Apache Trout Monitoring Plan: Revised

2024

Credit: AZGFD

Credit: D. Dauwalter

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: Revised

2024

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Note: This revision of the 2017 Plan was undertaken to: address a few editorial issues; incorporate information on the stream length bias of NHD, and how to correct for it, when extrapolating field data to obtain streamwide estimates of abundance (Appendix A); specify the estimator used to estimate adult Apache Trout abundance within sample units (reaches); specify use of the successive differences variance estimator for systematic samples, and update the assessment of relative precision of streamwide estimates of abundance and sample sizes required to meet monitoring goals using data collected under the 2017 Plan (from 2016 – 2023)(Appendix B).

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1. Purpose of the Plan

The Apache Trout *Oncorhynchus apache* is native to the White Mountains of east-central Arizona with a historical range that includes the White, Black, and Little Colorado river drainages above 1,800-m elevation (Figure 1; USFWS 2009; USFWS 2022). 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). In 2023, the U.S. Fish and Wildlife Service proposed to remove the Apache Trout from the List of Endangered and Threatened Wildlife; as of April 2024 no final rule has been issued.

The Apache Trout Recovery Plan has a goal of implementing actions necessary to delist the Apache Trout, with a specific objective of establishing and 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 to monitor, prevent, and control disease and/or causative agents, parasites, and pathogens that may threaten Apache Trout (USFWS 2009).

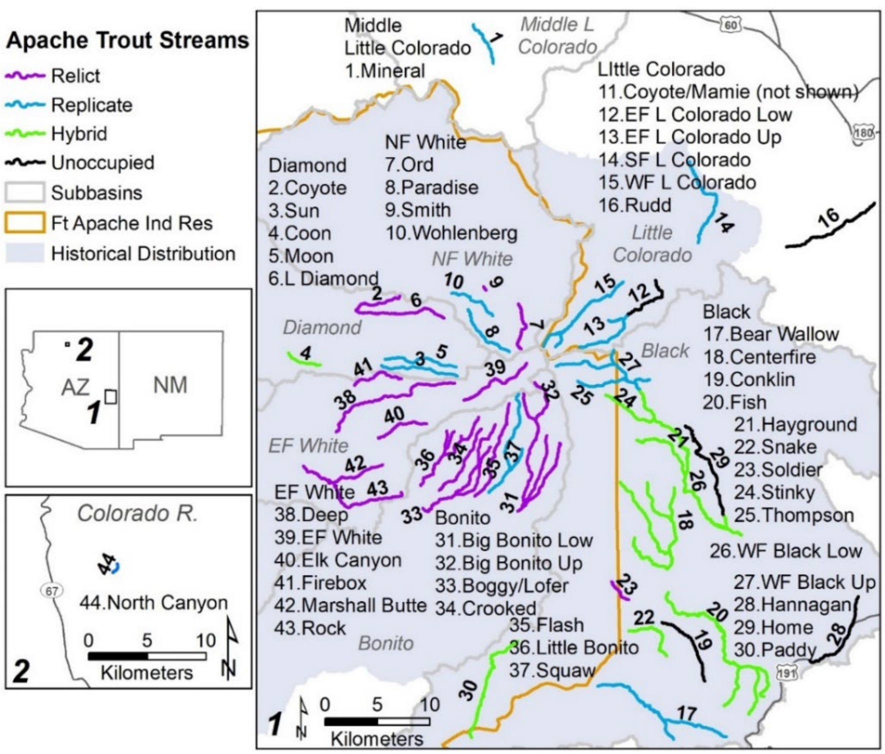


Figure 1. Map of Apache Trout streams with Relict, Replicate, and Hybridized populations, and streams currently unoccupied, Sin east-central Arizona. Figure from USFWS (2022).

Given the existing delisting criteria, or even in the event of delisting, there is a need to monitor Apache Trout populations to obtain 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). Inferential monitoring based upon obtaining unbiased estimates of population attributes (e.g., population abundance) obtained periodically over time allows for the strongest inferences regarding the status, and changes in status, of populations. 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 each population, and thus the species, to inform ESA 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, a main goal (Goal 1) of this plan is to accurately and precisely estimate the status of Apache Trout populations on a 5-year interval. This allows 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 an explicit monitoring goal (see below). The primary objective under Goal 1 is focused on estimating the abundance of adult Apache Trout (≥130-mm TL) in each population, whereas secondary objectives are focused on estimating the distribution of juveniles and adults and distribution of recruitment for each population. The plan also recommends ancillary stream habitat data collection, coincident with population monitoring, that will aid in the interpretation of population monitoring data. Monitoring the effectiveness of conservation barriers is Goal 2, with the primary objective being detection of non-native salmonids upstream of conservation barriers. The goals and objectives for this monitoring plan 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 (α = 0.20) no larger than 40% of estimated abundance every 5 years or sooner (i.e., if $\widehat{N}$ = 100, then confidence interval width should be ≤ 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 (α = 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 extent (sample units or reaches) with Apache Trout recruitment present (1 or more individuals <80mm TL) with an 80% confidence level (α = 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 habitats.

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, and 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 or reaches) 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 at least once every 5 years (Figure 2).

3a. Sampling Design

It is rare that all individuals in a population can be enumerated. Therefore, populations are sampled (sampled statistically, not as in field sampling) to draw inferences (make generalizations) about them (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) 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.

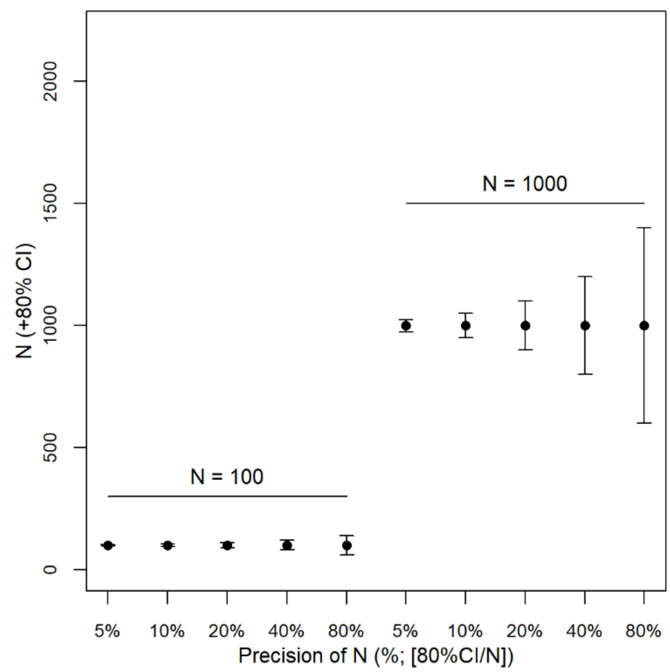


Figure 2. Example of confidence intervals at 5, 10, 20, 40, and 80% of estimated abundance ($\hat{N}$) for two levels of $\hat{N}$ ($\hat{N}$ = 100, and $\hat{N}$ = 1000).

Summary

1. **Identify Apache Trout Streams for Monitoring**: Identify Apache Trout populations for monitoring.
2. **Define Sampling Frame per Population**: For each population, determine the sampling frame by identifying the maximum potential habitat extent used by Apache Trout across all monitoring years, including any suitable tributary habitat, and then computing how many sample units are potentially available to be sampled within the sampling frame (i.e., divide habitat extent by sample unit [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 (e.g., downstream conservation barrier) 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 5 years, and populations should be monitored at approximately the same time of the year during years in which monitoring occurs.

Sampling Design Definitions

Definitions of the 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 (adult [≥130-mm TL], juvenile [≥80-mm TL], or age-0 [<80-mm TL] depending on objective).

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 available sampling units that could be included in a sample. For Apache Trout monitoring the sampling frame is all potential sample units (e.g., 100-m reaches) available to be selected (a sample) for field sampling of a population in the defined habitat extent.

Sample: List of sampling units selected for sampling. For Apache Trout monitoring the sample is a list of all sample units (reaches) systematically selected (a sample) for field sampling of a 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 identified and monitored separately. Identification of individual streams or sets of streams containing demographically isolated Apache Trout populations should be based on past monitoring data (e.g., Johnson 2011; Dauwalter et al. 2017), expert opinion, and known reintroductions while considering the influence of conservation barriers, habitat, and other factors on demographic isolation. Populations have been identified in the 2022 Species Status Assessment for the Apache Trout (Figure 1; USFWS 2022).

Define Sampling Frame per Population

Habitat Extent: Once the streams containing each demographically isolated Apache Trout population have been identified for monitoring, the sampling frame for each population should be defined. The sampling frame should be based on the maximum extent of stream habitat available to an Apache Trout population. The habitat extent available to each population has been defined in the 2022 Species Status Assessment for the Apache Trout (USFWS 2022) based on presence of conservation barriers, past population monitoring data (Johnson 2011; Dauwalter et al. 2017), physical habitat suitability (Petre and Bonar 2017), thermal suitability (Recsetar and Bonar 2013; Recsetar et al. 2014; Petre and Bonar 2017), exploratory sampling, and expert opinion (Figure 1; USFWS 2022). However, the lengths of these habitat extents are based on National Hydrography Dataset 1:24,000 scale flowlines, which are known to underestimate the length of stream occupied by Apache Trout by 11% on average (Dauwalter et al. 2022). As a result, the lengths of these habitat extents used to define the sampling frame should be adjusted using a correction factor as outlined in Appendix A. Adjusting for Bias in Stream Lengths to Determine Sample Frame. Delineating the extent of habitat potentially occupied by an Apache Trout population is an important step because it defines the sampling frame from which sample units (reaches) will be selected for monitoring and it is the extent to which monitoring data are extrapolated for streamwide estimates of adult Apache Trout abundance (see 3c. Computing the Monitoring Metrics). Underestimating the extent results in underestimates of streamwide estimates of abundance, that is, population size (Dauwalter et al. 2022).

Importantly, the sampling frames should be held constant over the timeframe of monitoring (i.e., used for each monitoring period; see Monitoring Schedule). For example, inter-annual variability in climate may result in a portion of the stream being dry in a particular year (Robinson et al. 2004), 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 the calculations of adult abundance and proportion of reaches occupied by juveniles or adults will consider that these sample units are unoccupied (Apache Trout abundance is assumed to be 0 at sites that are dry). 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. Goal 1, Objective 2 accounts for changes in distribution, aka site occupancy, within defined habitat extents due to interannual changes in habitat conditions.

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 when populations occupy streams 2-km or more of habitat, and 50-m be used when populations occupy <2-km of habitat (e.g., Smith). The 100-m reach length is based on fisheries convention, but also because one might 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 of individuals in suitable habitat units (e.g., pools). However, it is preferable to sample shorter reaches (e.g., 50-m) in populations with small habitat extents (e.g., <2-km) to ensure replication and good spatial coverage is possible. In special circumstances, longer reaches (200-m) may be needed to properly represent channel morphology and habitat sequencing (ruffle-run-pool) in larger streams (Ord in 2019).

Sampling Frame: Once a standard reach length is determined for a stream, 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 sample unit (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 (the sample of sample units).

Determine Number of Sampling Units Needed

The number of sample units (reaches) to be included in a sample must be determined before monitoring can occur. The number of sample units in the sample (sample size), among-unit variation in abundance, the proportion of the sample frame sampled (by way of a finite population correction of the variance estimate), and Type I error rate all determine the precision (confidence interval) of streamwide abundance estimates (Scheaffer et al. 2012).

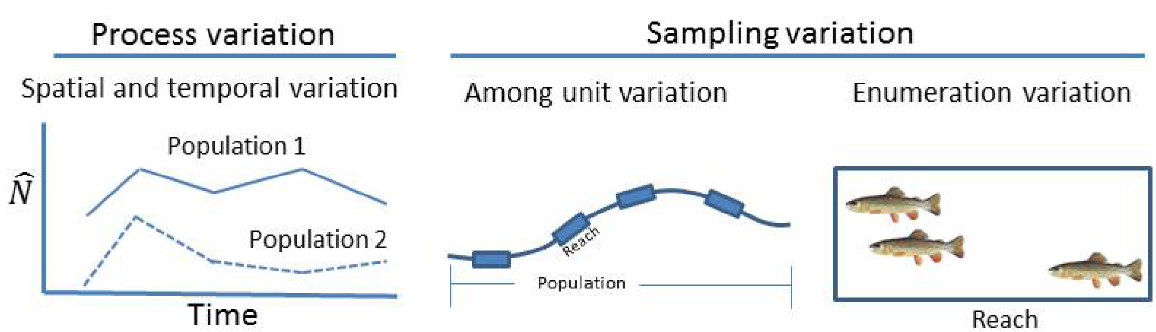
Sample a 100-m Reach Every 0.5 km: The recommended number of sampling units should be based on sampling a 100-m reach every 0.5 km along the defined habitat extent, which equates to sampling approximately 20% of the habitat and larger habitat extents require a larger sample size. This systematic sampling frequency follows the 2017 Plan and is based on the precision goal (Goal 1, Objective 1) and analysis of monitoring data collected from 2016 and earlier (Appendix B in Dauwalter et al. 2017).

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 over space and time due to changing environmental conditions. 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 estimating 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., Burnham 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 (5 reaches) within the stream and not every sample unit in the sample frame (20 reaches; 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.



In some cases, past monitoring of 100-m reaches every 0.5 km may yield consistently imprecise abundance estimates for a population. This suggests a larger sample size may be needed, i.e., more reaches need to be sampled, to meet the precision objective for monitoring (Goal 1, Objective 1). The 2017 Plan has been implemented since 2016 and there are eight years of monitoring data through 2023 according to the 2017 Plan (Appendix B). These data can be used to ascertain the sample sizes (number of reaches) needed to meet the precision objective for each population during future monitoring years. The following recommendations, in order, are suggested for establishing sample sizes needed to meeting abundance precision goals:

1) Sample a 100-m reach every 0.5 km along the defined habitat extent, which equates to sampling approximately 20% of the habitat extent using a systematic sampling design. The number of reaches will depend on the maximum habitat extent available to a population.
2) If the population has been monitored according to the 2017 Plan (2016 to present) and streamwide abundance estimates are imprecise (80% confidence interval is >40% of the estimate), then use the sample size estimated for meeting the precision goal for that population in Table 2B (Req. n column) of Appendix B. As an example, if it is determined that there are 50 sample units (reaches) in the sampling frame (e.g., 5000-m of habitat to be sampled using 100-m reaches) and it is determined that 40% needs to be sampled, the number of units to sample is 50 x 0.40 = 20 units (systematically sampled).
3) If past monitoring data are only available before 2016 and were collected using the BVET design as included in Appendix B of the 2017 Plan, then use the percent of habitat required for that population in the Req. % column in Table 6B of 2017 Plan (Dauwalter et al. 2017).
4) If no past monitoring data are available for a population, then use the percent habitat required in Table 2B (or Table 6B in 2017 Plan) from a population that occupies similar habitat nearby (use professional judgement). Sample size requirements are scaled to habitat extent, so be sure to use percent habitat required and convert percent habitat required to a sample size.
5) If the sample size or percent habitat requirements in Table 2B are logistically not feasible and it is expected that very few sites will be occupied by Apache Trout, establish a 100-m reach every 0.5 km and consider implementing an adaptive sampling approach based on initial site occupancy (see Box 2; Appendix B).

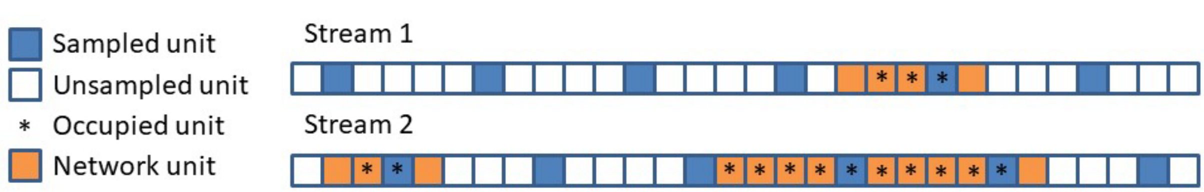
Select a Systematic Sample of Units for Monitoring

Systematic Sampling: After determining the sample size needed, sampling units should be selected using a systematic sample (Figure 3). Systematic sampling involves randomly identifying the first sampling unit, and then selecting every kth sampling unit thereafter where k refers to the interval at which samples are chosen; geographic information system (GIS) software can create a systematic sample on a polyline (stream). Systematic sampling can be easier to implement than random sampling when done in a GIS, on a map, or in the field. Systematic sampling also ensures thorough spatial coverage of sampling units within the sampling frame whereas random sampling can result in clustering of sample units. Finally, data generated from a systematic sample approximates a random sample if the sampling units do not exhibit ordering or periodicity or are not otherwise correlated in space (Scheaffer et al. 2012).

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.



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 k by reach length (e.g., if k = 2, then 2 x 100-m reach = 200 meters upstream from the downstream habitat extent boundary). Second,

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

select the remaining sampling units systematically by identifying every k^{th} sampling unit upstream. Using the previous example, if k = 5 and 10 of 50 sampling units need to be selected from the sampling frame, 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 (reach) length (e.g., 5 x 100-m = 500-m); this distance represents the beginning (downstream boundary) of the next sample unit upstream, and that if the desire is to sample 20% of the habitat then 100-m reaches (downstream reach boundary to downstream reach boundary; Figure 3) should be spaced every 0.5 km (or 500 m). 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 reches are established for the first monitoring time period, the same reaches can be revisited in future monitoring years.

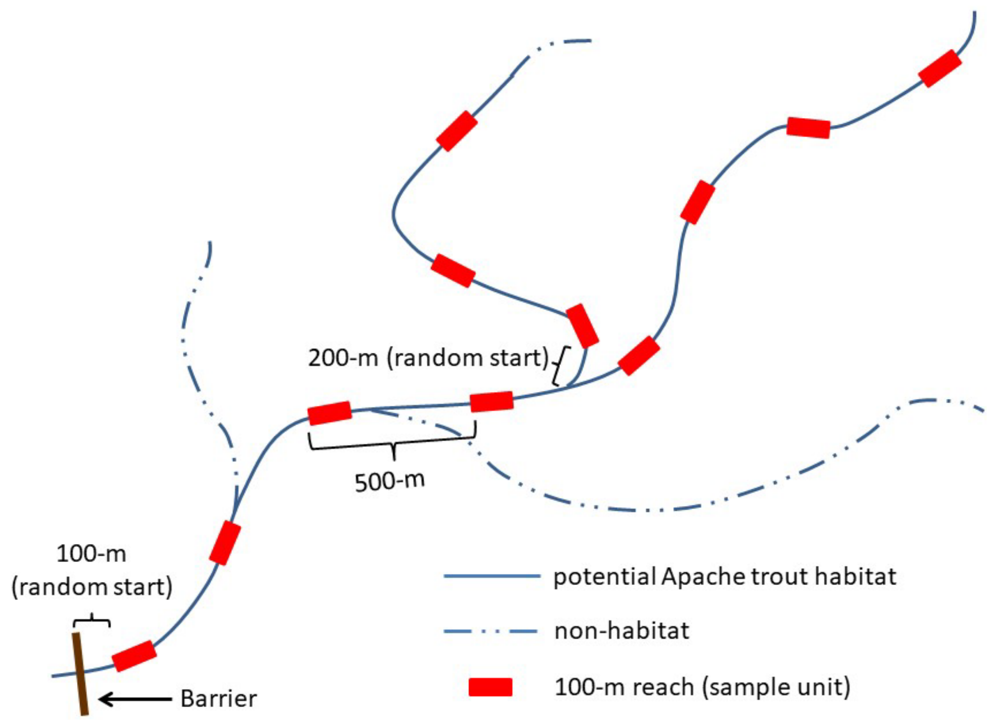


Figure 3. Schematic of a systematic sampling design for a hypothetical population. The first downstream sampling unit (100-m reach) was randomly selected and the remaining sample units were spaced every 5th unit (500-m), upstream from the downstream sample unit boundary. The same sample unit selection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

Monitoring Schedule

Each Apache Trout population should be monitored at least once every 5 years. Populations can be scheduled as groups of populations referred to as panels. Then, all populations in a panel can be sampled during the same year, and different panels of populations are monitored each 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 in 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 survey 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 to 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 5 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 for unbiased estimates of the monitoring metrics to be computed, and this involves multiple-pass depletion electrofishing at each sample unit (reach) per population. These data will allow estimation of adult Apache Trout abundance and document presence of age-1 and older Apache Trout, and presence of age-0 Apache Trout per monitoring objectives under Goal 1. Physical habitat will also be measured within the sample unit as ancillary data to help interpret population monitoring metrics among populations 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). Alternatively, measure in the field the upstream distance from the previous sample unit to the next sample unit.
2. **Define Sampling Unit (Reach)**: Measure upstream from the downstream boundary the predetermined length of the sample unit (e.g., 100-m reach) to identify the upstream reach boundary, being careful not to scare fish from the reach. Set 6.35-mm bar mesh block nets at each reach boundary.
3. **Fish Sampling**: Employ daytime multiple-pass backpack electrofishing within the reach by sampling fish using the 100-Watt method, 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.
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 (e.g., downstream extent of occupied habitats, area burned by wildfire).

Navigate to Sampling Unit (Reach)

Prior to entering the field, compile a list of all sampling units for each population 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 (or 50-m) upstream along the streambank taking care not to frighten fish from the reach and install 6.35-mm (0.25-in) bar mesh block nets at the upstream and downstream reach boundaries. Salmonids have been shown to frequently move upstream and out of defined reaches during electrofishing (Peterson et al. 2005), so block nets need to be used to meet the closed population assumptions associated with removal abundance estimates (Seber 1982; Peterson et al. 2005). Block nets should be secured in a way that ensures full closure of the reach throughout all 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 escape over it. 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. Again, take care to avoid scaring 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. See Reynolds and Kolz (2012) for more comprehensive discussion of electrofishing safety.

During population monitoring, the 100-watt method will be used to sample Apache Trout in the enclosed reach (Meyer et al. 2021). The 100-watt method is a protocol that maximizes capture efficiency in small streams while minimizing electrofishing injury to salmonids. It does so by standardizing backpack electrofishing power output to an average of 100 W. To apply the method, electrofisher settings need to be determined outside the sample reach before sampling begins. To determine the correct settings, first set the backpack electrofisher to pulsed DC, a pulse frequency of 60 Hz, and duty cycle of 25% (or pulse width at 4 ms). Second, outside of the reach to be electrofished locate a riffle and pool that are representative of habitat in the reach. Third, in the riffle start with low voltage (e.g., 200 V) and adjust voltage, with the anode at midwater depth, upward to achieve an average power output of 70-100 W. Fourth, determine average power output in the representative pool, again with the anode at midwater depth; it should be 100-130 W. The average power output between the riffle and pool readings should be ~100 W. If the average is higher or lower than 100 W, adjust the voltage accordingly until average power output of ~100 W is achieved. On rare occasions, duty cycle and pulse frequency may need to be modified slightly to achieve 100 W. Reynolds and Kolz (2012) is an excellent reference for understanding the theory and practice of electrofishing. Be sure to record the electrofisher settings and resulting average power output, the adjustments needed, observations of fish behavior, and ambient water conductivity and temperature. During past Apache Trout monitoring, conductivities averaged 45 µS/cm (range = 3 – 836 µS/cm) and stream temperatures averaged 10.1°C (SD = 5.6°C). The 100-watt method may not yield desired effectiveness when electrofishing to remove non-native fishes.

Once testing shows electrofishing settings to achieve 100 W of power output on average, begin sampling by conducting two electrofishing passes in the sample unit (reach). 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 to the water 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 and encourage recovery. The amount of electrofishing effort should remain constant across all electrofishing passes 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.

The number of electrofishing passes is typically three but is dependent on the pattern of catch across successive passes. If no Apache Trout are collected in pass 1 and 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 or pass 2, then at least a third pass should be conducted. 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 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, 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 sample unit (reach). See Appendix B for estimates of capture probabilities from past Apache Trout monitoring.

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

Each Apache Trout collected during an electrofishing pass should be measured to the nearest millimeter total length (TL; Figure 5). If large numbers of fish cannot be processed quickly, or if individual fish cannot be measured accurately because of a tendency to move erratically, 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 because few sample units are occupied (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. Multiply mean wetted width by reach length to obtain the area sampled. Linear (number per 100-m) and areal (number per 100-m^2) 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 (Dauwalter et al. 2023), 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; Petre and Bonar 2017).

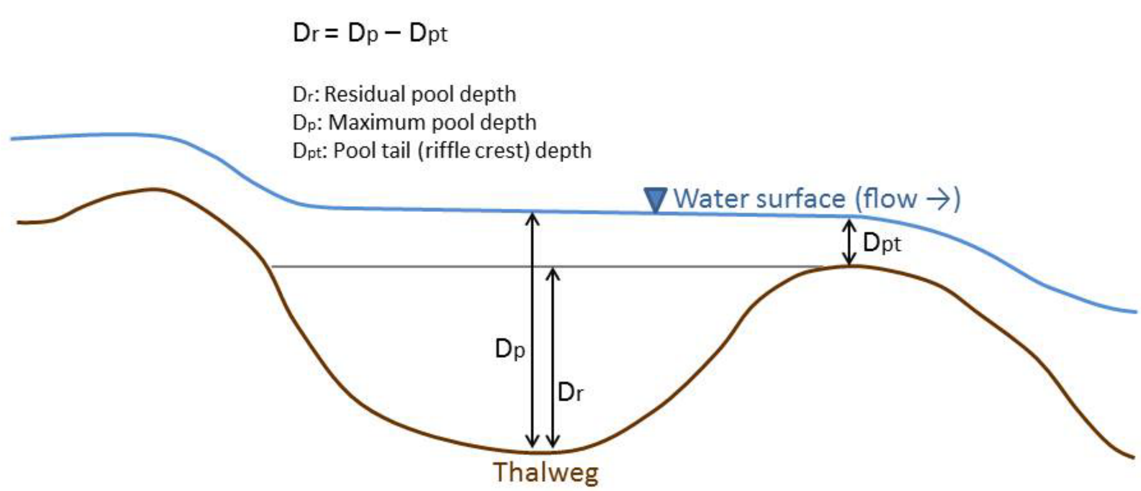


Figure 6. Schematic showing measurement of residual pool depth by measuring the maximum pool depth along the thalweg and subtracting off the depth of pool tail (riffle crest) at the thalweg.

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). Harper (1978) found the smallest female Apache Trout with eggs, indicating maturity, to be 130-mm TL.

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. It is recommended to use the general k-pass estimator presented by Burnham in the commonly used software MicroFish (Van Deventer and Platts 1989). The estimator is available in the FSA package in Program R (R Core Team 2015; Ogle 2017). The Zippin (Zippin 1958) and Carle-Strub (Carle and Strub 1978) removal estimators are also commonly used. Below is an example `output` from the Program **R code** needed to estimate abundance of trout with the Burnham 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="Burnham")      #submit 'ct' to removal function, specify estimator, assign to 'est'
summary(est)                              #submit removal command output 'est' to summary function
```

```
     Estimate Std. Error
No 65.0000000   7.460927
p   0.3666667   0.083503
```

```
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 Burnham removal estimator, that there was between 50 and 82 trout in the sample unit (reach) when it was electrofished (best estimate is No=65 fish). The estimator also tells us that the average capture probability for an individual fish on a given pass, *p*, is 36.7% (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 when habitat complexity is high and capture probability is low to moderate, 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 ($\widehat{N}$): The information at sample units (reaches) is then summarized and extrapolated to the sample frame (occupied extent) to a streamwide estimate of adult Apache Trout (≥130mm TL) abundance. The streamwide abundance estimate, ($\widehat{N}$), is computed by multiplying the mean number of adult Apache Trout across all sampling units (reaches) 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 (reach) 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:

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

where N_i and n_i are as defined above, and s_i^2 is the variance of differences in abundance of adult Apache Trout across successive sample units (reaches) in population *i* (Scheaffer et al. 2012, page 238). In the case of successive differences, s_i^2 is computed as: $s_i^2 = \frac{1}{2n_i(n_i-1)} \sum_{j=1}^{n_i-1} d_j^2$, where $d_j = y_j - y_k$, the difference in adult abundance between successive sample units *j* and *k* (Scheaffer et al. 2012, page 238), and all other terms are 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 $\hat{V}\left(\widehat{N}_i\right)$ as specified is based on the successive differences in adult abundance between sample units from a systematic sample and not the variance estimator for a simple random sample that will overestimate variance when systematic data exhibit ordering or linear trends (see Scheaffer et al. 2012).

The variance $\hat{V}\left(\widehat{N}_i\right)$ 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:

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

and the lower 80% confidence limit as:

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

where $\hat{V}(N_i \bar{y}_i)$ 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.

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

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 by 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.

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 if 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*.

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 ($\hat{N}_i$) 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:

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

where $\hat{N}_i$ 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 $\hat{N}_{total}$ is:

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

where $\hat{V}(\hat{N}_i)$ is the variance for the abundance estimate ($\hat{N}_i$) 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 of Apache Trout populations by non-native salmonids 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 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. Non-Native Trout Sampling:

Non-Native Trout 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 to detect any non-native salmonids present. 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 be 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 (see Appendix B). 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 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 non-native trout 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 non-native trout during electrofishing pass *j*. Similarly, the probability of detecting at least one individual of $\widehat{N}$ total non-native trout is:

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

where *d* is as defined above and $\widehat{N}$ is the estimated number of non-native trout in the reach (typically unknown).

As an example, assume five Brown Trout circumvented a conservation barrier and invaded a conservation population of Apache Trout upstream. To detect this invasion of Brown Trout, 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) x (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 listing and delisting processes (USFWS 2022). It can also serve as a post-delisting monitoring plan if Apache Trout are removed from the List of Endangered and Threatened Wildlife.

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References

Armantrout, N. B. 1998. Aquatic habitat inventory terminology. American Fisheries Society, Bethesda, Maryland.

Bonar, S. A., S. Contreras-Balderas, and A. C. Iles. 2009. An introduction to standardized sampling. S. A. Bonar, W. A. Hubert, and D. W. Willis, editors. Standard methods for sampling North American freshwater fishes. American Fisheries Society, Bethesda, Maryland.

Carle, F. L., and M. R. Strub. 1978. A new method for estimating population size from removal data. Biometrics 34:621-630.

Cook, N., F. J. Rahel, and W. A. Hubert. 2010. Persistence of Colorado River cutthroat trout populations in isolated headwater streams of Wyoming. Transactions of the American Fisheries Society 129:1500-1510.

Dauwalter, D., B. Giordano, Z. Jackson, J. Johnson, M. Lopez, and T. Stephens. 2017. Apache Trout monitoring plan: a monitoring plan for small and isolated trout populations. Trout Unlimited, Arlington, Virginia.

Dauwalter, D. C., and W. L. Fisher. 2007. Electrofishing capture probability of smallmouth bass in streams. North American Journal of Fisheries Management 27:162-171.

Dauwalter, D. C., W. L. Fisher, and K. C. Belt. 2006. Mapping stream habitats with a global positioning system: accuracy, precision, and comparison with traditional methods. Environmental Management 37:271-280.

Dauwalter, D. C., T. Gatewood, Z. J. Jackson, J. Barney, and Z. S. Beard. 2022. Digital hydrography underestimates stream length and leads to underestimates of trout population size. North American Journal of Fisheries Management 42:994-1002.

Dauwalter, D. C., R. Gonzalez, T. Gatewood, Z. S. Beard, and Z. Jackson. 2023. Resiliency of Apache Trout habitats in a warmer and drier climate future. Transactions of the American Fisheries Society 152:361-380.

Dauwalter, D. C., F. J. Rahel, and K. G. Gerow. 2010. Power of revisit monitoring designs to detect forest-wide declines in trout populations. North American Journal of Fisheries Management 30:1462-1468.

Dunham, J. B., G. L. Chandler, B. E. Rieman, and D. Martin. 2005. Measuring stream temperature with digital data loggers: a user's guide. General Technical Report RMRS-GTR-150WWW. U. F. Service, Boise, Idaho. http://www.fs.fed.us/rm/pubs/rmrs_gtr150.pdf.

Dunham, J. B., A. E. Rosenberger, R. F. Thurow, C. A. Dolloff, and P. J. Howell. 2009. Coldwater fish in wadeable streams. Pages 119-138 *in* S. A. Bonar, W. A. Hubert, and D. W. Willis, editors. Standard methods for sampling North American freshwater fishes. American Fisheries Society, Bethesda, Maryland.

Gotelli, N. J. 1998. A primer of ecology. Sinauer Associates, Inc., Sunderland, Massachusetts.

Hansen, M. J., T. D. Beard Jr., and D. B. Hayes. 2007. Sampling and experimental design. Pages 51 - 120 *in* C. S. Guy, and M. L. Brown, editors. Analysis and interpretation of freshwater fisheries data. American Fisheries Society, Bethesda, Maryland.

Harper, K. C. 1978. Biology of a southwestern salmonid, *Salmo apache* (Miller 1972). Pages 99 - 111 *in* J. R. Moring, editor. Proceedings of the wild trout - catchable trout symposium. Oregon Department of Fish and Wildlife, Research and Development Section, Eugene, Oregon.

Hawkins, C. P., J. L. Kershner, P. A. Bisson, M. D. Bryant, L. M. Decker, S. V. Gregory, D. A. McCullough, C. K. Overton, G. H. Reeves, R. J. Steedman, and M. K. Young. 1993. A hierarchical approach to classifying stream habitat features. Fisheries 18:3-12.

Isaak, D. J., D. L. Horan, and S. Wollrab. 2013. A simple protocol using underwater epoxy to install annual temperature monitoring sites in rivers and streams. RMRS-GTR-314. F. S. U.S. Department of Agriculture, Rocky Mountain Research Station, Fort Collins, Colorado.

Jennings, C. A., B. L. Sloss, B. A. Lasee, G. J. Burtle, and G. R. Moyer. 2012. Care, handling, and examination of sampled organisms. Pages 163-221 *in* A. V. Zale, D. L. Parrish, and T. M. Sutton, editors. Fisheries techniques, Third edition. American Fisheries Society, Bethesda, Maryland.

Johnson, J. L. 2011. Apache trout population assessments on the Fort Apache Indian Reservation 2007-2010 summary report. USFWS-AZFWCO-PT-11-020, Arizona Fish and Wildlife Conservation Office, U.S. Fish and Wildlife Service, Pinetop, Arizona.

McDonald, L. L. 2004. Sampling rare populations. W. L. Thompson, editor. Sampling rare or elusive species. Island Press, Washington, DC.

McMullin, S. L., and E. Pert. 2010. The process of fisheries management. Pages 133-155 *in* W. A. Hubert, and M. C. Quist, editors. Inland fisheries management in North America, Third edition. American Fisheries Society, Bethesda, Maryland.

Meyer, K. A., L. V. Chiaramonte, and J. B. Reynolds. 2021. The 100-watt method: a protocol for backpack electrofishing in small streams. Fisheries 46:125-130.

Meyer, K. A., and B. High. 2011. Accuracy of removal electrofishing estimates of trout abundance in Rocky Mountain streams. North American Journal of Fisheries Management 31:923-933.

Meyer, K. A., D. J. Schill, E. R. J. Mamer, C. C. Kozfkay, and M. R. Campbell. 2014. Status of redband trout in the Upper Snake River basin of Idaho. North American Journal of Fisheries Management 34:507-523.

Ogle, D. H. 2017. FSA: Fisheries Stock Analysis. R package version 0.8.17.

Peterson, J. T., N. P. Banish, and R. F. Thurow. 2005. Are block nets necessary? Movement of stream-dwelling salmonids in response to three common survey methods. North American Journal of Fisheries Management 25:732-743.

Peterson, J. T., and C. P. Paukert. 2009. Converting nonstandard fish sampling data to standardized data. Pages 195-215 *in* S. A. Bonar, W. A. Hubert, and D. W. Willis, editors. Standard methods for sampling North American Freshwater Fishes. American Fisheries Society, Bethesda, Maryland.

Peterson, J. T., R. F. Thurow, and J. W. Guzevich. 2004. An evaluation of multipass electrofishing for estimating the abundance of stream-dwelling salmonids. Transactions of the American Fisheries Society 133:462-475.

Petre, S. J., and S. A. Bonar. 2017. Determination of habitat requirements for Apache trout. Transactions of the American Fisheries Society 146:1-15.

R Core Team. 2015. R: a language and environment for statistical computing. The R Foundation for Statistical Computing, Vienna, Austria.

Recsetar, M. S., and S. A. Bonar. 2013. Survival of Apache Trout eggs and alevins under static and fluctuating temperature regimes. Transactions of the American Fisheries Society 142:373-379.

Recsetar, M. S., S. A. Bonar, and O. G. Feuerbacher. 2014. Growth and survival of Apache Trout under static and fluctuating temperature regimes. Transactions of the American Fisheries Society 143:1247-1254.

Reynolds, J. B., and A. L. Kolz. 2012. Electrofishing. Pages 305-361 *in* A. V. Zale, D. L. Parrish, and T. Sutton, editors. Fisheries Techniques, 3rd edition. American Fisheries Society, Bethesda, Maryland.

Robinson, A. T., L. D. Avenetti, and C. J. Cantrell. 2004. Evaluation of Apache Trout protection actions. Arizona Game and Fish Department, Phoenix, Arizona.

Roper, B. B., J. L. Kershner, E. Archer, R. Henderson, and N. Bouwes. 2002. An evaluation of physical stream habitat attributes used to monitor streams. Journal of the American Water Resources Association 38:1637-1646.

Rosenberger, A. E., and J. B. Dunham. 2005. Validation of abundance estimates from mark-recapture and removal techniques for rainbow trout captured by electrofishing in small streams. North American Journal of Fisheries Management 25:1395-1410.

Saunders, W. C., K. D. Fausch, and G. C. White. 2011. Accurate estimation of salmonid abundance in small streams using nighttime removal electrofishing: an evaluation using marked fish. North American Journal of Fisheries Management 31:403-415.

Scheaffer, R. L., W. Mendenhall III, R. L. Ott, and K. G. Gerow. 2012. Elementary survey sampling, Seventh edition. Brooks/Cole, Boston.

Seber, G. A. F. 1982. The estimation of animal abundance and related parameters, 2nd edition. Charles Griffith, London.

Sweka, J. A., C. M. Legault, K. F. Beland, J. Trial, and M. J. Millard. 2006. Evaluation of removal sampling for basinwide assessment of Atlantic salmon. North American Journal of Fisheries Management 26:995-1002.

Thompson, W. L., G. C. White, and C. Gowan. 1998. Monitoring vertebrate populations. Academic Press, San Diego, California.

Urquhart, N. S., and T. M. Kincaid. 1999. Designs for detecting trend from repeated surveys of ecological resources. Journal of Agricultural, Biological, and Environmental Statistics 4:404-414.

USFWS. 1967. Native fish and wildlife - endangered species. Federal Register 32 (48):4001.

USFWS. 1975. "Threatened" status for three species of trout. Federal Register 40 (137):29863-29864.

USFWS. 2009. Apache trout recovery plan. Second Revision, U.S. Fish and Wildlife Service, Albuquerque, New Mexico.

USFWS. 2022. Species status assessment for the Apache Trout *Oncorhynchus apache*. U.S. Fish and Wildlife Service, Flagstaff, Arizona. https://www.fws.gov/media/apache-trout-species-status-assessment-2022#:~:text=This%20Species%20Status%20Assessment%20(SSA,and%20threats%2C%20and%20conservation%20actions.

Van Deventer, J. J., and W. S. Platts. 1989. MicroFish 3.0. INT-254. Intermountain Research Station, Boise, Idaho.

Wing, M. G., A. Eklund, J. Sessions, and R. Karsky. 2008. Horizontal measurement performance of five mapping-grade global positioning system receiver configurations in several forested settings. Western Journal of Applied Forestry 23:166-171.

Zippin, C. 1958. The removal method of population estimation. Journal of Wildlife Management 22:82-90.

Appendix A. Adjusting for Bias in Stream Lengths to Determine Sample Frame

The habitat extent occupied by Apache Trout population represents the sample frame from which sample units (reaches) are selected for monitoring. These habitat extents have been defined in the 2022 Species Status Assessment (SSA) for the Apache Trout (USFWS 2022) using National Hydrography Dataset (NHD) 1:24,000 flowlines. However, the NHD flowlines are known to be generalized representations of streams that underestimate the stream length in the field (Figure 7A). This includes Apache Trout streams where stream length is underestimated more in meadows than forested sections (Dauwalter et al. 2022). For Apache Trout streams, these NHD stream lengths are underestimated, on average, by 11.1%, which can lead to underestimates of streamwide estimates of adult Apache Trout abundance (Dauwalter et al. 2022).

When the habitat extents as defined in the SSA (USFWS 2022) are used to define the sample frame for a population their length must be corrected to account for the fact that they underestimate stream length in the field (Dauwalter et al. 2022). The NHD length must be divided by the length of stream observable in aerial imagery, such as National Agricultural Imagery Program (NAIP), or measured in the field to obtain an unbiased length estimate. For example, the habitat extent occupied by Apache Trout in Coyote Creek on the Fort Apache Indian Reservation is 5.1 km as measured using NHD 1:24,000 flowlines. Using NAIP imagery, the NHD was shown to be 0.88 of the length of Coyote Creek in aerial imagery, i.e., underestimated by 12%. Thus, the sample frame for Coyote Creek should be based on the NHD length divided by the NHD/NAIP imagery length ratio (Table 1A) to achieve an adjusted and unbiased habitat extent length: Adjusted length (km) = NHD Length (km) / (NHD/NAIP Ratio) = 5.1 / 0.88 = 5.8 km. The adjusted length of habitat extents for a subset of Apache Trout populations are presented in Table 1A (from Dauwalter et al. 2022). If the NHD/NAIP length ratio hasn't been estimated for an Apache Trout population or stream that is to be monitored, the mean NHD/NAIP ratio can be used to adjust the NHD length to define the sample frame for a population (N_i in section: Adult Abundance).

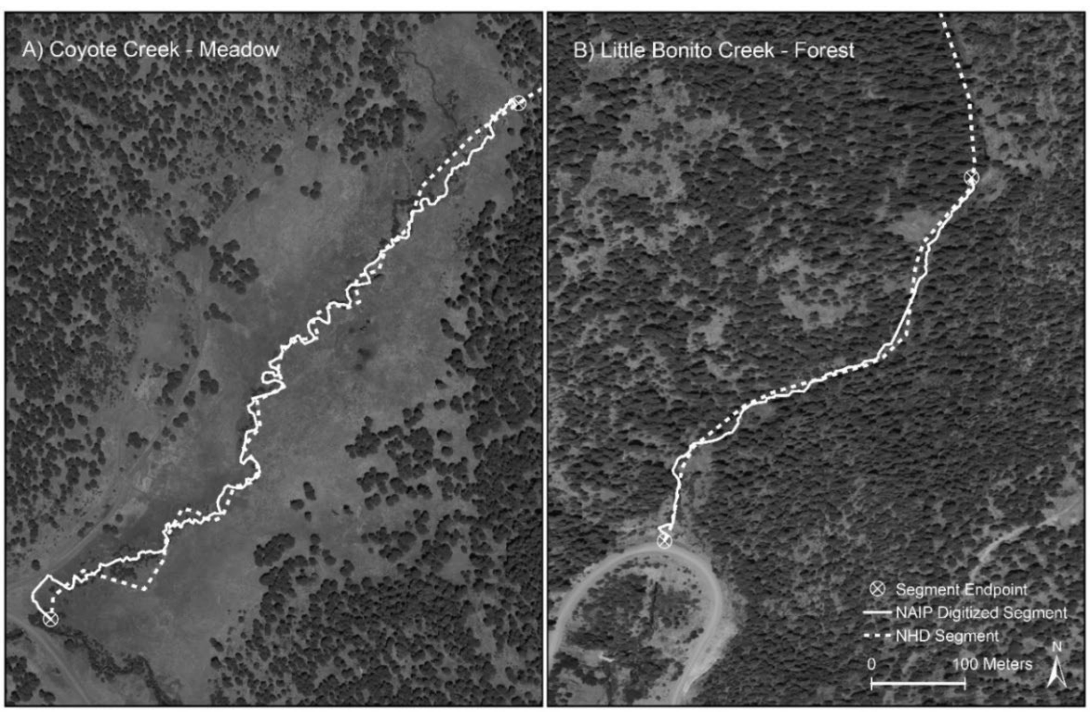


Figure 7A. Example of stream segments from NHD and digitized from 2017 NAIP imagery for a meadow segment in Coyote Creek (A) and a forested section in Big Bonito Creek (B), Arizona. From Dauwalter et al. (2022).

Table 1A. Ratio of NHD length to NAIP digitized length, occupied stream length measured using NHD, adjusted occupied length using NHD / NAIP ratio, and % difference of lengths (100×NHD-NAIP/NAIP) for Apache Trout streams in the White Mountains of Arizona. From Dauwalter et al. (2022).

Stream / Population	NHD/NAIP Ratio	NHD Length (km)	Adjusted Length (km)	% Difference
Coyote	0.88	5.1	5.8	-12.1
Deep	0.89	14.6	16.3	-10.4
Flash	0.81	10.4	12.9	-19.4
Little Bonito	0.90	14.8	16.4	-9.8
Mineral	0.94	4.7	5.0	-6.0
Ord	0.84	5.6	6.7	-16.4
Paradise	0.88	6.5	7.4	-12.2
S. Fk. Little Colorado	0.96	10.6	11.0	-3.6
Soldier	0.89	2.7	3.0	-10.0
Squaw	0.90	13.7	15.2	-9.9
W. Fk. Black	0.89	18.6	20.8	-10.6
W. Fk. Little Colorado	0.88	14.3	16.3	-12.3
Mean	0.89	10.1	11.4	-11.1
Median	0.89	10.5	11.9	-10.5
SD	0.04	5.1	5.7	4.1

Appendix B. Investigating the Precision of $\widehat{N}$

Monitoring of stream salmonid populations requires a sampling framework that allows stream survey data to be extrapolated to the entire length of occupied habitat to estimate parameters of a populations at the stream scale (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 number of trout per reach (statistical sample) is often extrapolated to the entire habitat extent (km) based on how many reaches are available to sample (statistical population) to obtain a total population size ($\widehat{\boldsymbol{N}}$). 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, 2) the number of sites surveyed (sample size), 3) the size of the sample frame (habitat extent), and 4) Type I error rate (false positive). In finite population sampling, the variance is adjusted by the fraction of the total statistical population sampled, termed the finite population correction (fpc). The fpc effectively reduces the variance by the proportion of habitat that was actually sampled, which has no uncertainty. For example, in a 2-km stream there are 20 possible 0.1-km (100-m) reaches available to be selected for sampling. If 10 are selected for sampling, then 50% of the available habitat will be sampled $(N - n) / N = (20 - 10) / 20 = 0.5$, or 50%. If expressed as a unit of stream length: $(L - \sum l_i) / L = (2km - (10 \times 0.1km)) / 2km = 0.5$, or 50%. If you sampled the entire stream the only uncertainty would come from enumeration methods (removal estimator)(Box 1).

The primary objective of Goal 1 for Apache Trout monitoring per 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 ($\widehat{N}$)(see Figure 2). Since precision can be controlled by monitoring a certain number of sample units (reaches, or sites) given the site-to-site variance in trout abundances typically observed, one can use past monitoring data to assess the number of reaches (sample size) needed to meet the precision levels stated in the monitoring Goal 1, Objective 1.

Objectives

1. Evaluate how the number of sample units (reaches) sampled and proportion of total stream length sampled influence the relative precision of stream-wide estimates of adult Apache Trout abundance using data collected under the 2017 Plan (Dauwalter et al. 2017).
2. Evaluate if the relative precision of stream-wide estimates of adult Apache Trout abundance is influenced by the proportion of sample units occupied by adult Apache Trout using data collected under the 2017 Plan (Dauwalter et al. 2017).
3. Determine the number of sample units (reaches) needed to achieve an 80% confidence interval that is 40% or less of the streamwide adult Apache Trout abundance estimates using data collected under the 2017 Plan (Dauwalter et al. 2017).

Methods

We used Apache Trout monitoring data collected from 2016 to 2023 under the 2017 Plan (Dauwalter et al. 2017) to evaluate the effects of sample size (number of sample units [reaches]) on the relative precision of stream-wide estimates of adult Apache Trout (≥130-mm TL) abundance. Briefly, Apache Trout have been monitored since 2016 using the first version of this plan at 36 population-years whereby sample units (reaches, or sites) were sampled systematically and the number of sites (sample size) was equivalent to sampling approximately 20% of the sample frame (habitat extent)(Dauwalter et al. 2017). Three pass removal (depletion) electrofishing was conducted at each sample unit (100-m reach), and abundance estimates for adult Apache Trout were made at each sample unit using the Burnham estimator in the FSA package of Program R (Ogle 2017).

Streamwide estimates of adult Apache Trout abundance ($\widehat{N}$) 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, successive differences variance estimator, number of reaches sampled, and the finite population correction described above (section 3c. Computing the Monitoring Metrics) and in 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 multiplied by 100 (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, the proportion of stream sampled (across all reaches), and proportion of sample units occupied (% occupancy). Significance of correlations was assessed at α = 0.10.

Data collected from 2016 to 2023 Apache Trout monitoring efforts were also used to understand sample size requirements (and % of habitat sampled) needed to achieve the precision objective for adult Apache Trout abundance (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 relative precision objectives for adult Apache Trout abundance estimates based on the number of sampling units available in the sampling frame (*N*), s^2 is the successive differences variance in abundance between adjacent sample units (reach to reach) (Scheaffer et al. 2012, page 238), and *D* is:

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

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 Type I error rate divided by 2 for each side of the confidence interval to match the total α = 0.2 for an 80% confidence interval that encompasses both sides of the estimate ($\widehat{N}$). 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) on either side 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 assuming reach length is the same as in past monitoring.

Results and Discussion

The 36 population-years of Apache Trout monitoring data collected from 2016 to 2023 under the first version of this plan are in Table 2B (also Figure 1; Figure 8B). Median capture probabilities of adult Apache Trout in sample units were typically above 0.7 (Figure 9B). All populations were monitored using a systematic sample, and most with sample units (reaches) being 100-m in length; exceptions include Smith Creek in 2018 and 2023 where sample units were 50-m in length with the exception of one longer reach in 2018, and Ord Creek in 2019 where reaches were 200 m in length (Table 2B). Monitoring was typically based on sampling 20% of the habitat and resulted in 0.5-km spacing of sample units in the systematic sample (Figure 10B). The shortest habitat extent sampled and used for extrapolation was 0.65-km in Smith Creek in 2018 and 2023, whereas Boggy/Lofer was 22.1-km (Table 2B). Estimates of $\widehat{N}$ ranged from 0 adult Apache Trout (≥130-mm TL) in several streams and up to 2,289 adult Apache Trout in the East Fork White River in 2021 (Figure 11B). The 80% confidence intervals ranged from 0 individuals in in several streams where no Apache Trout were sampled, up to 465 individuals in Boggy/Lofer in 2022 (Figure 11B). Relative precision (80% CI / $\widehat{N}$) ranged from 0% in Smith Creek 2018 up to 264% in Mineral Creek in 2017 (Table 2B).

Spearman rank correlations revealed that the relative precision of $\widehat{N}$ was significantly negatively correlated with the number of reaches sampled as expected (top panel of Figure 12B). The significant negative correlation 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 Coon (2022), Coyote (2018), Firebox (2012), Mineral (2017), South Fork Little Colorado (2017), and others (see Figure 12B and Table 2B). The relative precision of streamwide estimates of adult Apache Trout abundance was strongly and negatively correlated to the proportion of sample units occupied by adult Apache Trout, suggesting that patchy populations, typically those of small size, result in high reach-to-reach variance in adult Apache Trout abundance and wide confidence intervals (Figure 13B).

The sample sizes needed to reach precision objectives for past monitoring years ranged from 3 (East Fork White 2021 and Ord 2019) to 97 (South Fork Little Colorado 2017; Table 2B). When sample sizes needed were converted to percent of habitat that needs sampling, percentage of habitat needed ranged from 3% (East Fork White 2021) to 92% (Firebox 2018 and Coon 2022). Streams requiring a high percentage of habitat to be sampled only collected Apache Trout at a small percentage of sample units (reaches) during that monitoring year, resulting in considerable uncertainty in streamwide estimates of abundance (Figure 13B). Patchy distributions as revealed by a low percent occupancy suggests that sampling 20% of the habitat extent is insufficient to meeting monitoring Goal 1, Objective 1 for small patchy populations (Figure 14B). 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 of most populations except in streams where abundance is low and, thus, the percent occupancy is low leading to high variance (Table 2B; see below); in these situations an adaptive sampling approach (Box 2) may require less effort and resources than a high and logistically unfeasible sample size as suggested in Table 2B.

Table 2B. Monitoring survey characteristics and stream-wide estimates of Apache Trout abundance ($\widehat{N}$; ≥130-mm TL) per population and year. Population extent (km), number of sites sampled (n), mean site length (m), mean abundance of Apache Trout ≥130-mm TL per site ($\overline{y}$), successive differences standard deviation in abundance across sites (s), proportion of extent surveyed, stream-wide estimates of Apache trout ≥130-mm TL abundance ($\widehat{N}$), lower and upper 80% confidence limits, and relative precision (80% confidence interval / ($\widehat{N}$); %), estimated of number of sites required for 80% CI to be within 40% of $\widehat{N}$ (Required n), and proportion of habitat required 80% CI to be within 40% of $\widehat{N}$ (Required %).

Population	Year	Extent (km)[a]	Sites (n)	Mean Site Length (m)	Prop. Sampled	p	$\overline{y}$	s[b]	$\widehat{N}$	Lower 80% CL	Upper 80% CL	80%CI/ $\widehat{N}$ (%)	Req n	Req %
Aspen	2019	15.2	19	100.0	0.12	0.95	5.58	3.62	848	691	1005	36.9	17	11
Big Bonito Upper	2017	3.8	5	100.0	0.13	1.00	16.00	5.15	608	488	728	39.6	5	13
Boggy/Lofer	2022	22.1	46	100.0	0.21	0.37	3.54	5.45	783	578	989	52.5	69	31
Coon	2022	4.0	9	100.0	0.22	0.11	1.11	3.54	44	10	102	258.6	37	92
Coyote	2018	5.8	13	100.0	0.22	0.08	0.15	0.58	9	2	20	248.3	53	91
Coyote	2023	5.8	13	100.0	0.22	0.23	0.77	1.78	45	11	79	152.7	47	81
Crooked	2016	8.6	10	100.0	0.11	0.80	3.20	2.62	282	187	376	67.0	23	26
Crooked	2023	8.6	17	100.0	0.19	1.00	16.41	5.07	1444	1315	1574	17.9	4	5
Deep	2016	16.4	10	100.0	0.06	0.90	2.80	2.62	459	278	640	78.8	33	20
Deep	2021	16.4	32	100.0	0.20	0.84	3.50	2.60	574	486	662	30.8	21	13
E Fk Little Colorado	2020	10.9	19	98.9	0.17	0.32	1.37	2.04	151	88	213	82.7	52	47
East Fork White	2021	9.2	17	99.7	0.18	1.00	24.88	6.94	2289	2103	2476	16.3	3	3
Elk Canyon	2018	6.1	12	100.0	0.20	0.25	0.25	0.30	15	9	22	84.9	32	52
Elk Canyon	2023	6.1	13	100.0	0.21	0.38	2.92	2.50	178	128	229	56.8	22	36
Firebox	2018	6.1	13	100.0	0.21	0.08	0.08	0.29	5	1	11	248.9	56	92
Firebox	2023	6.1	13	100.0	0.21	0.00	0.00	0.00	0	0	0	--	--	--
Flash	2019	12.8	24	100.0	0.19	0.58	1.38	1.82	176	119	233	64.3	48	38
Little Bonito	2019	16.4	23	91.3	0.13	0.74	2.35	2.63	423	301	544	57.5	42	23
Little Diamond	2023	10.8	23	100.0	0.21	0.00	0.00	0.00	0	0	0	--	--	--
Marshall Butte	2017	6.2	7	100.0	0.11	0.43	2.43	3.03	151	56	245	125.6	35	56
Mineral	2017	5.0	9	100.0	0.18	0.11	0.11	0.35	6	1	13	264.3	45	90
Mineral	2022	5.0	10	100.0	0.20	0.00	0.00	0.00	0	0	0	--	--	--
Moon	2023	8.7	18	100.0	0.21	0.28	0.39	0.59	34	19	48	85.5	48	55
Ord	2019	6.7	11	200.0	0.33	1.00	19.64	4.89	668	611	724	16.8	3	9
Paradise	2018	7.4	13	100.0	0.18	0.15	0.15	0.41	11	2	22	179.8	60	81
Paradise	2023	7.4	14	100.0	0.19	0.57	2.21	1.62	164	125	203	47.3	18	24
Rock	2022	14.8	32	100.0	0.22	0.22	1.16	1.33	171	131	211	47.2	41	28
S Fk Little Colorado	2017	11.0	18	100.0	0.16	0.06	0.17	0.73	18	3	41	250.8	97	88
S Fk Little Colorado	2021	11.0	23	100.0	0.21	0.43	4.96	5.64	545	394	697	55.6	37	34
Smith	2018	0.8	12	54.2	0.81	0.25	0.25	0.30	4	3	5	42.1	12	80
Smith	2023	0.8	5	50.0	0.31	0.00	0.00	0.00	0	0	0	--	--	--
Soldier Springs	2017	3.0	3	100.0	0.10	1.00	8.33	4.72	250	123	377	101.6	13	43
Soldier Springs	2021	3.0	6	100.0	0.20	1.00	18.67	7.77	560	437	683	43.8	7	23
Thompson Upper	2019	2.1	3	100.0	0.14	0.00	0.00	0.00	0	0	0	--	--	--
W Fk Little Colorado	2018	16.3	24	100.0	0.15	0.50	2.08	1.22	340	290	389	29	14	9
Wohlenberg	2022	5.7	12	100.0	0.21	0.50	2.25	1.85	128	92	165	57.1	20	35

[a]Extent corrected for stream length bias (see Appendix A; Dauwalter et al. 2022).

[b]New use of the successive differences standard deviation may result in slight differences in standard deviation, confidence intervals, confidence interval precision, required sample size, and required percent of habitat than those reported in the 2017 Plan and the 2022 Species Status Assessment for the Apache Trout which used a standard deviation for a random sample (Scheaffer et al. 2012; USFWS 2022).

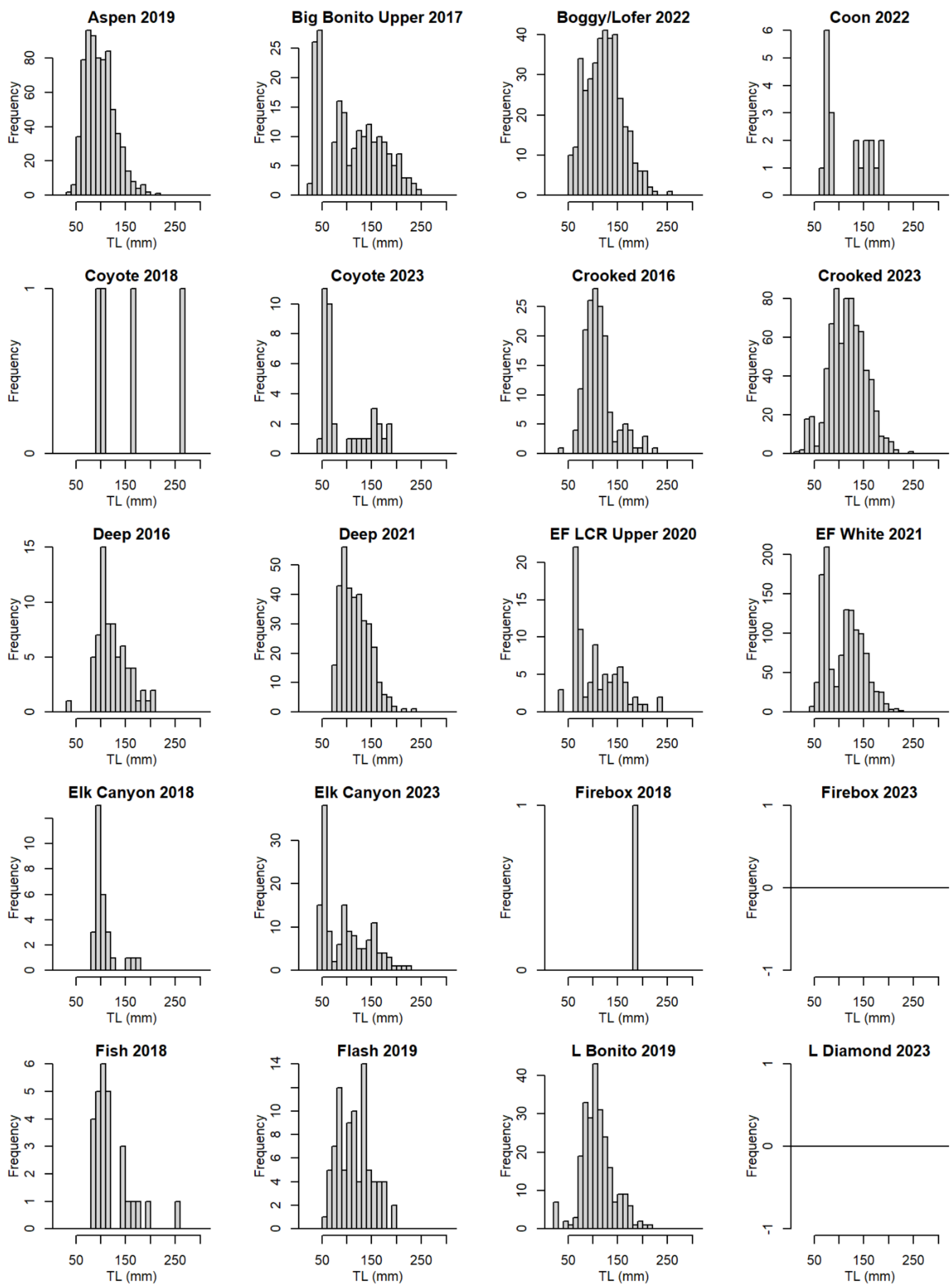


Figure 8B: Length frequency histograms for Apache Trout by population and year. Aspen 2018 monitoring was terminated due to inclement weather, and West Fork Black Upper 2019 was only sampled on the Fort Apache Indian Reservation. Data are from surveys using systematic sampling at 100-m sites according to the 2017 Plan (Dauwalter et al. 2017)

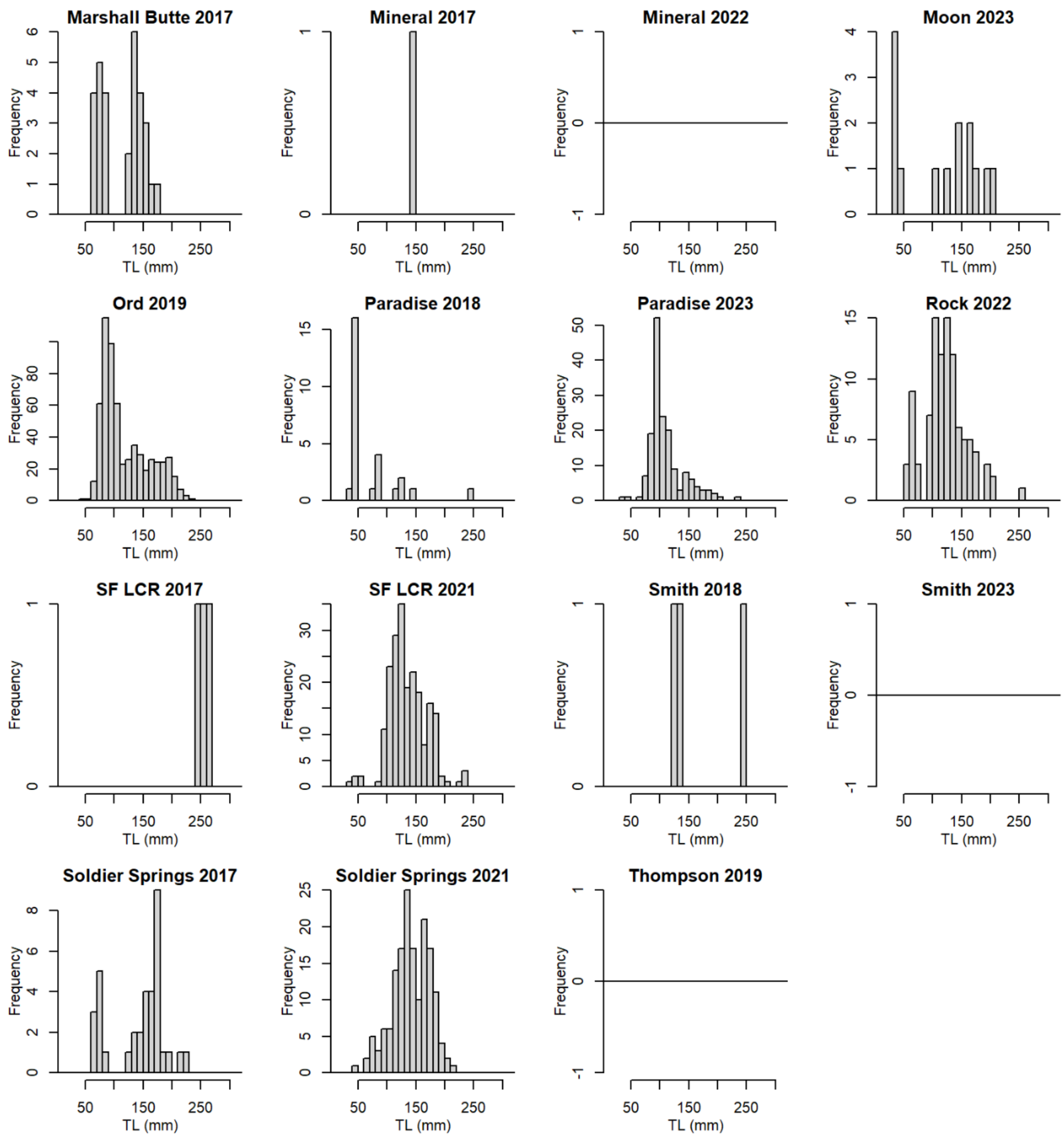


Figure 8B: Continued.

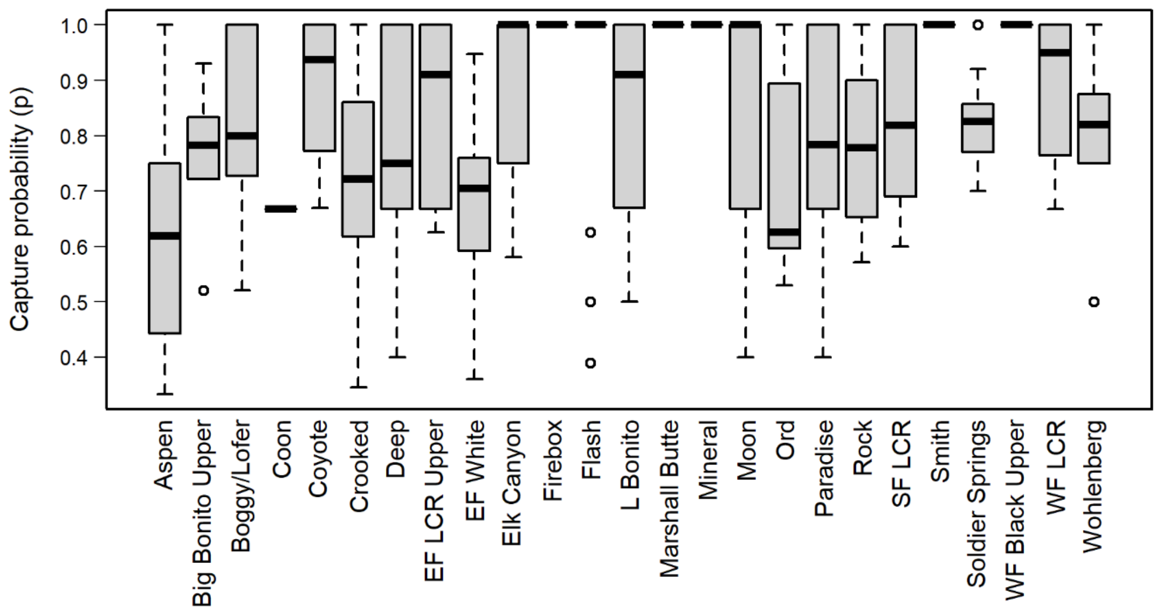
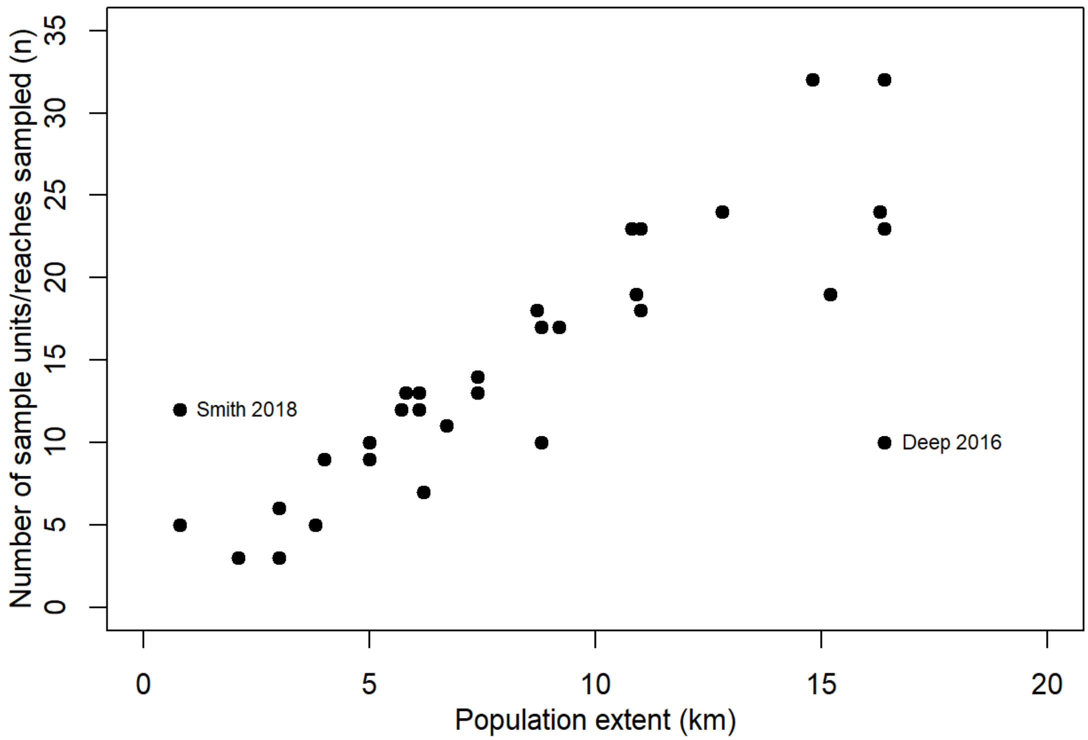


Figure 9B. Box plots of capture probability of ≥130-mm TL Apache Trout estimated using a Burnham removal estimator for 100-m monitoring sites sampled using multiple-pass backpack electrofishing. Data are from monitoring surveys using systematic sampling at 100-m sites according to the 2017 Apache Trout monitoring plan (Dauwalter et al. 2017).



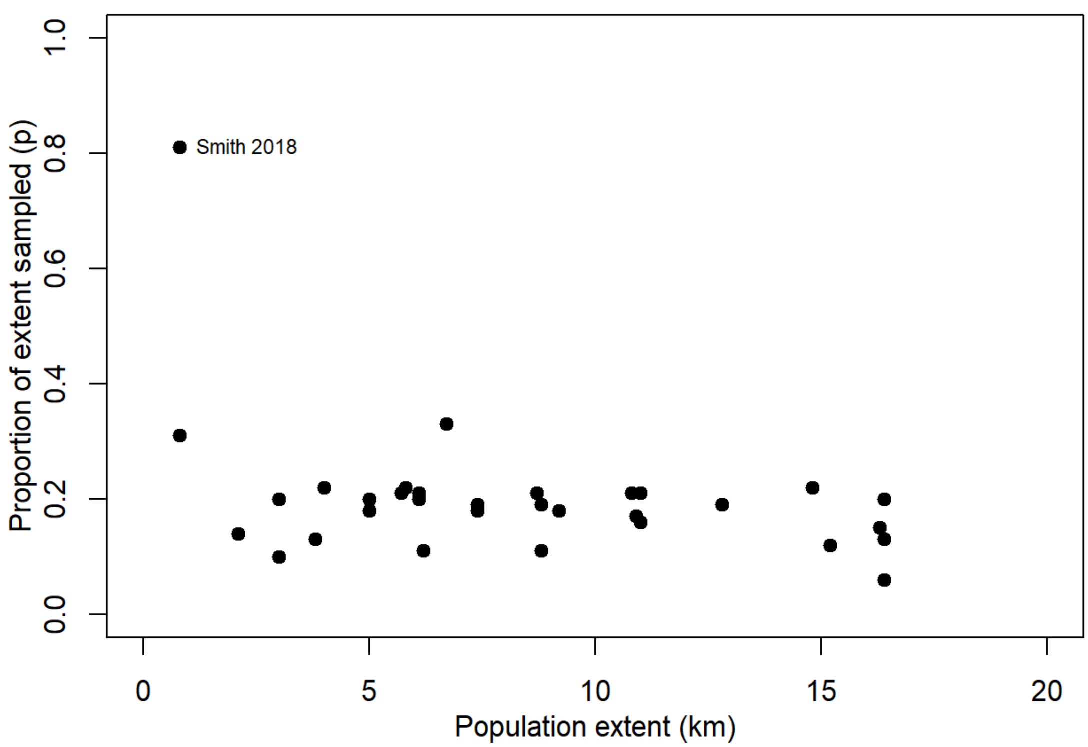


Figure 10B: Associations between population extent and sampling effort during monitoring surveys, expressed as number of sites sampled (top panel) and proportion of extent sampled (bottom panel). Data are from surveys using systematic sampling at 100-m sites according to the 2017 Apache Trout monitoring plan (Dauwalter et al. 2017).

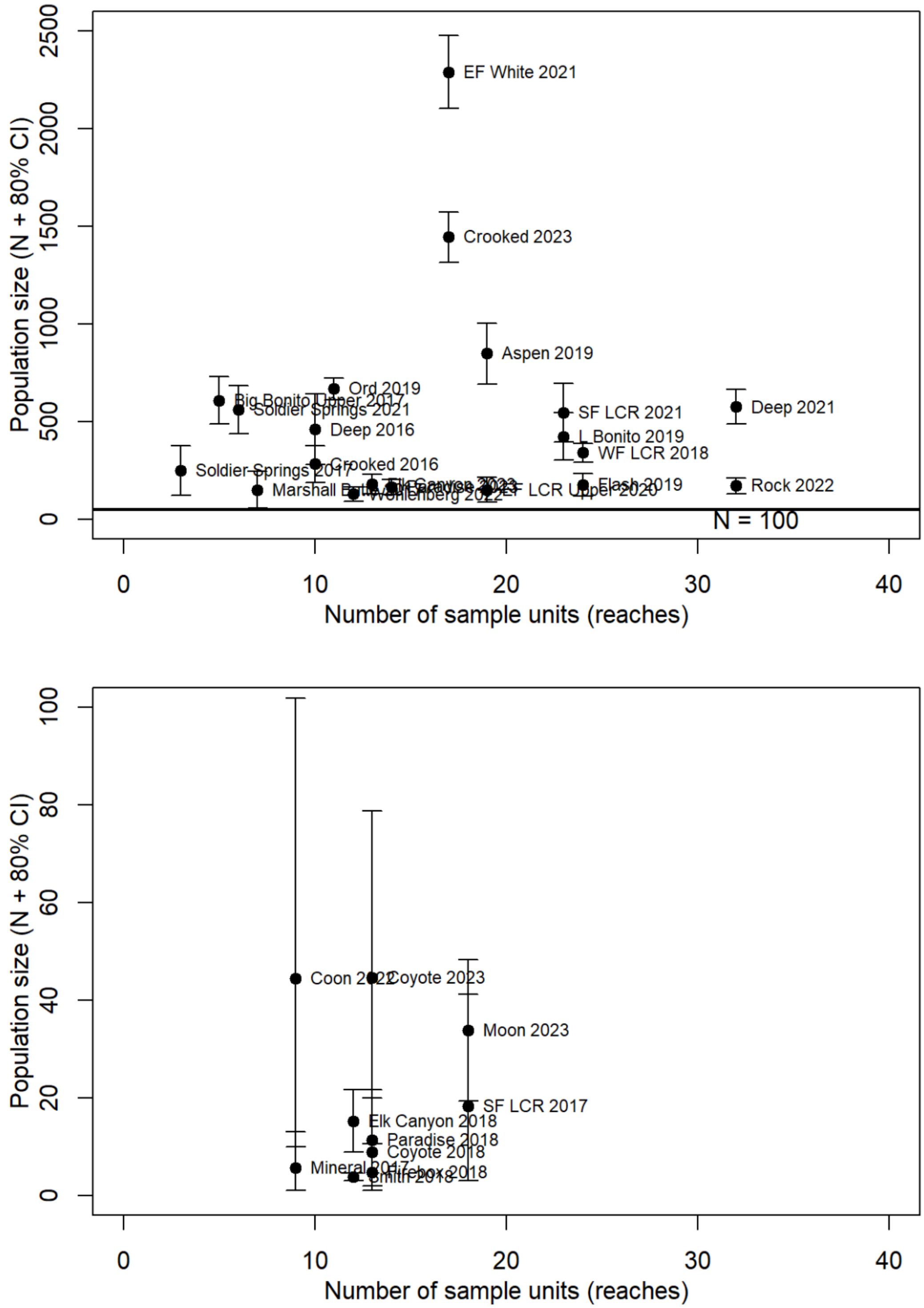
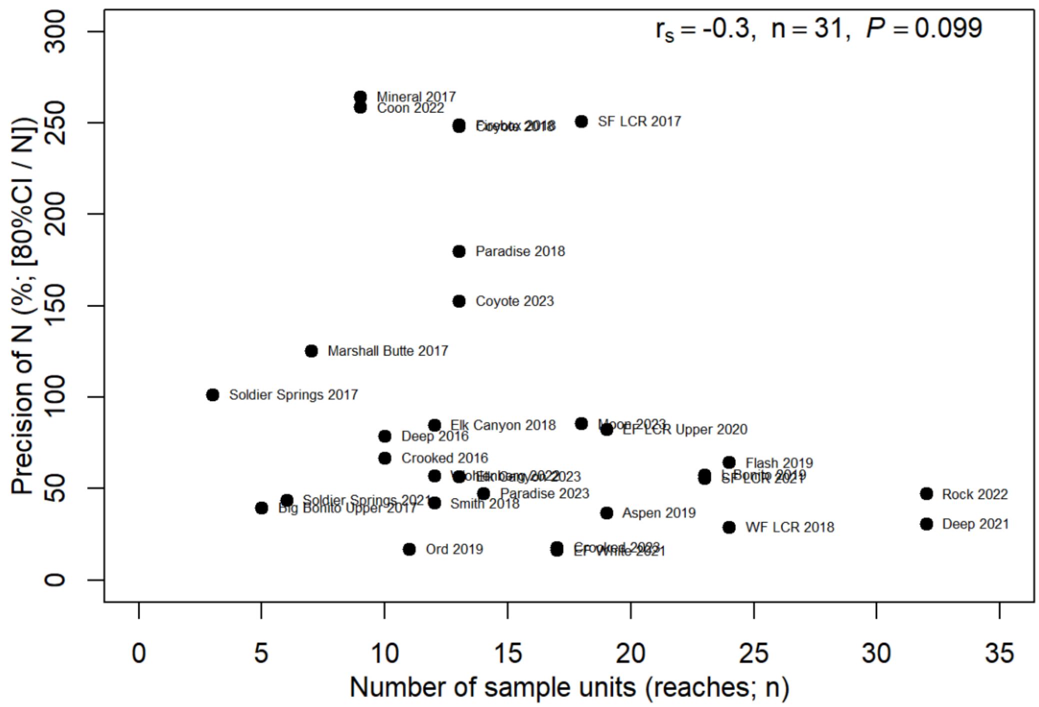


Figure 11B: Population size ($\widehat{N}$) versus number of sites sampled for adult (≥130-mm TL) Apache Trout populations surveyed in different years. Top panel shows populations with $\widehat{N}$ greater than 100 individuals. Bottom panel shows populations with $\widehat{N}$ less than 100 individuals. Data are from surveys using systematic sampling at 100-m sites according to the 2017 Apache Trout monitoring plan (Dauwalter et al. 2017).



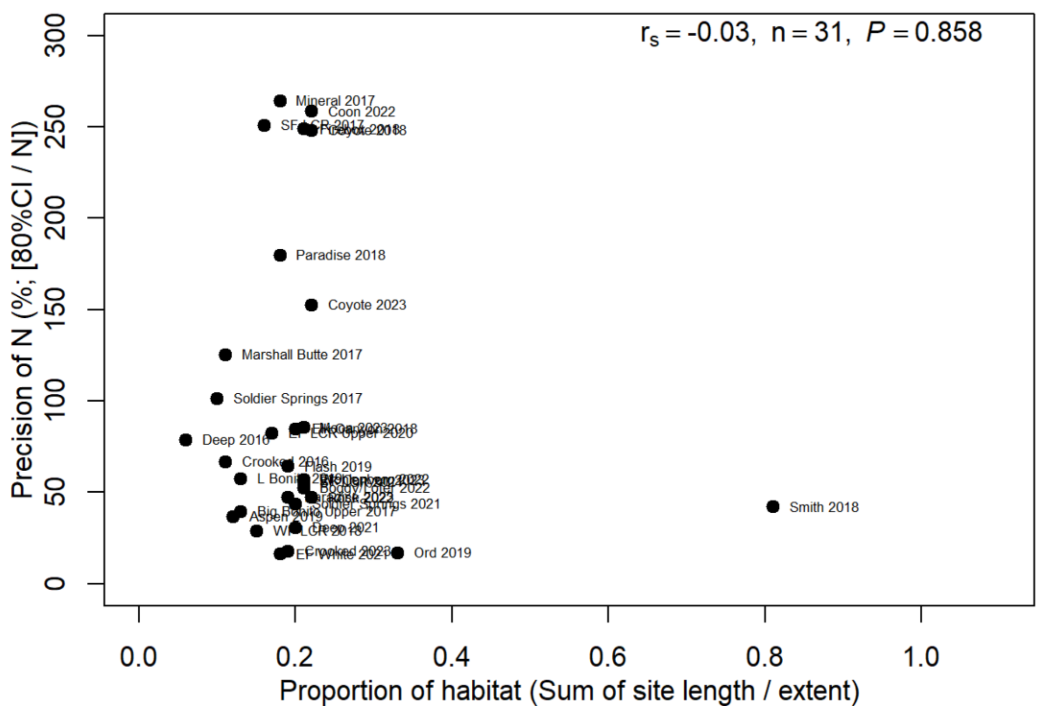


Figure 12B: Precision of adult (≥130-mm TL) Apache Trout population size estimate, expressed as ratio of 80% confidence interval divided by population size estimate ($\widehat{N}$), versus number of sample units (top panel) and proportion of habitat ([sum of survey site lengths / total habitat extent]; bottom panel) sampled for Apache Trout populations surveyed from 2016 to 2023. Data are from surveys using systematic sampling at 100-m reaches according to the 2017 Plan (Dauwalter et al. 2017).

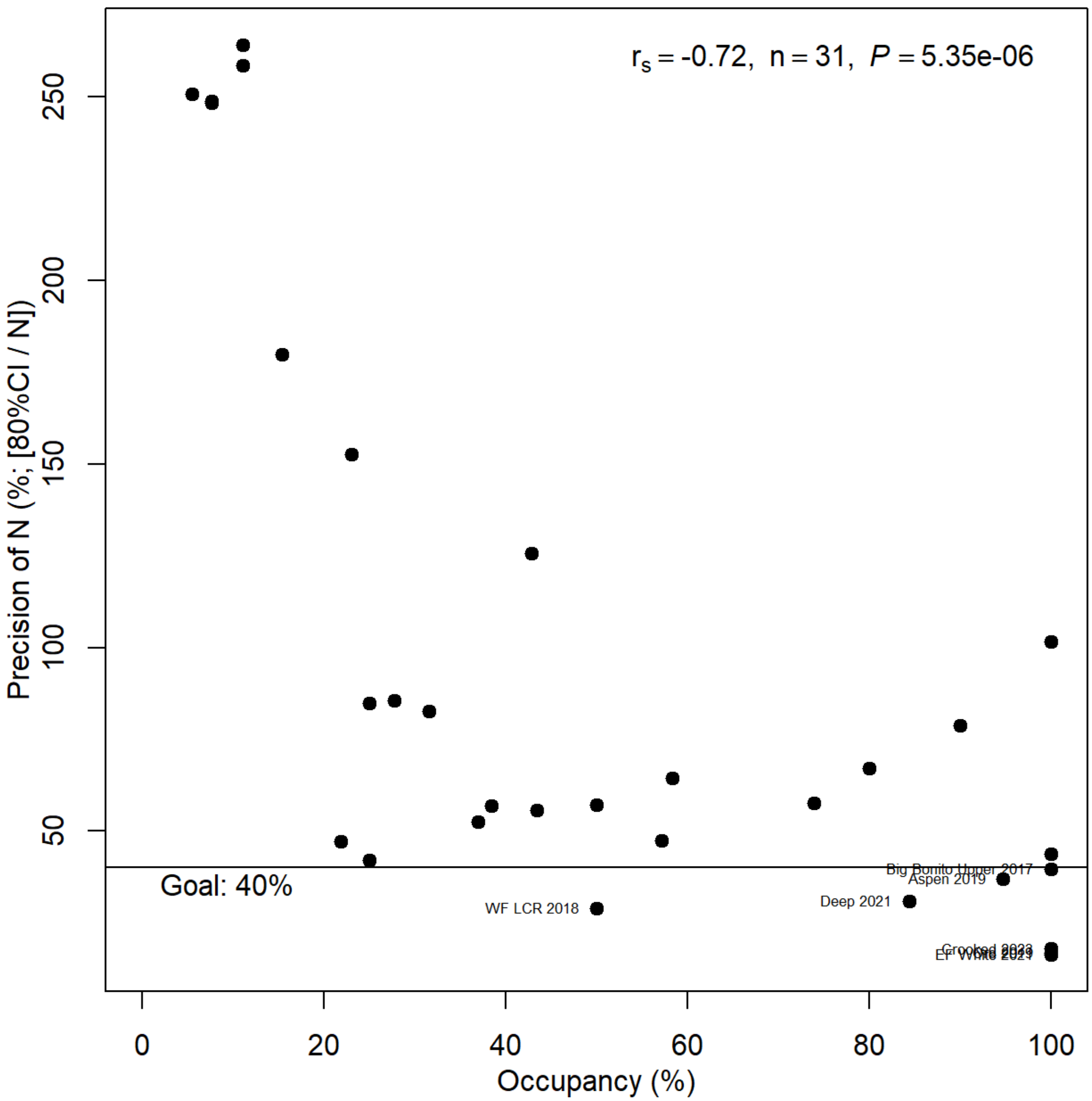


Figure 13B. Precision of adult (≥130-mm TL) Apache Trout population size estimate, expressed as ratio of 80% confidence interval divided by population size estimate ($\widehat{N}$), versus percent of sample units (reaches) occupied by adult Apache Trout for Apache Trout populations surveyed from 2016 to 2023. Data are from surveys using systematic sampling at 100-m reaches according to the 2017 Plan (Dauwalter et al. 2017).

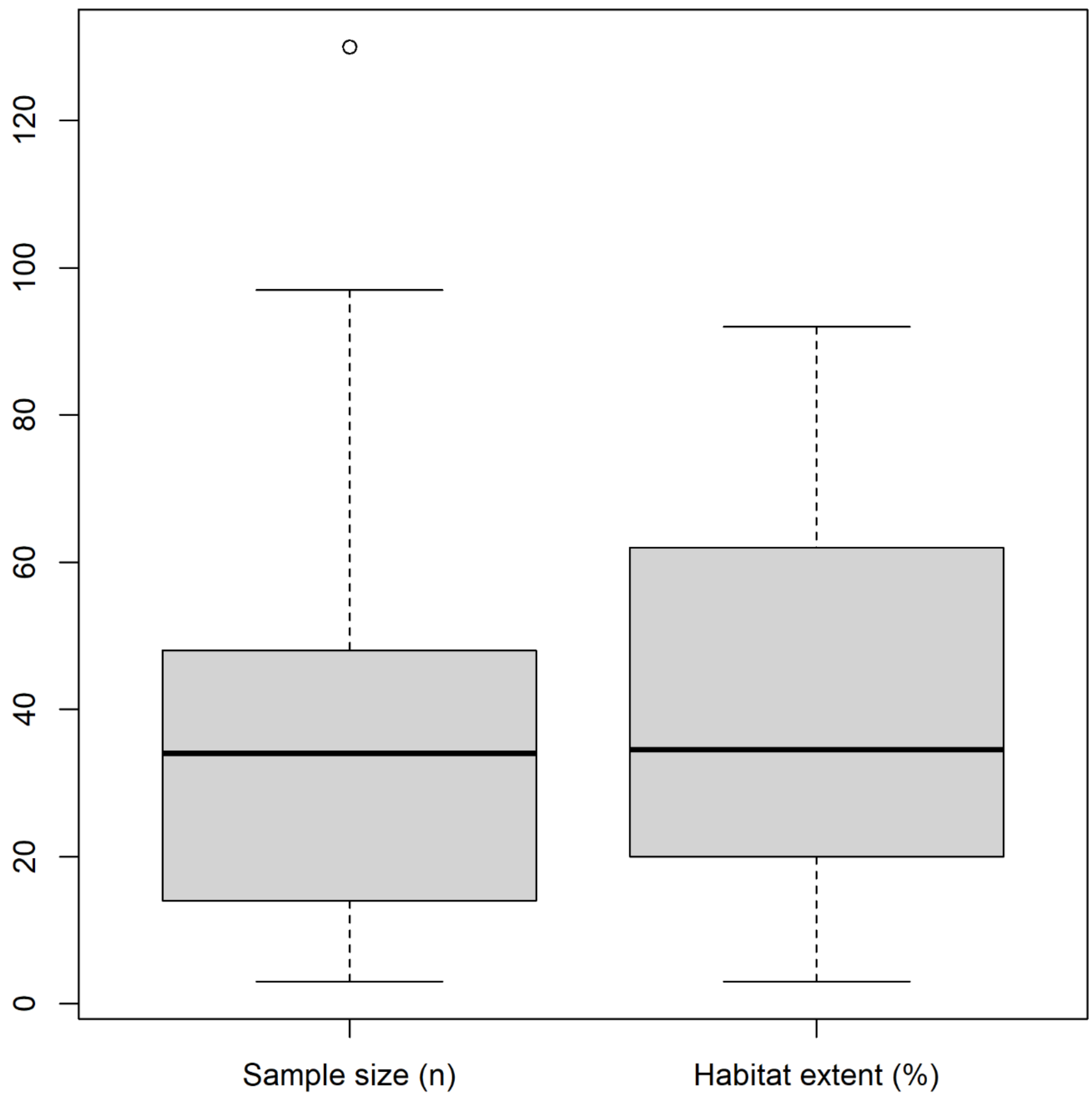


Figure 14B. Box plots showing the sample size (number of sample units [reaches]) required, and percent of habitat extent required, to be sampled to meet the monitoring Goal 1, Objective 1 of relative precision of streamwide estimates of adult Apache Trout (≥130-mm TL) abundance having a ratio of 80% confidence interval divided by population size estimate ($\widehat{N}$) less than or equal to 40% using 28 population-years of monitoring data from 2016 to 2023 (see Table 2B). Data are from surveys using systematic sampling at [typically] 100-m reaches according to the 2017 Plan (Dauwalter et al. 2017).

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 be 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* ($\hat{N}_t$), thus giving more weight to the trends of larger, more abundant populations.

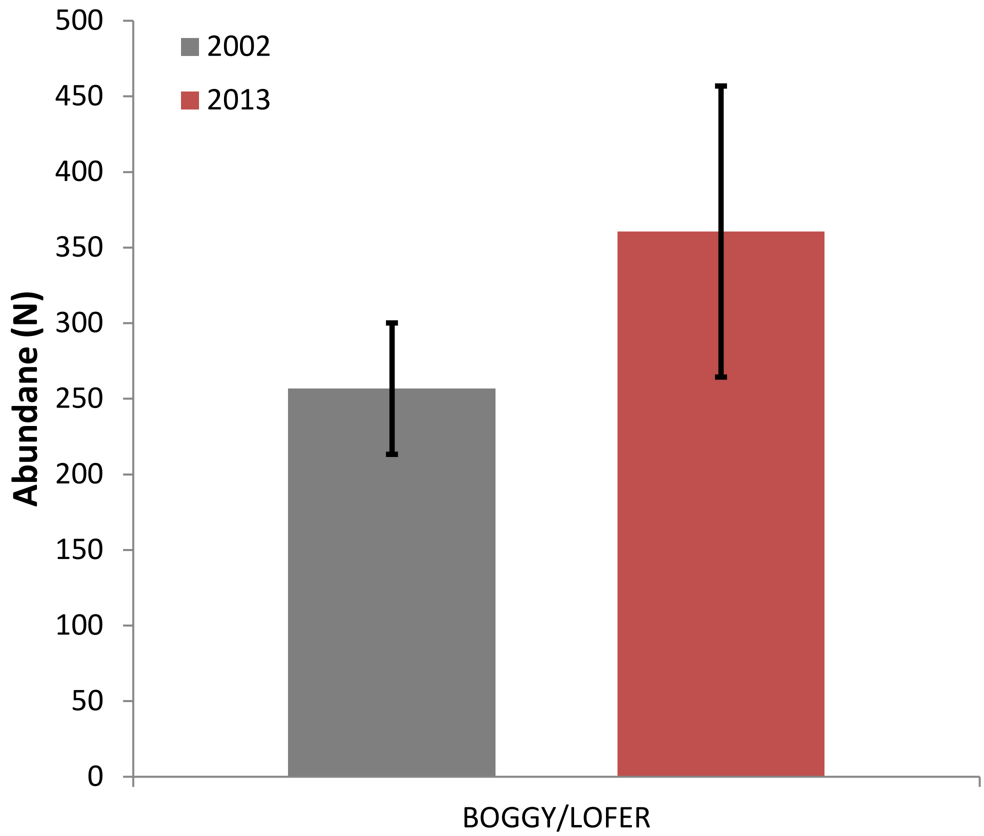


Figure 15C. Examples of population trend estimates (*r*) between two time periods for an Apache Trout stream. Abundance estimates of 361 in 2013 and 257 in 2002 yield an estimate of *r* = 1.031 or 3.1% increase in abundance per year.