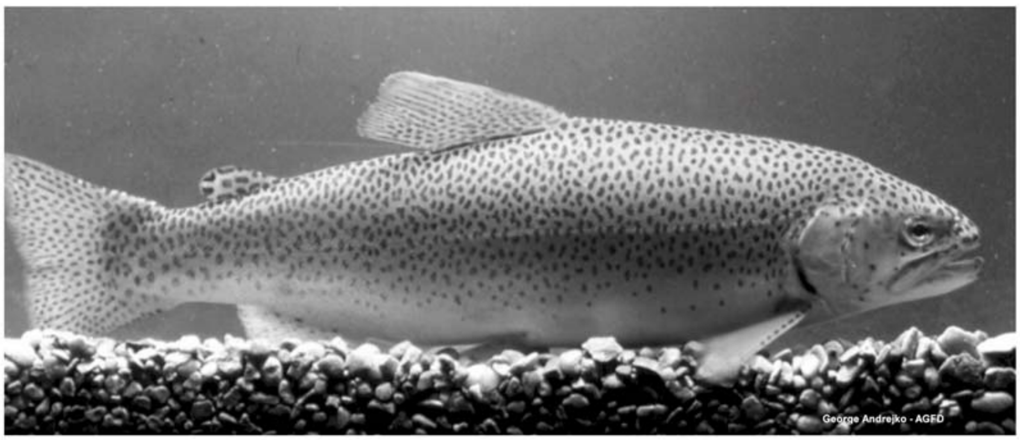

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Evaluation of Apache Trout Habitat Protection Actions

Federal Aid in Sport Fish Restoration
Project F-14-R

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Arizona Game and Fish Department Mission

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INTRODUCTION

Apache trout (*Oncorhynchus apache*) is a federally threatened salmonid native to headwaters of the Little Colorado, Black, and White rivers in east-central Arizona. Decline of Apache trout to threatened status was attributed to over-fishing, habitat degradation and negative interactions (predation, competition and hybridization) with introduced nonnative salmonids (USFWS 1983). Although over-fishing is no longer considered a threat, habitat degradation and negative interactions with nonnative salmonids continue to threaten Apache trout, and it is towards these threats that recovery actions are directed.

Logging, grazing, mining, reservoir construction, agricultural practices and road construction all have played some role in degrading riparian-aquatic habitat (USFWS 1983). Alteration of logging practices, removal of roads, and exclusion of livestock from riparian areas (either by fencing or disallowing grazing) are examples of actions directed at restoring riparian and stream habitat. With respect to livestock exclusion, over 100 miles of stream on U.S. Forest Service lands have been fenced to restore riparian and Apache trout habitat.

Barrier placement, in conjunction (when necessary) with chemical piscicide treatement (renovation) and subsequent stocking of pure Apache trout into the stream above the barrier, is the primary method to isolate Apache trout from nonnative salmonids. Since 1979, barriers have been erected in 13 streams within Apache-Sitgreaves National Forest, and will be erected in at least three more streams by 2006.

While barrier construction began in 1979 and livestock exclusion began in the mid-1980s, the efficacy of these recovery actions at increasing Apache trout abundance and improving habitat condition had not been evaluated. We therefore initiated a study to evaluate the efficacy of riparian fencing and barriers. Our study had two major objectives to address these recovery actions. One was to evaluate if the exclusion of livestock from riparian areas had improved riparian and stream habitat and increased Apache trout production, condition and food resources. Sub-objectives were to: a) determine habitat used by Apache trout, b) determine if habitat use is correlated with time elapsed since fencing (i.e., as recovery increases), and c) determine if restored (fenced) areas contain more Apache trout habitat than what was available prior to fencing. The second major objective was to evaluate the effectiveness of constructed barriers at preventing upstream movement of nonnative salmonids into reaches occupied by Apache trout. A sub-objective was to determine if Apache trout move downstream past barriers.

STUDY AREAS

Study streams are located in east-central Arizona (Figure 1) within the Apache-Sitgreaves National Forest and are headwaters of the Little Colorado, Black, and Blue rivers (Figure 1). Within the Little Colorado River Basin, the streams include the Coyote-Mamie creek system, which drains Escudilla Mountain; Mineral Creek, which begins from springs below Green’s Peak; and Lee Valley Creek, which drains Mount Baldy. Within the Black River drainage study streams included the West Fork of the Black River-Burro Creek-Thompson Creek complex; the Fish-Double Cienega-Corduroy creek complex; and Bear Wallow, Conklin, Hayground, Home, Soldier, Snake and Stinky creeks. Coleman Creek is located in the Blue River drainage.

The effectiveness of livestock exclusion was studied on Mineral, Coyote, Soldier, Conklin, Fish, Double Cienega, and Corduroy creeks. Mineral, Coyote, and Soldier creeks contain allopatric populations (no other salmonid species present) of genetically pure Apache trout above their fish barriers. Conklin, Fish, Double Cienega, and Corduroy creeks contain Apache trout and Apache-rainbow hybrids, but no other salmonid species above fish barriers (at least at the initiation of the study; Jim Novy, Arizona Game and Fish Department, personal communication). Study streams were either never renovated or had been renovated more than 20 years before our study. Each of the streams has meadow, intermediate, and headwater- canyon reach types (Rosgen 1985; Clarkson and Wilson 1995). Most of the fenced reaches are meadows, but some intermediate and canyon reaches also were fenced. All study streams had reaches that were sampled at least twice prior to cattle exclusion. In order to make meaningful comparisons between pre- and post-fencing periods, we targeted sites in meadow and intermediate reaches during the post-fencing period because cattle graze primarily in meadows. Canyon areas tend to have rocky substrates that are less prone to erode due to trampling, and typically have less forage for cattle.

Apache trout habitat use was assessed in Coyote, Mineral, Stinky, Soldier, Coleman,

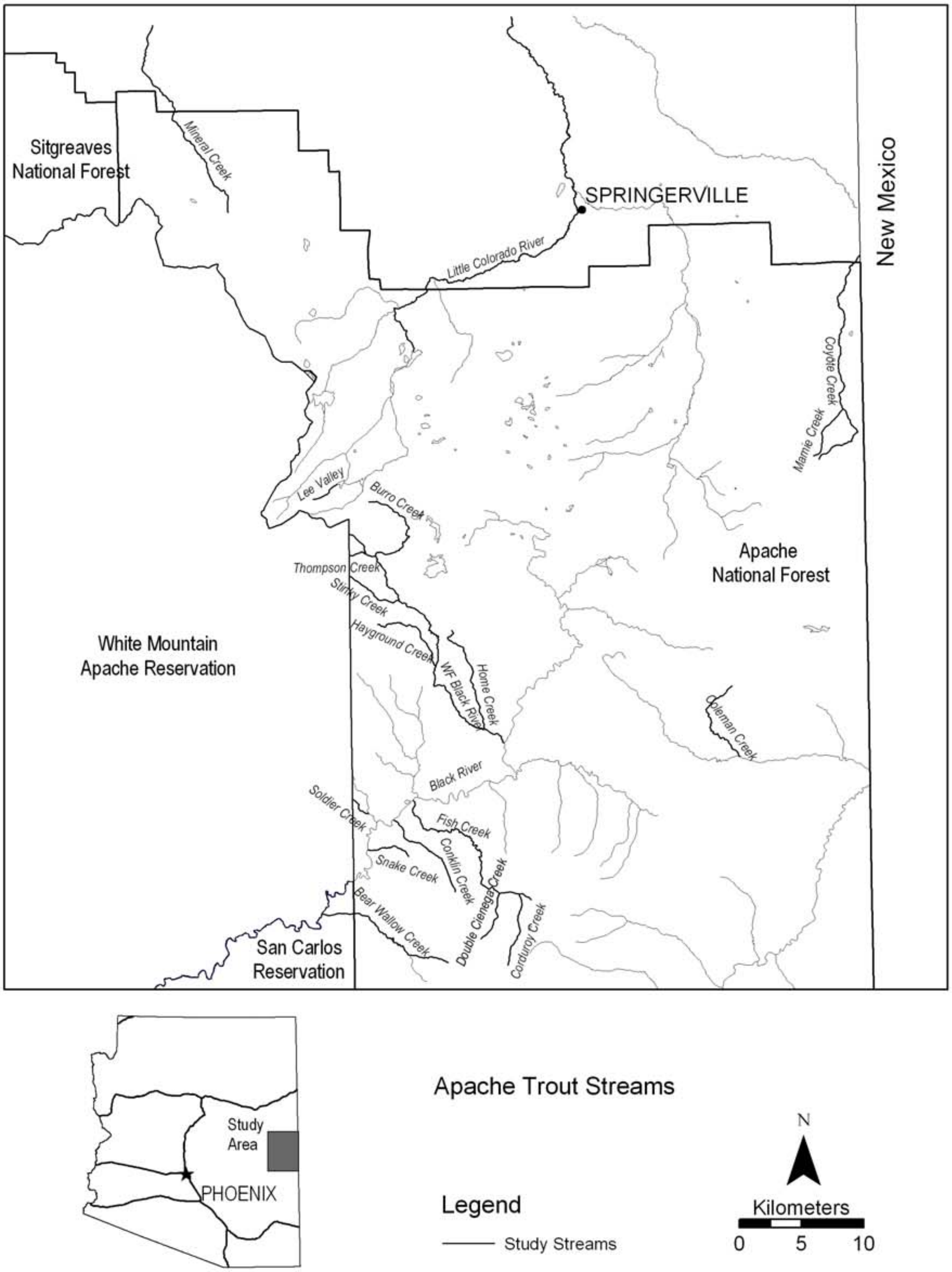

Figure 1. Map of study area showing study streams.

Thompson, and Burro creeks and the West Fork of the Black River. These streams were selected because genetically pure Apache trout was thought to be the only salmonid species present, and a broader range of habitat may be utilized by a species when interspecific competition is absent (Cunjak and Green 1982; Kitcheyan 1999). However, one stream (Stinky Creek) was found to contain brown trout (*Salmo trutta*) after initiation of our study. Habitat sampling was restricted to upstream of constructed or natural (Soldier and Coleman creeks) barriers. Streams were fenced 4 to 14 years prior to sampling, although 41% of the reaches sampled still experience short-term grazing. Habitat improvement structures (e.g., logs) were installed over the last 75 years on several of the study streams: 304 dispersed through out West Fork Black River complex, 35 in Mineral Creek, and 478 in Coleman Creek. Apache trout were the only fish species present in Coyote, Mineral, Coleman, and Soldier creeks. The fish assemblage in West Fork Black River-Thompson Creek-Burro Creek complex was comprised of Apache trout, speckled dace (*Rhinichthys osculus*) and desert sucker (*Catostomus clarki*).

We evaluated the effectiveness of constructed barriers in Bear Wallow, Conklin, Fish, Hayground, Home (two barriers), Snake, and Stinky creeks, and West Fork Black River (two barriers) in the Black River Drainage, and Coyote, Lee Valley, and Mineral creeks in the Little Colorado River Basin. The barriers on Mineral, Hayground, and Stinky creeks are gabion (wire mesh baskets filled with cobble) construction. The barriers on Bear Wallow, Home, and Coyote creeks and West Fork Black River are gabions reinforced with masonry. Conklin Creek barrier is a culvert with a grate plus a gabion, and the barrier on Snake Creek has a grate plus gabions. The barrier on Lee Valley Creek is concrete block construction.

METHODS

Fencing

Based on a literature review (Platts 1991; Fleischner 1994; Rinne 1999; and others), we developed a conceptual model of the effects of livestock grazing on production of Apache trout and riparian and aquatic habitat. Our conceptual model is that livestock overgrazing negatively affects riparian and lotic ecosystems and excluding livestock from riparian areas allows recovery of riparian and lotic ecosystems to a nominal state. Relative to grazed areas (pre-fencing), areas where livestock are excluded (post-fencing) will: 1) have greater production of Apache trout; 2) have Apache trout with greater condition; 3) have steeper and more stable banks, more undercut banks, less fine substrates and embedded larger substrates, smaller width:depth ratios, deeper near-shore areas, and more riffle habitat; 4) have more cover; 5) have more dense and diverse riparian vegetation; 6) have greater production of aquatic invertebrates; and 7) have a greater proportion of terrestrial insects in water column samples.

Model predictions were tested by comparing historical data collected before livestock were excluded from streams with data we collected during the post-fencing period. Fish and environmental variables were surveyed using General Aquatic Wildlife System (GAWS) protocols (USFS 1990, Clarkson and Wilson 1995) in pre- and post-fencing periods. Clarkson and Wilson (1995) completed the first set of GAWS surveys during 1987-1990. They established 1 to 3 fixed sites (subjectively chosen or at systematic intervals) in meadow, intermediate, and headwater-canyon reaches (classified based on criteria of Rosgen 1985). Number of sites within a reach depended on the length of the reach; shorter reaches required fewer sites in order to get a representative sample. Clarkson and Wilson (1995) determined that 50-m sites provided equivalent results to 100-m sites and so sampled 50-m sites after their first year of study. Regional Arizona Game and Fish Department biologists sampled the same sites in a second and in a few instances a third set of surveys between 1991 and 1996; additional 50-m sites were established in some reaches. The purpose of these historical surveys was to monitor Apache trout populations and stream habitat. Subsequent to these historical surveys, riparian areas were fenced to exclude livestock, and grazing was disallowed near some streams.

We sampled the established 50-m sites in meadow and intermediate reaches each year (2001-2003) during the post-fencing period. In addition, during each post-fencing year, one or more of our study streams was selected and all sites in all reaches (including canyon) were sampled to monitor Apache trout populations and habitat condition.

For fish sampling, each 50-m site was blocked off at both ends with 3-mm mesh seines. Three depletion passes were made through the site with a backpack electrofisher (a Coffelt unit during pre-fencing and a Smith-Root unit during post-fencing), and fish were captured with dip nets. After each pass, fish were identified to species, weighed (± 1 g), measured for total length (± 1 mm), identified to sex, injected with a passive integrated transponder (PIT) tag if >99 mm total length (TL), and released below the site. Fish weights missing due to equipment failure or windy conditions were estimated from a length-weight regression equation calculated using data from all streams.

At each 50-m site, numerous habitat variables were measured along five perpendicular-to-flow transects spaced equally at 10-m increments (Table 1). Riparian condition was assessed using USFS Region 3 Riparian Scorecard (USFS 1989), beginning with the second set of pre-fencing surveys. In the riparian zone of each site, tree overstory, shrub midstory, and understory components were rated (0-4) based on canopy closure, age classes, and species. An overall index of riparian condition was calculated as the sum of those ratings.

Aquatic invertebrates (Apache trout food resources) were collected once during the pre-fencing period and twice during the post-fencing period. During the pre-fencing period, six samples were collected from the 'best' riffle areas within a site using a Surber sampler, and samples were only collected at the furthest downstream site on a reach. Densities (total and for each taxa) and total biomass were expressed as means of six samples; raw data was lost. During the post-fencing period, aquatic and drifting terrestrial invertebrates were collected from each site within each reach using a Hess sampler. At each site, three samples were collected from random locations in riffles or rapids and combined into a composite sample, and three samples were collected from random locations in glides or pools and combined into a composite sample. In the laboratory, invertebrates were identified to family level, counted and weighed (g wet weight).

To evaluate effects of local drought on our study results, we retrieved discharge data from the US Geological Survey Water Resources Division website (http://waterdata.usgs.gov/az/nwis). We used data from the Black River gage 09489500 near Point of Pines to estimate runoff patterns in Fish, Double Cienega, Corduroy, Conklin, and Soldier creeks, and from the Little Colorado River gage 0934000 above Lyman Lake near St. Johns, AZ to estimate runoff patterns in Coyote and Mineral creeks. We consider data from these two stream gages to be representative of upstream discharge even though water is withdrawn above both gages for industrial, urban, and/or agricultural use.

Data Analyses. An insufficient number of control sites (continued grazing) and no reference sites (where grazing has never been allowed) existed to utilize a before-and-after treatment and control design, therefore we used a before-and-after treatment paired comparison study design. We used repeated measures ANOVA with an orthogonal contrast to compare variables of interest before to after fencing. We did not include Apache trout condition, riparian condition or benthos densities or biomass in these comparisons because we did not have sufficient sample size to make meaningful comparisons. We only included sites in our analysis if they had been sampled twice prior to fencing and three times after fencing; our final sample size was 30 sites (Table 2).

To assess Apache trout condition, we calculated Fulton's condition factor,

$$K = \left(W/L^3\right) \times 10^5$$

where W is weight in grams and L is length in millimeters (Ricker 1975), for Apache trout greater than 100 mm TL.

Environmental variables recorded as ratings were converted to percentage of maximum value prior to statistical analyses. A habitat condition index (HCI), a multivariate rating of aquatic habitat, was calculated for each site as

$$HCI = \frac{(PM + H + G + C + S + V)}{6}$$

where PM = percent pool measure, H = percent high quality pool width, G = percent gravel - cobble width, C = percent bank cover, S = percent bank soil stability, and V = percent bank vegetation stability. Pool measure is a rating of the total sample width in pool and riffle elements (when percent of the stream that is pools (P) is equal to 50 then the rating (PM) is 100, if P < 50, PM = 100-((50-p)x2), if P > 50, PM = 100 – ((p-50)x2). It is assumed that a 1:1 ratio of pools to riffles is the best for trout (USFS 1990). Percent high quality pools is the percent of total width in pools that have pool ratings less than 4 (Table 1). Other variables used to compute HCI are described in Table 1.

Table 1. Habitat characteristics recorded along transects at each 50-m sampling site; definitions from Clarkson and Wilson (1995).

Habitat measure	Description
Channel gradient	Slope (± 0.5%) between transects measured with a clinometer and a stadia rod
Channel width	Distance (± 0.1 m) between banks along transect at the points where bank full discharge is indicated
Stream width	Distance (± 0.1 m) along a transect between shores, including individual substrate particles above water completely surrounded by water
Water depth	Depth (± 0.01 m) recorded at each shore and at 25, 50, and 75% of transect width
Maximum depth	Deepest (± 0.01 m) point along transect
Riffle width	Transect width (± 0.1 m) accounted for by riffle, run, or cascade habitat
Pool width	Transect width (± 0.1 m) accounted for by pool or glide habitat
Boulder width	Transect width (± 0.1 m) accounted for by boulders (256-4,096 mm in diameter)
Cobble width	Transect width (± 0.1 m) accounted for by cobbles (64 - 256 mm in diameter)
Gravel width	Transect width (± 0.1 m) accounted for by gravel (2 - 64 mm in diameter)
Sand-silt width	Transect width (± 0.1 m) accounted for by sand and silt (0.004 - 2 mm in diameter)
Other substrate width	Transect width (± 0.1 m) accounted for by other bottom material (clay, detritus, etc.)
Embeddedness	Percent of gravel and larger substrate perimeter covered or surrounded by sand and smaller substrate within the stream 5 m above and below transect, rated as: 5, < 5%; 4, 5-25;, 3, 26-50%; 3, 51-75%; 1, > 75%
Ungulate damage	Percent of each streambank 5 m above and below transect grazed and trampled by ungulates, rated as: 4, 0-25%; 3, 26-50%; 2, 51-75%; 1, > 75%
Bank soil stability	Percent of each streambank surface 5 m above and below transect covered by vegetation or substrate classes, and percent of bottom that is affected by scouring or deposition, rated as: 4, > 80% plant cover, 65% covered by boulders, < 25% of bank eroding, and <5% of stream bottom affected by scouring and deposition; 3, 50-79% plant cover, 40-65% covered by large substrates (boulder and cobble), < 50% of bank eroding, 5-30% of stream bottom affected by scouring and deposition; 2, < 25% plant cover, 20-40% large substrates (mostly cobble), 50-75% of bank eroding, 30-50% of stream bottom affected by scouring and deposition; 1, < 25% plant cover, < 20% large substrates (mostly pebbles), > 75% of bank eroding, > 50% of bottom affected by scouring and deposition
Bank vegetation stability	Percent of each streambank surface 5 m above and below transect covered by vegetation or substrate classes, rated as: 4, > 80% covered by vegetation or boulders and cobble; 3, 50-79% covered by vegetation or by gravel and larger substrates; 2, 25-49% covered by vegetation or by gravel and larger substrates; 1, < 25% covered by vegetation or by gravel and larger substrates
Bank cover	Class of vegetation on or above each streambank 5 m above and below transect, rated as: 4, shrubs dominant; 3, trees dominant; 2 grasses and forbs dominant; 1, streambank devoid of vegetation cover
Undercut bank width	Distance (± 0.1 m) along transect from furthest protrusion of bank to the furthest undercut of the bank
Bank angle	Angle formed by downward sloping stream bank as it meets the water surface, measured with clinometer and meter stick, ranges from 0 to 180 degrees, with those less than 90 degrees being undercut banks
Canopy density	Percent canopy closure (area of sky over the stream channel that is screened by vegetation), recorded with a densiometer at 4 points (30 cm perpendicular from each shore, and at the center of the transect facing upstream and downstream) 30 cm above water surface
Pool rating	1 = pool length or width greater than average stream width, and pool depth ≥ 0.67 m and abundant cover, or pool depth ≥ 1 m and little to no cover; 2 = pool length or width greater

Habitat measure	Description
	than average stream width, and 0.67 - 1 m depth with little or no cover, or < 0.67 m depth with intermediate or abundant cover; 3 = pool length or width greater than average stream width and pool depth < 0.67 m with little or no cover, or pool length or width equal average stream width and pool depth < 0.67 m with intermediate or no cover; 4 = pool length or width equal to average stream width and depth equal to average stream depth with no cover, or pool length or width less than average stream width, and pool depth is ≤ 0.67 m or average stream depth with intermediate to abundant cover; 5 = pool length or width less than average stream width and pool depth is equal to average stream depth, with no cover

To help interpret results of repeated measures ANOVAs, we used Pearson's correlation to examined relationships between the stressor variable, ungulate damage, and response variables (e.g., Apache trout density and biomass and various habitat measures) using pre-fencing data from all sites (not just the repeated measures data); post-fencing data was not included because ungulate damage decreased to near zero during this period. In addition, we used Pearson's correlation to assess Apache trout-habitat associations using data from all sites with Apache trout in the pre-and post-fencing periods. Only benthos data from the post-fencing period was used in this latter analysis because pre-fencing data was collected with different equipment and summarized differently than the post-fencing data.

Habitat Use

Apache trout habitat use was surveyed in two to three streams each year from 2001 to 2003 in spring (May) and late summer to autumn (August-October; Table 3). In each stream we sampled two or more of the GAWS reaches described above. Each time a reach was sampled, a random location within the reach was selected as the beginning point for sampling. Beginning at that point, surveyors electroshocked, using a Smith-Root Model 15-C backpack unit, upstream in a single pass to the end of the reach or until Apache trout were captured in 10 separate sites. Sites were defined by habitat type (e.g. riffle, run, cascade, and pool) as described in McCain et al. (1990) and Bisson and Montgomery (1996). At times, fish were captured at places where two or more habitat types were found across the width of the stream, in which case all habitat types were recorded.

Environmental characteristics were measured within each site along five perpendicular-to-flow transects, placed so that they encompassed the area where fish were first observed, rather than at the capture point, to minimize the effects of electrotaxis (Gatz et al. 1987). Transects were spaced 0.5 m apart, with the central transect bisecting observed fish location. If multiple fish were captured within the site and the length of the habitat was greater than 2.0 m, additional transects were added at 0.5-m intervals. Width (cm) of the stream at each transect was recorded. Depth (cm), current velocity (cm/s), and presence and types of substrate (bedrock, boulder, cobble, pebble, gravel, sand, silt, and debris), and presence and types of cover (undercut bank, instream vegetation, over-hanging vegetation, woody debris, or boulder) were recorded at five points along each transect at 0.25, 0.5 and 0.75 times the width of each transect and 10 cm from each shoreline (minimum of 25 points per site). To describe available habitat, a random location within 50 m of each fish capture site was measured for environmental characteristics in the same fashion as used sites (at five points along five perpendicular to flow transects spaced 0.5 m apart). Sample sizes are indicated in (Table 4).

Number of Apache trout captured per site, total length (mm) and weight (g) of each fish were recorded. Fish > 99 mm TL were injected with PIT tags for the barrier evaluation study (described below). After processing, fish were returned alive to the site from which they were captured.

Data Analyses - We used a three-way multivariate analysis of variance (MANOVA) and subsequent univariate ANOVAs to evaluate differences in used and available habitat among streams and seasons. Substrate categories were ranked as (1 = silt, 2 = sand, 3 = gravel, 4 = pebble, 5 = cobble, 6 = boulder, and 7 = bedrock) prior to analysis. Means (width, depth, current velocity, ranked substrate size, and width:depth ratio) and percents (percent of transect points with cover, each cover type, and eddy flows) for environmental characteristics measured at each site were calculated for each season. Percents were arc-sine transformed prior to analysis to address assumptions of normality. These means and arc-

Table 2. Number of sites surveyed in each reach of each stream, years surveyed before fencing, and year fenced; each site was surveyed post-fencing during 2001, 2002, and 2003. According to Clarkson and Wilson (1995) reaches with slopes < 2% were classified as meadows and those 2-6% were classified as intermediate.

Stream	Reach	Mean gradient (%)	# sites	Years surveyed before fencing		Year Fenced
Coyote	4	2.9	3	1990	1995	1996
	6	1.7	3	1990	1995	1996
Soldier	3	4.5	3	1989	1996	1999
Mineral	1	4.4	1	1991	1996	1996
Conklin	3	3.8	2	1988	1995	1996
Corduroy	2	3.2	3	1987	1995	1996
	3	3.0	2	1987	1995	1996
Double Cienega	2	3.0	2	1987	1995	1996
	3	1.5	1	1987	1995	1996
Fish	3	3.5	8	1987	1995	1996
	4	3.2	2	1987	1995	1996

sine transformed percents of the dependent variables were used in the three-way MANOVA. Univariate ANOVAs were evaluated if the MANOVA (Wilk's lambda) had a $P < 0.05$. We considered nonsignificant results to indicate that Apache trout used habitat similar to what is available. We examined differences in used versus available habitat among streams for four Apache trout size classes (0-99, 100-149, 150-199, >200 mm TL) with MANOVA, but found no significant differences among size classes, and therefore, did not include size class in our analysis.

We used contingency table analysis and G-tests (Sokal and Rohlf 1981) to compare used versus available habitat types (i.e., pool, run, riffle, etc) for each stream.

Barriers

We considered a barrier to have failed to serve its purpose if nonnative salmonids were found above the barrier. We used two approaches to detect barrier failure.

Historical Evaluation. We reviewed historical fish survey data (Arizona Game and Fish Department unpublished data) to determine if non-native salmonids were captured above a barrier after renovation and re-stocking. For streams with two barriers (West Fork Black River and Home Creek), the presence of nonnative salmonids above each barrier was noted. We counted Lee Valley Creek twice because it had been renovated twice; therefore our total sample size was 13 barriers. We quantified barrier failure percent as the ratio of the number of failed barriers to total number examined multiplied by 100. We broke the barrier failure rate into three categories: low was 0-33.3%, moderate was 34-66.6% and high 67-100%.

Mark-Recapture. For the mark-recapture component of our study, we captured and marked salmonids below barriers and subsequently surveyed above the barriers to detect marked fish. We first marked fish in autumn, 2000, and during all subsequent trips (spring and autumn 2001, autumn 2002, and summer and autumn 2003) we marked fish below the barriers and surveyed above barriers.

We captured and marked salmonids within the reach that extends downstream from the barrier for a distance of 300 m, or until the mouth of the stream (whichever was shorter). Because more fish species and a greater abundance of fish were present downstream from barriers, we divided the 300-m reach into three 100-m sections to more thoroughly sample the reach and to reduce stress on the fish. The reach was divided into sections with block nets (3 mm–mesh seines) set across the width of the stream. Each section was electrofished (Smith-Root model 15-C backpack shocker) from downstream to upstream using two passes. All salmonids were weighed (± 1 g) and measured for

Table 3. Habitat use study streams, reaches, type of reach, year fenced, and year sampled. Meadows reaches have wide valleys with primarily grass vegetation and mean gradient < 2%, intermediate reaches have narrow U-shaped valleys with mean gradient 2-6% and a variety of vegetation types, and canyon reaches have V-shaped valleys with mean gradient > 6% and a variety of vegetation types. Grazing (G) is still permitted on some streams; GS = short term (< 1 month per year) grazing still permitted.

Stream	Reach	Reach Type	Year fenced	Year sampled
Coleman	3	Canyon	1992	2002
Coleman	4	Canyon	1992	2002
Coyote	3	Canyon	G	2003
Coyote	4	Meadow	1996	2003
Mineral	1	Intermediate	1996	2001
Mineral	2	Canyon	1987	2001
Mineral	3	Intermediate	1987	2001
Soldier	1	Canyon	1999	2003
Soldier	2	Canyon	1999	2003
Soldier	3	Meadow	1999	2003
Burro	1	Meadow	G	2002
Burro	2	Meadow	G	2002
Thompson	1	Meadow	GS	2002
Thompson	2	Meadow	GS	2002
Thompson	3	Meadow	GS	2002
West Fork Black	7	Meadow	GS	2002
West Fork Black	8	Canyon	GS	2002

total length (± 1 mm). Salmonids ≥50 mm were marked with a coded-wire tag injected into the caudal peduncle and the adipose fin was clipped. Apache trout in Conklin and Fish creeks and the West Fork Black River were scanned for PIT tags in order to detect downstream movement past barriers; marked upstream from barriers during the fencing evaluation portion of this study. Apache trout captured downstream from barriers in all streams were scanned for PIT tags. After processing, all fish were placed back into the stream alive.

Surveys for marked fish above the barriers were conducted in spring (2001 and 2003) and autumn (2001-2003). On each stream, we made one pass with the electrofisher from the barrier upstream 500 - 800 m. All salmonids captured were scanned for a coded-wire tag and processed as above. In order to better detect downstream down stream movement of Apache trout, a PIT tag was injected into all unmarked Apache trout ≥100 mm total length captured above barriers during summer 2003. All other fish species were processed as above. Native fish were released back into the stream alive. Nonnative salmonids were sacrificed.

Table 4. Number of used and available sites sampled during spring and autumn in six White Mountain streams, Arizona. No Apache trout were captured in Burro Creek, so no used or available sites were sampled.

	Spring		Autumn		Total
Stream	A	U	A	U	
Coleman	20	20	18	18	76
Coyote	13	13	11	11	48
Mineral	6	6	10	10	32
Soldier	17	17	20	20	74
Thompson	20	20	30	30	100
West Fork Black	20	20	20	20	80
Total	96	96	109	109	410

Bear Wallow and Snake creeks were chemically treated with antimycin to remove all fish above barriers just after our last survey (autumn 2003). All salmonids obtained during these renovations were scanned for coded-wire tags and measured for total length (mm). Apache trout ≥100 mm were scanned for PIT tags.

Data Analyses. For Apache trout captured during our mark-recapture surveys, we examined the effect of constructed barriers on condition and size structure. We calculated Fulton's condition factor and then compared condition above to below barriers for each stream and year with t-tests. Apache trout were categorized into 10-mm length classes and length frequency histograms were plotted for fish above and below barriers. To assess differences in size structure, the frequency of fish ≤ 80 mm TL and > 80 mm TL was compared above to below barriers for each year on each stream (where sample sizes were large enough) with G-tests (Sokal and Rohlf 1981).

RESULTS

Fencing

Apache trout densities and biomass were greater before fencing than after fencing (Figure 2). We did not calculate a repeated measures ANOVA of Apache trout condition (K) because of missing data from the first pre-fencing surveys. However, paired t-tests of condition between the second pre-fencing survey (1995 - 1996) and the first (2001), second (2002), and third (2003) post-fencing surveys all yielded insignificant ($P > 0.05$) differences.

Few environmental variables differed from pre- to post fencing. Ungulate damage, embeddedness, percent bank cover, and the habitat condition index decreased after fencing (Figures 2 and 3); 83% of the sites had observable ungulate damage before fencing compared to 15% after fencing. Percent riffles and the width:depth ratio increased significantly from pre- to post-fencing, but values from the second pre-fencing survey were virtually identical to post-fencing values (Figure 2). None of the other environmental variables differed between pre- and post-fencing periods (Figures 2 and 3).

Although we did not detect many differences in variables of interest between pre- and post-fencing periods, correlation analysis of ungulate damage with Apache trout and environmental variables using pre-fencing data from all sites (including canyon reaches) yielded interesting results (Table 5). Ungulate damage was positively correlated with percent embeddedness, bank angle, and the pool:riffle ratio, and negatively correlated with Apache trout condition, undercut bank width, percent undercut banks, percent bank cover, percent bank vegetation stability, percent bank soil stability, percent canopy density, riparian condition, habitat condition index, shore depth, mean maximum depth, percent rubble- boulder substrates, percent riffles, gradient, and invertebrate densities (Table 5); although sample size for invertebrate densities is low.

Examination of all data from both periods showed that Apache trout measures were correlated with a variety of habitat measures but none of the

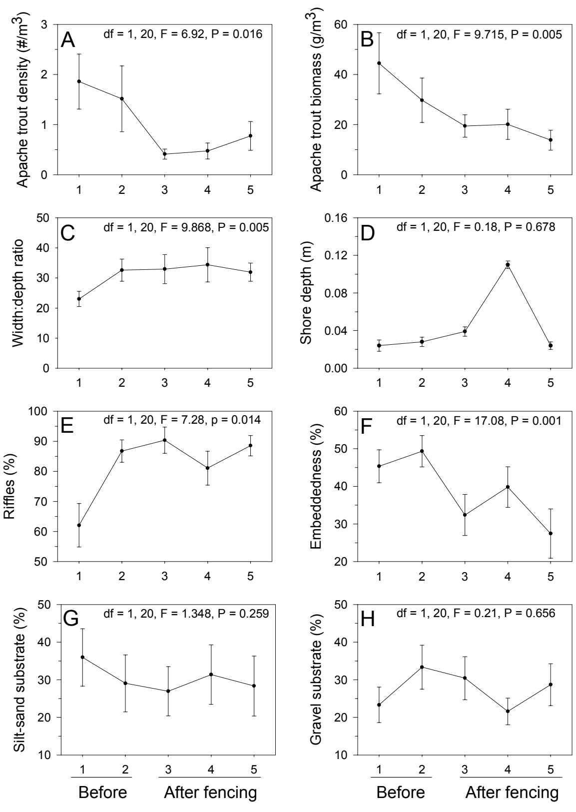

Figure 2. Means and standard errors of Apache trout and aquatic habitat characteristics measured on six White Mountain, Arizona streams before and after livestock were excluded from streams.

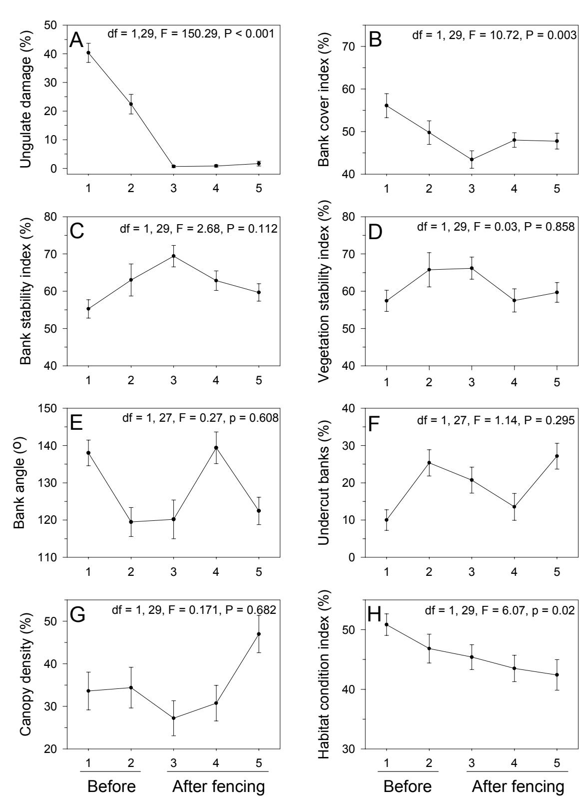

Figure 3. Means and standard errors of riparian habitat characteristics measured on six White Mountain, Arizona streams before and after livestock were excluded from streams.

Table 5. Correlations of ungulate damage index with Apache trout and habitat variables before fencing.

Variable	N	r	P
Apache trout density (#/m^3)	116	-0.09	0.313
Apache trout biomass (g/m^3)	116	-0.11	0.244
Apache trout condition (K)	88	-0.24	0.022
Apache trout maximum total length (mm)	117	0.02	0.813
Embeddedness (%)	161	0.29	<0.001
Undercut bank width (m)	160	-0.18	0.021
Undercut banks (%)	158	-0.27	0.001
Bank angle (º)	158	0.21	0.007
Bank cover (%)	163	-0.35	<0.001
Bank soil stability (%)	163	-0.62	<0.001
Bank vegetation stability (%)	163	-0.59	<0.001
Canopy density (%)	163	-0.38	<0.001
Overhanging vegetation (%)	109	-0.33	0.001
Riparian area (ha)	162	0.15	0.061
Riparian condition	98	-0.42	<0.001
Habitat condition index (%)	163	-0.39	<0.001
Shore depth (m)	158	-0.31	<0.001
Mean maximum depth (m)	134	-0.29	0.001
Mean depth (m)	158	-0.11	0.163
Width:depth ratio	158	-0.01	0.942
Water width (m)	158	-0.04	0.611
Gravel substrate (%)	158	0.14	0.074
Rubble-boulder substrate (%)	158	-0.20	0.013
Silt substrate (%)	158	0.08	0.323
Percent pools	158	0.15	0.068
Pool measure (%)	158	0.04	0.586
Percent high quality pools	158	-0.13	0.115
Percent riffles	158	-0.20	0.012
Pool:riffle ratio	153	0.23	0.005
Channel gradient (%)	105	-0.36	<0.001
Invertebrate density (#/m^2)	13	-0.58	0.039
Invertebrate biomass (g/m^2)	13	0.16	0.608

significant ($P < 0.05$) correlation coefficients were greater than 0.4 (Table 6). Apache trout densities (#/m^3) were negatively correlated with percent undercut banks, shore depth, maximum depth, mean depth, and water width, and positively correlated with bank soil stability and bank vegetation stability. Apache trout biomass (g/m^3) was positively related to embeddedness, bank vegetation stability, and percent silt-sand substrates and negatively related to maximum depth, mean water depth, water width, and percent of rubble and boulder substrates. Apache trout condition was positively correlated with percent undercut banks. Maximum length of Apache trout was positively correlated with embeddedness, maximum depth, and mean water depth.

During the post-fencing period, Apache trout densities (#/m^3) were positively related to invertebrate density (#/m^2; r = 0.80, P < 0.001, N = 22) and Apache trout biomass (g/m^3) was positively related to invertebrate densities (#/m^2; r = 0.53, P < 0.011, N = 22), but no significant correlations with Apache trout condition or maximum length and invertebrate densities and biomass were detected. Apache trout densities or biomass were not significantly correlated (all P > 0.05) with the proportion of terrestrial to total insect densities or biomass.

Stream flows were mostly above average during the pre-fencing period, whereas the post-fencing period was dominated by less than average stream flows (Figure 4). Of the 31 GAWS sites sampled, two were dry in 2001 (one each in Conklin and Corduroy creeks), eight in 2002 (three in Double Cienega Creek and five in Corduroy Creek), and four in 2003 (three in Double Cienega Creek and one in Corduroy Creek). None of the sites were dry during the pre-fencing surveys.

Habitat Use

Habitat characteristics at sites with Apache trout differed from available sites and differed among streams and seasons; the three-way MANOVA yielded significant (P < 0.05) stream, availability-use, and season main effects, and significant interactions between stream and availability-use. The two-way interaction between availability-use and season was not significant, nor was the three-way interaction among season, stream, and availability-use, indicating that Apache trout habitat use did not differ between seasons.

Univariate tests for availability-use main effects showed used sites were significantly wider and deeper, had slower current velocities, more percent eddy flows, lower width:depth ratios, more percent boulder and undercut bank cover, and less instream vegetation cover than available sites (Figures 5 and 6), but there were no differences between used and available sites for ranked substrate size and percent total cover. However, significant interactions between availability-use and stream were found for stream width (F = 4.096, P =

Table 6. Correlations of Apache trout and habitat measures in White Mountain streams, all sites, 1987-2003. Only environmental variables with significant ($P < 0.05$) correlations are shown.

Environmental variable		Density (#/m^3)	Biomass (g/m^3)	K	Maximum total length (mm)
Embeddedness (%)	*r*	0.06	0.15	-0.14	0.25
	P	0.425	0.030	0.075	0.000
	N	205	205	159	207
Undercut banks (%)	*r*	-0.14	-0.09	0.17	0.01
	P	0.044	0.183	0.038	0.877
	N	205	205	158	206
Bank soil stability (%)	*r*	0.17	0.12	0.06	-0.07
	P	0.018	0.079	0.418	0.347
	N	205	205	159	207
Bank vegetation stab. (%)	*r*	0.16	0.16	0.13	0.01
	P	0.018	0.022	0.103	0.891
	N	205	205	159	207
Shore depth (m)	*r*	-0.17	-0.08	0.15	0.06
	P	0.014	0.278	0.062	0.387
	N	205	205	158	206
Maximum depth (m)	*r*	-0.36	-0.31	0.10	0.23
	P	0.000	0.000	0.207	0.002
	N	179	179	153	179
Mean water depth (m)	*r*	-0.31	-0.27	0.09	0.30
	P	0.000	0.000	0.237	0.000
	N	205	205	157	205
Width:depth ratio	*r*	-0.07	-0.10	-0.06	-0.12
	P	0.317	0.167	0.426	0.082
	N	205	205	157	205
Water width (m)	*r*	-0.34	-0.38	0.04	0.12
	P	0.000	0.000	0.639	0.082
	N	205	205	158	206
Rubble-boulder substrate (%)	*r*	-0.09	-0.19	-0.06	-0.07
	P	0.211	0.006	0.468	0.325
	N	205	205	158	206
Silt substrate (%)	*r*	0.05	0.21	0.07	0.12
	P	0.489	0.002	0.386	0.083
	N	205	205	158	206
Percent high quality pools (ratings < 4)	*r*	-0.09	0.03	0.05	0.03
	P	0.194	0.694	0.528	0.684
	N	205	205	158	206

Environmental variable		Density (#/m^3)	Biomass (g/m^3)	K	Maximum total length (mm)
Habitat condition index	*r*	0.03	0.02	0.05	-0.01
	P	0.704	0.740	0.545	0.846
	N	205	205	159	207
Channel gradient	*r*	0.04	0.03	-0.01	0.01
	P	0.640	0.750	0.885	0.908
	N	143	143	121	144

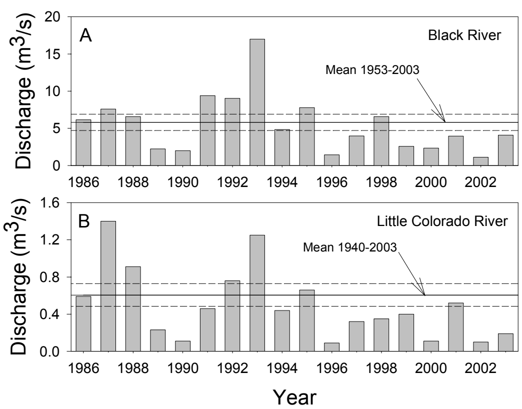

Figure 4. Mean annual discharge in the Black River at USGS gage 09489500 below the pumping plant, near Point of Pines, and in b) the Little Colorado River at the USGS gage 09384000 above Lyman Lake near St. Johns. The Black River is representative of Fish, Double Cienega, Corduroy, Conklin, and Soldier creeks, whereas the Little Colorado River is representative of Coyote Creek and Mineral creeks.

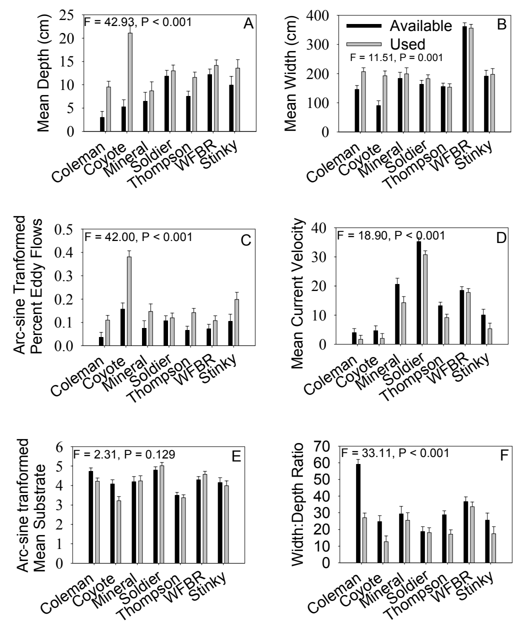

Figure5. Habitat characteristics at sites with Apache trout and at available sites in six study streams. The F and P values are for the univariate ANOVA main effects comparing available to used habitat characteristics; df = 1 and 413 for all tests.

0.001), depth (F = 7.130, $P < 0.001$), ranked substrate size (F = 2.812, $P = 0.017$), percent eddy flows (F = 4.672, $P < 0.001$), and the width:depth ratio (F = 7.828, $P < 0.001$); df = 5, 381 for all. In all six streams, used sites were deeper, had more % eddy flows, and less width:depth ratios than available streams, but the magnitude of these differences differed among streams (Figure 5). Used sites were wider than available sites in Coleman, Coyote, Mineral, and Soldier, but slightly narrower than available sites in Thompson Creek and West Fork Black River (Figure 5). Used sites had smaller substrates than available sites in Coleman, Coyote, Stinky, and Thompson creeks, but slightly larger substrates in Mineral and Soldier creeks and West Fork Black River (Figure 5).

Categorical analysis of habitat type showed that used sites differed from available sites in proportions of habitat types in each stream (Figure 7). In all streams except the West Fork Black River, used sites had a greater proportion of pool habitat than did available sites. In West Fork Black River used sites had more complex habitat than available sites.

Visual examination of Figures 5 - 7 indicated that available habitat in streams that are still grazed (West Fork Black, Thompson, and parts of Coyote) was similar to that in streams where no grazing is permitted, or at least no obvious pattern could be

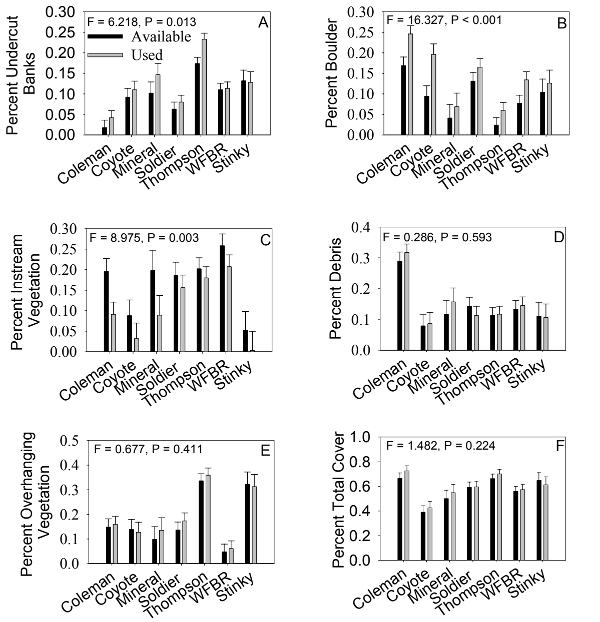

Figure 6. Mean arc-sine transformed percents of each cover type at sites in six study streams. The F and P values are for the univariate ANOVA main effects comparing available to used habitat characteristics; df = 1 and 413 for all tests.

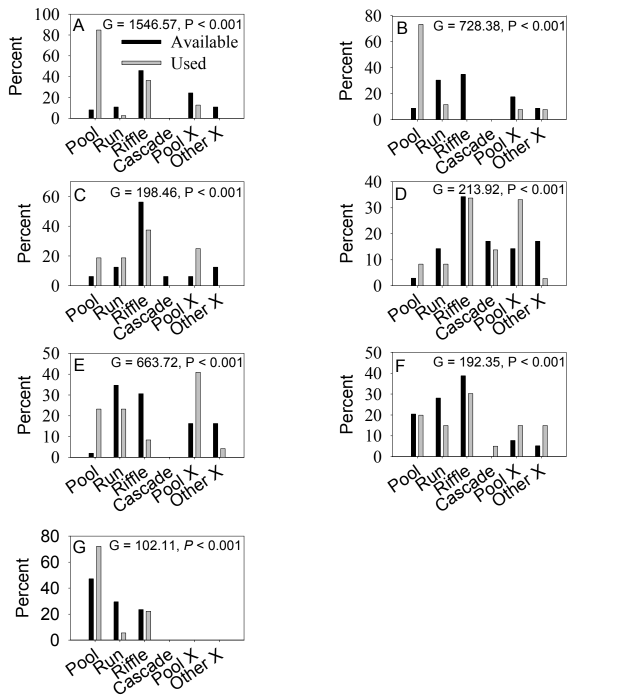

Figure 7. Percentage of habitat types in used and available sites in each stream: a) Coleman Creek, b) Coyote Creek, c) Mineral Creek, d) Soldier Creek, e) Thompson Creek, f) West Fork Black River, and g) Stinky Creek. Pool X refers to the combination of pool plus another habitat type, and Other X refers to any other combination of habitat types. The G and *P* values are for the test comparing frequency of habitat types between used and available sites.

detected. The patterns of used versus available habitat were similar in all streams, regardless of whether or not they were still grazed.

No Apache trout were captured in Burro Creek, and so no used or available sites were measured. Reaches 1 and 2 on Burro Creek had greater grazing pressure than any other reach on any stream.

Barriers

Historical Data. Subsequent to renovation and re-stocking with Apache trout, non-native salmonids were found above barriers on Stinky, Hayground, and Bear Wallow creeks, above both barriers on West Fork Black River, and above the barrier on Lee Valley creeks after both renovations. This represents a 64 % failure rate. Most of these barriers were in need of repair or reconstruction when non-native trout were captured above them. The upper West Fork Black River barrier had small leaks that were not considered to have compromised the integrity/success of the barrier.

Mark-recapture study. A total of 1,436 salmonids were marked with CWT and adipose fin clips below barriers in the six study streams (Table 7). One CWT Apache-rainbow hybrid (268 mm TL) was captured ~8 km above the barrier in Fish Creek in August 2002. One CWT brown trout (195 mm TL) was captured ~ 300 m above the Fish Creek barrier in August 2003. No CWT fish were captured above barriers on any other streams and none of the salmonids scanned following the renovations of Snake and Bear Wallow creeks during autumn 2003 had CWT.

Condition of Apache trout ≥ 100 mm TL above barriers was only found to be significantly different from that below barriers in two instances. Condition of Apache trout below barriers was less than that above barriers in Snake Creek during 2001 (n = 71, $P = 0.003$) and in West Fork Black River during 2002 (n = 165, $P = 0.02$); condition below was 1.0 ± 0.09 and 0.92 ± 0.10 respectively and above was 1.1 ± 0.14 and 1.04 ± 0.12 respectively. A greater proportion of smaller fish (< 80 mm)

Table 7. Total of marked and recaptured salmonids below 6 artificial barriers on 6 streams.

	Number of Salmonids below barriers	
Stream	Marked	Recaptured
Bear Wallow	95	18
Conklin	264	18
Fish	294	3
Snake	88	16
Stinky	84	17
West Fork Black River	611	165
Total	1436	237

were found below the barrier than above the barrier (Figure 8) in Conklin Creek during 2001 and 2003, in Fish Creek during 2003 and in West Fork Black River during 2002. In Snake Creek during 2003 the opposite was found with 88% of the fish above the barrier being ≤ 80 mm TL. No other significant differences were found in 6 other contingency tables analyses.

Conklin Creek below the barrier was dry at the time of our survey in autumn 2002. All brown trout were mechanically removed from Stinky Creek in June 2002, and were inadvertently not checked for coded wire tags before being moved to another stream, therefore subsequent surveys above the barrier were not conducted in autumn 2002 or 2003.

Apache Trout Movements. Four hundred and ninety one Apache trout were PIT tagged above the study barriers; 57 above the Conklin Creek barrier, 320 in Fish and Corduroy creeks above the Fish Creek barrier, 18 above the Stinky Creek barrier, and 96 above the upper West Fork Black River barrier. No PIT tagged fish were captured below any of the barriers; i.e., we did not detect downstream movement of Apache trout past barriers. Our sample size is relatively low for Stinky Creek, and many of the fish marked in the GAWS sites on the other streams were several kilometers above the barriers, potentially hindering our ability to detect downstream movement past the barriers.

In our fencing and habitat use evaluation, we also PIT marked 40 Apache trout in Coyote Creek, 141 in Soldier Creek, 40 in Coleman Creek, and 23 in Mineral Creek, for a total of 735 in all study streams. Only 23 fish were recaptured; nine each in Fish and Soldier creeks, three in Coyote Creek, and one each in Stinky and Conklin creeks. Of these, 14 did not move (captured within 50 m of where they were marked). For the nine fish that moved,

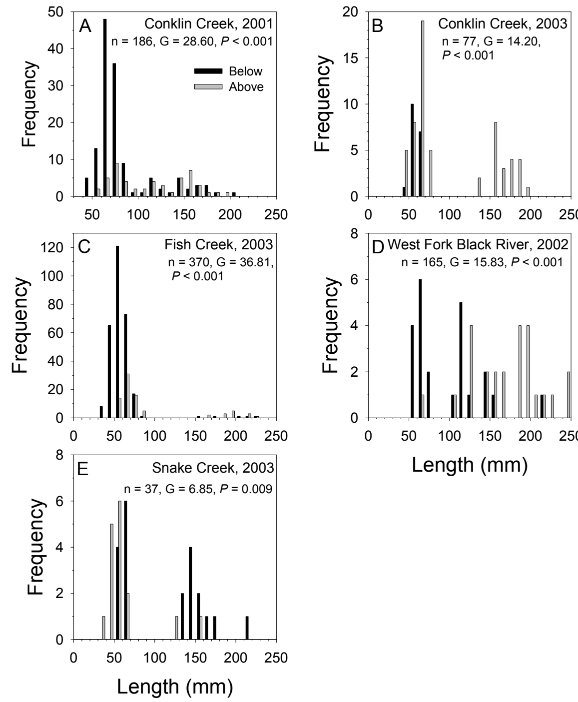

Figure 8. Length frequencies of Apache trout above and below barriers in a) Conklin Creek during 2001, b) Conklin Creek during 2002, c) Fish Creek during 2003, d) West Fork Black River during 2002, and e) Snake Creek during 2003. Results (G-tests with P-values) of comparisons of frequency of fish ≤ 80 mm and > 80 mm between below and above barriers is given.

seven moved upstream (90, 200, 390, 390, 440, 580, and 590 m), and two moved downstream (500 and 510 m).

DISCUSSION

Evaluation of Fencing

As expected, ungulate damage decreased dramatically following livestock exclusion, even though elk had free access to study reaches before and after exclusion. As predicted, embeddedness of cobble and boulder substrates decreased following fencing. However, contrary to our predictions, we did not detect any other positive effects of excluding livestock on Apache trout or on Apache trout habitat. As such, our study lent little support to conventional wisdom (Fausch et al. 1988; Platts 1991; Fleischner 1994) that grazing negatively impacts fish populations and stream habitat. It may be that grazing levels prior to fencing were not high enough to negatively impact stream habitat and hence Apache trout production. However, we do not believe this to be the case because during the pre-fencing period, relationships between ungulate damage and habitat variables met several of our predictions, although most correlations were relatively weak ($r < 0.5$). For instance, Apache trout biomass and condition increased as ungulate damage decreased, supporting predictions 1 and 2. Stream morphology–grazing relationships were as expected; percent undercut banks and undercut bank width, bank stability, shore depth, maximum depth, percent riffles, gradient, and the habitat condition index increased with decreasing ungulate damage (prediction 3). Percent bank cover, canopy density and undercut banks increased (prediction 4), as did riparian condition (prediction 5) and invertebrate density (prediction 6) with decreasing ungulate damage.

Why then did we not continue to see these relationships following livestock exclusion? Negative effects of drought likely overshadowed any positive effect of excluding livestock from Apache trout streams. Drought can reduce numbers of fish in a stream or even result in localized extirpations (Matthews and Marsh-Matthews 2003), and can hinder plant establishment and growth. Eastern Arizona experienced a drought during our post-fencing evaluation period. Large sections of some of our streams went dry in 2002 and part of 2003, decreasing the amount of available habitat for Apache trout, and possibly resulting in mortality of some individuals. Even the years of the second pre-fencing survey had precipitation below average. Many of the predicted changes (model predictions 3 - 6) are dependent on recovery of riparian vegetation to stabilize banks, provide cover, and provide food and substrate for insects. Because of the drought, five to eight years may not have been enough time to detect improvement in fish populations and habitat conditions in response to livestock exclusion.

Our data indicate that Apache trout responded to changes in habitat from pre- to post-fencing periods as expected. Similar to Wada (1991) and Kitcheyan (1999), our habitat use results indicated that Apache trout select for pools and pool-like habitat, similar to rainbow trout (*Oncorhynchus mykiss*) and cutthroat trout (*Oncorhynchus clarki*) (Hearn and Kynard 1986; Rosenfeld and Bass 2001; Dare et al. 2002). We found Apache trout used habitat that was wider, deeper, had slower current velocities, more percent eddy flows, lower width:depth ratios, more percent boulder and undercut bank cover, and less instream vegetation cover than what was available. Therefore, because percent pools decreased (inverse of percent riffles) and the width:depth ratio increased from pre- to post-fencing period, less suitable habitat was available in the post-fencing period and Apache trout densities and biomass decreased.

Results of our fish-habitat association analysis of the GAWS data seemed contradictory to our habitat use results, but still suggest the importance of pools. Depth was negatively correlated with Apache trout densities ($\#/m^3$), and positively correlated with Apache trout maximum length, indicating that a few big fish tend to occupy the deepest pools, possibly excluding smaller conspecifics, a pattern seen in other salmonids (Moyle and Baltz 1985; Fausch and White 1986; Hughes 1992). Similarly, the negative correlation of Apache trout density ($\#/m^3$) with percent undercut banks, and the positive correlation of Apache trout condition with percent undercut banks, indicate that few fish that are in good condition are found in sites with undercut banks.

Evaluation of Barriers

Constructed barriers have been moderately successful at preventing the upstream invasion of non-native salmonids into Apache trout waters. In the six streams where barriers failed, we hypothesize that the presence of non-native fishes above the barriers was primarily due to volitional movement by fish rather than angler transport. Bear Wallow Creek is remote, so angler transport is a less likely mechanism of non-native invasion than volitional movement of the non-native fishes past this barrier. Stinky, Lee Valley, Fish and Hayground creeks require short hikes (more than 1.5 km) to access the barriers so angler transport of non-native fish above these barriers is possible but not as probable. Angler transport of non-native fishes upstream of the barriers is most likely in the West Fork Black River, a popular fishing area readily accessible to anglers. However, all of the barriers that failed were either in obvious need of structural repair or had design flaws (not tall enough or the interstitial spaces among cobbles had not yet filled with fine sediment (evidenced by flow of water through the barrier), and hence it is likely that non-native salmonids found above these barriers moved on their own accord.

Failure of constructed barriers to prohibit invasion of non-native salmonids has been reported in other studies. Young et al. (1996) reported that non-native trout breached barriers in 20 Colorado River cutthroat trout streams. Improper design and maintenance may have enabled brook trout to scale some barriers (Young et al. 1996). The barrier on Fish Creek was constructed at the top of a small step cascade, and the vertical rise from the last step to the top of the barrier was only 1 m, which would hardly be a barrier during high flow events; this flaw was noted in the draft environmental assessment for barrier construction and renovations (unpublished environmental assessment, US Forest Service) and subsequent to our study the barrier height was increased by 0.6 m. In addition, there were deep pools below the barriers on Snake Creek and Bear Wallow creeks, which may make it possible for fish to jump over these barriers. Barriers have been reported to lose their physical integrity following high stream flows (Avery 1978; Thompson and Rahel 1998). A flood in 1983 damaged the barrier on Bear Wallow Creek, and non-native rainbow and cutthroat trout (*Oncorhynchus clarki*) hybrids invaded before repairs were made.

Thompson and Rahel (1998) consider it especially important to minimize the size of interstitial spaces in gabion barriers by selection of appropriate rock size so silt and gravel can fill the remaining spaces between rocks. They concluded that brook trout were able to move upstream through the rocks of the gabion barrier because fine sediments had not filled in all the interstitial spaces. In our study, water flowed through the gabions in barriers that were not reinforced with mortar. For instance, water was often noticed flowing under the lower barrier on West Fork Black River, rather than over the spillway, necessitating barrier maintenance to stop the sub-spillway flows. Brown trout made it above this barrier, either by moving between the rocks, or by being transported by anglers. The downstream drop on this barrier was over two meters in height, and rocks placed at the bottom prevent pool formation below the barrier, so brown trout likely did not jump over this barrier. Stinky Creek also had leaks through the barrier.

There is little evidence that abundances and condition of threatened fishes are greater above than below isolation barriers. No increase in body condition or abundance was found in cutthroat trout in Wyoming with removal of brook trout above the barriers (Moyle and Sato 1991; Novinger and Rahel 2003). Smith and Laird (1998) found abundance and biomass of fishes above waterfalls were significantly lower than all other sites. In our study, condition of Apache trout did not differ above compared to below the barriers, except in two cases. In both cases, condition was lower downstream than upstream of the barrier, as might be expected due to negative indirect effects of predatory brown trout or competition for resources. We also found a greater proportion of smaller fish (≤ 80 mm) below barriers than above in some of our streams. We expected brown trout would preferentially prey on smaller Apache trout, so this result was opposite our expectations. It may be that adults that migrate upstream to spawn are stopped at the barrier and spawn there, resulting in the observed size distribution. However, Apache trout fry are reported to drift downstream (Harper 1978), so we would expect small fish to be more prone to downstream movement than large fish. Small fish that moved below the barrier would be prevented from moving upstream and hence smaller fish may accumulate downstream, resulting in the observed distributions. However, we expect a low survival to adult size in the presence of non-native trout (competition and predation). Streams that periodically go dry below the barriers, such as Conklin Creek, may tend to have smaller fish downstream from the barriers if most fish are dispersing from upstream, and if most dispersers are small fish; most of our PIT marked Apache trout did not move, but of those that did, 78% moved upstream. Fish that disperse downstream past barriers represent a loss of genetic material to the gene pool upstream, similar to a loss due to mortality. Conservation of a species depends on the preservation of its genetic diversity (Allendorf and Leary 1998).

The monetary costs of barrier strategy include not only the original construction of the barrier and stream renovation, but also periodic inspections and maintenance, and further actions when a barrier fails. A failed barrier may require minor or major repairs or total reconstruction. After a failed barrier has been repaired or modified, all fish must be removed from the upstream reach. Mechanical removal (e.g., electrofishing and netting) is not a cost effective means to remove all non-native competitors or predators. Five removals were required to successfully eliminate rainbow trout from a small southern Appalachian stream (Kulp and Moore 2000). The size and location of the stream, manpower needs, and the presence of sensitive native fish dictate the efforts involved to remove all non-native fish from a stream. However, if non-native invaders hybridize with the native species (as e.g. rainbow trout and Apache trout), or if a stream reach is to large and complex to ensure total removal of non-native competitors or predators, then renovation of the stream is necessary to ensure total removal of all non-native fishes. Subsequent to renovation, electrofishing surveys need to be conducted several times spanning several months or up to a year to ensure that all fish have been eliminated. Finally, fish have to be acquired either from a hatchery or relict donor population and restocked into the fish free waters above the barrier. Monetary costs accumulate with subsequent barrier failures. Cost of construction varies by size, location, proximity of rock source, type of barrier and contractor. Fish barriers in Arizona have ranged anywhere from US$150,000 for gabion barriers (personnel communication Scott Gurtin, Arizona Game and Fish Department native trout coordinator) to $3 million for a large solid concrete barrier on Aravaipa Creek. Barrier repairs on Apache trout streams have ranged from $3000-15,000. Compliance with the National Environmental Policy Act (NEPA) also substantially increases cost.

Direct costs of barrier failure due to angler transport are likely less than structural failure because structural repairs are not needed. However, indirect costs of increased law enforcement, stream closures, signs, and public education to solve the problem of anger transport will likely be significant and possibly more than costs of barrier repair and chemical re-treatment.

MANAGEMENT OPTIONS

Fencing

Apache trout biomass and densities actually decreased as ungulate damage decreased from pre- to post-fencing periods, but as mentioned before we think this is because available habitat decreased. We do not suggest that grazing had a positive effect on Apache trout habitat and production, but rather other factors such as drought had negative effects that overweighed any positive effects of excluding livestock. Because livestock have a direct effect on riparian vegetation and habitat, but only indirect effect on stream fishes, detecting a cause-and-effect relationship is problematic (Rinne 1999). Control sites (grazed throughout the study) or reference sites (ungrazed throughout the study) may have helped us tease apart the effects of the drought from those of grazing. We recommend that the livestock exclosures be maintained and the same sites in our study be monitored periodically in the future beyond the drought to detect long-term trends in Apache trout and habitat measures following exclusion of livestock.

Barriers

Although constructed barriers play a vital role in recovery of Apache trout, we question the effectiveness of gabion barriers. Our failure rate was moderate, but every time a barrier fails, action must be taken. Barrier failures have species conservation costs in addition to monetary costs. The recovery strategy (USFWS 1983) for Apache trout entails replication of relict populations into streams with constructed barriers. When a barrier fails and a renovation is required, the entire replicate is lost. It can take a year or more to replace a replicate. Filling in interstitial spaces and covering the entire gabion with concrete may minimize the chance that fish can pass upstream through the rocks in the barrier, and also increase the life of the barrier, and it would eliminate the possibility that fish are moving through interstitial rock spaces. A solid concrete, backfilled dam (so no upstream pool is created) may also have a longer life and require less maintenance than a gabion barrier. However, a solid concrete barrier would have a higher monetary cost, and getting equipment and supplies into remote streams would be problematic.

Angler transport might be minimized by restricting vehicle access, as was done on Stinky Creek (the road into and along the creek was closed), or by changing regulations, as was done for the West Fork of Black River (fishing was closed between the barriers and for 100 m below the lower barrier). Education may be the best means of minimizing angler transport, because a permanent law enforcement officer on site is not feasible. However, it is feasible to increase visitation by law enforcement. Installing remote cameras with motion control switches might be a way to increase surveillance without increasing officer visitation.

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For more detailed and technical presentations of methods, results and discussion of specific sections of this Technical Guidance Bulletin, the authors refer you to the following citations, which can be obtained by contacting:

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