FISH BARRIERS AND SMALL PLAINS FISHES: FISHWAY DESIGN RECOMMENDATIONS AND THE IMPACT OF EXISTING INSTREAM STRUCTURES

FINAL PROJECT REPORT

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The Front Range region of Colorado forms a transition zone between the Rocky Mountains and the western Great Plains. Urban and agricultural land uses in this area have changed its streams and led to the decline of many small-bodied plains fishes. One of the most detrimental changes is the fragmentation of stream systems by dams, weirs, and diversions. Fragmentation inhibits spawning migrations, ranging in search of sporadically available resources, and recolonization of empty habitats. Because removal of these structures is not feasible, it may be practical to reverse fragmentation by installing fishways or fish ladders that restore access to habitats upstream of fish barriers. As a result, distinct needs exist for fishway design specifications for small fishes and for monitoring protocols that determine whether existing instream structures are barriers to migration. We used a combination of laboratory studies and field studies to discover how to lessen the impact of instream structures on small-bodied plains fishes.

Effective fishway designs take a variety of variables into consideration, including hydraulic conditions, site topography, and, most importantly, the swimming and jumping ability and behavior of the target fish species. Therefore, we quantified the swimming and jumping ability of three representative plains fishes, the brassy minnow (*Hybognathus hankinsoni*), the Arkansas darter (*Etheostoma cragini*), and the common shiner (*Luxilus cornutus*) at 10, 17.5, and 25°C. Median endurance ranged from 33 minutes at 4 cm/s to 2.13 minutes at 32 cm/s for brassy minnows held at 10°C and from 197 minutes at 16 cm/s to 1.23 minutes at 64 cm/s for brassy minnows held at 17 and 25°C. Median endurance ranged from 200 minutes at 4 cm/s to 0.21 minutes at 32 cm/s for Arkansas darters at all temperatures. Median endurance ranged from 200 minutes at 16 cm/s to 0.5 minutes at 64 cm/s for common shiners at both temperatures. Based on our results, we recommend that current velocities between velocity refuges in fishways not exceed 64 cm/s for brassy minnows and common shiners and 32 cm/s for Arkansas darters. At 32 cm/s, maximum fishway length between velocity refuges should be 0.2 m or less to accommodate Arkansas darters, the weakest swimmer of the three species.

Jumping experiments showed that the presence of even a low vertical barrier (5 cm) dramatically reduced the probability of upstream movement of all three species. Brassy minnows did not jump at 10°C, while those held at 17.5°C jumped a maximum of 10 cm, and those held at 25°C jumped a maximum of 15 cm. Arkansas darters did not jump at any temperature. Common shiners held at 10°C jumped a maximum of 5 cm, and those held at 17.5°C jumped a maximum of 10 cm. Behavioral observations also indicated that even a submerged weir probably presents problems for bottom-oriented fish. Therefore, we recommend that fishways requiring jumping and those with weir-type structures be avoided in watersheds containing these species. Rock ramps may be appropriate for plains and transition-zone streams because the structurally and hydraulically diverse conditions in this type of fishway should allow upstream movement of numerous guilds of small fishes.

One of the main disadvantages of rock ramps is that they are very difficult to monitor because they span the width of the stream and lack confined entrances and exits. We monitored fish movements across two structures: a rock ramp fishway on the St. Vrain River near Longmont,

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Colorado, and a grade-control structure in Spring Creek in Fort Collins, Colorado. We electrofished both sites for one year and individually marked fish with visual implant elastomer (VIE) tags. Recapture rates were too low (17 out of 1340 individuals) in the St. Vrain River to allow quantitative estimates of movement rates. We measured water velocities on the ramp and concluded that it probably allows bidirectional movement of a variety of fishes despite some limitations. The grade-control structure in Spring Creek prevented upstream movement by most fish, and likely has negative effects on the fish community. We were able to successfully track individual fish movement across the structure in Spring Creek, a small stream (width = approximately 4 m). However, this method was not as successful in the St. Vrain River due to its large size.

Many fish populations exist as metapopulations, where the probability of local extinction is a function of patch size and the degree of isolation between patches. Fragmentation of stream systems effectively increases the probability of extinction by isolating small subpopulations and preventing ranging behavior, avoidance of stochastic events, and rescue by recolonization. Identification and modification of problematic instream structures would result in more robust fish communities by reducing fragmentation.

ACKNOWLEDGEMENTS

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SMALL-BODIED PLAINS FISHES AND URBAN AND AGRICULTURAL LAND USE: ISSUES AND EXPERIMENTAL APPROACH

Introduction

The Front Range of Colorado forms a transition zone between the Rocky Mountains and the rolling grasslands of the Great Plains. The Arkansas and South Platte Rivers and some of their smaller tributaries flow through these foothills (Fausch and Bestgen 1997), which are part of an extensive urban corridor. The Colorado Front Range is characterized by warm summers and cold winters; January temperatures average –3.2°C, and average July temperatures are 20.8°C (Hansen et al. 1978). Typical annual precipitation ranges from 100 cm/year in the mountains to 40 cm/year on the plains (Strange et al. 1999). The stream hydrograph in the transition zone is characterized by a spring snowmelt peak (May-July) followed by a low-flow period (August-April) punctuated by sharp peaks caused by localized rainstorms (Strange et al. 1999) (**Figure 1.1**). Streams on Colorado's eastern plains have a different hydrograph. Many of the streams in this area lack mountain headwaters. Instead, stream base flow is supplied by groundwater discharge, and additional input comes in the form of high-intensity, short-duration thunderstorms. The annual flow peak is smaller in magnitude but longer in duration in the spring and summer than in transition-zone streams. A similar low-flow period in late summer and early autumn occurs when high temperatures and arid conditions cause evaporation rates to exceed precipitation and groundwater discharge (**Figure 1.2**). This precipitation regime results in a large intra-annual and interannual variation in flow. Short and long-term variation in flow is much greater in plains streams than in transition-zone streams. Because flows in most Front Range and many eastern plains streams fluctuate so much, spring flows are usually diverted for irrigation or stored for use during the dry season (Wohl 2001). As a result, managed rivers have a runoff peak that is smaller in magnitude and longer in duration than flow peaks in natural systems. The subsequent release of this stored water for irrigation during the dry season causes higher base flows and many artificial flow peaks that mirror irrigation demand (Strange et al. 1999).

Despite intensive water management, these aquatic systems vary markedly within and between years in terms of water chemistry (Fausch and Bestgen 1997) and flow regime (Wohl 2001), producing a unique set of environmental challenges for fish. For example, on April 17, 2006, discharge in the Republican River was 56 cfs, but over the 71 years of record, flows on this date ranged from 12 to 102 cfs (www.usgs.com/nwis). Fishes native to transition-zone and plains streams have developed physiological and behavioral adaptations to allow them to persist in these systems. Most of these fishes are characterized by small body sizes (ca. < 100 mm total length), short lifespans (ca. 1 – 2 years), and early sexual maturity (Fausch and Bestgen 1997). Most species studied to date are capable of tolerating harsh environmental conditions such as hypoxia,

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high water temperatures, and sudden fluctuations in flow (Bramblett and Fausch 1991; Fausch and Bestgen 1997; Rabeni 1996; Scheurer et al. 2003b). Because their environment is unpredictable, and because resources do not have a uniform spatial or temporal distribution, plains and transition zone fishes migrate to locate spawning habitats, foraging opportunities, and refuge habitats (Fausch et al. 2002; Labbe and Fausch 2000). This ability to tolerate diverse environmental conditions does not, however, guarantee that the fishes can cope with the physical and functional changes caused by urbanization and water development. A minority of these species (e.g., the fathead minnow, *Pimephales promelas*) thrives in highly modified and disturbed systems (Page and Burr 1991), but populations of many native fishes have declined (Fausch and Bestgen 1997).

Soon after their arrival in this region, Euro-Americans began modifying rivers. Alterations of plains streams in the South Platte River basin began with hand-dug irrigation ditches in the 1840's. These were followed by the building of large canals in the 1860's and reservoir construction, groundwater pumping, and trans-basin diversions from the Western Slope of the Rocky Mountains (1885-1930) (Eschner et al. 1983). Streams on Colorado's Front Range have been dramatically modified as a result of agricultural and urban land use. For example, most of the water in the South Platte Basin is diverted for agricultural, household, or industrial use and then returned to streams (Strange et al. 1999) for transport to the next impoundment or diversion. The resultant changes to the stream that have the greatest effects on native fishes are water diversion, altered hydrologic cycles, pollution from excess nutrient inputs, and habitat fragmentation (Fausch and Bestgen 1997).

One main reason that fishes native to transition zone and plains streams may be in decline is that instream structures such as diversion dams and culverts act as barriers to upstream and downstream movements. In effect, these structures sever the ecological connectivity of the stream systems. Connectivity, or the availability of continuous, unfragmented stream habitat, is essential to the persistence of many fish species (Fausch et al. 2002). The focus of this research project was the development and refinement of tools to reestablish connectivity to plains streams on the Colorado Front Range. The design of effective fish ladders involves a number of essential elements such as the identification or selection of target species, the integration of design with fish swimming and jumping ability, and the necessity of monitoring after installation. We provide recommendations for fishway design for small-bodied fishes (based on data on their abilities) in **Chapter 2** of this work, and we evaluate the ability to monitor fish movement across existing instream structures (using electrofishing and visual implant elastomer marks) in **Chapter 3**.

Fishway design

Though small-bodied fish could move across most instream obstacles if they were fitted with a fishway or fish passage device, little attention has been given to the challenge of designing effective fish passage structures for plains stream species. Most fishways have been designed with large, strong swimming, jumping fishes, such as salmonids, in mind. Though some fishways have been built and evaluated for non-salmonid species such as lake sturgeon (*Acipenser fulvescens*) (Peake et al. 1997), yellow perch (*Perca fluviatilis*) (Katopodis et al. 1991), and minnows (e.g., chub *Leuciscus cephalus*, roach *Rutilus rutilus*) (Lucas 2000), these structures are still biased toward large fishes. Swimming performance in fishes depends on body length (Northcote 1998),

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so small-bodied plains fishes may not be able to negotiate the water velocities characteristic of fishways designed for larger species.

The main types of fishway are the pool-and-weir, vertical slot, Denil, and rock ramp (Clay 1995). The pool-and-weir fishway (**Figure 1.3**) consists of an ascending series of plunge pools and small waterfalls. Fish negotiate this structure by jumping from one plunge pool to the next. Some pool-and-weir fishways have an orifice in each weir to accommodate non-jumping species, but the position of the orifice in the weir can exclude certain species based on their preferred position in the water column. A vertical slot fishway is an inclined ramp with a series of weirs to control water velocity (**Figure 1.4**). Each weir has a slot that runs from the surface to the bottom of the water column, and the slots often alternate to further reduce water velocity. A Denil fishway is an inclined ramp fitted with protruding vanes that provide velocity refuges for fishes swimming upstream (**Figure 1.5**). A Denil fishway in the Grand River, Ontario, Canada allowed upstream movement of fishes ≥ 60 mm long under certain flow conditions (Bunt et al. 2001). The rock ramp fishway is essentially a steep, artificial riffle (**Figure 1.6**). These structures are built with a combination of concrete and boulders or natural rock. These fishways are common in Australia, for example, where they appear to be effective in allowing upstream movement of both large and small fishes (Harris et al. 1998).

Fish ladders differ in selectivity for jumping ability and swimming speed. For example, pool-and-weir fishways were designed with jumping fishes in mind, whereas vertical slot fishways allow fish to use any portion of the water column, provided that they can negotiate the high velocity current at the slot. Too often, non-salmonid fishes cannot use existing fishways because these structures were built without consideration of their swimming and jumping abilities (Mallen-Cooper 1999). A perfect fish ladder would facilitate passage past an instream obstacle for fish of all sizes and species. However, since this is not always feasible, fishways are typically designed for a few important "target species". The difficulty, therefore, lies in choosing the appropriate species. Presumably, if a fishway is designed with a small-bodied species in mind, other larger species will also benefit from the structure. Because each type of fishway is designed using data or assumptions about the swimming and jumping ability of a migrating fish, accurate data must be used. We address this issue in **Chapter 2** of this report. There, we describe a laboratory study on the swimming and jumping ability of three threatened Colorado plains fishes, the brassy minnow (*Hybognathus hankinsoni*), the Arkansas darter (*Etheostoma cragini*), and the common shiner (*Luxilus cornutus*).

Monitoring and assessment of existing fishways

Few fishways have been installed on Colorado's Front Range, and even fewer have been monitored for effectiveness. Furthermore, no studies have been conducted to determine the effects of instream structures on the native fish communities in urban streams in this region. Monitoring fish movement across existing fishways and instream structures is crucial to evaluating their effectiveness and negative impacts. Monitoring pool-and-weir, vertical slot, and Denil fishways presents less of a technical challenge because they usually have well-defined entrances and exits. Rock ramps, on the other hand, tend to span the width of the stream, so it is difficult to track fish movement across these large, structurally and hydraulically diverse fishways. There are also

numerous logistic difficulties associated with tracking small-bodied fishes. Most individuals are simply too small to mark with tags that allow electronic tracking (such as PIT and radiotelemetry tags), so alternate methods must be used. In **Chapter 3** of this report we describe a field study where we assessed the effects of two instream structures on fish movement rates. The first structure was a rock ramp fishway in the St. Vrain River in Longmont, Colorado, and the second was a vertical drop grade-control structure in Spring Creek in Fort Collins, Colorado.

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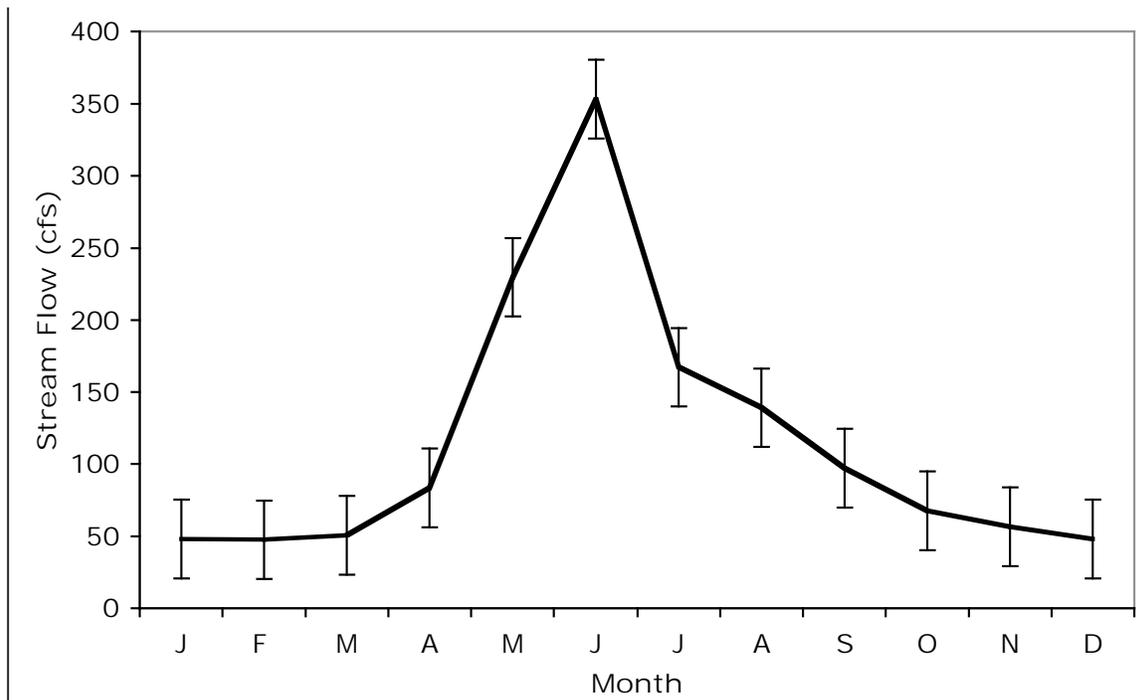
Figures

Figure 1.1a — An annual hydrograph of the St. Vrain River (drainage area 1,098 km^2, elevation 1,479 m above sea level) below Longmont, Colorado based on mean monthly flows in cubic feet per second (cfs). Note the predictable snowmelt peak in May-June. Data are from the U. S. Geological Survey gauge station #06725450.

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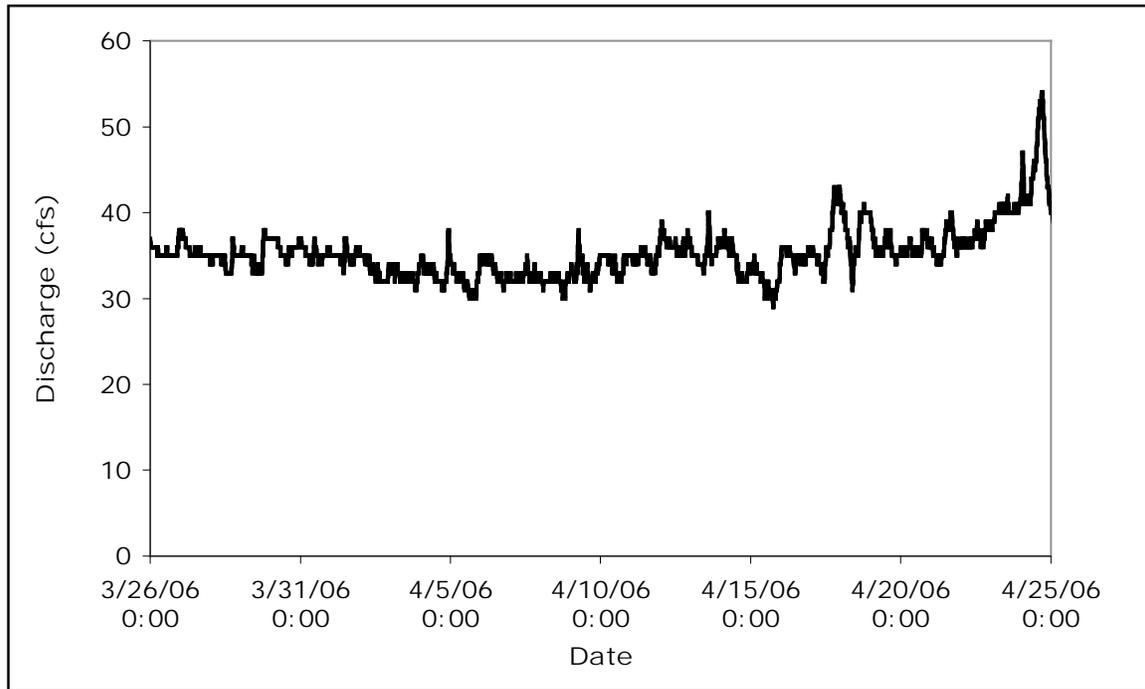
Figure 1.1b — Short-term variability in instantaneous stream flows (in cubic feet per second, cfs) on the St. Vrain River below Longmont, Colorado. Rains on 24 April 2006 accounted for the increased discharge on the following day. Data are from the U. S. Geological Survey gauge station #06725450.
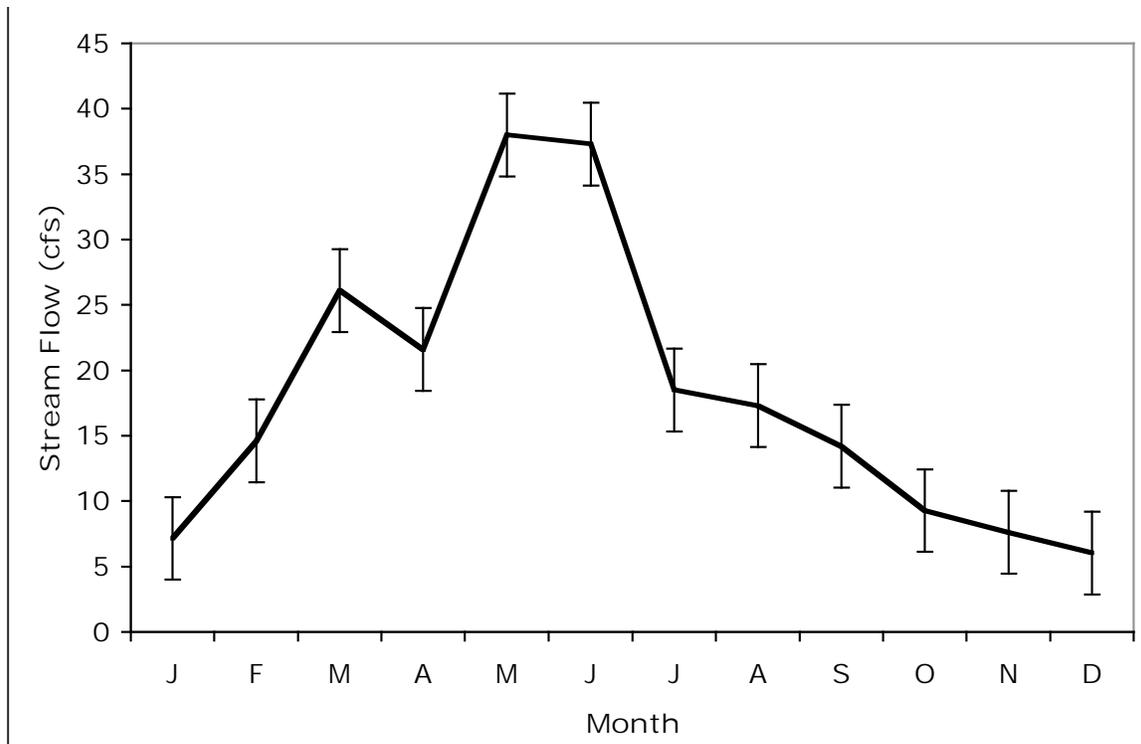
Figure 1.2a—An annual hydrograph of the Arikaree River at Haigler, Nebraska (drainage area 2,642 km^2, elevation 991 m above sea level) based on mean monthly flows in cubic feet per second (cfs). The annual peak occurs from precipitation input and groundwater discharge; this system lacks headwaters in the mountains. Data are from the U. S. Geological Survey gauge station #06821500.
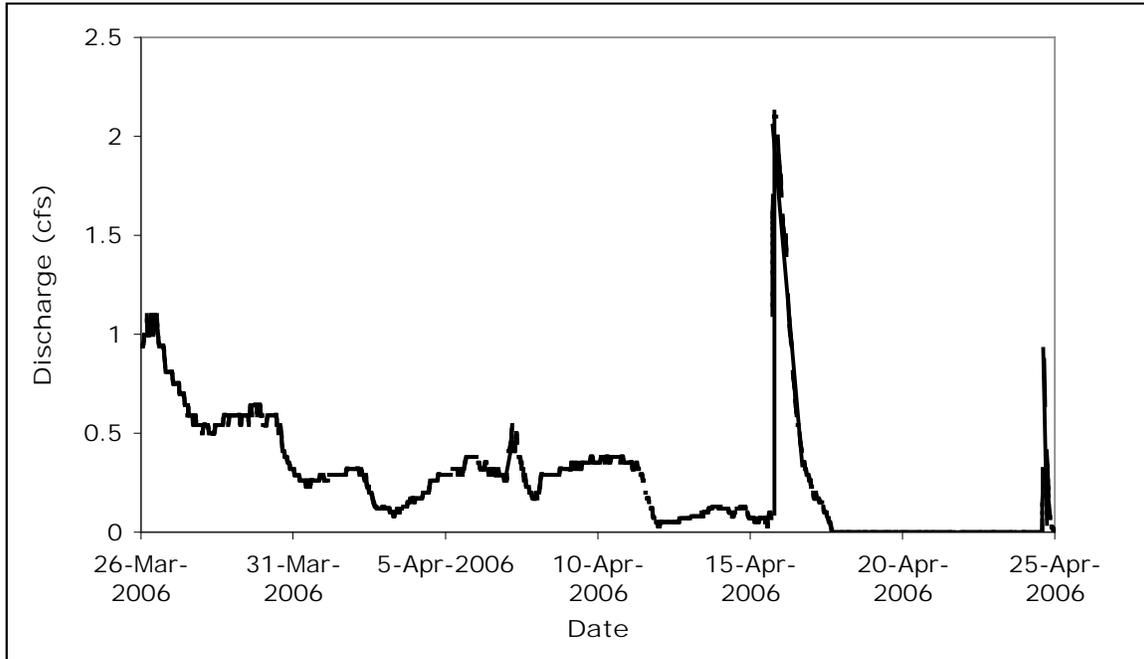
Figure 1.2b—Short-term variation in instantaneous flows on the Arikaree River. Minimal base flows were augmented by precipitation in mid-April. The interannual variation in these systems is greater than that of transition-zone streams. For example, on 17 April 2006, flows varied from 0 to 74 cubic feet per second (cfs) over the 73-year record. Data are from the U. S. Geological Survey gauge station #06821500.

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Figure 1.3 — Pool-and-weir fishway on the John Day Dam, Columbia River, USA. Fish using this structure must ascend it by jumping over each weir into the next pool. Photo: U. S. Department of Energy (www.energy.gov).

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Figure 1.4 — Vertical slot fishway on the St. Ours River, Quebec, Canada. This type of structure does not require jumping, but water velocities in the vertical slot sections can be high. Photo: Parcs Canada (www.pc.gc.ca).
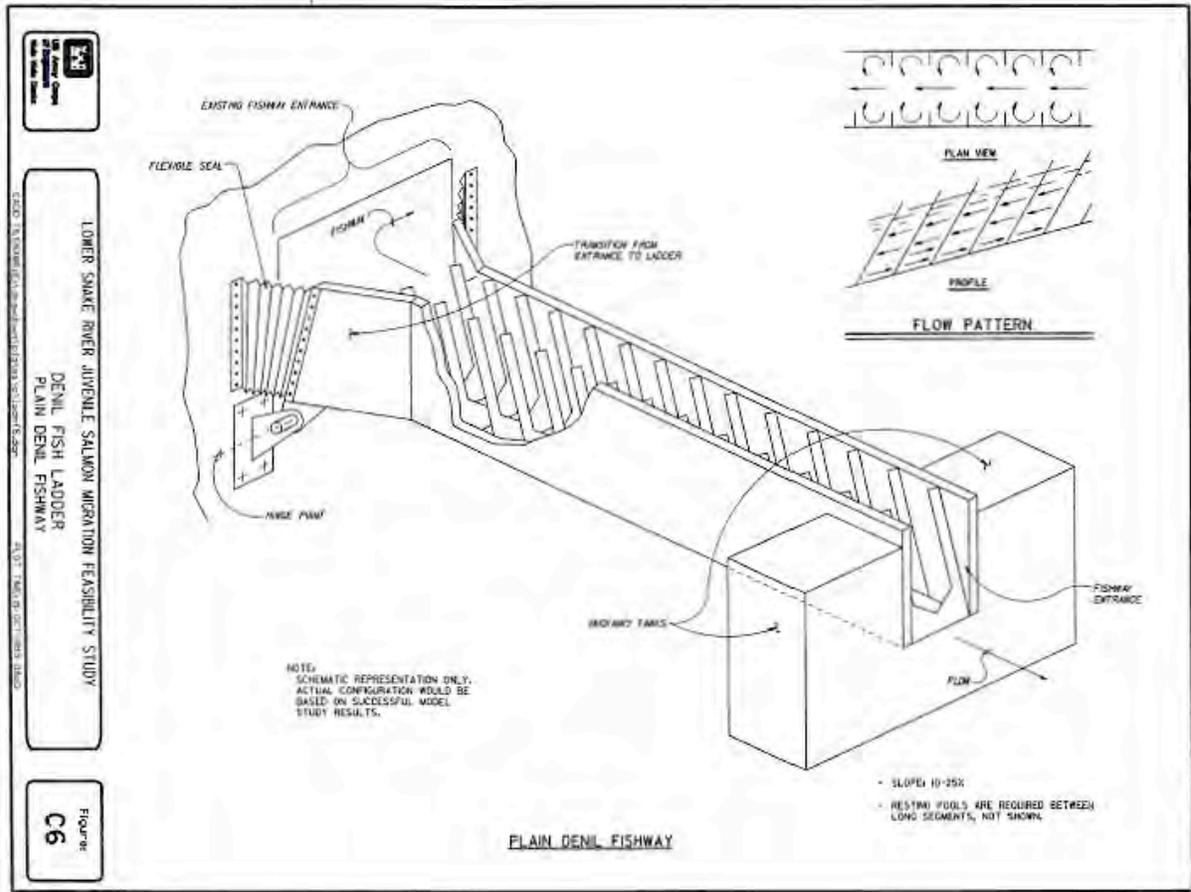
Figure 1.5—Diagram of a Denil fishway. Non-jumping fishes can ascend Denil fishways, and smaller fishes can use the spaces between the vanes to take refuge from high velocity water. However, high water velocities can exclude smaller fishes. Image: U. S. Army Corps of Engineers (www.usace.army.mil).

Figure 1.6—A rock ramp fishway on the St. Vrain River, Colorado, USA. These structures incorporate a number of depths and water velocities and allow the migration of non-jumping fishes. Photo: Chris Myrick.

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Introduction

The streams of Colorado's transition zone and eastern plains are unique environments. Streams with headwaters in the Rocky Mountains receive a predictable snowmelt peak, and those lacking mountain headwaters receive a peak resulting from spring rains and groundwater discharge. This peak varies greatly in magnitude and duration between years. Additional precipitation arrives during the monsoon season (June-August) in the form of short duration, high-intensity rainstorms. This results in a hydrograph that varies markedly within the year as well as between years. This highly variable hydrograph creates a changing mosaic of resources and habitats.

Most of Colorado's plains fishes are not endemic. Instead, they are generalists that can be found throughout many of the Great Plains drainages. In fact, Colorado's plains fish populations are unique in that they represent the westernmost limit of their geographic distributions (Fausch and Bestgen 1997). Habitat specialists that could develop a high degree of endemism are rare because of the stochastic nature of these systems. Because of the harsh abiotic conditions in these environments, plains fishes have survival strategies that include spawning migrations, ranging behaviors to locate resources and refugia, and temporary habitat use for spawning and foraging (Labbe and Fausch 2000; Scheurer et al. 2003b). Colorado's transition-zone and plains fishes have evolved to persist in a stream habitat that essentially consists of a series of dynamic, sporadically connected patches (Labbe and Fausch 2000). Therefore, stream connectivity is absolutely essential to the survival of these populations.

Unfortunately for many plains fishes, urban and agricultural land use and water development along Colorado's Front Range has resulted in the construction of thousands of instream structures including culverts, diversion dams, and grade-control structures. These structures transform continuous systems into a series of potentially fragmented habitats and restrict the ranging and migration behaviors that are central to the success of many plains fishes. As a result of this and other human impacts to streams, 14 species of plains fishes are listed in the state of Colorado as threatened, endangered, or of special concern, largely because of dramatic population declines (**Table 2.1**). Six of Colorado's 38 native plains species have been extirpated from the state (Scheurer et al. 2003a). Numerous studies have documented the importance of habitat connectivity to stream fishes (Nesler et al. 1988; Fausch et al. 2002), and others have implicated fragmentation as potential causes of extinction (Toepfer et al. 1999; Winston et al. 1991). A study on the Red River, Oklahoma reported that speckled chub (*Macrhybopsis aestivalis*), chub shiner (*Notropis potteri*), plains minnow (*Hybognathus placitus*), and Red River

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shiner (*N. bairdi*) were no longer collected in the North Fork of this stream after construction of Altus dam that restricted access to that reach (Winston et al. 1991). Though pre-dam data were sparse, these fishes are otherwise abundant in unregulated streams throughout southeast Oklahoma. Similarly, a 40-cm-high weir near the mouth of Sagentobel Creek, Switzerland caused the extinction of 8 of 10 of its fish species (**Table 2.2**) (Peter 1998).

Three species of particular concern in Colorado are the brassy minnow (*Hybognathus hankinsoni*), the Arkansas darter (*Etheostoma cragini*), and the common shiner (*Luxilus cornutus*). All three species are listed as threatened in Colorado by the Colorado Division of Wildlife, and growing efforts are being made to protect and stabilize remaining populations. In order to better predict how these fishes are affected by natural and anthropogenic challenges, it is necessary to first understand their basic ecological needs. To date, these small nongame fishes have not been extensively researched. The available data on their life histories and environmental tolerances are summarized below.

The brassy minnow reaches adult lengths of 50-90 mm and inhabits small, clear streams with sand or gravel substrate. Brassy minnows eat algae and organic matter and spawn from early May to mid-June (Scheurer et al. 2003b). The transition-zone streams on the Colorado Front Range represent the western limit of the brassy minnow's historic range; their historic distribution in Colorado included the Republican and Platte river basins (Scheurer et al. 2003a). Substantial decreases in numbers and distribution led to their listing as a threatened species in 1998 (Scheurer et al. 2003a). These fish rely upon system connectivity because their long-term persistence apparently depends upon their ability to recolonize empty habitats where populations went extinct due to desiccation or floods (Scheurer et al. 2003b).

The Arkansas darter, a spring-stream specialist, is thought to be a glacial relic in Colorado (Labbe and Fausch 2000). Arkansas darters have been listed as threatened in the state of Colorado since 1975 and were a candidate species for federal listing by the U. S. Fish and Wildlife Service (Loeffler and Krieger 1994). Arkansas darter populations consist mainly of age-0 and age-1 fish (Taber et al. 1986); age 1 fish collected in southeastern Colorado streams attained a length of 19-48 mm (Labbe and Fausch 2000). The largest adult collected by Taber et al. (1986) during their survey was 63 mm. These fish reproduce by burying eggs in silt-sand substrate when water temperatures are between 9 and 17°C (Taber et al. 1986; Labbe and Fausch 2000). Arkansas darters are thought to have a temperature tolerance similar to that of other plains fishes. At acclimation temperatures ranging from 20–30°C, their upper incipient lethal limit, or temperature at which they cannot survive indefinitely, ranged from 34 to 36°C (Smith and Fausch 1997). Although they are not lethal, temperatures of 27.5 and 30°C are physiologically stressful (Smith and Fausch 1997). Though historic records of Arkansas darter distribution are quite sparse, their range is thought to have contracted considerably since the late 1800's (Loeffler and Krieger 1994). The need for connectivity between habitat patches and the use of seasonally available habitats has also been documented for this fish (Labbe and Fausch 2000).

The common shiner is an omnivorous minnow with a typical adult size of 64-127 mm (Pflieger 1997; Trial et al. 1983a). This fish reproduces by laying adhesive eggs in gravel nests excavated by the male, or they use existing nests constructed by other species. This results in hybridization between common shiners and other minnows such as creek chub (*Semotilus atromaculatus*), and

rosyface shiner (*Notropis rubellus*) (Menzel 1978). Spawning migrations take place when water temperatures are between 12 and 16°C, and spawning occurs at temperatures ranging from 15.5 to 18.3°C (Raney 1940; Trial et al. 1983a). In Missouri, spawning activity occurs between April and June, and it peaks in May (Pflieger 1997). These fish prefer current velocities in the range of 20 cm/s (Trial et al. 1983a). Barila et al. (1982) conducted temperature preference trials on common shiner with acclimation temperatures of 6-30°C, and reported that the preferred temperatures ranged from 13 to 24°C, respectively (Barila et al. 1982). The final temperature preference of this species was determined by setting the preferred temperature equal to the acclimation temperature; the calculated temperature was 22.2°C (Barila et al. 1982). Common shiners require unchannelized reaches for reproductive success; a study of 13 reaches in the Red River drainage in Minnesota found that common shiner larvae were strongly associated with unchannelized habitats (Meneks et al. 2003). In Wyoming, common shiners were found in main channel habitats of streams ranging from 3.9 to 6.2 m in average width, and they appeared to prefer habitats with submerged macrophytes (Hubert and Rahel 1989).

Because persistence of these plains fishes depends upon stream system connectivity, fragmentation must be reversed in order to prevent further declines and allow species recovery. Possible solutions include the removal or modification of instream structures to allow fish passage. Instream structure removal to ensure survival of native fish species is generally not feasible because the structures provide benefits such as water storage and flood control to people living on the Front Range. Given this, perhaps the only viable option for restoring connectivity to Front Range streams is the construction of fish ladders. Traditionally, efforts elsewhere have focused on larger fishes, many of which are anadromous game species. To our knowledge, no fishways on Colorado's Front Range have been evaluated for their ability to allow upstream and downstream fish passage. Until recently, there were virtually no data on the swimming and jumping ability of small-bodied fishes. Therefore, it is likely that any fishways in this region were built based on the abilities of other fish species, or that they were designed without consideration of any swimming or jumping data. Locally relevant data must be used in fish ladder design if we are to effectively reverse stream fragmentation and the resulting declines in native fish populations. Therefore, there is a genuine need for data on the swimming and jumping abilities of Colorado's plains fishes.

Task Objectives

This phase of our project was designed to collect data on the jumping and swimming performance of three representative plains fishes. Our specific objectives were the following:

- Quantify the swimming and jumping performance of the brassy minnow, common shiner and the Arkansas darter as a function of water temperature.

- Use the fish swimming and jumping results to recommend maximum allowable water velocity, maximum allowable fishway length, and the most effective fishway type for structures designed to facilitate passage of these three fish species.

- Explored the applicability of these results to other species and systems.

Hypotheses

We formulated the following hypotheses with regards to the effects of temperature on the swimming and jumping performance of brassy minnows, Arkansas darters, and common shiners:

• Fish swimming performance will increase as water temperature increases from 10 to 25°C.

• Brassy minnow, common shiner, and Arkansas darter jumping ability will be function of waterfall height and plunge pool depth; it is probable that some or all of these fishes will not attempt to jump over small waterfalls.

Methods

Sources of Fish and Fish Transport

Brassy minnows were collected from a broodstock pond in the Tamarack State Wildlife area near Sterling, Colorado on 30 September 2004. These fish were originally collected from the South Platte River near Sterling and moved into this pond by the Colorado Division of Wildlife. They were transported back to the Colorado State University Foothills Fisheries Laboratory in a 600-L transport tank. Compressed oxygen was delivered to each compartment through a fine-pore diffuser. Sodium chloride (NaCl, 3 g/L) was added to the tank to alleviate osmotic stress, and Polyaqua™ was added (approximately 0.1 ml/L) to replace lost slime coat and to reduce the probability of infection. At the Foothills Fisheries Laboratory, fish were transferred into round 340-L polyethylene tanks containing 3 g/L of NaCl and 0.1ml/L of Polyaqua™.

Arkansas darters were obtained from the John Mumma Native Aquatic Species Recovery Center (NASRC) in Alamosa, CO on 9 May 2005. Hatchery employees transported them in sealed plastic bags inside coolers. At the Foothills Fisheries Laboratory, the bags were placed into the tanks described above. Fish were released from the bags when the water temperatures in the bags and the tanks equilibrated. Common shiners were also obtained from the NASRC, but they were hauled in transport tanks. We adjusted the temperature of the holding tanks in the laboratory to 1-2°C below the water temperature in the hauling tanks. Fish were transferred from the hauling tanks to the holding tanks with 5-gallon buckets. Sodium chloride and Polyaqua™ were added to the holding tanks in order to minimize mortality from osmotic stress or transport-related injuries.

A potential limitation of this project was our use of hatchery fish. The Arkansas darters were first-generation offspring of wild stock captured from the Big Sandy Creek drainage in eastern Colorado. The common shiners were first-generation offspring of wild stock captured from the St. Vrain River in Boulder and Weld Counties, Colorado. Therefore, the tested fish had not been subject to generations of artificial selection. In terms of observed behaviors during experiments, the Arkansas darters and common shiners reacted similarly to experimental stimuli as the wild-caught brassy minnows. For Arkansas darters, differences in swimming modes and behaviors in the artificial waterfalls were as expected for a bottom-oriented fish. It is perhaps fortunate that our experiments did not rely upon observations of complex social behaviors. Instead, we were testing physiological ability, which may not be as changeable as other traits. It is possible that wild fish are better conditioned than hatchery fish. In order to compensate for this, we rotated the tank

spray bars so that the water entering the holding tanks created a low-velocity current. The direction of the current was reversed weekly to ensure equal conditioning of the muscles on both sides of the fish. If differences in physical ability do exist between wild and captive Arkansas darters or common shiners, our estimates of swimming and jumping ability are probably conservative (McDonald et al. 1998).

Fish Care

Fish were held in 340-L round polyethylene tanks at the Colorado State University Foothills Fisheries Laboratory. The tanks received approximately 10 L/min of air-saturated water. Water temperature was controlled within 1°C of the target temperature with electronic mixing valves (George Fischer Inc.). Fish were randomly assigned to one of three rows of tanks and allowed to adjust to laboratory conditions and to recover from transport stress for one week. After this period, the temperatures were adjusted by 1°C per day until one row of tanks contained water at $10 \pm 1°C$, one contained $17.5 \pm 1°C$ water, and one contained $25 \pm 1°C$ water. Common shiners were acclimated to 10 or 17.5°C; the 25°C treatment was omitted due to constraints on time and fish availability. These temperatures reflect the natural temperature regime of streams on the eastern Great Plains of Colorado. Brassy minnows spawn during April and May, and their eggs hatch at temperatures ranging from 15.7 – 23.5°C (Scheurer 2002); Arkansas darters will spawn at temperatures between 8 and 17°C (Taber et al. 1986). Therefore, spawning migration of these species is most likely to occur at temperatures bracketed by the two lower treatments. Likewise, common shiners may begin spawning migrations and spawning activity at temperatures ranging between 10 and 17.5°C. Ranging behaviors or movements to seek refuge from high flows or low flows would likely occur at the two higher temperatures.

All fish were maintained on a natural photoperiod by using timers to control the lighting within the facility. The timer was adjusted weekly to match the natural photoperiod for the Fort Collins area (40.54°N). Tanks were covered with mesh and screened by an opaque enclosure to minimize disturbance from other activity in the laboratory. All fish were fed ad libitum rations twice daily. Brassy minnows and common shiners were fed a commercial flake food for omnivorous fish (Brine Shrimp Unlimited) and frozen adult brine shrimp, and Arkansas darters were fed live brine shrimp nauplii, frozen adult brine shrimp, and frozen bloodworms. Tanks were siphoned once daily.

Laboratory Assessment of Swimming Performance

Swimming endurance is related to water velocity, and fish swimming modes have been separated into three categories. Sustained swimming occurs over long periods of time (generally in excess of 200 minutes) and at low velocities, prolonged swimming occurs between 1 and 200 minutes and at higher velocities. Burst or sprint swimming is short-term (one minute or less), and involves the highest speeds a fish can attain (Beamish 1978). Trial velocities were tailored for each species at each temperature so that we could observe fish swimming in all three modes. Using these temperatures and trial speeds allowed us to build endurance curves for two fish species over a range of relevant temperatures and make recommendations for fishway design based on empirical data. Fish swimming performance was evaluated using the endurance test protocol described by Peake et al. (1997), wherein fish swim at a fixed velocity until they are fatigued. Trial velocities

were based on the mean total length of the fish held in the lab and the swimming ability of the fish (determined by pilot tests). Velocities were chosen so that they encompassed all three swimming modes. Brassy minnows held at 10°C and Arkansas darters at all temperatures were tested at 4, 8, 16, or 32 cm/s. Because brassy minnows were able to swim for much longer times at the two higher temperature treatments, we increased the velocities for the 17.5°and 25°C trials to 16, 32, 48 or 64 cm/s. Common shiners at both temperatures also swam at 16, 32, 48, or 64 cm/s (**Table 2.3**). Swimming performance measurements were made using Loligo Model 32 swim tunnels (Loligo Systems, **Figure 2.1**). The test temperature and water velocity were randomly selected, and fish were tested at their acclimation temperatures.

Individual fish were selected from the holding tanks, placed in the swim tunnel and allowed to recover from handling for 2 hours. During the recovery period, a current of 0.5 body lengths per second (bl/s) was used to provide rheotactic orientation. Because swimming performance experiments by Peake et al (Peake et al. 1997), Smiley and Parsons (Smiley and Parsons 1997), and Kolok (Kolok 1991) included acclimation times of 1 or 1.5 hours, we decided that 2 hours would be sufficient for recovery and acclimation to the flume. After 2 hours elapsed, the water velocity was increased to the test velocity. The water velocity was maintained until exhaustion, defined as the point when the fish was no longer able to swim and became impinged upon the rear screen of the swimming chamber. We assumed that a fish could swim indefinitely at a test velocity if it maintained position in the flume for more than 200 minutes. Although the cutoff time indicating sustained swimming is species-specific, 200 minutes is considered appropriate (Northcote 1998; Peake et al. 1997).

Once a fish became impinged, or if 200 minutes elapsed, the current was stopped. The fish was removed from the swim tunnel placed in an anesthetic bath containing 30 mg/L tricaine methanesulfonate (MS-222) and 3 g/L NaCl (to reduce osmotic stress). The fish was weighed to the nearest 0.01 g and measured (total length) to the nearest mm. Fish were placed in a recovery bath containing 3 g/L NaCl and 0.1 ml/L Polyaqua™ until they regained equilibrium. After recovery, they were placed in a separate tank reserved for tested fish. No fish were tested more than once to eliminate the confounding effects of training (Farlinger and Beamish 1978).

Based on a power calculation performed in JMP (SAS Institute) and data from a pilot study that measured the critical swimming speed of the small-bodied sand shiner (*Notropis stramineus*), the sample size per treatment was set at 20 fish. Therefore, our ideal sample size for each species was as follows:

20 fish per trial × 3 temperatures × 4 swimming speeds = 240 fish.

We were able to obtain enough brassy minnows to run all of the planned trials. The limited availability of Arkansas darters allowed me to run complete trials at 10 and 17.5°C and a small number of tests at 25°C. The limited availability of common shiners allowed me to run complete trials at 10 and 17.5 °C.

Fish that refused to swim or were reluctant to do so were classified as "non-performers". These individuals would cease swimming before exhaustion occurred or would not respond to the current, choosing instead to rest on the screen at the rear of the swimming chamber. Non-

performers are common in swimming performance studies and were not included in the data analyses. Instead, we recorded the number of sustained swimmers and the number of non-performers at each temperature and velocity combination. Then, we examined the numbers to see if either of these predictor variables affected the frequency of these outcomes.

Laboratory Assessment of Fish Jumping Ability

Fish jumping ability was measured with three artificial waterfalls (**Figure 2.2**) based on Kondratieff and Myrick's (2005) design (**Figure 2.2**). These are acceptable simulations of a pool-and-weir fishway. The upper and lower chambers were 0.6 m wide, 0.6 m long, and 1.2 m tall. Trials were run at 10, 17.5, and 25°C for both species. However, few trials were conducted for Arkansas darters held at 25°C because of limited numbers of fish.

In each trial, a group of fish was placed in the lower section of the apparatus, and water was pumped at 72 L/min into the upper reservoir and allowed to flow over the weir. Fish were given approximately 24 h to access the upper chamber of the apparatus. At the end of the experiment, fish from the upper and lower reservoirs were removed, separated by location, and processed as described for the fish swimming experiments. Our original intention was to test 45 pool depth (H_1 = 10, 20, or 30 cm) × weir height (H_2 = 0 to 20 cm in 5-cm increments) × temperature (10, 17.5 and 25°C) combinations (**Table 2.4**), but some combinations were omitted because of the limited number of fish. Due to the lack of published studies on the jumping ability of Colorado plains fishes, we ran pilot studies with lower weir heights (i.e., 0 and 5 cm) and increased the heights in subsequent trials. Fish were presented with increasing weir heights until they could not access the upper chamber. This height was taken to be the critical weir height. Once the critical weir height was reached, additional trials were run at the limiting height + 5 cm to confirm that a maximum height had been reached. Once the maximum jumping ability had been obtained, trials were assigned randomly. Fish activity was observed in the waterfalls several times daily to confirm that the fish were attempting to access the upper chamber of the apparatus. Because we observed attempts within minutes of starting a trial, and fish attempted to jump the weirs at all times of the day, no stimulus other than water flow was needed to induce movement. This behavior has been documented in other studies on fish jumping performance (Kondratieff and Myrick 2005; Kondratieff and Myrick 2006; Stuart 1962). In the event that fish were not found in the upper chamber, we assumed that they were unable to negotiate the weir.

Brassy minnows and common shiners were tested in groups of ten fish. Each combination of plunge pool and weir height was replicated four times with novel groups of fish. The sample size was based on studies measuring the jumping ability of Rio Grande cutthroat trout (*Oncorhynchus clarki virginalis*) and brook trout (*Salvelinus fontinalis*) (Kondratieff and Myrick 2005; 2006). We could not obtain enough fish to perform four replicate trials for each weir height x plunge pool combination, so some of the trials had fewer replicates (**Table 2.4**). The Arkansas darters were tested in groups of five to maximize the use of a limited number of fish. Nevertheless, we were able to perform four replicates of the trials at 10 and 17.5°C for weir heights of 0 and 5 cm. At 25°C, we were only able to run three trials, one with H_1=10 cm and H_2=0 cm, and two with H_1=10 cm and H_2=5 cm. Because of their small size (26-46 mm, μ = 36.9 mm), we used plunge pool depths of 5, 10, and 20 cm in the Arkansas darter trials.

Data Analyses

Analysis of Swimming Performance Data

Swimming performance data were examined by a survival analysis (proc LIFEREG, program SAS, or the survival platform in JMP). This method allowed us to include censored data (i.e., all instances where a fish swam for over 200 minutes and was interrupted by the end of the trial). Chi-square tests were used to test the significance of measured variables; predictors were considered significant at the $\alpha = 0.05$ level. Previous studies have found that temperature, trial velocity, total length, time of day, and photoperiod can affect swimming endurance, so these were all included in the analyses.

The survival analysis allowed us to estimate the time to failure (i.e., exhaustion) for the 50th percentile of fish. These data were used to determine the maximum allowable length of a fish ramp (or distance between velocity refuges on a fishway) at a given water velocity for each species. Once we developed endurance curves, we determined maximum fishway length using the equation:

$$v_f = v_s - \left(d \times E_{v_s}^{-1} \right)$$

where v_f = the water velocity in the fishway (cm/s), v_s = the swimming speed of the target species (cm/s), d = fishway length (cm), and E_{vs} = the endurance of the fish at the given velocity (seconds) (Peake et al. 1997).

Laboratory tests provided us with paired values of fish swimming speed (v_s) and endurance at that velocity (E_{vs}) that could be inserted into Peake's equation in combination with a range of fishway water velocities. It is important to note that v_f must be lower than v_s so that a fish can attain a positive ground velocity against the current in the fishway. Peake's equation can be solved for distance or for fishway water velocity; it produces acceptable combinations of the two variables that will allow fish passage.

Analysis of Jumping Performance Data

Our approach for analyzing fish jumping data was based on the techniques developed by Kondratieff and Myrick (2005). The probability of successful passage was modeled using fish length, weir height, plunge pool depth, photoperiod, and water temperature as predictors. These models were constructed using the logistic regression function in SAS (program PROC MIXED). This analysis also provided us with success probabilities (success was defined as movement into the upper chamber). Candidate models were ranked using the AIC_C selection criteria that accounts for small sample size (n = 4) (Anderson et al. 2001). If multiple models were supported by the data, success probabilities were determined by model averaging (based on the relative weights of the models). Because social interactions such as schooling behaviors were observed in these trials, we considered the trial group as an experimental unit. Although temperature was thought to be important in determining the probability of success, model constraints dictated that each of the three temperatures be modeled separately.

DEPARTMENT OF FISH, WILDLIFE, AND CONSERVATION BIOLOGY
FISH BARRIERS AND SMALL PLAINS FISHES: FISHWAY
DESIGN RECOMMENDATIONS AND THE IMPACT OF
EXISTING INSTREAM STRUCTURES

COLORADO STATE UNIVERSITY
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Results

Brassy Minnow Swimming Performance

Brassy minnows used three gaits and maintained a position in the water column throughout the duration of the tests. Most of the tested fish used the swim-and-coast swimming gait at low speeds. However, some fish did swim constantly, especially at higher speeds. Fish swimming at high velocities or nearing exhaustion used a burst-and-swim gait, suggesting that they were recruiting their white muscle to generate more power (Webb 1998). The number of nonperformers was much higher at 10°C at all velocities, and non-performers were rare at higher temperatures (**Figure 2.3**). The number of non-performers at 10°C was also higher at lower velocities. On the other hand, sustained swimmers were more common at higher temperatures and, within each temperature treatment, they were more prevalent at lower velocities (**Figure 2.4**).

According to the tests of fixed effects in proc LIFEREG, water temperature and trial velocity significantly affected swimming endurance (P<0.0001). The regression equation developed from the data was as follows:

$$E = 5.2455 - \left(1.8385 \bullet \log(v)\right) + \left(0.1448 \bullet T\right)$$

where E = endurance time (minutes), v = trial velocity (cm/s), and T = water temperature (°C). For each velocity, brassy minnows swam for shorter intervals at 10°C than at the other two temperatures (**Figure 2.5**). The 10°C treatment fish used sustained swimming modes at 8 cm/s, prolonged modes from 8 to 32 cm/s, and sprint modes above 32 cm/s. Velocities less than or equal to 32 cm/s were sustained swimming speeds for brassy minnows at 17.5 and 25°C. Although some were able to swim for extended periods at 48 cm/s, most fish swimming at speeds between 32 and 48 cm/s exhibited prolonged or sprint behaviors. All fish swimming at 64 cm/s sprinted for short periods of time.

Arkansas Darter Swimming Performance

Unlike the brassy minnows, Arkansas darters did not swim continuously. Instead, they attempted to hold their position on the bottom of the flume by using their fins as ailerons. If the current was strong enough to displace them, they used short bursts to regain their position. Darters were able to hold position without swimming constantly at 4 and 8 cm/s, but they were rarely able to do so at 16 cm/s. They never held position on the bottom of the swimming chamber at 32 cm/s. Non-performers were rare (n = 4 at 10 and 17.5°, all at 4 cm/s) in the darter swimming trials, probably because of their swimming behaviors at low speeds.

According to the tests of fixed effects in proc LIFEREG, only trial velocity significantly affected swimming endurance (P<0.0001) (**Figure 2.6**). The regression equation developed from the data was as follows:

$$E = 17.8989 - \left(13.3578 \bullet \log(v)\right),$$

where E = endurance time (minutes) and v = velocity (cm/s). Trial velocities of 16-32 cm/s seemed to represent a prolonged swimming speed, and velocities greater than 32 cm/s represented a sprint swimming speed.

Common Shiner Swimming Performance

Like brassy minnows, common shiners used three gaits and maintained a position in the water column. Tested fish used the swim-and-coast or constant swimming gaits at low speeds. Fish swam constantly at higher speeds, and those swimming at high velocities or nearing exhaustion used a burst-and-swim gait. The number of nonperformers was higher at 10°C at all velocities, and non-performers were rare at 17.5°C (**Figure 2.7**), suggesting a lack of motivation or the use of resting behaviors at lower temperatures. The number of non-performers at both temperatures was also higher at lower velocities. On the other hand, sustained swimmers were more common at higher temperatures and, within each temperature treatment, they were more prevalent at lower velocities (**Figure 2.8**).

According to the tests of fixed effects in proc LIFEREG, trial velocity, fish total length, and photoperiod significantly affected swimming performance (P< 0.0001) (**Figure 2.9**). The regression equation developed from the data was as follows:

$$E = 11.5 - \left(11.6148 \cdot \log(v)\right) + \left(0.0755 \cdot TL\right) + \left(0.3512 \cdot PP\right),$$

where E = endurance (min), v = velocity (cm/s), TL = fish total length (mm), and PP = photoperiod (hours of daylight). A velocity of 16 cm/s represented sustained swimming speeds, 32 cm/s represented a sustained or prolonged swimming speed, 48 cm/s represented a prolonged swimming speed, and 64 cm/s represented a sprint swimming speed.

Brassy Minnow Jumping Performance

Brassy minnows usually approached the weir from the side and, unlike trout, did not sprint from the bottom of the plunge pool to jump the obstacle. However, brassy minnows are capable of true jumping where their entire body clears the water, much like brook trout and Rio Grande cutthroat trout (Kondratieff and Myrick 2005; 2006). Pilot studies on other plains fish species such as longnose dace (*Rhinichthys cataractae*) and fathead minnows (Ficke and Myrick, unpublished data) demonstrated that these fish will attempt to swim up a weir face or up a column of water but their bodies do not completely exit the water. Fish were more active at higher temperatures, and the depth of the plunge pool appeared to play an important role in determining success.

At each temperature, the best-ranked models included only height terms: plunge pool depth (H_1) and weir height (H_2) were always present in the top model, and an interaction term between the two ($H_1 \times H_2$) was important at the two higher temperatures (**Table 2.5**, **Figures 2.10-2.12**). Brassy minnow jumping ability was highly temperature-dependent. Brassy minnows acclimated at 10°C did not jump any weirs with a height greater than zero, and the plunge pool depth did not affect the probability of movement over the weirs. Movement over the weir was lower at 10°C; the probability of success at 10°C with a 0-cm weir height was approximately 0.5, but at higher temperatures, it ranged from 0.5 to 0.95, depending upon plunge pool depth (**Figures 2.10-2.12**).

Temperature also affected maximum jump heights; at 17.5°C, fish jumped a maximum height of 10 cm, and at 25°C, fish jumped a maximum height of 15 cm. Plunge pool depth affected the probability of success at each weir height at these two higher temperatures (**Figures 2.11** and **2.12**). Although a 10 or 20-cm deep plunge pool seemed to increase success at all weir heights, increasing the weir heights beyond zero dramatically decreased success probabilities at all temperatures.

Arkansas Darter Jumping Performance

Arkansas darters did not jump at any temperature. Instead, they approached the weir from behind or from the side and attempted to swim up the water column. Often, the higher water velocity or the turbulence where the falling water entered the lower pool displaced the fish.

As with the brassy minnows, photoperiod and fish total length did not affect the probability of upstream movement. Although each temperature treatment was modeled separately, temperature did not have a large effect on movement probabilities. The probability of successful movement over a 0-cm weir was slightly higher at 17.5°C than it was at 10°C (**Figures 2.13** and **2.14**). The probability of upstream movement was affected by H_2 at both temperatures, but the data only supported a model containing H_1 and the interaction term ($H_1 \times H_2$) at 10°C. Essentially, the depth of the plunge pool was important in determining whether fish moved upstream over a 0-cm weir (**Table 2.6**). The darters were most likely to move upstream at 10°C at a pool depth of 10 cm. They may not have moved from the 5-cm plunge pool because they were seeking cover. Fish held in a 5-cm plunge pool were largely stationary, and they held position near the walls of the lower chamber or near the standpipe.

A 5-cm weir prevented upstream movement of all darters, regardless of plunge pool depth or test temperature. However, it appears that even a submerged weir structure may reduce movement rates. The probability of upstream movement over the 0-cm weirs was between 0 and 0.27 (depending upon H_1) at 10°C and 0.35 at 17.5°C (**Figures 2.13** and **2.14**). Arkansas darters were less than half as likely to move over a weir as brassy minnows were.

Common Shiner Jumping Performance

Jumping behavior was also rare for common shiners. These fish attempted to swim up the water column to access the upper chamber. However, they were more successful in negotiating the weir structures than Arkansas darters. Success probabilities at both temperatures were unaffected by photoperiod and strongly influenced by weir height. However, the data supported a model including total length at 10°C, and it supported an interaction term ($H_1 \times H_2$) at 17.5°C (**Table 2.7**). Although resolution between the top models at 10°C was poor, estimates produced by all three of them were essentially identical. At 10°C, a 5-cm weir nearly eliminated upstream movement (p = 0.02-0.35, **Figure 2.15**). At 17.5°C with a 0-cm weir, jumpers were more successful from 10 and 20-cm plunge pools. At 17.5°C with a 5-cm weir, jumpers were more successful from 20 and 30-cm plunge pools. A 10-cm weir nearly eliminated upstream movement (p = 0.01-0.04, **Figure 2.16**).

Recommendations for Fishway Length and Water Velocity

Because it is best to use conservative estimates for swimming performance in fish ladder design, we used the observed medians to obtain an estimate of maximum fish ladder length. For example, because we didn't conduct any trials at 10°C and 64 cm/s, we used the predicted 50th percentile endurance value to complete the curve. Observed medians were used whenever possible because the regression equation tended to overestimate swimming abilities of brassy minnows (**Table 2.8**). We constructed the curve to describe the maximum allowable fishway length based on brassy minnow performance at 10°C in the interest of using conservative estimates.

As with the brassy minnow endurance results, we compared the observed medians for Arkansas darter and common shiner performance with the predicted values from the equation. Estimates of Arkansas darter performance exceeded observed medians at all velocities except 32 cm/s (**Table 2.9**). Therefore, in the interest of using conservative estimates for fishway design, we used the predicted values from the regression equation for Arkansas darter sustained speeds (4 and 8 cm/s), the observed median values for prolonged speeds (16 cm/s), and the predicted values for sprint speeds (32 cm/s). Estimates of common shiner performance were also much higher than observed medians at sustained speeds, lower than observed medians at prolonged speeds, and essentially equal at sprint speeds (**Table 2.10**). For common shiners, we used regression output for sustained and prolonged speeds (16 and 32 cm/s), and the lower of the two observed medians at sprint speeds (48 and 64 cm/s).

Our results indicate that a fishway with a water velocity of 30 cm/s should be no longer than 3 m to accommodate brassy minnows at 10°C, 25 m long for brassy minnows at 17.5°C, 0.2 m long for Arkansas darters at any temperature, and 11 m for common shiners at any temperature (**Figures 2.17-2.19**). Fishways designed to accommodate sprinting abilities of brassy minnows and common shiners (water velocity = 60 cm/s) should not incorporate lengths greater than 3 m between velocity refuges for brassy minnows or 1.2 m for common shiners. The implications for fishway design are discussed in a subsequent section.

Discussion

Brassy minnow swimming endurance was dependent upon trial velocity and trial temperature. Brassy minnows held at 10°C swam for shorter times at each trial velocity than fish held at higher temperatures. Arkansas darter swimming performance depended only on trial velocity. These fish held their position on the bottom of the flume at lower trial velocities instead of swimming in the water column. Common shiner swimming performance was dependent upon trial velocity, fish total length, and photoperiod. Although brassy minnows jumped maximum heights of 10 cm at 17.5 and 15 cm at 25°C, brassy minnows held at 10°C and Arkansas darters held at any temperature did not jump. Common shiners jumped a maximum of 10 cm at both 10 and 17.5°C. The presence of any weir or vertical drop restricts upstream movement of all three species.

Predictive Ability of Swimming Endurance Regression Equation

We predicted endurance times for 50 percent of the brassy minnows at each tested velocity and temperature, but we caution that the predictive ability of the equations was limited. When the

predicted values for 50% success were compared to the observed medians, they overestimated swimming ability at all temperatures and most speeds (**Table 2.8**). At 10°C, predicted endurance was greater than the observed medians for all speeds, but it was within the same order of magnitude for higher velocities. At 17.5°C, the predicted values were lower than the observed ones for trials at 16 and 32 cm/s, but they were still greater than the observed medians for higher speeds. At 25°C, all predicted values greatly exceeded the observed medians. Observed medians of Arkansas darter swimming endurance were lower than the predicted 50th percentile by a factor of 3.25 at 16 cm/s (**Table 2.9**). They were higher than predicted values at 32 cm/s by a factor of 2.6. Observed medians for common shiners at both temperatures were lower than predicted values at 16 cm/s by a factor of 20 and higher than predicted values at 32 cm/s by a factor of 3.36. The observed median at 48 cm/s was lower than the predicted value by a factor of 2, and the observed and predicted values were equal at 64 cm/s (**Table 2.10**). At lower velocities, the overestimation of median endurance is an artifact of the survival analysis.

However, poor predictive value at higher velocities may have been due to the large individual variation in swimming performance. For example, endurance times for brassy minnows swimming at 48 cm/s at 25°C ranged from 0 to 200 minutes. Although it is possible that we failed to account for some predictive factor in these experiments, it is likely that the individual variation is the primary cause of the inaccuracy in the equations. This variability in performance within species is commonplace in swimming performance experiments (Bengtsson 1980; Billman and Pyron 2005).

Swimming Performance

Studies of swimming performance of other small plains fishes indicate that the swimming abilities of brassy minnows and common shiners are not unusual. The swimming ability of Topeka shiners (*N. topeka*) is similar to that of these two species. Tested fish (TL 30-55 mm) were able to sprint for short periods at water velocities of 75 cm/sec at 20°C (Adams et al. 2000). A 2005 study on the endurance of 15 North American minnows (30-60 mm standard length) at 20°C and at velocities of 67 cm/s reported mean endurance times ranging from 9 to 65 seconds (Table 2.11) (Billman and Pyron 2005). The median endurance times for our 64 cm/s trials were one minute for brassy minnows and 30 seconds for common shiners.

The ability of small-bodied fishes to swim at fast relative speeds, especially at higher water temperatures, is most likely a product of selective pressures of their environment and, to some extent, an artifact of their small size. Plains streams are characterized by a lack of permanent instream habitat structure and unpredictable hydrographs (Fausch and Bestgen 1997; Labbe and Fausch 2000; Scheurer et al. 2003b). Therefore, enhanced swimming ability may be the only option available to a small-bodied fish to prevent downstream displacement. This swimming ability would also make rapid recolonization of empty habitats after a flood possible.

The temperature-dependent swimming ability of the brassy minnow would not prevent its making successful spring migrations in Colorado plains streams. In the Arikaree basin, brassy minnows spawn in April-May (Scheurer 2002), so it is likely that they do migrate during this time. Water temperatures in Colorado's plains streams are approximately 10 – 20°C during this season (Labbe and Fausch 2000; Scheurer 2002). The Arikaree River has a wide floodplain, and, when discharge

increases, stream width and the availability of multiple low-velocity habitats increase. Therefore, although water temperatures may be suboptimal for high swimming performance, brassy minnows probably select low-velocity habitats through which to move. Later in the season, when water temperatures are higher and the hydrograph is affected by afternoon thunderstorms, swimming performance is improved and fish are able to avoid displacement during small floods.

Arkansas darters did not swim constantly at low speeds. Position holding behaviors are common, especially for bottom-oriented fishes. Station holding has also been documented for darters, *Percina* spp. and *Etheostoma* spp. (Matthews 1985; Toepfer et al. 1999), Sacramento suckers (*Catostomus occidentalis*) (Myrick and Cech 2000), Atlantic cod (*Gadus morhua*) (Gerstner 1998) and shovelnose sturgeon *Scaphirhynchus platorhynchus* (Parsons et al. 2003). Arkansas darters swimming performance is comparable to that of leopard darters; leopard darters (mean total length = 58 mm) were able to sprint against currents of 42 and 60 cm/s (Toepfer et al. 1999). Although Arkansas darters were not able to swim as quickly, they were smaller (average total length = 35 mm), and they swam at similar relative speeds.

These fish used short bursts of swimming to move or to maintain position. It is therefore not surprising that their performance was not temperature-dependent because they used white muscle to swim. Although many authors have linked red muscle performance with water temperature, there is less evidence for temperature-dependence in white muscle (Rome et al. 1992; Rome et al. 1990). For example, burst velocities in Atlantic salmon (*Salmo salar*) are not temperature-dependent (Booth et al. 1997), and temperature-dependence in brassy minnow swimming performance decreased dramatically at sprint speeds. In adult brown trout, white muscle was recruited to supplement red muscle at the same swimming speed, regardless of temperature (Day and Butler 2005). The long acclimation period between the adjustment of holding temperatures and the endurance trials (approximately 3 months) allowed the white muscle to acclimate to the water temperatures and function at peak efficiency in the trials.

Common shiner swimming performance was not temperature-dependent; this may have been due to the fact that they were acclimated to low temperatures for a long period of time (approximately 7 months) or because they are more tolerant of lower temperatures. It is possible that common shiners have a large zone of thermal insensitivity in which physiological abilities and metabolic functions remain constant; this has been documented for largescale suckers (Catostomus macrocheilus) between 10 and 16°C (Kolok et al. 1993). This would be an advantage in transition-zone streams, where high water velocities are concurrent with low water temperatures. Because common shiners begin to make spawning migrations at 12°C (Raney 1940; Trial et al. 1983), the ability to swim quickly at lower water temperatures would allow them to migrate more efficiently. However, the number of nonperformers at low water velocity and low water temperatures does indicate the possibility of resting behaviors or station holding on the substrate at 10°C.

When using Peake's equation to obtain fish ladder design parameters for Arkansas darter passage, it was best to use the predicted 50th percentile values. We focused on the higher speeds because they are most relevant, and although estimates at 16 cm/s were higher than the observed medians, structural differences between the swimming flume and a concrete fishway justify the use of the predicted values. Because of their design, the walls of the flume's swimming chamber had a very

small boundary layer. The chamber is designed so that operators can produce uniform flow and prevent swimming fish from taking refuge in the boundary layer. Very few of the tested darters (2 out of 20) were able to hold position on the bottom of the flume at 16 cm/s. A concrete fishway would have a rougher surface that would likely allow darters to hold position on the bottom at 16 cm/s. Therefore, they would not be required to ascend the entire span in a single sprint. Small benthic fishes such as the greenside darter (*Etheostoma blennoides*) have been observed using the boundary layer in order to ascend a Denil fishway (Bunt et al. 2001).

Because Arkansas darters are bottom-oriented fish, it is possible that a weir-type structure is incompatible with their swimming behaviors. If a bottom-oriented fish approaches the base of a submerged weir and tries to swim up it, its angle of approach to the high velocity current coming over the top of the structure is such that displacement is guaranteed. Multiple species of bottom-oriented fishes may experience similar problems with fishways involving weirs or baffles, even if the ability to jump is not required for successful use. For example, the swimming behaviors of adult white sturgeon (*Acipenser transmontanus*) were incompatible with the navigation of an experimental vertical-slot fishway. Fish did not swim up the current. Instead, they nosed along the baffle in the low velocity pool and entered the current from the side at the head of the baffle, thus becoming displaced (Cech, J. J. Jr., University of California at Davis, personal communication). Given the behaviors of the darters in these experiments, avoidance of weir-type structures and vertical slot fishways is probably prudent in watersheds containing benthic species.

Jumping Performance

Plains and transition-zone streams on Colorado's Front Range historically contained few permanent vertical barriers. Therefore, it appears that some of the resident fishes developed limited jumping abilities compared to fish in higher-gradient streams (e.g., salmonids). Those that can jump, such as brassy minnows, do jump well on a relative scale (i.e., brassy minnows jumped 2 to 3 body lengths at 17.5 and 25°C), but they are still limited by their small body size (a 10 cm brassy minnow would be limited by a 35-cm high obstacle). As a result, we cannot recommend the use of fish ladders that require jumping in these systems.

In addition, it appears that the plunge pool depth below a fishway is extremely important in determining its effectiveness. Plains streams were often extremely shallow, and our results showed that fish were more likely to negotiate weirs from a plunge pool with a depth of 20 cm. Brassy minnows may have been reluctant to jump from the 30-cm plunge pools for a number of reasons. First, their incentive to leave the 10 and 20 cm plunge pools may have been driven by the density of fish in the lower pool. It is possible that there was sufficient volume in the 30 cm pool to prevent crowding or intense interspecific interactions between the fish. Second, the fish may have perceived the 30-cm deep pool as a refuge habitat. Plains species take refuge in deep pools during the dry season (Labbe and Fausch 2000; Scheurer et al. 2003b). Finally, there may be a behavioral constraint that prevented these fish from swimming to the surface from a deep pool; they may remain in the middle or bottom of the water column to reduce the risk of predation by birds (Schlosser 1988). Given this, a deep plunge pool below a fishway may not be ideal for facilitating passage of brassy minnows. Deep pools may inhibit movement, and they also harbor nonnative predators (such as largemouth bass, *Micropterus salmoides*) that prefer lentic habitats. Shallow water below a fishway would be advantageous to plains fishes because they are equipped

to move through shallow areas whereas their nonnative competitors and predators may be less capable of doing so. Conversely, exceptionally shallow water below a fishway could also inhibit movement. Darters held in 5 cm plunge pools made very few attempts to move upstream across the weir. It is possible that cover-seeking behaviors prevented them from moving. If it were absolutely necessary to use very shallow plunge pools, provision of cover such as embedded cobble would probably facilitate passage. Plunge pool depth was not as important in determining common shiner jumping success. At a weir height of 5 cm, deeper plunge pools increased success, possibly because fish swam up the water column from below. At a weir height of 10 cm, too few fish jumped to make conclusions about the role of plunge pool depth in determining success.

Applicability of Results

Laboratory results do not always translate well to field situations. In this case, we subjected the fish to forced swimming trials to test physiological limits of swimming ability. It is possible that small fishes would avoid fishways that contained high water velocities even if their abilities would permit them to use the structure. However, given their behavior in the artificial waterfalls (i.e., constant attempts to move upstream by at least some individuals in the trial group), this seems unlikely. Migrations for spawning and avoidance of suboptimal conditions are probably directed enough that fish will attempt to use any structure.

Implications for Fish Ladder Design in Colorado

If one is to successfully design fish ladders for non-salmonid fishes, it is necessary to identify the species and size of migrating fish, quantify swimming ability, and design a structure based on this information (Mallen-Cooper 1999). Because of their small body size and short life span, age class and size of migrating plains fishes are less of an issue in these systems. Connectivity is necessary for persistence of brassy minnow (Scheurer et al. 2003b) and Arkansas darter (Labbe and Fausch 2000), and common shiners are known to make spawning migrations (Pflieger 1997; Raney 1940). We have successfully quantified the swimming abilities of these three species. We also argue that, because the swimming performance of most fishes is temperature-dependent, determining the timing of migration is also important. For example, brassy minnows in Colorado spawn in the spring, so it is likely that they migrate when water temperatures are approximately 10°C. Although it is unknown if they do migrate to spawn, it is still best to design fish ladders using conservative estimates of swimming performance (typically values for swimming at lower temperatures) to increase the probability of creating a successful structure. These fish may need to move more often in the summer and fall when habitats are drying, flooding events are more common, and water temperatures are higher, but if fishways were designed based on the swimming performance at these higher temperatures, creation of a seasonal barrier would be a substantial risk. Currently, too little is known about the ecology of these and other plains fishes to incorporate anything other than the most conservative estimate of swimming performance into fish ladder design.

Our results suggest that fish ladders should be designed with sprint speeds in mind. However, the small body size of Colorado's plains fishes imposes some important limitations. For example, in a fishway, sections with a water velocity of 30 cm/s could be 3 m long to accommodate brassy

minnows at 10°C, 25 m long for brassy minnows at 17.5°C, 0.2 m long for Arkansas darters at any temperature, and 11 m long for common shiners (Figures 2.17–2.19). These design specifications are based on continuous swimming ability, and they were created on the assumption that Arkansas darters would not be able to hold position on the bottom of the fishway. However, if the fishway were designed so that the roughness coefficient was increased (i.e., by using rough aggregates or by adding gravel to the surface of the fishway), darters would likely be able to cope with continuous spans longer than 0.2 m. Designing fish passage structures for small-bodied fishes involves some novel challenges; characteristics such as thickness of the boundary layer and the presence of small-scale turbulence will greatly affect the effectiveness of the structure.

Vertical slot fishways may be useful in plains streams for a number of reasons. Because the baffles forming the slot are the same height as the fishway itself, these structures continue to function regardless of water level (Clay 1995). Use of vertical slot fishways has been successful in facilitating movement of non-salmonid fishes in North America and Europe. Furthermore, a fish will attempt to use this type of structure regardless of its preferred position in the water column (Clay 1995). On the other hand, vertical slot fishways require high maintenance as the narrow slots can become fouled with debris (Clay 1995). Furthermore, the velocity of the water moving through the slots is often quite high. For example, the velocities in the Torrumbarry Weir vertical slot fishway in the Murray River, Australia can exceed 2 m/s (Mallen-Cooper 1999). These speeds would preclude movement of all of our tested species. Given this, perhaps a rock ramp fishway would be best for allowing passage of small plains fishes. These structures contain a number of water depths, velocities, and velocity refuges. Furthermore, small fishes (TL ≥ 30 mm) in the Murray-Darling river basin in Australia apparently have had some success in navigating structures such as these (Harris et al. 1998). They are also advantageous in that they lack a well-defined or constricted entrance. Smaller fishes attempting to use more intensively engineered fishways risk predation or exclusion due to the presence of larger fishes (Bunt et al. 1998). Like vertical-slot fishways, rock ramps can also accommodate a large number of water levels but require less maintenance than vertical slot fishways. A rock ramp would have lower water velocities near the edges of the structure; even though the water would be shallow here, it would not prevent passage of small-bodied target species.

The presence of flow was enough of a stimulus to encourage fish to attempt "upstream" movement in the artificial waterfalls. Furthermore, the fact that plains fishes use temporary habitats suggests that a large attraction flow (Clay 1995) may be less of an issue with these small-bodied species than it is with salmonids. However, this has not been experimentally tested and requires further exploration.

It appears that the swimming abilities of the brassy minnow and the Arkansas darter are similar to other species within their respective guilds. Given this, future research should focus on relatively unstudied guilds (e.g., benthic minnows and surface oriented fishes). A few representative species from each of these guilds would probably provide ample information for appropriate fish ladder design. We should also tailor our experiments to explore factors that are relevant to the studied fish. For example, the thickness of the boundary layer is important to benthic fishes, and its role should be explored in experiments involving them. Studying guilds of fishes would also allow us to identify the weakest swimmers in local fish assemblages. We should be using data on these weak swimmers for fish ladder design if we favor the conservative approach to building fishways.

Although the results of our jumping experiments are not surprising, we now have unequivocal evidence that a small obstruction (5-10 cm) can greatly restrict the movement of three state-threatened fish species. They also highlighted the importance of considering behavior in conjunction with swimming and jumping ability. We recommend that future studies take this into consideration. Behavioral observations can identify incompatibilities between fish behavior and fishway structures that could not be identified with traditional experiments that employ forced swimming. Our study also presents some new questions that need to be explored. The importance of boundary layer thickness, small-scale turbulence, and attraction flow should all be investigated, as they appear to be important in determining passage success. Knowledge of these roles would allow us to further refine fish ladder designs and increase fish ladder effectiveness.

Tables

Table 2.1. — The threatened, endangered and special concern fish species of Colorado's plains and transition-zone streams. The conservation status of each species is also listed: ST = state threatened, SE = state endangered, SC = species of special concern (not a formal listing category). Data from the Colorado Division of Wildlife (http://www.wildlife.state.co.us/wildlifespecies/speciesofconcern/fish/fishofconcern/htm).

Species	Scientific Name	Status
Arkansas darter	*Etheostoma cragini*	ST
brassy minnow	*Hybognathus hankinsoni*	ST
common shiner	*Luxilus cornutus*	ST
flathead chub	*Platygobio gracilus*	SC
Iowa darter	*Etheostoma exile*	SC
lake chub	*Couesius plumbeus*	SE
northern redbelly dace	*Phoxinus eos*	SE
plains minnow	*Hybognathus placitus*	SE
orangethroat darter	*Etheostoma spectabile*	SC
Rio Grande chub	*Gila pandora*	SC
Rio Grande sucker	*Catostomus plebeius*	SE
southern redbelly dace	*Phoxinus erythrogaster*	SE
stonecat	*Noturus flavus*	SC
suckermouth minnow	*Phenacobius mirabilis*	SE

Table 2.2. — Fish species captured below and above a 40 cm log weir on Sagentobel Creek, Switzerland. All species listed were captured below the barrier. Small-bodied fishes are indicated with an asterisk (After Peter 1998).

Common Name	Species Name	Family	Found above barrier	TL ≤ 150 mm
bleak	*Alburnus alburnus*	Cyprinidae	N	*
European eel	*Anguilla anguilla*	Anguillidae	N	
barbel	*Barbus barbus*	Cyprinidae	N	
threespine stickleback	*Gasterosteus aculeatus*	Gasterosteidae	N	*
gudgeon	*Gobio gobio*	Cyprinidae	N	*
European chub	*Leuciscus cephalus*	Cyprinidae	N	
Eurasian minnow	*Phoxinus phoxinus*	Cyprinidae	Y	*
roach	*Rutilus rutilus*	Cyprinidae	N	
brown trout	*Salmo trutta*	Salmonidae	Y	
tench	*Tinca tinca*	Cyprinidae	N	

Table 2.3—Treatments for brassy minnow, Arkansas darter, and common shiner swimming performance. Fish at a given acclimation temperature swam at one of four randomly chosen velocities that bracketed their sustained, prolonged, and sprint swimming gaits. Brassy minnows held at 17.5 and 25°C were tested at higher velocities due to their improved performance at these temperatures. Arkansas darters performed similarly at all three test temperatures, and common shiners performed similarly at both test temperatures.

Temperatures	Velocities (cm/s)			
	Brassy minnow			
10°C	4	8	16	32
17.5°C	16	32	48	64
25°C	16	32	48	64
	Arkansas darter			
10, 17.5, and 25°C	4	8	16	32
	Common shiner			
10°C	16	32	48	64

Table 2.4.—The sample size at each plunge pool depth and weir height combination used in the jumping trials. Limited fish availability and random assignments prevented all of the trials from being replicated four times. Arkansas darter availability was extremely low, so fish were tested in groups of five. Their small size led to the use of shallower plunge pools.

Brassy minnow

Temperature	H_1 (cm)	H_2 (cm)				
		0	**5**	**10**	**15**	**20**
10	**10**	4	3	4	--	--
	20	3	4	4	--	--
	30	3	4	4	--	--
17.5	**10**	4	4	4	4	1
	20	4	4	4	4	4
	30	4	4	4	4	3
25	**10**	4	4	4	4	4
	20	4	4	2	4	4
	30	4	4	4	3	3

Arkansas darter

Temperature	H_1 (cm)	H_2 (cm)				
		0	**5**	**10**	**15**	**20**
10	**5**	4	4	--	--	--
	10	4	4	--	--	--
	20	4	4	--	--	--
17.5	**5**	4	4	--	--	--
	10	4	4	--	--	--
	20	4	4	--	--	--
25	**10**	1	2	--	--	--

Common shiner

Temperature	**H_1(cm)**	**H_2 (cm)**				
		0	**5**	**10**	**15**	**20**
10	**10**	4	4	4	2	--
	20	4	4	4	2	--
	30	4	3	4	3	--
17.5	**10**	4	4	4	4	4
	20	4	4	4	4	4
	30	4	4	4	4	4

Table 2.5 — Set of top candidate models and associated AICc values for brassy minnow jumping. The best model is indicated with bold text. The second best models have little support according to their Δ AICc values. The interaction term between H_1 and H_2 was important at 17.5 and 25°C but not at 10°C.

Temperature	Variables	AICc	Δ AICc
10°C	**H_1, H_2**	**107**	**0**
	H_1, H_2, $H_1 \text{x} H_2$	111.2	4.2
17.5°C	**H_1, H_2, $H_1 \text{x} H_2$**	**147.4**	**0**
	H_1, H_2	151.8	4.4
25°C	**H_1, H_2, $H_1 \text{x} H_2$**	**55.9**	**0**

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FISH BARRIERS AND SMALL PLAINS FISHES: FISHWAY
DESIGN RECOMMENDATIONS AND THE IMPACT OF
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JUNE 2007

Table 2.6 — Set of top candidate models and associated AICc values for Arkansas darter jumping. The best model is indicated with bold text. The second best models have very little support according to their Δ AICc values. The interaction term between H_1 and H_2 was important at 10°C but not at 17.5°C. Small sample sizes at 25°C prevented me from obtaining any resolution among competing models, so trials run at this temperature were not included in the results.

Temperature	Variables	AICc	Δ AICc
	H_1, H_2	**0.8**	**0**
10°C	H_1, H_2, H_1xH_2	1	0.2
17.5°C	**H_2**	**102.9**	**0**
	PP, TL, H_1, H_2, H_1xH_2	5.5	0
25°C	TL, H_1, H_2, H_1xH_2	5.5	0
	H_1, H_2, H_1xH_2	6.9	1.4

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Table 2.7 — Set of top candidate models and associated AICc values for common shiner jumping. The best model is indicated with bold text. Although resolution between candidate models is poor, estimated success probabilities were similar with all of the models. At 10°C, the data supported models containing weir height (H_2), total length (TL), and plunge pool depth (H_1). At 17.5°C, there was support in the data for an interaction term between weir height and plunge pool depth.

Temperature	Variables	AICc	Δ AICc
	H_2	**172.1**	**0**
10°C	H_2, TL	172.7	0.5
	H_1, H_2	172.8	0.6
	H_1H_2, H_1xH_2	**161.2**	**0**
17.5°C			
	H_2	162.5	1.3

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Table 2.8 — Observed and predicted values of brassy minnow swimming endurance at three temperatures (degrees Celsius) and six velocities (cm/s). The regression equation overestimated swimming ability for many of the high-velocity trials that would be practical for fish ladder design. Therefore, observed median values provide a more conservative estimate of the sprinting abilities of these fish, and the regression output should be used with caution.

Velocity (cm/s)	Swimming Endurance (minutes)					
	Observed Medians			Predicted 50% Success		
	10°	17.5°	25°	10°	17.5°	25°
4	33	--	--	171	1076	6745
8	6	--	--	48	301	1886
16	3	198	198	13	84	527
32	2	198	198	4	24	147
48	--	3	198	2	11	70
64	--	1	2	1	7	41

Table 2.9 — Observed and predicted values of Arkansas darter swimming endurance at three temperatures (degrees Celsius) and four velocities (cm/s). The regression equation overestimated swimming ability for some of the high-velocity trials that would be practical for fish ladder design. Observed median values provide a more conservative estimate of the prolonged swimming abilities of these fish, and the regression output should be used with caution.

Velocity (cm/s)	Swimming Endurance (minutes)					
	Observed Medians			Predicted 50% Success		
	10°	17.5°	25°	10°	17.5°	25°
4	200	200	200	22907	26268	30121
8	200	200	200	370	392	415
16	1	1	2	5.96	6	6
32	0.3	0.2	0.2	0.1	0.1	0.1

Table 2.10 — Observed and predicted values of common shiner swimming endurance at two temperatures (degrees Celsius) and four velocities (cm/s). The regression equation accurately predicted swimming ability for some of the high-velocity trials that would be practical for fish ladder design. However, the regression output should be used with caution.

Velocity (cm/s)	Swimming Endurance (minutes)			
	Observed Medians		**Predicted 50% Success**	
	10°	17.5°	10°	17.5°
16	188	200	4022	4022
32	79	8	19	19
48	0.7	3	2	2
64	0.3	0.8	0.5	0.5

Table 2.11 — The swimming ability of 15 other North American minnows compared to that of the brassy minnow held at 3 temperatures. All other species swam at 20.5°C. SL = standard length, swimming velocity is in body lengths per second (BL/s), and mean endurance is in seconds. Data for all other species are from Billman and Pyron 2005. Standard deviations are presented for the Billman and Pyron data, and 95 percent confidence intervals are presented for the brassy minnow data.

Common Name	Species Name	Mean SL (mm)	Velocity (BL/s)	Mean endurance (67 cm/s)	SD or CI
silverjaw minnow	*Ericymba buccata*	51.7	13.0	8.5	6.1
redside dace	*Clinostomus elongatus*	39.9	16.8	10.5	9.5
rosyface shiner	*Notropis rubellus*	48.8	13.7	11.2	9.1
striped shiner	*Luxilus chrysocephalus*	41.5	16.1	13.1	9.9
sand shiner	*Notropis stramineus*	44.3	15.1	17.1	23.1
pearl dace	*Margariscus margarita*	39.3	17.0	17.5	17.8
bluntnose minnow	*Pimephales notatus*	45.1	14.9	17.7	14.9
spotfin shiner	*Cyprinella spiloptera*	44.9	14.9	19.3	20.1
fathead minnow	*Pimephales promelas*	41	16.3	20.1	14.7
creek chub	*Semotilus atromaculatus*	40	16.8	24.8	24.4
central stoneroller	*Campostoma anomalum*	38.9	17.2	33.6	19.3
longnose dace	*Rhinchthys cataractae*	39.4	17.0	51.8	53.0
blacknose dace	*Rhinichthys atratulus*	47.4	14.1	52.1	45.3
red shiner	*Luxilus cornutus*	47.6	14.1	57.2	55.2
hornyhead chub	*Nocomis biguttatus*	44.9	14.9	66.1	55.5
brassy minnow, 10°C	*Hybognathus hankinsoni*	38.7	16.5	63.0	(23, 172)
brassy minnow,17.5°C	*Hybognathus hankinsoni*	38.7	16.5	73.8	(64, 87)
brassy minnow, 25°C	*Hybognathus hankinsoni*	38.7	16.5	104.4	(74, 175)

Figures
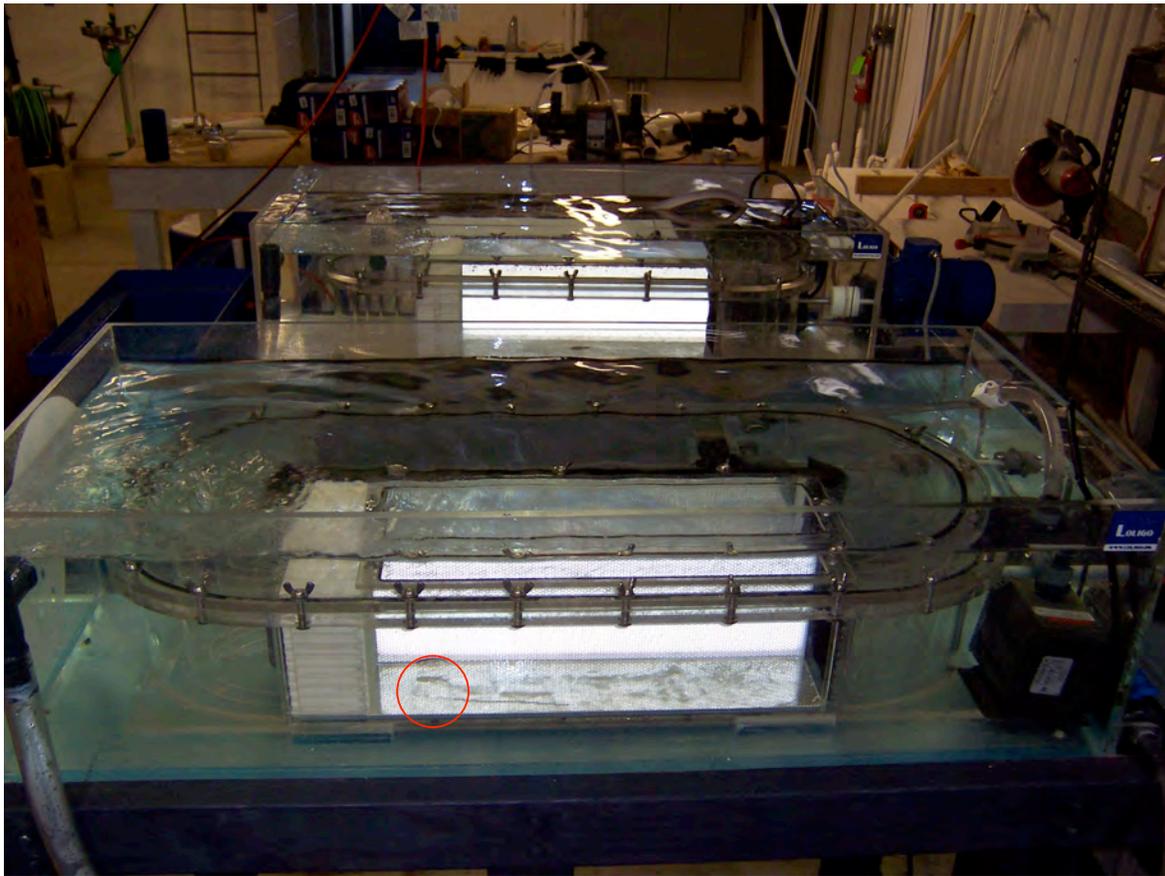
Figure 2.1 — Two Loligo Model 32 swim tunnels. A fish (circled in red) is placed in the white swimming chamber (height = 13.75 cm, length = 45.75 cm, and width = 14 cm), and it must swim to hold its position against the current in the apparatus.

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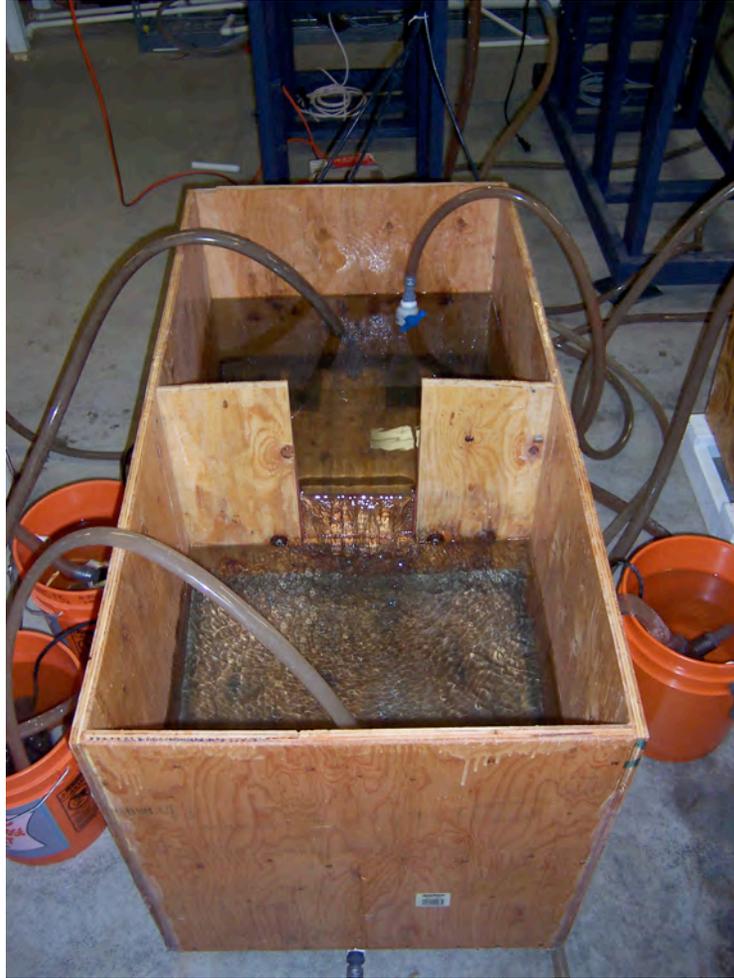
Figure 2.2 — A Kondratieff-type artificial waterfall used to test the jumping ability of small fishes. The orange buckets contain submersible pumps that pumped water from the lower chamber (bottom) into the upper chamber (top). The water then flowed over the weir back into the lower chamber.

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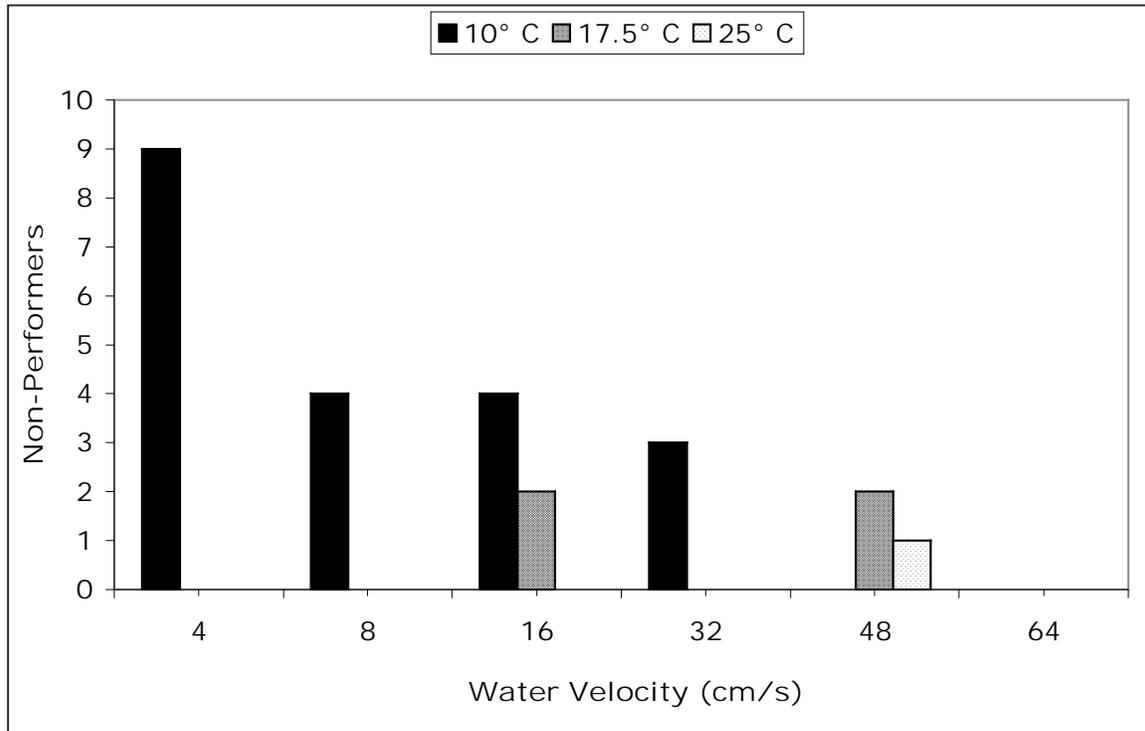
Figure 2.3 — Number of non-performing brassy minnows as a function of temperature and water velocity. Fish held at 10°C swam at speeds from 4 to 32 cm/s, and those held at higher temperatures swam at speeds from 16 to 64 cm/s.
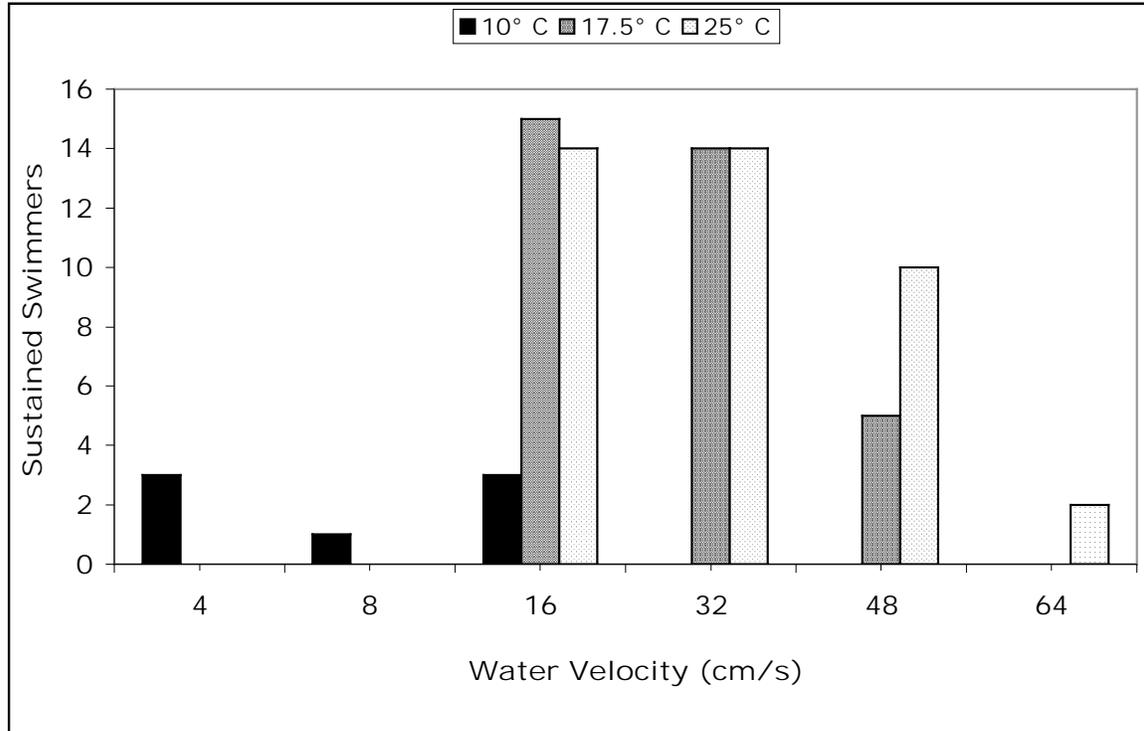
Figure 2.4 — Number of sustained swimmers (brassy minnows) as a function of velocity and water temperature. Fish held at 10°C swam at speeds from 4 to 32 cm/s, and those held at the two higher temperatures swam at speeds from 16 to 64 cm/s.
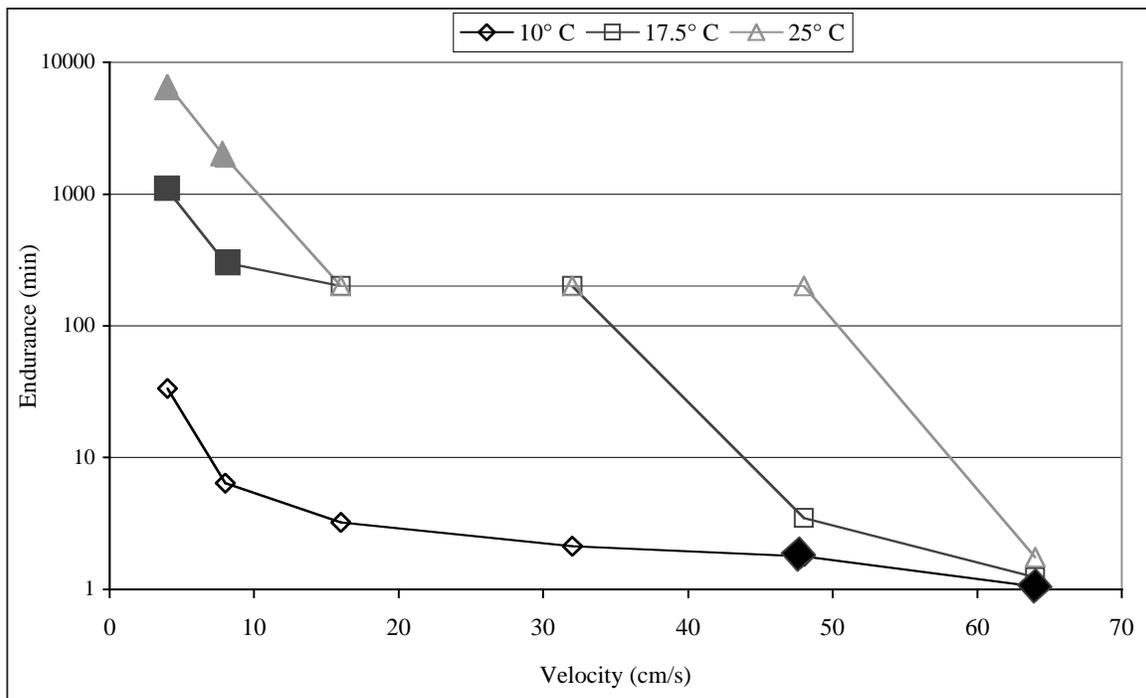
Figure 2.5 — Brassy minnow swimming endurance as a function of temperature and water velocity. At each velocity, brassy minnows tested at 10°C swam for shorter intervals than fish tested at the two higher temperatures. Temperature dependence was pronounced at sustained and prolonged swimming speeds, but it was minimal at sprint speeds. Large, closed symbols indicate the use of estimated values from the regression equation (fish were not tested at these combinations).
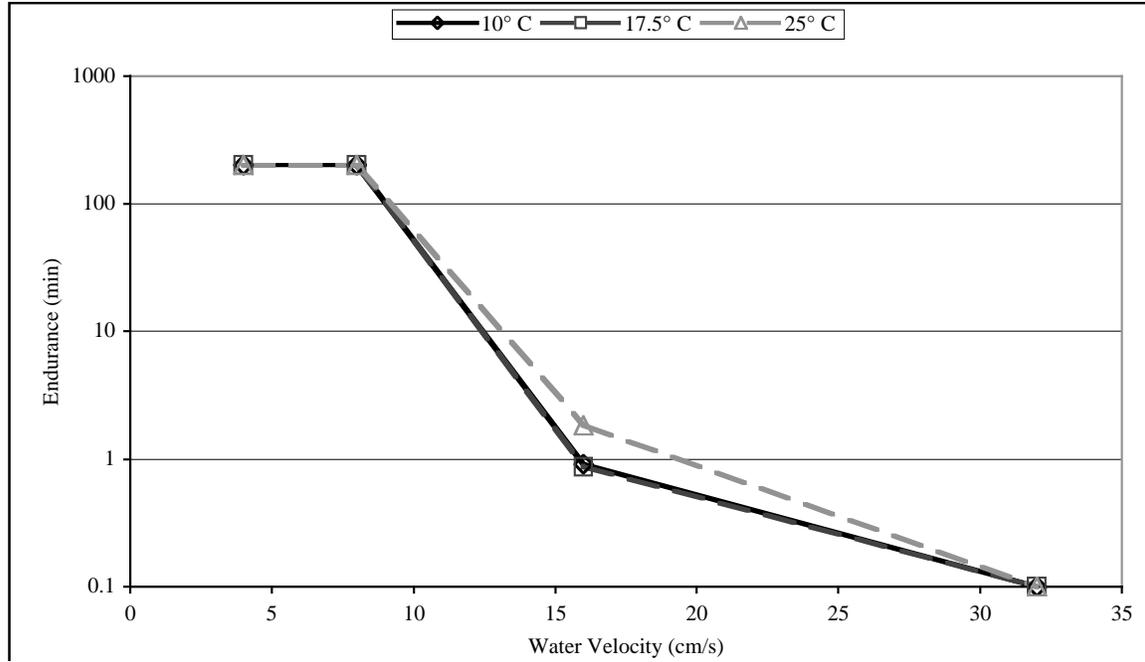
Figure 2.6 — Arkansas darter swimming endurance as a function of trial speed and temperature. Only trial velocity significantly affected swimming endurance. Observed medians were plotted at 4 and 8 cm/s to provide resolution on the graph, and observed medians were plotted at 16 cm/s because they were a more conservative estimate of swimming ability.
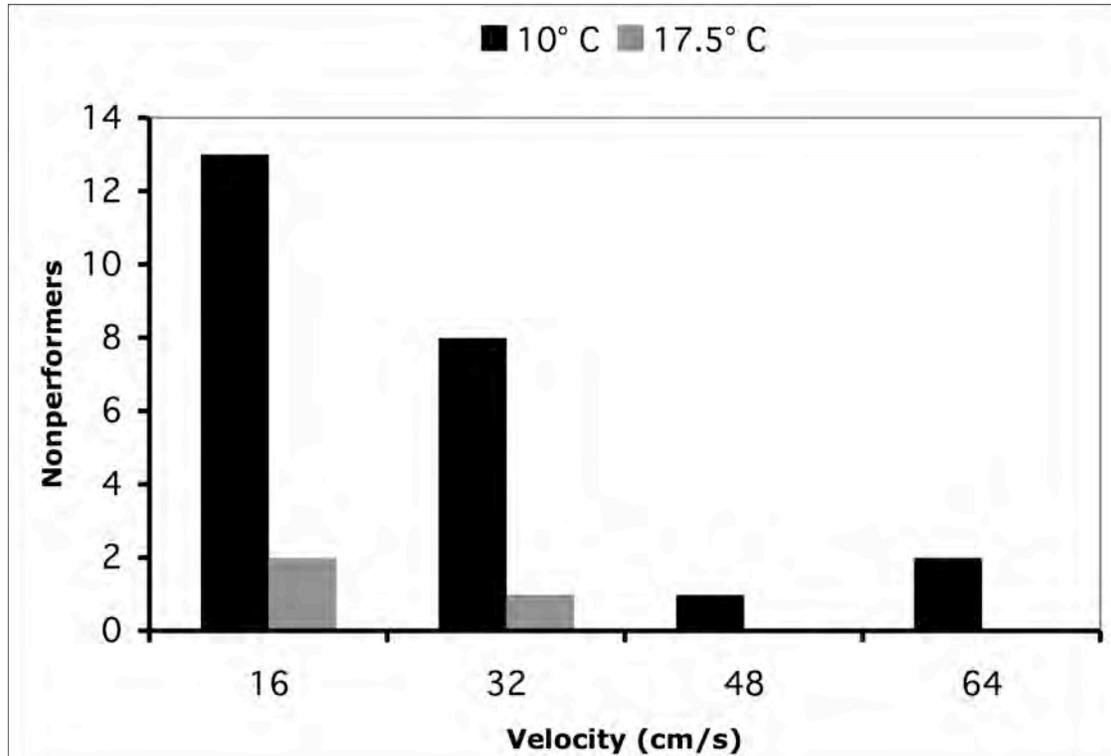
Figure 2.7 — Number of nonperforming common shiners at 10 and 17.5°C. Like brassy minnows, common shiners were less likely to swim at lower speeds at the lower water temperature.
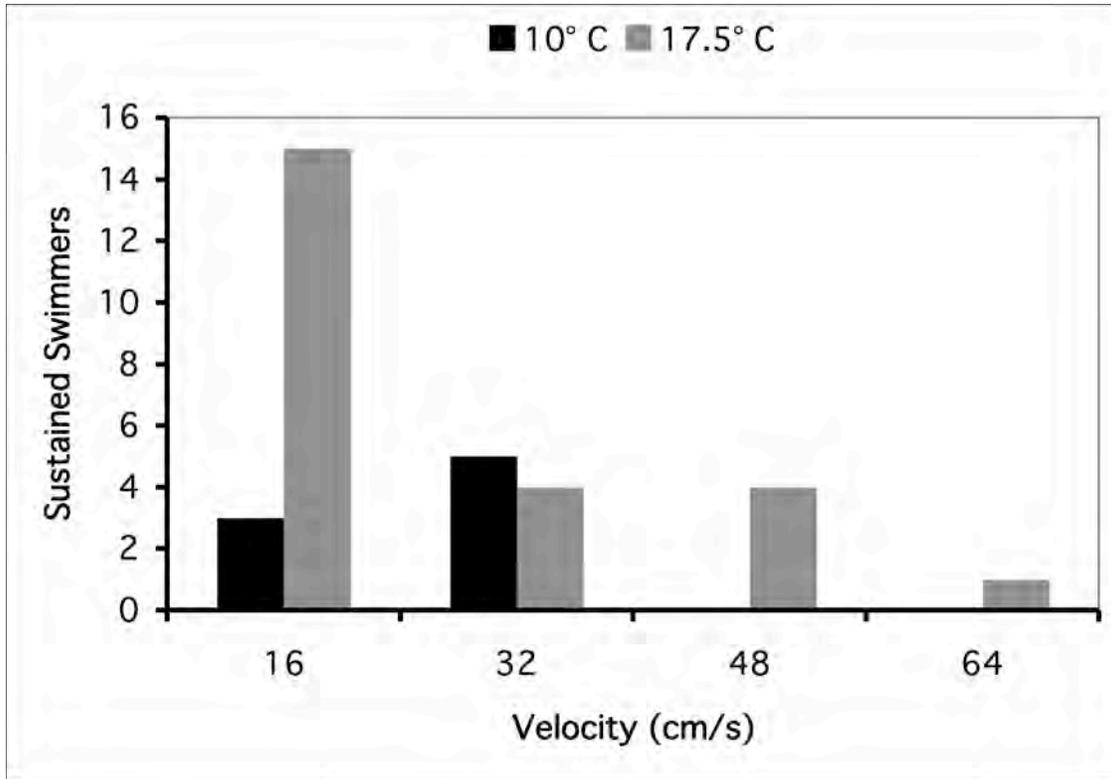
Figure 2.8 — Number of sustained swimmers (common shiners) as a function of velocity and water temperature. Fish held at 10 and 17.5°C swam at speeds from 16 to 64 cm/s.
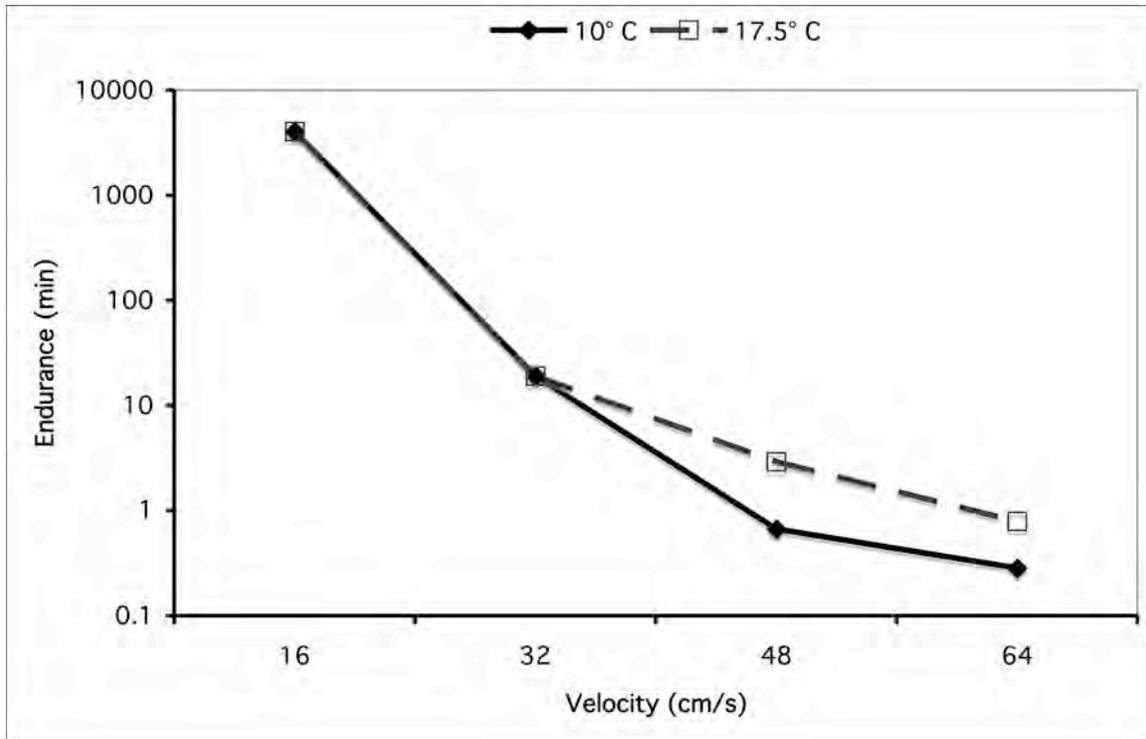
Figure 2.9 — Common shiner swimming endurance as a function of trial speed and temperature. Only trial velocity, fish total length, and photoperiod significantly affected swimming endurance. Predicted medians were plotted at 16 and 32 cm/s, but observed medians were plotted at 48 and 64 cm/s because they were a more conservative estimate of swimming ability.
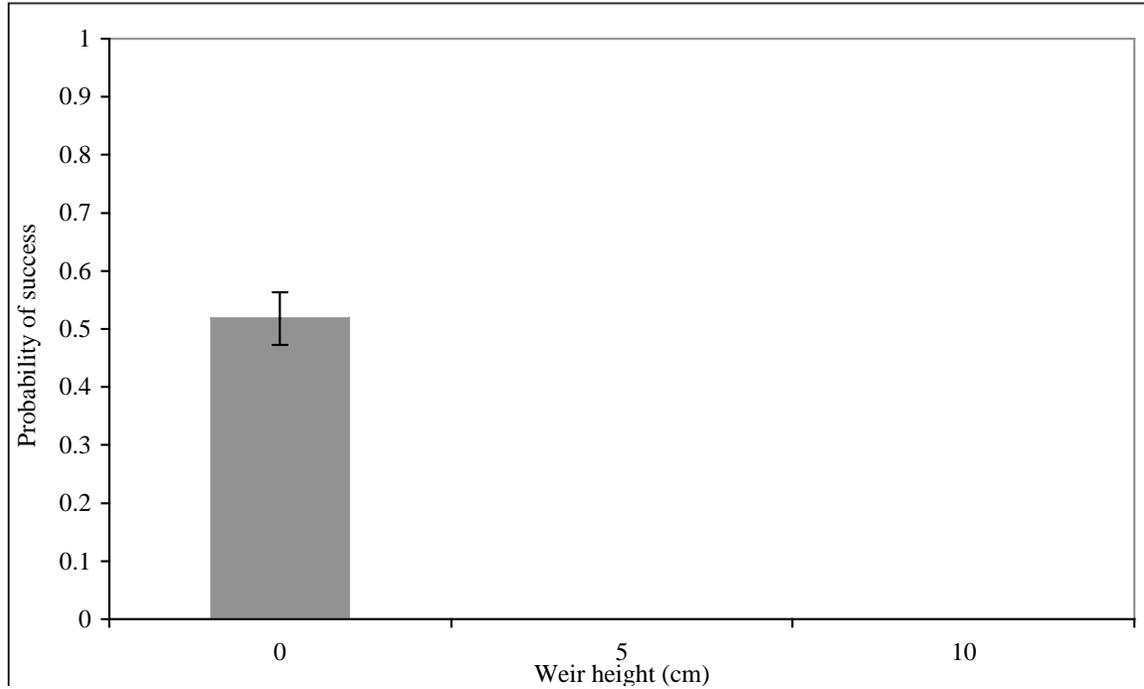
Figure 2.10 — Probability of brassy minnow jumping success as a function of weir height at 10°C. Success was defined as movement into the upper chamber. Plunge pool depth did not affect success probability, and no fish jumped over weirs that were greater than 0 cm in height. Even with a weir height of 0 cm, only about 50 percent of the brassy minnows moved into the upper chamber.
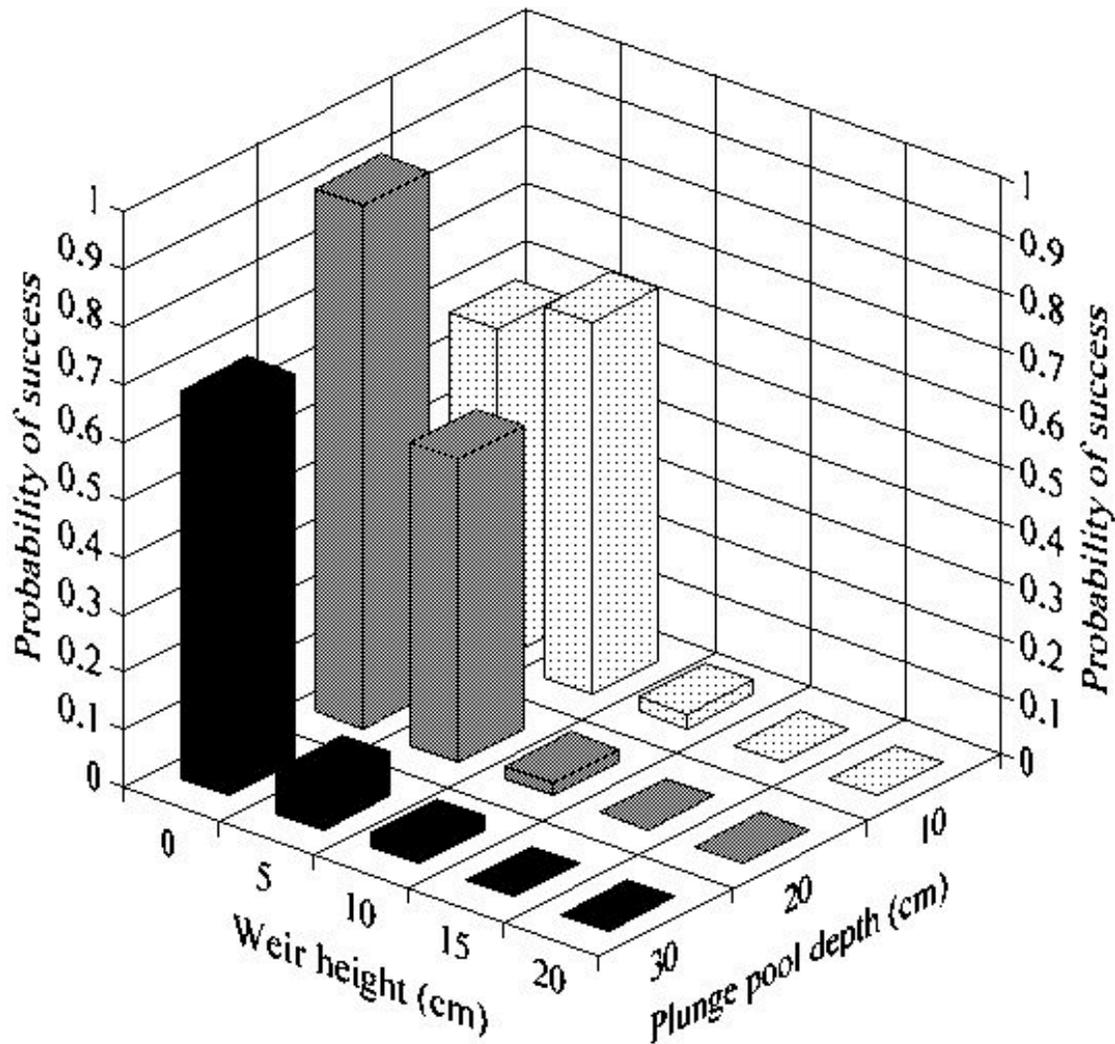
Figure 2.11 — Probability of brassy minnow jumping success as a function of weir height and plunge pool depth at 17.5°C. Success was defined as movement into the upper chamber. Fish were more likely to move upstream from shallower plunge pools, and they jumped a maximum height of 10 cm. At a weir height of 0 cm, 55-85 percent of the brassy minnows moved into the upper chamber.
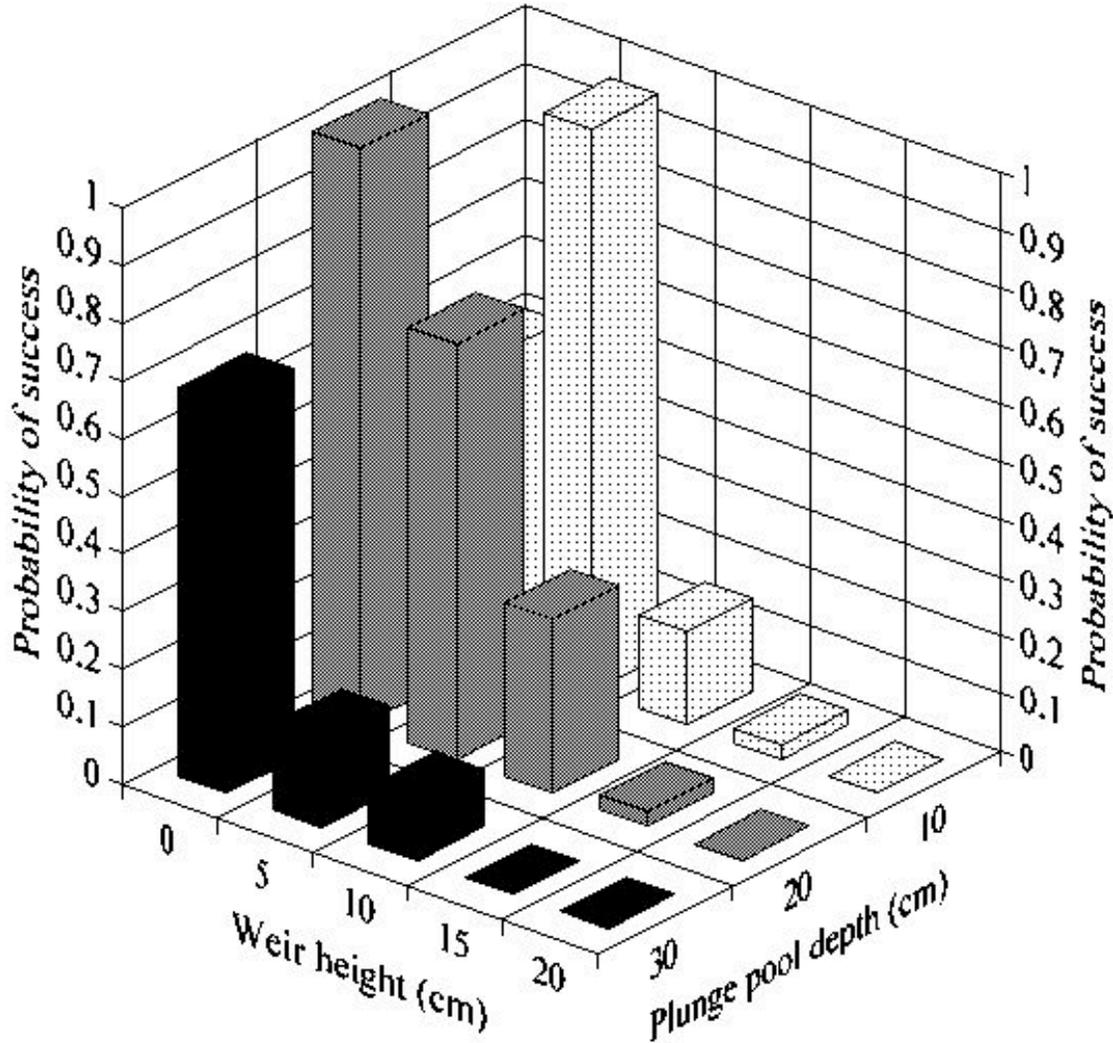
Figure 2.12 — Probability of brassy minnow jumping success as a function of weir height and plunge pool depth at 25°C. Success was defined as movement into the upper chamber. Fish were more likely to move upstream from shallower plunge pools, and they jumped a maximum height of 15 cm. At a weir height of 0 cm, 45-95 percent of the brassy minnows moved into the upper chamber.
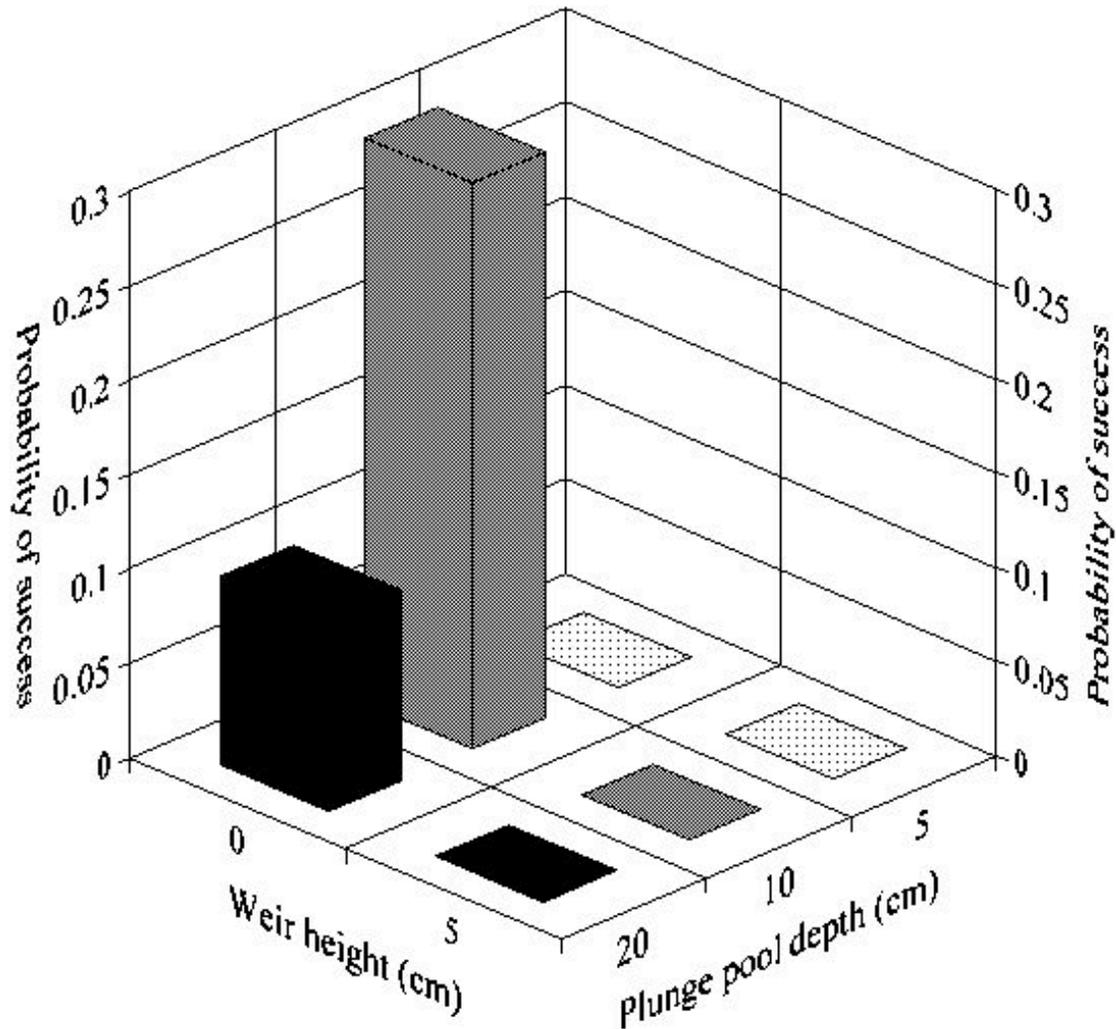
Figure 2.13 — Probability of Arkansas darter jumping success as a function of weir height and plunge pool depth at 10°C. Success was defined as movement into the upper chamber. Fish were more likely to move upstream from 10-cm plunge pools, and they did not move past a weir with a height greater than 0 cm. At a weir height of 0 cm, a maximum of 27 percent of Arkansas darters moved into the upper chamber.
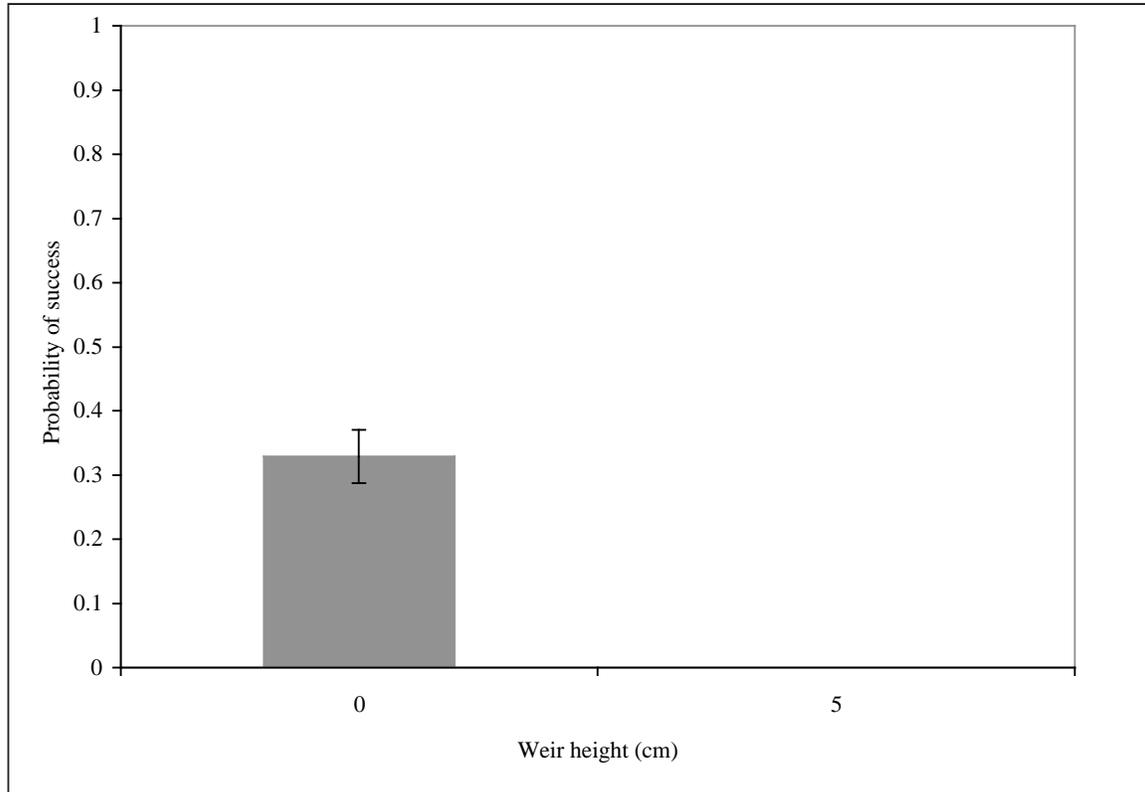
Figure 2.14 — Probability of Arkansas darter jumping success as a function of weir height at 17.5°C. Success was defined as movement into the upper chamber. Plunge pool depth had no effect on the probability of success, and fish did not move past a weir with a height greater than 0 cm. At a weir height of 0 cm, a maximum of 32 percent of Arkansas darters moved into the upper chamber.
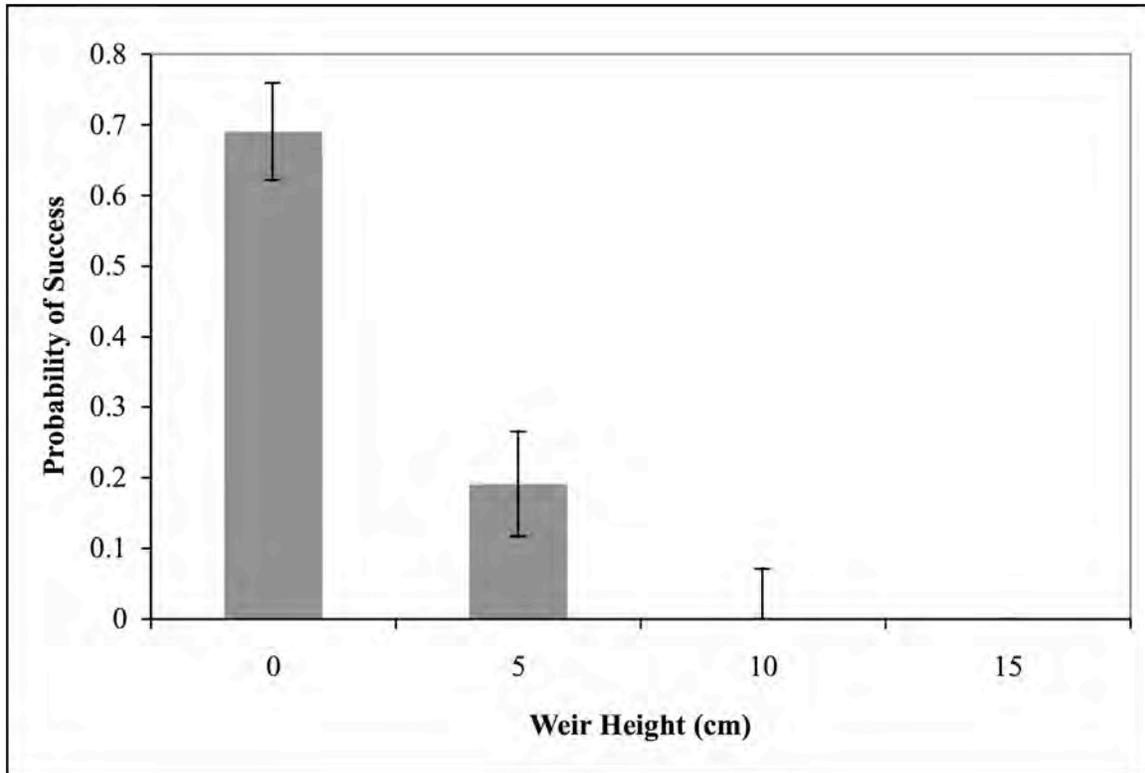
Figure 2.15.—Probability of common shiner jumping success as a function of weir height at 10°C.   Success was defined as movement into the upper chamber. A weir height of 5 cm restricted upstream movement, and a weir height of 10 cm eliminated it.

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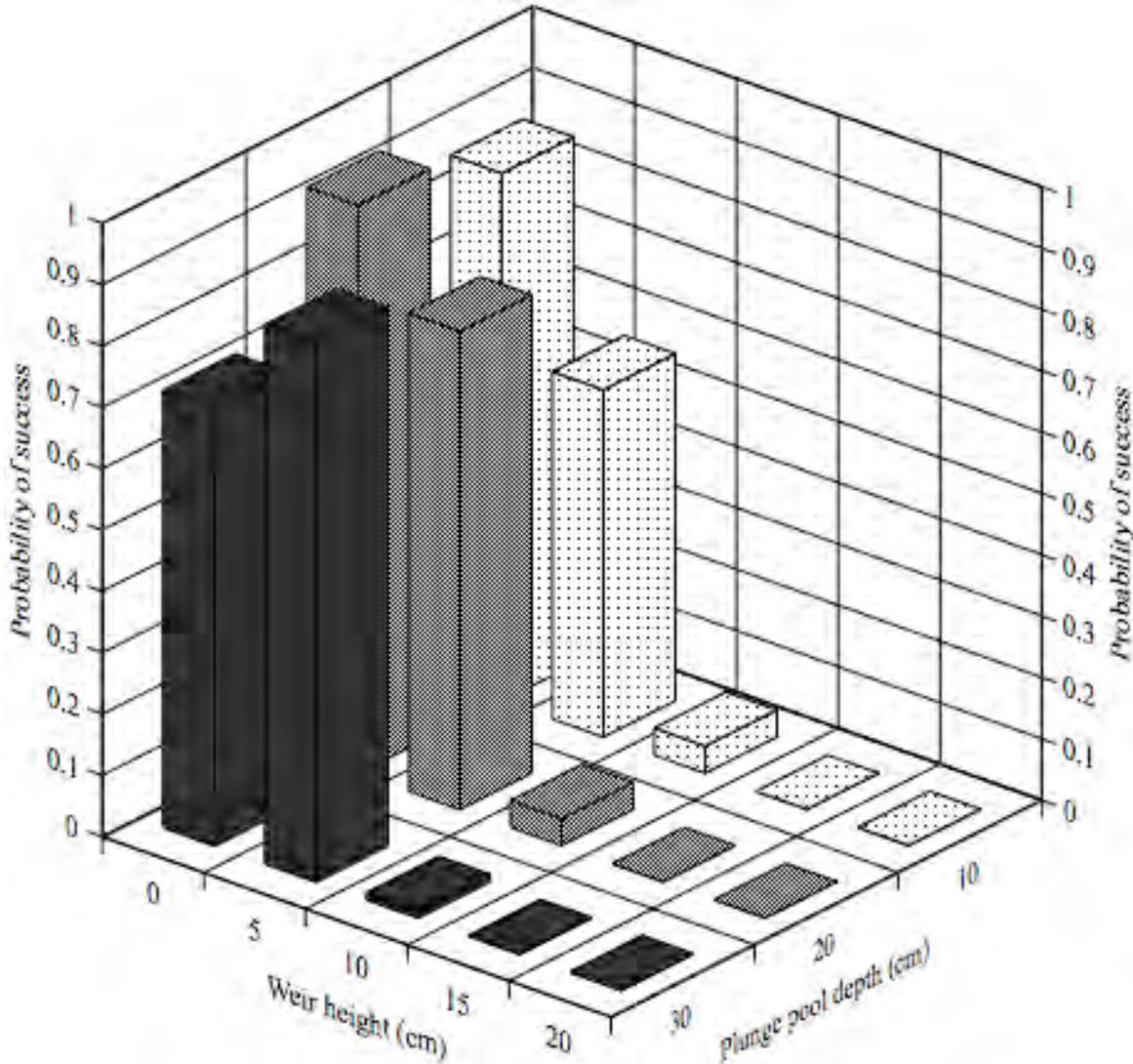
Figure 2.16 — Probability of common shiner jumping success as a function of weir height at 17.5°C. Success was defined as movement into the upper chamber. Both weir height and plunge pool depth had an effect on the probability of success, and AICc model selection indicated the presence of an interaction between H_1 and H_2. Fish did not move past a weir with a height greater than 10 cm. A weir height of 10 cm essentially eliminated movement into the upper chamber.

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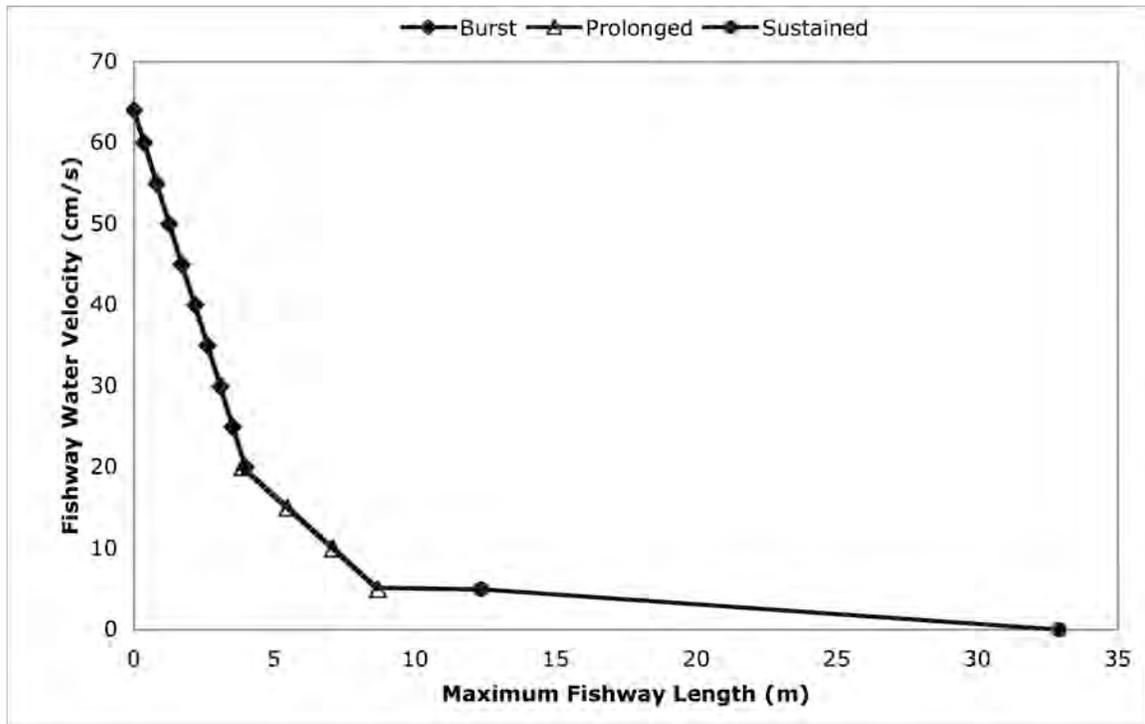
Figure 2.17 — Maximum allowable distances between velocity refuges and fishway water velocities for brassy minnows swimming at burst, prolonged, and sustained speeds at 10°C. Because endurance values were not observed for 64 cm/s, values estimated from the survival analysis were used. The observed median values for endurance at 32 and 8 cm/s were used to construct the prolonged and sustained portions of the curve, respectively. Any combination of distance between velocity refuges and water velocity underneath the curve is acceptable for allowing upstream passage of brassy minnows.

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Figure 2.18 — Maximum allowable distances between velocity refuges and water velocities for Arkansas darters swimming at burst and prolonged speeds at 17.5°C. The observed median for endurance at 16 cm/s was used to construct the prolonged swimming portion of the curve, and the regression equation prediction for 32 cm/s was used to construct the burst swimming portion of the curve (see results for details). The sustained swimming portion of this curve was omitted in order to retain some resolution at the burst speed scale. Any combination of distance between velocity refuges and water velocity under the curve is acceptable for allowing Arkansas darter passage.

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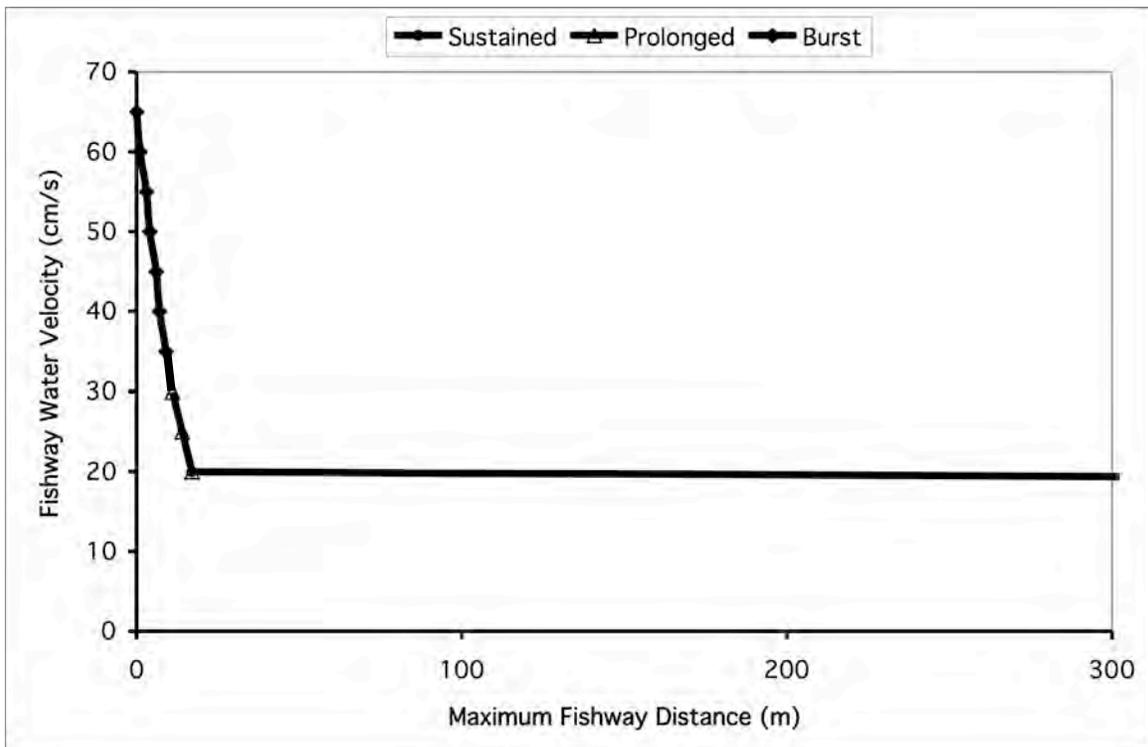
Figure 2.19 — Maximum allowable distances between velocity refuges and water velocities for common shiners swimming at burst, sustained, and prolonged speeds at 10 or 17.5°C. The predicted median for endurance at 16 cm/s was used to construct the prolonged swimming portion of the curve, and the observed median endurances at 48 and 64 cm/s were used to construct the prolonged and burst swimming portions of the curve (see results for details). Any combination of distance between velocity refuges and water velocity under the curve is acceptable for allowing common shiner passage.

A METHOD FOR MONITORING MOVEMENTS OF SMALL FISHES ACROSS INSTREAM STRUCTURES

Introduction

Streams in the urban corridor along Colorado's Front Range have been extensively modified by urban and agricultural land uses. These modifications include a damping of flow regimes, water withdrawals, pollution, and introduction of nonnative species (Rabeni 1996). The loss of stream connectivity due to thousands of impoundments, grade-control structures, and road crossings is one of the most significant human impacts on Front Range streams. Data on the actual number of structures in Front Range streams are rare. However, the prevalence of irrigation ditches and reservoirs in these systems provides some idea of the extent of fragmentation. For example, the St. Vrain River, a tributary to the South Platte, is 97 km long, and it drains an area of 1375 km^2 (Gerlek 1977). This basin contains 55 reservoirs that rely upon water diverted from its streams, and in 1970, the state engineer's office reported that 45 ditches diverted water from the St. Vrain and its tributaries (Gerlek 1977). Because at least some of these diversions can be assumed to involve instream structures, this stream has been changed from a longitudinally continuous system to a series of segments.

Removal of these structures is not feasible because they are an essential part of the infrastructure on the Front Range. Instream structures provide benefits ranging from diversion of irrigation water to grade control. The need for grade control in urban streams is fairly common, because urban streams are often channelized and straightened so that they will move water more efficiently during episodes of high flow, thus preventing property damage. However, when streams are straightened in this manner, grade-control structures are also necessary to prevent increased stream energy from causing downcutting and subsequent land loss. Grade control in urban streams is often achieved by installing vertical drop structures. Though these structures achieve the grade control and streambed stability objectives, they effectively fragment stream systems. Usually, the ecological needs of aquatic organisms are not considered when these structures are installed, and few studies have been conducted to determine their effects on local assemblages.

Because Colorado's Front Range streams contain thousands of grade-control and diversion structures, a need exists for information on their effects on the native fish community. Despite this abundance of structures, no attempts have been made to monitor movement across them or analyze their impact on fish assemblages. We studied two instream structures on Colorado's Front Range. The first is a vertical drop structure in Spring Creek in Fort Collins, CO, and the second is a rock ramp fishway in the St. Vrain River in Longmont, CO. The grade-control structure in Spring Creek is a vertical drop with an average height of 63 cm. The rock ramp fishway in the St. Vrain River presumably allows upstream movement of non-jumping fishes. Both of these structures were monitored for one year to determine their effects on the native fish communities.

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Rock ramp fishways have enormous potential to allow upstream movements of small fishes, because they incorporate multiple depths and water velocities, and non-jumping fishes can use them. These structures have shown some promise in allowing upstream movement of fishes as small as 30 mm in Australia (Harris et al. 1998). Because rock ramps tend to span the width of the stream, they are very difficult to monitor. Three monitoring methods have been used: comparing the fish community upstream and downstream of the structure (Harris et al. 1998), trapping migrating fish in weirs (Bunt et al. 2001), and tracking the movement of individual fishes (Bunt et al. 1999).

A comparison of the fish community immediately upstream and downstream of the structure is the least labor-intensive method of monitoring, but it provides the least amount of information. For example, if the stream habitat differs on either side of the structure, the fish community will obviously vary. This type of study will not clarify whether the difference in the assemblage is due to the habitat or to the presence of the structure. Trapping migrating fish in weirs is more accurate, but there are logistical problems associated with this approach. This technique is usually limited to studies involving large fishes because it is much easier to maintain a large-mesh weir in a stream. Small-mesh weirs are extremely difficult to anchor against stream current, debris removal is constantly necessary, and, in large warm water streams, it would be impractical to process every one of the thousands of small fish that pass through the study reach. The third method available involves tracking individually marked fish. Large fish can be marked with passive integrated transponder (PIT) tags and tracked with stationary antenna arrays (Aarestrup et al. 2003) or hand-held units (Quintella et al. 2005). Antenna arrays detect tagged individuals with remarkable accuracy: the flat-plate antenna installed at the Bonneville Dam First Powerhouse on the Columbia River tracked passing fish with 97% efficiency (Nunnallee et al. 1998). On the other hand, this method is costly: tags cost $2-5 each, and a flat-plate antenna array costs approximately $2000 (Greenberg and Giller 2000). It is also limited to fish larger than 100 mm TL due to the size of the tag. Therefore, smaller fishes must be individually marked with a different tagging system such as visual implant elastomers (VIE). Reaches above and below an obstacle are serially sampled through time to detect movement of marked fish between reaches within the study site. We evaluated the success of this relatively new marking method in tracking small fish movements across two instream structures on Colorado's Front Range.

Objectives

The goal of this phase of the project was to evaluate the effects of two of the common instream structures found in Colorado Front Range streams. These streams, which exist in highly-developed areas, are heavily modified yet continue to support (in most cases) a diverse native fish fauna. Our specific objective was the following:

• To investigate the movement of native eastern plains fishes past existing stream barriers and evaluate potential impacts to listed fish species.

Hypotheses

We formulated the following hypotheses with regards to fish passage at the two sites that we evaluated in this phase of the project:

- The rock-ramp fishway on the St. Vrain River will allow both downstream and upstream movement of native fishes.

- The drop structures on Spring Creek will allow downstream, but not upstream movement of native fishes.

Methods

Site and Reach Selection

Our first site was a rock ramp fishway on the St. Vrain River, Weld County, Colorado (**Figure 3.**1), and the second was a vertical drop structure (average drop = 63 cm) on Spring Creek, Larimer County, Colorado (**Figure 3.2**). We used mark-recapture techniques to track the movements of the individually marked fish. The two sites were sampled twice monthly every other month, weather permitting. The two-week interval between sampling occasions allowed enough time for the marked fish to mix evenly with the unmarked fish in the stream, yet it was a short enough time interval to maximize the recapture rates. Streams were sampled every other month so that we could examine both sites within the logistical limits of our project.

Our study segment on the St. Vrain consisted of a 100-m long reach below the rock ramp fishway and a 200-m long reach above it. Because of the width and water velocity of this stream, we were unable to place block nets in the water to separate the two reaches into smaller segments. Our study segment in Spring Creek consisted of three contiguous 100-m reaches below the drop structure and three contiguous 100-m reaches above it. This stream is much smaller than the St. Vrain (maximum width in the study segment = 5 m), and we were able to separate the two 300-m reaches into six 100-m sections during sampling. Sites were sampled by single pass electrofishing.

Fish Marking Techniques and Field Measurements

Captured fish were placed in an anesthetic bath containing tricaine methanesulfonate (MS-222, 30 mg/L), sodium chloride, 3 g/L, and Polyaqua™ (0.1 ml/L). Once fish were lightly anaesthetized (marked by loss of equilibrium), they were weighed to the nearest 0.1 g, measured (standard and total length) to the nearest 1 mm, and fish over 50 mm total length were given a unique mark with a visual implant elastomer (VIE tag, Northwest Marine Technologies). These unique marks allowed us to record individual capture histories for quantitative analysis. Fish were marked with an elastomer dye administered with a hand injection system. Each fish was given one, two, or three small marks, but no more than one mark was administered to any given site. We used the following sites for marking: caudal fin above the lateral line, caudal fin below the lateral line, anal fin, pelvic fin, and head behind the orbit (**Figure 3.3**). The right and left side of the fish could be marked for a total of ten sites. Using three marked sites per fish and four colors (fluorescent red,

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fluorescent orange, fluorescent green, fluorescent yellow), allowed us to administer an individual mark to each captured fish. The locations and marks of recaptured fish were recorded for analysis.

Processed fish were placed in a recovery bath containing Polyaqua™ (0.1 ml/L) to help restore the protective slime coat and seal the injection point (Swanson et al. 1996). This also reduced the risk of fungal and bacterial infections as a result of handling and marking. From the recovery bath, fish were moved to an instream live car for the remainder of the day to allow the elastomer to set and the injection point to heal.

Mark-recapture analyses assume perfect tag retention, so we tested this assumption in the laboratory. Thirty-five fathead minnows (*Pimephales promelas*) were given VIE marks corresponding to those we used in the field. They were held in the Colorado State University Foothills Fisheries Laboratory for 180 days. Mortality was independent of mark location and did not vary significantly between controls and marked fish. Tag retention rates were 100% in the laboratory (Hill and Myrick In Preparation). Data from marked and recaptured fathead minnows in Spring Creek were also examined to estimate field retention rates. Analysis results indicate that tag retention rates in the field are at least 92.4% (Hill and Myrick In Preparation).

The most important known cues for fish migration include water temperature (Northcote 1998; Prignon et al. 1998; Trial et al. 1983a), stream flow (Fausch and Bestgen 1997; Nesler et al. 1988; Northcote 1998; Taylor and Miller 1990), and photoperiod (Taylor and Miller 1990). We measured each of these during each sampling trip. Water temperature was measured with a hand thermometer upon arrival at the site (0800 – 0900 h) and around 1600 h to bracket the greatest thermal range (U. S. Forest Service, unpublished data). Stream flows were obtained from a U.S.G.S. gauge on the St. Vrain River upstream from our sampling site. On Spring Creek, they were estimated using a Marsh-McBirney current meter and a top-setting wading rod. Ten depth and flow measurements were taken in even increments across the stream channel, and the total stream flow was estimated using the equation

$$Q = \sum_{i=1}^{n} q_i$$

where Q = total stream flow in m^3/s , and q=the flow through each increment. This was determined by using the following equation:

$$q_i = v_i \times d_i \times \left(\frac{b_{i+1} - b_{i-1}}{2} \right)$$

where v_i=measured velocity in each increment, d_i=depth at the site of each flow measurement, and b=width of each portion of the cross section (Orth 1983). Photoperiod data specific to the study site latitude were obtained from U.S. Naval Observatory data.

We took longitudinal (**Figures 3.4** and **3.5**) and cross-sectional (**Figures 3.6** and **3.7**) stream profiles of the two study segments so that we could characterize them and compare them. We also measured water velocities on the St. Vrain rock ramp. We focused on the restricted points between boulders through which small fishes would have to sprint to ascend this structure. These high-velocity areas would be the limiting factor in determining whether fish could successfully ascend the ramp.

Data Analyses

Mark-recapture data were analyzed using a multi-strata open population model in program MARK (White and Burnham 1999). In order to quantify the effects of the drop structure on Spring Creek, we ran two analyses. We attempted to estimate baseline movements in the absence of physical barriers with our first analysis by examining only the three 100-m reaches downstream of the drop structure. The second analysis included the entire study segment, which contained the large drop structure (in the middle of the study site) and a small cascade (without vertical drops) located between the second and third 100-m reach above the drop structure (Figures 3.8-3.9). The strata were defined as the sections between the instream structures: the first stratum was the 300 m reach downstream of the drop structure, the second was the 200 m reach between the drop structure and the small cascade, and the third was the 100 m reach above the small cascade.

The multi-strata model in program MARK produces transition probabilities, which estimate movement rates. A transition probability is the probability that an individual captured in a given stratum at one time will be recaptured in another stratum during a subsequent sampling effort. No analytical method can discriminate between individuals that have left the study area permanently, those that were in the study area and not captured, or those that died. Therefore, transition probabilities are likely to underestimate movement rates, and we can only make inferences regarding the detected movement rates in Spring Creek.

For the downstream reach, we tested the hypothesis that only distance was important by comparing a model with two transition probabilities ($A^1{\rightarrow}A^2 = A^2{\rightarrow}A^3 = A^2{\rightarrow}A^1 = A^3{\rightarrow}A^2$; $A^1{\rightarrow}A^3 = A^3{\rightarrow}A^1$) to one with four transition probabilities. The alternative model emphasized a difference between upstream and downstream movements ($A^1{\rightarrow}A^2 = A^2{\rightarrow}A^3$; $A^2{\rightarrow}A^1 = A^3{\rightarrow}A^2$; $A^1{\rightarrow}A^3$; $A^3{\rightarrow}A^1$). For the entire study reach, we tested the hypothesis that transition probabilities were a function of distance, not of barrier presence. We set the upstream transition probabilities equal to their corresponding downstream transition probabilities ($A{\rightarrow}B = B{\rightarrow}A$; $A{\rightarrow}C = C{\rightarrow}A$; $B{\rightarrow}C = C{\rightarrow}B$) and compared this model with the barrier model (six transition probabilities, all unique). Models were ranked with Akaike's Information Criterion, adjusted for small sample sizes (AICc).

Results

St. Vrain River

We captured and marked 1,115 fish of 21 species on the St. Vrain River (**Table 3.1**). Of these, only 17 were recaptured. Only one fish marked above the rock ramp was recaptured below it; no fish marked below the structure were recaptured above it. These low recapture rates prevented us from quantitatively analyzing the data gathered from the St. Vrain River. Instead, we examined the rock ramp carefully and assessed its effectiveness qualitatively. Further study of this structure would be necessary to determine whether or not it allows upstream and downstream movement of a variety of fishes. However, measured velocities on the rock ramp ranged from 0.33 to 1.01 m/s, and 75 percent of the measured velocities were 0.68 m/s or less. Based on the swimming abilities

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of the brassy minnows, Arkansas darters, and common shiners tested in the laboratory (**Chapter 2**), these velocities should allow passage of at least some small-bodied fishes.

Spring Creek

We recaptured and positively identified 520 of the 2,994 fish (representing 15 species; **Table 3.2**) we captured and marked on Spring Creek on at least one occasion. Of these, only 6 individuals (1 percent) moved upstream past the drop structure, and only 12 (2.3 percent) moved downstream across it. The multi-strata models we used are characterized by a large number of estimable parameters and therefore require a large amount of data. We caught sufficient numbers of white suckers (*Catostomus commersoni*) and creek chubs (*Semotilus atromaculatus*) to estimate their transition probabilities between reaches, but we had insufficient data for the other species. Only white suckers and creek chub moved upstream past the barrier (**Table 3.3**). Although we were able to estimate these movement probabilities, we could not relate them to flow or water temperature, as the models in program MARK containing transition probabilities that changed over time never reached convergence.

Detected movement in Spring Creek was a function of distance and the presence or absence of instream structures (**Table 3.4**, **Figures 3.10-3.11**). Detected movements greater than 100 m were relatively rare in the unrestricted reaches below the drop structure; the transition probabilities for movements over 100 m were 0.080 (upstream direction) and 0.025 (downstream direction) (**Figure 3.10**). Although many of the recaptured fish in Spring Creek did not move long distances, it is clear that the instream structures in this system reduce upstream movement (**Figure 3.11**). The transition probability that estimated movement between strata separated by the large drop structure was more than 34 times lower than the one estimating movements greater than 100 m in unrestricted strata (0.0025 vs. 0.080). Fish moving upstream past the large drop structure ranged from 58-161 mm in total length (**Table 3.3**). These fish were small to medium-sized in comparison to all of the captured creek chub and white suckers. The average drop from the top of the structure to the plunge pool below was 63 cm. Of the fish that moved upstream past the structure, the largest white sucker would have had to jump over 4 body lengths and the largest creek chub would have had to jump over 5 body lengths to clear it. Brook trout of similar size (100-150 mm) were able to jump approximately 4-5 body lengths from plunge pools of 10 cm or more (Kondratieff and Myrick 2006). Because the pool below the drop structure offers little to no staging opportunity for jumping fishes (i.e., it is very shallow and small), it is unlikely that the largest of the upstream movers actually jumped the barrier. In fact, the success of the smaller fishes leads me to hypothesize that these fish moved upstream by swimming under or around the structure. The presence of a hole or pipe under or around the large drop structure is possible because during sampling, we observed frequent undercutting of the riprap on the stream banks.

On the other hand, the transition probability that estimated upstream movement across the small cascade was slightly lower than transition probabilities between unrestricted reaches (0.037 vs. 0.080). No recaptured fish moved from stratum A upstream to stratum C, which prevented program MARK from properly estimating a transition probability. This value was fixed at zero to allow estimation of other parameters. This lack of apparent upstream movement appears to be a function of barrier presence, not of distance. Models in MARK that emphasized the importance of distance had no support from the data (**Table 3.4**). It is apparent that the combination of the drop

structure and the small cascade essentially eliminated upstream movement in this segment of Spring Creek. Movement downstream does not seem to be impaired by the structures, as downstream transition probabilities over the large barrier and the small cascade were similar to those seen in unrestricted reaches (and similar to or greater than what was seen in the raw data). As a result, these structures bias the movement of fishes in Spring Creek in a downstream direction and likely have negative impacts on the local fish assemblage.

Discussion

St. Vrain River

Although this structure appears to accommodate fish of all sizes, the lower pool below the fishway may present some unanticipated problems to small migrating fishes. First, the southern portion of the lower pool harbors many nonnative predators such as largemouth bass (*Micropterus salmoides*) and white crappie (*Pomoxis annularis*). We captured hundreds of piscivorous sunfish in the St. Vrain River (**Table 3.2**), and most of them were found in this pool. These fishes could prey heavily on the smaller fishes attempting to use the structure. Because these predators prefer lentic areas, elimination of this lower pool would remedy the problem and should not reduce the energy dissipation capacity of the structure. Second, there is a large eddy on the northern side of the lower pool. Eddies below fishways tend to disorient fish so that they are delayed or prevented from gaining entrance to the structure (Clay 1995). Wing walls have been used to eliminate eddies in traditional fishways (Clay 1995). Though this structure already contains a wing wall in the form of a row of small boulders, its extension or enlargement might solve the problem.

Based on the results of our swimming performance experiments and the measured velocities between boulders on the St. Vrain rock ramp, brassy minnows and common shiners should be able to swim upstream past this structure. Arkansas darters cannot sprint at these velocities, but their tendency to use the boundary layer may allow them to ascend a structure such as this one. Although this structure probably prevents the movement of some small-bodied fishes, it is unlikely that it prevents it altogether. Therefore, a structure such as this is preferable to a vertical drop structure. The rock ramp fishway also maintains a slope that is more typical of a transition-zone stream. Instead of a series of vertical drops to achieve grade control, this structure incorporates an incline that is more accommodating towards non-jumping fishes (**Figure 3.4**).

Spring Creek

The transition probabilities from the MARK analysis of the entire study reach departed from the observed data: upstream movements were underestimated, and downstream movements were overestimated. This is probably due to artifacts of our sampling method. For all recaptures (i.e., some individuals were recaptured more than once), we positively identified 34.2 percent of the white suckers and 37.8 percent of the creek chubs. Many of these recaptured fishes could not be positively identified because they were either marked incorrectly on initial capture or misread upon recapture. It is far more likely that most of these fish were misread as it was easy to confuse the elastomer dye colors, especially if they had been injected under pigmented skin. It is also

possible that some fish lost marks. Tag retention rates in these two species may have been lower than it was in fathead minnows. Unidentifiable individuals probably led to biased estimates of capture probabilities and therefore transition probabilities. However, the raw data and the analytical results do agree in that downstream movement across the large barrier was higher than upstream movement (**Table 3.5**).

The length of our study reach on Spring Creek was dictated, in part, by the logistical constraint of the project. We suspect that many of the fish that we marked left the study area, leading us to underestimate the success rate of fish attempting to cross the structure in an upstream direction. Few studies have been conducted to quantify the movements of resident stream fish, but their results indicate that they do move surprising distances. Telemetry studies involving larger fish indicate that individuals are capable of moving extremely long distances. For example, radio-tagged Colorado pikeminnow (*Ptychocheilus lucius*) migrated distances of 100-350 km to spawn (Tyus et al. 1987). A radiotelemetry study on adult cutthroat trout (total length 170-237 mm) indicated that they moved over 500 m during the summer months (Fausch and Young 1995). Observations of smaller fish also indicate that they can move substantial distances in short periods of time; a school of plains minnows (*Hybognathus placitus*) migrating upstream in the South Canadian River, New Mexico in 1989 moved 250 m in 15 minutes (Fausch and Bestgen 1997). However, because we were able to show that very few of the recaptured fish negotiated the artificial waterfall, we are confident that it does pose a substantial barrier to the upstream migration of fishes. This structure prevents local movements, and there is no reason to believe that it does not have similar effects on long-distance movements. The analysis of movement rates in the unrestricted downstream reaches indicates that many of the fish that were recaptured in Spring Creek did not move long distances, perhaps because of the fragmented nature of that system, but transition probabilities estimating movements of more than 100 m were 0.080 and 0.025. The transition probability between strata separated by the drop structure was at least an order of magnitude lower than this, suggesting that it may impede important processes necessary for completion of the life history of some fish species.

For many animals, including stream salmonids, the distribution of movements within a population is leptokurtic (Gowan and Fausch 1996). The percentage of fish that move a given distance is small, but 30 to 50 percent of them may move long distances, thereby driving important biological processes. For example, when a 50-year flood caused a 50% reduction in fish abundance in Kings Creek, Konza Prairie Biological Station, Kansas, the fish community recovered, presumably due to recolonization, within three months (Fritz et al. 2002). Long-distance movements also maintain gene flow and drive rescue effects for small, isolated populations (Stacey and Taper 1992). The Spring Creek study clearly demonstrated that the instream structures bias fish movement in a downstream direction, thereby introducing an artificial constraint on the fish community.

The channel work in our study segment has modified Spring Creek so that its longitudinal profile resembles that of a mountain stream (**Figure 3.5**). The warm water fish species typical of transition-zone streams have not evolved to cope with physical features such as large vertical drops. The Spring Creek fish assemblage has lost some of its more sensitive species (i.e. common shiner, *L. cornutus*), presumably because of channel modifications such as channelization and inadvertent construction of fish barriers (Fausch, K. D., Colorado State University, personal communication). Under current conditions, a catastrophic event such as a chemical spill or

dewatering in one of the upstream segments could result in the local extinction of most of the fishes because of the lack of effective connectivity with downstream reaches.

Assessment of Sampling Methods

Tracking individual fish using VIE tags and repeated electrofishing is a novel method of estimating movement across rock ramps, and this study illustrated the strengths and limitations of that approach. The first thing that affected our success was stream size. On Spring Creek, we were able to collect enough data to run a quantitative analysis. Within the study reach, the stream was a maximum of 5 m wide (**Figure 3.7**) and thus was easy to sample with a single backpack shocker. Conversely, the study segment on the St. Vrain River was approximately six times as wide (Figure 3.6) and much harder to sample. We used a single backpack electrofisher to capture fish in this stream. It was nearly impossible to efficiently sample the large volume of water in this stream with a single backpack unit, so our capture probabilities were low. Also, high water in the spring prevented us from sampling on every scheduled occasion, and capture probabilities were therefore further reduced. Future sampling of streams this size should involve multiple backpack units or an electric seine (Bayley et al. 1989).

Another major consideration is that individually marking thousands of small fish was labor intensive. We found that a minimum crew size of five was necessary to mark hundreds of fish per day. The most time-consuming part of processing the fish was marking fish and identifying recaptured fish. Identifying a previously marked fish took about the same amount of time as marking a newly captured one, though the amount of time needed for both operations decreased with experience.

Each individual mark was a unique combination of marking sites and elastomer colors, but care had to be taken in placing and reading marks to avoid misidentification. The colors used were similar enough that green was sometimes mistaken for yellow, and red was sometimes mistaken for orange. This is a known challenge with VIE marks (Curtis 2006), and it can be alleviated by using more disparate colors and checking ambiguous colors with the provided blue LED light. We discovered that there was some potential for human error in both administering and reading marks. Despite excellent tag retention rates (approximately 92 percent in the field), we weren't always able to positively identify recaptured fish because they had either been incorrectly marked when they were first caught or because their marks were misread upon recapture. Carefully trained crews using standardized marking and mark-reading methods would minimize this source of error. Tag reading accuracy does not appear to increase with reader experience (Curtis 2006). Therefore, potential bias could be minimized in future studies by identifying the most accurate tag readers with pilot studies before commencing fieldwork. Despite the fact that our mark-recapture approach using VIE marks and a backpack electrofishing unit requires a large crew, we were able to collect enough data to quantitatively analyze movement probabilities of the fishes in Spring Creek by spending only four days every other month at the sampling site. On larger systems, such as the St. Vrain River, the use of a larger crew, multiple backpack shockers, and a more frequent sampling schedule (four days each month) could increase capture and recapture rates to make quantitative analyses of movements rates feasible.

Administering and reading individual marks provided benefits that outweighed the cost of extra effort. Transition or movement probabilities can only be estimated by tracking individual capture histories (White and Burnham 1999). Programs such as MARK and SAS, both of which can estimate these parameters, only recognize individual capture histories. Even if one were content with the information provided by batch marking, marking protocols would necessarily become complex to accommodate multiple capture sites and occasions. Furthermore, marking recaptured individuals would be difficult.

Tables

Table 3.1 — Fish species captured on the St. Vrain River between December 2004 and October 2005. With the exception of one brook stickleback, all listed fish were 50 mm long or greater. Smaller fish were released into the stream without being processed and were not included in the analysis. A total of 1340 individuals of 21 species were captured.

Species	Number captured	Total length range (mm)
black crappie (*Pomoxis nigromaculatus*)	8	74-182
bluegill (*Lepomis macrochirus*)	24	50-131
brassy minnow (*Hybognathus hankinsoni*)	1	52
black bullhead (*Ameiurus melas*)	2	50-58
brook stickleback (*Culea inconstans*)	48	48-71
common carp (*Cyprinus carpio*)	153	62-680
fathead minnow (*Pimephales promelas*)	90	50-74
gizzard shad (*Dorosoma cepedianum*)	16	81-97
green sunfish (*Lepomis cyanellus*)	20	51-130
Iowa darter (*Etheostoma exile*)	13	50-57
Johnny darter (*Etheostoma nigrum*)	16	53-70
largemouth bass (*Micropterus salmoides*)	412	71-220
longnose dace (*Rhinichthys cataractae*)	15	59-107
longnose sucker (*Catostomus catostomus*)	1	111
mosquitofish (*Gambusia affinis*)	52	51-58
orangespotted sunfish (*Lepomis humilis*)	3	50-64
red shiner (*Cyprinella lutrensis*)	51	55-73
sand shiner (*Notropis stramineus*)	299	51-75
white crappie (*Pomoxis annularis*)	18	55-165
white sucker (*Catostomus commersoni*)	87	51-305
yellow perch (*Perca flavescens*)	11	76-216

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Table 3.2 — Fish species captured on Spring Creek between September 2004 and July 2005. With the exception of one brook stickleback and one fathead minnow, all fish listed were 50 mm or greater. Smaller fish were returned to the stream without being processed, and they were not included in the analysis. A total of 2994 individuals of 15 species was captured.

Species	Number Captured	Total Length range (mm)
bluegill (*Lepomis macrochirus*)	5	92-102
bigmouth shiner (*Notropis dorsalis*)	2	61-76
brook stickleback (*Culea inconstans*)	54	49-69
common carp (*Cyprinus carpio*)	1	57
creek chub (*Semotilus atromaculatus*)	951	51-240
fathead minnow (*Pimephales promelas*)	480	44-80
green sunfish (*Lepomis cyanellus*)	94	62-186
Iowa darter (*Etheostoma exile*)	8	50-70
Johnny darter (*Etheostoma nigrum*)	43	53-73
longnose sucker (*Catostomus catostomus*)	13	94-237
largemouth bass (*Micropterus salmoides*)	64	44-87
longnose dace (*Rhinichthys cataractae*)	294	44-117
rainbow trout (*Oncorhynchus mykiss*)	1	280
sand shiner (*Notropis stramineus*)	26	55-80
white sucker (*Catostomus commersoni*)	958	51-360

Table 3.3 — Captured fish in Spring Creek that moved upstream across the large drop structure. Only white suckers and creek chub were recaptured above the barrier after having been captured below it. The fish that moved upstream were small to medium in size.

Species	Number of fish moving	Total lengths (mm)
creek chub (*Semotilus atromaculatus*)	2	58, 118
white sucker (*Catostomus commersoni*)	4	68, 69, 129, 161

Table 3.4 — Top models from the MARK analysis of transition probabilities in Spring Creek. The best model for the lower section of the study area (no physical barriers) shows that the data support the role of distance and direction in determining transition probabilities. The model differentiating between upstream and downstream movement has more support from the data than the one that equated upstream and downstream movements of the same distance. The best model for the entire study area (separated by physical barriers) supports barrier presence as the primary factor in determining transition probabilities. The model that incorporated distance (upstream and downstream movements equal) had no support from the data.

Study Reach	Model	AICc	Δ AICc	Relative Weight
Downstream of drop structure	Distance and direction	4012.8	0	0.786
	Distance	4015.4	2.6	0.214
Entire study reach	Barrier presence	5773.9	0	1
	Distance	5865.2	91.3	0

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Table 3.5 — Comparison of raw data and model output. Despite some discrepancy between the two, it is clear that upstream movement is less frequent than downstream movement, at least among the recaptured fishes.

Direction of Movement	Raw Data	Model Output (Transition Probabilities)	
		ψ (Strata A and B)	ψ (Strata A and C)
Upstream (A to B or C)	0.011	0.002	0
Downstream (C or B to A)	0.026	0.066	0.180

Figures

Figure 3.1 — A rock ramp fishway on the St. Vrain River, Colorado, USA. These structures incorporate a number of depths and water velocities, and they allow migration of non-jumping fishes. Photo: Christopher Myrick.

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Figure 3.2 — The vertical drop structure on Spring Creek. This structure has an average drop height of 63 cm that prohibits non-jumping and small-bodied jumping fishes from moving upstream. Photo: Ashley Ficke
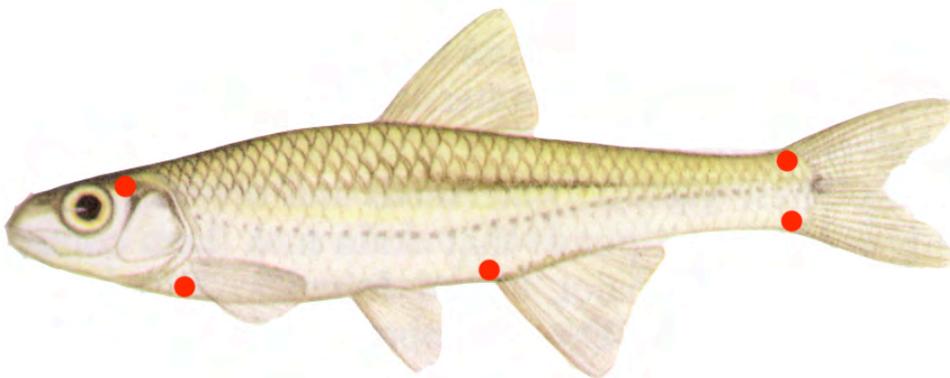
Figure 3.3 — Marking sites for captured fish on Spring Creek and the St. Vrain River. Each fish had 10 sites (five on each side), and we used four colors of elastomer. By giving each individual three marks that were a unique combination of sites and colors, we were able to individually mark nearly 3000 fish in Spring Creek and over 1300 in the St. Vrain River.
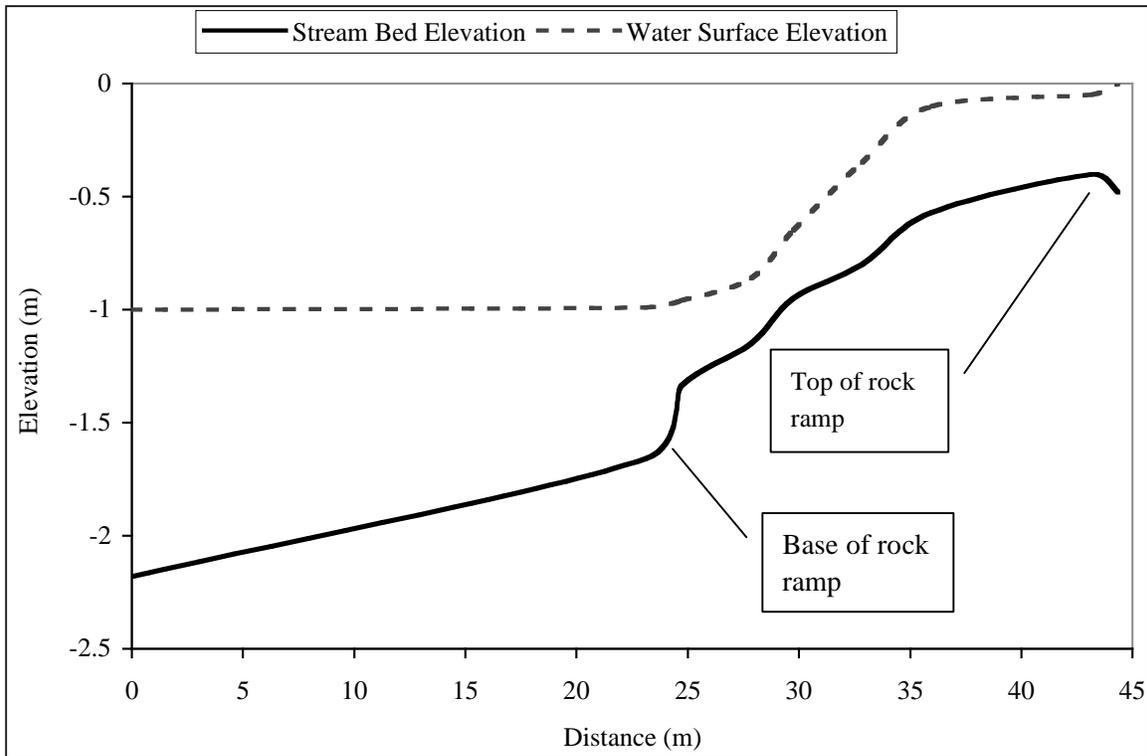
Figure 3.4 — A longitudinal section of the St. Vrain River from approximately 25 m below the rock ramp fishway to the top of the structure. Large boulders at the base of the rock ramp prevent formation of a deep plunge pool.

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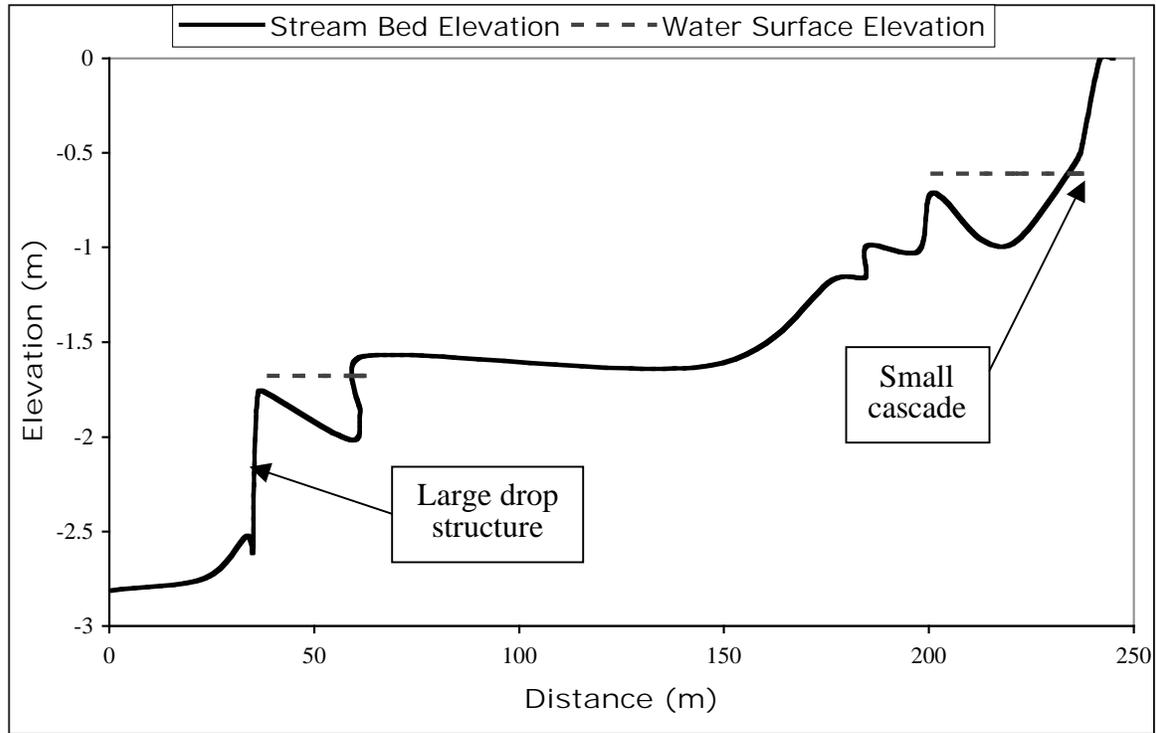
Figure 3.5 — A longitudinal section of the engineered portion of Spring Creek from below the large drop structure to the upstream end of the study reach. Sinuosity was very low, and multiple grade-control structures were installed to compensate for the change in slope involved in straightening the stream channel. Water surface elevation was added to indicate the large deep glides in this section.
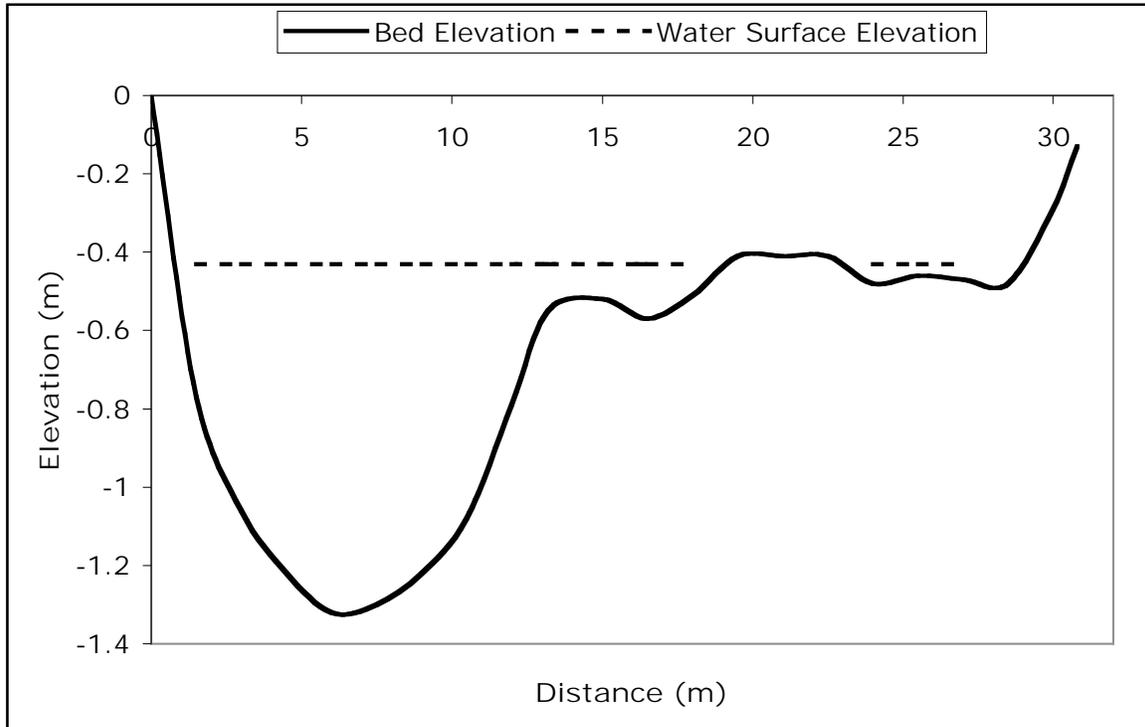
Figure 3.6 — A cross section of the St. Vrain River below the rock ramp fishway. This stream was approximately 30 m wide and was not channelized.

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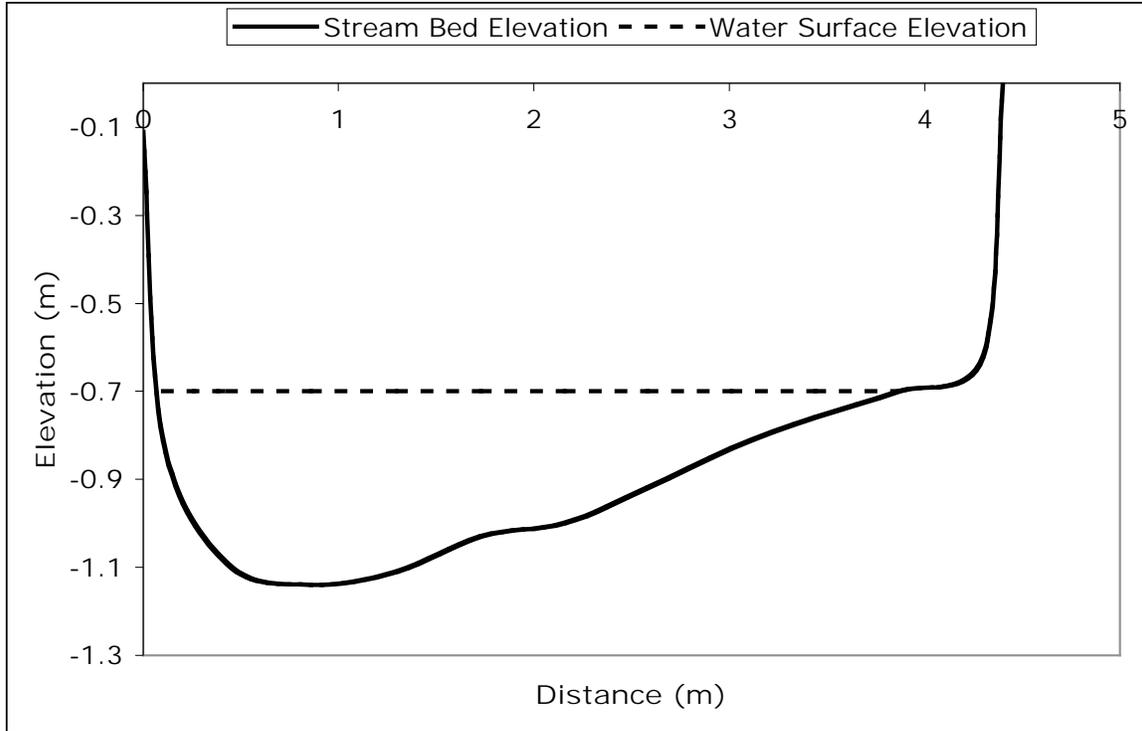
Figure 3.7 — A cross section of Spring Creek above the large drop structure. Stream width rarely exceeded 5 m in width in our study reach, and it was channelized from the drop structure to the upstream end of our study reach.
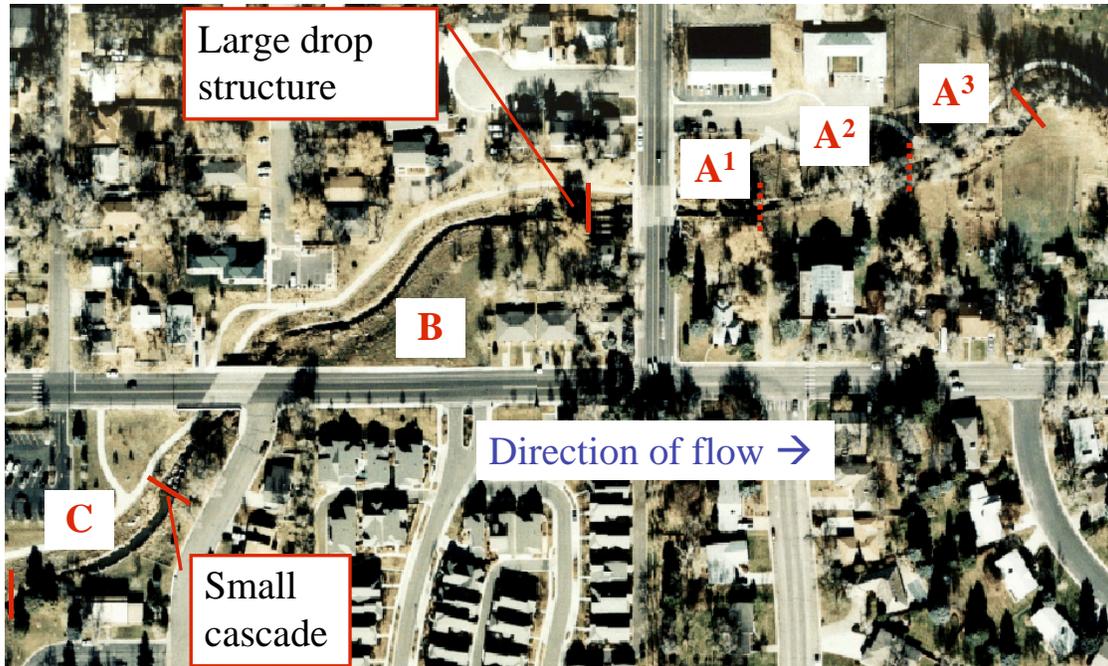
Figure 3.8 — Aerial view of the Spring Creek study site. We conducted two analyses on the data collected from Spring Creek. First, we estimated transition probabilities between strata A^1, A^2, and A^3; this provided information about movement rates in the absence of a physical barrier. In the second analysis, we treated A^1, A^2, and A^3 as a single stratum and estimated transition probabilities between strata A, B, and C. Potential barriers separated these three strata: the large drop structure is between strata A and B, and the small cascade is between strata B and C.

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Figure 3.9 — The small cascade on Spring Creek. This structure incorporates a section (circled in red) that can be ascended by non-jumping fishes.
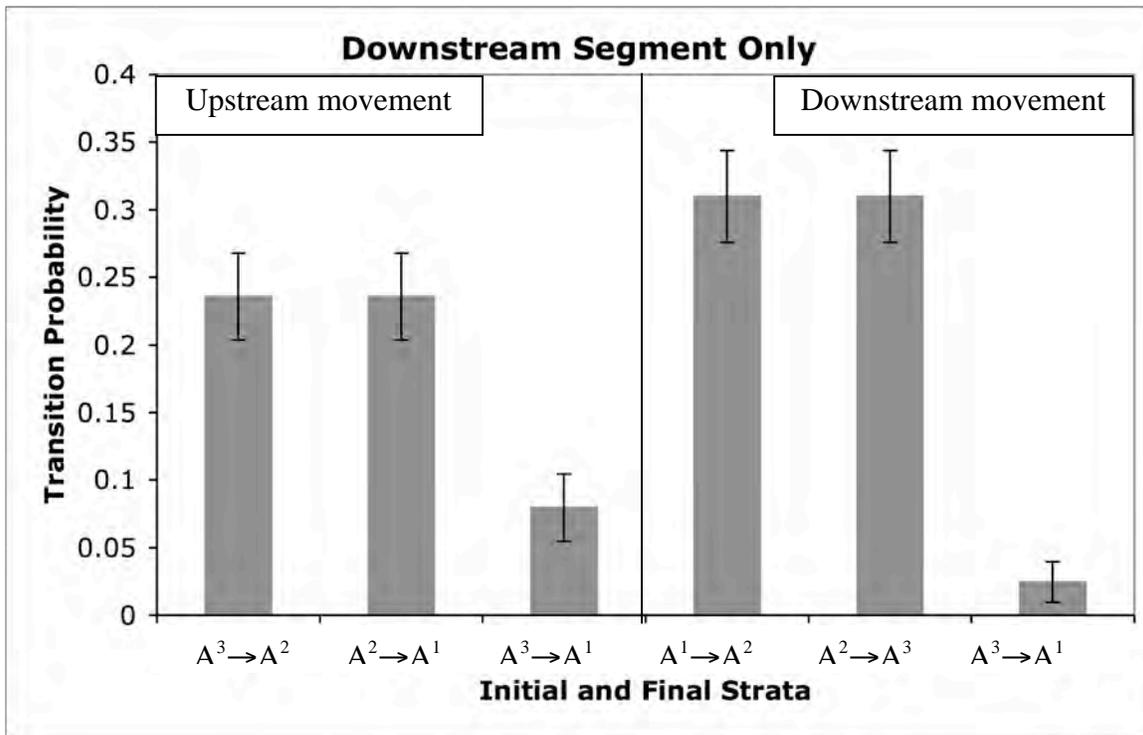
Figure 3.10 — Movement detected in Spring Creek downstream of the large drop structure. Detected movements in excess of 100 m were relatively rare. However, the short length of the study reach prevented detection of long-distance movements and may have led to the underestimation of local movements.

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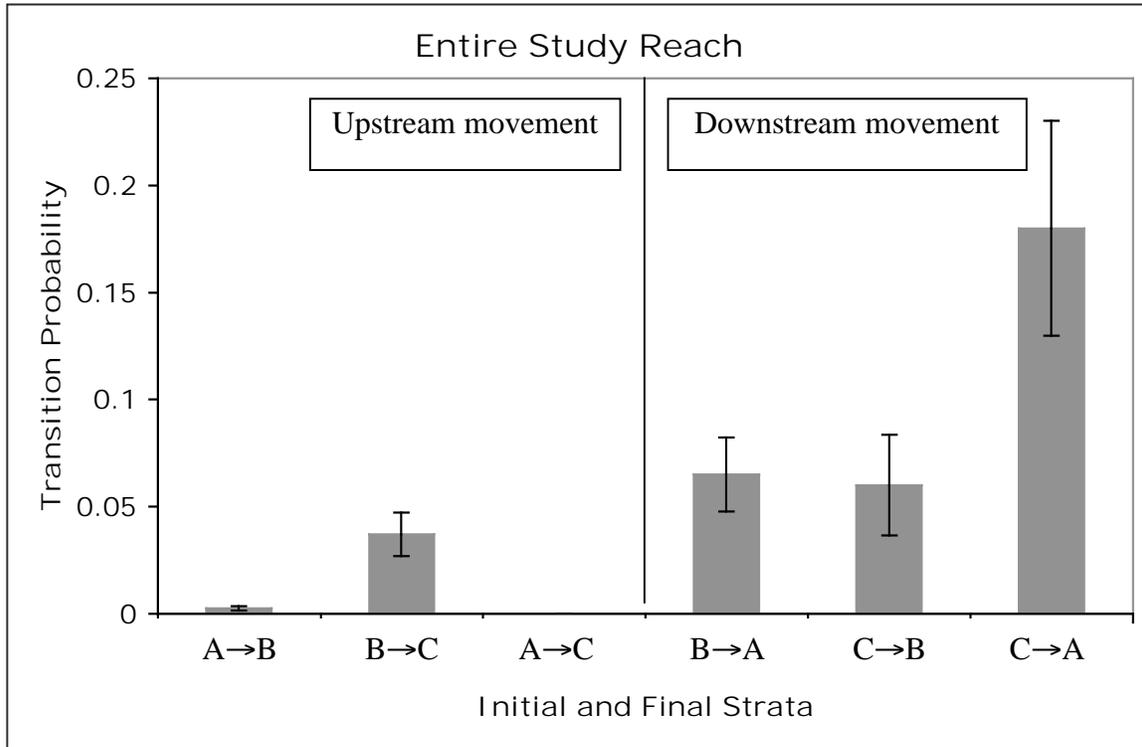
Figure 3.11 — Movement detected between Spring Creek reaches separated by grade-control structures. The drop structure and the small cascade drastically reduced apparent upstream movement rates but did not affect downstream movement rates.

CONCLUSIONS AND RECOMMENDATIONS

There is mounting evidence that stream fishes exist in a metapopulation framework (Rieman and Dunham 2000; Scheurer 2002; Schlosser and Angermeier 1995). This is significant because the persistence of a population depends primarily upon patch size, isolation, and colonization rate (Hanski and Gilpin 1991; Levins 1969). Fish in plains and transition-zone streams have evolved to cope with a certain degree of habitat fragmentation. For example, in the Big Sandy drainage, streams are often intermittent in the dry season and frozen to the bottom during the winter (Labbe and Fausch 2000). This natural fragmentation is seasonal and intermittent; with the onset of precipitation or warmer temperatures, migration between patches can occur. In transition-zone streams intermittency is less common. Unfortunately, land uses in these regions have led to a dramatic increase in both the frequency and duration of fragmentation. Changing a stream from a longitudinally continuous system to a series of fragments has consequences for fish metapopulations. It increases isolation of patches, thereby increasing the extinction risk of local populations and decreasing the probability that a patch will be revitalized by immigrants (Brown and Kodric-Brown 1977). Monitoring programs such as the one described above can identify problem structures that disrupt the spatial population structure of resident fish species. These structures convert once-connected subpopulations into isolated populations, thereby increasing the probability of local extinctions. Identifying and modifying them so that they will permit upstream and downstream movement of resident fishes will allow fishes to better cope with the other challenges that come with urban and agricultural land use. Thus, the reversal of fragmentation will help ensure the sustainability of fish communities in streams affected by urbanization or agricultural uses.

A new perspective is necessary if we are to successfully maintain warm water fish communities on the Front Range. For example, much research has focused on the swimming and jumping ability of trout and salmon, but these data are irrelevant in these systems as salmonids are rare in transition-zone streams and absent from plains streams. The mark-recapture study on Spring Creek clearly demonstrated the impact that even a small (63 cm) drop structure can have on fish movements and highlighted the need for the widespread installation of adequate fish passage structures. Historically, the needs of small-bodied fishes have been ignored, but new fishway designs will effectively allow them to negotiate otherwise impassable obstacles. Our work has shown that although some transition zone species may have the ability to jump, their small size reveals a need for instream structures that lack vertical drops entirely. On the other hand, they are capable of swimming at impressive relative velocities and absolute velocities that could easily be incorporated into fishways. Their small body size requires designers to address a smaller scale that has perhaps been heretofore neglected; small fish use the boundary layer to move through high-velocity sections of a fishway, but they are also vulnerable to displacement by small amounts of turbulence.

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Although we have made recommendations concerning the structure, water velocity, and length of fishways for these species, further study is required to determine small-scale design elements such as boundary layer thickness. Engineered structures such as vertical slot fishways or Denil fishways require careful study and design, may represent expensive retrofits to existing instream structures, and may not work for all the fish found in a stream system. Rock ramp fishways may also be expensive to install, but are less difficult to design and by their very nature, provide fish with a diversity of options for upstream passage. Resource managers would be wise to carefully study the success of existing rock ramp fishways and apply those lessons, and designs, to potential obstacles in other plains and transition zone systems. Once in place, any fishway needs to be monitored to evaluate its efficiency, and, if necessary, its basic design altered until it meets its intended purpose.

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