers, 200 mm TL for flannelmouth suckers, and 150 mm TL for roundtail chubs. Fish equal to or greater than the length at maturity were termed adults, and fish less than the length at maturity were termed subadults.

One of the difficulties commonly encountered when designing resource selection studies and subsequently establishing the association of animals with features of their habitats lies in the ability to measure the availability of habitats (Manly et al. 2002). Though certain habitats may exist within a landscape, there are frequently abiotic constraints that limit the ability of individuals in the population to access these habitats.

One model that accounts for the differential availability of habitats to individual species based on abiotic constraints is that of hierarchical faunal filters (Matthews 1998). Within this model, a series of faunal filters at multiple spatial scales influence the occurrence of species. These faunal filters can be thought of as hierarchical in that biotic, or particularly abiotic, constraints act to structure the present and abundance of species within a community at several spatial scales. For example, basin-scale faunal filters such as water temperature have been shown to limit the availability or suitability of habitats to warmwater fishes above threshold elevations (Rahel and Hubert 1991; Carter and Hubert 1995; Quist et al. 2004b). Though this model has taken many forms in describing both terrestrial and aquatic ecological patterns of

species occurrence (Poff 1997; Matthews 1998), the fundamental idea that multi-scale processes affect the presence and abundance of fishes within a community has broad applicability.

I identified faunal filters that influence the availability of habitats for the three target species and each length class based on thresholds of occurrence at basin and reach scales. The single basin-scale threshold of occurrence was elevation which likely acted as a surrogate for many basin-scale processes that limited the availability of warmwater habitats to the target species. At the reach-scale, a measure of remnant pool isolation was used to eliminate reaches that were unavailable to the target species at the time of sampling due to a lack of water, calculated as the proportion of channel length that was dry within a reach. A second reach-scale filter termed "sampled area" was measured as the area of a reach that was sampled by electrofishing and seining, excluding areas of riffle habitat that were not sampled. This filter was considered a surrogate for the size of the stream through the sampled reach. This set of three faunal filters, or thresholds of occurrence, was used to eliminate from subsequent analyses of resource selection those habitats that were considered unavailable to a particular species or length class of a species based on thresholds of occurrence (Figure 3.2).

Analysis of resource selection by two length groups of each target species at the reach and channel-unit-scales was performed using the information-theoretic approach. In this approach, evidence for a set of *a priori* hypotheses expressed as models was evaluated by ranking each model, estimating the formal likelihood of each model, and measuring precision that incorporates model selection uncertainty (Anderson et al. 2000; Anderson et al. 2001a; Anderson et al. 2001b; Anderson and Burnham 2002). Given the limited literature on habitat use by bluehead suckers, flannelmouth suckers, and roundtail chubs in small streams, the set of models that was considered was not limited to published habitat associations. The *a priori*

hypotheses that are supported by the literature or represent plausible biological hypotheses are presented for each life stage of the three target species. Hypothesized selection for an attribute is denoted by (+) and hypothesized avoidance is denoted by (-):

*Hypothesis 1.* Subadult bluehead sucker relative abundance in a reach is related to the proportion of dry stream channel (-), percentage of rock substrates throughout the reach (+), and the size of the stream through the reach (+). Subadult bluehead sucker relative abundance within channel units is related to the depth of the channel unit (-), the amount of instream cover present (+), and the mesohabitat type of the channel unit (+ glide).

*Hypothesis 2.* Adult bluehead sucker relative abundance in a reach is related to the proportion of dry stream channel (-), the percentage of rock substrates throughout the reach (+), and the size of the stream through the reach (+). Adult bluehead sucker relative abundance within channel units is related to the depth of the channel unit (+), the amount of instream cover present (+), and the mesohabitat type of the channel unit (+ pool).

*Hypothesis 3.* Subadult flannelmouth sucker relative abundance in a reach is related to the proportion of dry stream channel (-), percentage of rock substrates throughout the reach (-), and the size of the stream through the reach (+). Subadult flannelmouth sucker relative abundance within channel units is related to the depth of the channel unit (-), the amount of instream cover present (+), and the mesohabitat type of the channel unit (+ pool, + glide).

*Hypothesis 4.* Adult flannelmouth sucker relative abundance in a reach is related to the proportion of dry stream channel (-), the percentage of rock substrates throughout the reach (+), and the size of the stream through the reach (+). Adult flannelmouth sucker relative abundance within channel units is related to the depth of the channel unit (+), the amount of instream cover present (+), and the mesohabitat type of the channel unit (+ pool).

*Hypothesis 5.* Subadult roundtail chub relative abundance in a reach is related to the proportion of dry stream channel (-), percentage of rock substrates throughout the reach (-), and the size of the stream through the reach (+). Subadult roundtail chub relative abundance within channel units is related to the depth of the channel unit (-), the amount of instream cover present (+), and the mesohabitat type of the channel unit (+ pool, + glide).

*Hypothesis 6.* Adult roundtail chub relative abundance in a reach is related to the proportion of dry stream channel (-), the percentage of rock substrates throughout the reach (-), and the size of the stream through the reach (+). Adult roundtail chub relative abundance within channel units is related to the depth of the channel unit (+), the amount of instream cover present (+), and the mesohabitat type of the channel unit (+ pool).

The set of variables to be included in multiple-regression models was limited by examining correlations among variables. Where substantial ($r > 0.20$) correlations were present among measured variables, variables were chosen based on their biological significance. A set of uncorrelated variables were then used to compute multiple-regression models to predict the relative abundance of a species or length class as indicated in the *a priori* hypotheses. Prior to computing models, all proportional habitat measures were transformed using the arcsine square root transformation (Ramsey and Schafer 2002). These measures included proportions of rock substrate, dry stream channel, and channel unit area with submergent vegetation.

Multiple-regression models were computed to describe the physical features of habitats influencing the index relative abundance (CPUE = fish/100 m) of each species and length class computed as (log(CPUE+1)), given by

$$\hat{y} = \beta_0 + \beta_1 x_1 + \beta_2 x_2 + \ldots \beta_p x_p$$

where $\beta_0$ = the regression constant, $\beta_p$ = regression coefficients, and $x_p$ = independent variables.

Model selection was performed by calculating Akaike's Information Criterion, modified for small sample size ($AIC_c$), given by

$$AIC_c = n * \log_e(RSS/n) + 2(K) + (2K(K+1))/(n-K-1)$$

where n is the sample size, RSS is the residual sum of squares of the fitted model, and $K$ is the number of unknown parameters that must be estimated.

The $AIC_c$ values were then rescaled such that the model with the minimum $AIC_c$ had a value of zero given by

$$\Delta_i = AIC_c - \min AIC_c$$

This rescaling allowed for a strength of evidence comparison and scaled ranking among competing models. The likelihood of a model given the data was calculated by computing a normalized simple transformation of $\Delta_i$ given by

$$w_i = \frac{\exp(-\frac{1}{2}\Delta_i)}{\sum_{r=1}^{R}\exp(-\frac{1}{2}\Delta_i)}$$

where $w_i$ is the Akaike weight and can be interpreted as the approximate probability that model $i$ is, in fact, the Kullback-Liebler best model in the set of models considered.

Multiple-regression models were computed for each species and length class at the reach and channel-unit scales. The reach scale was defined as the set of reaches that were sampled and variables computed from measurements over each reach. The reach scale included a set of models computed using data from all sampled reaches with no stratification among the three stream segments. The channel-unit scale was defined as the set of three types of channel units (e.g., pools, glides, and runs) from which individual habitat features were measured. Two sets of analyses were conducted at the channel-unit scale, one that included all channel units sampled over all reaches and a second for each of the three stream segments independently.

In each of the reach and channel-unit analyses, competing models were identified from among the array of variables in the global model (Anderson et al. 2000). All models within 10% of the Akaike weight of the top-ranked model were considered competing models. Model-averaged estimates of effect size were generated by computing the weighted average among all competing models for each parameter given by

$$\hat{\bar{\theta}} = \sum w_i \hat{\theta}_i$$

where $\hat{\theta}_i$ is the parameter estimate from competing model $i$ and $w_i$ is the Akaike weight of model $i$. All inferences were based on these averaged parameter estimates, rather than a single best model (Burnham and Anderson 1998).

Plots were generated from averaged multiple-regression models to illustrate the effects of individual model parameters on the index of relative abundance of each length class of each of the three target species. These plots were generated for the range of values observed for a particular model parameter within those habitats considered available to each species and length class while holding other model parameters constant at the mean values observed within available habitats.

## Results

Three variables were chosen for inclusion in the global reach-scale models for each species and length class in order to assess the *a priori* hypotheses. Given the prevalence of isolated remnant pool habitats during the summer and fall of 2003 and 2004, and the potential for stream drying to affect the abundance of each of the three species, the first variable selected for inclusion in the reach-scale models was the proportion of the length of the reach that was dry.

Given the reported associations of the target species with habitats typified by rock substrates, the proportion of rock substrates within the reach was included in the global model as a measure of substrate. Sampled area was chosen as the best measure of stream size (Table 3.2). A set of seven models was developed consisting of all combinations of the three variables. These models were computed using multiple-linear regression with the relative abundance (log(CPUE + 1)) of a species in a length class as the response variable.

Two sets of substantial correlations were observed among the habitat variables measured at the channel unit scale. The three measures of channel unit size were correlated ($r \geq 0.41$, $n = 447$). Measures of cover were frequently correlated with measures of channel unit area with the exception of submergent vegetation. Additionally, measures of deep water habitat were correlated with all measures of channel unit size ($r \geq 0.35$, $n = 447$) and overhanging cover ($r \geq 0.32$, $n = 447$; Table 3.2).

Three habitat variables were identified for inclusion in the global channel-unit-scale models for each species and length class in order to assess the *a priori* hypotheses. The maximum depth (m) of the channel unit was selected as the most intuitive measure of channel unit depth that was not correlated with all measures of cover. Subsequently, proportion of channel unit area ($m^2$) containing 50% or greater coverage of submergent vegetation was selected as the single measure of instream cover that was not correlated with this measure of channel unit depth. Channel unit mesohabitat type (pool, glide, or run) was selected as a measure of channel morphology. A set of seven models were developed consisting of all combinations of the three variables. These models were computed using multiple regression with the relative abundance (log(CPUE + 1)) of a species in a length class as the response

variable where CPUE = fish/100 m. Models were computed for each of the three segments independently for each species in a length class as well as for the total study area.

Fifty-seven reaches were sampled across the study area. The proportion of the reach length that was dry at the time of sampling, ranged from zero (channel units connected by life streamflow) to one (entire reach was dry) with a mean value of 0.18. The percentage of rock substrates within the reaches ranged from 0 to 48% with a mean of 10%. The sampled areas varied widely among reaches with a range of 0 to 823 $m^2$ and a mean area of 380 $m^2$. Adult bluehead suckers were sampled in 51% of the reaches while subadult bluehead suckers were sampled in 76% of the reaches. Adult flannelmouth suckers were sampled in 36% of the reaches and subadult flannelmouth suckers were sampled in 49% of the reaches. Adult roundtail chubs were found in 66% of the reaches while subadult roundtail chubs were found in 75% of the reaches.

A total of 447 channel units were sampled among the 57 reaches. These channel units were classified as pools (n = 283), glides (n = 150), or runs (n = 14). The proportion of submergent vegetation within these channel units ranged from zero to 0.47 with a mean of 0.05. The average maximum depth of the channel units was 0.51 m with a range of 0.05 to 2.14 m.

*Bluehead suckers*

*Subadult*. No subadult bluehead suckers were detected above 2,228 m in elevation. This elevation was established as the basin-scale faunal filter above which habitat conditions become unsuitable based on basin-scale processes. Three reaches occurring above this elevation were eliminated from the set of available reaches for the purpose of resource selection analysis (Table 3.3). Bluehead sucker subadults were not detected in six reaches with greater than 82% of the reach that was dry and these reaches were eliminated from the set of available reaches. Subadult

bluehead suckers were not detected in four reaches containing less than 19 $m^2$ of sampled area and these reaches were similarly eliminated from the set of available reaches.

The best reach-scale model accounting for variation in the index of abundance of subadult bluehead suckers included sampled area and rock substrates ($w_i$ = 0.72). One model was considered to be a competitor and was represented by the global model ($w_i$ = 0.20; Appendix B1). The average model (Table 3.4) indicated that the relative abundance of subadult bluehead suckers was weakly associated with the sampled area and the proportion of reach length that was dry at the time of sampling (Figure 3.3), and strongly associated with rock substrates (Figure 3.4).

One additional faunal filter was identified at the channel-unit-scale based on minimum channel unit depth. Bluehead sucker subadults were not detected in channel units less than 0.12 m deep (Table 3.3). Nine channel units less than 0.12 m deep were removed from the set of channel units available for the purposes of resource selection analyses.

The best channel-unit-scale model accounting for variation in the index of abundance of subadult bluehead suckers across all stream segments was the single-variable model that accounted for channel unit type. Three competing models were identified, all of which contained the channel unit type ($\Sigma w_i$ = 1.00). Maximum depth ($\Sigma w_i$ = 0.26) and submergent vegetation ($\Sigma w_i$ = 0.26) were each present within two competing models (Appendix B2). The averaged model (Table 3.5) indicated positive association with pool habitats in relation to runs, avoidance of glides in relation to runs, slight avoidance of maximum depth (Figure 3.5), and slightly positive association with submergent vegetation (Figure 3.6).

Models accounting for variation in the abundance of bluehead sucker subadults at the channel unit scale differed among the three segments in both the predictor variables in the best

models and the effect size and direction of predictions within the averaged model. The best models within Segments 1 ($w_i$ = 0.40) and 2 ($w_i$ = 0.42) were the single-variable models with channel unit type. Within Segment 3, the best model was the single-variable model of maximum depth ($w_i$ = 0.33; Appendices B8-B10). Parameter estimates from the averaged model for Segment 1 showed positive associations with both pool and glide habitats in relation to run habitats, positive association with maximum depth, and positive associations submergent vegetation (Table 3.6). Within Segment 2, the averaged model indicated positive association with pool habitats and avoidance of glide habitats in relation to run habitats, positive association with maximum depth, and avoidance of increasing proportions of reach length that was dry at the time of sampling (Table 3.7). The averaged model for Segment 3 showed positive associations with both pool and glide habitats in relation to run habitats, positive association with maximum depth, and slight avoidance of submergent vegetation (Table 3.8). The relative importance of model parameters varied little among stream segments (Table 3.9), consistently indicating the importance of pool and glide habitats.

*Adult.* No bluehead sucker adults were detected above 2,228 m in elevation. This value was established as the basin-scale threshold above which habitat conditions became unsuitable. Three reaches occurring above this elevation were eliminated from the set of available reaches for the purpose of resource selection analysis. Bluehead sucker adults were not detected in nine reaches where greater than 54% of the reach length was dry or in eleven reaches containing less than 127 $m^2$ of sampled area (Table 3.3). These reaches were similarly eliminated from the set of available reaches.

The best reach-scale model accounting for variation in the abundance of adult bluehead suckers was the single-variable model with rock substrate ($w_i$ = 0.91). There were no competing

models (Appendix B1). The best model showed a single positive association with rock substrates (Figure 3.4, Table 3.4).

Bluehead sucker adults were not detected in channel units less than 0.25 m deep (Table 3.3). Sixty channel units shallower than this threshold were removed from the set of channel units available for the purposes of resource selection analyses based on this channel-unit-scale faunal filter.

The best model accounting for variation in the abundance of bluehead sucker adults at the channel-unit-scale across all three segments was the global model ($w_i$ = 0.62). Two competing models were identified that included the channel unit type ($\Sigma w_i$ = 0.93), maximum depth ($\Sigma w_i$ = 0.81), and submergent vegetation ($\Sigma w_i$ = 0.74; Appendix B2). The averaged model (Table 3.5) indicated positive association with pool and glide habitats in relation to run habitats, positive association with maximum depth (Figure 3.5), and positive association with submergent vegetation (Figure 3.6).

Models accounting for variation in the relative abundance of adult bluehead suckers differed among segments. The best model within Segment 1 was the global model ($w_i$ = 0.43). Within Segment 2, the best model included the channel unit type and maximum depth ($w_i$ = 0.33). The best model within Segment 3 was the single variable model with channel unit type ($w_i$ = 0.26; Appendices B3-B5). Parameter estimates from the averaged model for Segment 1 showed a positive association with pool habitats and slight avoidance of glide habitats in relation to run habitats, as well as positive associations with both maximum depth and submergent vegetation (Table 3.6). Within Segments 2 and 3, the averaged models indicated positive associations with both pool and glide habitats in relation to run habitats, as well as positive associations with maximum depth and submergent vegetation (Tables 3.12 and 3.13). The

relative importance of pool and glide habitats as well as the maximum depth of the channel unit within Segments 1 and 2 was contrasted by the importance of all habitat variables within Segment 3 (Table 3.9).

*Flannelmouth suckers*

*Subadults.* No subadult flannelmouth suckers were detected above 2,195 m in elevation. This value was established as the basin-scale threshold above which habitat conditions become unsuitable based on basin-scale processes. Ten reaches occurring above this elevation were eliminated from the set of available reaches for the purpose of resource selection analysis. Flannelmouth sucker subadults were not found within six reaches where greater than 81% of the reach length was dry and seven reaches with less than 79 $m^2$ of sampled area (Table 3.3). These reaches were eliminated from the set of available reaches.

The best model accounting for variation in the abundance of subadult flannelmouth suckers at the reach-scale was the single-variable model with rock substrates ($w_i = 0.35$). Five additional competing models contained various combinations of rock substrates ($\Sigma w_i = 0.86$), the proportion of reach length that was dry at the time of sampling ($\Sigma w_i = 0.38$), and sampled area ($\Sigma w_i = 0.33$; Appendix B1) were found. The averaged model (Table 3.4) showed a weak association with the sampled area, weak association with the proportion of reach length that was dry at the time of sampling (Figure 3.3), and a slightly positive association with rock substrates (Figure 3.4).

Flannelmouth sucker subadults were not detected within channel units less than 0.12 m maximum depth (Table 3.3). This channel-unit-scale faunal filter was used to remove nine channel units from the set of channel units available for resource selection analyses.

The best channel-unit-scale model accounting for variation in abundance of subadult flannelmouth suckers contained the channel unit type and maximum depth ($w_i$ = 0.64). One additional competing model was the global model ($w_i$ = 0.43; Appendix B2). The averaged model (Table 3.5) showed a positive association with both pool and glide habitats in relation to run habitats, a slightly positive association with maximum depth (Figure 3.5), and a slightly positive association with submergent vegetation (Figure 3.6).

Channel-unit-scale models accounting for variation in relative abundance of subadult flannelmouth suckers differed among segments one and two. Within Segment 1, the best model was the single variable model with maximum channel unit depth ($w_i$ = 0.41). The best model for Segment 2 included the channel unit type and submergent vegetation ($w_i$ = 0.56; Appendices B3-B5). Parameter estimates from averaged models for both segments indicated positive associations with both pool and glide habitats in relation to run habitats, as well as positive associations with maximum depth and submergent vegetation (Table 3.6). The relative importance of model parameters varied among Segments 1 and 2 (Table 3.9). Within Segment 1, the maximum depth of the channel unit was the most important variable ($\Sigma w_i$ = 1.00), occurring in all competing models. Within Segment 2, pool and glide habitats ($\Sigma w_i$ = 0.99) were considered more important.

*Adults*. No flannelmouth sucker adults were detected above 2,228 m in elevation. This value was established as the basin-scale threshold above which habitat conditions become unsuitable based on basin-scale processes. Three reaches occurring above this elevation were eliminated from the set of available reaches for the purpose of resource selection analysis. Flannelmouth sucker adults were not found within five reaches where greater than 83% of the

reach length was dry or six reaches containing less than 53 $m^2$ of sampled area (Table 3.3). These reaches were eliminated from the set of available reaches.

The best model accounting for variation in the abundance of adult flannelmouth suckers at the reach-scale was also the single variable model with sampled area ($w_i$ = 0.35). Five additional competing models contained various combinations of rock substrates ($\Sigma w_i$ = 0.74), the proportion of reach length that was dry at the time of sampling ($\Sigma w_i$ = 0.39), and sampled area ($\Sigma w_i$ = 0.36; Appendix B1) were observed. The averaged model (Table 3.4) showed a weak positive association with sampled area, a weak association with the proportion of reach length that was dry at the time of sampling (Figure 3.3), and a slightly positive association with rock substrate (Figure 3.4).

Flannelmouth sucker adults were not detected within channel units less than 0.34 m maximum depth (Table 3.3). This channel-unit-scale faunal filter was used to remove 126 channel units shallower than this threshold from the set of channel units available for resource selection analyses.

The best channel-unit-scale model accounting for variation in the abundance of adult flannelmouth suckers across all three segments was also the single-variable model with maximum depth ($w_i$ = 0.43). Three additional competing models were identified consisting of various combinations of maximum depth ($\Sigma w_i$ = 1.00), channel unit type ($\Sigma w_i$ = 0.39), and submergent vegetation ($\Sigma w_i$ = 0.26; Appendix B2). The averaged model (Table 3.5) showed avoidance of both pool and glide habitats in relation to run habitats, a positive association with maximum depth (Figure 3.5), and a slightly positive association with submergent vegetation (Figure 3.6).

Models accounting for variation in the abundance of adult flannelmouth suckers differed among segments. Within Segment 1, the best model included maximum depth and channel unit type ($w_i$ = 0.65). The best model for Segment 2 was the single variable model with maximum depth ($w_i$ = 0.33; Appendices B3-B5). Parameter estimates from the averaged model for Segment 1 indicated avoidance of both pool and glide habitats in relation to run habitats, as well as positive associations with both maximum depth and submergent vegetation (Table 3.6). Within Segment 2, the averaged model showed positive associations to both pool and glide habitats in relation to run habitats, a positive association with maximum depth, and avoidance of submergent vegetation (Table 3.7). The relative importance of model parameters varied by stream segment (Table 3.9). The maximum depth of the channel unit was considered an important model parameter in Segments 1 ($\Sigma w_i$ = 0.98) and 2 ($\Sigma w_i$ = 0.88) while pool and glide habitats were of more importance in Segment 1 ($\Sigma w_i$ = 0.98) than in Segment 2 ($\Sigma w_i$ = 0.49).

*Roundtail chubs*

*Subadults*. No roundtail chub subadults were detected above 2,228 m in elevation. This value was established as the basin-scale threshold above which habitat conditions become unsuitable based on basin-scale processes. Three reaches occurring above this elevation were eliminated from the set of available reaches for the purpose of resource selection analysis. Roundtail chub subadults were not found in three reaches where greater than 94% of the reach length was dry (n = 3) or three reaches with less than 17 $m^2$ of sampled area Table 3.3). These reaches were eliminated from the set of available reaches.

The best model accounting for variation in the abundance of subadult roundtail chubs at the reach scale included the sampled area and proportion of reach length that was dry at the time of sampling ($w_i$ = 0.68). The global model was considered a competing model ($w_i$ = 0.32;

Appendix B1). The averaged model (Table 3.4) showed a weak association with sampled area and a strong positive association with both the proportion of reach length that was dry at the time of sampling and rock substrates (Figures 3.3 and 3.4).

Roundtail chub subadults were not detected in channel units less than 0.12 m maximum depth. This channel-unit-scale faunal filter was used to remove nine channel units shallower than this threshold from the set of channel units available for resource selection analyses (Table 3.3).

The best channel unit model accounting for variation in the abundance of roundtail chubs subadults across all three segments included maximum depth and channel unit type ($w_i$ = 0.71). One competing model was identified, the global model ($w_i$ = 0.15; Appendix B2). The averaged model (Table 3.5) showed a positive association with both pool and glide habitats in relation to run habitats, a strong positive association with maximum depth (Figure 3.5), and avoidance of submergent vegetation (Figure 3.6).

Models accounting for variation in relative abundance of roundtail chub subadults at the channel-unit-scale differed among segments. The best model for Segment 1 was the global model ($w_i$ = 0.64). The best models for Segments 2 ($w_i$ = 0.30) and 3 ($w_i$ = 0.21) included the channel unit type and maximum depth (Appendices B8-B10). The averaged model for Segments 1 and 2 indicated positive associations with both pool and glide habitats in relation to run habitats as well as positive associations with both maximum depth and submergent vegetation (Tables 3.11 and 3.12). Within Segment 3, the averaged model indicated avoidance of pool and glide habitats in relation to run habitats, positive association with maximum depth, and avoidance of submergent vegetation (Table 3.8). The relative importance of model parameters showed similar patterns among the three stream segments (Table 3.9). Pool and glide habitats

were consistently considered important model parameters. The maximum depth of the channel unit was considered most important within Segment 1 ($\Sigma w_i = 0.98$).

*Adults*. No roundtail chub adults were detected above 2,228 m in elevation. This value was established as the basin-scale threshold above which habitat conditions become unsuitable based on basin-scale processes. Three reaches occurring above this elevation were eliminated from the set of available reaches for the purpose of resource selection analysis. Roundtail chubs adults were not found in five reaches where greater than 82% of the reach length was dry or in four reaches with less than 19 $m^2$ of sampled area (Table 3.3). These reaches were eliminated from the set of available reaches.

The abundance of adult roundtail chubs at the reach scale was accounted for best by the sampled area and proportion of reach length that was dry at the time of sampling ($w_i = 0.67$). The global model was considered a competing model ($w_i = 0.28$; Appendix B1). The averaged linear-regression model (Table 3.4) showed a weak association with sampled area and positive associations with the proportion of reach length that was dry at the time of sampling and rock substrates (Figures 3.3 and 3.4).

Roundtail chub adults were not detected within channel units less than 0.23 m maximum depth. This channel-unit-scale faunal filter was used to remove 50 channel units shallower than this threshold from the set of channel units available for resource selection analyses.

The best channel-unit-scale model accounting for variation in the abundance of roundtail chub adults across all three segments was represented by the global model ($w_i = 0.64$). One competing model contained channel unit type and maximum depth ($w_i = 0.32$). The averaged linear-regression model (Table 3.5) indicated a positive association with both pool and glide

habitats in relation to run habitats, a strong positive association with maximum depth (Figure 3.5), and a positive association with submergent vegetation (Figure 3.6).

Models accounting for variation in relative abundance of roundtail chub adults at the channel-unit-scale also varied among segments. The best models for Segments 1 ($w_i$ = 0.32) and 2 ($w_i$ = 0.60) were represented by the global model. The best model for Segment 3 was represented by the single variable maximum depth ($w_i$ = 0.36; Appendices B3-B5). Parameter estimates from the averaged models for Segments 1 and 2 indicated positive associations with pool and glide habitats in relation to run habitats, a positive association with maximum depth, and a positive association with submergent vegetation (Tables 3.11 and 3.12). Within Segment 3, the averaged model indicated a positive association with pool habitats and avoidance of glide habitats in relation to run habitats, a positive association with maximum depth, and a positive association with submergent vegetation (Table 3.8). The relative importance of model parameters varied among stream segments (Table 3.9). The maximum depth of the channel unit was of most importance in Segment 1 ($\Sigma w_i$ = 1.00), while submergent vegetation was of most importance in Segment 2 ($\Sigma w_i$ = 0.87) and maximum channel unit depth was of most importance in Segment 3 ($\Sigma w_i$ = 0.78).

## Discussion

There was evidence of meaningful biological associations of native fishes with measured habitat features in the upper Muddy Creek watershed. Within this study, the variables chosen for inclusion in reach- and channel-unit-scale models were surrogates for larger classes of physical habitat variables. For example, maximum channel unit depth was correlated with nearly all measures of stream size, with the exception of the variable termed "sampled area." Similarly,

correlations among the several measures of cover necessitated the selection of a single variable, submergent vegetation, for inclusion within channel-unit-scale models. However, the variables chosen for inclusion in global models at the reach- and channel-unit-scale were thought to be of high biological significance and, therefore, interpretation of the associations of the target species with these physical habitat attributes should increase our understanding of the life history and habitat requirements of the three species.

Another consideration when interpreting the results is my designation of subadult and adult length classes for each species. As the fish present in the upper Muddy Creek watershed are generally smaller than those in larger streams, it was not possible to establish length classes by using age at maturity statistics available for larger streams. Refinement of the age at maturity for populations of each of the target fishes within the study area is likely to increase the resolution of models accounting for associations of the two distinct life stages with physical features of their habitats.

Among the three segments of the upper Muddy Creek watershed, the variation in channel-unit-scale models predicting the abundance of each species and length classes underscores the importance of segment-scale influences on resource selection. Variation in the relative importance of model parameters among stream segments suggests that habitat features that may be limited in their availability within one segment may be abundant within another. For example, the maximum depth of the channel unit was consistently identified as an important parameter within Segment 1. Segment 1 experienced extensive stream drying and consisted of isolated pool habitats during much of the sampling period. The depth of these remnant pools, therefore, was likely a key feature allowing the persistence of each of the target species within Segment 1. Within Segment 2, much of which consisted of perennial streamflow conditions

during sampling, pool and glide habitats were generally considered more important than maximum channel unit depth, suggesting that the depth of channel units may not be limiting within this segment. Fausch et al. (2002) described a common limitation of our understanding of stream fish ecology where knowledge is lacking at the segment scale. Obtaining a continuous view of stream processes and the way in which fishes relate to various components of physical habitats at multiple spatial scales remains one of the challenges that will need to be overcome to take a holistic approach to fish community and habitat management.

*Bluehead suckers*

Bluehead sucker subadults and adults showed a strong association with rock substrates in the upper Muddy Creek watershed, similar to the association with hard substrates reported by Gorman (1994). The variable describing remnant pool isolation (i.e., the proportion of reach length that was dry) was not included in any of the competing models accounting for the abundance of bluehead sucker adults and, therefore, was not considered to be of importance, however, the threshold value of 53% of the reach length as dry channel suggests that adult bluehead suckers were intolerant of moderate to high levels of remnant pool isolation.

As the maximum depth of channel units increased, there appeared to be little influence on the abundance of either length class of bluehead suckers in the upper Muddy Creek watershed. Beyers et al. (2001) documented bluehead sucker use of deep pool habitats in the Colorado River using radio telemetry, suggesting some selection for such habitats. The proportion of the channel unit area occupied by submergent vegetation appeared to influence the abundance of adult bluehead suckers, but had little to no influence on subadult bluehead suckers in the upper Muddy Creek watershed.

Positive associations with rock substrates and pool habitats suggest that bluehead suckers of both length classes were selecting rock substrates and utilizing deep pool habitats where they were available. These areas were generally associated with pool-riffle complexes within the upper Muddy Creek watershed and were most prevalent within a relatively short section typified by perennial streamflow occurring from roughly 5 km downstream from the irrigation diversion upstream to the confluence with McKinney Creek (see Figure 1.1).

*Flannelmouth suckers*

The basin-scale threshold of elevation in the upper Muddy Creek watershed was lower for flannelmouth suckers (2,195 m) than for either bluehead suckers or roundtail chubs (2,228 m). This threshold was similar to the elevational limits of the warmwater fish assemblage in the Bitter Creek watershed of southwestern Wyoming (2,192 m) reported by Carter and Hubert (1995).

Flannelmouth suckers were not abundant within the upper Muddy Creek study area, probably decreasing the resolution with which habitat associations could be identified, but habitat associations generally supported the *a priori* hypotheses. Adult and subadult flannelmouth suckers showed positive associations with rock substrates in the upper Muddy Creek watershed, and the association was more pronounced for the adult length class. Large flannelmouth suckers have been previously described to occur over rock substrates (Holden and Stalnaker 1975a). Flannelmouth sucker subadults showed associations with both pools and glides. Subadults have been shown to utilize lower velocity habitats with fine substrates in the San Juan River, New Mexico (Gido and Probst 1999). However, flannelmouth sucker adults showed avoidance of both pools and glides in favor of faster velocity run habitats in the study area. Flannelmouth suckers also showed positive associations with increasing maximum channel

unit depths for both length classes in concordance with the observations of Beyers et al. (2001) and avoidance of increasing proportions of reach length that were dry at the time of sampling.

Flannelmouth suckers showed associations with submergent vegetation in the upper Muddy Creek watershed, though the association was much stronger for the subadult length class. Considering the general lack of backwater habitats within the study area, which are generally associated with subadult habitat within larger streams of the upper Colorado River basin (Gido and Probst 1999), submergent vegetation may be functioning as a form of cover for subadult flannelmouth suckers.

*Roundtail chubs*

Based on basin-scale thresholds of occurrence, roundtail chub subadults showed the greatest tolerance among the three target species for high proportions of the reach length being dry in the upper Muddy Creek watershed, thus showing an association with isolated remnant pools. Most prevalent among the habitat associations of roundtail chubs was the strong association of both subadults and adults to reaches with isolated remnant pools and rock substrates. Within areas containing few remnant pools, the largest and most permanent remnant pools tended to be found in close proximity to hydrologic controls comprised of rock substrates.

In concordance with reported habitat associations, adults in the upper Muddy Creek watershed were positively associated with slow-velocity habitats, such as deep pools and glides, and cover in the form of submergent vegetation (Bestgen and Probst 1989). Roundtail chub subadults were similarly associated with pool and glide habitats (Barrett and Maughan 1995), but were negatively associated with submergent vegetation.

*Conclusions*

Thresholds of occurrence showed differences in the availability of habitats to the three species and two length classes. Though adult bluehead sucker appeared intolerant of isolated remnant pool habitats, both length classes of flannelmouth suckers and roundtail chubs, as well as subadult bluehead suckers, occurred in reaches with high proportions of reach lengths that were dry at the time of sampling (i.e., isolated pool habitats). Within available habitats, both bluehead suckers and flannelmouth suckers showed either weak or negative associations with reaches having isolated pools. Both length classes of all three of the target species showed positive associations with rock substrate. With the exception of adult flannelmouth suckers, both length classes of the three target species also showed positive associations with pool habitats in relation to run habitats.

# Chapter IV

## Considerations for the Management of Bluehead Suckers, Flannelmouth Suckers, and Roundtail Chubs in the Upper Muddy Creek Watershed

### Introduction

The occurrence of bluehead suckers, flannelmouth suckers, and roundtail chubs, as well as other native fishes (i.e., speckled dace, Colorado River cutthroat trout, and mountain sucker), within the upper Muddy Creek watershed suggests that it may be one of the highest conservation priorities for native fishes in Wyoming. Partnerships and momentum generated as a result of ongoing restoration activities in coldwater habitats within the watershed further increase the value of the upper Muddy Creek watershed as a conservation unit for native fishes. Specifically, the work of the Muddy Creek Coordinated Resource Management (CRM) group has resulted in greatly improved habitat conditions for two native coldwater fishes, Colorado River cutthroat trout and mountain sucker. Incorporation of factors affecting the viability of populations of the native warmwater fishes into cooperative management efforts in the watershed may enable fish community conservation. My project has identified several factors to be considered in conservation planning efforts, including the potentially adverse effects of introduced fishes and movement barriers on the movements of native fishes, as well as key physical habitat associations.

Two forms of management could be considered for the maintenance or enhancement of populations of the target species – active and passive approaches. Passive management approaches are related primarily to planning activities and regulation of anthropogenic uses. These activities typically entail regulation of human uses of fishes and their habitats or

maintenance of watershed processes that form desired habitat conditions. Active management approaches are more tangible and entail the direct manipulation of either fish assemblages or fish habitats in order to meet management objectives. Examples of active management include the removal of an undesirable species and applying instream habitat prescriptions with the intention of increasing the quality of habitats for target species.

## Active Management Opportunities

*Fish community management*

Given the likely adverse effects of exotic fishes on the native fish community of the upper Muddy Creek watershed, removal of exotic fishes should be given consideration in conservation planning. The feasibility and benefits of such removal efforts should be considered in light of biological, economic, and social costs (Beamesderfer 2000; Quist and Hubert 2004). Many of these costs and benefits vary among removal techniques. For example, mechanical removal of creek chubs and white suckers from the study area would likely constitute a fixed economic cost.

If introgression is a threat to native suckers within the study area, white sucker removal alone may not be sufficient to restore the native genome of bluehead suckers and flannelmouth suckers (Rhymer and Simberloff 1996). Given this scenario, chemical removal of the entire fish community and subsequent reintroduction of genetically pure native fishes from neighboring populations may be considered, but would generate biological costs. For example, chemical treatments would remove exotic fishes and those native fishes thought to have experienced introgression, but would also remove species such as roundtail chubs and speckled dace for which there is no evidence of introgressive hybridization with introduced species. Neither

roundtail chubs nor speckled dace are known to hybridize with any of the exotic species in the upper Muddy Creek watershed. In particular, roundtail chubs are thought to be a high conservation priority within Wyoming given their extremely limited distribution. The upper Muddy Creek watershed contains the highest abundances of this species currently known in Wyoming.

Experiences from the Colorado River basin have suggested that the foremost problem limiting the success of nonnative removal efforts is the lack of a single holistic approach to ecosystem recovery. Additional limitations have been attributed to sampling efficiency and other logistical constraints (Tyus and Saunders 2000). The streams of the upper Muddy Creek watershed are small in relation to other habitats that have been targeted for nonnative fish removal efforts in the Colorado River basin. As a result, sampling efficiencies within this system are generally high due to a lack of habitat complexity and small stream size. Logistical constraints are also limited in the upper Muddy Creek watershed due to abundant road access. By incorporating the native warmwater fish community into ongoing restoration activities within the upper Muddy Creek watershed, many of the limitations plaguing removal efforts in large-river habitats may be surmountable. Given the abundance and diversity of native fishes in the upper Muddy Creek watershed, steps to remove the threats posed by exotic fishes should be considered within a cohesive ecosystem recovery plan that considers the biological, social, and economic costs of nonnative fish removal scenarios.

*Habitat management*

Significant variability among stream segments within channel-unit-scale models for each of the three species and both subadults and adults in the upper Muddy Creek watershed suggests that segment-scale influences of movement barriers on the availability of limited resources may

be of importance. For example, the importance of maximum channel unit depth within Segment 1 suggests that providing access to deep remnant pool habitats may be important during drought conditions. Providing for movements among stream segments that allow for unimpeded utilization of required or limited resources may be an appropriate management action. By removing impediments to the movement of fishes among segments, the useable area available to each of the target species could be functionally increased.

Though the ultimate effects of movement barriers on the viability of populations of the three target species remain unknown in the upper Muddy Creek watershed, habitat management prescriptions that act to increase the connectivity of the stream network should be considered. However, the removal of a barrier to movement at the downstream boundary of the study area would facilitate invasion into the study area by additional exotic fishes, such as fathead minnows, known to occupy Muddy Creek below the downstream boundary (Beatty 2005). Conversely, the removal or reconstruction of instream structures in the middle of the study area to allow fish movement would act to reconnect the stream network and possibly increase access of native fishes to required habitats.

An additional consideration relates to ongoing nonnative fish removal efforts within the upper portions of the watershed. Barriers below these areas should be maintained to preclude reinvasion of the restored areas by introduced fishes occupying downstream areas. However, the placement of fish barriers in Muddy Creek to facilitate reintroduction of native coldwater fishes has fragmented populations of the three target species, as demonstrated by the presence of the three target species within the currently identified reintroduction area (see Figure 1.1, Segment 3). It is not known if the remaining individuals of the three target species within Segment 3 represent viable populations, or if they require connectivity to adjacent segments. While some

barriers within the watershed are desirable for the removal of exotic fishes or prevention of future invasions by exotic fishes, new instream structures should be avoided where they may act to fragment existing populations into smaller isolated populations, thus decreasing population viability.

## Passive Management Opportunities

Active habitat management approaches are common where the management goal is to increase the abundance of a single target species. However, where fish community or ecosystem management is the goal, a detailed understanding of the diverse habitat requirements of each of the species present is necessary to ensure maintenance of suitable habitats. As observed during this study, the warmwater fish community within the upper Muddy Creek watershed exhibited diverse habitat associations both among species and among subadults and adults. Incorporation of these diverse habitat associations into an active habitat management plan that prescribes instream treatments to increase habitat suitability would be difficult and possibly inappropriate given the limited predictive capability of physical habitat models.

Thresholds of occurrence that represented abiotic constraints to the distributions of two length classes of the three target species provided understanding related to the suitability of physical habitats. Though thresholds of elevation provide little to managers in regards to active management opportunity, they can be though of as surrogates for a much larger class of basin-scale habitat characteristics. These characteristics likely include cold water temperature, small stream size, large amounts of overhanging cover, and several other physical characteristics of what is frequently referred to as coldwater habitat. For the three target species, maintenance of

warmwater habitats, with their own characteristic habitat features such as warm water temperatures, high turbidity, and submergent vegetation may be of importance.

Thresholds of stream drying are more tangible. For example, adult bluehead suckers were never observed where greater than 54% of a reach's length was dry. However, each of the remaining species and length classes were present in reaches where greater than 81% of the reach was dry. Except in the case of adult bluehead suckers, given the limited number of reaches containing greater than 81% of their length as dry channel, this feature did not appear to significantly influence the distribution of either length class of the three target species.

The two most consistent habitat associations among subadults and adults of all three of the target species were positive associations with both rock substrates and pools. Threats to the persistence of these two habitat components include sedimentation, channel incision, and attenuation of hydrologic variation. Sedimentation results from the addition of fine sediments such as silt to stream channels, leading to an increase in the proportion of fine sediments and a decrease in the availability or suitability (i.e., embeddedness) of remaining rock substrates. Increased sediment delivery to stream channels can result from the construction of roads and associated alteration of surface drainage patterns, high-intensity livestock grazing in riparian areas (Armour et al. 1991), and other surface disturbances which increase upland and riparian soil erosion. Though the biological effects of sedimentation include a variety of ecological interactions (Waters 1995), sedimentation could act to embed or cover the limited rock substrates in the upper Muddy Creek watershed.

Channel incision occurs when the base elevation of the stream channel adjusts to account for an alteration of geomorphic parameters such as sediment supply, flow volume, or channel roughness (e.g., riparian vegetation). Channel incision has been shown to simplify channel

geometry and result in the loss of pool habitats (Shields et al. 1994). The construction of impoundments can affect both the sediment supply and flow volume under which the channel dimensions have evolved. Discharge of waters can similarly affect the flow volumes that have resulted in the observed channel dimensions. Removal of riparian vegetation can also act to destabilize stream channels by decreasing the roughness of the channel boundary.

Attenuation of hydrologic variation occurs when peaks in streamflow and sediment transport are dampened and baseflows are increased, most commonly resulting from the storage of water within impoundments. Peak discharges are largely responsible for observed channel dimensions and mesohabitat variability (pool-riffle complexes). Changes to fish communities and habitats resulting from the construction of impoundments have been reported for all three of the target species (Vanicek 1967; Bestgen and Crist 2000).

Another consideration in the management of habitats for the three target species is the role of stream drying. When considering the results of thresholds of stream drying and the output of multiple-regression models, bluehead suckers and flannelmouth suckers were most common in areas with perennial streamflow while roundtail chubs were common in areas with high degrees of stream drying as well as areas with perennial streamflow. In this case, maintenance of a diversity of habitat conditions may provide suitable habitats for all three of the target species.

The maintenance of critical habitat components such as rock substrates and pools is likely the role of passive management approaches. For example, avoiding actions that may lead to increased sedimentation, channel incision, and attenuation of hydrologic variation is likely to ensure the persistence of rock substrates and pool habitat. Given our limited understanding of the potential outcomes of physical habitat restoration activities, maintenance of the watershed

processes that lead to instream habitat diversity, such as naturally extreme hydrologic events (i.e., droughts and floods), is the most reliable method of ensuring the persistence of habitats required by each of the three target species (Beechie and Bolton 1999).

## Summary

The abundance of bluehead suckers, flannelmouth suckers, and roundtail chubs, as well as other native warmwater and coldwater fishes in the upper Muddy Creek watershed, make it one of the highest conservation priorities for native fishes in Wyoming. Introduced creek chubs and white suckers in the watershed may be affecting populations of the three target species through competitive and predatory interactions as well as through hybridization. Active management of exotic fishes will be an important consideration when designing holistic ecosystem management plans. The removal of barriers to the upstream movement of fishes would act to reconnect the stream network and may increase the viability of populations of the three target species. Avoidance of additional habitat fragmentation would ensure the connectivity of populations and availability of diverse habitats. Though the trends of populations of the target species remain unknown, the abundance of each of the target fishes within the study area in relation to other areas of Wyoming suggests that habitats remain suitable. Passive management of the watershed processes that have created and maintained habitats within the upper Muddy Creek watershed required to fulfill the diverse habitat requirements of the native warmwater fish community should enable conservation at a significant scale.

Progress made towards meeting habitat objectives for native coldwater fishes underscores the utility of collaboration among diverse agencies and individuals. The Muddy Creek CRM

group, comprised of representatives from the BLM, WGFD, conservation organizations, county and local governments, as well as local landowners, has been effective at designing and implementing improved grazing systems within the coldwater portions of the upper Muddy Creek watershed. Expansion of the habitat objectives of this group to include the diverse requirements of the native warmwater fish assemblage would act to increase the feasibility of implementing a holistic ecosystem-based approach to fish community management. As new economic development opportunities are identified within the watershed, such as coalbed methane, involvement of industry representatives will be necessary if progress is to continue to be made towards meeting habitat objectives.

# Chapter V

## Research Needs

The genetic identification and quantification of introgression of the white sucker genome into populations of bluehead suckers and flannelmouth suckers should become a high priority. Initial results from microsatellite DNA and AFLP analyses used to develop species diagnostics have been encouraging, but additional work is needed to confirm preliminary results (Dave McDonald, University of Wyoming, personal communication). Species-diagnostic AFLP markers have proven particularly encouraging for distinguishing among the native species, white suckers, and hybrid individuals. Additional work is needed to determine if introgression of the white sucker genome into populations of bluehead suckers and flannelmouth suckers has occurred. One way to answer this question is through a controlled breeding experiment where white suckers are spawned with each of the native species of catostomids. Their first-generation offspring could subsequently be backcrossed with pure individuals of each of the native species to assess their fertility. Genetic analyses could be conducted for the pure individuals and each generation of hybrid offspring, validating previous AFLP analyses and refining the genetic signal of hybrid individuals in order to test populations of the two native catostomids from other watersheds for signs of introgression with white suckers.

In addition to the identification and quantification of introgressive hybridization using species-diagnostic genetic markers, future efforts should assess the levels of genetic variation present both within and among populations of bluehead suckers and flannelmouth suckers in the Colorado River basin. Knowledge of these levels of genetic variation will be important when

considering the genetic similarity of populations within reintroduction planning efforts that necessitate population viability analyses.

The ability to model basic population parameters such as growth rates, mortality rates, and recruitment is limiting our understanding of the population dynamics of the three target species. An age and growth study has been initiated to begin to answer these basic questions (Mike Quist, Wayne Hubert, Frank Rahel, University of Wyoming, personal communication). This study seeks to identify both lethal and non-lethal methods for aging individuals of the three species. Upon completion of this research, subsequent efforts should focus on applying length-at-age statistics to populations of the three target species within the upper Muddy Creek watershed in order to model population parameters and enable the creation of monitoring protocols that consider trends in these parameters.

Several studies have documented substantial movements by bluehead suckers, flannelmouth suckers, and roundtail chubs related to spawning or other seasonal movements among habitats such as larval drift (Bezzerides and Bestgen 2002). Given the potential for habitat fragmentation to affect access to required habitats or to influence source-sink dynamics, an understanding of the movements of the three species within the upper Muddy Creek watershed is needed. The demographic consequences of instream structures as they relate to movements among spawning and refuge habitats is the focus of an ongoing research effort (Mike Quist, Wayne Hubert, Frank Rahel, University of Wyoming, personal communication). The results of this effort will be useful to identify where active management prescriptions may be applied in order to reconnect habitats and to identify additional research needs pertaining to the movement of the three target species within the upper Muddy Creek watershed.

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Table 1.1. Fish species known to occur in the Colorado River drainage in Wyoming.

| Common name | Scientific name |
|---|---|
| **Native species** | |
| Bluehead sucker | *Catostomus discobolus* |
| Flannelmouth sucker | *Catostomus latipinnis* |
| Mountain sucker | *Catostomus platyrhynchus* |
| Mottled sculpin | *Cottus bairdi* |
| Bonytail* | *Gila elegans* |
| Roundtail chub | *Gila robusta* |
| Colorado River cutthroat trout | *Oncorhynchus clarki pleuriticus* |
| Mountain whitefish | *Prosopium williamsoni* |
| Colorado pikeminnow* | *Ptychocheilus lucius* |
| Speckled dace | *Rhinichthys osculus* |
| Razorback sucker* | *Xyrauchen texanus* |
| **Exotic species** | |
| White sucker | *Catostomus commersoni* |
| Lake chub | *Couesius plumbeus* |
| Common carp | *Cyprinus carpio* |
| Utah chub | *Gila atraria* |
| Leatherside chub | *Gila copei* |
| Channel catfish | *Ictalurus punctatus* |
| Burbot | *Lota lota* |
| Smallmouth bass | *Micropterus dolomieui* |
| Sand shiner | *Notropis stramineus* |
| Golden trout | *Oncorhynchus aguabonita* |
| Rainbow trout | *Oncorhynchus mykiss* |
| Kokanee | *Oncorhynchus nerka* |
| Fathead minnow | *Pimephales promelas* |
| Longnose dace | *Rhinichthys cataractae* |
| Redside shiner | *Richardsonius balteatus* |
| Brown trout | *Salmo trutta* |
| Brook trout | *Salvelinus fontinalis* |
| Lake trout | *Salvelinus namaycush* |
| Creek chub | *Semotilus atromaculatus* |
| Arctic grayling | *Thymallus arcticus* |

*Probably extirpated in Wyoming.

Table 2.1. Location of study area boundaries and potential movement barriers in the upper Muddy Creek study area during 2003 and 2004. Positions represent post-processed coordinates collected using a Trimble GeoXT receiver. All coordinates are presented in the UTM Zone 13, NAD 27 projection. Horizontal precisions represent 68% probability that the position lies within the specified distance.

| Type | Location description | Easting | Northing | Horizontal precision (m) | Std dev |
|---|---|---|---|---|---|
| Downstream boundary | headcut stabilization structure | 271659.52 | 4592962.59 | 2.44 | 0.0005 |
| Potential movement barrier | irrigation diversion structure | 288505.61 | 4595035.49 | 0.31 | 0.0002 |
| Movement barrier | gabion fish barrier - Muddy Creek | 293009.68 | 4593321.83 | 1.20 | 0.0009 |
| Movement barrier | gabion fish barrier - Littlefield Creek | 296500.54 | 4591483.32 | 1.82 | 0.0007 |
| Culvert | impassible road culvert | 296246.91 | 4588690.85 | 1.62 | 0.0005 |
| Upstream boundary | known coldwater habitat | 299875.59 | 4594174.67 | 2.83 | 0.0001 |

Table 2.2. Morphological characters used to distinguish among bluehead suckers, flannelmouth suckers, and hybrid individuals sampled during the summer and fall of 2003 and 2004 in the upper Muddy Creek study area. Adapted from keys found in Baxter and Stone (1995).

| | Morphological character | | | | | |
|---|---|---|---|---|---|---|
| Species | Scale pattern | Dorsal fin shape | Caudle peduncle depth | Head length | Cartilaginous biting ridge | Lateral notch in lips |
| Bluehead sucker | fine | not sickle-shaped | shallow | short | present | yes |
| Flannelmouth sucker | fine | sickle-shaped | shallow | long | absent | no |
| White sucker | large | not sickle-shaped | deep | intermediate | absent | no |

Table 3.1. Stream segments and number of randomly selected 200-m reaches sampled during the summer and fall of 2003 and 2004 in the upper Muddy Creek study area.

| *Segment* | *Length (km)* | *Random reaches* |
|---|---|---|
| 1 | 42.8 | 30 |
| 2 | 26.7 | 20 |
| 3 | 12.3 | 7 |
| All segments | 81.8 | 57 |

Table 3.2. Correlations among reach-scale and channel-unit-scale habitat attributes measured during the summer and fall of 2003 and 2004 within the upper Muddy Creek study area.

**Reach-scale**

| | Variables | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| Variables | Wetted area | Sampled area | Reach length | Wetted length | Sampled length | Average wetted width | Percent reach length dry | Percent riffle habitat |
| Sampled area | 0.97 | | | | | | | |
| Reach length | 0.49 | 0.48 | | | | | | |
| Wetted length | 0.87 | 0.86 | 0.49 | | | | | |
| Sampled length | 0.80 | 0.87 | 0.49 | 0.94 | | | | |
| Average wetted width | 0.93 | 0.90 | 0.31 | 0.66 | 0.61 | | | |
| Percent reach length dry | -0.62 | -0.61 | 0.17 | -0.77 | -0.70 | -0.52 | | |
| Percent riffle habitat | 0.30 | 0.09 | -0.04 | 0.32 | 0.01 | 0.22 | -0.40 | |
| Percent rock substrate | 0.18 | 0.11 | 0.04 | 0.11 | 0.03 | 0.23 | -0.09 | 0.26 |

**Channel-unit-scale**

| | Variables | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| Variables | Surface area | Length | Mean width | Maximum depth | Maximum residual pool depth | Area of deep water habitat | Area of deep water in residual | Overhaning cover | Submergent vegetation |
| Length | 0.90 | | | | | | | | |
| Mean width | 0.67 | 0.41 | | | | | | | |
| Maximum depth | 0.54 | 0.42 | 0.58 | | | | | | |
| Maximum residual pool depth | 0.53 | 0.44 | 0.57 | 0.92 | | | | | |
| Area of deep water habitat | 0.67 | 0.50 | 0.43 | 0.59 | 0.52 | | | | |
| Area of deep water in residual pool | 0.55 | 0.39 | 0.35 | 0.49 | 0.42 | 0.96 | | | |
| Overhanging cover | 0.36 | 0.37 | 0.15 | 0.25 | 0.25 | 0.32 | 0.35 | | |
| Submergent vegetation | -0.14 | -0.13 | -0.17 | -0.11 | -0.13 | -0.04 | 0.01 | -0.01 | |
| Brush | 0.25 | 0.27 | 0.14 | 0.31 | 0.31 | 0.17 | 0.19 | 0.33 | -0.04 |

Table 3.3. Thresholds of occurrence for two length classes of three species within the upper Muddy Creek watershed during the summer and fall of 2003 and 2004.

| | Habitat feature | | | |
|---|---|---|---|---|
| Length class | Maximum elevation (above mean sea level) | Maximum percent of reach dry (%) | Minimum sampled area ($m^2$) | Minimum depth (m) |
| | | **Bluehead sucker** | | |
| Adult | 2,228 | 53.04 | 127.15 | 0.25 |
| Subadult | 2,228 | 81.10 | 19.20 | 0.12 |
| | | **Flannelmouth sucker** | | |
| Adult | 2,186 | 82.08 | 53.46 | 0.34 |
| Subadult | 2,195 | 81.10 | 79.48 | 0.12 |
| | | **Roundtail chub** | | |
| Adult | 2,228 | 81.10 | 19.20 | 0.23 |
| Subadult | 2,228 | 93.05 | 17.46 | 0.12 |

Table 3.4. Multi-model averaged parameter estimates (β), their associated precision (SE), and the sum of the Akaike weights ($\Sigma w_i$) from candidate multiple-linear-regression models containing each model parameter predicting the abundance of three species and two length classes at the reach scale. Multiple-linear-regression models are given by: $\hat{y} = \beta_{\text{intercept}} + \beta_{\text{sampled area}}(X_{\text{sampled area}}) + \beta_{\text{rock substrates}}(X_{\text{rock substrate}}) + \beta_{\text{proportion of reach length dry}}(X_{\text{proportion of reach length dry}})$.

| | Intercept | | | Sampled area | | | Rock substrate | | | Proportion of reach length dry | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Length class | $\beta$ | SE | $\Sigma w_i$ | $\beta$ | SE | $\Sigma w_i$ | $\beta$ | SE | $\Sigma w_i$ | $\beta$ | SE | $\Sigma w_i$ |
| **Bluehead sucker** | | | | | | | | | | | | |
| Subadult | 1.47 | 0.38 | 0.93 | -0.0017 | 0.00061 | 0.93 | 3.93 | 0.70 | 0.93 | 0.02 | 0.53 | 0.20 |
| Adult | 0.12 | 0.22 | 0.91 | | | | 2.69 | 0.64 | 0.91 | | | |
| **Flannelmouth sucker** | | | | | | | | | | | | |
| Subadult | 0.45 | 0.25 | 0.98 | 0.0005 | 0.00056 | 0.33 | 1.37 | 0.62 | 0.86 | -0.47 | 0.42 | 0.38 |
| Adult | -0.01 | 0.26 | 1.00 | 0.0011 | 0.00051 | 0.75 | 0.85 | 0.57 | 0.52 | -0.38 | 0.38 | 0.36 |
| **Roundtail chub** | | | | | | | | | | | | |
| Subadult | 0.09 | 0.45 | 1.00 | 0.0054 | 0.00083 | 1.00 | 0.88 | 0.92 | 0.32 | 3.81 | 0.52 | 1.00 |
| Adult | 0.26 | 0.34 | 0.95 | 0.0019 | 0.00063 | 0.95 | 0.58 | 0.71 | 0.28 | 1.84 | 0.48 | 0.95 |

Table 3.5. Multi-model averaged parameter estimates (β), their associated precision (SE), and the sum of the Akaike weights ($\Sigma w_i$) from candidate multiple-linear-regression models containing each model parameter predicting the abundance of three species and two length classes among all segments at the channel unit scale. Multiple-linear-regression models are given by: $\hat{y} = \beta_{intercept} + \beta_{pool}(X_{pool}) + \beta_{glide}(X_{glide}) + \beta_{maximum\ depth}(X_{maximum\ depth}) + \beta_{submergent\ vegetation}(X_{submergent\ vegetation})$. Insufficient numbers of flannelmouth suckers were detected (n=1) to compute linear-regression models within this segment.

| | Intercept | | | Pool | | | Glide | | | Maximum depth | | | Submergent vegetation | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Length class | $\beta$ | SE | $\Sigma w_i$ | $\beta$ | SE | $\Sigma w_i$ | $\beta$ | SE | $\Sigma w_i$ | $\beta$ | SE | $\Sigma w_i$ | $\beta$ | SE | $\Sigma w_i$ |
| **Bluehead sucker** | | | | | | | | | | | | | | | |
| Subadult | 0.82 | 0.42 | 1.00 | 0.47 | 0.42 | 1.00 | -0.13 | 0.43 | 1.00 | -0.02 | 0.31 | 0.26 | 0.01 | 0.01 | 0.26 |
| Adult | -0.25 | 0.35 | 0.93 | 0.51 | 0.35 | 0.93 | 0.13 | 0.34 | 0.93 | 0.52 | 0.23 | 0.81 | 0.69 | 0.33 | 0.74 |
| **Flannelmouth sucker** | | | | | | | | | | | | | | | |
| Subadult | -0.24 | 0.31 | 0.92 | 0.39 | 0.32 | 0.92 | 0.10 | 0.32 | 0.92 | 0.63 | 0.19 | 0.92 | 0.39 | 0.30 | 0.43 |
| Adult | -0.02 | 0.24 | 1.00 | -0.44 | 0.39 | 0.41 | -0.38 | 0.39 | 0.41 | 0.76 | 0.19 | 1.00 | 0.02 | 0.33 | 0.26 |
| **Roundtail chub** | | | | | | | | | | | | | | | |
| Subadult | -0.36 | 0.53 | 0.99 | 1.68 | 0.54 | 0.99 | 1.20 | 0.54 | 0.99 | 2.11 | 0.43 | 0.99 | -0.27 | 0.56 | 0.29 |
| Adult | -0.68 | 0.37 | 0.96 | 0.48 | 0.37 | 0.96 | 0.12 | 0.37 | 0.96 | 1.69 | 0.24 | 0.96 | 0.63 | 0.34 | 0.64 |

Table 3.6. Multi-model averaged parameter estimates (β), their associated precision (SE), and the sum of the Akaike weights ($\Sigma w_i$) from candidate multiple-linear-regression models containing each model parameter predicting the abundance of three species and two length classes within Segment 1 at the channel unit scale. Multiple-linear-regression models are given by: $\hat{y} = \beta_{intercept} + \beta_{pool}(X_{pool}) + \beta_{glide}(X_{glide}) + \beta_{maximum\ depth}(X_{maximum\ depth}) + \beta_{submergent\ vegetation}(X_{submergent\ vegetation})$.

| | Intercept | | | Pool | | | Glide | | | Maximum depth | | | Submergent vegetation | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Length class | $\beta$ | SE | $\Sigma w_i$ | $\beta$ | SE | $\Sigma w_i$ | $\beta$ | SE | $\Sigma w_i$ | $\beta$ | SE | $\Sigma w_i$ | $\beta$ | SE | $\Sigma w_i$ |
| **Bluehead sucker** | | | | | | | | | | | | | | | |
| Subadult | -0.03 | 1.11 | 1.00 | 0.79 | 1.11 | 1.00 | 0.14 | 1.11 | 1.00 | 0.19 | 0.31 | 0.30 | 0.60 | 0.48 | 0.43 |
| Adult | -0.21 | 0.47 | 1.00 | 0.23 | 0.59 | 0.73 | -0.01 | 0.59 | 0.73 | 0.49 | 0.19 | 0.85 | 0.58 | 0.58 | 0.73 |
| **Flannelmouth sucker** | | | | | | | | | | | | | | | |
| Subadult | -0.31 | 0.30 | 1.00 | 0.30 | 0.74 | 0.29 | 0.26 | 0.74 | 0.29 | 0.86 | 0.20 | 1.00 | 0.39 | 0.33 | 0.42 |
| Adult | 1.85 | 0.61 | 0.98 | -2.11 | 0.60 | 0.98 | -2.13 | 0.60 | 0.98 | 0.70 | 0.22 | 0.98 | 0.27 | 0.32 | 0.32 |
| **Roundtail chub** | | | | | | | | | | | | | | | |
| Subadult | -1.43 | 1.69 | 0.98 | 3.08 | 1.67 | 0.98 | 2.40 | 1.68 | 0.98 | 2.42 | 0.44 | 0.98 | 1.28 | 0.71 | 0.64 |
| Adult | -1.09 | 0.78 | 1.00 | 0.84 | 1.10 | 0.64 | 0.52 | 1.09 | 0.64 | 2.28 | 0.33 | 1.00 | 0.75 | 0.50 | 0.52 |

Table 3.7. Multi-model averaged parameter estimates (β), their associated precision (SE), and the sum of the Akaike weights ($\Sigma w_i$) from candidate multiple-linear-regression models containing each model parameter predicting the abundance of three species and two length classes within Segment 2 at the channel unit scale. Multiple-linear-regression models are given by: $\hat{y} = \beta_{intercept} + \beta_{pool}(X_{pool}) + \beta_{glide}(X_{glide}) + \beta_{maximum\ depth}(X_{maximum\ depth}) + \beta_{submergent\ vegetation}(X_{submergent\ vegetation})$.

| | Intercept | | | Pool | | | Glide | | | Maximum depth | | | Submergent vegetation | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Length class | $\beta$ | SE | $\Sigma w_i$ | $\beta$ | SE | $\Sigma w_i$ | $\beta$ | SE | $\Sigma w_i$ | $\beta$ | SE | $\Sigma w_i$ | $\beta$ | SE | $\Sigma w_i$ |
| **Bluehead sucker** | | | | | | | | | | | | | | | |
| Subadult | 1.73 | 0.80 | 1.00 | 0.21 | 0.99 | 0.76 | -0.33 | 0.99 | 0.76 | 0.29 | 0.54 | 0.37 | -0.11 | 1.14 | 0.31 |
| Adult | -0.25 | 0.75 | 1.00 | 0.90 | 0.86 | 0.83 | 0.30 | 0.86 | 0.83 | 1.00 | 0.43 | 0.79 | 1.12 | 0.83 | 0.46 |
| **Flannelmouth sucker** | | | | | | | | | | | | | | | |
| Subadult | -0.21 | 0.59 | 0.99 | 0.90 | 0.60 | 0.99 | 0.16 | 0.61 | 0.99 | 0.30 | 0.36 | 0.32 | 1.71 | 0.75 | 0.82 |
| Adult | -0.17 | 0.64 | 0.98 | 0.27 | 1.01 | 0.49 | 0.65 | 1.03 | 0.49 | 0.88 | 0.40 | 0.88 | -0.30 | 0.84 | 0.28 |
| **Roundtail chub** | | | | | | | | | | | | | | | |
| Subadult | 0.36 | 1.07 | 0.97 | 1.90 | 1.08 | 0.97 | 0.83 | 1.08 | 0.97 | 1.04 | 0.62 | 0.57 | 1.83 | 1.24 | 0.50 |
| Adult | -0.42 | 0.79 | 0.87 | 0.80 | 0.79 | 0.87 | 0.13 | 0.80 | 0.87 | 0.76 | 0.40 | 0.60 | 2.06 | 0.77 | 0.87 |

Table 3.8. Multi-model averaged parameter estimates (β), their associated precision (SE), and the sum of the Akaike weights ($\Sigma w_i$) from candidate multiple-linear-regression models containing each model parameter predicting the abundance of three species and two length classes within Segment 3 at the channel unit scale. Multiple-linear-regression models are given by: $\hat{y} = \beta_{intercept} + \beta_{pool}(X_{pool}) + \beta_{glide}(X_{glide}) + \beta_{maximum\ depth}(X_{maximum\ depth}) + \beta_{submergent\ vegetation}(X_{submergent\ vegetation})$. Insufficient numbers of flannelmouth suckers were detected (n=1) to compute linear-regression models within this segment.

| | Intercept | | | Pool | | | Glide | | | Maximum depth | | | Submergent vegetation | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Length class | $\beta$ | SE | $\Sigma w_i$ | $\beta$ | SE | $\Sigma w_i$ | $\beta$ | SE | $\Sigma w_i$ | $\beta$ | SE | $\Sigma w_i$ | $\beta$ | SE | $\Sigma w_i$ |
| **Bluehead sucker** | | | | | | | | | | | | | | | |
| Subadult | 0.71 | 0.52 | 1.00 | 0.55 | 0.57 | 0.60 | 0.14 | 0.62 | 0.60 | 0.07 | 1.27 | 0.39 | -0.07 | 0.98 | 0.37 |
| Adult | 0.04 | 0.42 | 1.00 | 0.40 | 0.46 | 0.48 | 0.36 | 0.49 | 0.48 | 0.40 | 1.02 | 0.39 | 0.68 | 0.73 | 0.49 |
| **Roundtail chub** | | | | | | | | | | | | | | | |
| Subadult | 0.15 | 0.12 | 1.00 | -0.21 | 0.13 | 0.68 | -0.21 | 0.13 | 0.68 | 0.34 | 0.28 | 0.45 | -0.29 | 0.21 | 0.54 |
| Adult | -0.47 | 0.42 | 0.98 | 0.30 | 0.42 | 0.47 | -0.07 | 0.42 | 0.47 | 1.73 | 0.79 | 0.78 | 0.40 | 0.62 | 0.28 |

Table 3.9. Relative importance of channel-unit-scale model parameters (sum of Akaike weights, $\Sigma w_i$) predicting the abundance of the three target species among three stream segments within the upper Muddy Creek watershed.

| | Habitat variable | | | |
|---|---|---|---|---|
| Segment | Pool | Glide | Maximum Depth | Submergent vegetation |
| | | **Bluehead sucker - subadult** | | |
| 1 | 1.00 | 1.00 | 0.30 | 0.43 |
| 2 | 0.76 | 0.76 | 0.37 | 0.31 |
| 3 | 0.60 | 0.60 | 0.39 | 0.37 |
| | | **Bluehead sucker - adult** | | |
| 1 | 0.73 | 0.73 | 0.85 | 0.73 |
| 2 | 0.83 | 0.83 | 0.79 | 0.46 |
| 3 | 0.48 | 0.48 | 0.39 | 0.49 |
| | | **Flannelmouth sucker - subadult** | | |
| 1 | 0.29 | 0.29 | 1.00 | 0.42 |
| 2 | 0.99 | 0.99 | 0.32 | 0.82 |
| | | **Flannelmouth sucker - adult** | | |
| 1 | 0.98 | 0.98 | 0.98 | 0.32 |
| 2 | 0.49 | 0.49 | 0.88 | 0.28 |
| | | **Roundtail chub - subadult** | | |
| 1 | 0.98 | 0.98 | 0.98 | 0.64 |
| 2 | 0.97 | 0.97 | 0.57 | 0.50 |
| 3 | 0.68 | 0.68 | 0.45 | 0.54 |
| | | **Roundtail chub - adult** | | |
| 1 | 0.64 | 0.64 | 1.00 | 0.52 |
| 2 | 0.87 | 0.87 | 0.60 | 0.87 |
| 3 | 0.47 | 0.47 | 0.78 | 0.28 |

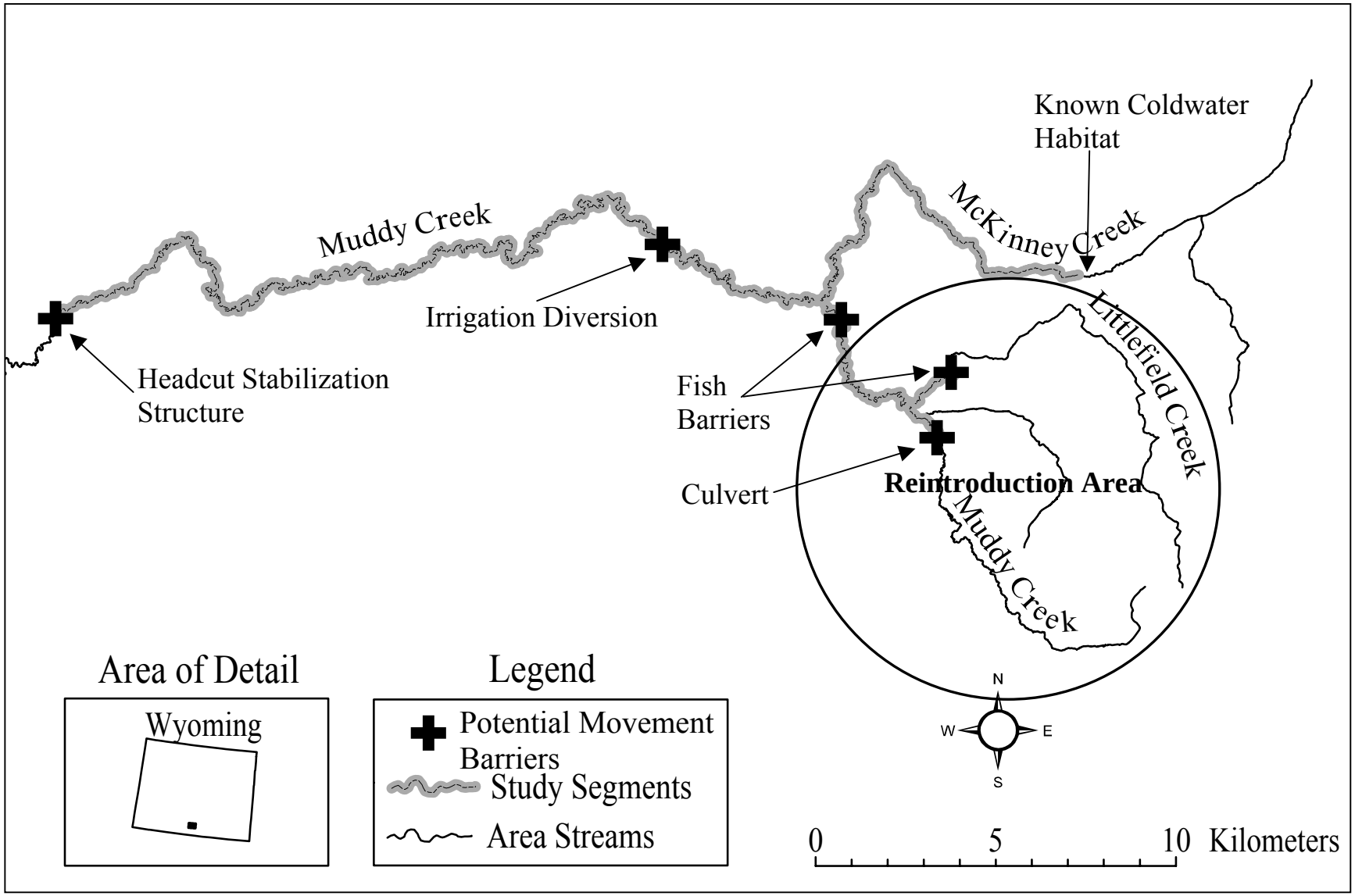


Figure 1.1. Study area and associated potential movement barriers within the upper Muddy Creek watershed of southern Carbon County, Wyoming.

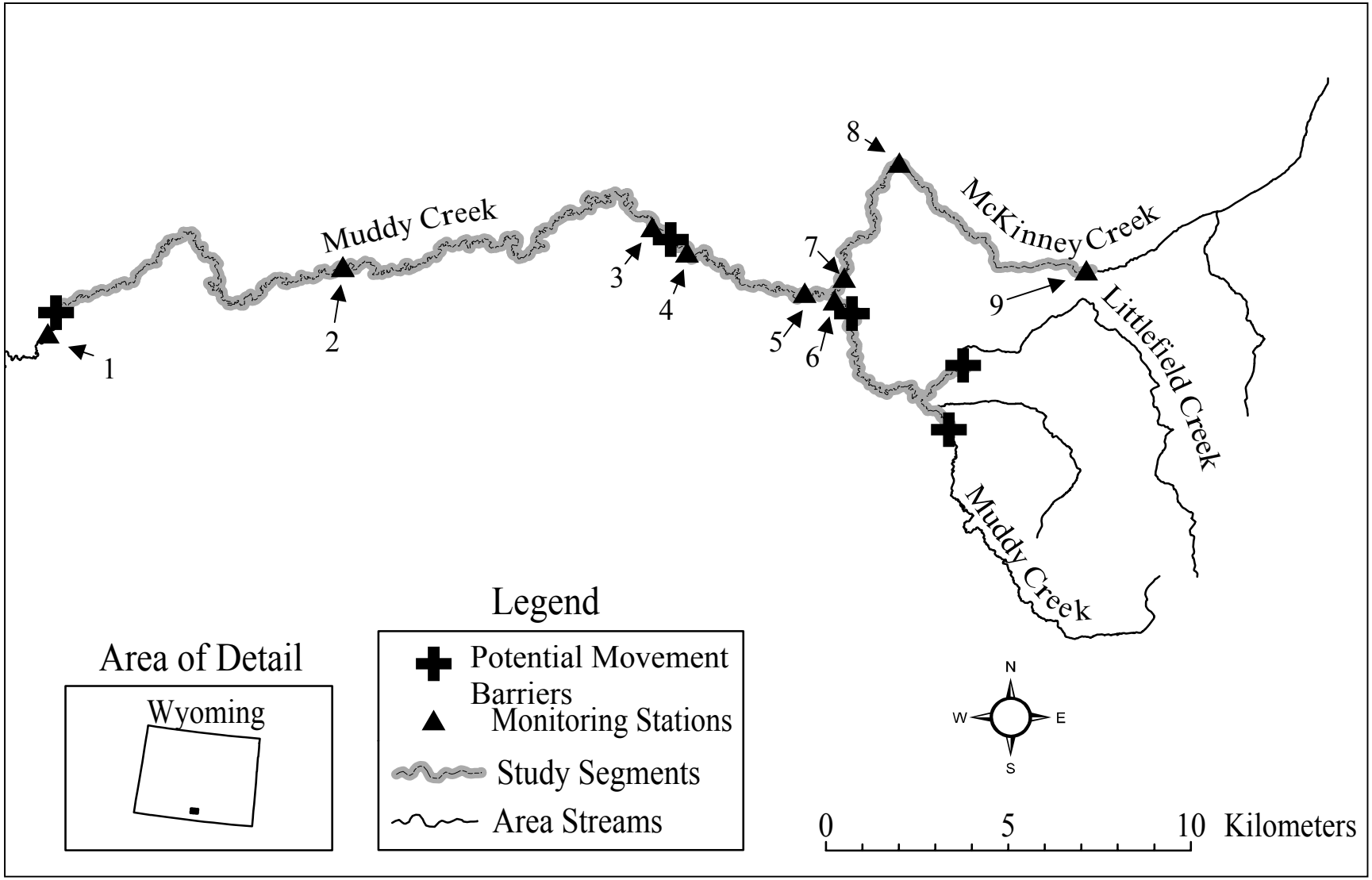


Figure 2.1. Hydrologic monitoring sites in relation to potential movement barriers within the upper Muddy Creek study area. Site details are presented in Appendix A.

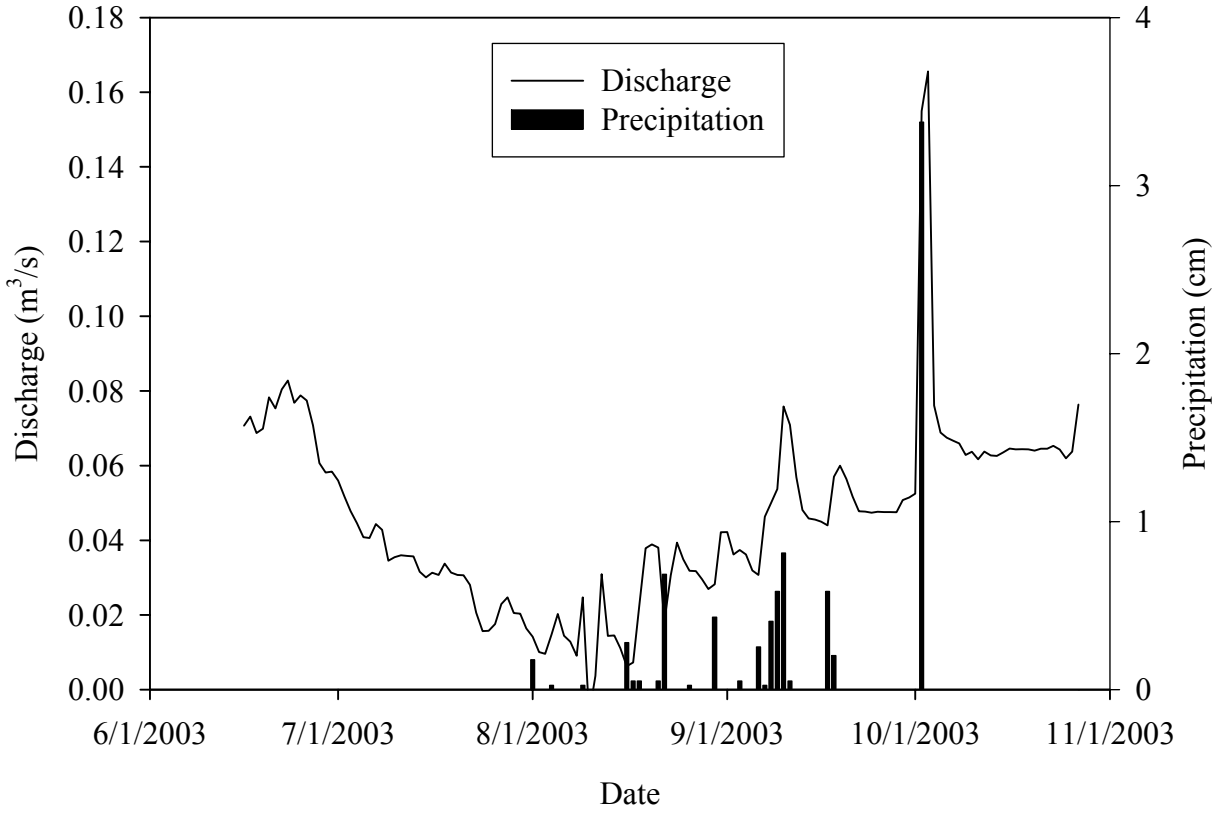


Figure 2.2. Mean daily discharge measured in the upper Muddy Creek study area at monitoring site six and daily precipitation measured at nearby monitoring site five during the summer and fall of 2003.

Figure 2.3. Mean daily water temperature measured in the upper Muddy Creek study area at monitoring sites one and two during the summer and fall of 2003

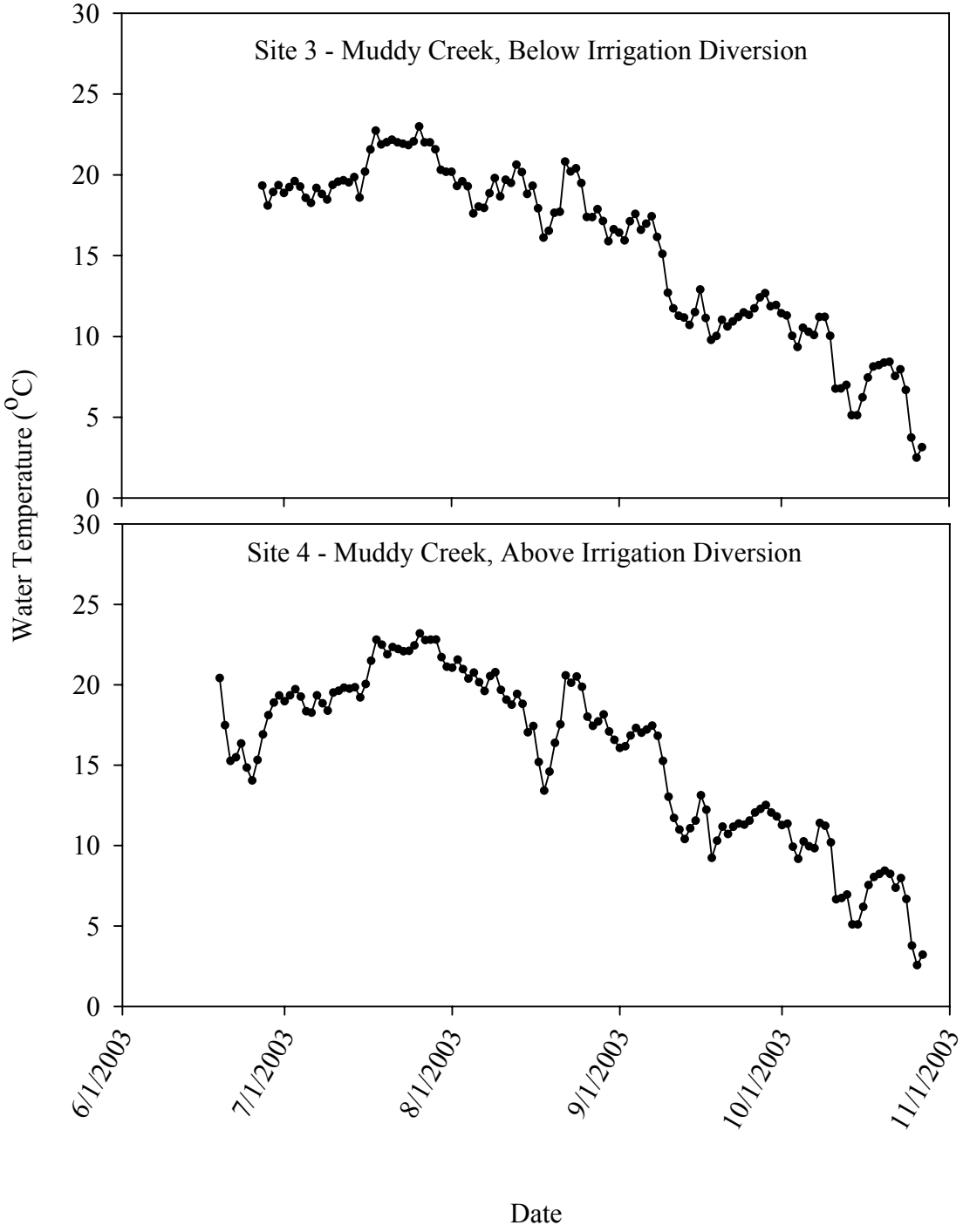


Figure 2.4. Mean daily water temperature measured upper Muddy Creek study area at monitoring sites three and four during the summer and fall of 2003

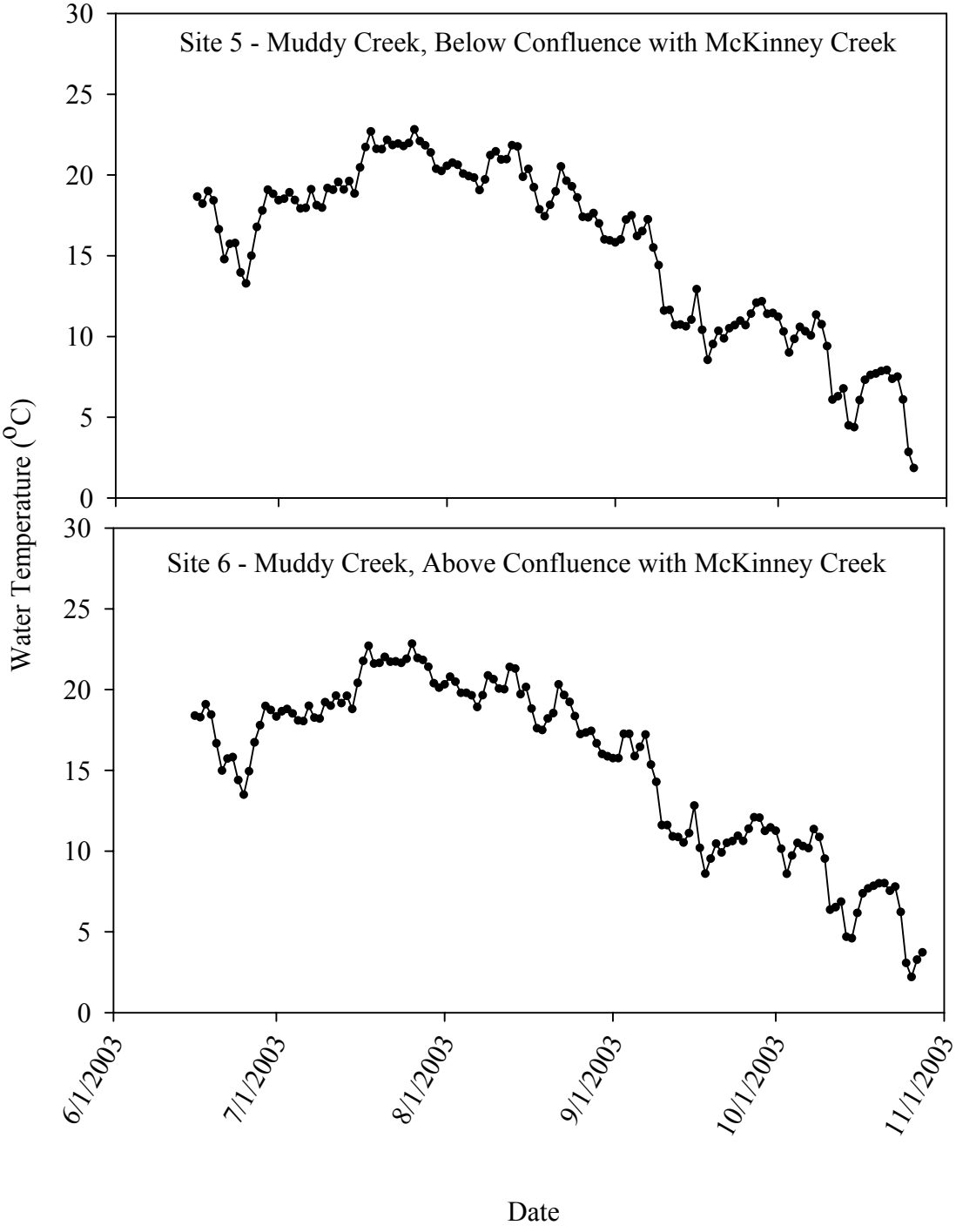


Figure 2.5. Mean daily water temperature measured in the upper Muddy Creek study area at monitoring sites five and six during the summer and fall of 2003

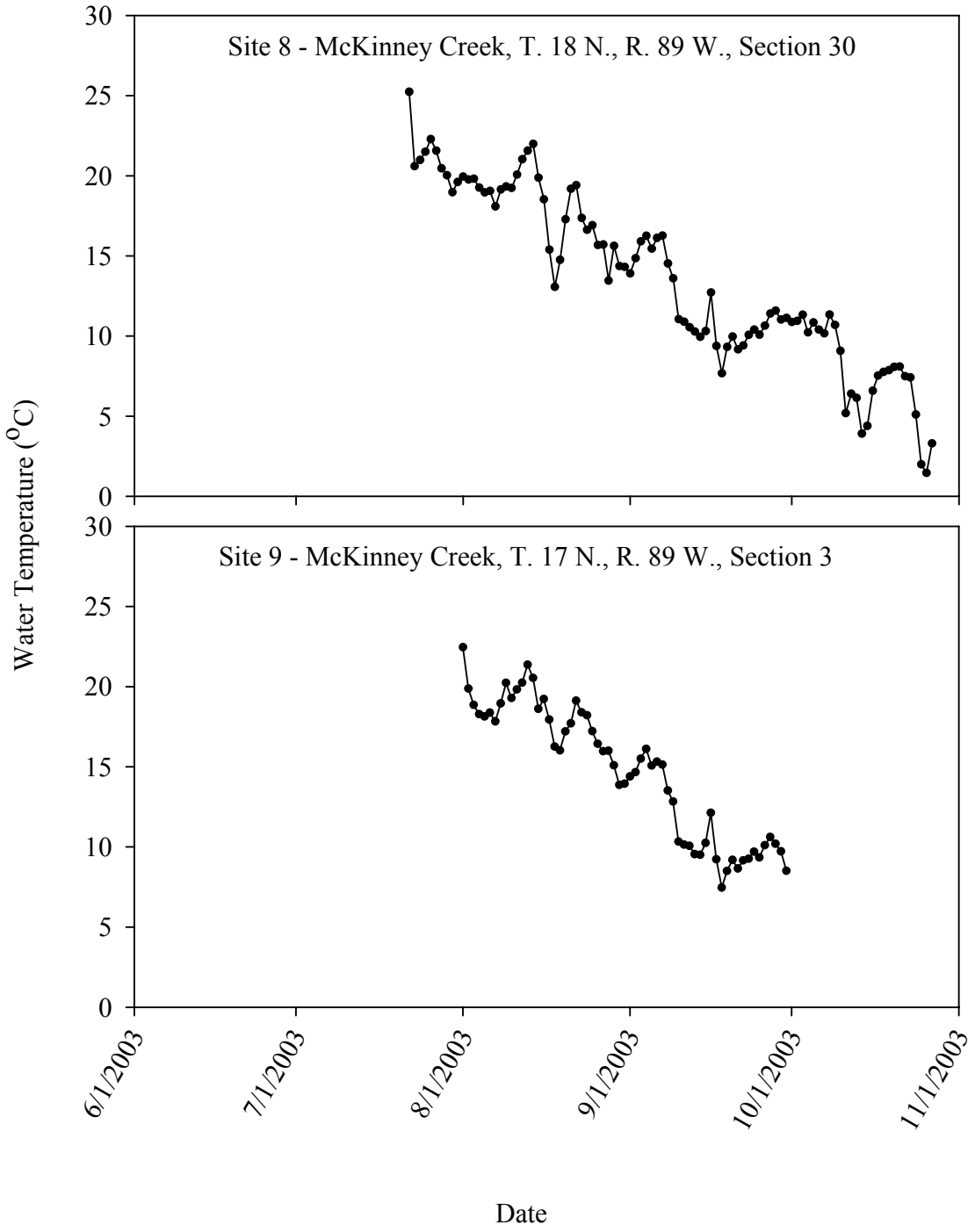


Figure 2.6. Mean daily water temperature measured in the upper Muddy Creek study area at monitoring sites eight and nine during the summer and fall of 2003.

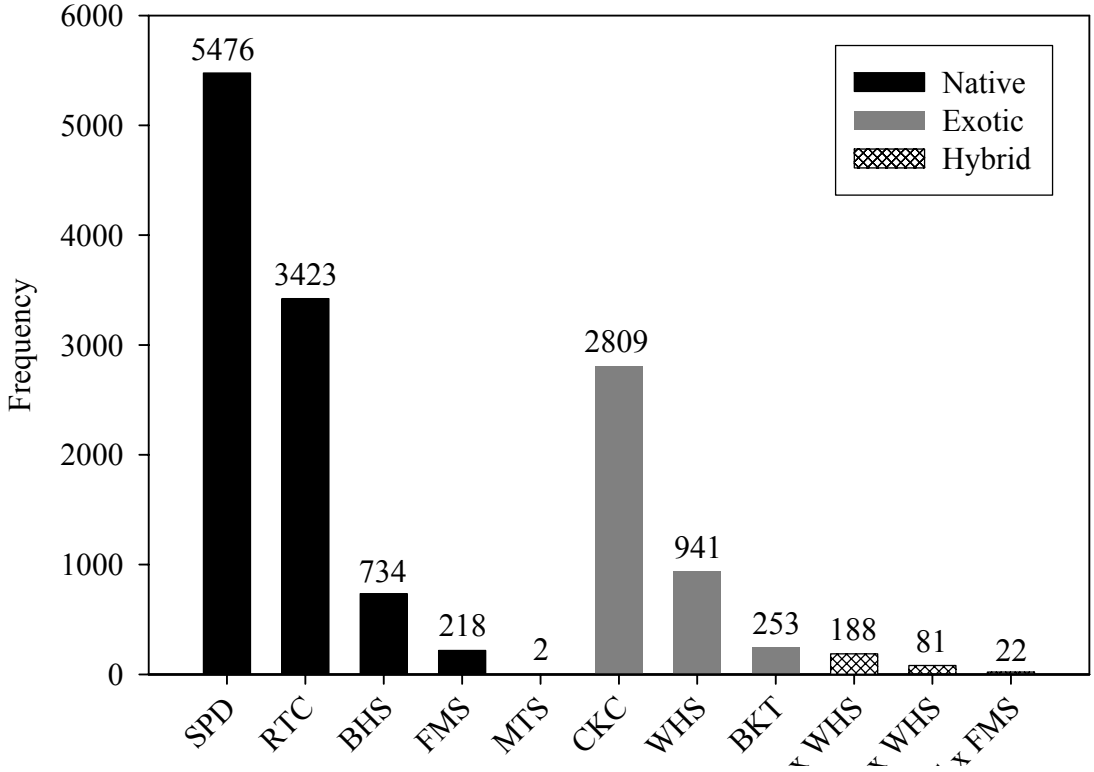


Figure 2.7. Total community composition of sites sampled within the upper Muddy Creek watershed during the summer and fall of 2003 and 2004. SPD = speckled dace, RTC = roundtail chub, BHS = bluehead sucker, FMS = flannelmouth sucker, MTS = mountain sucker, CKC = creek chub, WHS = white sucker, BKT = brook trout.

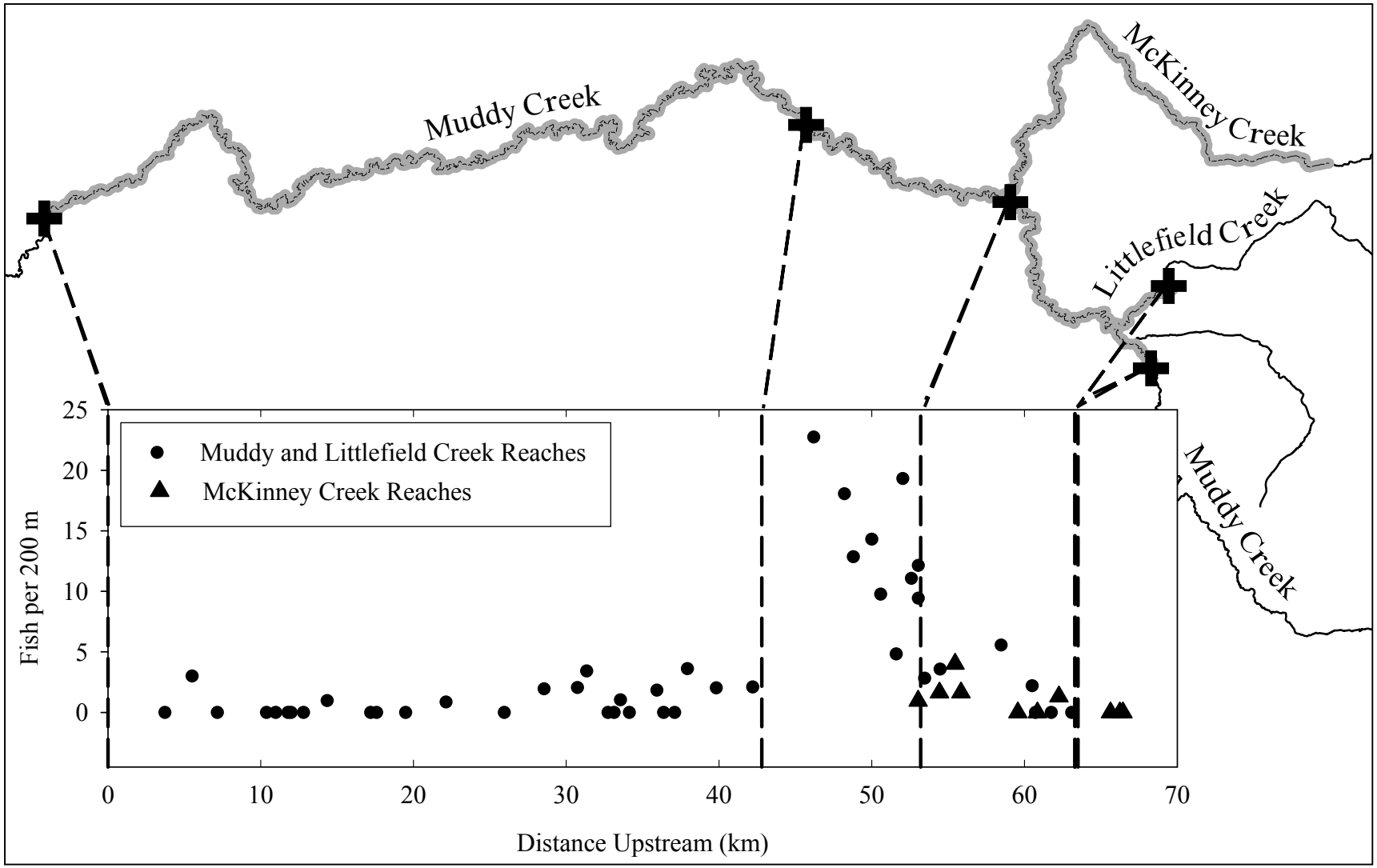


Figure 2.8. Distribution and relative abundance of bluehead sucker ≥ 150 mm TL sampled during the summer and fall of 2003 and 2004 in the upper Muddy Creek study area in relation to potential movement barriers.

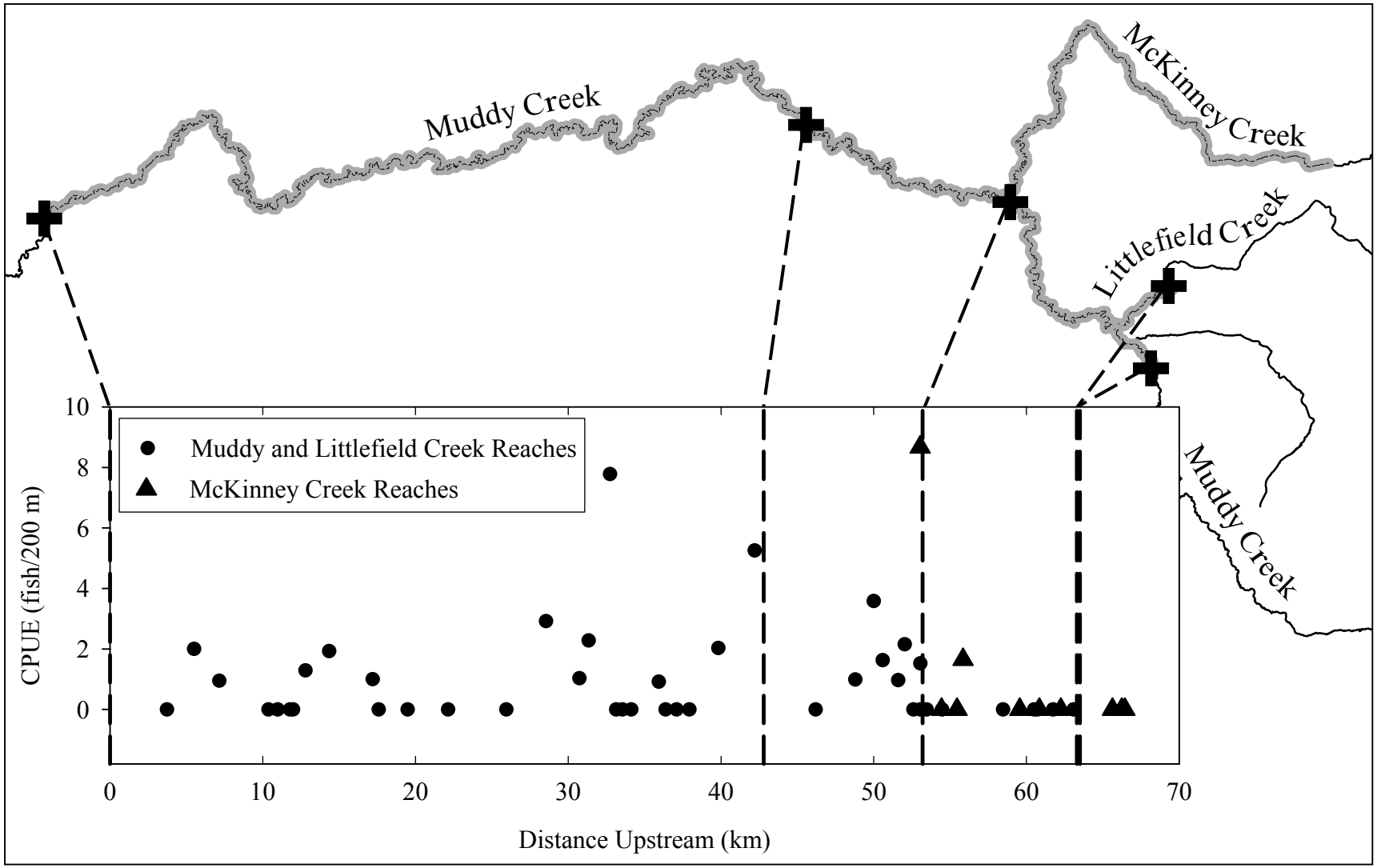


Figure 2.9. Distribution and relative abundance of flannelmouth sucker ≥ 200 mm TL sampled during the summer and fall of 2003 and 2004 in the upper Muddy Creek study area in relation to potential movement barriers.

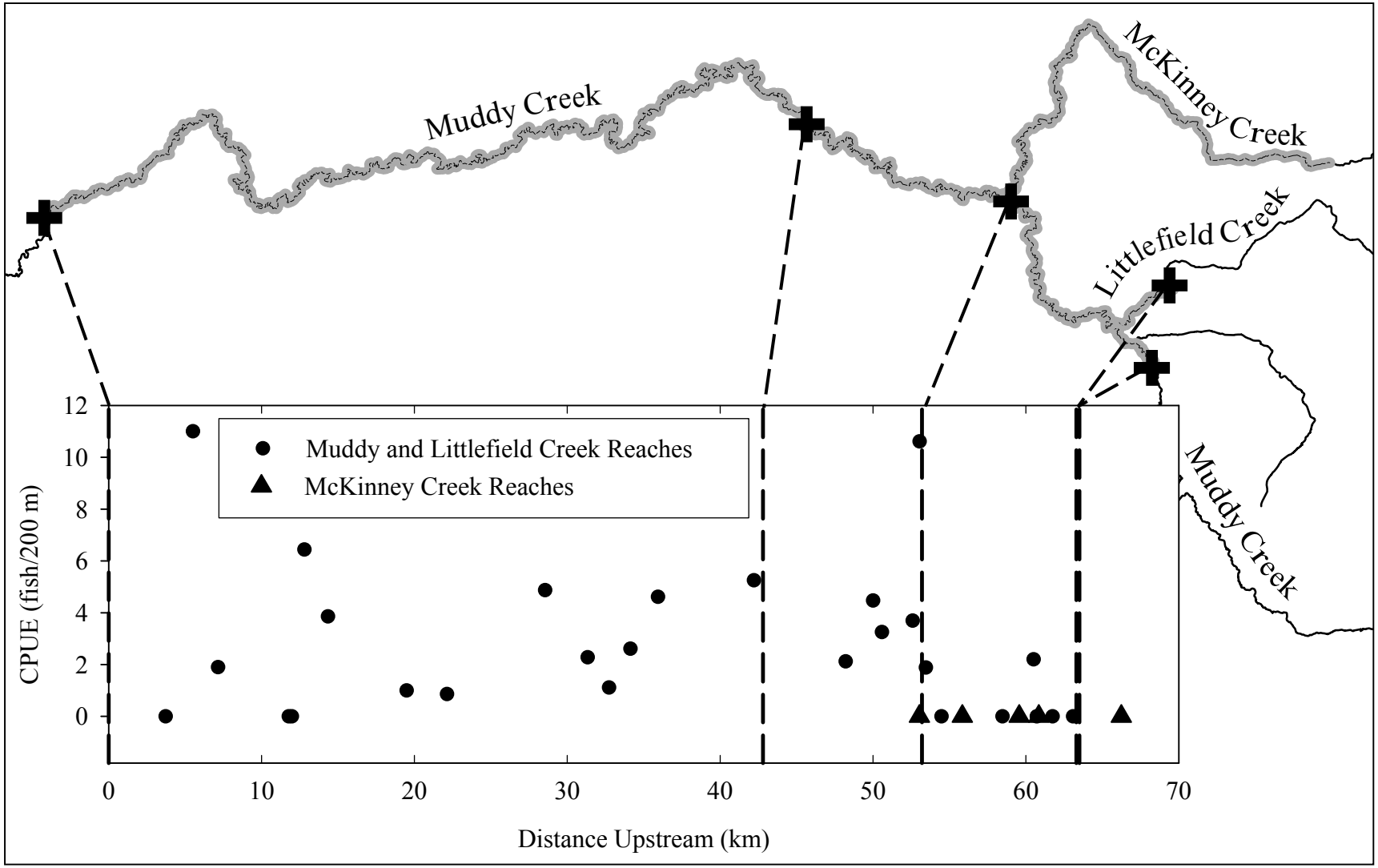


Figure 2.10. Distribution and relative abundance of roundtail chub ≥ 150 mm TL sampled prior to August during the summer of 2003 and 2004 in the upper Muddy Creek study area in relation to potential movement barriers.

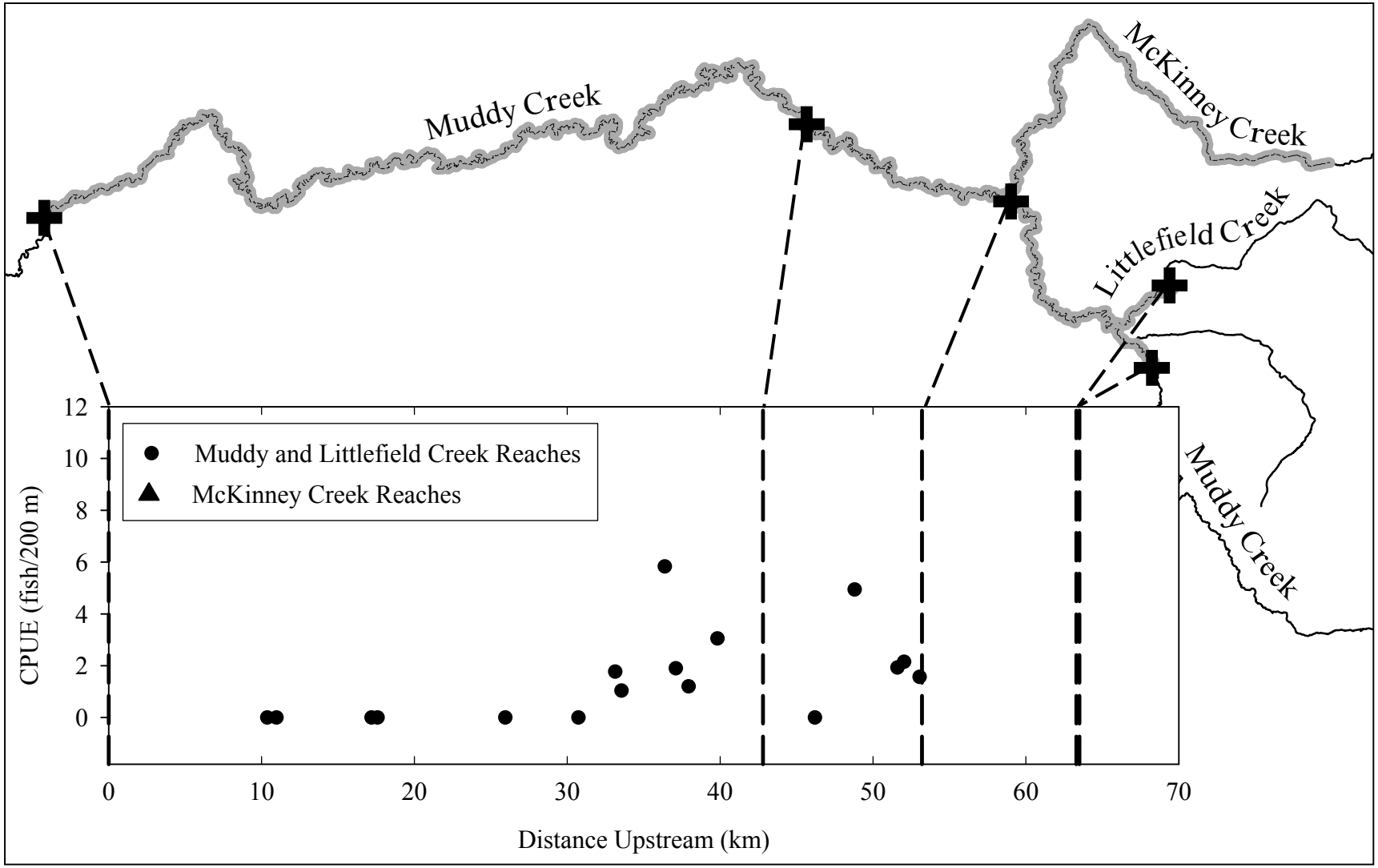


Figure 2.11. Distribution and relative abundance of roundtail chub ≥ 150 mm TL sampled subsequent to July 31 during the summer and fall of 2003 and 2004 in the upper Muddy Creek study area in relation to potential movement barriers.

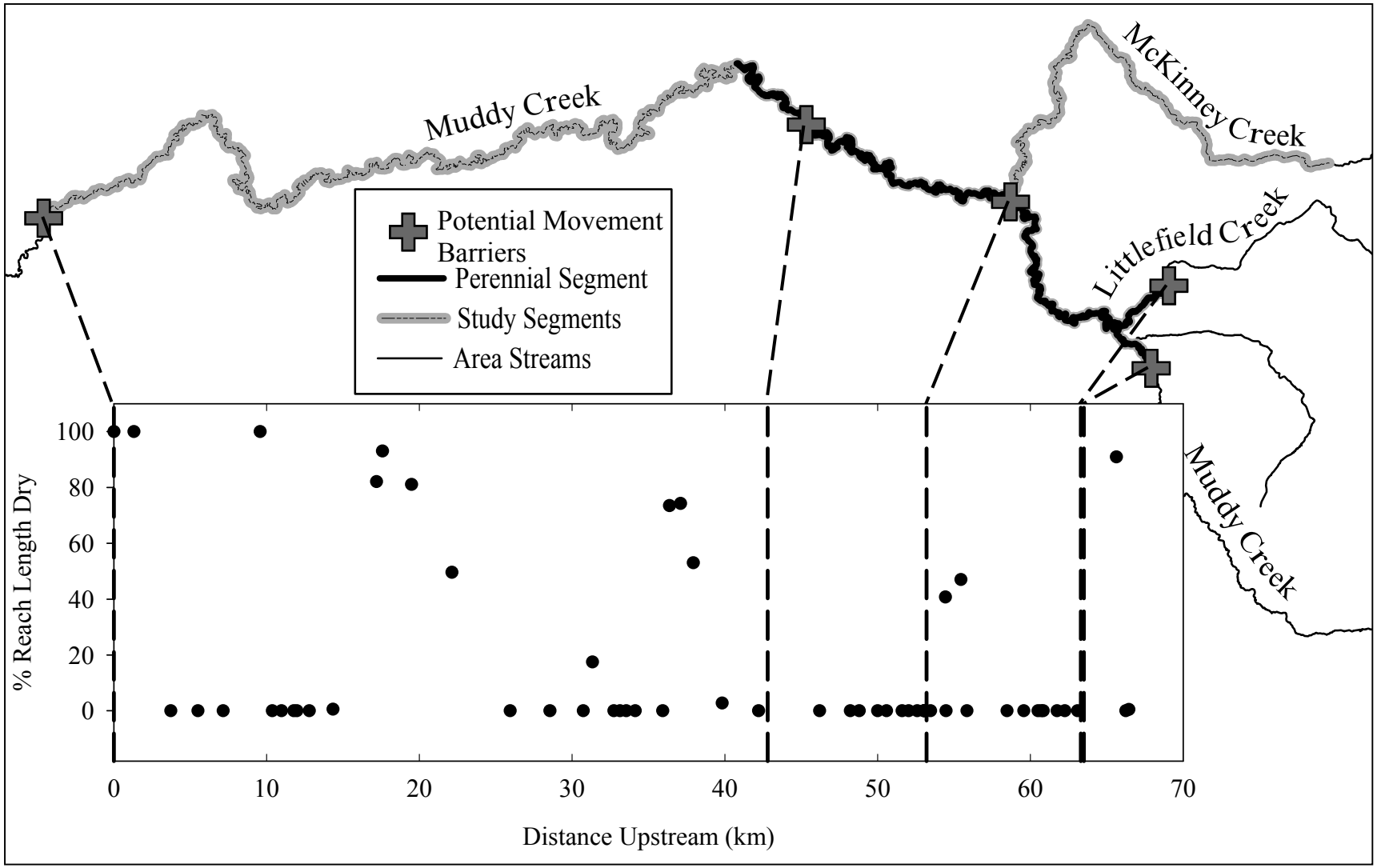


Figure 2.12. Proportion of reach length that was dry at the time of sampling during the summer and fall of 2003 and 2004 in the upper Muddy Creek study area in relation to potential movement barriers.

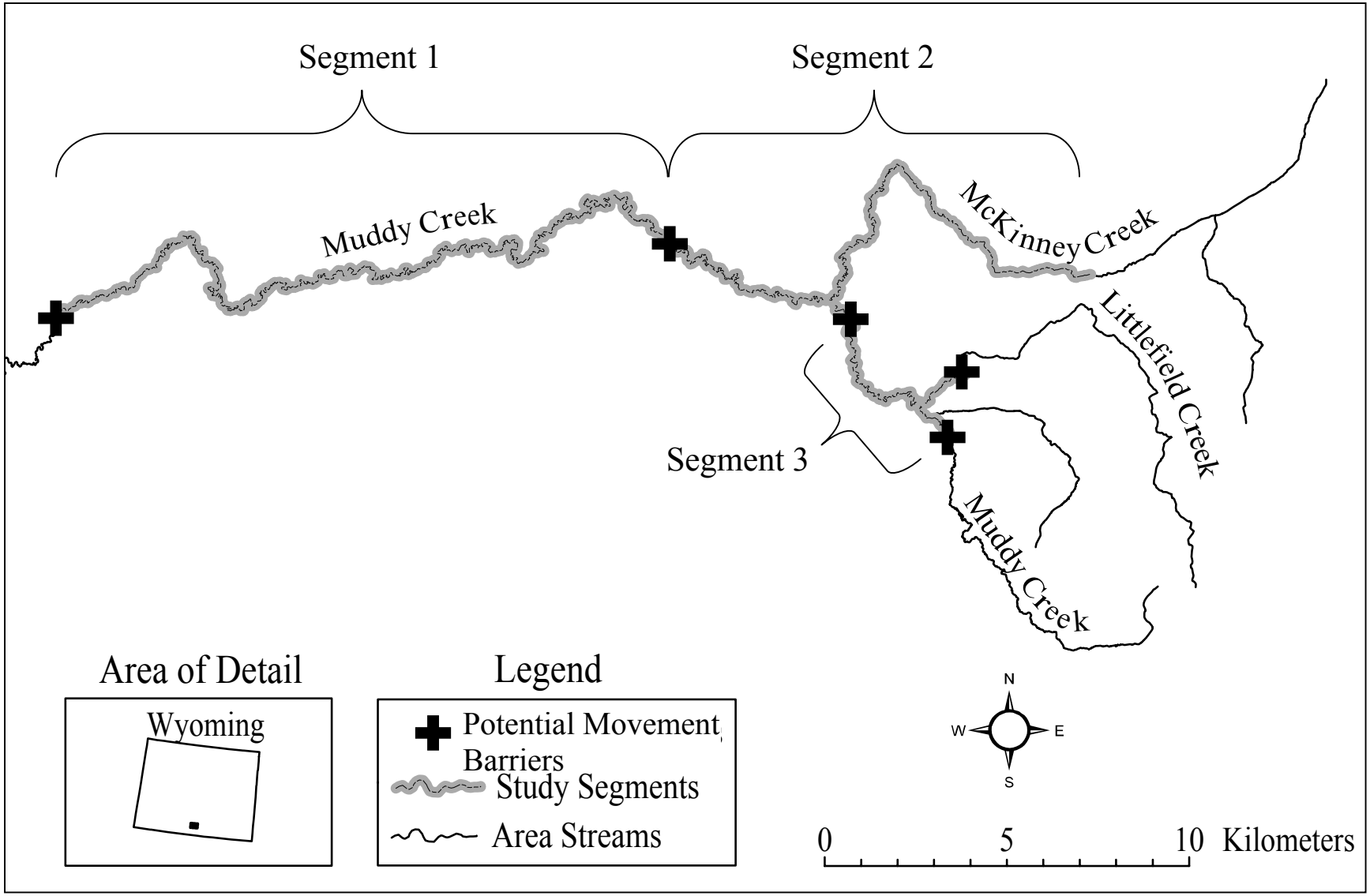


Figure 3.1. Stream segments identified within the upper Muddy Creek study area during the summer and fall of 2003 and 2004 in relation to potential movement barriers.

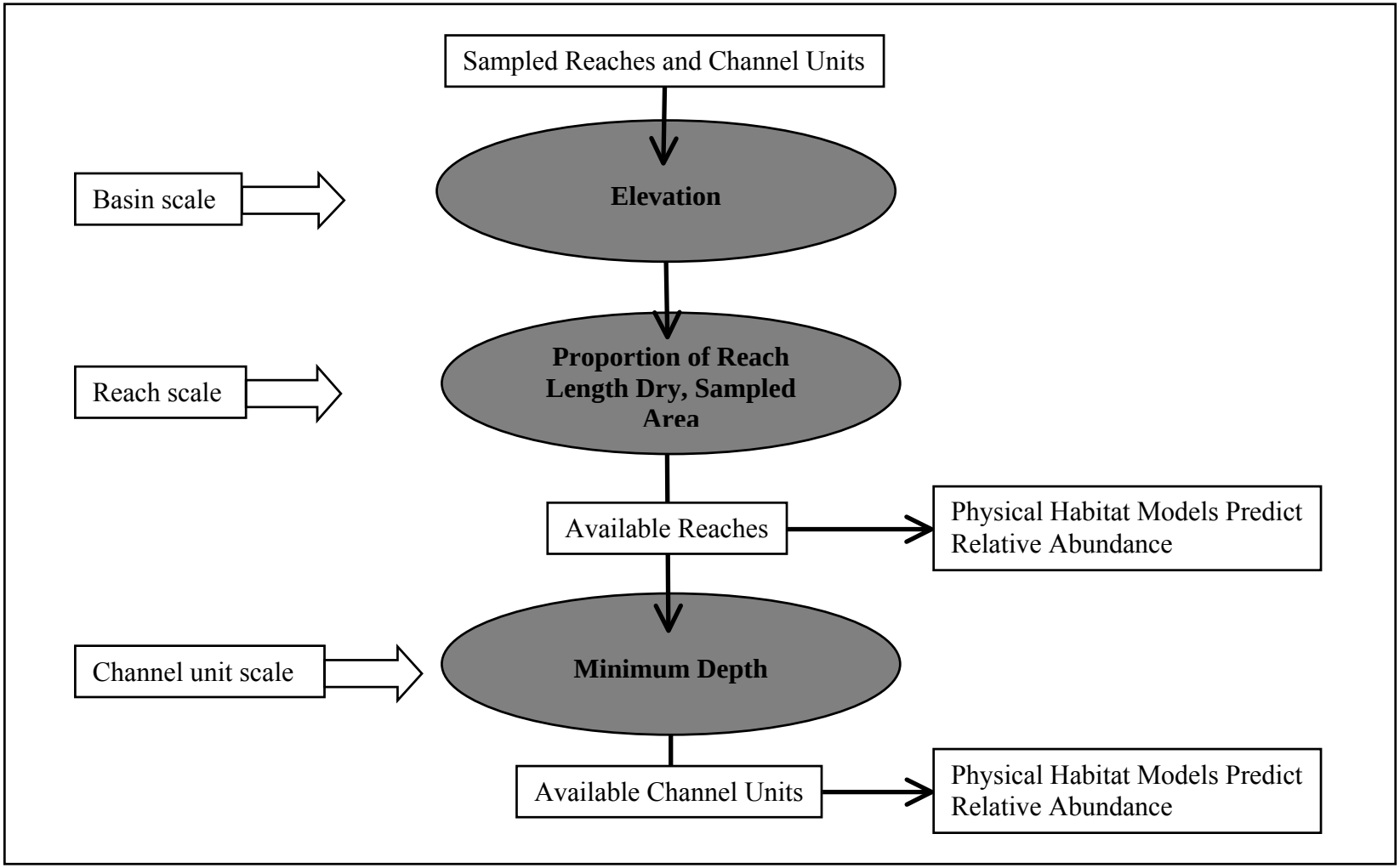


Figure 3.2. Abiotic faunal filters consisting of thresholds of occurrence at three spatial scales used to refine reach- and channel-unit-scale habitat availability prior to modeling the relative abundance of the three target fishes and two length classes as a function of physical habitat parameters.

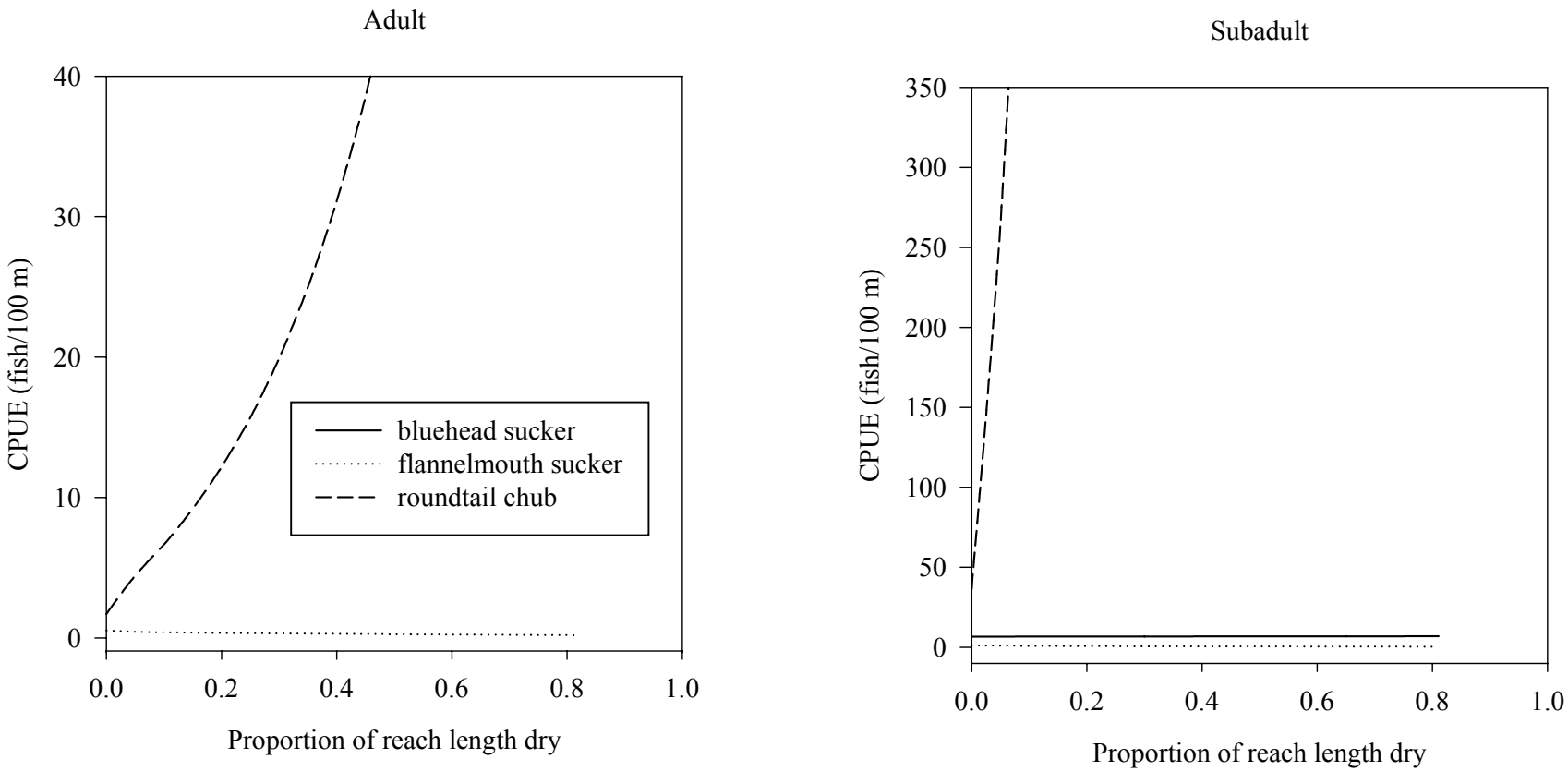


Figure 3.3. Relative abundance of two length groups of three species as a function of the proportion of reach length dry at the reach scale. Plots were generated using the averaged multi-model linear-regression function for both length groups of the three species below threshold values. The plot of adult bluehead sucker was omitted due to this variable's absence from candidate regression models.

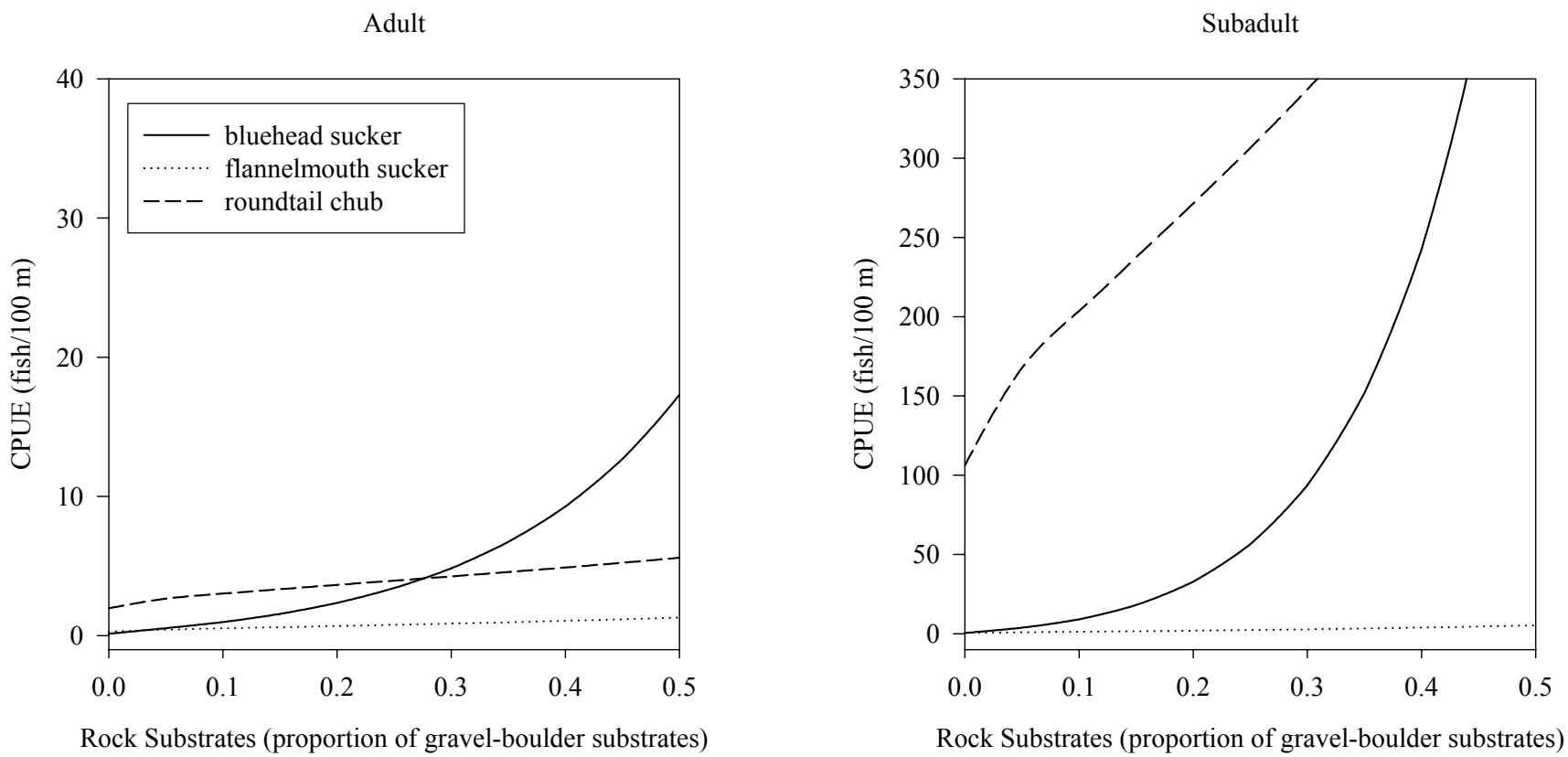


Figure 3.4. Relative abundance of two length groups of three species as a function of the prevalence of rock substrates at the reach scale. Plots were generated using the averaged multi-model linear-regression function for both length groups of the three species.

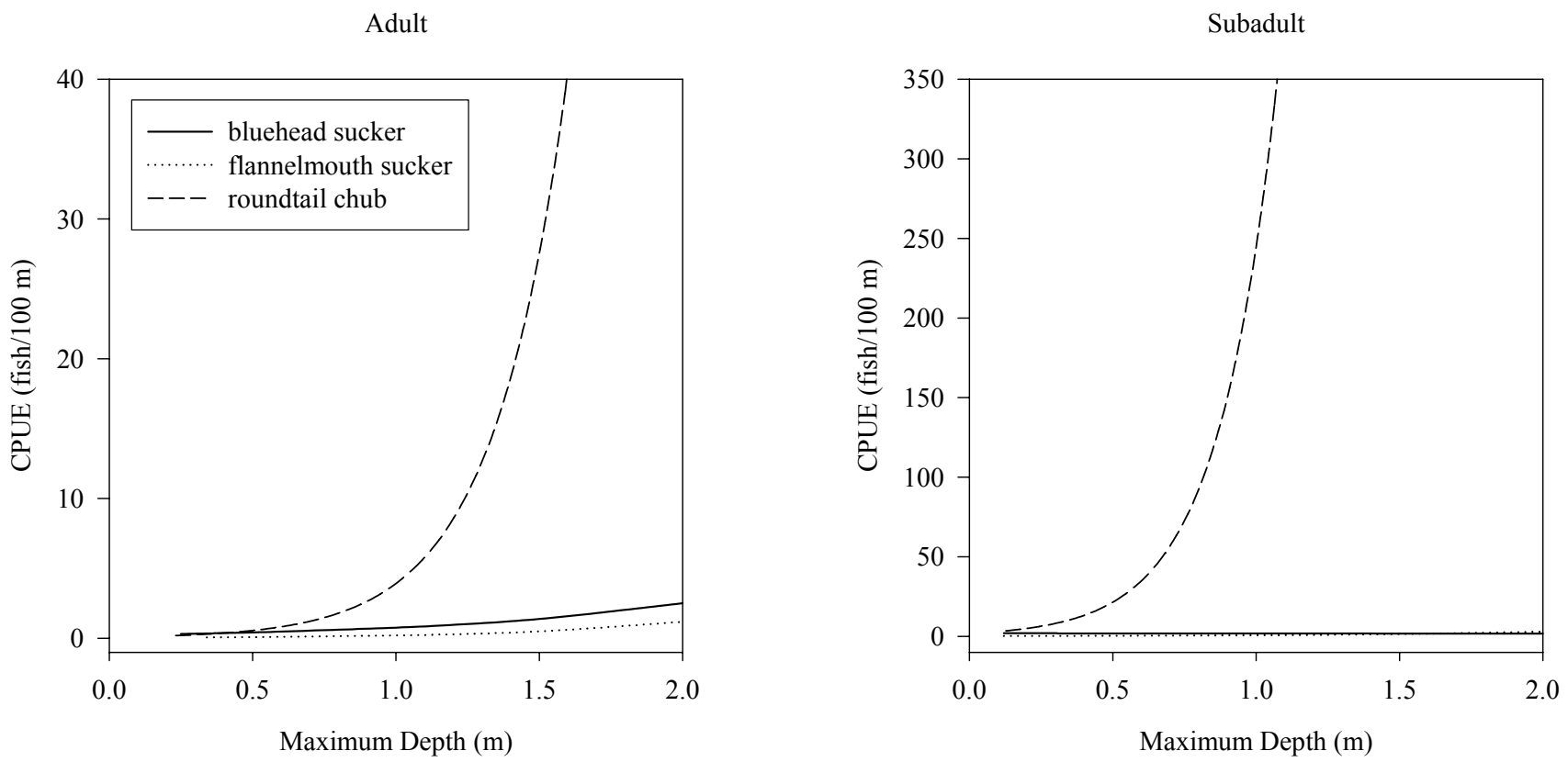


Figure 3.5. Relative abundance of two length groups of three species as a function of maximum channel unit depth. Plots were generated using the averaged multi-model linear-regression function for both length groups of the three species above minimum depth thresholds.

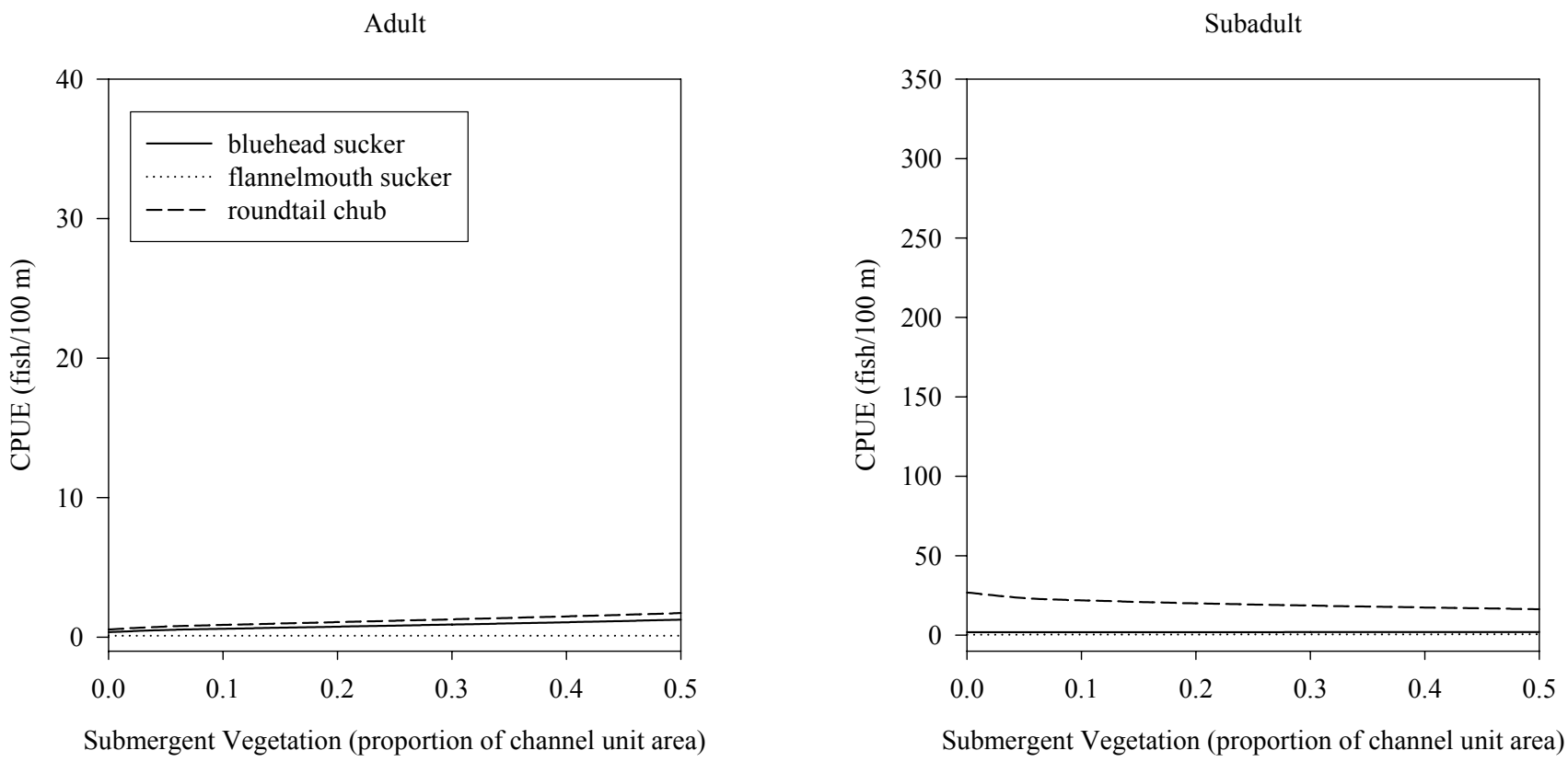


Figure 3.6. Relative abundance of two length groups of three species as a function of the prevalence of submergent vegetation at the channel unit scale. Plots were generated using the averaged multi-model linear-regression function for both length groups of the three species.

Appendix A1. Location of monitoring sites established during the spring and summer of 2003 within the upper Muddy Creek study area. Positions represent post-processed coordinates collected using a Trimble GeoXT receiver. All coordinates are presented in the UTM Zone 13, NAD 27 projection. Horizontal precisions represent 68% probability that the position lies within the specified distance.

| Site number | Type | Location description | Northing | Easting | Horizontal precision (m) | Std dev |
|---|---|---|---|---|---|---|
| 1 | Precipitation gauge | Muddy Creek below headcut stabilization structure | 4592914.04 | 271694.28 | 2.10 | 0.0004 |
| 1 | Gaging station | Muddy Creek below headcut stabilization structure | 4592897.85 | 271722.98 | 3.03 | 0.0000 |
| 2 | Gaging station | Muddy Creek below Olson Draw | 4594266.30 | 279520.91 | 3.24 | 0.0001 |
| 2 | Precipitation gauge | Muddy Creek below Olson Draw | 4594286.61 | 279520.49 | 2.17 | 0.0000 |
| 3 | Gaging station | Muddy Creek below irrigation diversion | 4595119.03 | 288442.17 | 3.51 | 0.0002 |
| 4 | Gaging station | Muddy Creek above irrigation diversion | 4595014.80 | 288445.15 | 2.96 | 0.0001 |
| 5 | Precipitation gauge | Muddy Creek downstream of McKinney Creek | 4593525.89 | 292661.74 | 2.93 | 0.0001 |
| 5 | Gaging station | Muddy Creek downstream of McKinney Creek | 4593543.40 | 292662.28 | 2.94 | 0.0000 |
| 6 | Gaging station | Muddy Creek fish barrier | 4593381.63 | 292978.60 | 2.60 | 0.0001 |
| 7 | Gaging station | Lower McKinney Creek | 4593524.12 | 292916.36 | 3.78 | 0.0002 |
| 8 | Temperature logger | Lower McKinney Creek | 4597224.68 | 294758.55 | 3.52 | 0.0005 |
| 9 | Temperature logger | Upper McKinney Creek | 4594174.67 | 299875.59 | 2.83 | 0.0001 |

Appendix A2. Location of reaches sampled during the summer and fall of 2003 and 2004 within the upper Muddy Creek study area. Positions represent post-processed coordinates collected using a Trimble GeoXT receiver. Coordinates are presented in the UTM Zone 13, NAD 27 projection. Horizontal precisions represent 68% probability that the position lies within the specified distance.

| Location ID | Date sampled (Month/day/year) | Easting | Northing | Horizontal precision (m) | Std dev |
|---|---|---|---|---|---|
| s1r1 | 9/30/2003 | 271669.78 | 4592973.54 | 2.65 | 0.0015 |
| s1r19 | 7/17/2003 | 273872.74 | 4594053.29 | 1.94 | 0.0009 |
| s1r28 | 7/11/2003 | 274693.53 | 4594905.46 | 2.45 | 0.0007 |
| s1r7 | 8/5/2003 | 272376.65 | 4593403.41 | 2.05 | 0.0002 |
| s2r15 | 8/12/2003 | 276064.92 | 4594208.39 | 2.62 | 0.0010 |
| s2r18 | 9/29/2003 | 275976.84 | 4593744.98 | 2.58 | 0.0005 |
| s2r22 | 8/7/2003 | 275951.22 | 4593599.90 | 2.82 | 0.0002 |
| s2r26 | 7/16/2003 | 276192.68 | 4593400.33 | 1.97 | 0.0004 |
| s2r27 | 7/2/2003 | 276312.53 | 4593302.62 | 2.02 | 0.0057 |
| s2r3 | 7/1/2003 | 275401.16 | 4595266.64 | 2.19 | 0.0009 |
| s2r31 | 7/14/2003 | 276633.42 | 4593317.16 | 2.56 | 0.0135 |
| s2r38 | 7/15/2003 | 277322.78 | 4593605.11 | 2.36 | 0.0124 |
| s2r53 | 8/27/2003 | 278554.73 | 4593944.63 | 2.43 | 0.0001 |
| s2r55 | 10/2/2003 | 278776.68 | 4593947.04 | 3.55 | 0.0032 |
| s3r1 | 7/29/2003 | 279696.14 | 4594252.14 | 0.40 | 0.0000 |
| s3r14 | 7/24/2003 | 280841.56 | 4594195.02 | 2.69 | 0.0012 |
| s3r33 | 9/16/2003 | 282571.02 | 4594863.12 | 2.92 | 0.0001 |
| s3r46 | 7/23/2003 | 283883.25 | 4594867.70 | 5.21 | 0.0159 |
| s3r57 | 9/23/2003 | 284322.55 | 4594524.29 | 2.71 | 0.0014 |
| s3r60 | 7/28/2003 | 284689.87 | 4594718.10 | 2.75 | 0.0001 |
| s3r67 | 6/26/2003 | 284905.79 | 4595295.94 | 2.08 | 0.0001 |
| s3r69 | 9/24/2003 | 285016.16 | 4595334.54 | 1.71 | 0.0041 |
| s3r70 | 9/18/2003 | 285281.05 | 4595441.90 | 2.44 | 0.0005 |
| s3r74 | 7/21/2003 | 285504.72 | 4595673.63 | 2.70 | 0.0012 |
| s3r83 | 7/27/2003 | 286340.29 | 4595959.02 | 1.89 | 0.0006 |
| s3r85 | 8/6/2003 | 286149.51 | 4596163.95 | 2.90 | 0.0006 |
| s3r88 | 8/26/2003 | 286418.37 | 4596153.21 | 3.00 | 0.0180 |
| s3r93 | 8/14/2003 | 286859.06 | 4596157.80 | 2.53 | 0.0004 |
| s4r18 | 7/3/2003 | 288292.11 | 4595311.23 | 2.02 | 0.0008 |
| s4r5 | 8/4/2003 | 287244.00 | 4596009.29 | 2.23 | 0.0000 |
| s5r18 | 10/8/2003 | 289695.93 | 4594358.64 | 2.37 | 0.0005 |
| s5r28 | 6/19/2003 | 290332.19 | 4594065.02 | 0.34 | 0.0001 |
| s5r31 | 10/7/2003 | 290430.16 | 4593742.28 | 2.80 | 0.0018 |
| s5r37 | 7/18/2003 | 291306.97 | 4593566.36 | 3.07 | 0.0028 |
| s5r40 | 7/25/2003 | 291475.54 | 4593592.00 | 2.71 | 0.0080 |
| s5r45 | 8/13/2003 | 292034.26 | 4593508.53 | 1.98 | 0.0005 |
| s5r47 | 9/25/2003 | 292383.94 | 4593508.24 | 3.24 | 0.0066 |
| s5r50 | 7/30/2003 | 292590.93 | 4593436.82 | 0.34 | 0.0000 |

Appendix A2, cont.

| Location ID | Date sampled (Month/day/year) | Easting | Northing | Horizontal precision (m) | Std dev |
|---|---|---|---|---|---|
| s6r1e | 7/26/2003 | 292883.89 | 4593478.00 | 3.11 | 0.0005 |
| s6r1l | 10/1/2003 | 292883.89 | 4593478.00 | | 0.0005 |
| s7r1 | 6/17/2003 | 292896.33 | 4593493.82 | | 0.0000 |
| s7r13 | 8/22/2003 | 293314.24 | 4594535.60 | 2.15 | 0.0044 |
| s7r15 | 6/18/2003 | 293191.55 | 4594654.97 | 0.39 | 0.0001 |
| s7r33 | 7/22/2003 | 294248.67 | 4596520.71 | 2.52 | 0.0003 |
| s7r40 | 7/31/2003 | 294614.89 | 4597129.95 | 0.31 | 0.0001 |
| s7r46 | 8/21/2003 | 295415.38 | 4596517.07 | 2.45 | 0.0020 |
| s7r64 | 8/11/2003 | 297248.94 | 4594813.52 | 4.44 | 0.0001 |
| s7r67 | 6/30/2003 | 297396.42 | 4594275.37 | 3.17 | 0.0086 |
| s7r68 | 8/20/2003 | 297569.58 | 4594252.04 | 2.55 | 0.0028 |
| s7r8 | 8/25/2003 | 293271.07 | 4594123.13 | 3.22 | 0.0029 |
| s8r2 | 6/28/2004 | 293024.66 | 4593183.19 | 1.94 | 0.0001 |
| s8r27 | 7/20/2004 | 294091.05 | 4590843.01 | 2.42 | 0.0001 |
| s8r37 | 6/15/2004 | 295016.44 | 4590795.83 | 2.20 | 0.0001 |
| s8r38 | 6/16/2004 | 295173.61 | 4590783.46 | 2.49 | 0.0001 |
| s8r50 | 6/29/2004 | 296133.58 | 4589850.39 | 1.79 | 0.0001 |
| s8r7 | 7/2/2004 | 293566.47 | 4593015.81 | 1.80 | 0.0002 |
| s9r1 | 7/21/2004 | 295586.06 | 4590702.63 | 1.91 | 0.0000 |

Appendix B1. Competing multiple-linear-regression models predicting the abundance of three species and two length classes at the reach scale. Models are ranked in order of decreasing Akaike weight ($w_i$). The model with the largest Akaike weight represents the best model from the set of candidates. Models within 10% of the Akaike weight of this best model are presented as competing models.

| Length class | Rank | Model parameters | $K$ | n | RSS | $AIC_c$ | $\Delta_i$ | $w_i$ | $R^2$ |
|---|---|---|---|---|---|---|---|---|---|
| | | **Bluehead sucker** | | | | | | | |
| Subadult | 1 | sampled area, rock substrates | 4 | 45 | 27.84 | -12.60 | 0.00 | 0.72 | 0.50 |
| | 2 | sampled area, rock substrates, proportion reach length dry | 5 | 45 | 27.84 | -10.07 | 2.54 | 0.20 | 0.50 |
| Adult | 1 | rock substrates | 3 | 44 | 22.96 | -22.01 | 0.00 | 0.91 | 0.30 |
| | | **Flannelmouth sucker** | | | | | | | |
| Subadult | 1 | rock substrates | 3 | 40 | 18.65 | -23.85 | 0.00 | 0.35 | 0.11 |
| | 2 | rock substrates, proportion reach length dry | 4 | 40 | 17.87 | -23.09 | 0.76 | 0.24 | 0.15 |
| | 3 | sampled area, rock substrates | 4 | 40 | 18.04 | -22.71 | 1.14 | 0.20 | 0.14 |
| | 4 | sampled area, rock substrates, proportion reach length dry | 5 | 40 | 17.73 | -20.78 | 3.07 | 0.07 | 0.15 |
| | 5 | proportion reach length dry | 3 | 40 | 20.32 | -20.43 | 3.42 | 0.06 | 0.03 |
| | 6 | sampled area | 3 | 40 | 20.38 | -20.30 | 3.54 | 0.06 | 0.03 |
| Adult | 1 | sampled area | 3 | 40 | 15.84 | -30.40 | 0.00 | 0.30 | 0.12 |
| | 2 | sampled area, rock substrates | 4 | 40 | 14.94 | -30.26 | 0.14 | 0.28 | 0.17 |
| | 3 | proportion reach length dry | 3 | 40 | 16.77 | -28.10 | 2.30 | 0.09 | 0.07 |
| | 4 | sampled area, proportion reach length dry | 4 | 40 | 15.77 | -28.09 | 2.31 | 0.09 | 0.13 |
| | 5 | rock substrates, proportion reach length dry | 4 | 40 | 15.80 | -28.01 | 2.39 | 0.09 | 0.12 |
| | 6 | sampled area, rock substrates, proportion reach length dry | 5 | 40 | 14.86 | -27.84 | 2.55 | 0.08 | 0.18 |
| | 7 | rock substrates | 3 | 40 | 17.07 | -27.41 | 2.99 | 0.07 | 0.05 |
| | | **Roundtail chub** | | | | | | | |
| Subadult | 1 | sampled area, proportion reach length dry | 4 | 51 | 60.93 | 17.94 | 0.00 | 0.68 | 0.57 |
| | 2 | sampled area, rock substrates, proportion reach length dry | 5 | 51 | 59.77 | 19.43 | 1.49 | 0.32 | 0.57 |
| Adult | 1 | sampled area, proportion reach length dry | 4 | 48 | 32.88 | -9.23 | 0.00 | 0.67 | 0.27 |
| | 2 | sampled area, rock substrates, proportion reach length dry | 5 | 48 | 32.40 | -7.44 | 1.79 | 0.28 | 0.28 |

Appendix B2. Competing multiple-linear-regression models predicting the abundance of three species and two length classes among all segments at the channel unit scale. Models are ranked in order of decreasing Akaike weight ($w_i$). The model with the largest Akaike weight represents the best model from the set of candidates. Models within 10% of the Akaike weight of this best model are presented as competing models.

| Length class | Rank | Model parameters | $K$ | n | RSS | $AIC_c$ | $\Delta_i$ | $w_i$ | $R^2$ |
|---|---|---|---|---|---|---|---|---|---|
| | | **Bluehead sucker** | | | | | | | |
| Subadult | 1 | channel unit type | 3 | 405 | 881.01 | 320.82 | 0.00 | 0.54 | 0.04 |
| | 2 | channel unit type, maximum depth | 4 | 405 | 881.00 | 322.86 | 2.04 | 0.19 | 0.04 |
| | 3 | channel unit type, submergent vegetation | 4 | 405 | 881.01 | 322.86 | 2.04 | 0.19 | 0.04 |
| | 4 | channel unit type, maximum depth, submergent vegetation | 5 | 405 | 881.00 | 324.91 | 4.09 | 0.07 | 0.04 |
| Adult | 1 | channel unit type, maximum depth, submergent vegetation | 5 | 344 | 300.69 | -36.11 | 0.00 | 0.62 | 0.09 |
| | 2 | channel unit type, maximum depth | 4 | 344 | 304.65 | -33.67 | 2.44 | 0.18 | 0.07 |
| | 3 | channel unit type ,submergent vegetation | 4 | 344 | 305.44 | -32.78 | 3.32 | 0.12 | 0.07 |
| | | **Flannelmouth sucker** | | | | | | | |
| Subadult | 1 | channel unit type, maximum depth | 4 | 332 | 244.41 | -93.57 | 0.00 | 0.50 | 0.09 |
| | 2 | channel unit type, maximum depth, submergent vegetation | 5 | 332 | 243.11 | -93.27 | 0.29 | 0.43 | 0.10 |
| Adult | 1 | maximum, depth | 3 | 254 | 146.36 | -133.93 | 0.00 | 0.43 | 0.06 |
| | 2 | channel unit type, maximum depth | 4 | 254 | 145.56 | -133.25 | 0.69 | 0.31 | 0.06 |
| | 3 | maximum depth, submergent vegetation | 4 | 254 | 146.36 | -131.87 | 2.06 | 0.15 | 0.06 |
| | 4 | channel unit type, maximum depth, submergent vegetation | 5 | 254 | 145.56 | -131.18 | 2.75 | 0.11 | 0.06 |
| | | **Roundtail chub** | | | | | | | |
| Subadult | 1 | channel unit type, maximum depth | 4 | 416 | 1404.18 | 514.17 | 0.00 | 0.71 | 0.15 |
| | 2 | channel unit type, maximum depth, submergent vegetation | 5 | 416 | 1403.36 | 515.98 | 1.81 | 0.29 | 0.15 |
| Adult | 1 | channel unit type, maximum depth, submergent vegetation | 5 | 379 | 429.08 | 57.19 | 0.00 | 0.64 | 0.21 |
| | 2 | channel unit type, maximum depth | 4 | 379 | 432.99 | 58.58 | 1.38 | 0.32 | 0.20 |

Appendix B3. Competing multiple-linear-regression models predicting the abundance of three species and two length classes within Segment 1 at the channel unit scale. Models are ranked in order of decreasing Akaike weight ($w_i$). The model with the largest Akaike weight represents the best model from the set of candidates. Models within 10% of the Akaike weight of this best model are presented as competing models.

| Length class | Rank | Model parameters | $K$ | n | RSS | $AIC_c$ | $\Delta_i$ | $w_i$ | $R^2$ |
|---|---|---|---|---|---|---|---|---|---|
| | | **Bluehead sucker** | | | | | | | |
| Subadult | 1 | channel unit type | 3 | 196 | 234.16 | 40.99 | 0.00 | 0.40 | 0.08 |
| | 2 | channel unit type, submergent vegetation | 4 | 196 | 232.37 | 41.57 | 0.57 | 0.30 | 0.09 |
| | 3 | channel unit type, maximum depth | 4 | 196 | 233.81 | 42.78 | 1.79 | 0.16 | 0.08 |
| | 4 | channel unit type, maximum depth, submergent vegetation | 5 | 196 | 231.79 | 43.18 | 2.19 | 0.13 | 0.09 |
| Adult | 1 | channel unit type, maximum depth, submergent vegetation | 5 | 169 | 55.19 | -178.75 | 0.00 | 0.43 | 0.11 |
| | 2 | maximum depth, submergent vegetation | 4 | 169 | 56.37 | -177.31 | 1.44 | 0.21 | 0.10 |
| | 3 | channel unit type, maximum depth | 4 | 169 | 56.56 | -176.73 | 2.02 | 0.15 | 0.09 |
| | 4 | channel unit type, submergent vegetation | 4 | 169 | 56.89 | -175.77 | 2.98 | 0.10 | 0.09 |
| | 5 | maximum depth | 3 | 169 | 57.91 | -174.85 | 3.90 | 0.06 | 0.07 |
| | 6 | channel unit type | 3 | 169 | 57.96 | -174.72 | 4.04 | 0.06 | 0.07 |
| | | **Flannelmouth sucker** | | | | | | | |
| Subadult | 1 | maximum depth | 3 | 190 | 100.42 | -115.03 | 0.00 | 0.41 | 0.10 |
| | 2 | maximum depth, submergent vegetation | 4 | 190 | 99.66 | -114.39 | 0.65 | 0.30 | 0.11 |
| | 3 | channel unit type, maximum depth | 4 | 190 | 100.25 | -113.26 | 1.77 | 0.17 | 0.10 |
| | 4 | channel unit type, maximum depth, submergent vegetation | 5 | 190 | 99.55 | -112.49 | 2.55 | 0.12 | 0.11 |
| Adult | 1 | channel unit type, maximum depth | 4 | 152 | 52.11 | -154.45 | 0.00 | 0.65 | 0.15 |
| | 2 | channel unit type, maximum depth, submergent vegetation | 5 | 152 | 51.86 | -153.04 | 1.41 | 0.32 | 0.15 |
| | | **Roundtail chub** | | | | | | | |
| Subadult | 1 | channel unit type, maximum depth, submergent vegetation | 5 | 207 | 558.16 | 215.63 | 0.00 | 0.64 | 0.23 |
| | 2 | channel unit type, maximum depth | 4 | 207 | 567.22 | 216.86 | 1.23 | 0.34 | 0.22 |
| Adult | 1 | channel unit type, maximum depth, submergent vegetation | 5 | 182 | 207.07 | 33.83 | 0.00 | 0.32 | 0.29 |
| | 2 | channel unit type, maximum depth | 4 | 182 | 209.52 | 33.86 | 0.03 | 0.32 | 0.28 |
| | 3 | maximum depth, submergent vegetation | 4 | 182 | 210.70 | 34.88 | 1.05 | 0.19 | 0.28 |
| | 4 | maximum depth | 3 | 182 | 213.51 | 35.20 | 1.37 | 0.16 | 0.27 |

Appendix B4. Competing multiple-linear-regression models predicting the abundance of three species and two length classes within Segment 2 at the channel unit scale. Models are ranked in order of decreasing Akaike weight ($w_i$). The model with the largest Akaike weight represents the best model from the set of candidates. Models within 10% of the Akaike weight of this best model are presented as competing models.

| Length class | Rank | Model parameters | $K$ | n | RSS | $AIC_c$ | $\Delta_i$ | $w_i$ | $R^2$ |
|---|---|---|---|---|---|---|---|---|---|
| | | **Bluehead sucker** | | | | | | | |
| Subadult | 1 | channel unit type | 3 | 142 | 386.38 | 148.31 | 0.00 | 0.42 | 0.02 |
| | 2 | channel unit type, maximum depth | 4 | 142 | 386.30 | 150.40 | 2.09 | 0.15 | 0.02 |
| | 3 | channel unit type, submergent vegetation | 4 | 142 | 386.37 | 150.43 | 2.12 | 0.14 | 0.02 |
| | 4 | maximum depth | 3 | 142 | 392.93 | 150.70 | 2.39 | 0.13 | 0.01 |
| | 5 | submergent vegetation | 3 | 142 | 395.92 | 151.78 | 3.46 | 0.07 | 0.00 |
| | 6 | channel unit type, maximum depth, submergent vegetation | 5 | 142 | 386.28 | 152.55 | 4.23 | 0.05 | 0.02 |
| | 7 | maximum depth submergent vegetation | 4 | 142 | 392.77 | 152.76 | 4.45 | 0.04 | 0.01 |
| Adult | 1 | channel unit type, maximum depth | 4 | 129 | 174.64 | 47.40 | 0.00 | 0.33 | 0.13 |
| | 2 | channel unit type, maximum depth, submergent vegetation | 5 | 129 | 172.17 | 47.72 | 0.32 | 0.28 | 0.14 |
| | 3 | channel unit type, submergent vegetation | 4 | 129 | 177.65 | 49.60 | 2.20 | 0.11 | 0.11 |
| | 4 | channel unit type | 3 | 129 | 180.83 | 49.76 | 2.36 | 0.10 | 0.10 |
| | 5 | maximum depth | 3 | 129 | 180.98 | 49.87 | 2.47 | 0.10 | 0.10 |
| | 6 | maximum depth, submergent vegetation | 4 | 129 | 179.00 | 50.58 | 3.18 | 0.07 | 0.11 |
| | | **Flannelmouth sucker** | | | | | | | |
| Subadult | 1 | channel unit type, submergent vegetation | 4 | 115 | 112.85 | 6.19 | 0.00 | 0.56 | 0.15 |
| | 2 | channel unit type, maximum depth, submergent vegetation | 5 | 115 | 112.25 | 7.77 | 1.57 | 0.26 | 0.15 |
| | 3 | channel unit type | 3 | 115 | 118.39 | 9.55 | 3.36 | 0.10 | 0.11 |
| | 4 | channel unit type, maximum depth | 4 | 115 | 117.15 | 10.49 | 4.30 | 0.07 | 0.12 |
| Adult | 1 | maximum depth | 3 | 87 | 84.22 | 3.46 | 0.00 | 0.33 | 0.05 |
| | 2 | channel unit type, maximum depth | 4 | 87 | 82.18 | 3.53 | 0.07 | 0.32 | 0.07 |
| | 3 | maximum depth, submergent vegetation | 4 | 87 | 84.08 | 5.52 | 2.05 | 0.12 | 0.05 |
| | 4 | channel unit type, maximum depth, submergent vegetation | 5 | 87 | 82.04 | 5.63 | 2.17 | 0.11 | 0.07 |
| | 5 | channel unit type | 3 | 87 | 87.82 | 7.10 | 3.64 | 0.05 | 0.01 |
| | 6 | submergent vegetation | 3 | 87 | 88.21 | 7.49 | 4.02 | 0.04 | 0.00 |
| | | **Roundtail chub** | | | | | | | |
| Subadult | 1 | channel unit type, maximum depth | 4 | 142 | 456.74 | 174.19 | 0.00 | 0.30 | 0.13 |
| | 2 | channel unit type, maximum depth, submergent vegetation | 5 | 142 | 450.53 | 174.39 | 0.20 | 0.27 | 0.14 |
| | 3 | channel unit type, submergent vegetation | 4 | 142 | 458.53 | 174.74 | 0.55 | 0.23 | 0.13 |
| | 4 | channel unit type | 3 | 142 | 467.09 | 175.25 | 1.06 | 0.18 | 0.11 |
| Adult | 1 | channel unit type, maximum depth, submergent vegetation | 5 | 133 | 154.76 | 30.63 | 0.00 | 0.60 | 0.18 |
| | 2 | channel unit type, submergent vegetation | 4 | 133 | 159.22 | 32.25 | 1.62 | 0.27 | 0.16 |

Appendix B5. Competing multiple-linear-regression models predicting the abundance of three species and two length classes within Segment 3 at the channel unit scale. Models are ranked in order of decreasing Akaike weight ($w_i$). The model with the largest Akaike weight represents the best model from the set of candidates. Models within 10% of the Akaike weight of this best model are presented as competing models. Insufficient numbers of flannelmouth suckers were detected (n=1) to compute linear-regression models within this segment.

| Length class | Rank | Model parameters | $K$ | n | RSS | $AIC_c$ | $\Delta_i$ | $w_i$ | $R^2$ |
|---|---|---|---|---|---|---|---|---|---|
| | | **Bluehead sucker** | | | | | | | |
| Subadult | 1 | maximum depth | 3 | 87 | 84.22 | 3.46 | 0.00 | 0.33 | 0.05 |
| | 2 | channel unit type, maximum depth | 4 | 87 | 82.18 | 3.53 | 0.07 | 0.32 | 0.07 |
| | 3 | maximum depth, submergent vegetation | 4 | 87 | 84.08 | 5.52 | 2.05 | 0.12 | 0.05 |
| | 4 | channel unit type, maximum depth, submergent vegetation | 5 | 87 | 82.04 | 5.63 | 2.17 | 0.11 | 0.07 |
| | 5 | channel unit type | 3 | 87 | 87.82 | 7.10 | 3.64 | 0.05 | 0.01 |
| | 6 | submergent vegetation | 3 | 87 | 88.21 | 7.49 | 4.02 | 0.04 | 0.00 |
| Adult | 1 | channel unit type | 3 | 46 | 35.86 | -4.88 | 0.00 | 0.26 | 0.02 |
| | 2 | submergent vegetation | 3 | 46 | 35.98 | -4.74 | 0.15 | 0.24 | 0.02 |
| | 3 | maximum depth | 3 | 46 | 36.49 | -4.09 | 0.79 | 0.17 | 0.00 |
| | 4 | channel unit type, submergent vegetation | 4 | 46 | 35.25 | -3.27 | 1.62 | 0.11 | 0.04 |
| | 5 | maximum depth, submergent vegetation | 4 | 46 | 35.43 | -3.04 | 1.84 | 0.10 | 0.03 |
| | 6 | channel unit type, maximum depth | 4 | 46 | 35.86 | -2.48 | 2.40 | 0.08 | 0.02 |
| | 7 | channel unit type, maximum depth, submergent vegetation | 5 | 46 | 35.14 | -0.89 | 3.99 | 0.04 | 0.04 |
| | | **Roundtail chub** | | | | | | | |
| Subadult | 1 | channel unit type, maximum depth | 4 | 67 | 6.31 | -149.62 | 0.00 | 0.21 | 0.07 |
| | 2 | submergent vegetation | 3 | 67 | 6.55 | -149.41 | 0.21 | 0.19 | 0.04 |
| | 3 | channel unit type | 3 | 67 | 6.55 | -149.39 | 0.23 | 0.19 | 0.04 |
| | 4 | channel unit type, submergent vegetation | 4 | 67 | 6.34 | -149.36 | 0.26 | 0.18 | 0.07 |
| | 5 | channel unit type, maximum depth, submergent vegetation | 5 | 67 | 6.22 | -148.26 | 1.37 | 0.10 | 0.08 |
| | 6 | maximum depth | 3 | 67 | 6.74 | -147.48 | 2.14 | 0.07 | 0.01 |
| | 7 | maximum depth, submergent vegetation | 4 | 67 | 6.55 | -147.19 | 2.43 | 0.06 | 0.04 |
| Adult | 1 | maximum depth | 3 | 64 | 48.19 | -11.76 | 0.00 | 0.36 | 0.09 |
| | 2 | channel unit type, maximum depth | 4 | 64 | 47.42 | -10.51 | 1.25 | 0.19 | 0.11 |
| | 3 | maximum depth, submergent vegetation | 4 | 64 | 47.73 | -10.09 | 1.67 | 0.15 | 0.10 |
| | 4 | channel unit type | 3 | 64 | 49.46 | -10.09 | 1.67 | 0.15 | 0.07 |
| | 5 | channel unit type, maximum depth, submergent vegetation | 5 | 64 | 46.96 | -8.77 | 2.99 | 0.08 | 0.11 |
| | 6 | channel unit type, submergent vegetation | 4 | 64 | 49.45 | -7.83 | 3.94 | 0.05 | 0.07 |