



Research article

Integration of new methodologies in monitoring native fish habitat and populations in the southwestern U.S.

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ABSTRACT

Native fish populations and their habitats are declining in the southwestern United States. The upper Verde River (UVR) is one of a shrinking number of remaining perennial stream systems where multiple populations of native fish species still exist. This study utilized unoccupied aerial vehicle (UAV), terrestrial lidar, real-time kinematic (RTK) survey technology and environmental DNA (eDNA) sampling to assess current riparian and in-stream habitat conditions and compare them to an assessment conducted in 2009 to assess current suitability and long-term trends in populations and habitat conditions of native fish.

Average stream discharge within the UVR has fallen over 40%, from a high of 30.6 cubic feet per second (cfs) in 1995 to recent lows in 2023 of just over 18 cfs. Woody vegetation cover has increased within the riparian zone by approximately 30% since 2007 and cattail backwater areas, which support non-native fish and amphibian populations, now comprise nearly 9.4% of total riparian cover after being nearly absent in 2007. Analysis of eDNA fish samples suggest that populations of all native fish species are no longer detected within the first 10 km (kms) of the river, with Spikedace (*Meda fulgida*) appearing to have been extirpated from the UVR. Groundwater pumping in the upper watershed along with recent long-term drought conditions are most likely associated with the decline in baseflow and are resulting in the expansion of deep-rooted woody species and cattails along the streambanks, confining the stream channel and reducing suitable habitat for native fish species.

1. Introduction

Native fish populations in the southwestern U.S. have been in decline for many decades due to a multitude of factors including loss of suitable habitat related to dam construction, dewatering of stream channels from groundwater pumping, impacts from post-fire flooding events, as well as competition and predation from introduced species (Hedden et al., 2022; Olden and Poff, 2005; Propst et al., 2008; Ruhf et al., 2016; Ruppert et al., 1993). More recently, increasing temperatures due to climate change in the region have been intensifying the effects of episodic drought periods. Climate models suggest that the Southwestern U.S. has entered a period of consistently drier conditions due to climate change, which will intensify aridification in the region (Wahl et al., 2022). Protection and restoration of existing native fish habitat is critically needed.

Many water development projects in the southwestern United States store or move water from a source to nearby population and agricultural centers. However, they have often had unintended consequences of

promoting the movement of non-native fish species within and across watershed boundaries. One example is the Central Arizona Project (CAP), a network of pipelines and aqueducts which delivers water from the Colorado River in western Arizona to central and southern Arizona. As part of mitigation undertaken by the U.S. Bureau of Reclamation (BOR) beginning in 1994 under Section 7 of the Endangered Species Act (ESA) following construction of the CAP, the BOR began taking efforts to discourage the establishment of non-native fish species in the region and conserving existing native populations. It is important to continuously monitor the impacts of such efforts. This study examines long-term native and non-native fish population trends.

Considered as an “ecologically critical area”, the Upper Verde River (UVR) provides critical habitat for plants, animals, native fish, and rare reptile species (Arizona Department of Water Resources, 1994; Neary et al., 2012). Populations of native fish species on the UVR include Sonora Sucker (*Catostomus insignis*), Desert Sucker (*Catostomus clarki*), Spikedace (*Meda fulgida*), Roundtail Chub (*Gila robusta*), Longfin Dace (*Agosia chrysogaster chrysogaster*) and Speckled Dace (*Rhinichthys*

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oculus), which have been steadily declining on the river for several decades (Rinne, 2012; Stites and Chmiel, 2022). Previous research suggests the decline is due to the loss of suitable habitat for native fish, which is typically made up of a mosaic of high-gradient and low gradient riffles, as well as a mix of glides and runs. Shallow pools are slowly giving way to long, deep pool habitats, more suitable for non-native predaceous fish species such as redeye bass (*Micropterus coosae*) (Rinne and Miller, 2006). Today, the primary threats to native fish populations on the upper Verde are predation/competition from non-native fish species and habitat alteration related to changes in hydrology (Neary et al., 2012; Rinne and Miller, 2006). Similar to other western states, Arizona has seen large increases in urban areas and population growth, which increases demand on available surface and groundwater resources and poses serious threats to native fish populations in the Southwest (Crimmins et al., 2023). Since 2000, populations in the UVR region have grown at nearly twice the average of the rest of the United States, increasing nearly 20 percent (U.S. Census Bureau). Because there are no significant surface water resources in the surrounding watershed, much of population growth has relied on groundwater pumping of local aquifers. Since 1940, groundwater levels in Little Chino Valley aquifer, an important source of baseflow to the UVR, have receded by more than 30 m (100 ft) in the margins of the basin (Arizona Department of Water Resources, 2021). Perhaps not surprisingly, non-native fish species appear better adapted to the reduced flow conditions, out-competing native species and putting further stress on existing populations (Gido et al., 2013; Ruhf et al., 2016). Recently, the construction of a pair of fish barriers has been proposed by the BOR on the Upper Verde River (UVR) to help conserve native fish populations.

The main objective of this study was to utilize new technologies for monitoring and integrate new datasets with legacy datasets in quantifying changes in habitat and native species populations since 2012. To accomplish this, we utilized an unoccupied aerial vehicle (UAV) to collect vegetation data across the entire floodplain at each of ten long-term monitoring sites, thereby replacing the previous woody vegetation plots which were limited to 10 m × 40 m strips on either stream-bank. UAV remote sensing is increasingly utilized in vegetation monitoring (Villarreal et al., 2025; Sankey et al., 2024, 2021 ab, 2019, 2018), although riparian vegetation has been less commonly characterized via UAV sensors. The high spatial, temporal, and spectral resolution enabled by UAV remote sensing offer unique opportunities to accurately detect riparian species and efficiently monitor traditional vegetation plots and larger areas of up to 10 ha per flight (Sankey, 2025).

Another emerging tool for geomorphic monitoring is terrestrial lidar. Given its high spatial and temporal resolution, lidar sensors can provide more accurate river assessments compared to traditional monitoring tools (Wheaton et al., 2010). Lidar data can be used at a wide range of spatial scales to quantify temporal changes in river channels (Kasprak et al., 2019; Anderson, 2019), characterize geomorphic processes, and inform policy and management decisions (Passalacqua et al., 2015; Waskiewicz et al., 2017). We used terrestrial lidar to capture elevations across the entire floodplain, 20 m above and 20 m below the existing single line transects. In addition, Real-Time Kinematic (RTK) survey technology was used to improve previous geomorphology survey methods, greatly increasing the number and accuracy of survey points collected within the stream channel at each long-term monitoring plot. Lastly, environmental DNA (eDNA) sampling for the presence and absence of key fish species was employed along the entire study length of the river corridor. Comparisons were made to data collected in 2009 to quantify changes in geomorphology, riparian vegetation and fish populations.

2. Materials and methods

2.1. Study area

The upper Verde River lies within the transition zone between the Basin and Range and Colorado Plateau provinces. The landscape displays lithologies common to both provinces, interspersed throughout the riparian corridor and uplands. Granitic materials are most common to the south, while limestone, sandstone, and basalt rock types are most common north of the river (Krieger, 1965). The UVR section of the Verde River Valley is semi-arid in nature with precipitation averaging less than 460 mm (18.0 in) per year (Blasch et al., 2006). Except for higher terrain to the north that provides streamflow for Sycamore Canyon and Hell Canyon, two major tributaries of the UVR, most of the precipitation occurs as rainfall rather than as snow. Monthly precipitation varies by over a factor of five from the spring dry period (13 mm or 0.5 in) to the summer monsoon period (70 mm or 2.8 in). Over the past century, rainfall in the UVR region has gone through several cycles of wet and dry periods. Analysis of rainfall records since 1900 has shown that the UVR is in a lower rainfall cycle that started in 1994 and that snowfall for the UVR and Middle Verde watersheds has been mostly below normal since 1955. Potential evapotranspiration rates average more than 1520 mm (60.0 in yr⁻¹), which creates the semi-arid conditions.

Baseflows in the UVR have continued to decline since the mid-1990s. The 120-day moving average has fallen over 40 % from a high of 30.6 cubic feet per second (cfs) in 1995 to recent lows in 2023 of just over 18 cfs (Fig. 1). The combination of increasing average temperature and below average precipitation over the last four decades suggests the likelihood of the beginning of a period of consistently warmer and drier conditions in the southwestern United States (Wahl et al., 2022). Less water input due to groundwater pumping and increased evapotranspiration due to higher average temperature will continue to put increasing pressure on baseflow inputs in the upper Verde River from the Big and Little Chino aquifers.

Wirt et al. (2005) estimated that contributions to the Verde River from the Big Chino Valley aquifer make up about 81 % of the total flow in the river with approximately 19 % coming from the western part of the Coconino Plateau, contributions from the Little Chino subbasin are now negligible due in part to the extensive groundwater pumping that has already occurred. Although the Little Chino and Big Chino Valleys route all surface-water drainage to the UVR above Hell Canyon, there have been on-going discussions between local, State, and Federal officials and scientists over the fate of groundwater within these aquifers and the potential impact that additional groundwater pumping might have on the remaining UVR baseflows (Barks, 2023). In addition to hydrologic alteration, there is also modification of sediment processes in the UVR watershed. Two dams exist on Granite Creek, a major tributary to the UVR, including the Granite Creek Dam constructed in 1915 which created Watson Lake and later Willow Creek Dam constructed in 1935 which created Willow Lake, both were constructed for irrigation purposes but today primarily support recreation and wildlife. In addition to dams on Granite Creek, there are also multiple gravel mining operations currently in operation above the confluence with the Verde River. Sullivan Dam was constructed on the main stem of the Verde River in 1939 which created Sullivan Lake, however, within a decade the lake had mostly filled with sediment. Today, the remnants of Sullivan Dam remain in place, continuing to alter sediment movement from the upper watershed to the main channel of the UVR.

Fish barriers have been proposed for the UVR. They are considered to be one of the only feasible short-term solutions to meet the immediate needs of native fish conservation within the region (Clarkson and Riley, 2012). They consist of concrete drop-structures installed across the stream channel intended to separate upstream native fish populations from downstream, non-native populations. Following the construction of a barrier, all fish are removed from the upstream portion of the targeted stream, typically through the application of a piscicide deployed at

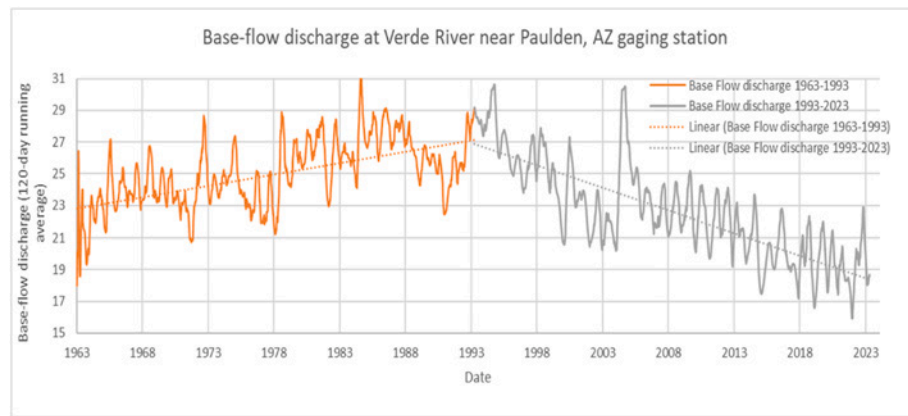


Fig. 1. Average base-flow discharge for the Upper Verde River since 1963, when record keeping at this location began. Data taken from the USGS stream gauge at Paulden, Arizona. Credit Joel Unema, USGS URL: [https://waterdata.usgs.gov/nwis/monthly?.](https://waterdata.usgs.gov/nwis/monthly?)

the head of stream reach, which euthanizes all fish within the reach (Pham et al., 2013). Afterwards, only desirable native species are restocked above the barrier, which then serves to prevent the future upstream movement of any non-native fish species.

Monitoring of the UVR began in the early 1990's in support of the Spikedace Recovery Plan (Marsh, 1991) which identified the uppermost section of the Verde River as one of the few remaining populations of Spikedace. Long-term stream monitoring sites were established and were intended to be read on a reoccurring basis by U.S. Forest Service to track changes on the river and help inform management decisions being made regarding the protection of native fish species. However, in the ensuing decade, the Spikedace was thought to be extirpated from the UVR and other populations of native fish species appeared to be in decline. Due to budget constraints, monitoring was discontinued by the

U.S. Forest Service following the publication of a detailed summary report in 2012 (Neary et al., 2012). This has left a substantial gap in knowledge regarding the timing and causes of many of the changes that have occurred on the river within the last decade.

2.2. Site selection

For this study, a subset of the original 44 monitoring plots, established by the USFS, were selected for remeasurement based on the proximity of some sites to one another as well as the greater amount of data collected at individual plots. To increase similarity among sites, each plot originally consisted of riffle, run, and pool habitat with a latitudinal geomorphology transect established across the riffle habitat at each plot. The project separated the study area into two sections,

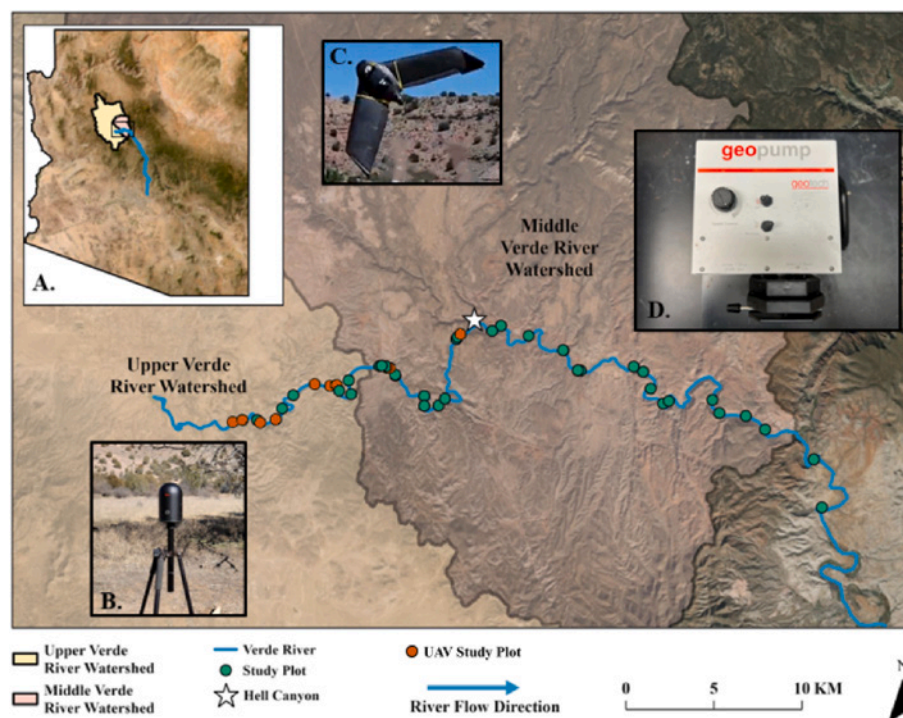


Fig. 2. The project study area extends from the confluence with Granite Creek on the western end to near the confluence of Sycamore Creek on the eastern end. Green dots represent all 44 long-term monitoring plots established by the U.S. Forest Service in the early 1990s. Red dots represent plots which were surveyed with a UAV A. The Verde River flows through central Arizona before joining the Salt River near Phoenix, Arizona. B. A Leica BLK360 terrestrial lidar scanner was used to survey 20 of the 44 long-term monitoring plots. C. Sensefly Ebee X UAV platform equipped with a multispectral sensor was used to survey 10 long-term monitoring sites D. Environmental DNA samples were taken using a geostatic pump and filter to detect the presence or absence of key fish species at 29 monitoring sites.

naturally divided by Hell Canyon, a major tributary of the upper Verde River. The upper section is defined by wide valley bottoms, historically containing large wetland areas comprised of sedges and rushes, with channel substrates typically ranging from sands and silts to cobble. The section below Hell Canyon contains stream reaches confined by steep canyons. Water inputs from the upper Verde Valley sub-basin, the portion of this study which extends from Hell Canyon down to Sycamore Canyon, while subject to the same climatic trends are less impacted by human development and may prove to be more resistant to anthropogenic inputs (Garner et al., 2013). The width of the valley bottoms is reduced in many reaches and the diameter of channel substrate increases, containing greater amounts of course cobble and boulders. The proposed locations of two fish barriers also coincides with the delineation of our upper and lower sections. The proposed site for the lower most barrier is just above the confluence with Sycamore Canyon, the lower most reach of our study. The upper barrier is proposed just above the Hell Canyon confluence, the mid-point of our study (Fig. 2).

The USFS began monitoring the long-term sites in the early 1990s and re-read the plots in 1997, 2001/02, and 2009. The 2009 data was used to compare to the new remote sensing data collected via unmanned aerial vehicle (UAV) and terrestrial lidar scanner (TLS) sensors in this study. The sites are distributed over the entire 50 km of the UVR from Paulden, AZ to Sycamore Canyon, but at uneven distance intervals. We resurveyed 20 of the 44 plots with previous geomorphology, woody vegetation as well as updated real-time kinematic (RTK) survey methods (Wyrick et al., 2014). We successfully imaged 17 of these plots with the ground-based, terrestrial lidar scanner (three other plots were excluded from the analysis due to missing 2009 data to match) (Tango et al., 2024). Of these 17 plots, we then imaged ten with a UAV multispectral sensor in this study.

2.3. UAV data acquisition and analysis

We imaged ten of the UVR plots in summers 2021 and 2022 with a Sensefly Ebee X UAV platform equipped with a multispectral sensor that images in four spectral bands: green, red, red edge, and near-infrared, centered at 550, 660, 735, and 790 nm, respectively (SenseFly, Lausanne, Switzerland). The Sensefly Ebee X flew at 70m AGL in interlaced, perpendicular lines with 85 % and 90 % latitudinal and longitudinal overlaps, respectively (Sankey et al., 2021a,b; Donager et al., 2021; Belmonte et al., 2019). Each flight lasted for ~25 min, on average, centered at each transect location. Flight mission planning and data parameters were customized and executed in SenseFly eMotion 3 software (SenseFly, Lausanne, Switzerland). The resulting multispectral image had 14 cm spatial resolution spanning, on average, a 10 ha (~22 acres) centered at each transect location. Each UAV image was subset to focus on the transect area and excluded distorted areas along the edges of the multispectral image.

Using a random forest classifier in the machine learning classification tool, a total of eight different target cover types were classified: 1) water, 2) gravel bars, 3) herbaceous cover, which consisted of forbs and sedges/rushes (*Cyperus esculentus*, *niger* and spp.), 4) cattail species (*Typha* spp.), 5) riparian woody cover which combined all woody species, 6) junipers (*Juniperus monosperma*) outside of the narrow riparian corridor, 7) bedrock/bare ground, and 8) shadows (Fig. 3). We focused on exposed water surface area and gravel bars because their relative abundance (percent cover) is an important indicator of riparian habitat condition.

We also emphasize cattail (*Typha* spp.) detection because cattails can rapidly encroach into the shallow riverbanks resulting in high density and cover, which deteriorate habitat quality for native species by supporting non-native fish and amphibian species (Bansal et al., 2019). High density cattail stands can also capture sediment, which can further narrow the river channel (Neary et al., 2012). The UAV image-derived land cover type classification was then used to train and classify a coarser spatial and spectral resolution image from the National Aerial

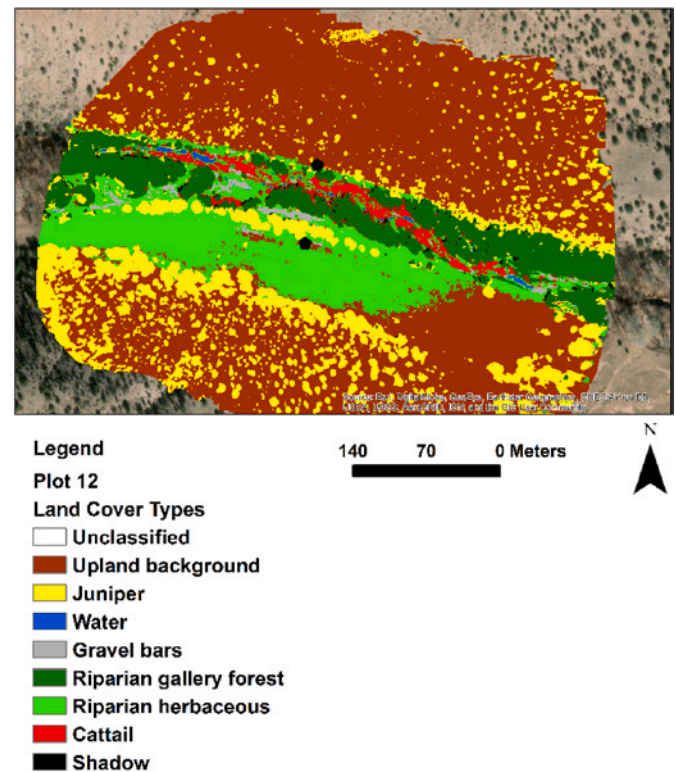


Fig. 3. Multispectral aerial vegetation survey using Unmanned Aerial Vehicle.

Imaging Program (NAIP), which covered the entire stretch of the UVR including the subset of plots imaged by the UAV and TLS.

2.4. National Aerial Imaging Program data classification

We used manned airborne images from the US National Aerial Imaging Program (NAIP) for the entire UVR corridor to examine riparian-woody vegetation cover changes across a larger spatial extent beyond the UAV image extents. The riparian-woody vegetation cover change over time was of specific interest given its significant role and impact on riparian habitat conditions. The NAIP image is typically collected for each US state, on average, once every five years providing an opportunity for riparian-woody-cover change estimates, whereas our UAV images represent a single year at each transect. We used the available NAIP images in 1 m spatial resolution from 2019 to 2007 in four spectral bands: blue, green, red, and near-infrared to classify three different land cover types: 1) water, 2) bare ground, and 3) woody vegetation cover (Fig. 4). After the 2007 and 2019 NAIP images were classified, we compared the classification outputs via a pixel-to-pixel change detection to only examine changes in riparian-woody- vegetation cover.

2.5. Field-based riparian woody vegetation classification

In the field, we used a modified Daubenmire methodology (Bonham et al., 2004) along both right and left streambanks to classify species, density, diameter at breast height (DBH) and height of each woody stem. This method uses five plots, 5 m × 8 m in dimension, established along 40 m of streambank with the permanent single-line geomorphology transect acting as the midpoint. Importance values (IVs) indicating how dominant a species is in a certain location were calculated by summing relative density and relative dominance and scaling the result between 0 and 200.

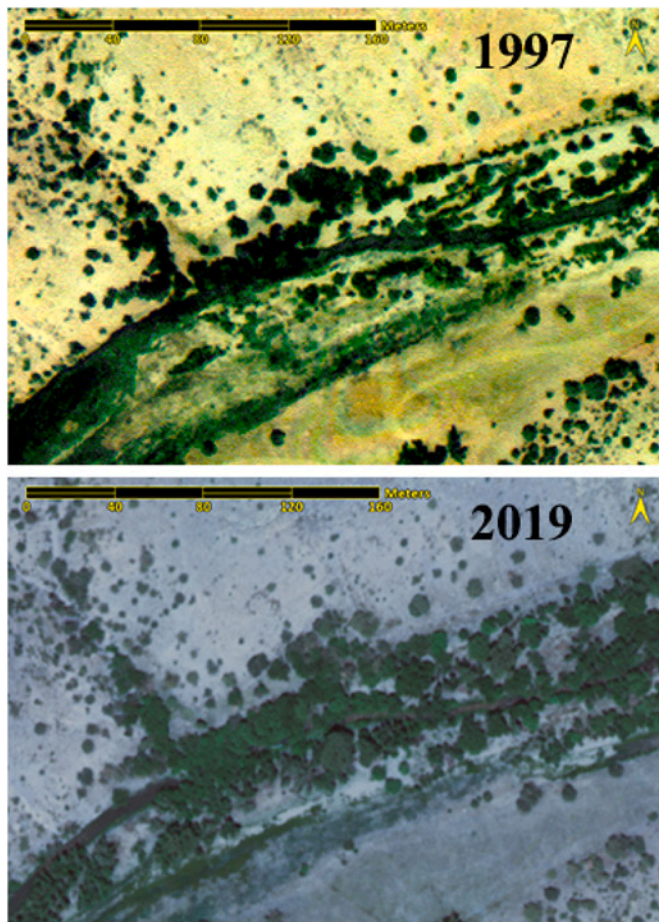


Fig. 4. Aerial images from 1997 (top panel) and 2019 (bottom panel) illustrating riparian woody vegetation cover increases over time along the Upper Verde River corridor.

2.6. Terrestrial lidar data and geomorphic change detection

To quantify geomorphic changes over time, we completed two different comparisons: 1) USFS survey-USFS survey comparison, and 2) USFS survey-TLS survey. In summers 2021 and 2022, we imaged nine plots with a Leica BLK360 terrestrial lidar scanner to compare to transect-based riparian cross-section data collected by the US Forest Service in 2009, which provided opportunities to estimate geomorphic changes since 2009 at matching locations, whereas the other transect locations had shifted slightly over time. Our lidar scan extents included the entire lengths of the single line transects, which varied between 30m and 115m, and a consistent 40m widths. As a result, the lidar-imaged areas ranged in dimensions and total area, but with constant 40 m widths. A total of 15–25 individual scans from different scan positions were acquired at each plot to cover the entire spatial extent. The Leica BLK360 scanner laser pulse is centered at 830 nm wavelength and has a maximum range of 60m with a vendor-reported accuracies of 8 mm at 20 m.

The 15–25 individual scans were registered and mosaicked together to create a single point cloud with a total of 30–60 million points at each plot location (Fig. 5). The point cloud data were then used to create digital elevation models (DEM) centered on the USFS line transect location for each plot. The original USFS single line transects were permanently marked with stakes on both north and south banks of the channel with a measuring tape stretched between the two stakes. Using a laser level, the USFS measured the vertical distance from the tape to the ground at typically 1–5m intervals starting on the north bank of each plot. The 2009 vertical distances were then used to create a single-line

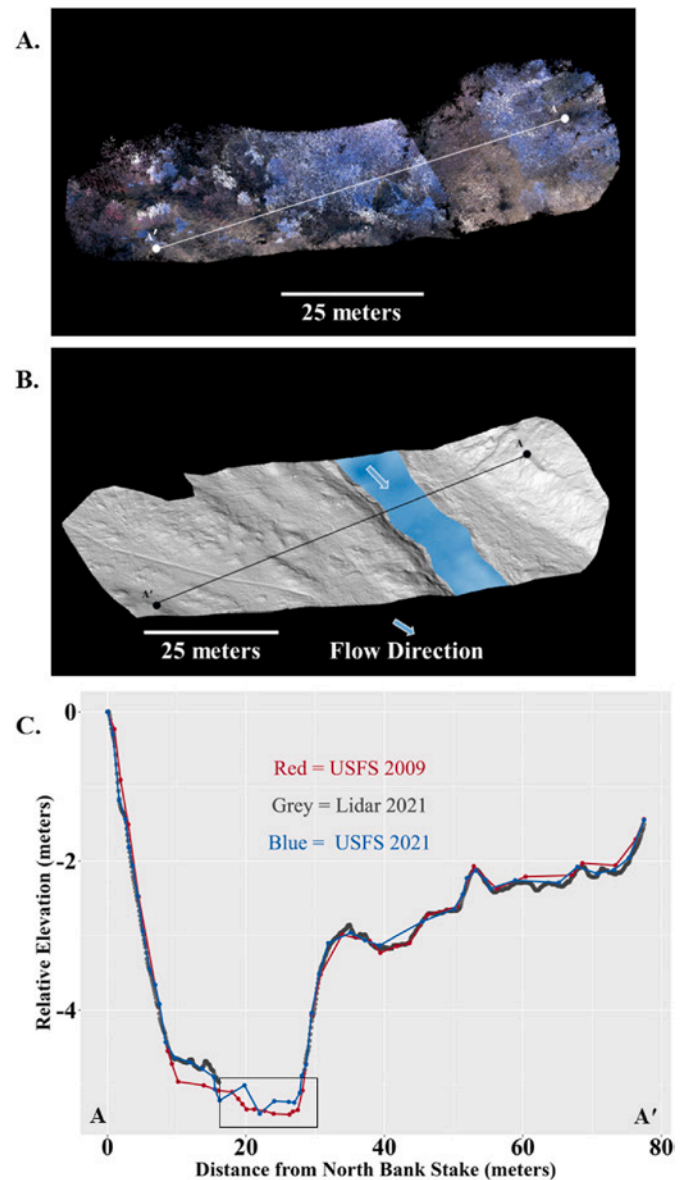


Fig. 5. A. Mosaicked lidar-derived point-cloud of riparian vegetation along a long-term monitoring transect B. Lidar-derived digital elevation model (DEM) from the same transect. C. Cross section showing the 2009 and 2021 USFS data plotted with the lidar-derived cross section data collected in 2021. The black box indicates the absence of useable lidar data as the lidar sensor does not capture data below the water's surface. Subsurface data were collected traditional surveying methods in 2009 and 2021.

cross-section of the channel geometry and we then compared them to the lidar-derived DEMs to estimate geomorphic changes along the UVR over the thirteen-year period between 2009 and 2022.

2.7. Channel morphology change detection

Stream cross-sectional data were collected in 2021/22 with a laser level and staff rod in the field and later digitized into excel spreadsheets. Elevation points were collected at least every meter along the transect. Other important points along the single line transect were recorded including bankfull, water's edge, water depth, and thalweg. Comparison of 2009–2021/22 data included 16 single line transects. Entrenchment ratios (ER) were also collected and it compares the width of the bankfull channel to the width that the water surface would have been during a typical flood. This ratio is often used to help determine energy

dissipation, stability, and stream channel evolution. ER was calculated as flood prone width/bankfull width. Flood prone width is calculated as 2x the bankfull maximum depth. Width-to-Depth ratio was calculated as the bankfull width/average bankfull depth. Width-to-Depth ratios (W/D) are key to understanding the distribution of available energy within a channel, and the ability of various discharges occurring within the channel to move sediment.

Plots that are located above Hell Canyon include plots 5–23 while plots 32–44 are located below Hell Canyon down to the confluence with Sycamore Canyon. A paired *t*-test was performed on the ER and W/D comparison between 2009 and 2021/22. A two-sample unequal variance *t*-test was performed on the ER and W/D comparisons above and below Tri-Canyon. Entrenchment Ratio (ER) and Width to Depth (W/D) ratio variables were calculated from the data in the excel spreadsheets. ER of 16 stations in 2009 compared to 2021/22.

2.8. Real-time kinematic survey

Bathymetric surveys were conducted at 19 long-term monitoring plots using RTK GPS in 2023. Survey data was captured to characterize the below water bathymetric profile of the stream channel. Surveys focused on capturing breaks in lateral and cross channel morphology. Characterization and estimate of the stream channel habitat area was defined as riffle, pool, run, cattail backwater and large rocks. Each plot was surveyed from 20 m upstream and downstream of previously measured cross channel transects yielding approximately 40-m reaches centered on the historic single line transects. When mapped in ArcGIS, these additional points create a three-dimensional survey layer, which allows for the classification of instream habitat features (Wyrick et al., 2014) and their percent cover (Fig. 6).

Pools were classified as any depths typically over 1.8m and/or lower flow velocities, with generally larger stream widths. Pools were also identified in plots where water's edge was relatively level. Runs were identified by deeper stream channels than riffles, approx. ≥ 0.5 m depth but typically shallower than pools within a given reach, with small changes in water edge elevations. Riffles were identified by water edge to thalweg depths of generally <0.5 m, with simultaneous decreases in elevation of thalweg and water edge and often preceded by an upstream increase of thalweg elevation to the start of a riffle and termination of a pool or run designation.

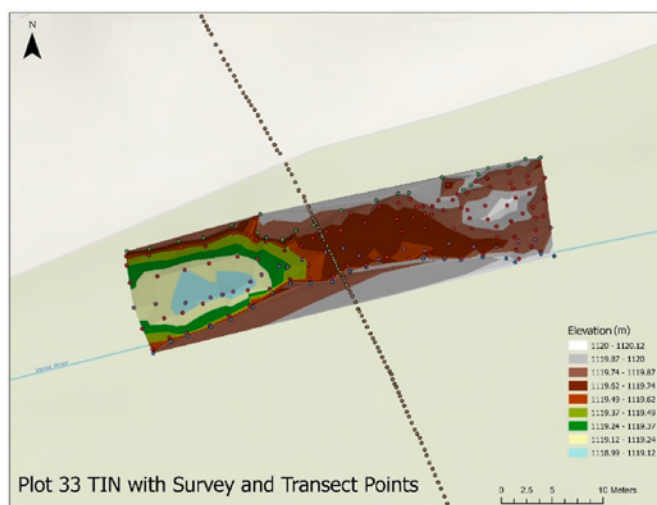


Fig. 6. Image and profile created from RTK survey conducted at long-term monitoring plot depicting in-stream pool (light blue/green), run (red) and riffle (grey) habitats.

2.9. Fish population and eDNA sampling

In November of 2021 and 2022, a total of 29 plots located along the entirety of the study area were sampled using eDNA sampling methods developed by RMRS and the National Genomics Center for Wildlife and Conservation (Carim et al., 2016). The method requires 5 L of stream water to be drawn through a Whatman® 1827-047 Glass Microfiber Binder-Free Filter (Thermo Fisher Scientific Inc.), 1.5 μ m, 3.7 s/100 ml flow rate, Grade 934-AH, 47 mm diameter, which is then analyzed for DNA. Negative controls were included in the DNA extraction process by incorporating samples containing filtered distilled water and in the qPCR analyses process by using distilled water as a replacement of the DNA template in a triplicate reaction. Specific sampling locations were located within the 40-m plot areas for long-term monitoring in well-mixed water, near the center of the channel. Sampling areas were also co-located with long-term, quantitative sampling sites using electro-fishing methods established by the USFS in the 1990s.

Sample analysis was conducted to determine the presence/absence of four native species including Sonora Sucker (*Catostomus insignis*), Desert Sucker (*Catostomus clarki*), Spikedace (*Meda fulgida*) and Roundtail Chub (*Gila robusta*). All species are present in the river year-round. The limit of detection (lowest concentration with >95 % amplification success) of each assay was 10 mtDNA copies/rxn with successful detection of target DNA in all six replicates; however, each assay also detected target DNA in five out of six replicates at 2 mtDNA copies/rxn (83.3 % detection success) (Dysthe et al., 2016). Two native species still present in the river, the Longfin Dace (*Agosia chrysogaster chrysogaster*) and Speckled Dace (*Rhinichthys osculus*) were not screened because the DNA assays for these two species were not available. However, once developed, eDNA analysis can be re-run to detect the presence of these species.

3. Results

3.1. Fish populations

Backpack electrofisher sampling conducted in 2006 by the USFS from plots 10 - 13 found all native species sampled for in this study were present, apart from Spikedace, and the three main species Desert Sucker, Sonora Sucker, and Roundtail Chub combined made up a significant proportion of the 7084 total fish sampled: 32.3 %, 35.29 % and 3.19 %, respectively (Rinne, 2012). Electrofisher data collected by the Arizona Game and Fish Department in 2022 using canoe mounted electrofishing methods which included plots 10 - 13 as well as additional reaches along the upper most stretch of the UVR (Stites and Chmiel, 2022), although not directly comparable to the data collected in 2006 because the difference in method (backpack vs. canoe mounted electrofisher) and time spent sampling, the data still indicate a distinct declining trend in native fish populations. Of the 364 fish sampled in 2022, only 4 were native fish, suggesting a decrease in relative abundance from 70 % in 2006 to 1 % of total fish sampled in 2022.

These data are supported by the results of our eDNA sampling in 2021/22 which did not detect native species at eight of the twelve uppermost sites. Desert Sucker was detected at the first site near the river's origin. Roundtail Chub was detected at two sites near kilometers 3 and 4 from the confluence with Granite Creek. The next positive Desert Sucker detection occurred approximately 10 kms downstream and the first positive detection of Sonora Sucker occurred nearly 11 kms downstream. Spikedace were not detected at any site sampled for this study and is presumed to be extirpated (Rinne, 1999). Both species of sucker were detected at the seventeen remaining sites except for Desert Sucker at a single site near kilometer twelve. Roundtail Chub was detected at eleven of the remaining seventeen sites, continuously present from the confluence with Hell Canyon to the confluence with Sycamore Creek, which makes up the lower half of the study area (Fig. 7).

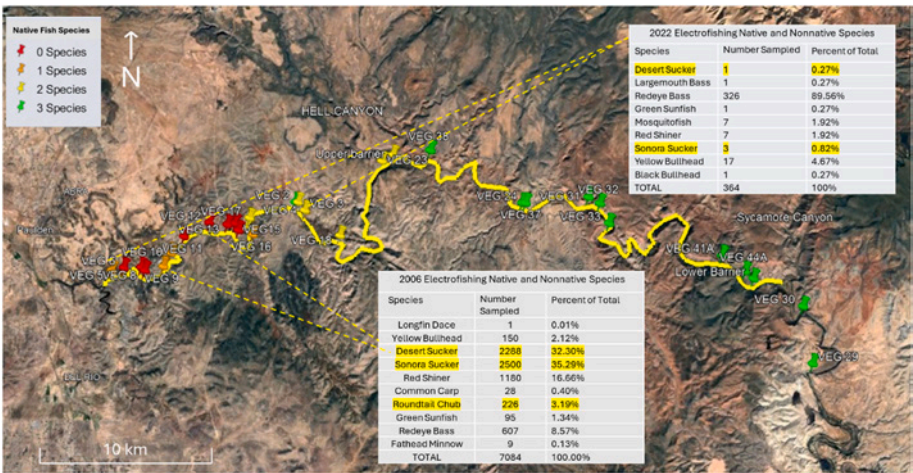


Fig. 7. The lower table shows the total number of Desert Sucker, Sonora Sucker, and Roundtail Chub captured during backpack electrofisher sampling in 2006 along the stretch of the Verde River within the dashed lines, 2288, 2500, and 226, respectively. The upper table depicts canoe electrofisher sampling conducted by The Arizona Game and Fish Department in 2022 within the dashed lines where only a handful of native fish were captured. Environmental DNA sampling along the same stretch of river in 2022 for this study only detected the presence of Roundtail Chub at Site 10, no other native species sampled for were detected at other sites within this reach. Color coded pins represent the number of native species detected at each sampling site using eDNA in 2022 throughout the entire study area: red = 0 species, orange = 1 species, yellow = 2 species and green = 3 species. The fourth species sampled for, Spikedace, was not detected at any site in 2006 or 2022.

3.2. UAV image classification

We successfully classified eight different cover types at the ten plots imaged by the UAV. The UAV image extent covered both the riparian corridor and surrounding upland vegetation and bedrock/bare ground. Within the riparian corridor defined by the flood prone area, riparian woody vegetation was the most common cover type covering on average 53.8 % of the narrow sub-section of the images. Herbaceous vegetation (sedges/rushes) was the second most common cover type comprising 22.1 % of the riparian corridor. In addition, on average 9.4 % of the riparian corridor was covered with cattails, which are considered a nuisance species. Water surface and gravel bars only covered 4.7 % and 6.2 % of the riparian corridor, respectively. While these cover type estimates are from a single data and do not indicate changes, the estimated respective area extents are consistent with trends, observed in the field, of potentially increasing cattail cover and decreasing channel width. The 2021–2022 UAV images will provide important benchmark data for future estimates of land cover-type changes.

Outside of the narrow riparian corridor, many of the UAV images (10 ha, on average) covered the steep slopes and upland bedrock/bare ground areas. Within these larger extents, we additionally classified exposed bedrock/bare ground as well as juniper. The mean areas for each cover type (%) across the entire larger UAV image extent (~10 ha) at each plot location is summarized in Table 1.

3.3. NAIP imagery

We used the coarser spatial resolution NAIP images to quantify

woody cover change from 2007 to 2019 due to the lack of any previous UAV data. This comparison indicates that riparian-woody vegetation cover has increased approximately 30 %. Specifically, 6 % of the area classified as bare ground in 2007 and 23 % of the area classified as water in 2007 were classified in 2019 as riparian woody cover indicating that riparian-woody vegetation may have encroached on the river channel resulting in narrowing over time.

A similar trend of riparian-woody vegetation cover is observed from a few older aerial photographs we acquired. For example, Fig. 4 compares manned aerial images from 1997 versus 2019. The figure illustrates that riparian-woody cover has increased over the 24-year period. While this trend may not be observed everywhere, it is observed in many portions of the areas imaged in 1997. This trend of increasing riparian-woody vegetation is notable given that invasive tamarisk and Russian olive trees have been removed along the riparian corridor over the same 24-year period.

3.4. Changes in woody stem density 2009–2021/22

For the purposes of this study, woody vegetation data was used to ground-truth aerial survey and UAV identification data and to examine potential changes in species dominance along the river since the last vegetation data was collected in 2009. Starting in 2007, the U.S. Forest Service began mechanically removing tamarisk (*Tamarix* spp.) from lands under their management along the river. The results of our data show that tamarisk was the 6th most dominant species in 2009, but no tamarisks were detected within plots in 2021/22 during our field-based survey. Goodding’s willow (*Salix gooddingii*) was the most dominant

Table 1

UAV multispectral image-derived land cover types and their mean area distribution (in percent) across the entire UAV image extent of approximately 10 ha, on average, including the riparian corridor and adjacent upland areas.

Plots	Water	Gravel bars	Herbaceous Cover	Cattails	Riparian woody cover	Juniper	Upland bedrock/bareground	Shadow
1–2	0.6	1.7	4.4	5.5	5.8	7.3	34.3	38
12	0.3 %	0.9 %	18 %	2.8 %	10.5 %	16 %	51 %	0.4 %
13–14	0.4	0.8	0.7	0.5	10	20.4	63	3.8
7–8	0.4	1	1.6	1.4	14	6.6	74	0.9
9	1.5	2	1.1	3.5	8.9	11	69	3
10	0.8	0.5	2.6	1.5	14.6	9.1	62	8.7
23	1.8	1.6	0.4	0.5	5	24	65	1.3
Mean	0.8	1.2	4.4	2.2	9.9	13.5	59.8	8

species within the plots in 2009 and remained the most dominant in 2021/22. Coyote willow (*Salix exigua*) was the 7th most dominant species in 2009 but became the 2nd most dominant in 2021/22. Velvet ash (*Fraxinus velutina*) was 2nd most dominant in 2009 and became the 3rd most dominant species in 2021/22. Coyote willow, Gooding's willow, and Velvet ash are the three most dominant species within the 26 plots studied.

3.5. Lidar-derived geomorphic changes

A total of 17 plots were successfully surveyed with the terrestrial laser scanner (Tango et al., 2024). Each plot was scanned with 15–25 scans from different scan-position locations. The individual scans at each plot were all successfully registered together to create a full image of the area. Lidar points within the channel were not included in any analysis because the near-infrared laser of the lidar scanner does not penetrate through water to the channel bed. Topographic data from the 2009 USFS survey were compared to the 2021–2022 lidar data for geomorphic change detection, which indicated that there were statistically significant changes at all 9 plots available for comparison (Table 2). Other than Plot #16, more deposition than erosion occurred at all plots, and most of the changes, whether deposition or erosion, ranged between 0 cm and 50 cm. The deposition largely occurred in the floodplain nearest the channels. The notable exception is again Plot #16, which has eroded on the south bank further away from the channel. Importantly, the results were consistent with the change detection results from the comparison between the 2009 and 2021 USFS survey data.

3.6. Geomorphic survey

Channel width to depth ratios ranged from 8.71 to 86.77 m with a mean of 34.7 m in 2009 compared with a range of 8.97–71.38 m and a mean of 29.4 m in 2022. Statistical comparison ($\alpha = 0.05$) showed that this change was a statistically significant decrease in W/D ratio (narrowing and deepening of the channel) between 2009 and 2021/22 (p -value = 0.04). Entrenchment ratios that are between 1.0 and 1.4 are considered entrenched, between 1.41 and 2.2 to be moderately entrenched, and >2.2 are slightly entrenched. Only three plots (11, 16, and 44) in 2009 and two plots (13 and 16) in 2021/22 were moderately to slightly entrenched. All other plots in 2009 and 2021/22 were not entrenched according to the Rosgen Classification System (Rosgen, 1994).

3.7. RTK habitat classification

Of the total instream habitat across all plots surveyed, pools represented 37.6 % of total stream reaches with run habitats representing 32.7 %, riffles habitats representing 13.7 %, shallows and cattails 15.5 %, with large rocks making up the remainder of 0.4 %. The plots within the upper section of river from Granite Creek to Hell Canyon contained

all the cattail/backwater habitat surveyed.

4. Discussion

4.1. Riparian vegetation change

The results of this study suggest that an approximate 40% reduction in baseflow over the last two decades has coincided with an approximately 30% increase in woody species and nearly 10% increase in cattail cover, especially in the upper reaches of the river above Hell Canyon. The changes in vegetation have served to confine the stream channel in many areas (Figs. 8 and 9). Reductions in base flow in other studies have also been linked to vegetation encroachment and channel confinement. In these cases, a lack of large scouring flood events (Dean and Topping, 2019; Grams et al., 2020) and dewatering of wet streamside areas, allows for the expansion of deeper-rooted, woody species (Sankey et al., 2015) which confine the channel and can create a reinforcing feedback loop, focusing stream energy, which continues to deepen the channel (Dean and Topping, 2019). It appears that these shifts in vegetation are influencing channel shape and size on the Upper Verde away from wider, more shallow channels to narrower, deeper channels, illustrated by the reduction in exposed water surface in NAIP imagery comparisons over the last fifteen years, which results in less suitable habitat for native fish species.

Merritt (2013) found that floodplain development is associated with flow duration, frequency, and vegetation establishment. During periods of lower flows, floodplain accretion is often associated with increases in bank vegetation. Plots surveyed in this study have experienced a mean vertical change of +0.3m suggesting that the small to medium-sized flows which have occurred over the last decade have primarily deposited sand and gravel near the channel (Tango et al., 2024), which has been trapped by streambank vegetation. Decreasing width to depth ratios suggest that the channel has become deeper while the adjacent streambanks have aggraded. Flooding events that occur with regular periodicity (<25-year return intervals) appear to be concentrating energy within the channel which disconnects the channel from the floodplain and serves to reinforce the confinement of the stream channel in many areas. While we do not have previous instream habitat data to compare newly gathered RTK data to, comparisons of repeat photography through time suggest that this cycle is leading to a reduction in the number and size of shallow riffle habitats, which support several native species including Spikedace, Speckled Dace, Longfin Dace and Desert Sucker (Rinne and Miller, 2006).

Research along other river systems within the region have shown similar trends (Scamardo et al., 2022). Dean and Topping (2024) recently documented an example of the effect of vegetation encroachment on the geomorphology within Moenkopi Wash, a tributary to the Little Colorado River. The results of the study suggest that reductions in total annual flow and annual peak flows dating back to the 1940s coincided with the establishment of dense woody-vegetation along the streambank resulting in channel narrowing and vertical aggradation of the streambank. These changes have had far reaching effects on how sediment moves through the river system. In this case, it has reduced the amount of sediment ultimately supplied downstream to the Colorado River because much of the available material is locked up in the floodplain and inaccessible during peak flow events due to confinement by woody vegetation. In the case of the Little Colorado River, they suggest that a re-widening of the channel is unlikely to occur on its own and could require large-scale vegetation removal to reverse the encroachment on the channel. This may also hold true on the Verde River, where results from this study documented the establishment of dense stands of willow and ash trees in the last two decades.

4.2. Evaluation of new methods

The use of UAV, RTK and terrestrial lidar technology undoubtedly

Table 2

Summary of the geomorphic changes observed from the lidar data at 9 plots with matching USFS survey data from 2009.

Plot	Reach	Avg. Vert Change	Min Deposition/Max Erosion	Max Deposition
6	#1	0.27m	−0.34m	0.90m
9	#1	0.10m	−0.37m	0.30m
10	#1	0.37m	0.13m	0.76m
16	#1	−0.22m	−0.37m	0.15m
18	#1	0.25m	0.18m	0.60m
23	#1	0.10m	−0.23m	0.23m
33	#2	0.34m	0.11m	0.97m
41	#2	0.28m	0.13m	0.59m
44	#2	0.29m	−0.36m	0.54m



Fig. 8. Photographs of Plot 10 located within the uppermost section of the UVR looking upstream from a. 1997, b. 2004, c. 2009 and d. 2021 illustrating the narrowing of the stream channel due to changes in cattail density.

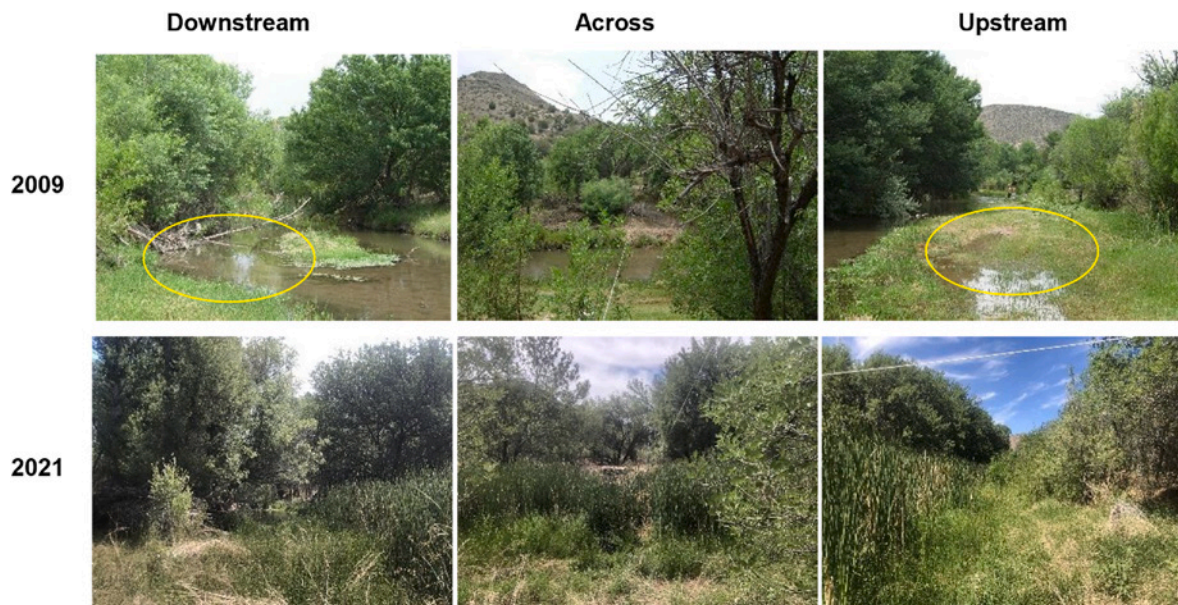


Fig. 9. Photos taken of Plot 9 in 2009 vs. 2021 looking downstream, across and upstream from a similar perspective. Expansion of cattails obscures the view of the channel and dominates the wetted perimeter in 2021. The transformation of side channels (circled) into dense cattail stands illustrates the loss of in-channel habitat and narrowing of the stream channel.

collected more data in the field in a shorter amount of time than traditional methods which include labor intensive field-based survey methods and aerial photogrammetry from an airplane. Consideration should be given to the amount of computer processing time and hardware requirements needed to work with such large datasets. We processed our datasets on individual desktop computers to examine each plot individually and extract statistical summaries. Larger areas imaged by UAVs and terrestrial lidar or many plot areas being simultaneously processed would require cluster computing or cloud computing. For some management agencies, the instruments may not currently be an option as the initial investment in hardware and instrumentation can cost up to \$30,000 (Sankey et al., 2019). However, after the initial

investment, land managers can collect repeated data consistently and efficiently at user-defined frequency for little additional investment. This translates to reduced costs over time and allows adaptive management with detailed quantitative monitoring data. In particular, the high spatial resolution and geolocational accuracy of UAV and lidar data can reveal smaller and nuanced changes from sediment deposition or erosion (Sankey et al., 2024; Solazzo et al., 2018) or in vegetation cover (Sankey et al., 2018, 2019, 2021a,b, 2024, 2025) than cannot be easily observed from field-based measurements. Furthermore, these methods and associated data can be combined with historical legacy datasets from field-based transects to document statistically significant landscape-scale changes (Tango et al., 2024). Such ability to combine

various monitoring datasets to quantify landscape-scale changes in vegetation and stream morphology is invaluable in aiding in management decision making. As more streamlined image processing workflows are increasingly becoming available, it makes these technologies more easily adoptable. Future studies can use the UAV, lidar, and RTK data demonstrated in this study as a benchmark dataset to continue high spatial resolution monitoring on the Upper Verde River.

Similarly, the use of eDNA sampling methods in this study allowed for sampling across a larger area with fewer personnel. As is often the case in studies that span multiple decades, sampling methods may vary as resources and personnel change. In the case of this study, resources were no longer available to conduct backpack electrofishing as had previously been done. While eDNA sampling is a powerful tool for monitoring the presence and absence of select species there are limitations to understanding population trends without quantitative sampling (Rees et al., 2014). For this reason, we could not statistically analyze fish populations from 2006 and were limited to only pointing out the trend we observed of a steep, downward decline of native populations in the upper portions of the river. However, the use of the eDNA method did allow more frequent sampling of the study area, resulting in more options of when to sample during the year. It also allows for cross verification of electrofishing results and the expansion of the area being sampled for relatively low cost. For this reason, we suggest that a combination of the approaches is ideal but varies depending on the management objectives.

4.3. Implications for restoration

The lower section of our study area from Hell Canyon to Sycamore Canyon appears to show greater resistance to change compared to the upper half, as evidenced by the lack of new cattail establishment and persistence of native fish species. This section of river contains more canyon-bound reaches, which contribute more cobble and gravel bed-load material to the channel, helping to maintain riffle habitats preferred by several native fish species (Rinne and Miller, 2006). The surrounding watershed along this portion of river has also been less impacted by human development and appears to still maintain adequate hydrologic (Beisner et al., 2018) and sediment processes (Rinne and Miller, 2006) to continue to provide suitable native fish habitat. While competition and predation from nonnative aquatic species, including several fish species, bullfrogs (*Lithobates catesbeianus*) and crayfish (*Orconectes virilis*) still exists, it appears that maintaining habitat integrity creates greater resilience for native fish populations.

In contrast, many reaches in the uppermost section of this study flow through broad, alluvial valleys in which geomorphology is more readily influenced by changes in streamside vegetation (Neary et al., 2012). The development of long, deep pools within many reaches of the UVR appear correlated with the confinement of the channel by cattails and woody vegetation which produces habitat less suitable for native species and more favorable for non-native fish and other undesirable aquatic species. The entrenchment of stream channels and disconnection with the floodplain has been linked in other cases to losses in habitat heterogeneity and decreases in total aquatic productivity, impacting important plant and animal species including sensitive salmon and trout species in other studies (Schneider et al., 2022).

While effective at reducing non-native intrusion, fish barriers do not address underlying habitat degradation. Techniques such as Processed-based and Stage 0 restoration which use natural processes to physically manipulate impacted channels by attempting to reconnect the stream channel with the adjacent floodplain may be more effective (Ciotti et al., 2021; Hinshaw et al., 2022) at increasing wetted area, which increases habitat diversity and benefits targeted species (Beechie et al., 2010). Further investigation of the differences in hydrologic and geomorphic processes between the upper and lower sections of the study area may help determine the applicability of future restoration methods and their potential impact on restoring habitat suitability for native fish

species.

5. Conclusions

The disappearance of native fish species from the upper 10 km of the UVR coincides with a 40% reduction in baseflow and 30% increase in woody species within the riparian area, suggesting that in the absence of direct human impact, changes in hydrologic and geomorphic processes are most likely responsible for reducing suitable habitat for native aquatic species within the river. To date, there has not been a comprehensive sediment budget and targeted hydrologic analysis of the upper section of the river which is essential to better understand the influence of these physical processes on ecology of the river. Other activities such as groundwater pumping and gravel mining within major tributaries may also be influencing these processes and should be considered. While the continuing loss of baseflows may weigh disproportionately on the future ability of the river to support native species, restoring functionality between the main channel and the adjacent floodplain may provide for the retention of native species habitat in at least some portions. Similarly, the identification and conservation of hydrologic and geomorphic processes which continue to support native fish populations within existing sections of the river, may also prove essential to the protection of what remains of native species habitat.

CRedit authorship contribution statement

Jackson M. Leonard: Writing – review & editing, Writing – original draft, Supervision, Resources, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Temuulen Ts Sankey:** Writing – review & editing, Writing – original draft, Supervision, Resources, Project administration, Methodology, Investigation, Formal analysis, Conceptualization. **Lauren L. Tango:** Writing – review & editing, Writing – original draft, Software, Methodology, Investigation, Formal analysis. **Roger W. Tyler:** Methodology, Investigation, Formal analysis, Data curation.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Data availability

Data will be made available on request.

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