Apache Trout Monitoring Plan

2017

Credit: AZGFD

Apache Trout populations are influenced by both short-term, stochastic events such as wildfire and long-term non-native species and land use impacts. Because of the need to determine the status of individual populations after stochastic events as well as assessing long-term changes periodically over time, the goals and objectives outlined in this plan are based on accurately and precisely estimating the status of Apache Trout populations on a 5-year interval.

A monitoring plan for small and isolated trout populations

Apache Trout Monitoring Plan

2017

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1. Purpose of the Plan

The Apache Trout *Oncorhynchus apache* is a salmonid native to Arizona with a historical range that includes the White, Black, and Little Colorado River drainages above 1,800-m elevation (USFWS 2009). Apache Trout populations have declined over time due to land management (forestry, livestock grazing, agriculture, mining), water uses (withdrawal and reservoir construction), and introductions of non-native species. The species is currently listed as threatened (since 1975) under the Endangered Species Act or ESA (USFWS 1975); the species was originally listed as endangered in 1967 (USFWS 1967).

The Apache Trout Recovery Plan has a goal of implementing necessary actions to delist the Apache Trout, with a specific objective of establishing or maintaining 30 discrete, self-sustaining, and genetically pure Apache Trout populations within the species' historical range (USFWS 2009). The criteria used to meet the recovery objective are four-fold: 1) habitat sufficient for all life history requirements for 30 self-sustaining discrete populations of genetically pure Apache Trout has been established and protected through plans and agreements with responsible land and resource management entities; 2) 30 discrete and genetically pure Apache Trout populations have been established and determined to be self-sustaining as evidenced by multiple year classes and evidence of periodic natural reproduction that persists under the range of variation experienced naturally; 3) appropriate angling regulations are in place to protect Apache Trout populations while complying with federal, state, and tribal regulations; and 4) necessary agreements are in place with the U.S. Fish and Wildlife Service, Arizona Game and Fish Department, and the White Mountain Apache Tribe (USFWS 2009).

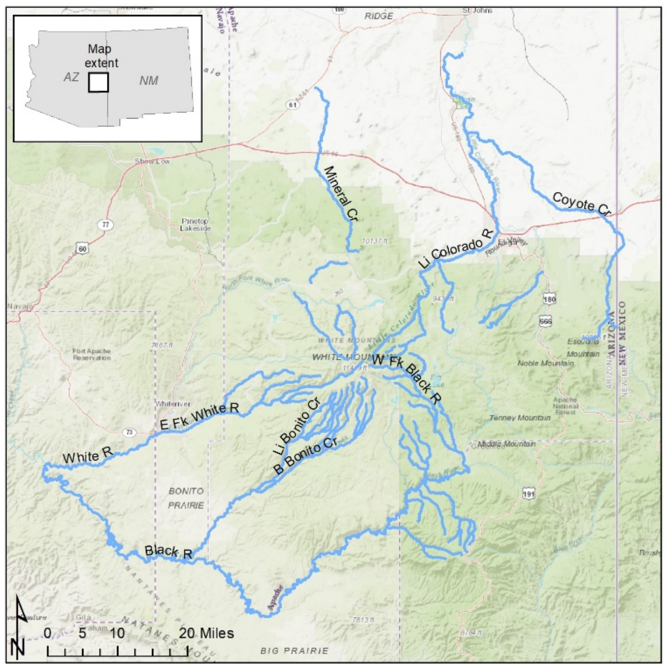


Figure 1. Map of Apache Trout streams in Arizona.

Given the existing delisting criteria there is a need to monitor Apache Trout populations to obtain up-to-date, accurate, and precise estimates of abundance, distribution, and recruitment periodically over time (Figure 1). Monitoring is defined as the repeated assessment of status of some quantity or attribute over a specified time period (Thompson et al. 1998). In relation to population monitoring, inferential monitoring whereby unbiased estimates of population attributes (e.g., population abundance) are obtained periodically over time allows for the strongest inferences regarding the status, and changes in status, of populations over time. These unbiased estimates for individual populations can be scaled up to understand the status and trends of a species range-wide if all populations are sampled or if the sampled populations are representative of all populations of a species. The goals and objectives of this Apache Trout Monitoring Plan are designed around obtaining unbiased estimates of relevant population attributes to help understand the status of individual populations, and thus the species, to inform ESA delisting and fisheries management decisions.

2. Goals and Objectives

Apache Trout populations are influenced by both short-term, stochastic events such as wildfire, and the longer term influences of non-native species and land and water use impacts (USFWS 2009). Because of the need to determine both population status soon after stochastic events and assess long-term changes periodically over time, the goals and objectives outlined in this plan are based on accurately and precisely estimating the status of Apache Trout populations on a 5-year interval. As such the goals allow for the immediate status assessment of individual populations while also facilitating the evaluation of long-term trends in populations even though trend monitoring is not explicitly a monitoring goal (see below). In addition, the primary objectives are focused on Apache Trout populations (estimating abundance, distribution, and recruitment), but the plan recommends ancillary stream habitat data collection that will aid in the interpretation of population monitoring data. Monitoring the effectiveness of conservation barriers is a second monitoring goal. The goals and objectives of are:

Goal 1: Estimate Apache Trout population status: Accurately and precisely estimate population monitoring metrics for individual Apache Trout populations every 5 years or sooner.

Objective 1 (Primary) - Apache Trout population abundance. Estimate the adult (>130-mm TL) abundance ($\widehat{N}$) of each Apache Trout population with an 80% confidence interval ($\alpha = 0.20$) no larger than 40% of estimated abundance every 5 years or sooner (i.e., if $\widehat{N} = 100$, then confidence interval width ≤ 40; see Figure 2).

Objective 2 (Secondary) - Apache Trout population distribution. Estimate the proportion of habitat (sample units or reaches) with age 1 and older Apache Trout present (1 or more individuals >80-mm TL) with an 80% confidence level ($\alpha = 0.20$) that is within 0.2 of the estimated proportion every 5 years or sooner.

Objective 3 (Secondary) - Apache Trout population recruitment. Estimate the proportion of habitat (sample units or reaches) with Apache Trout recruitment present (1 or more individuals <80mm TL) with an 80% confidence level ($\alpha = 0.20$) that is within 0.1 of the estimated proportion every 5 years or sooner.

Goal 2: Assess Conservation Barrier Effectiveness: Detect the invasion of Apache Trout populations by non-native salmonids upstream of conservation barriers.

Objective 1 (Primary): Detect non-native salmonid presence: Detect the presence of non-native salmonids within 1-km upstream of conservation barriers with a 99% probability soon after stochastic events (e.g., floods) that compromise barrier effectiveness or that otherwise allow invasion by non-native salmonids from downstream.

3. Apache Trout Monitoring

The basic monitoring elements for Goal 1 of this plan comprise a sampling design *and* field sampling procedures that yield data for estimating population parameters that are explicit to the monitoring objectives (abundance, distribution, recruitment). The sampling design defines how Apache Trout populations should be identified and then sampled in terms of the number and distribution of locations where field sampling will occur (i.e., sample units) and how frequently field sampling should occur over time for each population. The field sampling procedures define the field techniques employed at each location to estimate a population parameter. Together, these two elements are designed to meet the monitoring goal and objectives that are based on obtaining accurate and precise estimates of Apache Trout population abundance, distribution, and recruitment for each population every five years or sooner (Figure 2).

3a. Sampling Design

It is rare that all individuals in a population can be enumerated. This, then, requires then that populations are sampled (sampled statistically, not as in field sampling) in order to draw inferences (make generalizations) about a population as a whole (Hansen et al. 2007; Bonar et al. 2009). A sampling design is defined as the protocol for obtaining estimates of population attributes (often called parameter estimates) for a sampled population, and these estimates should be accurate (unbiased) and precise (high certainty) in order to make strong inferences regarding a population parameter from sample data (Thompson et al. 1998). Sampling variation in the data and sample size influence precision (Box 1), and the sampling design is focused on these elements.

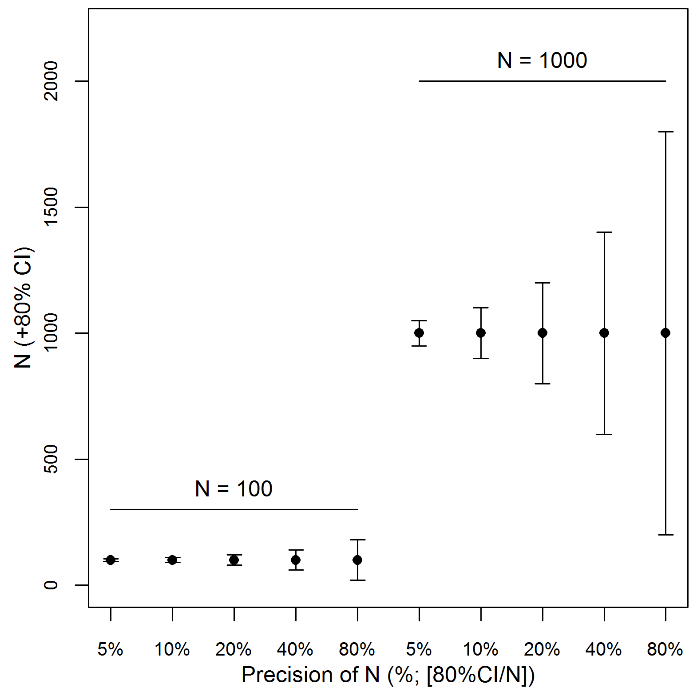


Figure 2. Example of confidence intervals at 5, 10, 20, 40, and 80% of estimated abundance ($\widehat{N}$) for two levels of $\widehat{N}$ ($\widehat{N}$ = 100, and $\widehat{N}$ = 1000).

Summary

1. **Identify Apache Trout Streams for Monitoring**: Identify demographically unique Apache Trout populations.
2. **Define Sampling Frame per Population**: For each population, determine the sampling frame by identifying the maximum potential habitat used by Apache Trout across all monitoring years, including any suitable tributary habitat, and then computing how many sample units fit within the sampling frame (i.e., divide habitat extent by reach length).
3. **Determine Number of Sampling Units Needed**: Determine number of sampling units to sample by using a sample size estimator that incorporates variance in abundance from reach to reach from past data, acceptable level of error from monitoring objectives, and the size of the sampling frame.
4. **Identify Downstream Sample Unit**: Randomly select the downstream-most sample unit for a population by randomly selecting a number from 0 to k, where k = the number of sample units in the sampling frame divided by the number of sampling units needed. Multiply the random number by the standard reach length to determine the distance upstream from the downstream boundary of available habitat at which to locate the first sample unit.
5. **Systematically Identify Additional Sample Units**: Systematically locate each additional k^{th} sample unit upstream from the first sample unit. Multiply k by the standard reach length to get the standard spacing distance.
6. Monitoring Schedule: Monitor each population at the same sample units (reaches) every five years, and populations should be monitored at approximately the same time of the year during years in which monitoring occurs.

Sampling Design Definitions

Elements of a sampling design (Thompson et al. 1998), and their application to Apache Trout monitoring:

Element: Individual, object, or time of interest. For Apache Trout monitoring the sampling element is an individual Apache Trout.

Sampling Unit: A site or plot containing a unique collection of elements. For Apache Trout monitoring the sampling unit is an individual stream reach (e.g., 100-m reach) within a population.

Sampling Frame: Complete list of sampling units. For Apache Trout monitoring the sampling frame is all potential sample units (reaches) available to be selected for field sampling for a population.

Sample: List of sampling units selected for sampling. For Apache Trout monitoring the sample is a list of all sample units (reaches) selected for sampling for a given population.

Target Population: All elements in a defined space and time interval. For Apache Trout monitoring the target population is all individual Apache Trout in a population over the duration of monitoring.

Identify Apache Trout Streams for Monitoring

All streams containing extant Apache Trout populations that are demographically independent should be monitored separately. Identification of individual streams or sets of streams containing demographically isolated Apache Trout populations should be based on past monitoring data (Johnson 2011), expert opinion, and known reintroductions while considering the influence of conservation barriers, habitat, and other factors on demographic isolation.

Define Sampling Frame per Population

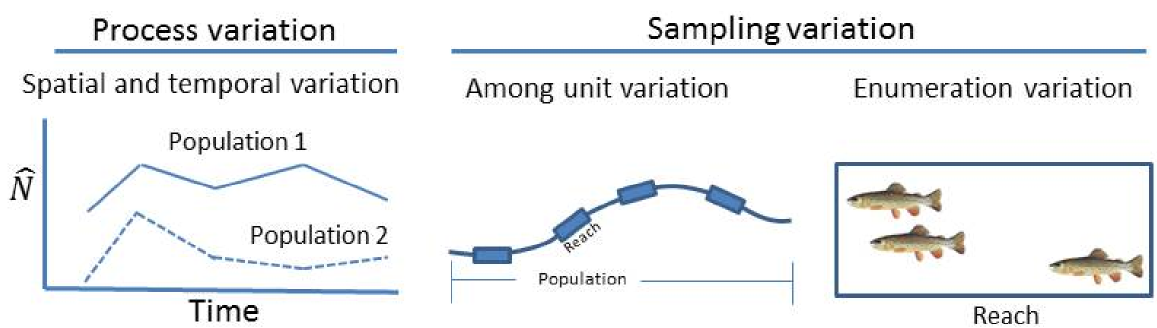
Habitat Extent: Once the streams containing each demographically isolated Apache Trout population have been identified for monitoring, the maximum extent of stream habitat potentially used by Apache Trout should be delineated as the basis for defining the sampling frame for each population to be used over the timeframe of monitoring. The maximum habitat extent should be identified using presence of conservation barriers, past population monitoring data (Johnson 2011), physical habitat suitability (Petre and Bonar 2017), and thermal suitability (Recsetar and Bonar 2013; Recsetar et al. 2014). It is important to consider inter-annual variability in habitat conditions across longer timescales when defining this maximum habitat extent as it will form the basis of the sampling frame that will likely remain constant from year to year. For example, inter-annual variability in climate may result in some of the stream being dry in a particular year, and thus no habitat being available to Apache Trout (or for field sampling) at that location. However, incorporating this habitat into the sampling frame will still yield accurate estimates of abundance and distribution because even though some sample units (reaches) are not sampled (because they are dry) the calculations of adult abundance and proportion of sites containing Apache Trout will take in to account that these sample units are unoccupied. In contrast, if certain segments of a stream are always dry or are known to never be used by a population then those segments should be omitted from the sampling frame. Defining the sampling frame for a population may be difficult and somewhat subjective and can be done using past field data and experience. However, it is an important step because this is the extent to which monitoring data are extrapolated to compute the number of adult Apache Trout in the stream (see 3c. Computing the Monitoring Metrics).

Box 1: A Note on Process and Sampling Variation:

Population abundance varies over space and time as result of environmental and demographic stochasticity that results from changes in weather, soils, vegetation, topography, predator and prey populations, ability to find mates, and other factors (Thompson et al. 1998). Together this spatial and temporal variation is referred to as ***process variation***, and is why you see differences in population parameters, such as abundance, in different areas and at different time periods.

Monitoring data often appears to vary randomly over space and time. This is, in part, driven by process variation due to population biology. However, because we cannot usually census an entire population (e.g., count each individual) during every monitoring time period this variation in the data also includes ***sampling variation***. Sampling variation reflects both among-sample-unit (reach-to-reach) variation and enumeration variation (variance in a removal or mark-recapture estimate due to incomplete detection).

As an example, a common approach to determining trout population size in small streams is to sample multiple reaches (sample units) across the stream, and estimate trout abundance in each reach using multiple-pass electrofishing and a removal estimator (e.g., Zippin estimator). To estimate the number of trout in the stream ($\widehat{N}$), the average number of trout across all reaches is multiplied by (i.e., extrapolated to) the number of sample units potentially available for sampling in the sampling frame for the population. This is equivalent to the number of 100-m reaches that could potentially occur within the defined habitat extent (e.g., 2,000-m extent / 100-m reach = 20 total reaches available to be sampled). The confidence interval around this stream-wide estimate of population size ($\widehat{N}$) reflects both components of sampling variation because the number of trout was estimated and not censused in each reach using multiple-pass electrofishing (enumeration variation) and trout were sampled only in certain reaches (20) within the stream and not the entire stream (among unit variation). The influence of sampling variation on confidence intervals of a stream-wide population estimate can be managed by increasing detection probability during electrofishing surveys by being as thorough and efficient as possible to reduce enumeration variation and increasing the number of reaches surveyed (i.e., increasing sample size) to reduce among unit variation.



Sample Unit (Reach) Length: To define the sampling frame from the habitat extent, a standard sample unit (reach) length needs to be defined. It is recommended that 100-m typically be used. This is based on fisheries convention, but also because one might also expect reach-to-reach variance (among-unit-variance in Box 1) in Apache Trout abundance to be higher if reaches are shorter because trout populations can exhibit patchiness or clustering. In very small streams, however, it may be preferable to sample shorter reaches (e.g., 50-m) to ensure replication is possible in small habitat extents, or longer reaches (200-m) may be needed to properly represent channel morphology and habitat sequencing in larger streams.

Sampling Frame: Once a standard reach length is determined, the sampling frame must be defined. Recall that the sampling frame for each Apache Trout population is a complete list of all sampling units (reaches) that could potentially be selected for sampling. This number of potentially available sample units is determined by dividing the habitat extent by the standard reach length to be used for field sampling. For example, if a population occupies 5-km of habitat and it is planned for sample units to be 100-m in length, then there are 5,000-m / 100-m = 50 potential sample units (reaches) in the sampling frame that could be selected for monitoring. The set of 50 sample units represents the sampling frame to which inferences will be made from monitoring data.

Determine Number of Sampling Units Needed

Monitoring should be conducted at a systematically selected set of sample units (reaches) per Apache Trout stream. Before a sample of units can be drawn from the sampling frame, it must be determined how many units are needed to meet precision objectives. The number of sample units (sample size), among-unit variation in abundance, and the proportion of the sample frame sampled (by way of a finite population correction of the variance estimate) all determine the precision of streamwide abundance estimates (Scheaffer et al. 2012). It is often recommended that past data be used in a sample size estimator to determine how many samples are required to estimate population parameters with a certain level of precision, such as that stated in Objective 1 where it is desired to estimate the streamwide abundance of adult Apache Trout with a confidence interval that is no larger than 40% of the abundance estimate (80%CI / $\widehat{N} \leq 0.40$). Using this precision goal and the among-unit variance in abundance from past monitoring data, the number of reaches needed for precise streamwide estimates of abundance have been computed (Table 6B). However, much of the past Apache Trout monitoring data were collected using a monitoring protocol – called the Basinwide Visual Estimate Technique (BVET) - that specifies short reach lengths (see Appendix B. Investigating the Precision of N). Because of the protocol differences, these sample sizes were converted into estimates of the percent of sampling frame that requires sampling to reach precision goals. The percent of sampling frame forms the basis of the finite population correction (fpc) used to reduce variance in many parameter estimates based on the proportion of sampling frame sampled (Scheaffer et al. 2012).

Because past monitoring data were collected using two different sampling approaches (BVET and systematic sampling), it is recommended that the sample size for monitoring be based on the best available information from the stream of interest in the following order: if past monitoring data are available and they were collected using a systematic sampling design, then use the sample size requirements as estimated for that stream (Req. n column in Table 6B). If past monitoring data were available but were collected using the BVET design, then use the percent of habitat required for that stream (Req. % column in Table 6B). If no past monitoring data are available, then use information from a similar stream (use professional judgement) or use the general guideline of 20 to 30 sites (25 to 50th percentile of Reg. n in Table 6B) or 20 to 30% of the habitat extent needed (25th to 50th percentile in Req. % in Table 6B). If the sample size or percent habitat requirements in Table 6B are logistically not feasible and it is expected that very few sites will be occupied by Apache Trout, choose a lower sample size and consider implementing an adaptive sampling approach (see Box 2). When percent habitat needed is used to determine sample size requirements, then multiply that percentage (but use a proportion) by the size of the sampling frame to return the sample size required. Using the previous example, if there are 50 units in the sampling frame (5000-m of habitat to be sampled using 100-m reaches) and it is determined that 20% needs to be sampled, the number of units to sample is 50 x 0.20 = 10 units.

Select a Systematic Sample of Units for Monitoring

Systematic Sampling: After determining the sample size needed, sampling units should be selected using a systematic sample. Systematic sampling involves randomly identifying the first sampling unit, and then selecting every k^{th} sampling unit thereafter where k refers to the interval at which samples are chosen. Systematic sampling can be easier to implement than random sampling when done in a geographic information system (GIS), on a map, or in the field. Systematic sampling also ensures thorough spatial coverage of sampling units within the sampling frame. Finally, data generated from a systematic sample approximate a random sample if the sampling units do not exhibit ordering or periodicity or are not otherwise correlated in space (Scheaffer et al. 2012).

Select a Sample of Sampling Units: Next, select a sample of units from the sampling frame. Since systematic sampling is recommended, sampling units (reaches) for Apache Trout population monitoring should be selected by randomly choosing the first, downstream most sample unit (reach) and then identifying every k^{th} sampling unit from the sampling frame. First, randomly select a number between 0 and k[1]. This number represents the number of units upstream from the downstream-most unit identified in the sampling frame (this unit is obvious if there is a conservation barrier); to convert this number into a distance, multiply the random number by reach length (e.g., 1 x 100-m reach = 100 meters). Second, select the remaining sampling units systematically by identifying every k^{th} sampling unit upstream. Using the previous example, if 10 of 50 sampling units needs to be selected, select every 5^{th} sampling unit working upstream (Figure 3). Again, to convert this into a distance to the next downstream reach boundary multiply k by the sample unit length (e.g., 5 x 100-m = 500-m); this distance represents the beginning (downstream boundary) of the next sample unit upstream. The location of each sample unit (reach) should be identified on a map, in a geographic information system (GIS), or in the field, and a global positioning system (GPS) coordinate associated with the downstream reach boundary should be recorded for each sample unit. Once sites are established for the first monitoring time period, the same reaches can be revisited in future surveys.

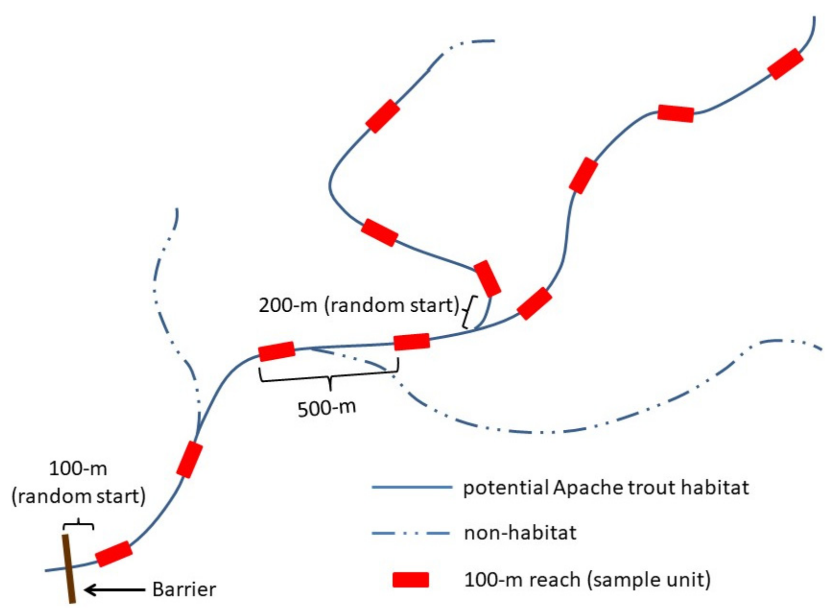


Figure 3. Schematic of a systematic sampling design for a hypothetical population. The first downstream sampling unit (100-m reach) was randomly selected and the remaining sample units were spaced every 5^{th} unit (500-m), upstream from the downstream sample unit boundary. The same sample unit selection process was used for the tributary.

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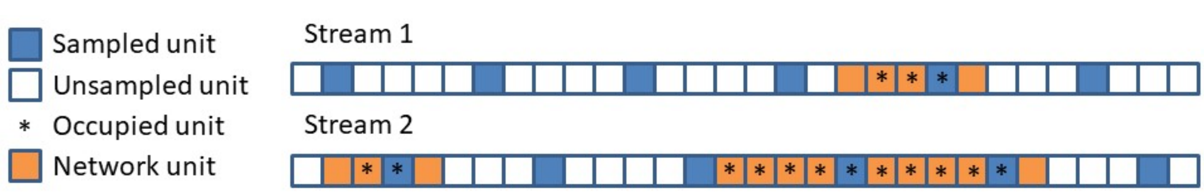
[1] If k = 10, then in Microsoft Excel type the following into an empty cell: =Randbetween(0,10)

Box 2: A Note on Adaptive Sampling for Rare Species and Small Populations:

Rare species or populations with low abundance can be difficult to sample and, thus, monitor because individuals are sparsely distributed across large areas (McDonald 2004). Sparseness in a random or systematic sample may result in few sample units (sites) being occupied by the focal species. Many unoccupied sites (zeros in the data) inflates variance estimates and, therefore, results in much uncertainty in estimates of population parameters such as stream-wide estimates of abundance. Adaptive sampling is a biologically intuitive sampling approach that can help overcome these issues. In adaptive sampling an initial random or systematic sample is drawn, field sampling for the target species occurs in the sample units initially selected, and then additional sampling occurs in new units adjacent to those occupied by the species in the initial sample. These new units are called network units. Network units continue to be added and sampled adjacent to occupied units until the unit cluster (both initial units and additional network units) is surrounded by unoccupied units. An important point is that the total sample size is not fixed prior to sampling. An advantage of adaptive sampling is that it can yield more precise parameter estimates than random or systematic sampling alone (Rosenberger and Dunham 2005; Scheaffer et al. 2012).

As an example of adaptive sampling, suppose a systematic sample of units (reaches) is selected on Stream 1 (selected units shown in blue below) and all units are sampled for Apache Trout in a first round of sampling. In a second round of sampling, additional network units are then added and sampled immediately adjacent (both upstream and downstream) to units occupied (*) by Apache Trout in the initial sample (network units in orange). New network units can continue to be added and sampled adjacent to occupied units until each unit cluster (group of adjacent occupied units) is surrounded by unoccupied units. Stream 2 shows a slight variation with a different starting point for a systematic sample and a different pattern of occupancy. When deciding whether to implement adaptive sampling after an initial systematic sample is collected, the likelihood of fish movement among reaches due to habitat features or elapsed time should be carefully considered as fish movement may bias occupancy and abundance estimates and override any precision benefits gained by adaptive sampling. There are other more complex variations of adaptive sampling, and Scheaffer et al. (2012) explain how adaptive sampling data should be analyzed to compute population means and totals (and their variances), such as for stream-wide estimates of trout abundance.



Monitoring Schedule

Each Apache Trout population should be monitored every 5 years or sooner. Populations can be scheduled as groups of populations referred to as panels, and then all populations in a panel can be sampled during the same year. This is commonly referred to as rotating panel design (Urquhart and Kincaid 1999). For example, one panel of populations sampled in year 1, another panel sampled in year 2, another in year 3, and so on with the initial panel being sampled again on year 6; an example of a rotating panel design is shown in Figure 4. When populations are resampled, the same reaches can be re-sampled during the years in which they are monitored.

Timing of Monitoring

Apache Trout populations should be monitored at approximately the same time of year each year in which monitoring occurs. Since Apache Trout reproduce once a year, new individuals are born and recruited into the population at approximately the same time each year. Mortality can occur throughout the year, even though it may be concentrated to certain times of the year. Thus, if a population was sampled in spring during one monitoring period and in fall the next (e.g., 5 years later), the difference in abundance observed from one year to the next may be due to mortality that occurs from spring to fall each year instead of long-term population trends. So, even though the exact timing of reproduction, recruitment, and mortality can vary among populations or within populations occupying habitats with a broad range of environmental conditions, monitoring populations at the same time during each monitoring period minimizes the amount of variability in monitoring data that can arise due seasonal population dynamics.

Panel	Year 1	Year 2	Year 3	Year 4	Year 5	Year 6	Year 7	...	Year *n*
Panel 1	X					X		...	
Panel 2		X					X	...	
Panel 3			X					...	
Panel 4				X				...	
Panel 5					X			...	X

Hypothetical Panels

Panel 1:	*Panel 2*:	*Panel 3*:	*Panel 4*:	*Panel 5*:
Big Bonito	Boggy/Lofer	Deep	Ord	Marshall Butte
Crooked	Coyote	EF White	Smith	Paradise
Flash	L. Bonito	Elk Canyon	Thompson	Wohlenberg
....	....	....	....	....

Figure 4. Example of a rotating panel design where each of five panels (groups) of populations are sampled every five years on a rotating basis.

3b. Field Sampling

All sample units identified for a population should be sampled during the year when monitoring is scheduled. Field sampling for Apache Trout monitoring is based on collecting data that allows the monitoring metrics to be computed, and this involves multiple-pass electrofishing at each sample unit (reach) per population. These data will allow estimation of adult Apache Trout abundance, presence of age-1 and older Apache Trout, and presence of age-0 Apache Trout. Physical habitat will also be measured within the sample unit as ancillary data to help interpret population monitoring metrics across populations and over time.

Summary

1. **Navigate to Sampling Unit (Reach)**: Enter the GPS coordinate for a sample unit into a GPS receiver, and navigate to the coordinate. This coordinate location represents the downstream boundary of the sample unit (reach).
2. **Define Sampling Unit (Reach)**: Measure upstream from the downstream boundary the predetermined length of the sample unit (reach) to identify the upstream reach boundary, being careful not to scare fish from the reach. Set 6.35-mm mesh block nets at each reach boundary.
3. **Fish Sampling**: Employ daytime multiple-pass backpack electrofishing within the reach by sampling fish in a way that maximizes capture efficiency (detection probability), exerting an equal amount of effort on each electrofishing pass, and counting and measuring fish between each pass. Conduct at least three electrofishing passes, unless no Apache Trout are collected during passes 1 and 2 then a third pass is unnecessary.
4. **Habitat Sampling**: Measure habitat attributes to compute mean wetted width, percent pool, and mean residual pool depth after electrofishing is completed. Monitor stream temperature in a temperature-sensitive area.

Navigate to Sampling Unit (Reach)

Prior to entering the field, compile a list of all sampling units to be monitored. The list should have a unique ID and the GPS coordinates for each sample unit. Use a GPS receiver or other means to navigate to the downstream boundary of the sample unit (reach) selected for sampling. If you have navigated to a GPS coordinate representing the downstream reach boundary and the coordinate is not on the stream itself because of signal interference from canyon walls or tree canopy, walk from the coordinate location perpendicular to the stream and that point should serve as the downstream reach boundary. It may be necessary to slightly move the downstream reach boundary if it is not possible to securely set a block net at this location. If the downstream reach boundary differs from the original coordinate, take a GPS reading at the exact location of the new downstream reach boundary and record the new coordinate for future identification. It is usually beneficial to use GPS averaging when collecting this coordinate location as averaging can reduce the effect of aberrant GPS reading due to multi-path effects or poor satellite geometry in challenging GPS terrain such as canyons or under tree canopies (Dauwalter et al. 2006; Wing et al. 2008). It is important to collect an accurate GPS waypoint at the exact location of the downstream reach boundary so that future field crews can navigate directly to the same sampling unit location.

Define Sampling Unit (Reach)

Once the downstream reach boundary is established, identify the upstream reach boundary by measuring 100-m upstream along the streambank taking care not to frighten fish. Since Apache Trout will be sampled using multiple-pass electrofishing to obtain a removal estimate, place 6.35-mm (0.25-in) bar mesh block nets at the upstream and downstream reach boundaries to meet the closed population assumptions associated with removal abundance estimates (Seber 1982), as salmonids have been shown to frequently move upstream and out of defined reaches during electrofishing (Peterson et al. 2005). Block nets should be secured in a way that ensures full closure of the reach throughout all three electrofishing passes without failure, including as debris (e.g., leaves, algae, small wood) dislodges and floats into the downstream block net during sampling. Secure the bottom of the block net (lead line) with rocks or other heavy material, and extend the top of the net above water to ensure fish cannot jump over the net. If needed, adjust the reach boundary to where a block net can be securely placed (e.g., move it from the middle of a deep pool to the riffle crest downstream), but make sure the final reach length along the thalweg is recorded if it deviates from 100-m. Take care not to scare fish from the sample reach when measuring reach length or securing block nets.

Fish Sampling

Fish will be sampled within the reach using daytime backpack electrofishing, which is a standard method of sampling fishes in coldwater streams in North America per American Fisheries Society standards (Dunham et al. 2009). Electrofishing should be conducted using a backpack electrofisher capable of pulsed direct current (DC) and configured with one anode and a rattail cathode. A minimum of two persons are needed to conduct sampling. One person will operate the electrofisher (the fisher) and another one or two persons will net fish (dip netters); additional electrofishers and netters maybe needed for larger streams to maximize sampling efficiency. Crew members should have polarized eye glasses to improve visibility below water and for eye protection. They should also wear waterproof waders, sturdy wading boots with good traction (sticky rubber, metal studs, etc), lineman's gloves, or any other agency-required personal protective equipment. Reynolds and Kolz (2012) also discuss various aspects of electrofishing safety. Before sampling begins, the settings on the electrofisher should be set and test sampling should be conducted outside of the study reach. Ensure the electrofisher is set to pulsed DC, and it is recommended to begin test sampling with a 30-Hz DC pulse at 12% duty cycle (4 ms) and 220-280 V (Dunham et al. 2009; Reynolds and Kolz 2012). If these settings are ineffective, then increase voltage incrementally at 100-V intervals to a maximum of 1,100 V. If electrofishing is still ineffective, decrease voltage to 300-V and then increase pulse frequency at intervals of 10-15 Hz to a maximum of 60 Hz (Dunham et al. 2009). These guidelines are intended to balance maximizing electrofishing efficiency while minimizing stress and electrofishing injury to the fish (Snyder 2003) and are generally in agreement with other electrofishing guidelines in waters containing salmonids listed under the Endangered Species Act (Schaeffer and Logan 2000). Reynolds and Kolz (2012) is also an excellent reference for understanding the theory and practice of electrofishing.

Once testing shows electrofishing settings to be effective, begin sampling by conducting two electrofishing passes in the sample unit (reach). If no Apache Trout are collected in pass 1 <u>and</u> pass 2, then no additional passes are needed; however, if it is perceived that sampling efficiency is low (i.e., large stream or complex habitat) or it is believed that Apache Trout may be in the reach then a third pass should be conducted. If one or more Apache Trout are collected during pass 1 <u>or</u> pass 2, then at least a third pass should be conducted. Each pass should be conducted in a zig-zag or herring bone pattern with attempts to sample all possible habitats and maximize capture efficiency (Dunham et al. 2009). Care should be taken to minimize any sampling bias due to preconceived notions of suitable habitat. The fisher should alternate application of current with periods of no current to avoid pushing fish ahead of the electrical field. The fisher should release the anode switch when fish are immobilized to minimize exposure to electricity and reduce the potential for injury. The primary netter(s) should closely follow the fisher, collect all Apache Trout immobilized during each electrofishing pass, and hold them in an aerated bucket or similar vessel to minimize fish stress. The amount of electrofishing effort should remain constant across all electrofishing passes in order to maintain assumptions of equal effort per pass as required by most removal estimators of abundance (Seber 1982). This can be done using the timer on the electrofisher. If the decrease of Apache Trout caught during successive electrofishing passes is erratic (e.g., Pass 1 = 16; Pass 2 = 15; Pass 3 = 3) or more trout are caught in later passes (e.g., Pass 1 = 32; Pass 2 = 10; Pass 3 = 12) then consider doing a fourth and

Figure 5. Measuring total length of an Apache Trout. Credit: AZGFD.

possibly fifth electrofishing pass. While many depletion electrofishing data show high probability of detection (i.e., high efficiency) resulting in a rapid depletion and precise abundance estimates (Appendix A. Evaluating a Need for Stratified Sampling), erratic depletion patterns can result in abundance estimates that have larger than desired confidence intervals; conducting additional electrofishing passes can improve the precision of abundance estimates for a given reach.

Each Apache Trout collected during each electrofishing pass should be measured to the nearest millimeter total length (TL). If large numbers of fish cannot be processed quickly, or if individual fish cannot be measured accurately because of a tendency to thrash, then consider using an anesthesia to sedate fish and reduce stress (Jennings et al. 2012). Fish can be released outside of the sample reach after each electrofishing pass if it can be assured that fish do not move into an adjacent, yet-to-be sampled sampling unit. Alternatively, sampled fish can be retained in a holding container on the streambank until all electrofishing passes have been completed, after which all fish can then be released back into the reach; however, one must ensure adequate conditions are present in the holding container to reduce stress and ensure fish survival. Holding fish streamside and placing them back in the reach after sampling may be desirable if adaptive sampling will occur in adjacent reaches (Box 2).

Habitat Sampling

Three easy-to-measure habitat attributes will be measured in each sample unit following electrofishing: mean wetted width, percent pool and residual pool depth; a fourth habitat variable, stream temperature, will be monitored in at least one location in each stream. Collecting data on these habitat attributes will facilitate the interpretation of population data within streams, across streams, and across years. Mean wetted width will allow the area of the sampling reach to be quantified, and pools have been shown to be important Apache Trout habitat (Petre and Bonar 2017). Percent pool habitat and residual pool depth have shown to be precisely and consistently measured among different field crews that receive proper training (Roper et al. 2002). Like all salmonids, stream temperature is a critical habitat element for Apache Trout that have been shown to have low survival at temperatures 25°C or greater (Recsetar and Bonar 2013; Recsetar et al. 2014).

Mean Wetted Width: Mean wetted width is defined as the average wetted stream width within the sample unit (reach) and should be measured by taking 5 to 10 stream width measurements within the reach. Width measurements should be taken systematically throughout the reach. Five measurements should suffice in a reach with uniform width, but 10 should be taken if stream width varies substantially throughout the reach. Sum the individual wetted width measurements and divide by the number of measurements taken to obtain the mean wetted width. Mean wetted width should be expressed in meters. Linear and areal fish densities are both common ways to report fisheries data (Dunham et al. 2009).

Percent Pool: Percent pool is defined as the percentage of the reach length comprised by pools. All pools in a sample unit (reach) should be identified as slow-water habitats (i.e., channel unit or mesohabitat) with less than 1% gradient that are normally deeper and wider than habitat units immediately upstream and downstream (Hawkins et al. 1993; Armantrout 1998), and the length of each individual pool should be measured. All pools that are at least as long as the stream is wide should be identified. To compute percent pool, sum the individual pool lengths, divide by the sample unit length (reach length), and multiply by 100 to obtain the percentage of the reach that is pool habitat.

Residual Pool Depth: Residual pool depth is the average residual pool depth for all identified pools in the reach. The residual pool depth of each identified pool is derived by subtracting the pool tail depth (thalweg at a hydraulic control, such as riffle crest) from the maximum pool depth (at thalweg; Figure 6). If a majority of the pool is within the reach but the pool tail or maximum pool depth is outside the reach it is appropriate to take the measurement outside of the reach boundary.

Stream Temperature: Stream temperature should be monitored continuously in a minimum of one location per Apache Trout stream. Temperature monitoring should occur in temperature sensitive areas or areas subject to warming, such as those impacted by livestock grazing or wildfire, but it is also important to consider site accessibility to easily download data or replace thermographs if they are lost (e.g., floods). Temperatures should be monitored continuously throughout the year; at a minimum they should be monitored continuously during the summer. One hour recording intervals are frequent enough to ensure near daily maximum temperatures are recorded. Thermographs should be deployed in housing that minimizes the effect of direct solar radiation on the thermograph, such as in a PVC housing that is epoxied to a rock or attached to a steel rod (Isaak et al. 2013). Temperature data should be summarized as average weekly maximum temperatures, or similar metrics, that are relevant to *in situ* habitat suitability for salmonids (Dunham et al. 2005).

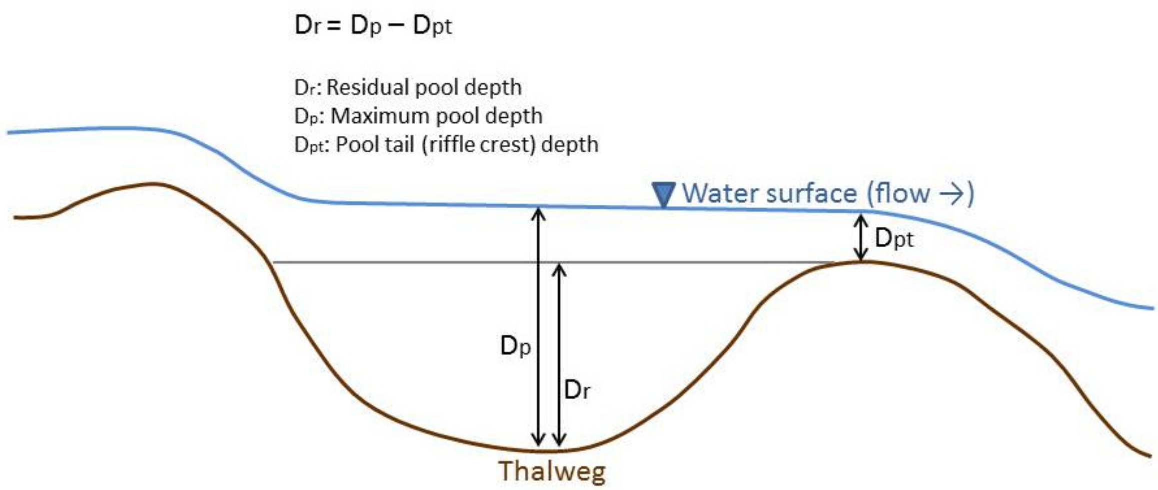


Figure 6. Schematic showing measurement of residual pool depth by measuring the maximum pool depth along the thalweg and subtracting off the depth of pool tail (riffle crest) at the thalweg.

3c. Computing the Monitoring Metrics

The metrics computed from monitoring data for each Apache Trout population reflect adult abundance (>130-mm TL), distribution of age-1 and older individuals (>80-mm TL), and distribution of reproduction and are explicitly linked to the primary and secondary objectives of monitoring Goal 1.

Adult Abundance ($\widehat{N}$)

The abundance of adults in each Apache Trout population will be computed by extrapolating the mean number of adult Apache Trout (number of Apache Trout >130mm TL) across all sample units (reaches) to the sampling frame (habitat extent).

Box 3: Bias in Removal (Depletion) Estimators:

Some studies have shown removal estimators of abundance to be biased low (12 to 88%) when computed using data from multiple-pass electrofishing in streams (Peterson et al. 2004; Rosenberger and Dunham 2005; Meyer and High 2011). This bias often results from violation of the assumption of constant capture probability (efficiency) among individuals (across all electrofishing passes) that is required for some commonly-used removal estimators. Heterogeneity in capture probability occurs because more fish that are easy to catch (they have a higher detection probability) are caught during event (pass) 1, and the more difficult to catch fish (they have a lower detection probability) remain during subsequent events (pass 2 and higher); the amount of bias has been shown to be influenced the same factors that affect capture probability: fish size, temperature, habitat complexity, etc. For example, when complex habitat is available the fish that use cover are harder to catch than those that do not, which results in easy-to-catch fish being collected more often during early electrofishing passes and harder-to-catch fish remaining during subsequent passes (referred to as heterogeneity in capture efficiency or detection). The bias in removal sampling, if present, will cause stream-wide or basin-wide estimates of abundance to be under-estimated (Sweka et al. 2006; Meyer et al. 2014).

What can be done about this bias? First it should be determined if bias exists by way of a pilot study where a removal estimate (Zippin, etc.) is compared to a known number of fish in streams similar to those being monitored. For example, fish could be collected, marked and allowed to recover, and then sampled using multiple pass electrofishing (Rosenberger and Dunham 2005; Dauwalter and Fisher 2007). Then the number of marked fish per pass could be used in a removal estimator to determine if the removal estimate contains the known number of initially marked fish (this assumes marked fish behave similarly to unmarked fish).

If bias is observed, then there are a few approaches to address it. One approach is simply to acknowledge that removal estimates are biased low, and therefore any streamwide estimates of abundance are probably underestimates. Meyer et al. (2014) used removal sampling to estimate abundance of Redband Trout *O. mykiss* in Idaho streams, and they acknowledged that their estimates of abundance are conservative due to bias in removal estimates observed in a separate study (Meyer and High 2011).

A second approach would be to develop a correction factor that could be used to calibrate the biased removal estimates (y / c; where y is the removal abundance estimate for a reach, and c is the correction factor) or develop an unbiased capture efficiency model that that can be used to estimate abundance from total catch (C / p; where C is total catch across all passes, and p is capture efficiency). The correction factor or capture efficiency model could be developed as part of the same study used to ascertain whether bias exists. Keep in mind that the correction factor should be developing using data from streams similar to those where the correction is to be applied, and the correction factor (c) or efficiency estimates (p) may need to be modeled as a function of factors known to affect capture efficiency (e.g., habitat complexity, stream size). Several authors describe how to calibrate fisheries data (Peterson et al. 2004; Rosenberger and Dunham 2005; Peterson and Paukert 2009).

A third approach is to use a more advanced abundance estimator for removal data. Saunders et al. (2011) used Program MARK to model abundance using an estimator that allowed capture efficiency to vary across electrofishing passes while also accounting for changing efficiency associated with fish size. These advanced estimators should be explored if needed, but may require the assistance of a biometrician.

Adult abundance per sample unit: The abundance of adult Apache Trout (number of Apache Trout >130mm TL) at each sample unit *i* within each population *j* (y_{ij}) should be estimated using a removal (or depletion) estimator. Commonly used depletion estimators are the Zippin estimator (Zippin 1958), which is employed in the commonly used software MicroFish (Van Deventer and Platts 1989). Zippin, Carle-Strub (Carle and Strub 1978), and other estimators are available in the FSA package (Ogle 2017) in Program R (R Core Team 2015). Below is an example `output` from the Program R **code** needed to estimate abundance of trout with a Zippin estimator using data from 4-pass electrofishing where 24 fish were caught in the first pass, 14 in the second pass, 10 in the third pass, and 7 in the fourth (# indicates annotated comments only):

```
require(FSA) #loads FSA package that contains removal function
ct <- c(24,14,10,7) #creates concatenated catch string as 'ct'
est <- removal(ct, method="Zippin") #submit 'ct' to removal command, specify estimator, assign to 'est'
summary(est) #submit removal command output 'est' to summary function
```

```
     Estimate Std. Error
No 66.0000000  8.0175268
p   0.3571429  0.0841718
```

```
confint(est) #summary command does not output confidence intervals, so submit 'est' to confint function
```

```
      95% LCI    95% UCI
No 50.2859363 81.7140637
p   0.1921691  0.5221166
```

Thus, we can say with 95% confidence, using the Zippin removal estimator, that there are between 50 and 82 trout in the reach electrofished (best estimate is No=66 fish). The estimator also tells us that the detection probability for an individual fish on a given pass, *p*, is 35.7% (95% CI: 19.2 to 52.2%). It is important to note that removal estimators of abundance have been shown to be biased low in some studies, and there are several ways that this bias, if present, can be evaluated and addressed in removal data collected as part of a monitoring program (see Box 3).

Extrapolate to Adult Abundance ($\widehat{N}$): Extrapolation of the mean abundance of adult Apache Trout will be done using an estimator where the mean number of Apache Trout across all sampling units (reaches) is multiplied by the number of units N_i in the sampling frame for population *i* (N_i= habitat extent, in meters, divided by reach length). The estimated abundance of adult Apache Trout (>130-mm TL) for population *i* is (from Scheaffer et al. 2012):

$$\widehat{N}_i = N_i \bar{y}_i = \frac{N_i \sum_{j=1}^{n_i} y_{ij}}{n_i}$$

where $\widehat{N}_i$is the estimated abundance of adult Apache Trout for population *i*, N_i is the total number of sampling units available in the sampling frame for population *i*, $\bar{y}_i$ is the mean number of adult Apache Trout (>130-mm TL) per sampling unit (reaches) across all sample units *j* sampled in population *i*, y_{ij} is the number of adult Apache Trout (>130-mm TL) in sample unit *j* in population *i*, and n_i is the number of sample units (reaches) sampled in population *i*.

The variance of $\widehat{N}_i$ is:

$$\widehat{V}(\widehat{N}_i) = \widehat{V}(N_i\bar{y}_i) = N_i^2 (\frac{s_i^2}{n_i}) \left(\frac{N_i - n_i}{N_i}\right)$$

where N_i and n_i are as defined above, and s_i^2 is the variance in abundance of adult Apache Trout across all sample units (reaches) in population *i*. Recall that s_i^2 is computed as: $s_i^2 = \frac{\Sigma(\bar{y}_i - y_{ij})^2}{n_i - 1}$, with all terms as defined above. The $\left(\frac{N_i - n_i}{N_i}\right)$ term is a finite population correction (fpc) that shrinks the observed variance by the proportion of the sampling frame (N_i) or habitat extent sampled across all sample units (n_i) for population *i*. It is important to note that $\widehat{V}(\widehat{N}_i)$ as specified is based on a simple random sample, and in order for a systematic sample to approximate the variance of a random sample it is assumed that adjacent sample units (reaches) are uncorrelated and that N_i is large; these assumptions can be evaluated when sufficient monitoring data are collected, and other variance estimators exist if assumptions are violated (see Scheaffer et al. 2012).

The variance $\widehat{V}(\widehat{N}_i)$ can then be used to compute 80% confidence bounds on $\widehat{N}_i$. The upper 80% confidence limit on $\widehat{N}_i$ can be computed as:

$$\widehat{N}_i + t_{\alpha=0.2/2,\mathrm{n}_i-1}\sqrt{\widehat{V}(N_i\bar{y}_i)}$$

and the lower 80% confidence limit as:

$$\widehat{N}_i - t_{\alpha=0.2/2,\mathrm{n}_i-1}\sqrt{\widehat{V}(N_i\bar{y}_i)}$$

where $\widehat{V}(N_i\bar{y}_i)$ and n_i are as defined above, and $t_{\alpha=0.2/2,\mathrm{n}_i-1}$ is the *t*-value from a *t*-distribution table at α = 0.2/2 = 0.1 and n_i-1 degrees of freedom[2]; α is divided by 2 for each side of the confidence interval to match the total α = 0.2 for an 80% confidence interval. Note that α = 0.2 (80% confidence interval) to match the level of precision stated in monitoring Goal 1, Objective 1.

Distribution

The distribution of Apache Trout will be quantified as the proportion of sample units (100-reaches) occupied by age-1 and older Apache Trout (≥80-mm TL). Age-1 and older Apache Trout occupancy in a sample unit (reach) is coded as y = 1 of one or more Apache Trout ≥80-mm TL are present using data from all electrofishing passes and y = 0 if none are present. The estimate of the proportion of sample units (reaches) in population *i* with age-1 and older Apache Trout present ($\hat{p}_i$) is:

$$\hat{p}_i = \bar{y}_i = \frac{\sum_{j=1}^{n_i} y_{ij}}{n_i}$$

where y_{ij} is the presence of one or more Apache Trout ≥80-mm TL (presence = 1, absence = 0) in electrofishing passes at sample unit *j* in population *i*, and n_i is the number of sample units (reaches) sampled in population *i*.

---

[2] In Microsoft Excel use the =TINV(probability,degrees_freedom) function, where probability = α/2 and degrees of freedom = n - 1.

The variance for the estimated proportion of sample units (reaches) occupied is:

$$\hat{V}(\hat{p}_i) = \frac{\hat{p}_i \hat{q}_i}{n_i - 1}\left(\frac{N_i - n_i}{N_i}\right)$$

where n_i is as defined above, $\hat{q}_i = 1 - \hat{p}_i$, N_i is the total number of sample units in the sampling frame for population *i*, n_i is the number of sample units reaches) sampled in population *i*. As mentioned previously, the $\left(\frac{N_i - n_i}{N_i}\right)$ term is a finite population correction that shrinks the observed variance by the proportion of sampling frame (N_i) sampled across all sample units (n_i).

The variance $\hat{V}(\hat{p}_i)$ can then be used to compute 80% confidence limits on $\hat{p}_i$. The upper 80% confidence limit on $\hat{p}_i$ can be computed as:

$$\hat{p}_i + t_{\alpha=0.2/2,\mathrm{n}_i-1}\sqrt{\hat{V}(\hat{p}_i)}$$

and the lower 80% confidence limit as:

$$\hat{p}_i - t_{\alpha=0.2/2,\mathrm{n}_i-1}\sqrt{\hat{V}(\hat{p}_i)}$$

where $\hat{V}(\hat{p}_i)$ and n_i are as defined above, and $t_{\alpha=0.2/2,\mathrm{n}_i-1}$ is the *t*-value from a *t*-distribution table at α = 0.2/2 = 0.1 and n_i-1 degrees of freedom; α is divided by 2 for each side of the confidence interval to match the total α = 0.2 for an 80% confidence interval. Note that α = 0.2 (80% confidence interval) to match the level of precision stated in monitoring Goal 1, Objective 2.

Reproduction

Similar to the distribution of age-1 and older Apache Trout, the distribution of Apache Trout reproduction will be quantified as the proportion of sample units (reaches) where age-0 Apache Trout are present across all electrofishing passes. Presence of reproduction in a sample unit (reach) is coded as y = 1 if one or more age-0 Apache Trout (<80-mm TL) are present and y = 0 if none are present. The estimate of the proportion of sample units (reaches) in population *i* with Apache Trout reproduction present ($\hat{p}_i$) is:

$$\hat{p}_i = \bar{y}_i = \frac{\sum_{j=1}^{n_i} y_{ij}}{n_i}$$

where y_{ij} is the presence of one or more age-0 Apache Trout (presence = 1, absence = 0) in all electrofishing passes in sample unit *j* in population *i*, and n_i is the number of sample units (reaches) sampled in population *i*.

The variance for the estimated proportion of sample units (reaches) with recruitment is:

$$\hat{V}(\hat{p}_i) = \frac{\hat{p}_i \hat{q}_i}{n_i - 1}\left(\frac{N_i - n_i}{N_i}\right)$$

where n_i is as defined above, $\hat{q}_i = 1 - \hat{p}_i$, N_i is the total number of sample units in the sampling frame for population *i*, n_i is the number of sample units (reaches) sampled in population *i*. As noted above, the variance ($\hat{V}(\hat{p}_i)$) can be used to compute 80% confidence intervals as noted in Goal 1, Objective 3.

Rangewide Abundance

Because Apache Trout conservation actions may include reconnecting individual populations, or rangewide estimates of abundance may be of interest, abundance estimates of individual populations ($\widehat{N}_i$) can be added to assess abundance (or changes in) across multiple populations. Here, the abundance estimates of individual populations can be summed across multiple populations or even rangewide. However, in order to quantify uncertainty the variance estimates for individual populations needs to be summed (variances have an 'additive' statistical property), and the summed variance is then used to construct confidence intervals. Total abundance is computed as:

$$\widehat{N}_{total} = \sum_{i=1}^{n} \widehat{N}_i$$

where $\widehat{N}_i$ is the estimated abundance of Apache Trout in population *i*, and *n* is the number of populations being incorporated in the total abundance estimate. The variance for $\widehat{N}_{total}$ is:

$$\hat{V}(\widehat{N}_{total}) = \sum_{i=1}^{n} \hat{V}\left(\widehat{N}_i\right)$$

where $\hat{V}\left(\widehat{N}_i\right)$ is the variance for the abundance estimate ($\widehat{N}_i$) of population *i*, and *n* is the number of populations being incorporated into the total abundance estimate. The variance estimate can then be used to compute confidence limits as shown above.

4. Non-Native Trout Invasion Monitoring

Monitoring for invasion of non-native trout should be focused on detecting their presence upstream of conservation barriers on an as-needed basis. This need could arise after stochastic events (e.g., floods) compromise barrier integrity or other factors suggest a conservation barrier has been ineffective and an injurious species (e.g., non-native salmonids) may have invaded an Apache Trout population upstream.

4a. Sampling Design

Monitoring for invasion is based on sampling a continuous reach from the barrier upstream for 1-km or more, and then determining whether invasion has occurred based on whether one or more non-native trout were observed. This determination should be made with a certain level of confidence. Since only one continuous reach is to be sampled, inferences regarding invasion only apply to the sampled area and cannot be extended beyond the sampled reach. Doing so would require a valid statistical sampling design for strong inference.

Summary

1. **Identify Barriers for Monitoring**: Identify all conservation barriers associated with extant or future Apache Trout populations.
2. **Identify Invasion Sample Unit**: Identify a 1-km continuous reach upstream of each barrier for sampling.

Identify Barriers for Monitoring

All conservation or natural barriers associated with extant or future Apache Trout populations should be inventoried and periodically to assess whether stochastic events or other factors have compromised their effectiveness at isolating Apache Trout populations from invasion by injurious species (e.g., non-native salmonids) residing downstream.

Identify Invasion Sample Unit

A sample unit immediately above the barrier should then be established. The sample unit should represent a continuous stream reach from the barrier at the downstream end to 1-km or more upstream.

4b. Field Sampling

Summary

1. Invasive Fish Sampling:

Invasive Fish Sampling

Once monitoring is triggered through a visual assessment and the sample unit (1-km reach) is established above the conservation barrier, then the sample unit should be thoroughly sampled using backpack electrofishing procedures. At least one electrofishing pass should be conducted whereby all habitats are thoroughly sampled in a way that maximizes capture efficiency. This could require more than one dedicated netter and possibly two electrofishers and two or more dedicated netters in larger streams. Electrofishing settings should optimized for immobilization of salmonids, but if Apache Trout are present settings should be set to avoid electrofishing injury. See Fish Sampling section above. If capture efficiency, based on past data or experience, is thought to be less than 80%, then at least a second electrofishing pass should be conducted. If experience suggests habitat conditions facilitate movement by invading fishes through the first 1-km above a barrier, then the sample units should be lengthened to encompass all habitat believed to be easily invaded. Targeted sampling above the formal sampling reach can also be conducted based on professional judgement.

4c. Computing Invasion Metrics

The probability of detecting at least one individual of the invading species (e.g., non-native trout) above a conservation barrier is dependent upon: 1) the detection probability (d_j) of an individual during one electrofishing pass, 2) the number of electrofishing passes conducted (k), and 3) the abundance of the invading species in the reach sampled ($\widehat{N}$) (typically unknown). If more than one pass is conducted, then the detection probability for an individual invader is:

$$d = 1 - \prod_{j=1}^{k}(1 - d_j)$$

where d is the detection probability of an individual invader held constant across all k electrofishing passes, and d_j is the detection probability of an individual invader during electrofishing pass j. Similarly, the probability of detecting at least one individual of $\widehat{N}$ total invaders is:

$$p_{\text{detection}} = 1 - (1 - d)^{\widehat{N}}$$

where d is as defined above, and $\widehat{N}$ is the estimated number of invaders in the reach (typically unknown).

As an example, assume five non-native brown trout permeated a conservation barrier and invaded a conservation population of Apache Trout upstream. To detect this invasion, assume two electrofishing passes were completed within a continuous 1-km reach upstream of the barrier. Also assume that the one-pass detection probability for each individual brown trout is 0.7 (remaining constant for both passes) and reflects field crew experience, stream size, and habitat complexity (each of these typically influence detection probability), that is, there is a 70% chance for each individual brown trout being detected within the sampled area during one electrofishing pass. Thus, the detection of each individual trout (assuming the detection probability is the same for all five trout) across the two electrofishing passes is: $d = 1 - (1 - 0.7) \times (1 - 0.7) = 1 - (1 - 0.7)^2 = 0.91$. Then, the probability of detecting one or more of the five invading brown trout in the sampled reach is: $p_{detection} = 1 - (1 - 0.91)^5 = 0.999$. As a result of this sampling effort, one would be almost certain that at least one brown trout would be detected during sampling, and the brown trout invasion (and barrier ineffectiveness) would be detected with nearly 100% certainty. Even if only one brown trout invaded upstream, the probability of detecting it given two electrofishing passes would be 0.91 or 91%. It is important to recognize that this detection probability only applies to the sampled area and not to the unsampled portion of habitat upstream; estimating the detection probability within the entire extent of Apache Trout habitat potentially invaded would require a statistically valid sampling design such as those mentioned above for monitoring Apache Trout populations.

Monitoring to Inform Management and Recovery

Fisheries management, including management of ESA-listed species, is often adaptive based on the current state of the fishery (or species). Necessarily, then, adaptive management requires knowledge of the current state of the fishery as well as continued monitoring and evaluation to determine if management actions are helping to accomplish goals and objectives (McMullin and Pert 2010). Endangered species management is a special case of fisheries management, but the basic tenants are the same. Management is guided by recovery plan goals and objectives, and monitoring and evaluation is required to determine whether those goals and objectives are being met and whether the species can be removed from the Endangered Species List. As outlined in the introduction, the Apache Trout Recovery Plan specifies goals and objectives that can be evaluated to determine if the species is recovered (USFWS 2009). This monitoring plan was developed with a goal of determining the current status of extant Apache Trout populations so that current and rigorous data are available to inform future recovery planning, management actions, and delisting processes.

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Appendix A. Evaluating a Need for Stratified Sampling

Stratified sampling is a type of sampling design whereby a statistical population is divided into homogeneous subgroups prior to sampling, and stratification can yield several benefits in biological monitoring. First, it can be used to increase the precision around estimates of population parameters, such as abundance, by allocating more samples (sampling units, sites, or reaches) to strata that are thought or known to be more variable across sample units (Thompson et al. 1998). Increasing the number of sample units in the more variable strata will reduce overall variance in the parameter estimate of interest (Scheaffer et al. 2012). Second, stratification may be used to ensure certain strata are included in sampling, such as stratifying by common and rare habitat strata to ensure at least some rare habitats are sampled for later comparisons (possibly using a different sampling design). For example, Roberts and Rahel (2008) used a stratified estimator to estimate the number of fish entrained in an irrigation canal because they wanted to ensure that all areas near canal head gates and siphons (the first stratum) were sampled because fish congregated in those areas, whereas the remaining portions of the canal (the second stratum) were sampled using a random sampling design.

Past monitoring data and field observations suggested that Apache Trout abundance may differ between confined and unconfined valleys (canyons versus meadows) and that there may be a need to stratify by valley type during monitoring. Thus, we used past Apache Trout monitoring data to evaluate the need for a stratified sampling design for future Apache Trout monitoring. We evaluated this need by assessing whether Apache Trout densities differed or were more variable among Rosgen stream types (a stream classification system that accounts for valley confinement; Rosgen 1996) and habitat type (channel unit or meso-habitat type). Differences in means and variances of Apache Trout densities by valley confinement or habitat type could indicate a need for stratification during monitoring, especially unequal variances could indicate need to allocate more samples to strata where densities are more variable among different stream reaches.

Objective

Evaluate differences in Apache Trout densities (both means and variances) among Rosgen channel types and habitat types (channel units).

Methods

The need for stratification during Apache Trout monitoring was evaluated through analysis of past monitoring data. Apache Trout streams were sampled according to the Basin Visual Estimation Technique (BVET)(Hankin and Reeves 1988; Dolloff et al. 1993) during two time periods: 2001-02, and 2012-13. During each survey Apache Trout streams were visually inventoried for all habitat types (see classifications below) and then a subset of habitat units of each type were selected for Apache Trout abundance surveys. First, each stream was divided into distinct reaches based on Rosgen stream channel types using a broad geomorphic characterization using topographic maps and other information (Rosgen 1994; Hartsell 2004). Rosgen channel types are based on channel entrenchment, channel pattern (single thread, multiple thread, anastomosed), channel slope, and channel shape (wide shallow vs. narrow deep) into classes: Aa+, A, B, C, D, E, F and G (Rosgen 1996). For example, Aa+ stream channels are very steep, deeply entrenched, torrent streams, whereas E streams are low gradient, meandering, riffle/pool streams with narrow and deep channels. Within each channel type all habitat units were classified as: Cascade, Riffle, Run, Pool or Complex (two habitat types occurring side-by-side); habitat types classified as Headwater during some surveys were reclassified as Riffle (J. Johnson, FWS, pers. comm.). Habitat units were classified based on surface turbulence, depth, and gradient (Armantrout 1998; Hartsell 2004). Every fifth habitat unit of each type was measured for average wetted width using three measurements. The length of all habitat units was also measured.

After habitat inventories, the abundance of adult Apache Trout (>130-mm TL) was estimated at a subset of inventoried habitats representing each type. Twenty percent of all inventoried habitat units were selected for electrofishing surveys. Two thirds of this subset were selected for multiple-pass electrofishing and the remaining one third were selected for single-pass electrofishing. Each selected unit was isolated with block nets and Apache Trout were sampling using multiple- or single-pass electrofishing. Electrofishing was conducted with a Smith-Root backpack electrofisher with one anode and one or two netters. All fish were measured for total length.

For habitat units sampled with multiple-pass electrofishing, the number of Apache Trout >130mm TL per pass was used to estimate adult Apache Trout abundance within each habitat unit using a Zippin depletion estimator used in FSA package of Program R (R Core Team 2015; Ogle 2017). For habitat units sampled with single-pass electrofishing, the number Apache Trout collected was divided by the mean capture efficiency for that stream and habitat type as estimated by the Zippin estimator from habitat units sampled with multiple-pass electrofishing.

A linear mixed-effects model was used to assess general differences in adult Apache Trout density (adult Apache Trout / 100-m) among habitat types and channel types. The response variable for the model was $\log_{10}$(Apache Trout / 100m +1), and both habitat type and channel type were included as covariates. Pool was used as the baseline habitat type, and Aa+ was used as the baseline channel type. Stream was included as a random effect to account for different Apache Trout densities among streams and make habitat type and channel type comparisons relative to within streams. Data from each year per stream was treated as an independent sample per stream. Candidate models included both habitat type and channel type together, models with either habitat type or channel type only, and also an intercept-only model (no habitat type or channel type effect). Candidate models were compared using Akaike's Information Criterion for small sample sizes (AIC_c), and Akaike weights (w_i) were used to assess model plausibility (Burnham and Anderson 2002). If model selection suggested differences in habitat type and channel type main effects, then Tukey contrasts were used to assess differences in densities among different levels of the main effects. Significance was assessed at $\alpha = 0.05$.

In addition to evaluating all streams together in a single analysis (the most powerful analysis), differences in means and variances in adult Apache Trout densities were evaluated independently by stream using separate analysis of variances. Again, the response variable was the adult Apache Trout density ($\log_{10}$(Apache Trout / 100m +1)). ANOVA was used to assess differences in mean adult (>130-mm TL) Apache Trout density by habitat type and Rosgen channel type. Separate ANOVAs were completed for each factor (habitat type or channel type) because not all habitat types occurred in all Rosgen channel types. A Fligner Test was used to assess differences in variances of adult Apache Trout density by habitat type and Rosgen channel type. Again, separate tests were completed for each factor (habitat type, channel type), and significance was assessed at $\alpha = 0.05$.

Results

Eighteen Apache Trout streams were sampled between the two time periods (2001-02, and 2012-13), and eight streams were sampled during both periods. Between both sampling periods (2001-02 and 2011-12), 2551 habitat units were sampled for Apache Trout. Most habitat units were sampled in Rosgen channel type A, followed by B channels (Table 1A). Likewise, riffles were the habitat type sampled most often, followed by pools (Table 1A).

Multiple age classes of Apache Trout were present in nearly all streams and years sampled, except for some streams where few fish were collected such as in Coyote Creek in 2001 and Firebox Creek in 2012 (Figure 7A). Adult Apache Trout densities ranged from 0.0 to 619 / 100-m across all streams and years, and were highly variable within each stream and year (Figure 8A; Figure 9A). The maximum density observed (619 per 100-m) was in Ord Creek in 2001 in a Complex habitat type in an E channel type. Estimated detection probabilities were often at or near 1.0, and were almost always above 0.5 (Figure 10A); the minimum observed was 0.27, and the 25th percentile was 0.75.

Model selection showed the linear mixed effects model with both Rosgen channel type and habitat type predictor variables to be the only plausible model illustrating that Apache Trout densities differed between at least some channel types and habitat types (Table 2A). Tukey contrasts showed adult Apache Trout densities to only be higher in Rosgen A channel types versus B channel types, but no significant differences between any other channel types (Table 3A; Figure 11A). Tukey contrasts also showed pools to have higher densities than all other habitat types ($P \leq 0.05$), and densities were generally lowest in riffles ($P \leq 0.05$; Table 4A; Figure 11A).

The ANOVAs completed individually for each population and year showed mean Apache Trout densities (>130-mm TL) to differ significantly ($P \leq 0.05$) among habitat types for 13 of 26 unique stream years (Table 5A). Fligner tests showed variances in densities to differ significantly among habitat types at 10 of 26 unique stream years (Table 5A). Densities also differed significantly ($P \leq 0.05$) among Rosgen channel types for 5 of 14 unique stream years where there was more than one channel type (12 of 26 unique stream-years only had one channel type; Table 5A). Fligner tests showed variance in densities to differ among Rosgen stream types in 4 of 14 unique stream years (Table 5A). Generally, significant differences in means and variances in Apache Trout densities among habitat types and Rosgen channel types were attributable to Apache Trout rarely occupying certain habitat and channel types, although the unoccupied types were not consistent between streams and years (Figure 8A; Figure 9A).

Discussion

As mentioned above, stratification during monitoring can be useful to ensure certain habitat types are sampled during monitoring, and stratification can be used to allocate sampling effort to manage variances. First, since the monitoring plan is centered around sampling standard reach lengths (e.g., 100-m reaches) and not individual channel units, stratification on habitat types is not an option despite differences in Apache Trout densities being significantly different among habitat types. Complementary habitat surveys will, however, be conducted in 50, 100, or 200-m reaches to monitor the percent of reach that is pool habitat – the habitat type where Apache Trout densities were highest. Second, Apache Trout densities were significantly higher in A versus B Rosgen channel types only. Since Apache Trout monitoring will occur systematically within each population, all channel types will be represented in monitoring unless they are rare; channel types are typically not unique to short stream segments (Rosgen 1996). Also, significant differences in the variance of Apache Trout densities among channel types was only observed in 3 of 14 streams (Crooked Creek [2002], Deep Creek [2002], and Ord Creek [2001, 2013]) with multiple channel types, and this appeared to largely result from a lack of occupancy of certain channel types and was inconsistent across years in Ord Creek (the only stream with differences in variances sample during two time periods). Therefore, it was decided not to use Rosgen channel type as a stratum in a stratified sampling design, and stratification in general is not warranted.

Table 1A: Count of habitat types by Rosgen channel type (depletion only sites) across 18 Apache Trout populations (26 population-years) sampled using BVET methods.

Rosgen	Habitat Type					
Channel Type	Pool	Cascade	Complex	Riffle	Run	Total
Aa+	62	46	53	85	31	277
A	462	120	204	624	256	1666
B	175	21	64	201	104	565
C	5	0	7	7	6	25
E	6	1	5	6	0	18
Total	710	188	333	923	397	2551

Table 2A: Model degrees of freedom (df), log-likelihoods, AIC_c, ΔAIC_c, and Akaike weights (w_i) for candidate models predicting Apache Trout density ($\log_{10}$[adult Apache Trout / 100m + 1]). Stream was included as a random effect to account for stream-level (population) differences in Apache Trout abundance.

Model	df	Log-likelihood	AIC_c	ΔAIC_c	w_i
Channel type + Habitat type	11	-2307.99	4638.1	0.00	1.00
Habitat type	7	-2320.66	4655.4	17.27	0.00
Channel type	7	-2397.76	4809.6	171.48	0.00
Intercept only	3	-2408.32	4822.7	184.57	0.00

Table 3A: Tukey contrasts of Rosgen channel type effects on adult (>130-mm TL) Apache Trout density.

Contrast	b_i	SE(b_i)	*P*-value
A – Aa+	0.098	0.048	0.203
B – Aa+	-0.077	0.055	0.582
C – Aa+	0.163	0.131	0.686
E – Aa+	0.220	0.153	0.560
B – A	-0.175	0.037	<0.001
C – A	0.065	0.125	0.982
E – A	0.122	0.146	0.906
C – B	0.240	0.127	0.280
E – B	0.297	0.151	0.244
E – C	0.056	0.192	0.998

Table 4A: Tukey contrasts of habitat type effects on adult (>130-mm TL) Apache Trout density.

Contrast	b_i	SE(b_i)	*P*-value
Cascade - Pool	-0.308	0.049	<0.001
Complex - Pool	-0.172	0.040	<0.001
Riffle - Pool	-0.392	0.030	<0.001
Run - Pool	-0.151	0.038	<0.001
Complex - Cascade	0.136	0.055	0.090
Riffle - Cascade	-0.084	0.048	0.387
Run - Cascade	0.156	0.054	0.028
Riffle - Complex	-0.220	0.038	<0.001
Run - Complex	0.021	0.044	0.990
Run - Riffle	0.241	0.036	<0.001

Table 5A: *P*-values from ANOVAs (testing for equal means) and Fligner tests (testing for equal variances) in adult (>130-mm TL) Apache Trout density by habitat type and Rosgen stream type for each stream and year. Analyses were not conducted when only one Rosgen channel type was present (--).

	Habitat type		Rosgen channel type	
Stream (Year)	ANOVA *P*-value	Fligner *P*-value	ANOVA *P*-value	Fligner *P*-value
Big Bonito (2001)	0.38	0.24	0.33	0.40
Boggy/Lofer (2002)	0.13	0.20	0.22	0.26
Boggy/Lofer (2013)	<0.01	<0.01	0.80	0.79
Coyote (2001)	0.33	0.32	--	--
Coyote (2012)	<0.01	<0.01	--	--
Crooked (2002)	0.40	0.38	0.01	0.01
Crooked (2012)	0.02	0.03	0.09	0.09
Deep (2002)	0.05	0.01	<0.01	<0.01
East Fork White (2002)	<0.01	0.29	--	--
Elk Canyon (2002)	0.01	<0.01	--	--
Elk Canyon (2012)	0.01	<0.01	--	--
Firebox (2012)	0.26	0.32	0.43	0.45
Flash (2013)	0.27	0.26	--	--
Little Bonito (2001)	0.06	0.08	0.94	0.69
Little Bonito (2013)	<0.01	<0.01	0.54	0.45
Marshall Butte (2012)	<0.01	<0.01	--	--
Ord (2001)	0.27	<0.38	<0.01	<0.01
Ord (2013)	0.01	0.27	<0.01	<0.01
Paradise (2012)	<0.01	<0.01	0.64	0.57
Smith (2002)	0.79	0.71	--	--
Smith (2012)	0.41	0.16	--	--
Soldier Springs (2002)	0.24	0.33	--	--
Soldier Springs (2012)	0.14	0.20	--	--
Thompson (ALL) (2002)	0.31	0.11	0.26	0.20
West Fork Black (2003)	0.01	0.04	<0.01	0.47
Wohlenberg (2013)	0.30	0.22	--	--

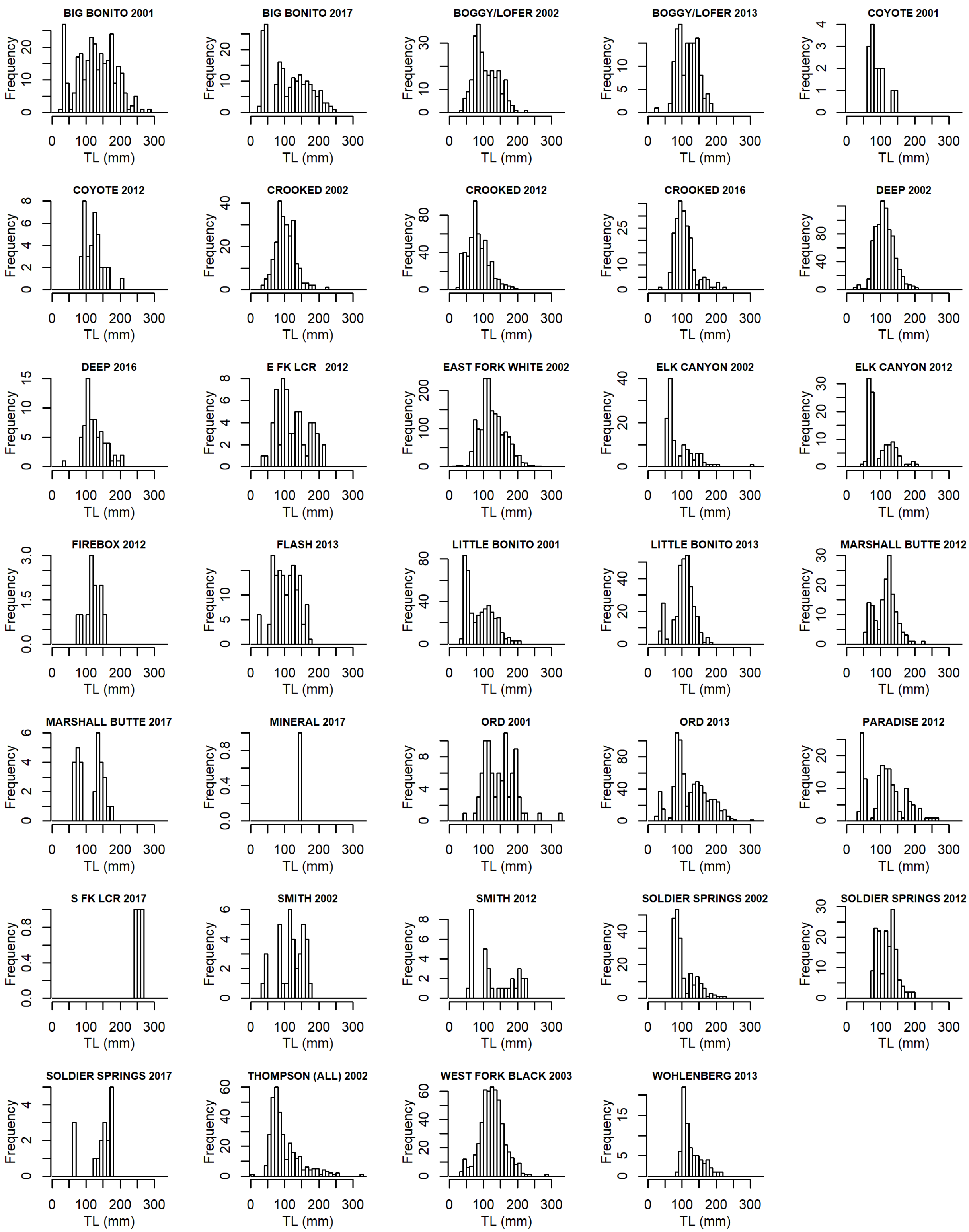

Figure 7A: Length frequency histograms for Apache Trout by population and year.

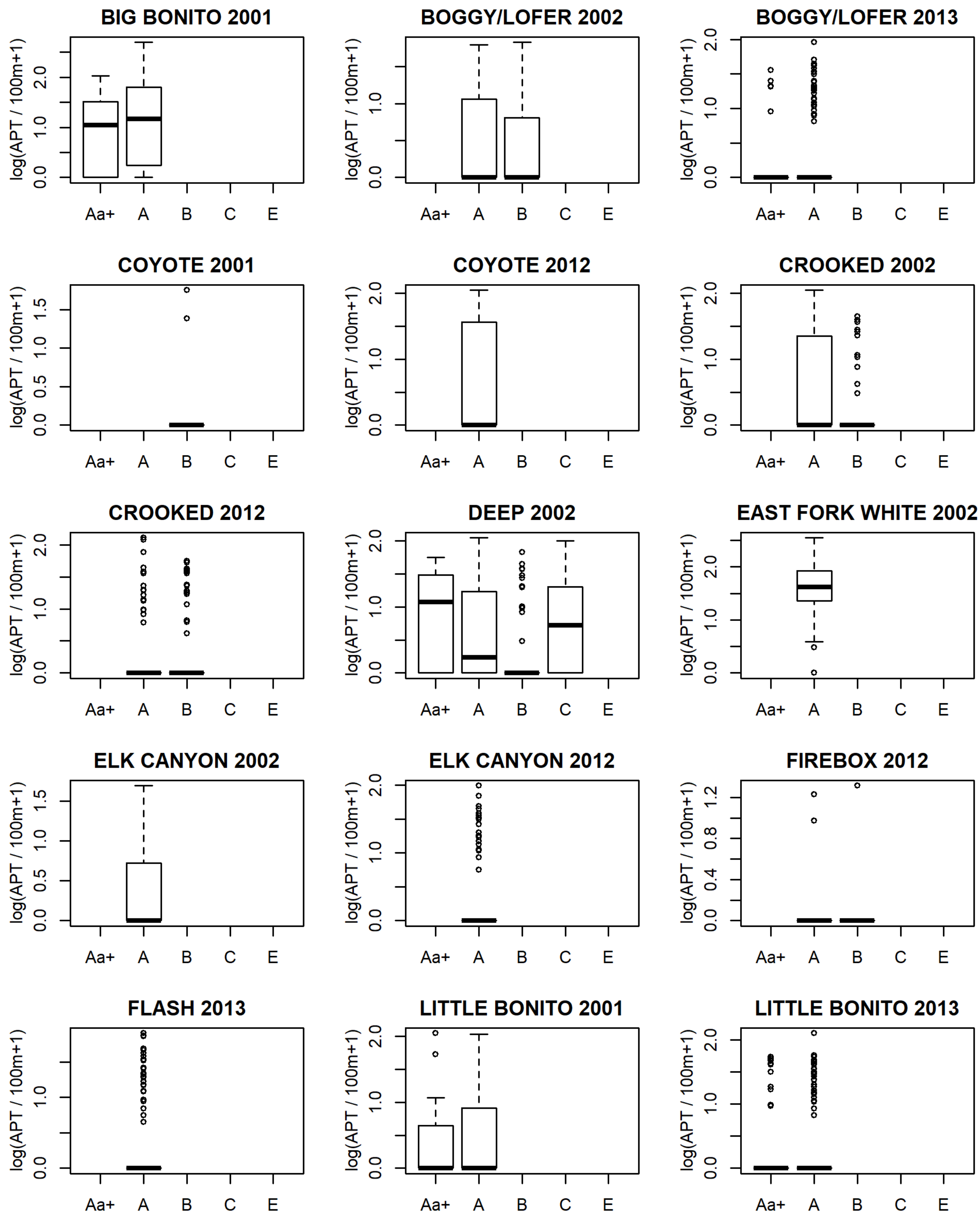

Figure 8A: Box plots of adult (>130-mm TL) Apache Trout density ($\log_{10}$[Apache Trout / 100-m +1]) by Rosgen channel type (Aa+, A, B, C, E) by population by year.

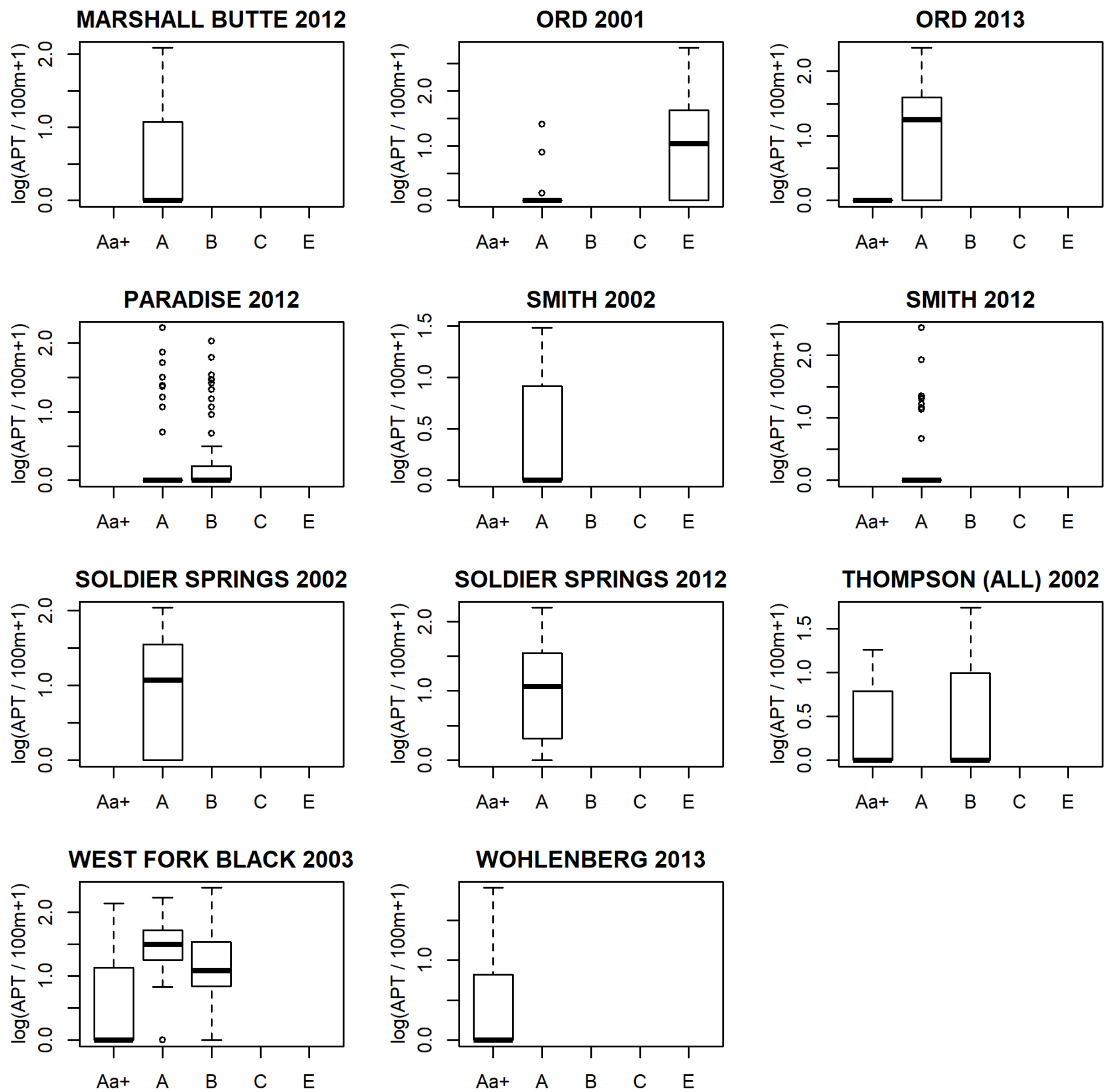

Figure 8A. Continued.

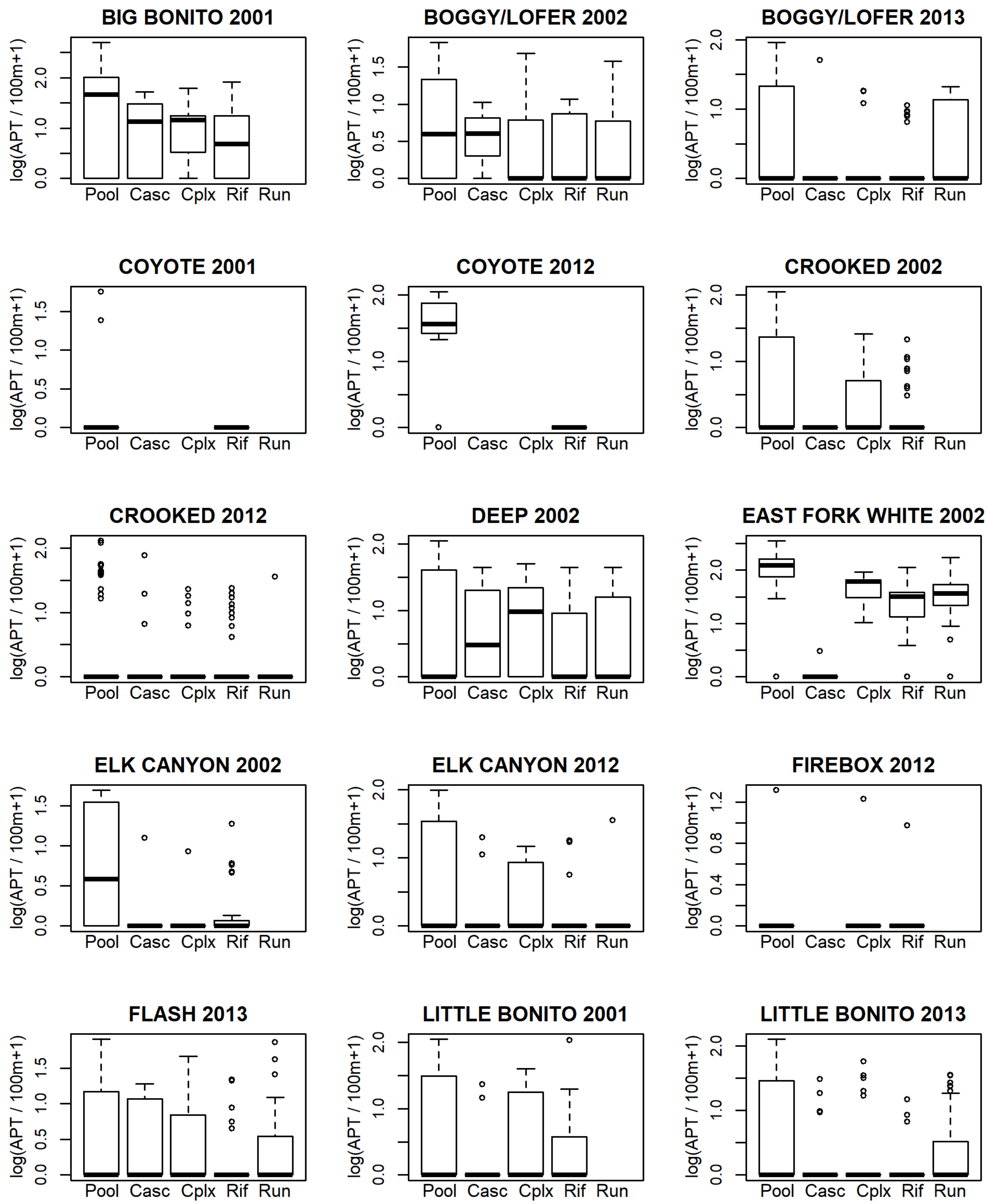

Figure 9A: Box plots of adult (>130-mm TL) Apache Trout density (log$_{10}$[Apache Trout / 100-m +1]) by habitat type (Pool, Cascade, Complex, Riffle, Run) by population by year.

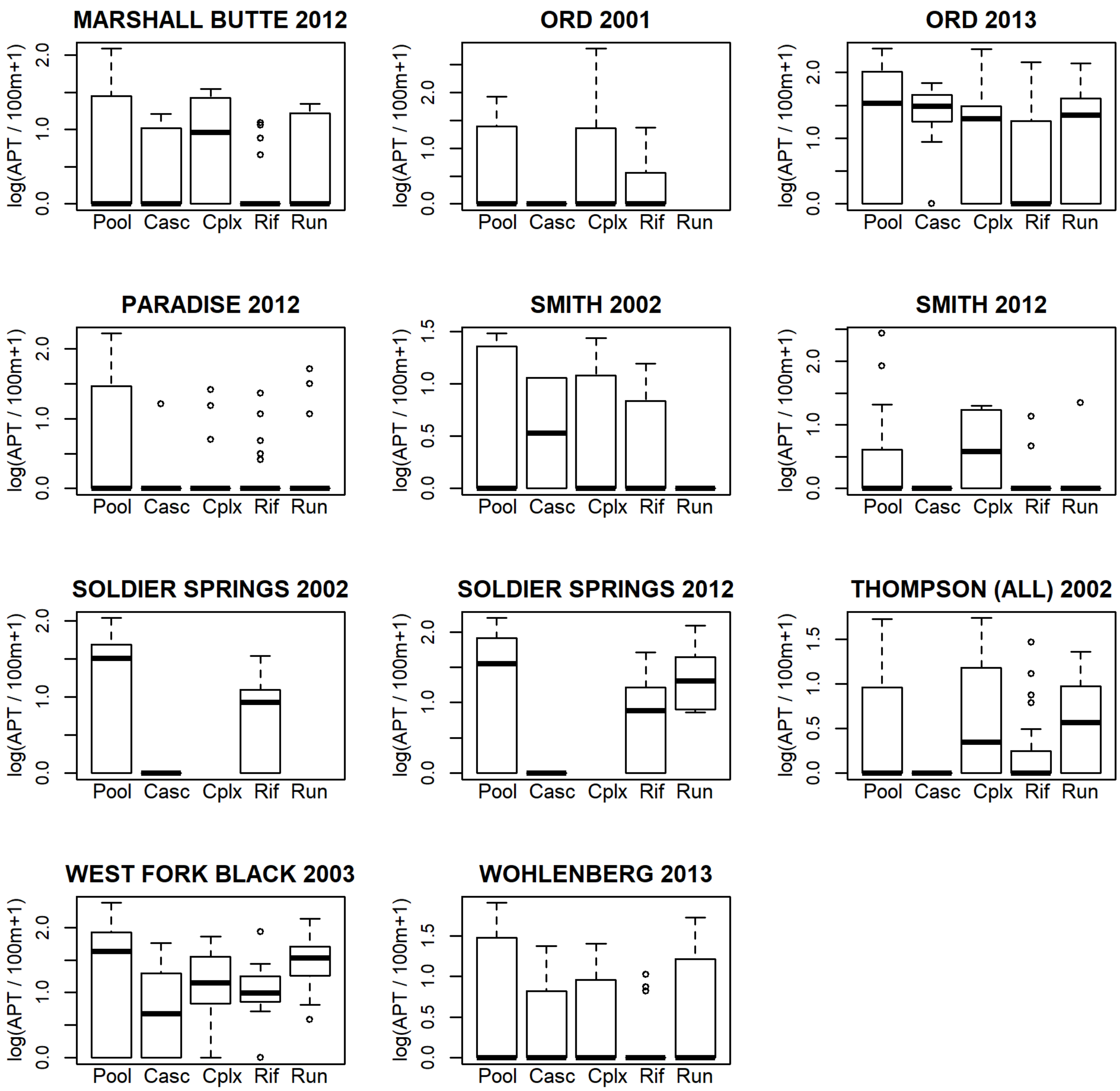

Figure 9A: Continued.

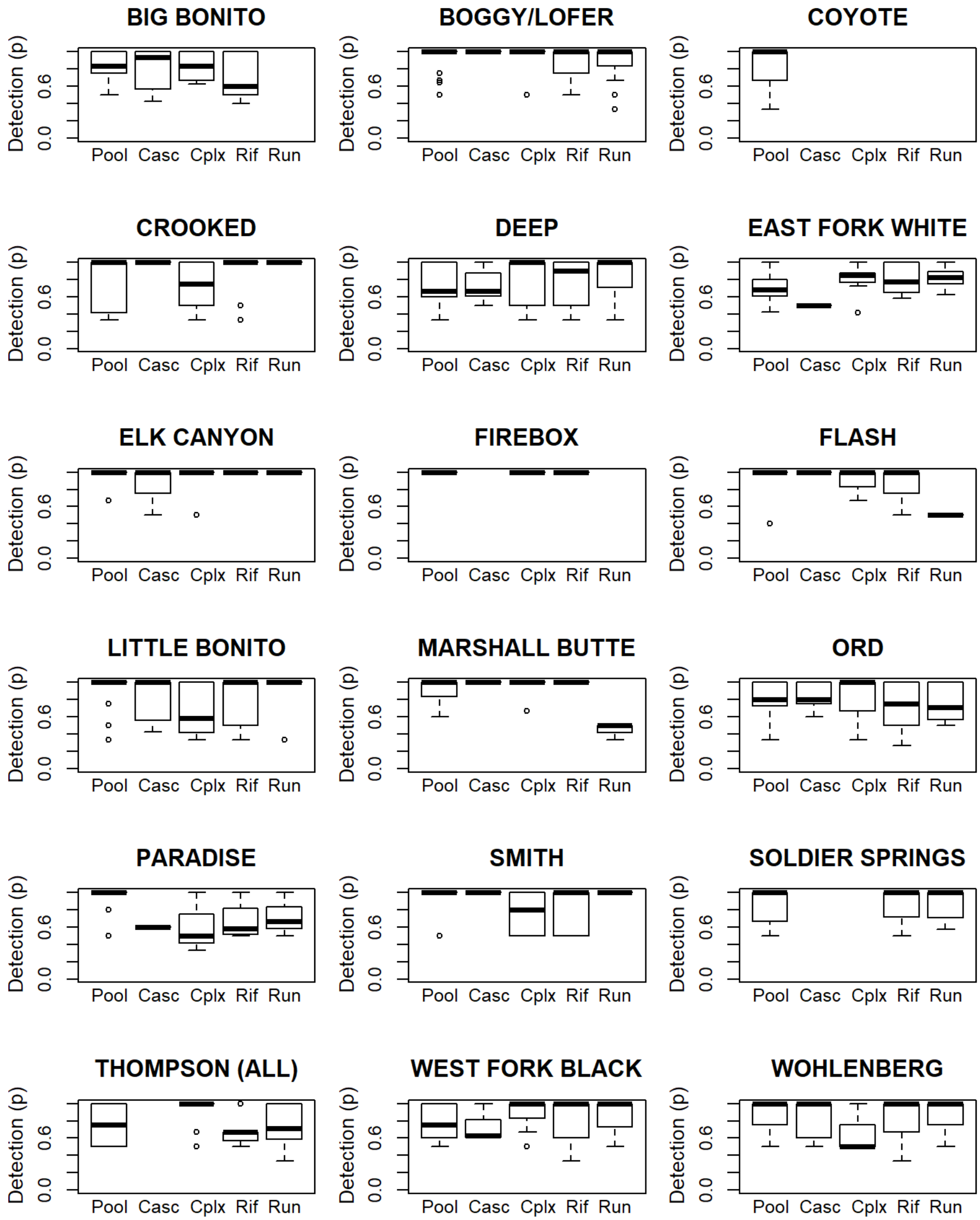

Figure 10A: Box plots of estimated detection probability (*p*) of adult (>130-mm TL) Apache Trout by habitat type (Pool, Cascade, Complex, Riffle, Run) by population. Data from FWS BVET surveys where depletion electrofishing was conducted.

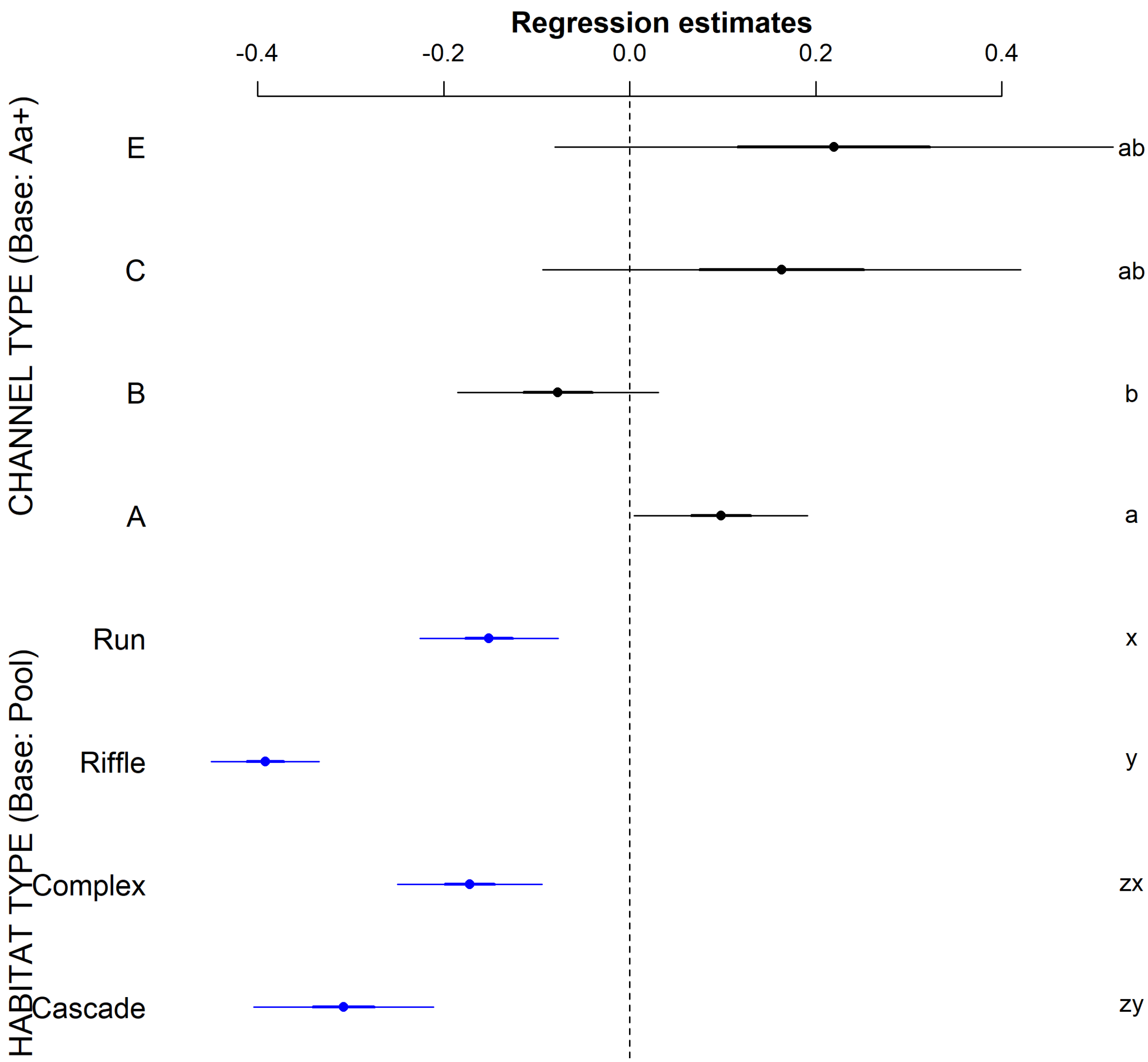

Figure 11A: Coefficient plot showing regression coefficients (± 2SE) from linear mixed model predicting Apache Trout density (log_{10}[adult Apache Trout / 100m + 1]) as a function of habitat type and Rosgen channel type with stream as a random effect. Channel types and habitat types with different letters were significantly different based on a Tukey contrast at *P* < 0.05. No channel types were significantly different from the baseline type: Aa+ (*P*<0.05). All habitat types were significantly different from the baseline type: Pool (*P*<0.05).

Appendix B. Investigating the Precision of N

Monitoring of stream salmonid populations requires a sampling framework that allows stream survey data to be extrapolated to the entire length of occupied habitat (Scheaffer et al. 2012). This is typically done by sampling multiple sites (reaches) on a stream that are selected in a way that allows for strong inferences (i.e., valid generalizations) to be made regarding population parameters. Random or systematic sampling are common ways to select reaches for trout population surveys (Cook et al. 2010). For example, the average linear density of trout per site (# / km) is often extrapolated to the entire habitat extent (km) to obtain a total population size ($\hat{\boldsymbol{N}}$). The precision of this estimate, expressed as a variance or confidence intervals, is dependent on several factors: 1) the unit-to-unit (reach-to-reach) variance in trout densities observed in the data, and 2) the number of sites surveyed (sample size). In finite population sampling, the variance is adjusted by the fraction of the total population sampled, termed the finite population correction (fpc). The fpc effectively reduces the variance by the proportion of habitat that was actually sampled and, therefore, there is no uncertainty. For example, in 2-km stream there are 20 possible 0.1-km reaches available to be selected for sampling. If 10 are selected for sampling, then 50% of the available habitat will be sampled (N – n) / N = (20 – 10) / 20 = 0.5. If expressed as a unit of stream length: $(L - \sum l_i)$ / L = (2km – (10 x 0.1km)) / 2km = 0.5, or 50%. If you sampled the entire stream the only uncertainty would come from enumeration methods (removal estimator).

The primary objective of Apache Trout monitoring as stated in this plan is to estimate the abundance of adult Apache Trout (>130-mm TL) with an 80% confidence interval that is within 40% of the abundance estimate (see 2. Goals and Objectives), that is, the target level of precision is proportional to estimated abundance ($\hat{N}$)(see Figure 2). Since precision can be controlled by monitoring a certain number of sites given the site-to-site variance in trout densities typically observed, one can use past monitoring data to assess the number of sites needed to meet the precision levels stated in the monitoring goals and objectives.

Objectives

1. Evaluate how the number of sites sampled and proportion of total stream length sampled influence the relative precision of stream-wide estimates of adult Apache Trout abundance during historical monitoring.
2. Evaluate if the relative precision of stream-wide estimates of adult Apache Trout abundance is influenced by reach length sampled during historical monitoring.

Methods

We used past Apache Trout monitoring data to evaluate the effects of sample size (number of sample reaches), cumulative reach length, and average reach length on the relative precision of stream-wide estimates of Apache Trout abundance. Briefly, Apache Trout have been monitored in the past using Basinwide Visual Estimation Technique (BVET) surveys in 18 streams in during two primary time periods, 2001-02 and 2012-13, resulting in 26 unique stream-year combinations (described in Appendix A. Evaluating a Need for Stratified Sampling). All individual habitats (channel units or mesohabitats) were inventoried per stream. Then the abundance of adult Apache Trout (>130-mm TL) was estimated at 20% of all inventoried habitat units using electrofishing surveys; two thirds of the 20% were sampled using multiple-pass electrofishing, and the remaining third were sampled using single-pass electrofishing. Abundance of adult Apache Trout were estimated with a Zippin removal estimator when multiple pass electrofishing was conducted, or by dividing the number of fish caught during a single electrofishing pass by the mean detection probability for that habitat type in each stream. Additional data were collected from five streams in 2016-17 according to this plan, where systematic sampling was used to select the location of 100-m sites where three-pass electrofishing was conducted; abundance estimates were made at each site again using the Zippin estimator.

Streamwide estimates of abundance for each stream and year were computed by multiplying the mean abundance of Apache Trout across all sample units (reaches) by the number of sample units available in the sampling frame (i.e., N = habitat extent / mean reach length). Variances were computed for a systematic sample using the number of sampling units in the sampling frame, unit-to-unit variance, and the finite population correction described above (section 3c. Computing the Monitoring Metrics) and in Scheaffer et al. (2012); a ratio estimator of population total and variance was explored to account for variable site lengths but it did not improve precision due to the surprising lack of association between site length and Apache Trout abundance (Scheaffer et al. 2012). The variance in stream-wide estimates was used to compute 80% confidence intervals. The relative precision of $\widehat{N}$ was computed by dividing the 80% confidence interval by the stream-wide estimate of abundance (100 x 80% CI / $\widehat{N}$). Spearman's rank correlations were used to assess the correlation between relative precision of $\widehat{N}$ and number of reaches sampled, relative precision of $\widehat{N}$ and the proportion of stream sampled (across all reaches), and relative precision of $\widehat{N}$ and average reach length. Significance of correlations was assessed at α = 0.05.

Data from past Apache Trout monitoring efforts were also used to understand sample size requirements (and % of habitat sampled) needed to achieve the abundance precision objective (100 x 80% CI / $\widehat{N}$ ≤ 40%) on a stream-by-stream basis. To do so, a sample size estimator was used (eq. 4.11 in Scheaffer et al. 2012):

$$n = \frac{N \cdot s^2}{(N-1)D + s^2}$$

where *n* is the sample size needed to meet abundance precision objectives based on the number of sampling units available in the sampling frame (*N*), s^2 is the variance in abundance from sample unit to sample unit (reach to reach), and *D* is:

$$D = \frac{\left(\widehat{N} \cdot \frac{0.40}{2}\right)^2}{\left(t_{\alpha=0.2/2,\mathrm{n}-1}\right)^2}$$

where $\widehat{N}$ is the streamwide estimate of abundance, and *t* is the *t*-value from a *t*-distribution table at α = 0.2/2 = 0.1 and n_i-1 degrees of freedom; α is divided by 2 for each side of the confidence interval to match the total α = 0.2 for an 80% confidence interval. Note that α = 0.2 (80% confidence interval) to match the level of precision stated in monitoring Goal 1, Objective 1. Also note that $\widehat{N}$ is multiplied by the target level of precision, that is an 80% CI that is 40% of $\widehat{N}$, but divided by 2 to halve the CI width so that it is represented as being within a certain bound (or distance) of $\widehat{N}$ as per Scheaffer et al. (2012). Since the sample size needed (*n*) is based both on s^2 and the finite population correction (1-*n*/*N*, but rearranged above), the sample size needed is also reported as the percent of sampling frame (percent of habitat extent) needed. This helps overcome the fact that past BVET monitoring data were often collected within much shorter sample units (reaches) than monitoring data collected more recently in 2016-17 as encouraged in this plan and any uncertainty associated with how s^2 is related to sample unit (reach) length (see Results).

The precision of stream-wide estimates of adult Apache Trout abundance was also evaluated on a stream-by-stream basis using bootstrapping. To obtain boot-strapped estimates, sites for each population were randomly selected, with replacement, to obtain samples of sites with samples sizes of: 3, 4, 5, 10, 15, 20, 25, 30, 35, 40, 50, 60, 70, 80, 90, and 100 sites. Each sample size was bootstrapped 1000 times. For each sample of sites, stream-wide estimates of abundance were computed (with 80% CI), as was the proportion of total habitat sampled.

Results and Discussion

Fish survey data summarized at the sample unit (reach) level is reported in Appendix A. For BVET sampling, the mean reach length across all streams and years averaged 12.6 m (range: 6.4 to 31.0-m). The shortest habitat extent sampled and used for extrapolation was 0.6-km in Soldier Springs in 2002, whereas Boggy/Lofer Creeks was 15.4-km (Table 6B). Estimates of $\widehat{N}$ ranged from 6 adult Apache Trout (>130-mm TL) in Mineral Creek in 2017 up to 4,439 Apache Trout in the East Fork White River in 2002 (Figure 12B). The 80% confidence interval range from 6 individuals in Smith Creek in 2012, up to 863 individuals in East Fork White River in 2002. Relative precision (80% CI / $\widehat{N}$) ranged from 19.4% in East Fork White River in 2002 up to 254% in Mineral Creek in 2017 (one Apache Trout collected in one reach).

Spearman rank correlations revealed that the relative precision of $\widehat{N}$ was significantly negatively correlated with both the number of reaches sampled (top panels of Figure 13B) and the proportion of habitat sampled (lower panels of Figure 13B) as expected. The significant negative correlations between relative precision of $\widehat{N}$ and number of reaches and proportion of habitat sampled appeared to be driven, in part, by the imprecise estimates of $\widehat{N}$ due to very low number of Apache Trout that were collected in a small number of reaches in Coyote (2001), Firebox (2012), Mineral (2017), South Fork Little Colorado (2017), and others (see Figure 8A and Figure 9A). The SD in abundance across reaches was surprisingly positively correlated with average reach length (Figure 14B). However, precision of N was not correlated with mean reach length (Figure 14B), suggesting that streamwide estimates of abundance were not influenced by the length of reaches surveyed. In general, approximately 30% of past monitoring has met the objective of having 80% confidence intervals that are 40% or less of the stream-wide estimate of abundance (Table 6B). Most of these past efforts, regardless of whether they were BVET or systematic (per this plan) sampling designs, targeted sampling 20% of the habitat extent occupied by the population. All of the very imprecise stream-wide estimates were due to Apache Trout occupying one or a few sites in the entire streams. It seems reasonable that continuing to sample 20 to 30% of available habitat should be the target level of sampling effort (number of sites or sampling units) during future monitoring except when stream-specific data suggest less sampling may meet precision goals (Table 6B; see below).

The sample sizes needed to reach precision objectives for past monitoring years ranged from 7 (Big Bonito 2017) to 305 (Boggy/Lofer 2013), the former sampled using systematic sampling at 100-m reaches and the latter sampled using the past BVET protocol (Table 6B). When sample sizes needed were converted to percent of habitat that needs sampling, percentage of habitat needed ranged from 5% (East Fork White 2002) to 89% (Mineral 2017 and South Fork Little Colorado R. 2017). Streams requiring a high percentage of habitat to be sampled only collected Apache Trout at one site during that monitoring year, which drastically inflates s^2 and creates considerable uncertainty in streamwide estimates of abundance.

Bootstrapped estimates of adult Apache Trout abundance showed lower relative precision (i.e., better precision) of stream-wide estimates as a higher proportion of the occupied habitat extent was sampled as expected (Figure 16B). The precision of stream-wide abundance estimates rarely reached 20% relative precision when Apache Trout were only collected at a few sites (e.g., Coyote 2001; Firebox 2012). In some cases, bootstrapped estimates suggested that sampling less than 20% of habitat during sampling may yield stream-wide estimates of abundance that meet precision goals. Examples are: in Deep Creek 2002 (7%), East Fork White River 2002 (5%), Ord Creek 2013 (9%), and West Fork Black River 2003 (10%) (Figure 16B).

Table 6B. Survey characteristics and stream-wide estimates of Apache Trout abundance ($\widehat{N}$; >130-mm TL) per population. Number of sites (n), mean site length (m), mean abundance per site, SD abundance per site, habitat extent (km), and proportion of extent noted per survey. Stream-wide estimates of Apache trout abundance, upper and lower 80% confidence limits, and relative precision of 80% confidence interval (%) given per survey. Estimates of number of sites (Required n) and proportion of habitat (Required %) required to achieve monitoring precision goals also reported. Note that negative lower 80% confidence limits were retained for computational clarity; often they will be replaced by observed catch.

Population	Year	Sites (n)	Site Length (m)	Mean No.	SD No.	Extent (km)	Prop. Extent	Streamwide N	Lower 80% CL	Upper 80% CL	80%CI/N (%)	Req. n	Req. %
BIG BONITO	2001	48	31	3.1	4.9	4.1	0.36	415.6	318	513.3	47	59	44
BIG BONITO	2017	5	100	15.8	6.42	3	0.17	474	358	590	49	7	23
BOGGY/LOFER	2002	99	19.6	0.7	0.99	7.7	0.25	256.7	213.3	300.2	33.8	77	20
BOGGY/LOFER	2013	189	10.9	0.3	0.78	15.4	0.13	360.6	264.4	456.9	53.4	305	21
COYOTE	2001	31	6.9	0.1	0.25	1.6	0.13	15	2.3	27.7	169.5	171	74
COYOTE	2012	20	12.1	0.6	0.75	2.9	0.08	145.8	93.8	197.8	71.3	54	22
CROOKED	2002	108	11.6	0.3	0.69	10.3	0.12	287.5	215.8	359.1	49.9	157	18
CROOKED	2012	224	8	0.2	0.55	9.6	0.19	242.3	190.7	293.9	42.6	248	21
CROOKED	2016	10	100	3.1	3.14	5.8	0.17	179.8	107.8	251.8	80	27	47
DEEP	2002	249	11.3	0.8	1.25	14	0.2	1020.9	908.2	1133.5	22.1	88	7
DEEP	2016	10	100	2.5	2.22	14	0.07	350	219.8	480.2	74.4	30	21
EAST FORK WHITE	2002	94	17.9	8.2	6.55	9.7	0.17	4438.8	4007.3	4870.4	19.4	26	5
ELK CANYON	2002	52	22.8	0.4	0.72	4.6	0.26	85.5	62.8	108.1	53	77	38
ELK CANYON	2012	87	8	0.3	0.63	3.9	0.18	139.1	101	177.1	54.7	141	29
FIREBOX	2012	57	10.5	0.1	0.23	4.6	0.13	23.2	7.3	39	137.2	280	64
FLASH	2013	117	7.7	0.4	0.98	5.2	0.17	251.6	180.6	322.6	56.4	199	30
LITTLE BONITO	2001	131	30.6	0.7	1.14	14.2	0.28	307.5	257.1	357.8	32.7	97	21
LITTLE BONITO	2013	283	6.4	0.2	0.52	11.7	0.16	354.5	287.9	421.1	37.6	254	14
MARSHALL BUTTE	2012	92	9.2	0.5	0.93	4.7	0.18	249.9	191.9	308	46.4	117	23
MARSHALL BUTTE	2017	7	100	2.1	3.67	3.6	0.2	77.1	13.7	140.6	164.5	29	81
MINERAL	2017	8	100	0.1	0.35	4.6	0.17	5.8	-1.5	13	253.9	41	89
ORD	2001	48	21	1.1	2.78	5.8	0.17	309.4	179.1	439.7	84.2	133	48
ORD	2013	143	9.9	2.4	3.16	7.9	0.18	1872.8	1629.8	2115.8	26	67	8
PARADISE	2012	108	13.7	0.5	1.58	6.7	0.22	248	163.8	332.2	67.9	220	45
S FK LCR	2017	18	100	0.2	0.71	9.4	0.19	15.7	-3.1	34.4	239.3	84	89
SMITH	2002	33	9.4	0.5	0.83	0.5	0.66	22.7	17.2	28.3	48.6	37	74
SMITH	2012	43	8.7	0.3	0.75	0.4	0.83	16.9	13.7	20.1	37.9	42	81
SOLDIER SPRINGS	2002	20	12.7	2.2	3.06	0.6	0.43	105.8	73.5	138	61.1	30	64
SOLDIER SPRINGS	2012	31	12.4	2	1.97	0.6	0.67	90.5	78.3	102.7	27	23	50
SOLDIER SPRINGS	2017	5	100	2.6	2.88	1.9	0.26	49.4	18.4	80.4	125.6	15	79
THOMPSON (ALL)	2002	84	15.9	0.8	1.4	6.7	0.2	340.8	266.4	415.2	43.7	97	23
WEST FORK BLACK	2003	85	16.6	2.8	3.08	8	0.18	1352.4	1163.4	1541.4	28	46	10
WOHLENBERG	2013	75	7.9	0.3	0.65	2	0.3	86.3	66.2	106.4	46.6	92	37

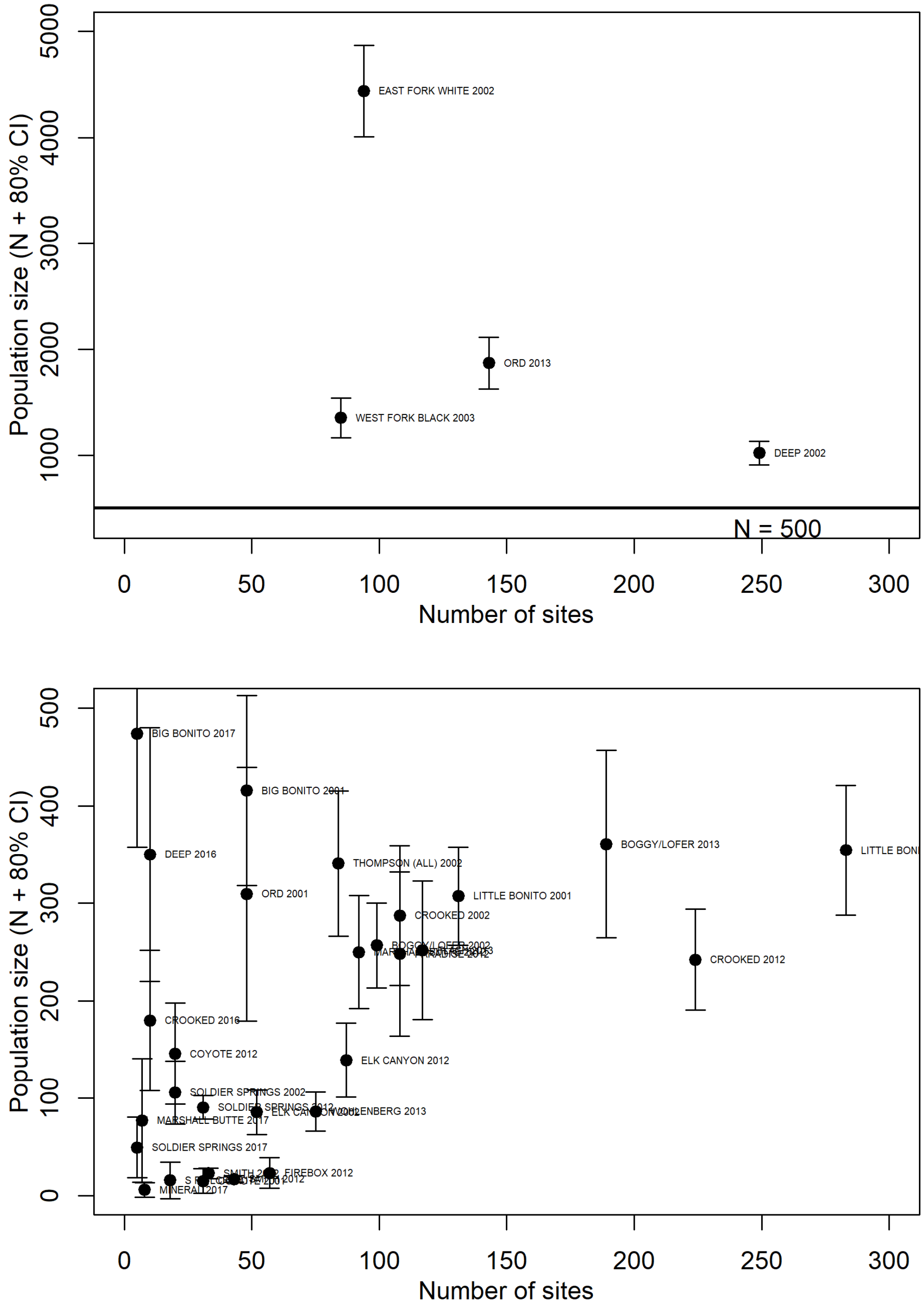

Figure 12B: Population size ($\widehat{N}$) versus number of sites sampled for adult (>130-mm TL) Apache Trout populations surveyed in different years. Top panel shows populations with $\widehat{N}$ greater than 500 individuals. Bottom panel shows populations with $\widehat{N}$ less than 500 individuals. Data are from U.S. Fish and Wildlife Service Basin Visual Estimation Technique (BVET) surveys, except surveys in 2016 and 2017 where systematic sampling at 100-m sites was conducted.

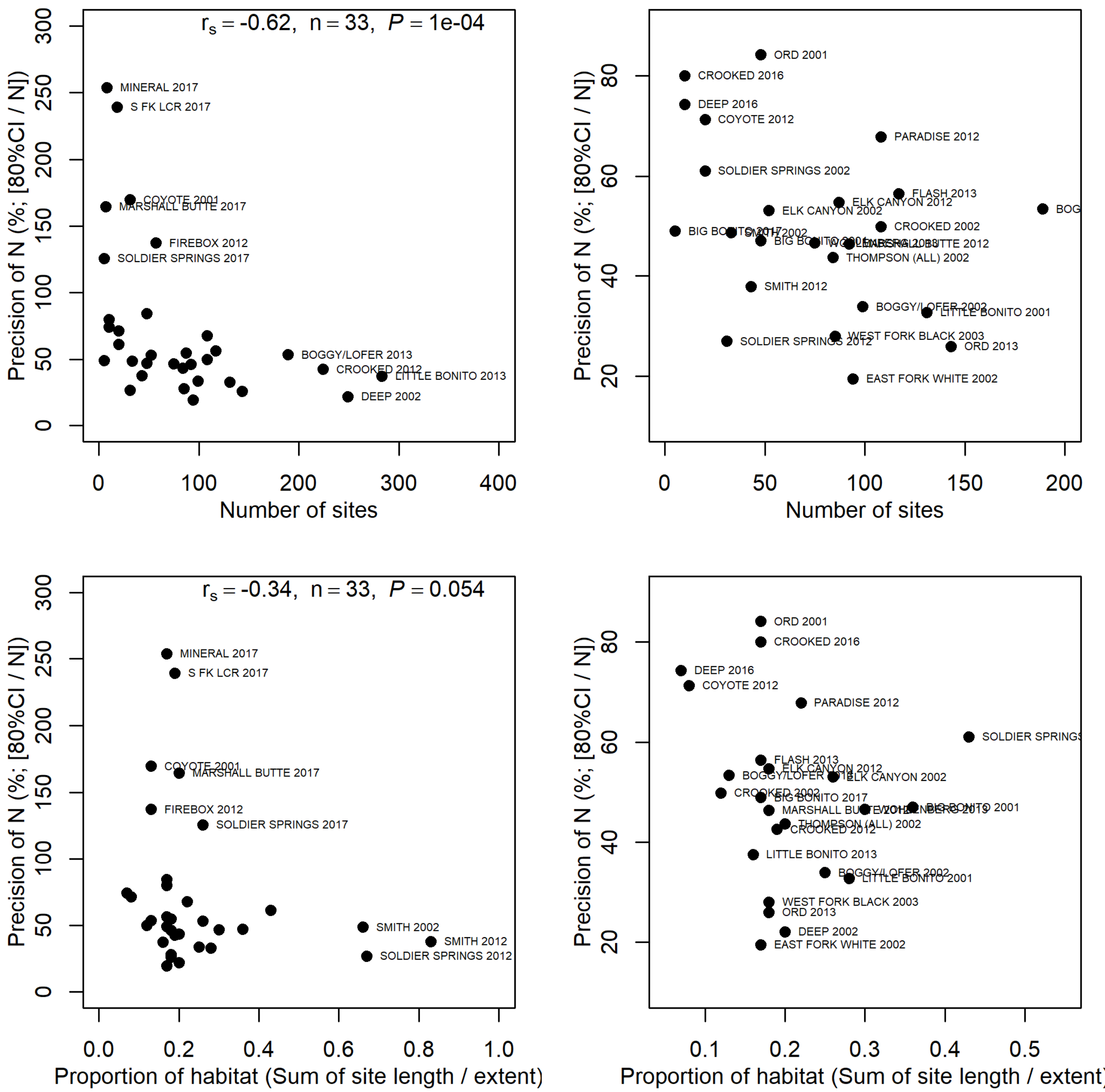

Figure 13B: Precision of adult (>130-mm TL) Apache Trout population size estimate, expressed as ratio of 80% confidence interval divided by population size estimate ($\widehat{N}$), versus number of sites (top panels) and proportion of habitat ([sum of survey site lengths / total habitat extent]; bottom panels) sampled for Apache Trout populations surveyed in different years. Right panels are zoomed in to data clusters of left panels. Spearman rank correlation statistics shown. Data are from U.S. Fish and Wildlife Service Basin Visual Estimation Technique (BVET) surveys, except surveys in 2016 and 2017 where systematic sampling at 100-m sites was conducted.

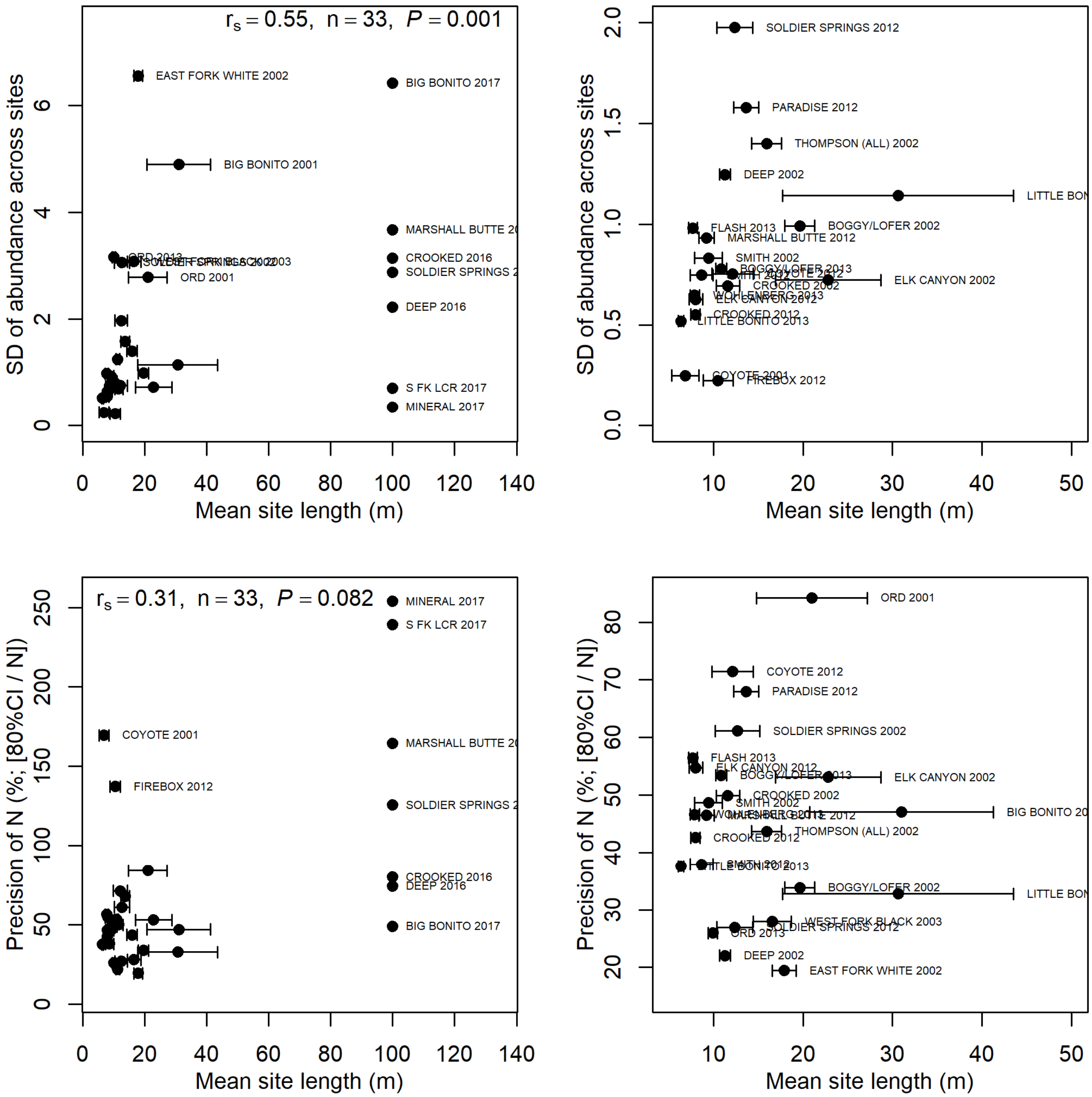

Figure 14B: SD of adult (>130-mm TL) Apache Trout abundance across sites versus average site length (±1 SE) (top panels), and precision of adult (>130-mm TL) Apache Trout population size estimate, expressed as ratio of 80% confidence interval divided by population size estimate ($\widehat{N}$), versus average sites length (±1 SE) sampled (bottom panels) for Apache Trout populations surveyed in different years. Right panels are zoomed in to data clusters of left panels. Spearman rank correlation statistics shown. Data are from U.S. Fish and Wildlife Service Basin Visual Estimation Technique (BVET) surveys, except surveys in 2016 and 2017 where systematic sampling at 100-m sites was conducted.

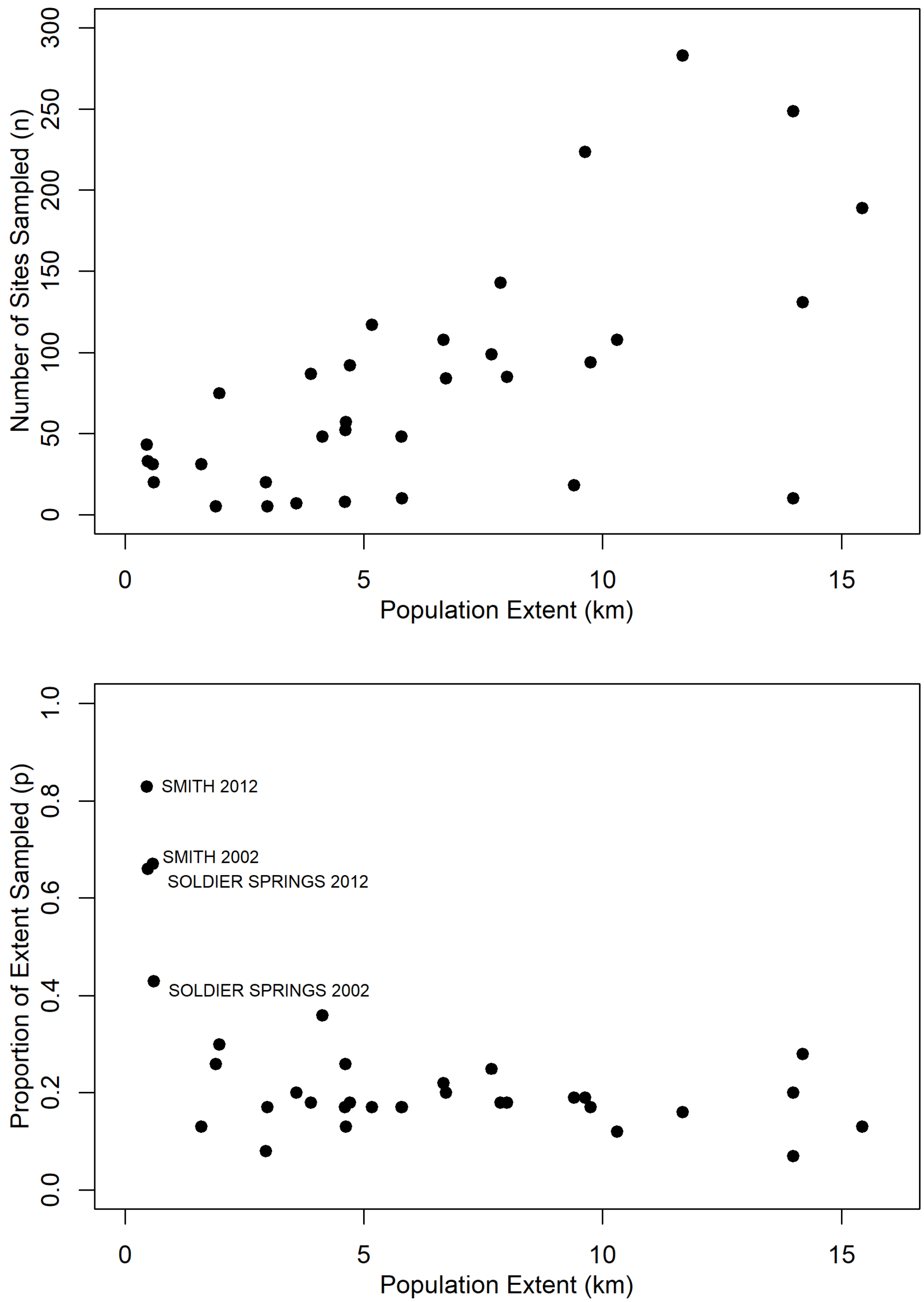

Figure 15B: Associations between population extent and sampling effort during BVET surveys, expressed as number of sites sampled (top panel) and proportion of extent sampled (bottom panel). Data are from U.S. Fish and Wildlife Service Basin Visual Estimation Technique (BVET) surveys, except surveys in 2016 and 2017 where systematic sampling at 100-m sites was conducted.

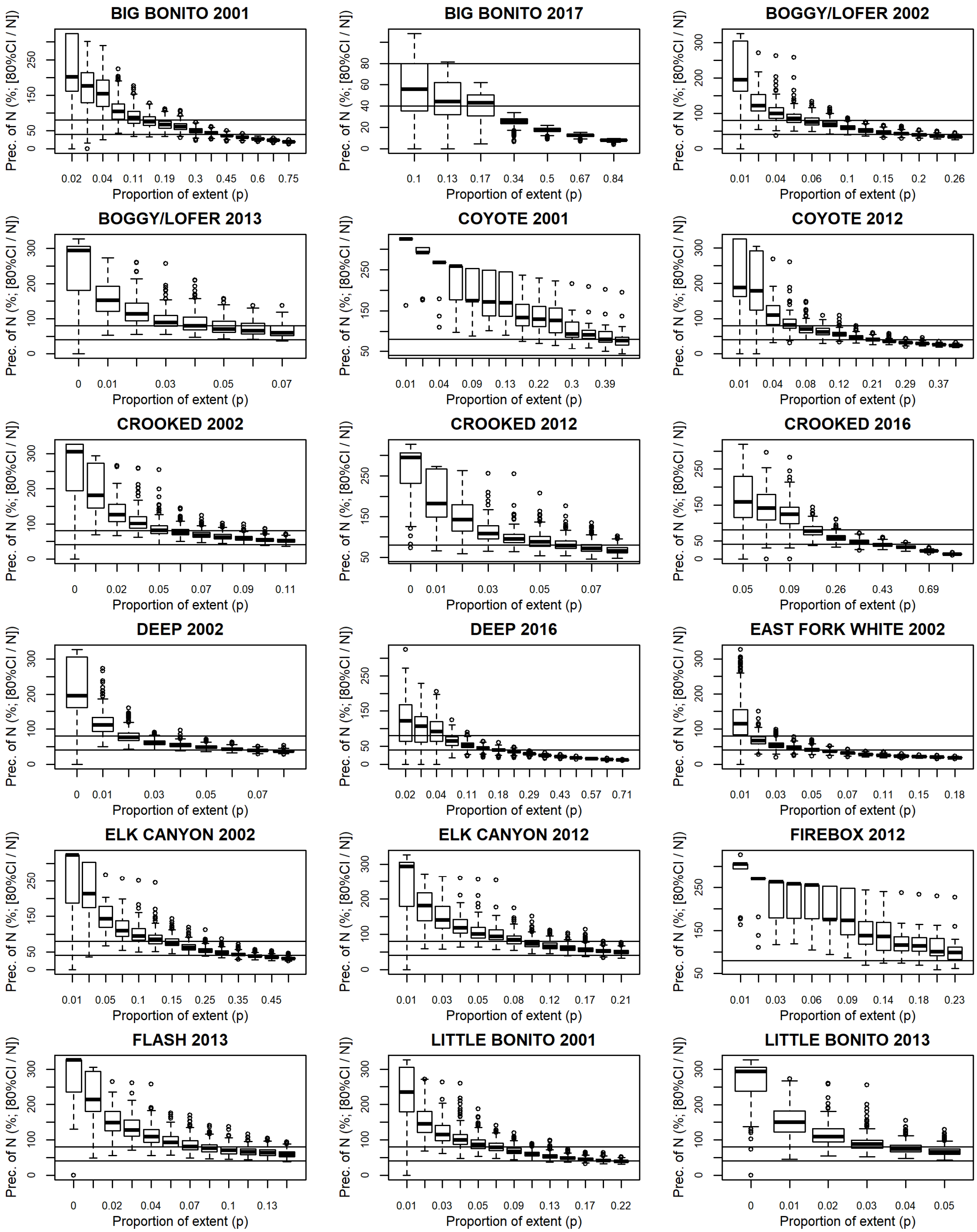

Figure 16B. Box plots showing bootstrapped estimates of precision of adult (>130-mm TL) Apache Trout population size estimate, expressed as ratio of 80% confidence interval divided by population size estimate ($\widehat{N}$), versus proportion of extent sampled (sum of sampled site lengths) for Apache Trout populations surveyed in different years. Data are from U.S. Fish and Wildlife Service Basin Visual Estimation Technique (BVET) surveys, except surveys in 2016 and 2017 where systematic sampling at 100-m sites was conducted.

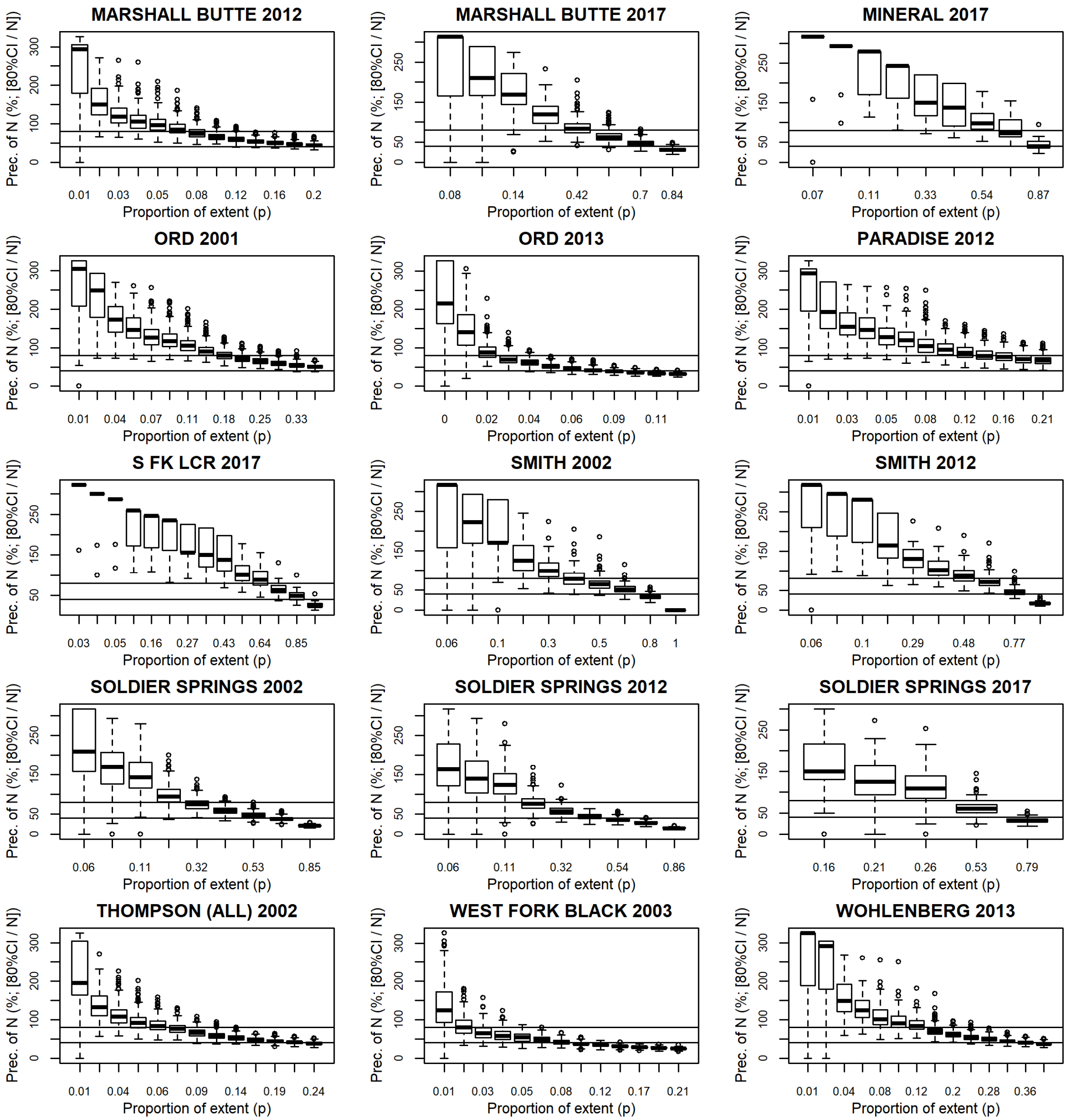

Figure 16B. Continued.

Appendix C. Trends in Single Populations and Rangewide Status

Although trend detection is not a monitoring objective (see 2. Goals and Objectives), for some purposes it might be useful to evaluate changes in population abundance between two or more time periods, or evaluate whether there are consistent trends across multiple populations (Dauwalter et al. 2010). There are several ways in which to evaluate trends.

Single Population Trends: The simplest way to evaluate trends in a single population is to compare abundance estimates for the population at time some initial time period *0* to abundance estimates for that population at a later time period *t*. A simple trend model commonly used for trend modeling is (Gotelli 1998):

$\widehat{N}_t = \widehat{N}_0 e^{rt}$

where $\widehat{N}_t$ is the population size at time *t*, $\widehat{N}_0$ is the initial population size, *e* is Euler's number (base of the $\log_e$), *t* is the time since the initial time period and for the purposes of this monitoring plan should represent years, and *r* is the intrinsic rate of population change and e^r is equivalent to λ ($e^r = \lambda$) and represents the proportional change in population size from the initial time period 0 to time *t*. The equation can linearized and solved directly when there are only two time periods:

$\log_e(\widehat{N}_t) = \log_e(\widehat{N}_0) + rt$

As a simple example, assume an estimate of abundance for a population was obtained in the years 2002 and 2009. Abundance was estimated to be 601 individuals in year 2002 ($\widehat{N}_{2002} = 601$) but was estimated to be 983 individuals in year 2009 ($\widehat{N}_{2009} = 983$). Using the linearized form of the equation above, the intrinsic rate of population change, *r,* can be solved as:

$\log_e(983) = \log_e(601) + r(2009 – 2002)$

or

$r = \log_e(983) – \log_e(601) / 7 = 0.0305$

Using Euler's number with *r* ($e^r = e^{0.0305} = 1.0310$) the data in this example shows there to be on average a 3.1% annual increase in abundance ($\widehat{N}$) over the time period from 2002 to 2009. Since there are only two time periods in for which to estimate of *r*, there is no way to estimate its error (e.g., standard error). Linear regression can be used to estimate *r* (and its standard error) with estimates of $\widehat{N}$ from three or more time periods (Year = independent variable (X); $\log_e(\widehat{N})$ = dependent variable (Y)).

Range-wide Population Trends: Sometimes it may be useful to evaluate the average trends across all populations to draw inference to rangewide population trends. In this case, one just has to estimate $\hat{r}_i$ for each population *i* and compute the average across all populations ($\hat{\hat{r}}$):

$\hat{\hat{r}} = \frac{\sum_{i=1}^{n} \hat{r}_i}{n}$

where $\hat{\hat{r}}$ is the average of all population-specific trend estimates $\hat{r}_i$ regardless of whether they are small or large populations, and *n* is the number of populations. For trends in the species status rangewide, it may be advisable to instead use a weighted average by weighting each individual trend estimate ($\hat{r}_i$) by the estimated population abundance of population *i* during the last monitoring time period *t* ($\widehat{N}_t$), thus giving more weight to the trends of larger, more abundant populations.

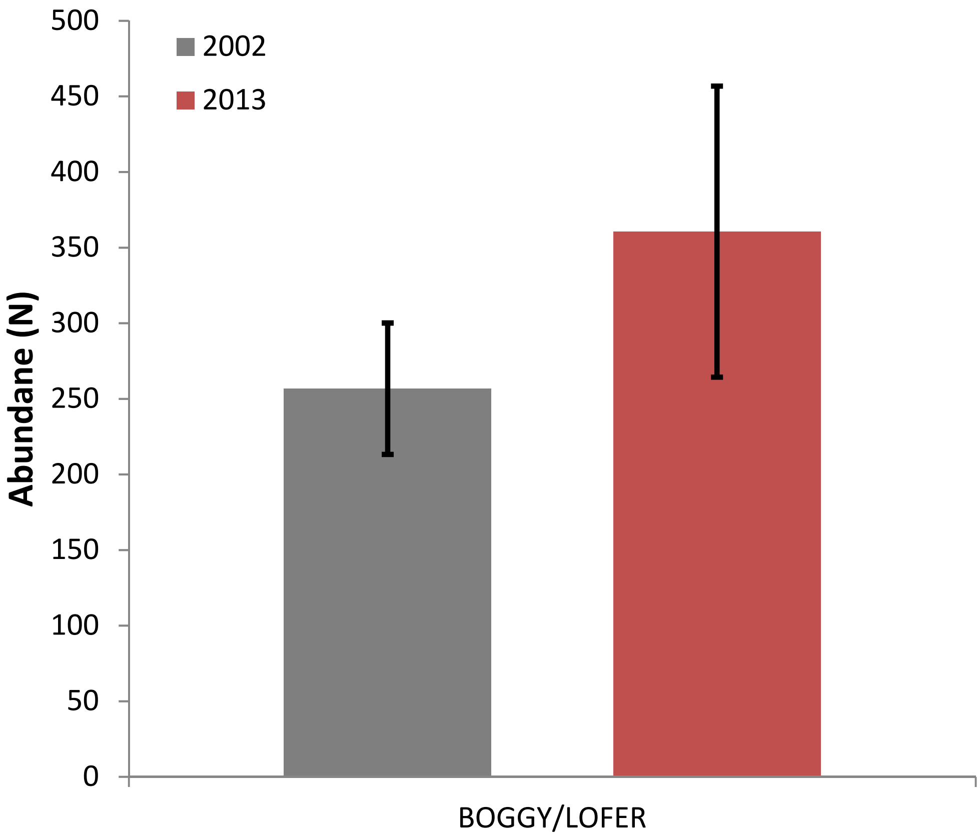

Figure 17C. Examples of population trend estimates (*r*) between two time periods for an Apache Trout stream. Abundance estimates of 361 in 2013 and 257 in 2002 yield an estimate of *r* = 1.031 or 3.1% increase in abundance per year.