2001 Fish Investigations in the Lee's Ferry Tailwater

2001 Annual Report

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EXECUTIVE SUMMARY

We present results of rainbow trout monitoring activities in the Lee's Ferry tailwater (Colorado River below Glen Canyon Dam, AZ) during 2001. We also describe modifications made to monitoring strategies and techniques which were implemented to improve long-term monitoring programs. Objectives and subsequent findings are as follows:

Objective 1: *Monitor the trout fishery in the Lee's Ferry reach to determine status and trends in abundance (population size and CPE), population structure (size composition and proportional stock density, PSD), growth rate and relative condition* (K_n).

Data collected during 2001 indicate the Lee's Ferry fishery is in a state of decline. High relative abundance of juvenile fish (< 305 mm) since 1997 coincide with declines in fish growth rates and proportional stock density [PSD; (# fish ≥ 406 mm TL/# fish ≥ 305 mm TL)*100]. Relative condition reached low levels unobserved since 1991. Over-recruitment, resource limitation (food, space) and density dependent growth have become problematic aspects of the fishery, particularly in relation to declining dam discharge since 1997. Flexibility in management objectives is needed to allow for maintenance of fish densities that will permit objectives for relative condition, growth and PSD to be attained over a range of flow regimes. Modifying flow regimes to limit rainbow trout recruitment should be evaluated as a means of making progress toward size structure, growth and condition objectives.

Objective 2: *Determine the relationship between Coffelt and Achilles electrofishing (EF) boats, measuring CPE,* K_n*, PSD and catch size distribution so that linkages between the historic data (Coffelt) and data collected with the new gear type (Achilles) can be established.*

Despite conflicting results among three paired gear comparison trials, we can infer that catchability of the Achilles electrofishing boat is higher than that of the Coffelt boat. Regression analysis indicates that CPE values obtained with the Coffelt boat are 72-83% of those collected by the Achilles, although there is much unexplained variation in this relationship. Uncertainties surrounding catchability of the two boats remain, particularly in relation to performance under various environmental conditions (flow regimen, weather, moon phase, food base elevation, etc.) and new EF output regulators. Alternative sampling strategies may need to be implemented to ensure continuity of the old (Coffelt-based) data series. For example, a single trip could be conducted annually with the Coffelt boat at the nine fixed sites to allow continuity of the old (pre-2002) data series. The remaining two or three trips per year could consist of sampling (N = 9 fixed and N = 27 random sample sites per trip) sampling trips conducted with the Achilles electrofishing boats.

Objective 3: *Divide existing sample sites into smaller units that are representative of the larger sites for permanent inclusion in an augmented, serially-alternating sampling design. Evaluate sample power gained by increasing sample size and predictive capability of habitat classifications.*

Most sub-sites were representative of their parent sample sites in that sub-site mean CPE values were of similar magnitude as grand means and 95% confidence intervals always overlapped substantially with those of grand means. We recommend moving forward during 2002 with selection of 27 random transects to augment the nine fixed sub-sites.

Sample power increased two to three-fold with increasing sample size from nine to 33 transects per trip during 2001. The present design appears capable of detecting moderate changes in CPE over the course of a single year.

Hydrologic unit, substrate type, shoreline type, and surface velocity were often correlated. Shoreline type (talus, cobble bars, cliff faces, sand bars) is the most useful variable, because it can be used to stratify catch rates for random site selection and is a surrogate variable for substrate type and hydrologic units. Extensive geographic coverage of shoreline type is also available in electronic format (GCMRC GIS database).

Objective 4: *Develop and refine a repeatable, efficient and precise snorkel survey protocol for quantifying fish observations along the longitudinal axis of the shoreline for use in monitoring of trout relative density. Develop a method to measure fish distribution along the cross-sectional axis of the river.*

Compared with previous surveys (1998-2000), protocols developed during 2001 provide a balance between repeatability, precision, efficiency, standardization of area surveyed, and agreement with electrofishing. However, the utility of snorkeling as a monitoring tool for fish abundance depends on its ability to reflect changes in the fish population over time with the same level of confidence demonstrated by electrofishing and AGFD creel surveys. It is apparent that we should expect some seasonal discrepancy between electrofishing and snorkeling trends. We recommend that snorkel surveys (longitudinal surveys of EF transects, cross-section surveys, and visibility tests) be continued during 2002, with emphasis placed on variables that affect catchability of snorkeling (diel and seasonal variance in visibility, turbidity, light availability, fish behavior).

Electrofishing CPE data from 2001 shows signs of saturation at or above 4.2 fish/min. AT these densities more fish can be counted visually than can be captured by netting. Mechanisms that cause this saturation appear to vary by habitat type. On cobble bars, fish are stunned effectively but there are often too many to bring into the boat during a sample; on sand bars, fish are not stunned effectively, but can be counted easily by snorkeling.

While snorkel/EF data comparisons during 2001 were not entirely conclusive, it is evident that catchabilities of both techniques vary among habitat type. For electrofishing, literature suggests that large particle substrate types generally favor electrofishing, whereas visibility tests in the present study suggests that fish are less visible in such areas, particularly over talus. Both hypotheses suggest that apparent trends in fish abundance among habitat types may be in part be caused by variable catchability coefficients. The extent to which these biases affect population estimates is unknown, and quantifying them would require additional electrofishing depletion experiments and more visibility tests. Surveys of fish distribution across the river channel take little time to complete and are essential for accurate population estimates.

Objective 5: *Estimate size of rainbow trout population in the Lee's Ferry tailwater using electrofishing CPE and snorkel count data; compare magnitude and variability of the two estimates.*

Using habitat-stratified population models, there are an estimated 91,000 RBT > 152 mm (95% CI = 55,000 – 149,000) in the Lees Ferry fishery based on near-shore estimates from electrofishing CPE. Use of abundance estimates from snorkel surveys in a similar model returns an estimate of 78,000 fish (95% CI = 54,000 – 114,000). Confidence intervals of the latter (snorkel-based) estimate are narrower than those from electrofishing, perhaps reflecting the ability to control sample error during individual transect snorkel surveys. Fish density modeled in relation to river mileage returns an estimate of 77,000 fish (95% CI = 44,000 – 110,000) based on electrofishing data, and 60,000 fish (95% CI = 31,000 – 93,000) based on snorkel data.

Similarity of population estimates based on EF and snorkeling is likely coincidental, and further emphasizes the need to understand variance components (of a seasonal nature, in this case) of snorkel surveys. Had the population estimates been calculated from the October data, for example, estimates from electrofishing would have undoubtedly been greater than estimates from snorkeling because the latter data series suggested a 70% decline in trout abundance, whereas EF data showed no change.

The accuracy of tailwater-scale population estimates depends heavily on accurate estimates of what proportion of fish along the cross-channel axis of the river is represented by densities in areas surveyed (either by snorkel or EF). Direct extrapolation—which makes the assumption that fish densities are uniform across the river channel—results in three to eight-fold overestimate of fish per river mile. The 1991 population estimate (which lacked cross-sectional fish density information) was used to formulate management recommendations, which state that a population consisting of 100,000 age 2+ rainbow trout should be maintained in the tailwater. Evidence from the present study indicates that this objective should be approximating 20,000 fish. The population of RBT in the tailwater today exceeds this objective by a factor of 3 to 4.5. Declining growth, size structure and condition indices coincide with increased trout abundance over the past decade and supports the conclusion that the system is overpopulated.

Existing objectives for growth, PSD and condition need to be reassessed for their attainability in relation to a given target population size. To ensure flexibility in management of the Lee's Ferry fishery, we should establish objectives for population size that are specific to a range of discharge regimes and favorable for attaining population size structure (i.e., PSD) and relative condition (K_n) objectives.

INTRODUCTION

Long-term Monitoring

Background: Impacts of regulated flow on the Lee's Ferry tailwater rainbow trout fishery have been a source of interest and concern for both managers and the public for several decades (Persons et al. 1985; Maddux et al. 1987; Reger et al. 1995, McKinney and Persons 1999, McKinney et al. 1999 a, d; McKinney et al. 2001 a; McKinney and Speas 2001). Rainbow trout were initially stocked in the Colorado River below Glen Canyon Dam (GCD) in 1964, and since that time, management efforts, dam operations and flow regimes have interacted to influence the trout community (Arizona Game and Fish Department [AGFD] 1996; Persons et al. 1985; Marzolf 1991; Reger et al. 1995; McKinney and Persons 1999; McKinney et al. 1999 a, c, d). The fishery continues to be a premier nationally and internationally recognized fishery.

In this report, we present results from 2001 monitoring activities in the Lee's Ferry tailwater and describe modifications of existing monitoring techniques implemented to improve the long-term monitoring program. Monitoring of the trout fishery by AGFD using standardized electrofishing (EF) procedures (Sharber et al. 1994) at fixed sampling locations was initiated in 1991 (McKinney and Persons 1999; McKinney et al. 1999 a, c, d; McKinney et al. 2001 a). Synthesis of existing information and a protocol evaluation conducted during 2000 (Lee's Ferry Protocol Evaluation Panel 2000; http://www.gcmrc.gov/pep/troutPEP.htm) suggested that the limited ability to detect short-term changes in RBT population variables is because of small sample sizes and reliance on fixed sampling sites.

This report summarizes results from field studies conducted during 2001 in the Lee's Ferry tailwater. Many of these studies were designed to address sample size and design concerns and to move toward the sample designs outlined in the GCMRC Sampling Protocol for Lee's Ferry Trout Fishery (http://www.gcmrc.gov/aq-prot.doc). This report primarily contains data collected during 2001, with occasional data from March 2002 presented for clarification.

OBJECTIVES

The following paragraphs outline specific objectives that are part of the overall objective of monitoring the trout fishery in the Lee's Ferry reach to determine status and trends in abundance (population size and CPE), population structure (size composition and PSD), growth rate and relative condition (K_n).

Paired Gear Comparison

Objectives: Establish linkages between the historic data and data collected with a new gear type by determining the relationship between two types of shocking boats (19' aluminum johnboat "Coffelt", and inflatable Achilles "sport boats"). Establish linkages between gear types so that electrofishing data from Lees Ferry can be compared with downstream data.

Hypotheses to be tested for gear calibration are:

Ho_1: Catch rate (fish per minute) by size class with the Coffelt EF boat is not different from that of the Achilles EF boat.

Ho_2: PSD and size structure of catches are not different for fish collected with the Coffelt and Achilles EF boats.

Ho_3: K_n is not different for fish collected with the Coffelt and Achilles EF boats.

Sub-site Habitat Classification, Intra-site Variation and Sample Power

Objectives: Divide nine established EF transects into smaller sub-sites and select those that are representative of the larger transect for permanent inclusion in a serially alternating sampling design. Determine intra-site variation (ISV) within each of the nine fixed sampling sites. Evaluate utility of habitat classification variables for predicting fish relative density in the Lee's Ferry tailwater. Evaluate post-hoc statistical power to detect short-term (intra-annual) trends gained by increasing sample size.

Snorkel Surveys

Objectives: Develop a protocol for using snorkeling in the Lee's Ferry tailwater to quantify rainbow trout abundance. Each of three trips focused on addressing specific questions related to snorkeling which included: Determine maximum underwater diver visibility over various substrates and ability to see fish and estimate their size. Conduct longitudinal snorkel surveys on existing EF transects and compare results to those from EF surveys. Develop and refine a repeatable, efficient protocol for quantifying fish abundance in near-shore areas. Develop a method to quantify fish distribution along the cross-sectional axis of the river. Conduct pilot cross-sectional SCUBA surveys in deep areas of the river to count fish.

Population Estimates

Objectives: Estimate size of rainbow trout population in the Lee's Ferry tailwater using electrofishing CPE data and snorkel survey count data and compare magnitude and variability of the two estimates.

METHODS

Long-term Monitoring

General Electrofishing Methods and Fish Handling

We used standard monitoring procedures for electrofishing (McKinney et al. 1999a) to assess changes in trout CPE, K_n, and PSD in the tailwater during 2001. Nine historically fixed electrofishing transects sampled since 1991 were divided into 33 new smaller transects each requiring about 500 seconds of electrofishing.

We measured maximum total length (mm) for all fish collected, and weights (g) for fish > 99 mm. We sexed fish based on manual extrusion of gametes. At all sampling locations, we scanned wild-spawned fishes >152 mm with Destron 400 KHz scanners for presence PIT tags, recorded recaptures, and implanted untagged fishes with PIT tags. We injected PIT tags ventrally into the fish body cavity from immediately posterior to the pelvic fin. We clipped adipose fins of all fish receiving PIT tags, as a secondary mark to evaluate tag loss. We scanned all adult and sub-adult trout for presence of a coded-wire tag (CWT) which would indicate a hatchery origin. A sub-sample of fish carrying CWTs were sacrificed for evaluation of growth rates. We anesthetized fish with a 10% solution of clove oil to prevent injury during handling, and allowed fish to recover in fresh water using live cages or net bags prior to release at the location of capture.

Data Analysis

We computed CPE as fish captured per minute of EF, and indexed size structure of the catch by calculating PSD (Anderson and Nuemann 1996; McKinney et al. 1999 a) as the ratio of "trophy" sized fish to the sum of trophy and "stock" sized fish, or

$$(\# \text{ fish} \geq 406 \text{ mm TL}/\# \text{ fish} \geq 305 \text{ mm TL})*100$$

Fish ≥ 406 mm are currently protected from harvest by AGFD regulations, and fish ≥ 305 mm are size at which most fish are sexually mature (McKinney et al. 1999 a). We also computed CPE for the following length categories: ≤ 152 mm (Age 1), 152-304 mm (ages 2-3), 305-405 mm (age 4+; stock size fish, PSD denominator), and 406-558 mm (age 4+; trophy size fish, PSD numerator; McKinney et al. 1999 a). We estimated length at age using procedures outlined by Wang (1998) and recapture data of wild-spawned fish implanted with PIT tags.

We determined relative condition factor (K_n; Anderson and Nuemann 1996) as

$$K_n = W/W'*100$$

where W′ is the standard weight relationship $10^{\wedge(-4.6\ +\ 2.856*\mathrm{Log10(length)})}$ derived by Persons (in McKinney et al. 1999 a, 2001 a) from Lee's Ferry RBT length and weight data.

We assessed trends in growth of wild-spawned and hatchery-reared RBT by plotting their average annual instantaneous growth (Busacker et al. 1990), calculated as

$$\ln(TL_r) - \ln(TL_m) / (t_r - t_m)$$

where TL_r is length at recapture, TL_m is mean length of fish at marking, and $(t_r - t_m)$ is time elapsed (days) between mark and recapture occasions. Growth rate trends among years (1991-1999 for stocked fish, 1998 – 2001 for wild-spawned fish) were evaluated using simple linear regression, and slopes for wild and stocked fish were compared using analysis of covariance (ANCOVA). Growth data for CWT fish from 2000 – 2001 are forthcoming, although rate of return on hatchery-reared fish has been ≤ 1% since 2000.

A sub-sample of 120 fish was sacrificed during October 2000 and March 2002 (N = 60 for each trip) and tested for whirling disease at AGFD Pinetop fish health lab. Tests consisted of a modification of the polymerase chain reaction method (Pat Lopez, personal communication).

Paired Gear Comparison

Field methods

We compared data collected with Coffelt and Achilles electrofishing boats by sampling with both boats on April 4-6, 8-20, October 1-3, 17-20 (2001), and March 14 and 27 of 2002. The March 2002 trip was designed to resolve ambiguous results on CPE obtained during the 2001 comparisons. Only fish counts and lengths were recorded during the 2002 comparison, whereas we complete fish processing protocols were followed on all other trips. Each trip consisted of two rounds of sampling at fixed sites with the second round of sampling following the first by two weeks (N = 33, 2001 samples) (N = 24, March 2002). Each boat sampled half of the available transects during the first round of sampling and then switched transects for the second sampling event. To avoid changes in flow regime between sample rounds, we restricted sampling during both rounds to weekdays within a given month. We controlled effects of diel flow variation by standardizing the order in which sites were sampled on a given night.

Data Analysis

Catch rates (CPE), relative condition, and PSD of trout captured by the two boat types were analyzed using ANOVA on log-transformed data using boat type (factor A, Achilles or Coffelt) as a fixed factor and sample periods (factor B, April 2001, October, 2001, March 2002) as random variables. If interaction term A x B was significant, metrics were then analyzed within sample periods by comparison of confidence intervals. We performed linear regression on each type of boat to estimate the magnitude of difference in size-specific CPE values for each boat. Preliminary analyses suggested the relationship of catch rates between the two boats varied by fish size, so we conducted analyses of CPE for fish ≤ 152 mm TL (young-of-year/age 1) and > 152 mm TL (age 1+). Frequencies of fish in the following size classes were also compared between boats by using a chi-square test for association: ≤ 152 mm, 152-304 mm, 305-405 mm, and

406-558 mm. All statistical tests were considered significant if resulting probabilities (P) were less than 0.05.

Sub-site Habitat Classification, Intra-site Variation and Sample Power

We subdivided existing EF transects into three or four sub-sites based on hydrologic units found within each transect. Hydrologic units (Pools/eddies, riffles, runs) were characterized based on terminology found in Orth (1983). Hydrologic unit classifications coincided closely with shoreline type as designated by the GCMRC GIS database for the Lee's Ferry tailwater. Principle shoreline types found in that database are talus slopes (TA), cliffs (CL), cobble bars (CB), sand bars (SB) and debris fans (DF). We categorized predominant substrate types for each sub-site according to a modified Wentworth (1922) scale. Sub-site habitat attributes are summarized in Table 1.

Sub-site boundaries were marked with reflective tape affixed to short lengths of PVC tubing. These markers can easily be seen at night with artificial light, but are cryptic in natural daylight. The position of the sub-site start and end points were set using a Garmin III GPS data recorder and recorded on orthomaps.

To assess variance of fish abundance with habitat variables, we conducted individual parametric analyses of variance on electrofishing CPE data (Achilles boat) and snorkel counts using habitat predictors as fixed factors and sample occasion as random factors. Relationships among the predictors were analyzed with correspondence analyses among all variables (SPSS, Inc. 2000).

We selected sub-sites for long-term monitoring (as fixed sites) by averaging catch rates for each over sampling trips and plotting their means and 95% confidence intervals. We then selected sub-sites that were most similar to the grand mean for each parent transect. We evaluated the increase in statistical power due to division of fixed sites ($N = 9$) into sub-sites ($N = 33$) by simulating power to detect intra-annual (i.e., trip by trip) change in CPE in relation to effect size (Cohen 1988) for the two sample sizes. We evaluated the statistical power of the two sample sizes ($N = 9$ and $N = 33$ sites) over effect sizes ranging from small (ca. 0.10) to large (ca. > 0.80; Cohen 1988) on a three year time scale. We then compared the results of the simulation with actual results obtained during 2001. We assumed desirable power to be $(1 - \alpha) = 0.80$ (Cohen 1988).

Snorkel Surveys

We conducted snorkel surveys in the Lee's Ferry tailwater on three occasions during 2001: May 14-16, June 19-21, and October 15-16. In general methods were unique to each trip and were designed to evaluate snorkeling as a sampling method.

We evaluated visibility and accuracy of size estimates in relation to three sets of independent variables: fish size, diver distance from fish (measured as percent maximum visibility), and substrate type. We conducted tests on divers by placing steel, cryptically colored fish silhouettes of known lengths (80, 150, 250, and 350 mm) into 5-10 m swim transects. Divers familiarized themselves with the underwater appearance and size of the fish targets for a few minutes prior to the tests, but had no knowledge of which target would be present on a particular trial. Starting roughly 10-20 m above the targets, divers drifted downstream much as they would during an actual fish count, their eyes toward the shore. We then asked divers whether or not they saw the target, and in which size range the target fell (0-100 mm, 100-200 mm, 200-300 mm, and 300-400 mm). We randomly varied target sizes among passes.

The second series of tests (5/15/01) was designed to evaluate effects of varied substrate (background) types on enumeration of targets. Diver distance was fixed at roughly 50-67% of maximum visibility. We randomly planted three targets per pass. We conducted these tests against cobble and talus substrates at RM –9.3 R and RM –10.2 R, respectively. We evaluated frequency of detection of targets and correct size determination among fish target sizes and substrate type with a chi-square test for association.

We experimented with methods to improve longitudinal snorkeling surveys including a PVC spacer (Schill and Griffith, 1984; Hillman et al. 1992; Thurow 1994) and swim bouys. A length of PVC tubing was used to keep divers equally spaced during a swim, A 6 m section of PVC tubing was used as a spacing device, whereby one diver held onto it at the midpoint (near shore or "inside" position) and the other at the end (offshore, "outside" position). As the divers swam, the free end of the tube was always within 1 m of the shore, and the tube itself was roughly perpendicular to the shoreline. Both divers kept their eyes toward the shoreline without scanning, and only counted fish between themselves and the shore or the next diver. Eight electrofishing sub-sites were snorkeled using this technique during 5/14 and 5/15. Characteristics of longitudinal transects sampled are listed in Table 1.

All fish counts from snorkel surveys conducted during 2001-2002 were transformed into density estimates by dividing numbers of fish counted by the product of average maximum visibility (m) and length of the electrofishing transect (m). We compared results from snorkeling to results from electrofishing surveys by regressing ln-transformed estimates of fish >152 mm per m^2 in each electrofishing transect to numbers of fish counted visually in the transect by snorkeling.

A team of three divers, a boat driver and a deckhand also conducted snorkel by swimming along a series of dive buoys anchored parallel to the shoreline in EF transects. The dive buoys enhanced repeatability and quantification of area surveyed. We placed buoys such that most fish residing near the shoreline would be visible to the divers and allowed maximum underwater visibility to define the area swept.

We controlled sampling error by swimming each transect three times. We minimized observer bias by having each diver swim each transect only once. Divers surveyed transects serially by looking toward the shoreline and counting all fish within a fixed field of vision. After several trial swims, we decided that fish ca. 152 mm or larger were mostly likely to be seen with reasonable repeatability. Maximum underwater visibility was defined as the maximum distance at which the smallest fish can be seen, and was measured periodically using a metal silhouette of a 152 mm fish. Using this method we surveyed 29 of 33 existing EF transects on June 19-21, 2001.

Cross-section surveys

To complete cross-section surveys of fish distribution, we used a 2 hp Aquascooter (R.S.W./D.I., Sedona, AZ), which is a lightweight, submersible, gasoline powered personal watercraft capable of ferrying a diver across the river with little effort. Fish were easily counted in this manner from one shoreline to the other. At roughly 10 m intervals, the diver relayed to those on a boat following the diver the number of fish he had seen in the preceding section of the transect. Each time fish were recorded, we also recorded river depth (m). Each cross sectional survey took about 10 minutes to complete and 20 cross sectional surveys were done. We also conducted two underwater cross sectional surveys using SCUBA gear at RM –2.8. Maximum depth in this area was about 14 m at a discharge of approximately 12,000 cfs. Divers recorded fish numbers at approximately 10 m intervals with pencils on plastic sleeves.

We compared percentages of fish occurring near the shoreline among shoreline types by performing a one-way ANOVA on arcsin-transformed data (Sokal and Rohlf 1973). We compared percent occurrence and absolute numbers of RBT with depth by using simple linear regression.

Population Estimates

We estimated rainbow trout population size in the Lee's Ferry tailwater using two different types of data: near-shore fish population estimates derived from CPE data collected during June, 2001, and snorkel survey data collected immediately prior to that time period. Both types of data were only for fish > 152 mm TL. Local fish population estimates (N_t) were estimated from EF data by using catchability coefficients (q) derived from depletion electrofishing experiments conducted during 2000 (Hilborn and Walters 1992), where

$$CPE = q\,(N_t).$$

Catchability coefficient q is some fraction of absolute fish abundance removed per unit of effort. We partially tested the assumption that catchability coefficients (q) for the Lee's Ferry reach were similar to those derived downstream by conducting three exploratory depletion runs near river mile (RM) –6.0 on 6/24/2001. Catchability estimates from these trials averaged 0.47, which fell within the range observed during previous research (mean q = 0.45, 95% CI = 0.40 – 0.50; N = 41), so we used these catchability coefficients to scale CPE values observed at Lee's Ferry during the 6/22-6/24 sample to local estimates of absolute fish abundance.

We used two different methods to expand the two types of near-shore fish density data:

1) Stratification and summation of fish abundance by habitat type (as indexed by river shoreline type with the GCMRC GIS database, Figure 18). Because of a

shortage of observations on cobble bars and cliff face shorelines, we used a resampling routine for these two substrate types (Resampling Stats, Arlington, VA; Blank et al. 2001) to generate 1000 shoreline-stratified combinations. We then sorted the resulting values in ascending order, selected the median value from the list as the population estimate, and the 2.5 % and 97.5 % quartile values to represent upper and lower 95% confidence interval boundaries (Blank et. al 2001).

2) Integration of a polynomial curve modeling fish abundance as a function of river mileage. Confidence intervals were estimated from the variance of the regression line.

RESULTS

Long-term monitoring

Relative abundance (CPE) of RBT varied among size groups. Relative abundance (CPE) of age 1 (< 152 mm) remained unchanged from levels observed in 2000, but CPE of ages 2-3 declined (Figures 2a and 2b). Relative abundance of stock-sized fish (305-405 mm; Figure 2c) fell to levels characteristic of 1995-1996, whereas quality-sized fish (>405 mm; Figure 2d) fell to minimal levels unobserved since monitoring began in 1991. Proportional stock density similarly declined to levels never before observed in the electrofishing time series (Figure 3). Relative condition declined to low levels observed during 1991 (Figure 4).

Instantaneous growth rates of stocked fish have been in decline since 1992 (R^2 = 0.22, N = 692, P < 0.001; Figure 5). Similarly, growth rates of wild-spawned (PIT carrying) fish displayed a significant but weak negative trend since 1998 (R^2 = 0.05, N = 102, P = 0.019; Figure 5). Slopes of regression lines modeling growth rate decline for both types of fish are not statistically different (ANCOVA, df = 2, F = 0.53, P = 0.58).

Rate of recapture of PIT-carrying fish during 2001 reached a maximum of 9.3% by mid-October, at which point 3236 fish had been tagged and fin-clipped (Figure 6). Loss rate of PIT tags (incidence of fin-clipped fish without PIT tags) among fish at large for 13 days during early- (4/4) to mid-April (4/18) 2001 was 11.1%. Observed loss rate among fish at large for 65 days (marked fish from early- to mid-April combined) was 23.5%.

Both sub-samples (October 2000 and March 2002; N = 60 each) of rainbow trout tested negative for the presence of *Myxobolus cerebralis* (Roger Sorensen, Jim Thompson, AGFD, personal communications).

Paired Gear Comparison

Catch rates of small fish using the Coffelt boat were either the same or slightly greater than those obtained using the Achilles boat during the April sample, but were significantly less than the Achilles boat during the October 2001 and March 2002 samples (Figure 7a). Interaction effects were significant ($P \leq 0.010$) for ANOVA of CPE over factors A (boat type) and B (sampling occasion) for small (≤ 152 mm TL) and large (> 152 mm TL) fish, and also for all sizes combined. Catch rates of large fish using the Coffelt boat were not different from those obtained using the Achilles boat during April 2001 and March 2002, but were significantly less during the October 2001 sample (Figure 7b). Regression analyses on all data combined indicate that on average, CPE of the Coffelt EF boat is 72-83% of the Achilles CPE ($0.41 \geq R^2 \geq 0.21$; N = 88; $P < 0.001$; Figures 8a-c). Chi-square statistic for comparison of frequencies of fish sizes between boats was not significant.

Sub-site Habitat Classification, Intra-site Variation and Sample Power

Analyses of variance indicated that electrofishing CPE varied significantly among levels of all habitat variables: shoreline type ($P = 0.001$), hydrologic unit ($P = 0.001$), substrate type ($P = 0.002$), and relative velocity ($P = 0.002$; Figure 9). Interaction effects between sample occasion and habitat variables were never significant. Among shoreline types, talus and cobble bars tended to be associated with higher CPE values than cliff faces and sand bars. Runs and riffles were associated with higher CPE values than pools. Catch per effort tended to be highest among cobble and boulder substrates, and lowest in areas of slowest ranked water velocity (Figure 9). In contrast to data from electrofishing, we noted no significant differences in fish/m^2 data obtained by snorkeling for any habitat variable.

Correspondence analysis revealed that shoreline types and substrate classification levels were closely aligned (P = 0.032, N =33), in that cliff and sand bar shorelines were associated with sand and organic substrates, cobble bars were associated with cobble-sized particles, and talus shorelines contained mostly large boulders. Shoreline type was also associated with hydrologic units (P = 0.021, N = 33), in that cliffs and sandbars were associated with pools, cobble bars were associated with riffles, and talus slopes were associated with runs.

Averaged CPE values of the Achilles boat among sampling occasions afforded comparisons between CPE values of sub-sites and parent transects (Figure 11).
Division of the existing nine fixed electrofishing sites into 33 smaller sites increased statistical power of the electrofishing monitoring design to detect short-term (ca. intra-annual) changes in CPE by a factor of 2.3 to 3.2, depending on magnitude of effect sizes (Figure 11). Despite marginal predicted power (0.60) for N = 97 samples during 2001, CPE among sample periods approached the significance level of 0.05 (actual P = 0.06).

Snorkel surveys

Maximum underwater visibility during the May 2001 snorkeling trip was 4.5 m (N = 6; SD = 0.5; Figure 12a). Table 2 contains results from the first set of tests (May 14) designed to evaluate fish target visibility and size estimation under ideal conditions, i.e. with no three-dimensional structure and a comparatively reflective background substrate. Under these conditions, we found no significant differences in percent detection of fish targets or percentage of correct size estimates among target sizes and distance from targets, although the sample size was relatively low (i.e., 25% of the chi-square cells did not contain enough observations). Detection of fish targets and size estimation at maximum visibility was less than 100% for targets < 350 mm in length, but nearly always 100% at 2/3 maximum visibility. We correctly estimated size of 80 mm fish targets at maximum visibility, but these targets were rarely detected at this distance.

Table 3 contains results from the second set of tests (May 15) designed to evaluate fish target visibility against various substrate types. Together with results from

May 14, frequency of detection (N = 169) and correct size determination of targets (N = 143) varied significantly among substrate types and target sizes ($P \leq 0.046$). Target visibility was poorest among boulder substrates (no more than 75% of any size category was seen), but slightly better among cobble substrates (≥ 87% seen for ≥ 150 mm targets). Size estimation of all fish target sizes was generally less than 100% correct for both bolder and cobble substrates, but entirely correct for sand/gravel substrates. Size estimation was poorest for 250 mm targets on cobble substrates.

Overall, 169 targets were actually detected visually (83%) in 204 trials, and of these 25 errors in size were made; of these errors, 72% were overestimates. All errors were made estimating targets 150 and 250 mm in length.

Longitudinal swims on electrofishing transects

Divers were easily able to maintain their distance from the shore and each other using the PVC tubing. Precision among replicate swims (coefficient of variation, CV) using this configuration was 0.26. Fish density estimates from this survey were not significantly correlated with fish captured by electrofishing during the previous month.

Mean maximum visibility in sunlit areas in June was 6.5 m (N = 6; SD = 0.6 m; Figure 12a). A logarithmic regression equation of the form $y = b_1 * Ln(x) + b_2$ (b_1, b_2 are constants) provided the best fit between fish density estimates from snorkel surveys (x) and CPE (y) data from electrofishing conducted on 6/22-6/24, 2001 (N = 29, $R^2 = 0.40$, $P < 0.001$; Figure 13a). Mean fish density in near-shore areas surveyed was 0.06 fish/m^2 (SD = 0.01). Coefficient of variation among individual counts by divers was 0.26. Maximum visibility (recorded once per day) was 5.7 m on 10/15 and 4.5 on 10/16 (Figure 12a). Like the June survey, electrofishing CPE (obtained during 10/1-10/3 and 10/17-10/19) was significantly predicted by $Ln((fish/m^2)+1)$ obtained by snorkel surveys ($R^2 = 0.16$, N = 31, $P < 0.001$; Figure 13b). Mean fish/m^2 during the October snorkel survey was 0.02 (SD = 0.02), significantly less than that observed during the June sample. Data from electrofishing surveys (Achilles boat) conducted concurrently with the June and October snorkel surveys did not show any significant decline in relative abundance (Figure 13b).

All data collected during June 2001 through March 2002 indicate an overall significant but weak logarithmic relationship between fish density estimates from electrofishing and snorkeling (Figure 13c; R^2 = 0.15, N = 87, P < 0.001). Within shoreline types, a logarithmic regression between fish density estimates from the two techniques were significant for within sand bars (R^2 = 0.19, N = 31, P = 0.014) but a linear regression fit the data better for cobble bars (R^2 = 0.77, N = 6, P = 0.20). Regressions within talus slopes and cliff faces were not significant.

Cross sectional surveys

While the greatest proportion of fish was observed near the shoreline in roughly half of the samples, there were about as many instances where fish distribution was relatively uniform or random across the channel (Figures 14-16). Fish frequently tended to avoid a well-defined thalweg. Percentage of fish occurring nearest the shoreline did not vary significantly by shoreline type or with river depth, although, but absolute numbers of RBT were weakly and negatively correlated with depth (R^2 = 0.04, N = 244, P = 0.0009). It was difficult to obtain accurate counts of fish nearest the shoreline. The combined effect of the swimmer, the aquascooter and the boat in such close proximity to the shoreline tended to scare fish out of the area, or at least out of sight of the diver (personal observation). Results from the two SCUBA cross section swims (Figure 16) revealed that fish were counted regularly across the channel among depths of 6-14 m.

Population estimates

<u>Habitat-stratified models</u>

Estimated population size of RBT > 152 mm based on near-shore estimates from electrofishing CPE was 91,000 fish (95% CI = 55,000 – 149,000; Figure 20a). Estimated population size based on near-shore snorkel counts was 78,000 (95% CI = 54,000 – 114,000; Figure 20b).

<u>Abundance vs. river mile models</u>

We modeled fish abundance estimates in relation to river mile by fitting a 5^{th}-order polynomial regression to both electrofishing (N = 33, R^2 = 0.37, P < 0.001; Figure 19a) and snorkeling data (N = 29, R^2 = 0.37, P < 0.001; Figure 19b). For models based on electrofishing data, integration produced an estimated population size (RBT > 152 mm) of 77,000 fish (95% CI = 44,000 – 110,000). For models based on fish counts from snorkel surveys, integration produced an estimated population size of 60,000 fish (95% CI = 31,000 – 93,000). Comparisons of estimates from the various types of data and computational methods are shown in Figure 21.

DISCUSSION

Long-term monitoring

Data collected during 2001 indicate that the Lee's Ferry fishery is in a state of decline. High recruitment to age 1+ since 1997 and concurrent decreased growth rates have resulted in a fish population comprised largely of small, mostly juvenile fish (<305 mm). Maturation of these fish and subsequent growth beyond 406 mm (inclusive of trophy-sized fish) is insufficient to maintain the desirable PSD values of 10 - 20, which were observed from 1995 through 2000. Continual decline in fish growth rates (and subsequent decline in body condition) over the previous four years supports the conclusion that over-recruitment and density dependant growth rate suppression remain problematic aspects of the fishery (McKinney and Speas 2001; McKinney et al. 1999 a, 2001 a).

McKinney et al. (1999 a, 2001) hypothesized that population dynamics of rainbow trout in the Lee's Ferry tailwater are governed largely by the magnitude of mean annual discharge and diel fluctuations in discharge. Daily net fluctuations have been restricted to a maximum of 8,000 cfs/day for the passed decade and mean discharge nearly doubled during 1991-1997 and then declined by as much from 1997 through the present (Figure 22). Standing stocks and consumption parameters of nearly all size classes of fish were positively correlated with mean daily discharge, and negatively correlated with magnitude of daily discharge fluctuations (McKinney et al. 1999 a). According to data from 1997 – 2000, high numbers of fish were maintained with modest fluctuations in other fishery indices (K_n, PSD) as long as discharge remained relatively high (i.e., above that observed during the first half of the 1990's). However, reductions in discharge, which began during 2000, have effectively lowered availability of resources (food, space) in relation to fish relative abundance (Figure 22).

Metrics of fish "health" (K_n, PSD, growth) have declined to the lowest levels observed since monitoring began in 1991. McKinney and Speas (2001) hypothesized that competition for food (and perhaps spatial) resources within the burgeoning RBT population was already occurring in 1991-1997. Relative abundance of trout during this time was increasing commensurate with higher mean discharge, and the former remained high despite declining discharge since 1997. Negative effects on fishery metrics stemming from competitive interactions have been exacerbated over the passed 2-3 years because of sustained high fish density and declining, flow-related habitat and food-base resources. Relative condition has declined, likely because of the inability of fish to maintain weight as adults when competition for food and space is most severe (McKinney and Speas 2001). Proportional stock density has likewise declined due to reductions in growth rates of stock-sized fish, which limits recruitment to "trophy" sizes.

Current management objectives call for fish to attain mean lengths of 457 mm by age 3, and maintain an overall relative condition score of 90. No updated assessment of compatibility of these objectives with target population size currently exists, but preliminary stock synthesis analyses suggest that they may be unattainable. Management objectives should be evaluated collectively in relation to annual flow volume and diurnal variability. McKinney et al (1999 a), for example, analyzed combinations of PSD, K_n and CPE observed from 1991-1997 and found that PSD and K_n were maximized (about 28 and 84, respectively) at CPE values of about 2.5 fish/min. More extensive analyses of this type in conjunction with stock synthesis modeling exercises are necessary to better understand interrelationships and attainability of management objectives. To progress toward PSD, growth and condition targets, modifications must be made to flow regimes to limit RBT recruitment. Evaluation of the wide range of combinations of mean flows, fish densities, size structure and condition observed since 1991 can aid in development of management objectives and options.

PIT tag loss in the present study substantially exceeded most values reported in the literature. For pacific salmonids, tag manufactures estimate tag loss rates from the body cavity to be less than 0.1 % (Scott McCutcheon, Biomark, Inc., personal communication; Prentice et al. 1990), although higher rates have been reported. Ombredane et al. (1998) reported loss rates of 3.4 % seven months after tagging for brown trout (*Salmo trutta*) and Buzby and Deegan (1999) reported instantaneous loss rates of 14% for artic grayling (*Thymallus arcticus*). While we try to ensure that PIT tags are entering the body cavity, it is very likely that some tags are actually inserted between the musculature and the skin near the pelvic girdle. Tag rejection is much more likely in this region of the fish (McCutcheon, personal communication). Rather than finding new insertion locations, improving tag retention requires more care in tagging techniques. We are currently attempting to address tag retention rates using varied body cavity tagging techniques at AGFD's Bubbling Ponds Hatchery in Cornville, AZ.

Low recapture rates using the new augmented serially alternating sampling design is perhaps a more serious issue than tag loss. Recapture rates among 33 fixed sites in 2001 never exceeded 10%, but recapture rates from recent electrofishing data (June 27-30, 2002) at 27 random sites and nine historically fixed sub-sites was less than 1%. This scarcity of recapture information might continue because of reduced size of fixed sites and reduced numbers of marked fish. Overall, recapture probabilities per sampling occasion are lower now than during the fixed-site sampling regime (1998 – 2001), and current recapture rates may not be sufficient to detect trends in growth.

Paired Gear Comparison

Gear comparisons conducted during April and October of 2001 gave conflicting results and were insufficient to demonstrate differences in catchability between the two types of electrofishing boats. During the first trial (April 2001), the data suggested that catchability of the Coffelt boat was higher than that of the Achilles boat, particularly for small fish. The most likely source of this variation during the April 2001 trial was electrical output. The Coffelt boat maintained an average of 238 V and 17 A on all samples, which are the levels commonly used for monitoring at Lee's Ferry since 1991 (McKinney et al. 1999 a, 2001 a). The Achilles boats, by contrast, maintained an average of 390 V and 14 A, which is more similar to settings used downstream of Lee's Ferry (AGFD, unpublished 2000 EF data from Grand Canyon; Lars Niemi and Pete Weiss, personal communication). Regardless of amperage, Achilles voltages were always greater than those of the Coffelt boat. Voltage gradients, are always greater with Achilles due to higher resistance (distance from anode to cathode). Another factor that probably influenced catch rates was weather. Strong winds and glare from the incandescent lights of the Achilles boat on the water may have made seeing small RBT more difficult than in the Coffelt boat, which is equipped with sodium lamps that illuminate water column more effectively than the Achilles lighting system.

Results from the October 2001 gear comparison strongly contradicted those of the April 2001 trial. In October, catchability of the Achilles boat was greater than that of the Coffelt boat for both small and large fish. We had standardized the power output of the two boats at 15-17 amps, but other factors intervened and likely made the two sampling periods incomparable. Weather was more favorable during October (no wind or precipitation) than in April. Also, in October the Coffelt EF output regulator used by the Achilles boat overloaded and caught fire briefly. No injuries resulted, but we were forced to use the spare regulator during the remainder of the trip. Prior to this incident, we assumed that the EF pulse regulators produced similar outputs, but we began to suspect that differences in output from the three available EF output boxes might explain differences in CPE between trips in April and October. We recommended a third, abbreviated, "tie-breaker" comparison to test the working hypothesis that using the same EF units from October could duplicate differences in CPE between the two boats.

Weather during the March 2002 gear comparison was similar to that in October 2001, and we utilized the same EF output regulators in each boat as we had previously. Results from the March 2002 trial were largely similar to those obtained during October.

Despite factors that are nearly impossible to control for (weather, electrical nuances, boat driver abilities and styles, netting crew skills) in these trials, we believe that catchability of the Achilles EF boat is higher than that of the Coffelt boat. Statistical differences in relative condition of fish between the two boats during October are probably not biologically relevant due to large sample sizes involved (N = 1946 length/weight pairs), or perhaps problems with the electronic scales.

High significance of the regression slope relating catch rates of the two boats suggests that use of the slope as a "conversion factor" in interpreting future CPE data from the Achilles boat may be plausible. However, large amounts of unexplained variation (i.e., $R^2 \leq 0.41$ for all comparisons) in the relationships between the two boats may compromise the accuracy of such conversions. If uncertainties surrounding differential catchability (and resulting time series data) using the two boats persist, particularly in relation to performance under various environmental conditions (weather, moon phase, food base elevation, etc.) or new equipment (EF output regulators), alternative sampling strategies may need to be adopted to that ensure continuity of the old (Coffelt-based) data series. For example, a single trip could be conducted annually where the Coffelt boat visits the nine fixed sites at a time of year that enables linkage with the old (pre 2002) data series. The remaining two or three trips would consist of serially augmented (fixed and random) sampling trips conducted using the Achilles electrofishing boats.

Sub-site Habitat Classification, Intra-site Variation and Sample Power

Results from correspondence analysis indicate that classification levels of habitat variables are correlated. Of all variables being recorded, shoreline type appears to be most useful, because it has predictive capability for trout relative abundance, it is a surrogate variable for substrate type and hydrologic units, and shoreline data is available for the entire tailwater in electronic format (GCMRC GIS database). Stratification of fish abundance data by shoreline type may also prove useful in other areas of the river corridor.

The lack of correspondence between habitat variables and fish density data from snorkel surveys is likely due to lack of catchability coefficients in calculations of fish density. Experimentation with visibility of fish targets among substrate types (see "Snorkel Surveys") demonstrated that fish targets were sighted at lower frequencies over cobble and talus substrates. Snorkel counts near such shoreline types likely underestimated fish density because no catchability coefficient was integrated into the density estimate. By contrast, the higher frequency of target sightings over sand/gravel would function to stabilize or lower estimates made adjacent to sand bars. As a result, fish densities appeared relatively uniform across shoreline types. Alternatively, electrofishing catchability among talus slopes and cobble bars could be higher than other habitats owing to the low conductivity of larger substrate particles in these areas (Reynolds 1983). Resolving the issue of whether certain habitats harbor more fish or are merely sampled more efficiently (i.e., habitat-specific catchability coefficients) will require additional analysis, although amount of additional data collection is uncertain.

Most sub-sites were easily selected and representative of parent transects in terms of their mean CPE and 95% confidence intervals. We recommend moving forward during 2002 with selection of random transects to add to the nine fixed sub-sites listed in the results section. Existing fixed electrofishing sub-sites are comprised of habitat types that are marginally representative of existing habitat. Monitoring data from past surveys over-represented sand bar habitats, and under-represented cobble bar and cliff face habitats. The impact of this misrepresentation on interpreting existing CPE data is probably slight, because CPE data from the three misrepresented habitat values tend to be statistically insignificant.

Sample power increased two to three-fold due to increasing sample size from nine to 33 transects per trip. As a result, the present design is capable of detecting moderate changes in CPE over the course of one year. Increased power is desirable because of continued experimentation with discharge as a resource management tool.

Snorkel surveys

Background:

Snorkel surveys are less intrusive than electrofishing (Nielsen 1998) and snorkel data can be used for evaluating electrofishing bias and catchability. High water transparency in the Lee's Ferry tailwater may also facilitate snorkeling and quantification of fish distribution across the river channel.

Prior to 2001, snorkel surveys were conducted only once per year. The purpose of these surveys was purely exploratory, and standardization of technique was minimal. Surveys typically consisted of three divers swimming EF transects in parallel lanes (one near the shore, the others further out in the channel) at least two times each. No attempts were made to quantify area swept, and divers frequently drifted into each other's lanes. Having up to three divers in the water at the same time tended to scare fish into another diver's field of vision, resulting in double counting. Correspondence with electrofishing CPE was also weak, probably due to small sample sizes.

To standardize swimming positions and quantification of area surveyed, in May 2001 we had divers hold onto a 20' section of PVC pipe, ensuring equal spacing throughout the survey. Swimming with the PVC pipe proved difficult and using the pipe to maintain a fixed distance from the shore only resulted in swimming too close to the varial zone, where few fish were counted. The results of the PVC swims did not correspond well with EF CPE, and we have since abandoned the technique.

Compared with previous surveys, the single-lane survey (current design developed during June 2001) has the best balance between repeatability, precision, efficiency, standardization of area surveyed, and agreement with EF CPE. A crew of five people can complete all 33 fixed EF transects in 2-3 days, which also includes cross-sectional swims. Divers need not be taken back to the top of the transect following a swim, which saves considerable time. Precision of the single lane method is easily quantifiable and comparable to previous surveys. Agreement with EF CPE is relatively consistent. Finally, using buoys and GPS technology should ensure quantification of area surveyed much more accurately than previous surveys.

Utility of snorkel surveys for monitoring fish density in the Lee's Ferry tailwater depends on its ability to reflect changes in the fish population over time with the same level of confidence and accuracy of electrofishing and/or AGFD creel surveys. Figure 13 suggests that we may expect seasonal discrepancies between electrofishing and snorkeling trends in fish abundance indices, but behavior of such trends over a period of a few years remains to be seen.

We recommend that snorkel surveys (longitudinal surveys of EF transects, cross-section surveys, and visibility test) be continued during 2002, with special emphasis placed on variables that potentially affect accuracy of the surveys (diel and seasonal variance in visibility, turbidity, light availability, fish behavior). Visibility tests have suggested that catchability coefficients (or $q_{snorkel}$, as indexed by percent detection of fish targets during the May 2001 sample) vary with river substrate type and fish size. For fish targets 150-350 mm, catchability declined as substrate particle size increases, reflecting the difficulty of sighting fish in complex, cryptically colored habitats. Site and sample period-specific differences in $q_{snorkel}$ form one of the largest sources of error in snorkel-based population estimates (Schill and Griffith 1984; Josh Korman, Ecometrics, Inc., personal communication). If a higher level of accuracy in snorkel data is desired, we need to further evaluate $q_{snorkel}$ and area swept at maximum visibility under varied sampling conditions (time of day and/or, ambient lighting, substrate types). Data on temporal and spatial variability of underwater visibility in the tailwater is limited, but there is strong evidence that it may vary seasonally, spatially and throughout the course of a day (Figures 12a,b). The impact of this variance on catchability is unknown but likely important in explaining discrepancies between electrofishing and snorkel counts such as that observed between June and October 2001 (Figure 13), when maximum underwater visibility apparently declined by about 25%. We recommend future surveys consist of at least three independent visibility tests per sample.

Seasonal and diel changes in fish behavior may also account for discrepancies between electrofishing and snorkeling data. Thurow (1994) recommended that snorkel surveys be conducted during seasons when fish movement is minimal and ranges are relatively static. In regulated streams, changing food base in relation to dam discharge likely dictates the daytime positioning of trout. Electrofishing samples are always collected at night, so behavior and positioning of RBT is likely governed primarily by factors such as moon phase (Robinson and Barraclough 1978). Seasonal or diel sources of variation associated with either sampling technique could cause bias in the respective time series (Figure 13).

Errors in estimation of fish size while snorkeling were generally overestimates, which reflect the characteristic 25% magnification effect of the water column (Professional Association of Diving Instructors 1999). Errors in sizing intermediate-sized fish suggest that reliable size estimation by 100 mm intervals may not be possible with snorkeling. Snorkeling will not produce size-composition data as accurate as those from electrofishing.

Complexity of the streambed cover (boulders, vegetation, some woody debris) precludes accurate counts of young-of-year fish using the protocol we developed Gardiner (1984) also noted that downstream swim surveys were generally ineffective for detecting small, cryptic fishes. Collection of observations on age-0 fish may be more effectively made through electrofishing.

Snorkel surveys of fish distribution across the river channel take little time to complete and are essential for accurate population estimates (Schill and Griffith 1984; Slaney and Martin 1987). However, some refinement of cross-section survey protocols is necessary, In several instances fish residing adjacent to the shoreline were frightened away before the divers could count them. We recommend the dive boat not follow the diver completely into shore during these surveys.

The relationship of fish density estimates from snorkel surveys with EF surveys appears to approach an asymptotic phase at high fish densities, which indicates that above a certain density more fish can be counted visually but not captured by electrofishing. Electrofishing CPE shows signs of saturation at or above about 0.05 fish/m^2, which is about 4.2 fish/min. EF. Inspection of electrofishing catch rates for fish > 152 mm for the Achilles boat to date partially supports this finding, in that 14% of all EF samples collected to date exceed CPE values of 4.2 fish/min. EF. The saturation effect is probably even more pronounced for the Coffelt boat, in which case only 4.5% of all EF samples exceed 4.2 fish/min. EF.

The mechanism behind the observed saturation in electrofishing data may vary by substrate type. Fish are effectively stunned and vulnerable to capture on cobble bars, but swift water limits the number that can be caught (AGFD, personal observations). As a result, the EF catch rates will not reflect actual densities, but such high densities of fish can be enumerated through snorkel surveys. “Saturation” in sand bar environments, by contrast, may only be apparent saturation. Relative to other habitats, sand bars usually harbor fewer fish that are often too deep to capture. Fine substrate size of sand bars is also not as conducive to efficient electrofishing as talus or cobbles (Reynolds 1983). Fish in sand bar environments are very visible to divers, but are usually in low enough densities that they are fully vulnerable to netting by EF.

While snorkel/EF data comparisons during 2001 were not conclusive, catchability of both techniques probably vary among habitat type. For electrofishing, literature suggests that large particle substrate types favor electrofishing (Reynolds 1983), whereas visibility tests in the present study suggests that fish are less visible in such areas, particularly over talus. The extent to which these biases affect population estimates is unknown, and quantifying them would require additional electrofishing depletion experiments and more visibility tests.

Population estimates

Prior lack of cross-sectional fish density information led to direct extrapolation of nearshore estimates across the river channel (Morgensen 1991, McKinney et al. 1999 a). This method was used in 1991 to formulate current management recommendations, which state that a population consisting of 100,000 age-2+ rainbow trout should be maintained in the tailwater (Arizona Game and Fish 2001). However, on average, direct extrapolation of data which makes the assumption that fish densities are uniform across the river channel, results in a three- to eight-fold overestimate of fish per river mile.

Estimates of population size from the current study have serious implications for revision of management objectives. Catch rate data from both standardized electrofishing and the AGFD creel census suggest that abundance of rainbow trout in the tailwater is about double what it was in 1991, when supposedly there were 100,000 age 2+ rainbow trout in the system. Adjusting the 1991 estimate downward by a mean factor of five from the present study amounts to a revised target of about 20,000 age 2+ fish. Ignoring this revision, could mislead managers or stakeholders into believing that since estimated population size in 2001 was roughly 90,000 fish, we are close to meeting our objectives for trout abundance, when in fact we have likely exceeded it. Declines in growth rate, relative condition and PSD support the conclusion that densities exceed system capacity. Justification of management actions to reduce trout abundance will be difficult if revisions of population estimate-based objectives are not completed.

Confidence limits for habitat-stratified population estimates based on snorkel survey data were narrower than those based on electrofishing data because they were based on mean counts from three replicate swims per sub-site, whereas electrofishing runs were not replicated within transects.

Electrofishing provides a relatively unbiased representation of the size structure of the trout population. Furthermore, the PEP recommended continuation of the AGFD creel survey at the expense of any other monitoring technique. We feel that snorkel surveys should be evaluated periodically in relation to the amount of confidence we have in snorkel data and the value that data adds to the existing monitoring program.

The similarity of population estimates based on EF and snorkeling is likely coincidental, but not entirely unexpected. In a review of nine studies comparing estimates from snorkeling with those from depletion electrofishing in small- to medium-sized streams, Thurow (1994) found that snorkeling tended to produce estimates that ranged from 75 to 110% of electrofishing estimates. Snorkel data from October suggested a 70% decline in fish density since June, whereas EF data suggested no change of any kind. The discrepancy between the two techniques underscores the importance of understanding factors affecting snorkeling catchability.

Estimation of population size using predictive components from substrate GIS databases did not return estimates significantly different from models based on longitudinal trends in fish abundance. Both techniques utilized the cross-channel expansion factors, suggesting that if this large source of variation is accounted for, expansion longitudinally or by habitat type account for about the same amount of additional variation. Stratification of expansions by habitat type seems a much more reliable expansion method, providing adequate geostatistical coverage is available and habitat variables have sufficient predictive power.

RECOMMENDATIONS

1. Evaluate management options in the Lee's Ferry tailwater fishery, with particular emphasis on flow regimes that may be employed to control RBT recruitment.
2. Develop flexible trout management objectives for fish densities such that objectives for relative condition, growth and PSD can be attained over a range of flow regimes. Objectives should be based on analyses of known combinations of mean flows, fish relative densities (CPE from AGFD creel, electrofishing), size structure (PSD), and relative condition (K_n) observed since 1991.
3. Update and re-parameterize stock synthesis model with more current data for aid in development of management objectives under recommendation (3) above.
4. Revise management objectives to reflect more accurate estimates of population size, and reassess attainability of growth, PSD and K_n objectives in relation to target population size. Such revisions should include recalculation of previous population estimates made during 1991 (Morgensen et al. 1991) and 1998

(McKinney et al. 1999 a) by expanding near-shore estimates from these two studies in the manner detailed herein.

5. Considerable unexplained variation exists in the relationship between efficiency of Coffelt and Achilles-type EF boats. If uncertainties surrounding catchability of the two boats persists, particularly in relation to performance under various environmental conditions, new equipment (Smith – Root EF output regulators), or new sampling designs (serially-augmented fixed/random designs proposed for 2002 an beyond), alternative sampling strategies may be needed to ensure continuity of the old data series. For example, a single trip could be conducted annually where the Coffelt boat visits the nine fixed sites at a time of year that enables linkage with the old (pre 2002) data series. The remaining two or three trips would consist of sampling trips conducted with the Achilles EF boats on fixed and random sites.
6. Further evaluate shoreline type as a means to stratify fish distribution and relative density for use in selecting random electrofishing sites in the Lee's Ferry tailwater, and eventually, areas downstream from Lee's Ferry.
7. Continue longitudinal snorkel surveys of electrofishing transects and comparisons of snorkel counts with electrofishing estimates in order to evaluate the reliability of snorkel-based estimates of trout abundance.
8. Continue and refine technique of cross-channel surveys of fish distribution, especially to improve accuracy of percentages of fish in near-shore areas for more accurate population expansions.
9. Develop a better understanding of physical (light availability, turbidity, background type) and diel/seasonal (fish behavioral) factors that influence catchability of snorkel surveys.
10. Evaluate information needs on catchability of electrofishing, snorkeling, and angling with specific emphasis on variance of catchability with sampling conditions (flows, habitat, timing).

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Table 1. Habitat characteristics of fixed electrofishing transects in the Lee's Ferry tailwater. Hydrologic unit codes: PO = pool, RU = run, RI = riffle (Orth 1983); Shoreline type codes: SB = sand bar, TA = talus, CB = cobble bar, CL = cliff; Substrate codes: 2 = sand, 6 = cobble, 8 = boulder, 11 = organic matter.

Sub-site	Upstream RM boundary	Side	Hydrologic unit	Velocity rank	Shoreline type	Substrate rank	Transect length (m)
3a	-14.9	R	PO	1	SB	2	122
3b	-14.8	R	PO	1	CL	2	173
3c	-14.7	R	RU	2	TA	8	185
3d	-14.6	R	PO	1	TA	8	132
4a	-14.22	L	RU	3	TA	8	206
4b	-14.1	L	RU	1	TA	8	175
4c	-14.02	L	RU	1	TA	8	169
4d	-13.9	L	RU	1	TA	8	209
5a	-12.3	L	PO	1	CB	6	131
5b	-11.98	L	RI	3	CB	6	346
5c	-11.75	L	RI	3	CB	6	232
6a	-10.41	R	PO	1	CL	2	148
6b	-10.35	R	PO	2	TA	8	160
6c	-10.26	R	PO	1	TA	8	161
6d	-10.2	R	PO	1	TA	8	162
9a	-7.22	L	PO	1	SB	2	176
9b	-7.12	L	PO	1	SB	2	170
9c	-7.01	L	PO	1	SB	2	139
9d	-6.92	L	PO	1	SB	11	163
10a	-5.82	R	PO	1	TA	8	149
10b	-5.75	R	PO	1	TA	8	124
10c	-5.69	R	RU	2	CL	8	209
10d	-5.58	R	RU	2	TA	8	127
12a	-4.3	L	RU	1	SB	2	176
12b	-4.2	L	RU	2	SB	2	315
12c	-4.02	L	PO	2	SB	2	325
13a	-3.23	R	RU	1	SB	2	119
13b	-3.15	R	RI	3	SB	6	320
13c	-2.9	R	PO	1	SB	2	178
15a	-1.3	L	PO	1	TA	2	191
15b	-1.2	L	PO	1	SB	2	176
15c	-1.1	L	PO	1	TA	2	148
15d	-1	L	PO	1	SB	2	140

Table 2. Sample size (N), percent detection (% seen) and percent correct size estimation (% corr.) of fish targets at various percentages of maximum visibility.

	Target Sizes											
	80 mm			152 mm			250 mm			350 mm		
% max. visibility	N	% seen	% corr.	N	% seen	% corr.	N	% seen	% corr.	N	% seen	% corr.
33	6	67	100	6	100	100	6	100	83	7	100	100
67	6	67	100	6	100	100	6	100	100	6	100	100
100	7	29	100	13	85	82	11	82	67	7	100	100

Table 3. Sample size (N), percent detection (% seen) and percent correct size estimation (% corr.) of fish targets among various substrate types. Diver distance was 50-67% of maximum visibility.

	Target Sizes											
	80 mm			152 mm			250 mm			350 mm		
Substrate	N	% seen	% corr.	N	% seen	% corr.	N	% seen	% corr.	N	% seen	% corr.
Sand/ Gravel	6	67	100	6	100	100	6	100	83	6	100	100
Cobble	--	--	--	24	88	81	15	87	38	15	100	87
Boulder	3	67	100	36	69	84	12	75	89	12	75	100

Table 4. Comparison of sampling attributes of snorkel surveys, Lee's Ferry tailwater 1998-2001.

Survey type, year	Repeatability	Precision (Coefficient of Variation, SD/mean*100)	Efficiency	Quantification of area surveyed	Agreement with EF
Pilot (1998-2000)	Poor	19% (bias present due to same diver/ swim lane)	Approximately 20-30 min./diver/transect	Poor; no standardized swim lanes	None or weak (N = 11)
PVC (May 2001)	Good	26% (bias present due to same diver/swim lane)	Approximately 20-30 min./diver/transect	Good; fixed distance between divers, shoreline	None (N = 8)
Single Lane (June 2001)	Good	26%; no diver bias	Approximately 10 min./diver/transect	Good; area delimited by visibility, swim lane well defined by buoys.	Acceptable, $R^2 = 0.33 – 0.37$; N = 29

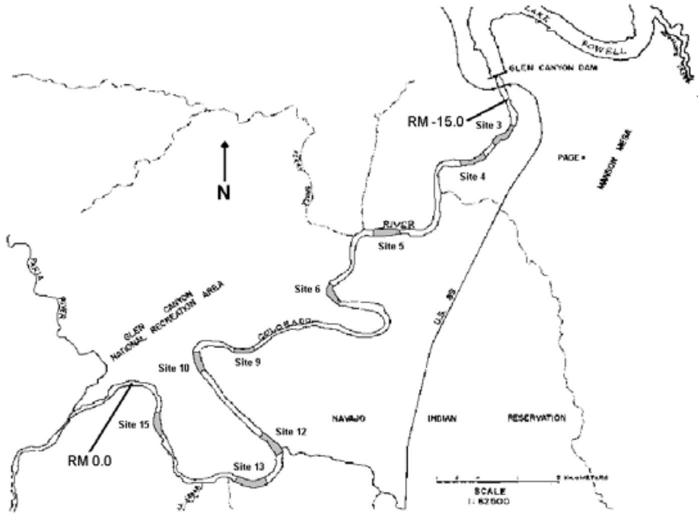

Figure 1. The Lee's Ferry tailwater below Glen Canyon Dam on the Colorado River in Northern Arizona. Fixed electrofishing transects sampled during 1991-2000 (shaded regions) were subdivided into 33 shorter transects during 2001.

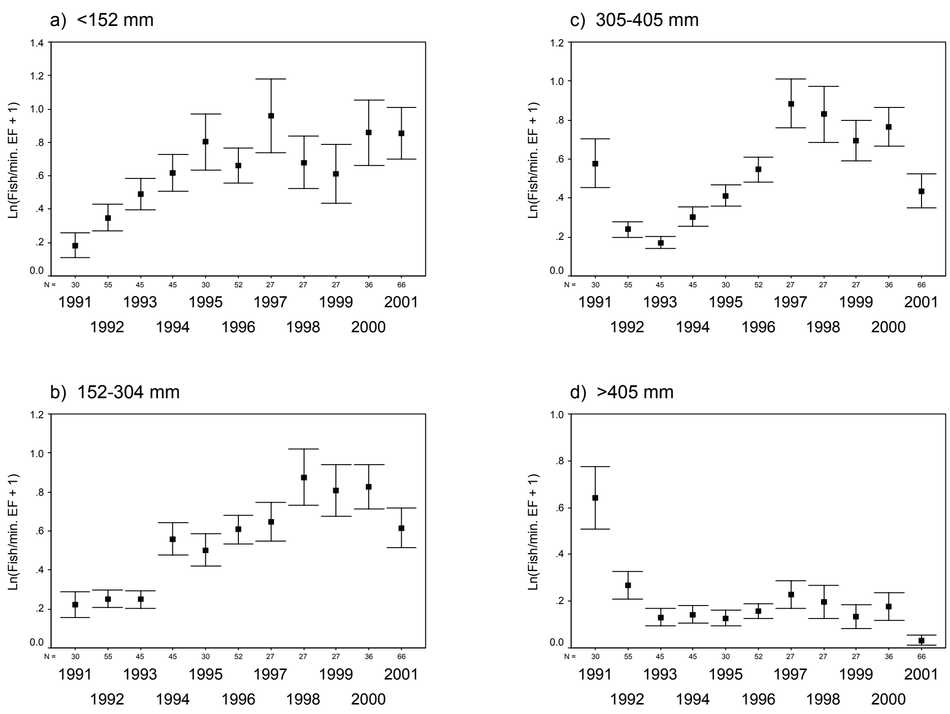

Figure 2. Log-transformed catch-per-effort (Coffelt boat) of RBT observed during 1991-2001 for (a) fish ≤ 152 mm, (b) 152-304 mm, (c) 305-405 mm, and (d) > 405 mm, Lee's Ferry tailwater, Colorado River, AZ. Bars represent 95% confidence intervals of the mean.

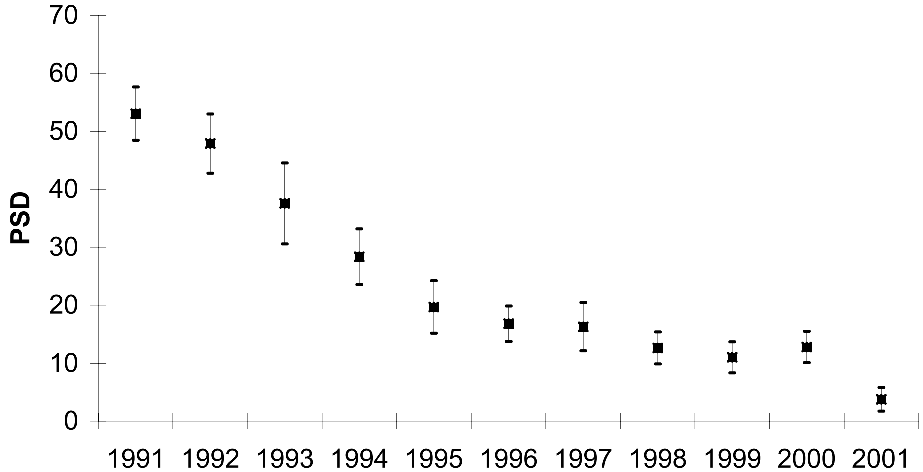

Figure 3. Proportional stock density ((# fish ≥ 406 mm TL/# fish ≥ 305 mm TL)*100) of RBT in the Lee's Ferry tailwater, Colorado River, AZ during 1991-2001. Bars indicate 95% confidence interval of the mean.

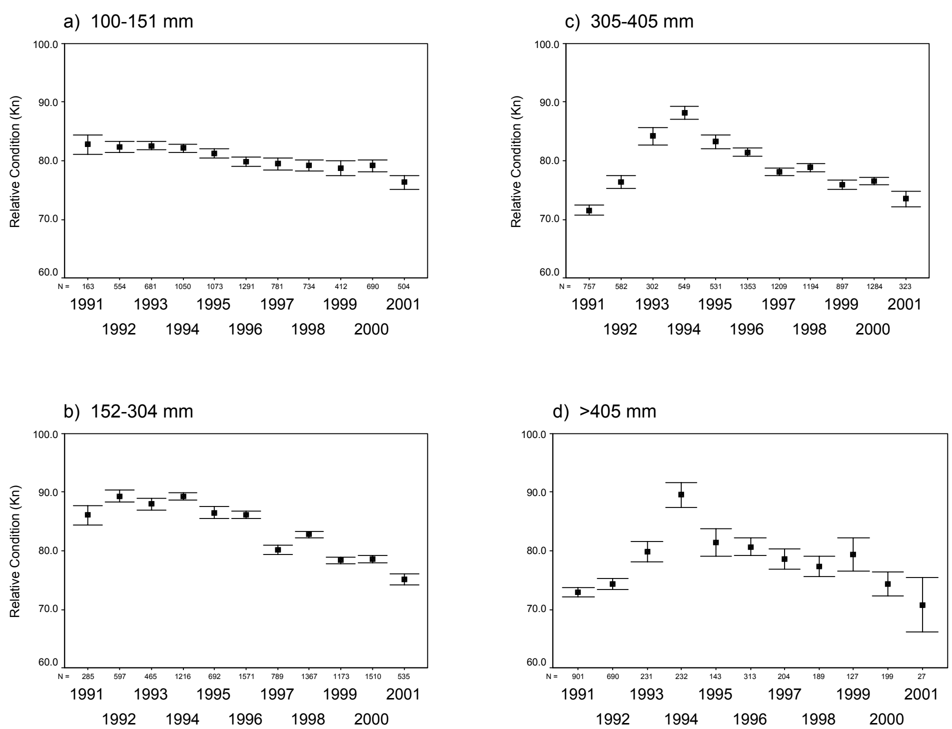

Figure 4. Relative condition (W/W′*100, where W′ = $10^{(-4.6 + 2.856*Log10(length))}$; data collected with Coffelt boat) for (a) fish 100-152 mm, (b) 152-304 mm, (c) 305-405 mm, and (d) > 405 mm, Lee's Ferry tailwater, Colorado River, AZ.. Bars indicate 95% confidence interval of the mean.

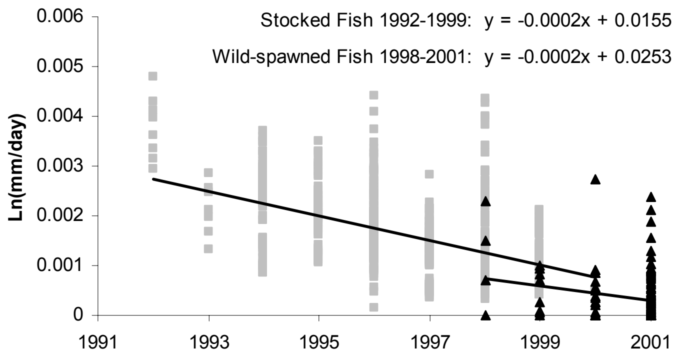

Figure 5. Instantaneous growth rates [Ln(mm)/day] of wild-spawned (triangles) and hatchery-reared (stocked fish; squares) in the Lee's Ferry tailwater, Colorado River, AZ.

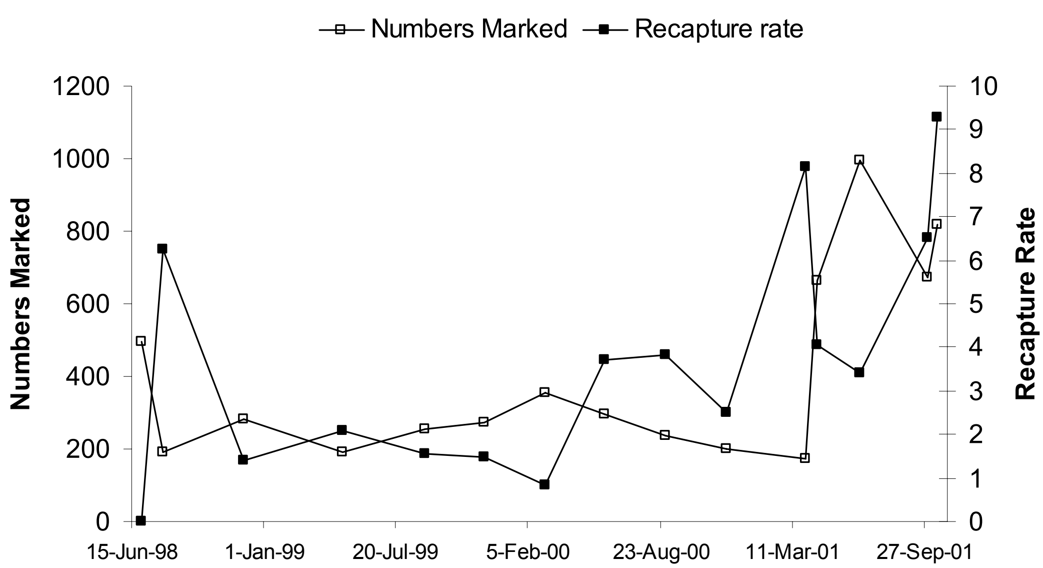

Figure 6. Number of RBT marked per sampling occasion (left axis) and recapture rate (number of marked fish captured/all taggable fish)*100) in the Lee's Ferry tailwater, Colorado River, AZ during 1998 – 2001.

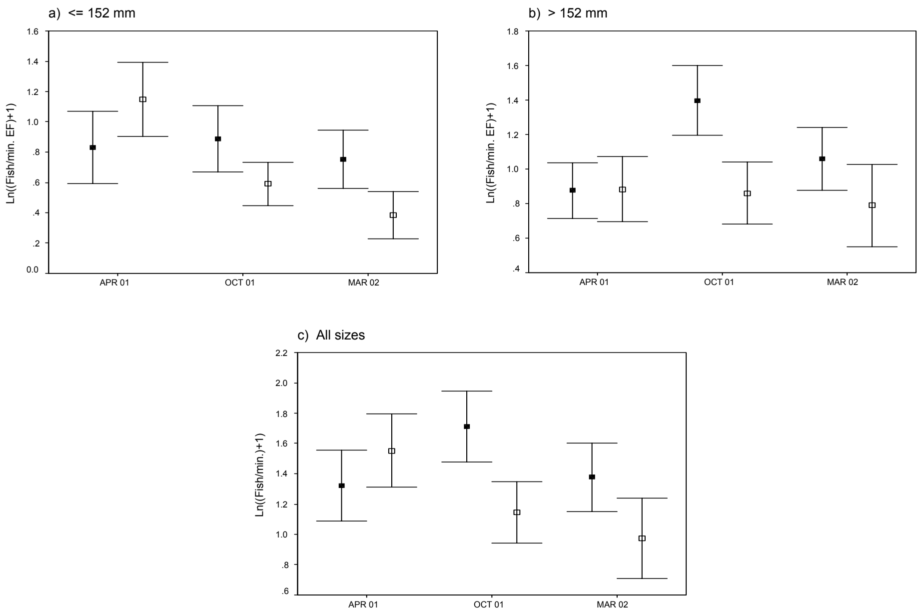

Figure 7. Comparison of CPE data collected with the Coffelt-type electrofishing boat (open symbols) and Achilles-type boats (closed symbols) for (a) fish ≤ 152 mm, (b) > 152 mm, and (c) all fish combined. Data were collected during three separate paired gear comparison trials (April and October 2001, March 2002) in the Lee's Ferry tailwater, Colorado River, AZ. Bars represent 95% confidence intervals of the mean.

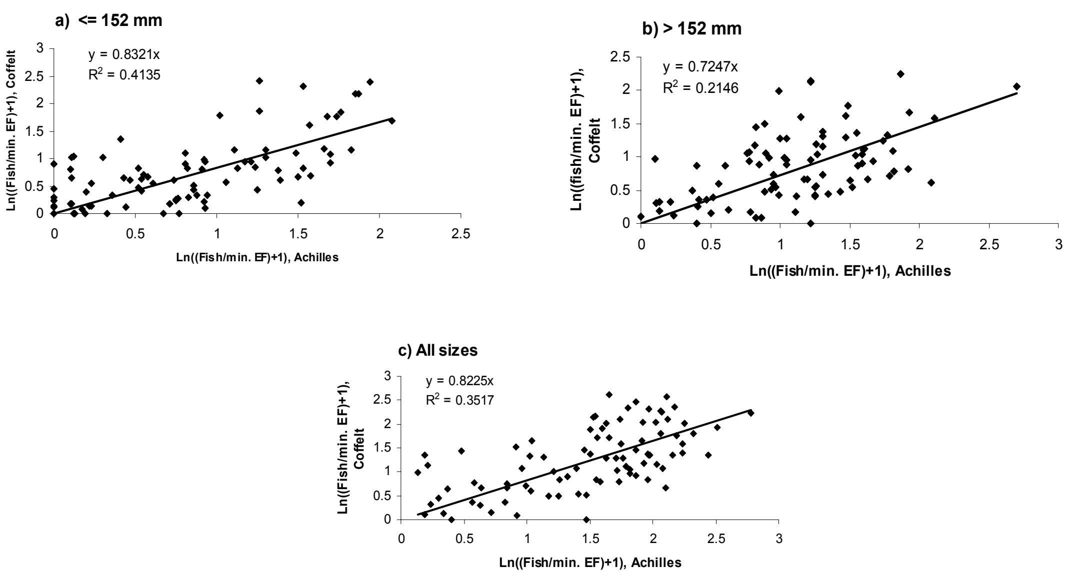

Figure 8. Regressions (forced zero intercept) of CPE data collected by the Coffelt-type electrofishing boat (y axis) and the Achilles-type electrofishing boat (x axis) for (a) fish ≤ 152 mm, (b) > 152 mm, and (c) all fish combined. All regressions are significant ($P < 0.001$)

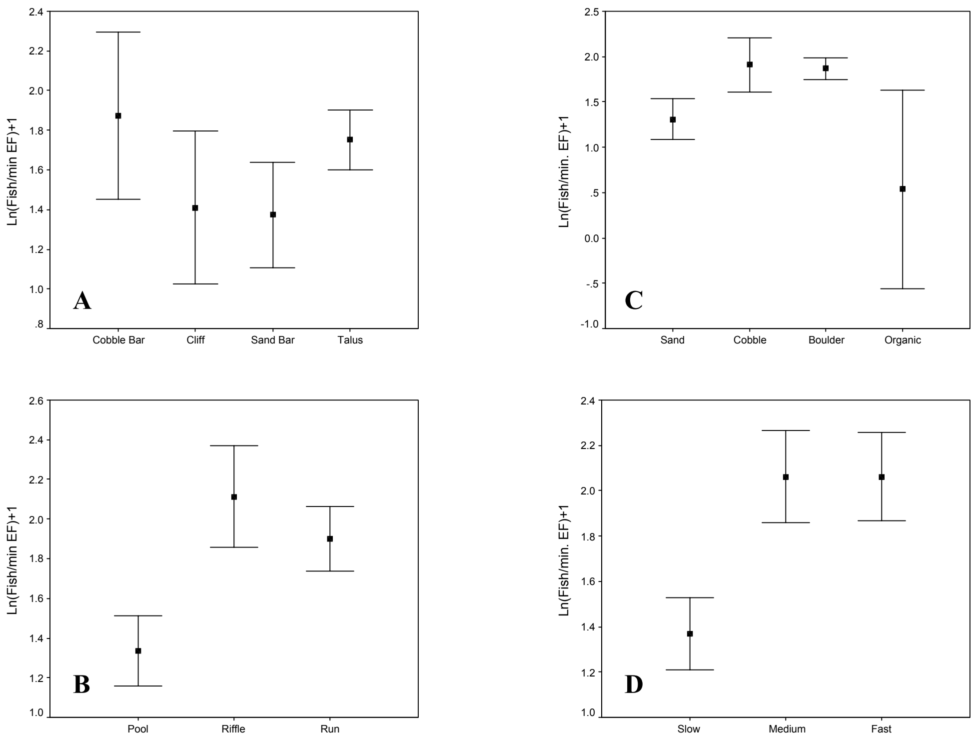

Figure 9. Log-transformed CPE data collected with the Achilles EF boat among (a) shoreline types (b) hydraulic unit, (c) substrate type, and (d) surface velocity in the Lee's Ferry tailwater, Colorado River, AZ during 2001. Bars represent 95% confidence intervals

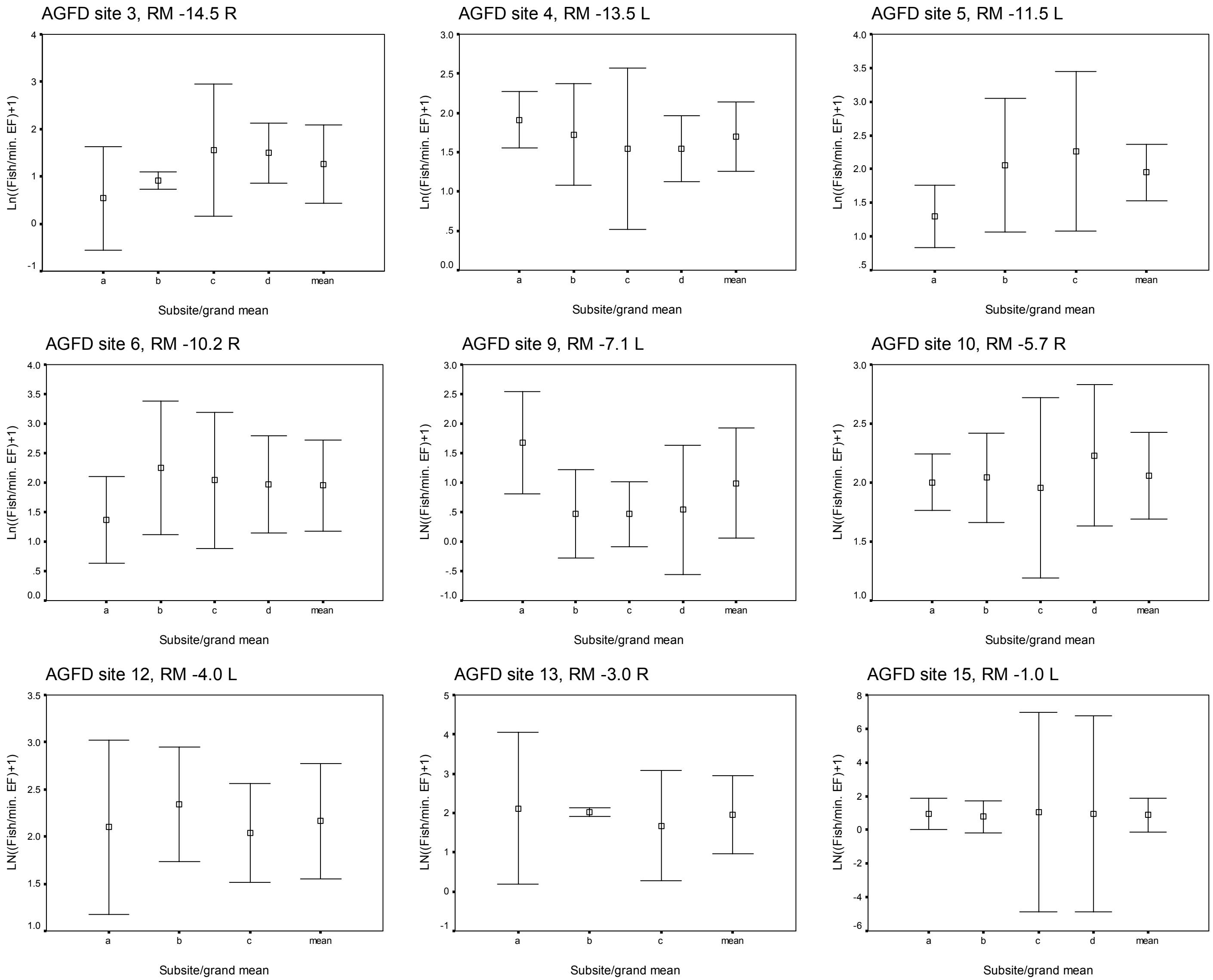

Figure 10. Mean CPE and 95% confidence intervals for electrofishing sub-sites and combined sub-sites averaged over three sampling occasions during 2001 (April, June, October). Data were collected with Achilles electrofishing boats.

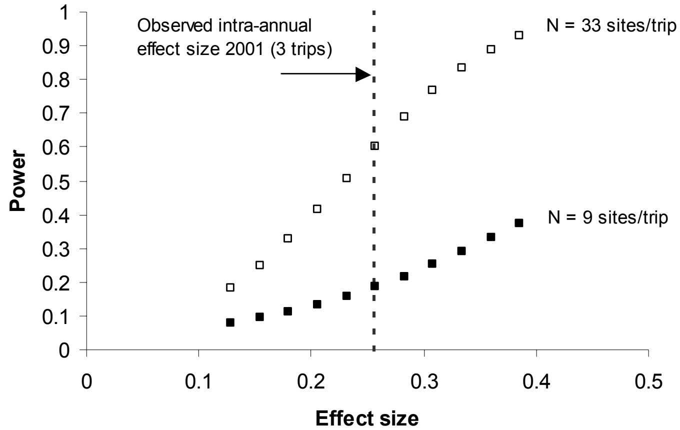

Figure 11. Statistical power to detect short-term (i.e., intra-annual) trends with N = 33 (open symbols) and N = 9 (closed symbols) sub-sites per electrofishing trip over various effect sizes, assuming three trips are conducted per year. Actual effect size observed during 2001 is represented by vertical, dashed line.

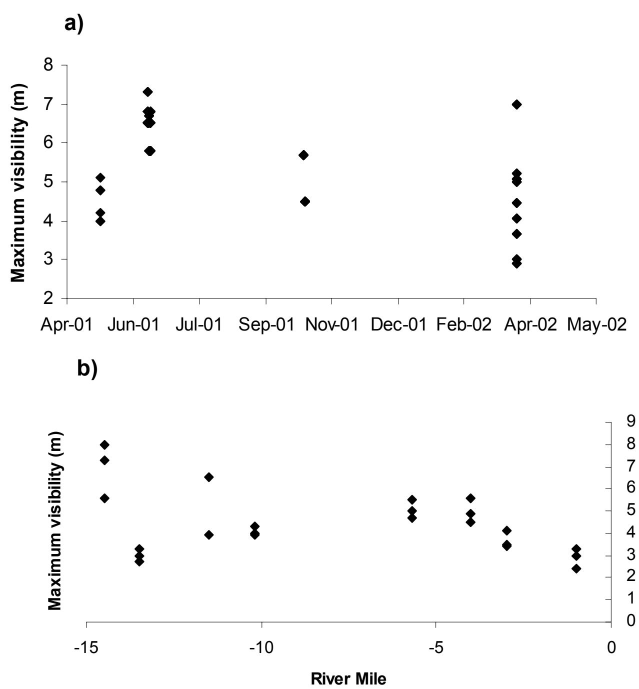

Figure 12. Maximum underwater visibility in the Lee's Ferry tailwater among (a) sampling occasions (May, June and October, 2001, and March 2002) and (b) river miles on March 2002.

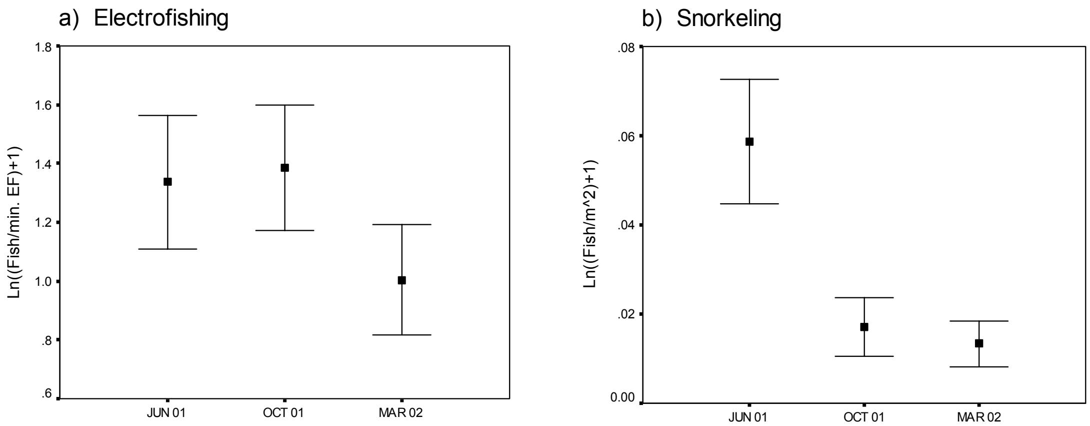

Figure 13. Seasonal (June 2001 – March 2002) trends in (a) log-transformed electrofishing CPE (Achilles boat), and (b) log-transformed fish density estimates from snorkel surveys, Lee's Ferry tailwater, Colorado River, AZ. Bars represent 95% confidence intervals of the mean.

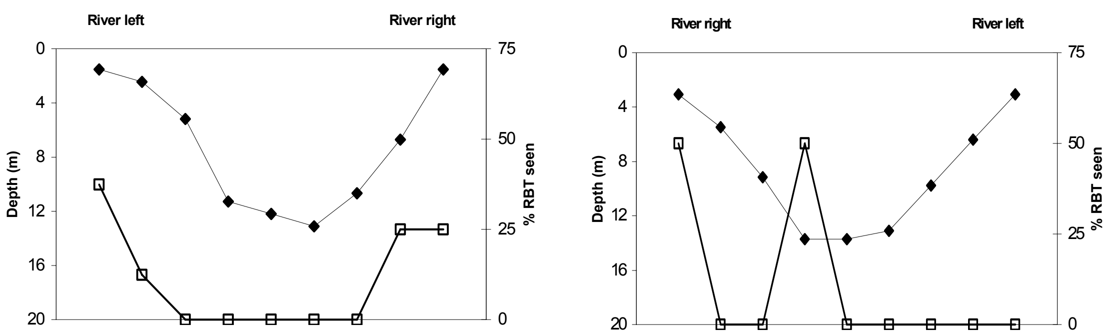

Figure 14. Schematics of cross-sectional distribution of rainbow trout (open symbols) and river depth (m; solid symbols) near RM -7.1 (sub-sites 9a-9d). X axis units are arbitrary and do not correspond to exact postion of dive observations.

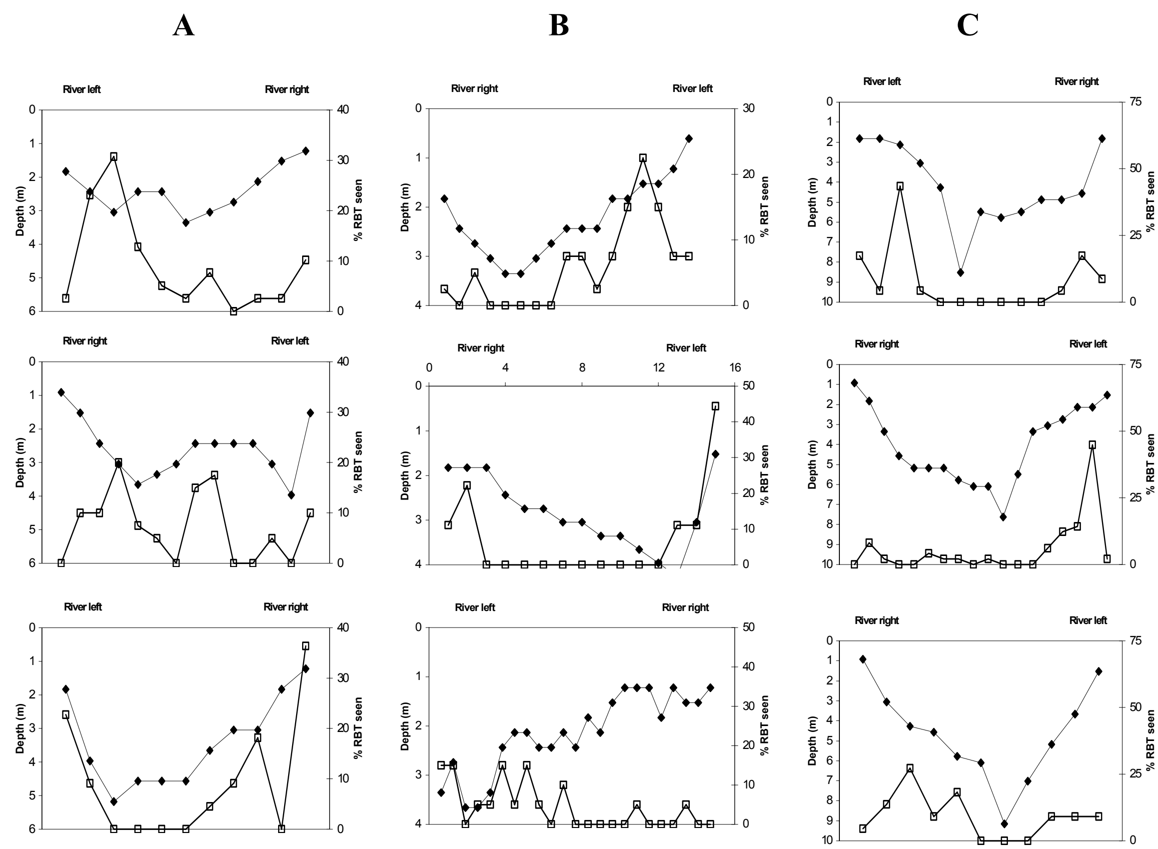

Figure 15. Schematics of cross-sectional distribution of rainbow trout (open symbols) and river depth (m; solid symbols) near(Column A) RM -4.0, (B) RM -3.0 , (C) RM -10.0. X axis units are arbitrary and do not correspond to exact postion of dive observations.

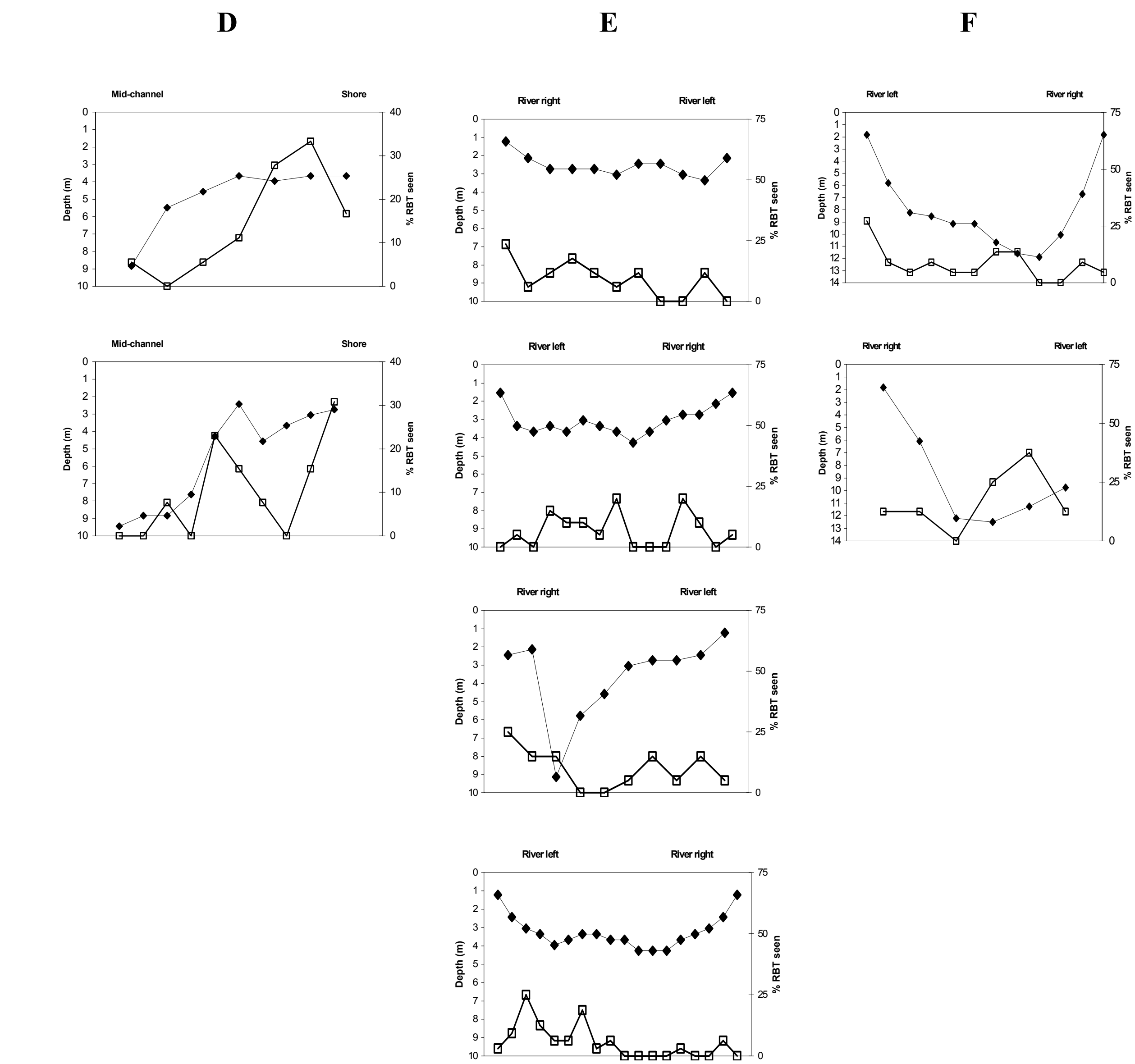

Figure 16. Schematics of cross-sectional distribution of rainbow trout (open symbols) and river depth (m; solid symbols) near (Column D) RM -1.0, (E) RM -6.0, and (F) RM -2.5. X axis units are arbitrary and do not correspond to exact postion of dive observations.

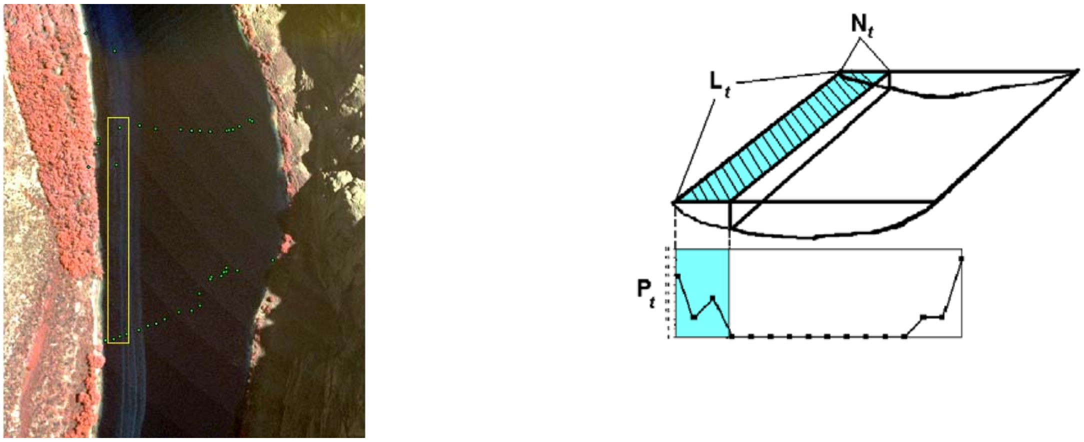

Figure 17. Population expansion technique showing (left) example of actual GPS coordinates of snorkel cross-section surveys (green dots) overlaying aerial photograph of stylized sample unit (EF/snorkel transect, yellow enclosure), and (right) schematic of variables used to estimate number of fish in areas inclusive of and adjacent to sample unit. N_t (hatch marked area) is estimated fish density in the sample unit, L_t is length of sample unit, and P_t is proportion of fish observed in sample unit relative to channel cross section.

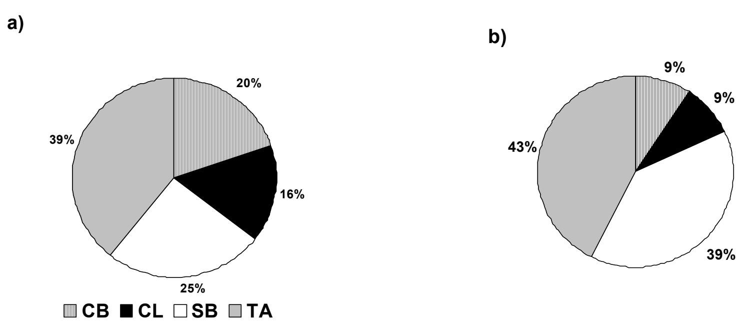

Figure 18. Proportions of shoreline types present in (a) the Lee's Ferry tailwater and (b) among electrofishing sub-sites sampled during 2001. CB = cobbler bar, CL = cliff, SB = sand bar, and TA = talus. Data are courtesy of GCMRC.

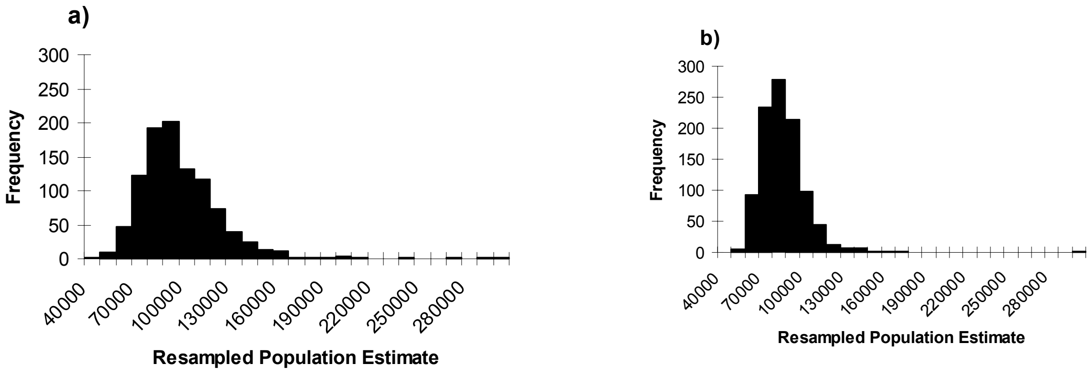

Figure 19. Frequencies of sequentially bootstrapped, habitat-stratified population estimates based on near-shore estimates of fish density from (a) electrofishing and (b) snorkel surveys. All data were collected during Jun 19-25, 2001, Lee's Ferry tailwater, Colorado River, AZ.

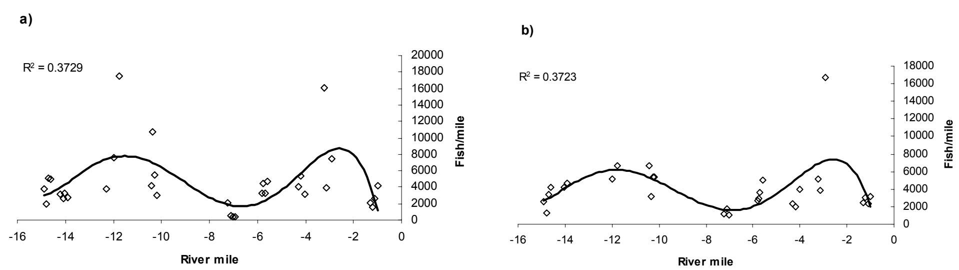

Figure 20. Estimates of fish per river mile in relation to river miles of the Lee's Ferry tailwater based on data from (a) electrofishing and (b) snorkel surveys. All data were collected during Jun 19-25, 2001, Lee's Ferry tailwater, Colorado River, AZ. Solid line represents a fifth-order polynomial regression line used to integrate population estimates.

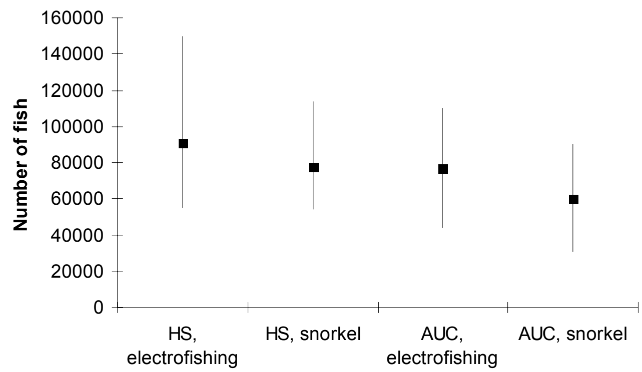

Figure 21. Estimates of mean population size of rainbow trout (> 152 mm) based on habitat-stratified (HS, x-axis) electrofishing data, habitat-stratified snorkel survey data, integrated (AUC, "area-under-curve" method) longitudinal electrofishing data, and integrated longitudinal snorkel survey data. Vertical bars represent 95% confidence intervals of the mean.

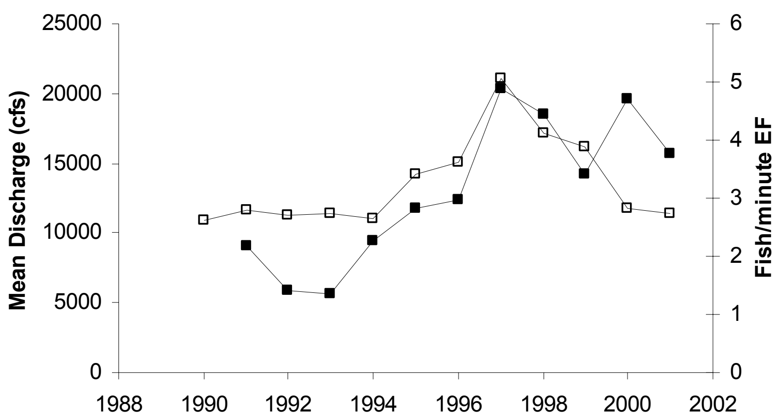

Figure 22. Untransformed electrofishing CPE (all fish combined, solid symbols) of rainbow trout and mean daily discharge from Glen Canyon Dam (open symbols), 1990-2001, Lee's Ferry tailwater, Colorado River, AZ.

Appendix 1. Sample size (N), means and standard errors (SE) CPE and relative condition (Kn) of rainbow trout in the Lee's Ferry tailwater, 1991-2001.

			CPE			Kn	
Group	year	N	Mean	SE	N	Mean	SE
<152 mm	1991	30	0.23	0.05	163	82.76	0.85
<152 mm	1992	55	0.49	0.07	554	82.32	0.48
<152 mm	1993	45	0.71	0.08	681	82.54	0.37
<152 mm	1994	45	0.99	0.12	1050	82.14	0.36
<152 mm	1995	30	1.46	0.20	1073	81.29	0.42
<152 mm	1996	52	1.09	0.12	1291	79.86	0.40
<152 mm	1997	27	2.05	0.36	781	79.46	0.55
<152 mm	1998	27	1.16	0.22	734	79.25	0.50
<152 mm	1999	27	1.05	0.20	412	78.75	0.61
<152 mm	2000	36	1.81	0.32	690	79.15	0.55
<152 mm	2001	66	1.96	0.30	504	76.34	0.58
152-304 mm	1991	30	0.27	0.04	285	86.09	0.83
152-304 mm	1992	55	0.30	0.03	597	89.34	0.53
152-304 mm	1993	45	0.30	0.03	465	87.99	0.53
152-304 mm	1994	45	0.82	0.09	1216	89.27	0.32
152-304 mm	1995	30	0.69	0.07	692	86.47	0.53
152-304 mm	1996	52	0.90	0.07	1571	86.17	0.32
152-304 mm	1997	27	0.97	0.10	789	80.11	0.40
152-304 mm	1998	27	1.56	0.19	1367	82.77	0.29
152-304 mm	1999	27	1.37	0.17	1173	78.35	0.32
152-304 mm	2000	36	1.42	0.14	1510	78.55	0.32
152-304 mm	2001	66	1.03	0.12	535	75.12	0.47
305-405 mm	1991	30	0.91	0.18	757	71.56	0.41
305-405 mm	1992	55	0.28	0.03	582	76.38	0.57
305-405 mm	1993	45	0.19	0.02	302	84.19	0.75
305-405 mm	1994	45	0.37	0.04	549	88.17	0.58
305-405 mm	1995	30	0.53	0.04	531	83.24	0.57
305-405 mm	1996	52	0.78	0.06	1353	81.46	0.35
305-405 mm	1997	27	1.54	0.16	1209	78.14	0.32
305-405 mm	1998	27	1.46	0.20	1194	78.89	0.36
305-405 mm	1999	27	1.07	0.11	897	75.87	0.42
305-405 mm	2000	36	1.25	0.12	1284	76.58	0.33
305-405 mm	2001	66	0.66	0.09	323	73.47	0.68
>405 mm	1991	30	1.05	0.20	901	72.90	0.40
>405 mm	1992	55	0.34	0.04	690	74.36	0.47
>405 mm	1993	45	0.15	0.02	231	79.89	0.87
>405 mm	1994	45	0.16	0.02	232	89.56	1.07
>405 mm	1995	30	0.14	0.02	143	81.45	1.22
>405 mm	1996	52	0.18	0.02	313	80.64	0.76
>405 mm	1997	27	0.27	0.04	204	78.60	0.91
>405 mm	1998	27	0.24	0.05	189	77.30	0.86
>405 mm	1999	27	0.15	0.03	127	79.35	1.46
>405 mm	2000	36	0.21	0.04	199	74.33	1.01
>405 mm	2001	66	0.04	0.01	27	70.74	2.25