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MANAGEMENT BRIEF

Digital Hydrography Underestimates Stream Length and Leads to Underestimates of Trout Population Size

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Abstract
Stream length is measured for many fisheries management applications. Characteristics of populations and habitats measured at field sites are commonly generalized to unsampled areas using estimates of stream length or stream network length. There are many ways to measure stream length, but map-based stream length measurements are commonly used in fisheries applications even though they are known to be biased. We evaluated how length of headwater streams in Arizona may be underestimated by the National Hydrography Dataset and how that bias influences streamwide abundance estimates for adult Apache Trout *Oncorhynchus apache*. As expected, stream lengths measured using National Hydrography Dataset flowlines underestimated true length revealed by National Agricultural Imagery Program imagery on average 11.1% (SD = 4.1%), and this bias was higher in meadow versus forested habitats. The observed bias led to streamwide estimates of adult Apache Trout abundance that were only 88% on average (SD = 5%) of those made with more realistic imagery-based stream measurements. As we have shown, high-resolution imagery, now widely available, can be used to assess and quantify stream length bias, and we conclude that it is important to assess whether this bias has the potential to negatively impact important fishery management decisions.

Obtaining accurate measurements of stream length is a prerequisite for many fisheries research, assessment, and monitoring applications. Sites of a given length (e.g., 50, 100, or 200 m), measured along the thalweg, are typically established for habitat or fish surveys (Bain and Stevenson 1999). Stream length occupied by a fish population, often referred to as patch size, is commonly used as a surrogate or rule of thumb to gauge population persistence or viability (Hilderbrand and Kershner 2000; Haak and Williams 2012; Al-Chokhachy et al. 2018). Accurate representations of entire stream networks, including the total length of streams, are also needed to establish probabilistic sampling designs to draw inferences on stream condition over large geographic areas, such as for the U.S.

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Environmental Protection Agency's Wadeable Streams Assessment (USEPA 2020). Discrepancies between actual and estimated stream length can, for example, lead to biased estimates of the status of fish populations and species when field data are extrapolated to entire streams or stream networks (Shepard et al. 2005; Meyer et al. 2014). It can also lead to increased ambiguity in applying the narrower definition of Waters of the United States rule put forth by the U.S. Environmental Protection Agency in 2018 (Colvin et al. 2019).

Stream length can be measured in a variety of ways. Sites used for fish and habitat surveys on the scale of 10–100 m are often measured along the thalweg using a tape measure (Bain and Stevenson 1999), but sites longer than 100 m can be difficult to measure accurately with a tape measure due to the difficulty of securing the tape along the thalweg on sinuous streams over longer distances. Lengths can be measured by computers using GPS-based maps of field sites (Dauwalter et al. 2006), or laser rangefinders can be used when only straight-line distance along a stream valley is needed. However, stream length measurements over 1 km are often based on streams as defined on topographic maps or in geospatial hydrography data sets, such as the National Hydrography Dataset (NHD) (USGS 2004).

The NHD is a digital vector geospatial data set of surface water features for the United States comprised of a data model with an underlying database, feature classes, attribute tables, and metadata (https://www.usgs.gov/core-science-systems/ngp/national-hydrography) (USGS 2004). The NHD has used two commonly used data models: one that represents 1:24,000 scale hydrography and another based on 1:100,000 scale hydrography. Flowlines are features that represent streams and other linear features, and streams are commonly based on digitization of topographic maps published by the U.S. Geological Survey and reflect the original map scale (1:24,000 or 1:100,000). Both topographic maps and the NHD have been widely used in fisheries applications (Hughes et al. 2006), but it is well known that the NHD underrepresents stream networks. For example, Elmore et al. (2013) found that the NHD underestimated stream density by 250%, particularly in urban areas and for headwater streams that can be buried by urban development or be too small to meet the minimum mapping size at the time of map generation. Topographic maps provide generalized representations of steams that smooth out stream sinuosity because of their scale and, therefore, underestimate stream length for sinuous segments (Morisawa 1957; Leopold et al. 1964).

Our goal was to understand how stream length may be underestimated in the NHD and thereby influence streamwide estimates of abundance for adult Apache Trout *Oncorhynchus apache*. It is common to use fish abundance data collected at sites and extrapolate them to the entire occupied length of stream using appropriate statistical estimators, as is the case for a monitoring plan recently developed for Apache Trout (Dauwalter et al. 2017a), and stream length is often measured using the NHD or paper maps for this purpose (Hankin and Reeves 1988; Cook et al. 2010; Zeigler et al. 2019). The Apache Trout is listed as threatened under the Endangered Species Act (USFWS 2009), and populations are often isolated above conservation barriers in small headwater streams to protect them from hybridization and competition with, and predation by, nonnative salmonids (Carmichael et al. 1995; Avenetti et al. 2006). Our objectives were twofold: (1) compare stream lengths measured using NHD 1:24,000-scale hydrography versus lengths derived from high-resolution aerial imagery, and (2) evaluate bias due to NHD-based stream length measurements in streamwide estimates of adult (≥130 mm TL) Apache Trout abundance, where field data are extrapolated to the stream length occupied. We expected NHD stream lengths for Apache Trout populations to be underestimated as shown for other geographies (Elmore et al. 2013), and because streamwide abundances are scaled to stream length occupied per a recently developed monitoring protocol for the species (Dauwalter et al. 2017a), we expect population abundances to be underestimated to a similar degree.

METHODS

We compared stream lengths and streamwide estimates of Apache Trout abundance on 12 and 11 streams, respectively, in the White Mountains region of eastern Arizona (Figure 1). Most of the study streams' headwaters are located on Mount Baldy, and geology shifts from felsic to mafic formations of volcanic origin as elevation decreases, but in some cases, streams course through glacial alluvial valleys (Long et al. 2006). Vegetation also shifts from spruce *Picea* spp. and fir *Abies* spp. to ponderosa pine *Pinus ponderosa* or mixed conifer as elevation decreases. Riparian vegetation is typically ponderosa pine or mixed conifer and willow *Salix* spp., alder *Aldus* spp., red-osier dogwood *Cornus stolonifera*, and other shrub species. Meadows occur from valley fill and other geomorphic processes and are often dominated by grasses, and streamside vegetation exists as grass, woody shrubs, or both but typically lacks large coniferous trees (Clarkson and Wilson 1995; Long et al. 2006). Clarkson and Wilson (1995) give a detailed description of upland and riparian vegetation of Apache Trout streams.

The extent of habitat occupied by an Apache Trout population was defined for each study stream. The upstream and downstream extent was typically defined using information on man-made conservation barriers or natural barriers such as waterfalls (typically downstream), the presence of perennial water, past Apache Trout survey data, and experience of biologists familiar with the fish-

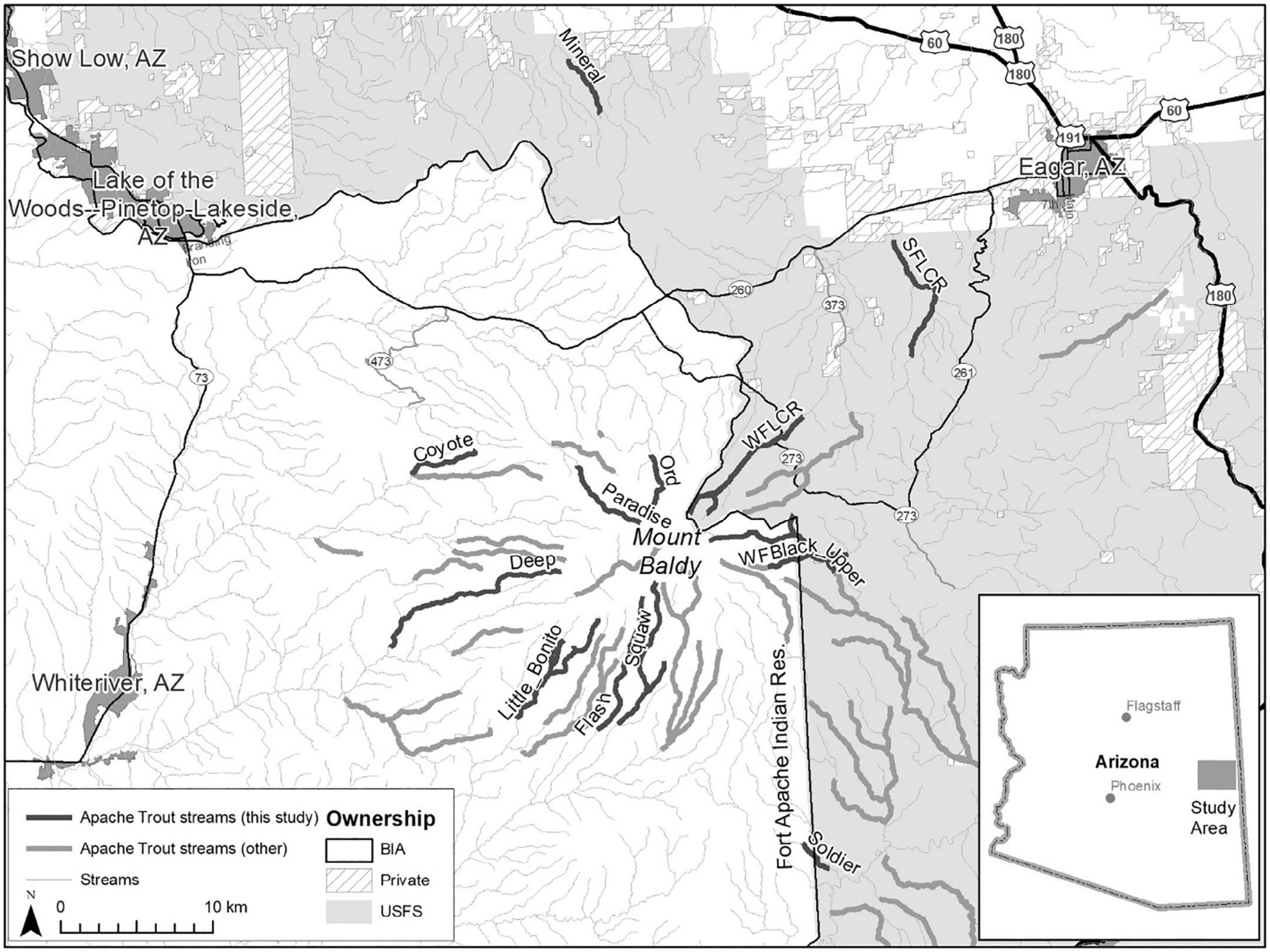

FIGURE 1. Apache Trout streams in the White Mountains of east-central Arizona. Dark and labeled Apache Trout streams were the focus of this study. BIA, Bureau of Indian Affairs; Res, Reservation; SFLCR, South Fork Little Colorado River; USFS, U.S. Forest Service; WF, West Fork; WFLCR, West Fork Little Colorado River.

bearing nature of Apache Trout streams. This extent was used to establish a systematic sampling design to monitor each Apache Trout population (Dauwalter et al. 2017a).

Stream length.— Upstream and downstream extents were used to trim the NHD high-resolution hydrography (1:24,000 scale) to occupied stream length for each population. In one case, the NHD underestimated headwater, fish-bearing extent and we digitized an additional stream segment (2.1 km) based on past fish surveys showing the presence of fish (West Fork Black River). The NHD was trimmed and dissolved to compute an NHD-based stream length. To approximate true stream length, 2017 National Agricultural Imagery Program (NAIP) imagery (1-m resolution) available for the study streams was used to digitize all stream segments per stream at 1:500 to 1:1,000 scale within meadow and forested reaches where the stream was visible (not hidden by canopy cover) in the imagery (Figure 2). Streams in meadows were generally visible, but forested reaches were often not due to coniferous canopy cover; segments not visible due to canopy cover were never digitized.

Segment lengths derived from the NHD and NAIP were compared for meadow and forested reaches. For these comparisons, the NHD was trimmed at the same upstream and downstream extent as that of the segment digitized using NAIP imagery. The average ratio of NHD-based length to imagery-based length (ratio = NHD kilometers/NAIP kilometers) was computed for each land cover type (meadow or forested) by stream, and the difference in ratio between land cover type across streams was compared using a Welch t-test for unequal variances at $\alpha = 0.10$; a paired t-test could not be used because both

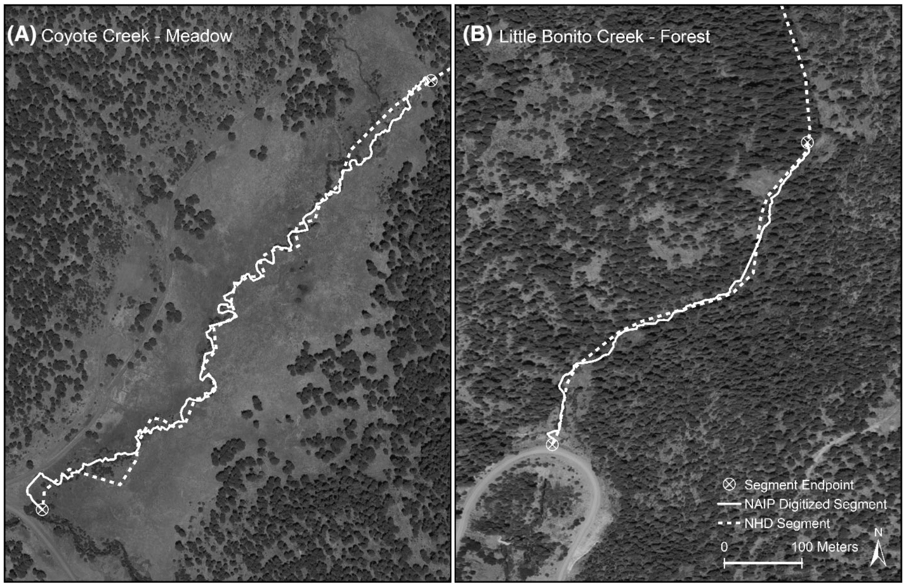

FIGURE 2. Example of stream segments from the NHD and digitized from 2017 NAIP imagery for **(A)** a meadow segment in Coyote Creek and **(B)** a forested section in Little Bonito Creek, Arizona.

land cover types were not always digitized due to visibility or were not always present on each stream. A stream-specific ratio of NHD/NAIP lengths was estimated using a weighted average, with the weight representing the total length of stream (NHD) that was forested or meadow along the NHD stream segment for each stream as observed in the aerial imagery. If a land cover type was present but not digitized (because no segments were visible), then the mean ratio for that land cover type across all streams was used for that stream. The stream-specific ratio was used to adjust the NHD-based stream lengths and to obtain an adjusted adult Apache Trout abundance estimate as described below.

Streamwide abundance estimates.— Each stream was sampled for Apache Trout from 2016 to 2019 using a systematic sampling design and backpack electrofishing within 100-m sites. Sites were typically established approximately every 0.5 km along the extent of habitat available to a population so that ~20% of habitat was sampled (0.1 km of every 0.5 km was sampled) (Dauwalter et al. 2017a). Monitoring sites were identified by starting at a random location within 0.5 km of the conservation barrier or other downstream extent landmark and then by identifying 0.1-km sites every 0.5 km along the stream thalweg using a tape measure or hip chain. For some streams in 2019, monitoring sites were established by measurement on aerial imagery on a computer desktop. The geographic coordinates were obtained for the downstream site boundary, and a GPS receiver was used to navigate to each site. Stream length bias was evaluated in the West Fork Black River because it is an important metapopulation (Williams and Carter 2009), but no surveys to estimate population size were conducted.

Adult Apache Trout (≥130 mm TL) were sampled within each monitoring site, isolated with 6.35-mm bar mesh block nets at the upstream and downstream site boundaries, using multiple-pass backpack electrofishing (200–500 V, 25–60 Hz) (Dunham et al. 2009). At least three passes were completed unless catch did not decline consistently across passes, which prompted additional passes to be made. Abundance estimates were made for adult Apache Trout, defined as individuals ≥130 mm TL (Harper 1976), using the removal function (Burnham method) in the FSA package in Program R (Ogle 2017).

Streamwide estimates of abundance were computed using the mean abundance of adult Apache Trout (≥130 mm TL) across sites (sampling units) that was then multiplied by the number of sampling units N_i available in the sampling frame for population i (N_i = occupied habitat extent, in kilometers, divided by 0.1-km survey site length). The estimated total abundance of adult Apache Trout for population i was computed as (from Scheaffer et al. 2012)

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

where $\hat{N}_i$ is the estimated abundance of adult Apache Trout for population i, N_i is the total number of sampling units (100-m sites) 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 (site) across all sample units j sampled in population i, y_{ij} is the number of adult Apache Trout in sample unit j in population i, and n_i is the number of sites (sampling units) sampled in population i.

The variance of $\hat{N}_i$ was computed as

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

where N_i and n_i are as defined above and s_i^2 is the variance in abundance of adult Apache Trout across all sites (sampling units) in population i. Recall that s_i^2 is computed as $s_i^2 = \frac{\Sigma\left(\bar{y}_i - y_{ij}\right)^2}{n_i - 1}$, with all terms as defined above. The $\left(\frac{N_i - n_i}{N_i}\right)$ term is a finite population correction 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; the 0.5-km systematic spacing and 100-m site length means that ~20% of occupied extent was typically sampled (i.e., 100 m out of every 500 m is sampled). To assess the effect of stream length based on the NHD versus NAIP imagery on streamwide abundance estimates, we computed streamwide estimates using estimates of stream length from the NHD or adjusted based on the stream-specific ratio of NHD length/NAIP imagery length; each effectively defines the sampling frame (N_i) as described above.

RESULTS

Stream Length

The NHD underestimated stream lengths observed in the NAIP imagery by 3.6% to 19.4% (Table 1). In total, 56 segments (27 forested and 29 meadow) were digitized from NAIP imagery across 12 streams (Figure 2), with an average of 4.7 segments (SD = 2.6) per stream. Based on the NHD, the percent of stream length flowing through meadow ranged from 0% to 55% (Table 1). There were 27 forested segments and 29 meadow segments that were digitized, and digitized lengths averaged 0.9 km (1 SD = 0.8) for meadows and 1.1 km (1 SD = 0.9) for forest (overall range from 0.1 to 4.2 km). The ratio of individual NHD lengths to NAIP digitized lengths was 0.87 on average (range = 0.55–1.06). Across the 12 streams, the average NHD/NAIP length ratio in meadow segments was less than in forested sections (t-test: $t = -3.69$, df = 11.9, $P = 0.003$); on average across streams, meadow segment ratios were 0.81 (1 SE = 0.02) and forested sections were 0.90 (SE = 0.01) (Figure 3). Length-weighted average of NHD/NAIP length ratios across the 12 populations was 0.89 (SD = 0.04) and ranged from 0.81 in Flash Creek to 0.96 in the South Fork Little Colorado River (Table 1).

Streamwide Abundance Estimates

The underestimation of stream length by the NHD resulted in streamwide abundance estimates being up to 19% less than those based on lengths adjusted using NAIP imagery that reflected the real length of streams (Table 2). The number of sites sampled per monitored population varied from 3 (Soldier Creek) to 24 (three populations), and the number of adult Apache Trout averaged from 0.11 Apache Trout per 100 m in Mineral Creek to 19.64 per 100 m in Ord Creek (Table 2). Streamwide estimates of abundance varied from 5 adult Apache Trout in Mineral Creek in 2017 to 764 in Squaw Creek in 2019 using NHD-based stream length. Streamwide estimates using NAIP-adjusted stream lengths varied from 6 adult Apache Trout in Mineral Creek in 2017 to 848 in Squaw Creek in 2019. Estimates using NHD-based length ranged from 81% to 100% of estimates made using NAIP-adjusted lengths (Table 2).

DISCUSSION

As we expected, stream lengths measured using high-resolution (1:24,000) NHD flowlines underestimated their true length revealed by aerial imagery. Not surprisingly, this bias was higher in meadows that tend to be more sinuous because they are unconfined and have lower gradients and stream energy (Knighton 1998), and the generalization of streams on topographic maps, even at 1:24,000 scale, reduces sinuosity of streams to maintain map aesthetics; these maps are the origin for the NHD data set. The bias we observed in NHD led to streamwide estimates of adult Apache Trout abundance that were in some cases only 81% of those made with more realistic stream length measurements. This negative bias has been noted when assessing the status of western native trout populations, but it has not been quantified as we have done here (Shepard et al. 2005; Meyer et al. 2014). The bias also has the potential to falsely trigger important management decisions for threatened Apache Trout and is directly applicable to the management of other species in similar systems (e.g., Gila Trout *Oncorhynchus gilae*).

Small adult populations of Apache Trout remained small regardless of whether the stream length used for extrapolation was measured using the NHD or NAIP, but management of more abundant populations occupying larger lengths of stream could be falsely triggered by negatively biased length estimates. For example, 500 adults has been one of three criterion used by the Arizona Game and Fish Department to determine whether a population can be opened to angling (Lopez and Hickerson 2014), and the point estimates of abundance for several study streams were near this criterion when the NHD was used to extrapolate survey data, whereas NAIP-adjusted estimates put abundance estimates further from this threshold (Ord Creek

TABLE 1. Number of segments digitized (with the number of meadow segments digitized in parentheses), percent meadow habitat, ratio of NHD length to NAIP digitized length, occupied stream length measured using the NHD, adjusted occupied length (NHD/NAIP ratio), and percent difference of lengths [100 × (NHD – NAIP)/NAIP] for Apache Trout streams in the White Mountains of Arizona.

Stream population and statistics	# segments (# meadow)	% meadow	NHD/NAIP ratio	NHD length (km)	Adjusted length (km)	% difference
Coyote	1 (1)	18.0	0.88	5.1	5.8	−12.1
Deep	2 (2)	11.3	0.89	14.6	16.3	−10.4
Flash	5 (3)	13.2	0.81	10.4	12.9	−19.4
Little Bonito	4 (0)	0.0	0.90	14.8	16.4	−9.8
Mineral	2 (0)	0.0	0.94	4.7	5.0	−6.0
Ord	2 (2)	54.6	0.84	5.6	6.7	−16.4
Paradise	12 (8)	30.8	0.88	6.5	7.4	−12.2
South Fork Little Colorado	9 (1)	1.0	0.96	10.6	11.0	−3.6
Soldier	2 (0)	0.0	0.89	2.7	3.0	−10.0
Squaw	2 (0)	0.0	0.90	13.7	15.2	−9.9
West Fork Black	9 (8)	48.5	0.89	18.6	20.8	−10.6
West Fork Little Colorado	6 (4)	33.5	0.88	14.3	16.3	−12.3
Mean	4.7 (3.6)	17.6	0.89	10.1	11.4	−11.1
Median	3.0 (2.5)	12.2	0.89	10.5	11.9	−10.5
SD	3.6 (2.9)	19.8	0.04	5.1	5.7	4.1

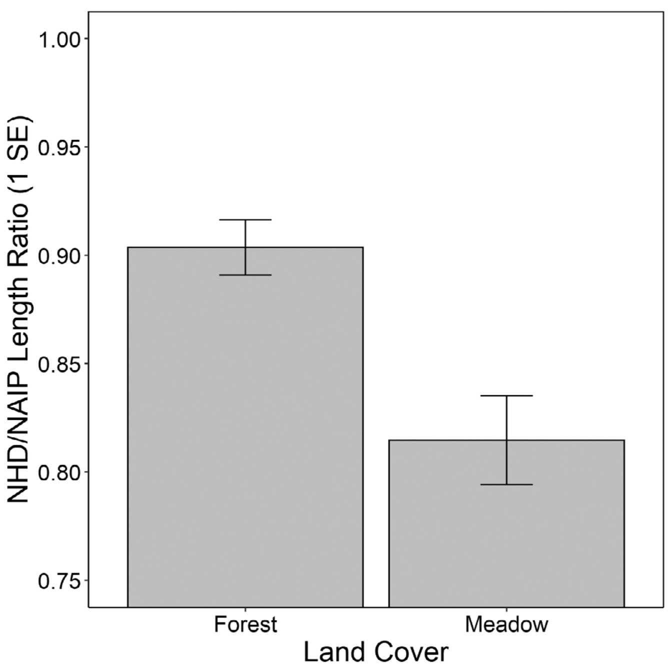

FIGURE 3. Ratio of stream length (error bars show SE) measured from NHD flowlines (streams) to streams digitized from 2017 NAIP imagery in meadow versus forested land cover. Ratios were significantly lower in meadow habitat (t-test: $t = -3.69$, df = 11.9, $P = 0.003$), showing that the length of meadow streams are underestimated more than the length of forested streams.

in 2019; Table 2). Future surveys may show more populations near this management threshold as trout population sizes are known to exhibit considerable interannual variability (Dauwalter et al. 2009; Dochtermann and Peacock 2010). Likewise, adult abundance estimates may be used as an approximation of the breeding population size and employed to evaluate whether a population may be susceptible to inbreeding depression or lack adaptive capacity and susceptible to drift due to low genetic variation based upon commonly used rules-of-thumb thresholds (e.g., 50:500 rule; at least 50 individuals to avoid inbreeding depression and at least 500 individuals to adapt and avoid drift), especially those isolated above barriers that lack the genetic and demographic benefits of occasional immigration (Franklin 1980; Rieman and Allendorf 2001; Whiteley et al. 2010). Furthermore, establishment of new Apache Trout recovery populations is typically reliant on translocations of adult Apache Trout from other populations. Adult abundance estimates are used to make decisions as to which populations can and cannot be a source of Apache Trout for establishing new populations elsewhere. Despite no populations in this study demonstrating adjusted abundance estimates increasing to over 500, the data we present clearly show that significant increases in estimates could be important relative to abundance thresholds used to trigger management in these systems, even if the thresholds are a convenient numeric threshold along a continuum and there is uncertainty associated with the point estimates of streamwide abundance.

While bias clearly does influence extrapolation of site-level data to an entire stream, in some cases the biased lengths may not be important. Stream length is often used as measure of patch size that represents a proxy for abundance or a surrogate for habitat complexity, number

TABLE 2. Number of sites sampled for Apache Trout, average number of adult Apache Trout (≥130 mm TL) per 100 m (with variance s_i^2 in parentheses) for population i, streamwide abundance estimates $\hat{N}_i$ (with bound of error estimation, B_i, in parentheses) of adult Apache Trout using NHD-based stream lengths or adjusted stream lengths based on digitized NAIP imagery, and the NHD/NAIP ratio of $\hat{N}_i$ (and ratio of B_i in parentheses). Bound of error estimation: $B_i = 2\sqrt{V(\hat{N}_i)}$ (Scheaffer et al. 2012). The percent difference is the percent of the abundance estimate using NHD-based stream lengths compared with NAIP-based lengths of occupied extent when computing streamwide abundances [100 × (NHD − NAIP)/NAIP].

Population, year sampled, and statistics	Sites (n)	Mean #/100 m $\bar{y}_i$ (s_i^2)	NHD $\hat{N}_i$ (B_i)	NAIP $\hat{N}_i$ (B_i)	$\hat{N}_i$ ratio (B_i ratio)	% difference
Coyote (2018)	5	0.40 (0.80)	20 (39)	23 (44)	0.87 (0.89)	−13.0
Deep (2016)	10	2.80 (6.84)	409 (233)	456 (261)	0.90 (0.89)	−10.3
Flash (2019)	24	1.38 (3.46)	143 (69)	177 (88)	0.81 (0.78)	−19.2
Little Bonito (2019)	24	2.25 (6.28)	333 (139)	369 (155)	0.90 (0.90)	−9.8
Mineral (2017)	9	0.11 (0.11)	5 (9)	6 (10)	0.83 (0.90)	−16.7
Ord (2019)	11	19.64 (198.10)	550 (185)	668 (237)	0.82 (0.78)	−17.7
Paradise (2018)	13	0.15 (0.14)	10 (12)	11 (14)	0.91 (0.86)	−9.1
South Fork Little Colorado (2017)	18	0.17 (0.50)	18 (32)	18 (34)	1.00 (0.94)	0.0
Soldier (2017)	3	8.33 (16.33)	225 (119)	250 (133)	0.90 (0.89)	−10.0
Squaw (2019)	19	5.58 (15.70)	764 (231)	848 (259)	0.90 (0.89)	−9.9
West Fork Little Colorado (2018)	24	2.08 (8.51)	298 (155)	340 (179)	0.88 (0.87)	−12.1
Mean	14.5	3.90 (23.34)	252.3 (111.2)	287.8 (128.5)	0.88 (0.87)	−11.6
Median	13.0	2.08 (6.28)	225.0 (119.0)	250.0 (133.0)	0.90 (0.89)	−10.3
SD	7.7	5.81 (58.25)	250.0 (84.2)	284.2 (97.2)	0.05 (0.05)	5.2

of unique habitats available, or the likelihood of refuges being present during harsh environmental conditions, such as drought (Hilderbrand and Kershner 2000; Peterson et al. 2014). This has led to development of models that incorporate stream length to gauge the likelihood of a population to persist into the future, a future that includes climate change, or success of translocations or reintroductions (Harig et al. 2000; Harig and Fausch 2002; Isaak et al. 2015). These models are typically developed based on stream length as measured using map-based representations of streams, including the NHD; as a result, underestimates of stream length are inherent in the data used to develop and apply the models. Thus, biased estimates of stream length are unlikely to impact their application and any conclusions drawn from them. One exception may be that streams with more meadow habitat than average may have habitat length underestimated when applying these general criteria. We did also have to digitize the headwater extent (2.1 km) of the West Fork Black River because past field survey data showed fish presence upstream of the NHD headwater extent, thus highlighting an issue demonstrated in other regions that may bias (low) map-based estimates of length (Elmore et al. 2013). Unfortunately, the West Fork Black River was the only stream with no recent population survey information to explore the influence of this NHD inaccuracy; we included it in the length comparison because the system contains extensive meadow habitat and has potential as a metapopulation (Table 1) (Williams and Carter 2009).

In addition to bias from measured stream lengths, removal estimates of abundance from multiple-pass electrofishing in streams can also be biased low due to heterogeneity in capture probability across electrofishing passes (Peterson et al. 2004; Rosenberger and Dunham 2005; Meyer and High 2011). This could potentially compound any bias in streamwide abundance estimates due to negatively biased stream lengths and cause streamwide abundance estimates to be even lower. However, this bias in removal estimators is greatly reduced when detection probabilities are high (e.g., >0.5) (Rosenberger and Dunham 2005; Sweka et al. 2006; Habera et al. 2010). For most Apache Trout streams, detection probability is typically high (>0.8) and consistent across habitat types (Dauwalter et al. 2017a), and our capture probabilities averaged 0.81. As such, bias associated with heterogeneity in or declining capture probably across successive passes for Apache Trout was likely minimal. Nevertheless, managers should be aware of the bias due to heterogeneity of capture probability and how it may compound any bias associated with using stream lengths measured from the NHD to extrapolate abundance estimates (Cook et al. 2010). Meyer and High (2011) estimated bias in abundance estimates from removal electrofishing due to decreasing capture probability over successive passes and they found that bias was a function of habitat covariates; however, they found mixed success in applying a correction factor to the biased estimates.

We conclude that it is important to assess whether accurate estimates of stream length are needed for management decisions. If so, it is important to understand whether lengths are biased for available paper or digital representations of streams, whether that bias differs across ecosystem types or other important landscape features, and whether it has the potential to impact estimates for fishery attributes, such as when local field data are being extrapolated to infer characteristics of entire streams as we illustrated with adult Apache Trout abundance. High-resolution imagery, such as NAIP, satellite imagery, and lidar (light detection and ranging) data, or other useful spatial data products are now widely available and can be used to quantify stream length bias to determine whether it may impact important fishery management decisions (Dauwalter et al. 2017b). Lidar data, where available, has for over a decade shown promise for mapping small headwater streams and other hydrologic features on the landscape (James et al. 2007).

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