RIPARIAN VEGETATION ALONG EPHEMERAL STREAMS IN ARID AND SEMI-ARID ENVIRONMENTS AND IMPLICATIONS FOR CLEAN WATER ACT JURISDICTION

by

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LIST OF ABBREVIATIONS

Abbreviation	Description
BLM	Bureau of Land Management
CWA	Clean Water Act
DEM	Digital Elevation Model
EPA	Environmental Protection Agency
GIS	Geographic Information System
IHA	Indicators of Hydrologic Analysis
NAM	North American Monsoon
NDVI	Normalized Difference Vegetation Index
NWPR	Navigable Waters Protection Rule
OHWM	Ordinary High Water Mark
OOB	Out of Bag
SWANCC	Solid Waste Agency of Northern Cook County
TNW	Traditionally Navigable Waterway
USACE	United States Army Corps of Engineers
USGS	United States Geological Survey
VPD	Vapor Pressure Deficit
WOTUS	Water of the United States

ABSTRACT

The Clean Water Act protects water quality, but its jurisdiction is limited to traditionally navigable waters and those tributaries that have relatively permanent flow (i.e., perennial flow) or a significant nexus (i.e., they significantly affect water quality in downstream waterways). In the arid southwestern United States, most streams are intermittent (flowing seasonally) or ephemeral (flowing only in response to precipitation), therefore they do not meet the relatively permanent standard. Determining whether a given ephemeral stream has a significant nexus to a downstream waterway is complicated, especially in arid regions where geomorphic evidence of flow events can persist for long periods of time. Riparian vegetation is used in field-based jurisdictional determinations, however the riparian vegetation of ephemeral streams in arid and semi-arid regions is understudied. This dissertation contributes to the existing empirical data on the riparian vegetation of ephemeral streams in the southwestern United States by examining the longitudinal spatial patterns of riparian communities along six ephemeral streams in New Mexico and Arizona using remote sensing. Normalized Difference Vegetation Index (NDVI) was used to determine the size and cover of riparian vegetation, which was then correlated to topographic, climatic, geomorphic, and streamflow characteristics. Riparian vegetation responds much more strongly to topographic and climatic variables than to streamflow variables. These results indicate that the riparian vegetation of ephemeral streams responds to different environmental factors compared to riparian vegetation along intermittent or perennial rivers. The results

presented here can inform Clean Water Act policies regarding ephemeral streams in the southwestern United States.

I. INTRODUCTION

Context

The Clean Water Act (CWA) regulates water quality in the United States, but protections are restricted to certain waterways called Waters of the United States (WOTUS). Protected waterways include territorial seas, interstate waterways, and traditionally navigable waterways (TNW), along with adjacent wetlands and certain tributaries. Determinations as to which tributary streams are jurisdictional WOTUS under the CWA are currently made based on two standards outlined in the Supreme Court decision on the case *Rapanos v. United States*, the relatively permanent standard and the significant nexus standard. To meet the relatively permanent standard, a given stream must have a continuous connection to downstream protected waterways via surface water for at least three months per year. The significant nexus standard can be met by one stream or by several streams together, but the stream(s) in question must affect the water quality of a downstream protected waterway.

Ephemeral (flowing only in direct response to precipitation) and intermittent streams (flowing seasonally) comprise the vast majority of total stream length in the arid and semi-arid southwestern United States (Goodrich et al. 2018), however water quality protections for them are tenuous. Headwater streams can contribute a majority of flow to downstream TNWs (Alexander et al. 2007), and inputs of pollutants and nutrients to headwaters can impact water quality far downstream (Freeman et al. 2007). For these reasons, the protection of headwaters, whether they are perennial or not, is crucial to maintaining good water quality in the southwestern United States.

In the past, the U.S. Army Corps of Engineers (USACE) and the Environmental Protection Agency (EPA), the agencies responsible for CWA regulation, have considered stream network connectivity in making jurisdictional decisions (Department of Defense and Environmental Protection Agency 2015). Under the 2015 Clean Water Rule, streams that had a hydrologic, geomorphic, or biologic nexus to downstream TNWs were considered jurisdictional (Alexander et al. 2015). The inclusion of connectivity analysis in making jurisdictional determinations would likely increase the number of protected streams and would have a positive impact on downstream water quality. An assessment of connectivity can guide water quality management decisions under any definition of WOTUS, particularly for local or state regulatory agencies, landowners, or businesses.

A refined understanding of connectivity is increasingly more important in a changing climate. Though climatic and hydrologic projections for the southwestern United States indicate that the climate will become warmer and drier and overall streamflows will decrease (Graf 1988; Seager et al. 2013; Allen et al. 2019), predictions for the North American Monsoon are less certain (Colorado-Ruiz et al. 2018). Some models predict more extreme precipitation events during future monsoons (Meyer and Jin 2017), which would result in higher-magnitude, and potentially longer-lasting flow events in ephemeral streams.

This dissertation integrates remote sensing and statistical analyses to evaluate longitudinal patterns in the riparian vegetation of ephemeral streams in the southwestern United States in order to determine whether such patterns are related to hydrologic processes. This region is particularly important to focus on as federal water quality protections in many streams in the southwestern United States are tenuous. Most streams

in the region are characterized as non-perennial, yet their infrequent flow may still have a significant impact on downstream water quality. The overarching research question I address here is: Can riparian vegetation of arid, ephemeral streams be used to assess hydrologic connectivity throughout arid stream networks and inform water quality management in the southwestern United States? This question is especially critical to this region where hydrologic connectivity is infrequent, yet highly important to surface water processes, ecological functioning, and contaminant storage and dispersal.

Literature review and theoretical framework

Though headwater streams supply much of the water to downstream navigable waterways (Alexander et al. 2007), they are often overlooked with regard to water quality protections. Non-perennial streams are believed by many land managers to make insignificant contributions to downstream water quality (Armstrong et al. 2012). However, flow events in ephemeral streams can mobilize accumulated material in the stream bed and initiate biogeochemical responses in alluvial soils (Larned et al. 2010).

Water quality management has important human health implications. Contaminants such as heavy metals can be dispersed via flowing water and introduced to drinking water supplies, exposing humans to toxic levels (Graf 1988). Humans can also be exposed to contaminants by consuming fish from polluted water sources. An analysis of mercury levels in fish throughout the United States found clusters of higher than expected mercury levels in the southwestern United States (Eagles-Smith et al. 2016). Consumption of fish from polluted waters can have serious effects on human health, and may disproportionately affect low-income communities (Pulford et al. 2017). Excellent water quality management also has benefits beyond the protection of human health,

including conservation of native biodiversity. Inputs of phosphorus and nitrogen, usually originating from agricultural or urban areas, leads to reductions in fish populations (Carpenter et al. 1998). Improvements in water quality made possible by the reduction of pollutant discharges are correlated to increased fish species richness (Perkin and Bonner 2016).

In the southwestern United States, many activities affect water quality including mining, agriculture, and urbanization. Mining has occurred in the area since before European contact, and today the region is mined for copper, coal, uranium, aggregates, and other minerals (Arizona Geological Survey 2019; NM Bureau of Geology & Mineral Resources 2019). Mining activities can impact water quality by introducing radioactive materials (Dias da Cunha et al. 2014), heavy metals (Gray et al. 2015), and acid mine drainage (Johnson and Hallberg 2003). Agricultural activities, including livestock grazing and tilled agriculture, introduce nutrients that alter water chemistry into streams. Inputs of nitrogen (used in fertilizer) into streams causes algal blooms, which then deplete dissolved oxygen in the water (Horrigan et al. 2002). The lack of oxygen in the stream is detrimental to fish and other aquatic organisms. Runoff from urban areas also contributes nutrients (e.g., nitrogen and phosphorus) to streams (Carpenter et al. 1998).

The importance of including structural and functional stream network connectivity in water quality management decisions cannot be overstated (Alexander et al. 2015). The impacts of pollutant discharge and other land management activities affecting water quality is best evaluated at a watershed scale (Sullivan, Rains, and Rodewald 2019) because the distances travelled by pollution inputs are largely unknown. The distances that sediments and pollutants are transported in dryland streams are

difficult to estimate because stream discharge does not increase downstream in arid regions (Graf 1988). An assessment of connectivity throughout the stream network can help determine the potential impacts of polluted headwaters streams on downstream water quality. I propose that an analysis of riparian vegetation patterns can indicate a relative degree of stream network connectivity.

The dependence of the riparian forest on surface and subsurface flow supports its role as a possible indicator of connectivity. Myriad definitions for connectivity exist, but most authors refer to it as exchanges and interactions among patches or components of a landscape (Liebowitz et al. 2018; Wohl et al. 2019). In stream networks, researchers focus on ecological, hydrological, and geomorphic connectivity. Ecological connectivity, a measure of interactions among patches of habitat, has been used in landscape ecology (Taylor et al. 1993) and biodiversity management (Ward et al. 1999) since the late twentieth century. In the context of ecological connectivity, riparian areas are often considered as corridors that allow faunal species to move among larger patches of habitat (Naiman et al. 1993). Hydrologic and geomorphic connectivity involve the movement of water and sediment through a system, and are dependent on factors influencing runoff and infiltration, such as topography, land use/land cover, and climate (Bracken and Croke 2007). In arid environments, assessments of geomorphic and hydrologic connectivity are complicated because differences in topographic relief and climate alter patterns and processes (Doyle and Bernhardt 2011). Runoff in arid environments is dominated by Hortonian overland flow from steep slopes, rather than by saturation flow that more frequently occurs in humid environments (Bracken and Croke 2007). Sediment transportation in arid environments is associated with higher-magnitude, lower-frequency

floods than those responsible for sediment transport in humid regions (Graf 1988). Because of these more complex factors, biological indicators can be used as a proxy for geomorphic or hydrologic indicators in establishing connectivity.

Surface water hydrology is a fundamental control on riparian vegetation. Poff et al. (1997) consider a stream's hydrology to be the key driver of fluvial processes, including those that govern riparian forest composition. The magnitude, frequency, timing, duration, and rate of change of flow events for a stream comprise the stream's flow regime, and biota associated with the stream are adapted to the flow regime (Poff et al. 1997). Hydrogeomorphic processes such as scouring and deposition create habitat for riparian vegetation (Bendix and Hupp 2000). In the southwestern United States, riparian trees are adapted to survive drought, utilize moisture during flow events, and tap into shallow riparian groundwater replenished by flow events. In this region, upland vegetation is typically quite sparse, while riparian vegetation is often more abundant.

Riparian vegetation generally increases in size and density along a transverse gradient from the uplands to the channel and from xeric to hydric sites (Stromberg, Wilkens, and Tress 1993; Sponseller and Fisher 2006). Differences in species composition exist along these same gradients, likely as a result of moisture patterns in the system and/or the presence of stream flow. These differences are likely also related to the presence of soil and/or alluvium in the channels and floodplains. Meso- and hydro-riparian forests occur along perennial or intermittent streams, and often rely on relatively consistent sources of flow including groundwater and seasonal snow melt runoff. These forests are typically composed of native trees with higher water needs, such as cottonwood (*Populus* spp.), willow (*Salix* spp.), walnut (*Juglans major*), and Arizona

sycamore (*Platanus wrightii*) (Lowe 1961), as well as exotic species such as saltcedar (*Tamarix* spp.) and Russian olive (*Eleagnus angustifolia*). Meso- and hydro-riparian tree species along perennial or intermittent streams rely on seasonal floods. Germination of these species coincides with spring high flows from snow melt runoff (Stromberg, Patten, and Richter 1991). Spring floods create habitat for seedlings by scouring away existing vegetation (Stromberg, Patten, and Richter 1991). Regular floods also replenish alluvial aquifers, which are needed to support mature trees through drier seasons (Lytle and Merritt 2004).

Xeroriparian forests thrive along ephemeral streams where access to surface water depends on rainfall run-off events that are short-lived and unpredictable, but deeper groundwater may be present. Xeroriparian forests are composed of upland plants that are adapted to drought and other harsh conditions (Beauchamp and Shafroth 2011). Such species include mesquite (*Prosopis* spp.), catclaw acacia (*Acacia greggii*), desert willow (*Chilopsis linearis*), and paloverde (*Parkinsonia* spp.) (Lowe 1961; Johnson and Lowe 1985; Johnson, Bennett, and Haight 1989). The relationships between surface run-off, surface-groundwater connections, flow regime, and xeroriparian vegetation is less clear than it is for meso- and hydro-riparian communities. However, there is some evidence that fluvial processes play a role in xeroriparian community composition, even on ephemeral streams (Douglas et al. 2018). Many xeroriparian species have deep roots, sometimes reaching tens of meters below the surface to access deep groundwater (Canadell et al. 1996). Others have laterally extensive roots that can absorb near-surface water over a broad area (Casper and Jackson 1997). Their root morphology, along with other life-history strategies, allows them to survive in areas where surface flow is

infrequent and unpredictable. Distinct from this aspect of their growth strategies, xeroriparian species use flowing water for seed dispersal, and along ephemeral streams surface flow can be the dominant downstream dispersal agent (Gaddis et al. 2016). Evidence of this dispersal mechanism is a reliable indicator of surface flow connectivity of ephemeral streams.

Analyses of remotely sensed imagery of riparian vegetation have been used to assess habitat (Villareal, van Reper, and Petrakis 2014) and hydrologic connectivity (Gallart et al. 2016; Hamdan and Schmeeckle 2016) in the southwestern United States. Satellite-derived vegetation indices are highly correlated to above-ground biomass in a variety of vegetation types (Jensen 1983). Normalized Difference Vegetation Index (NDVI) is a ratio of differences in reflectance of red and infrared energy (infrared – red/infrared + red) that is commonly used to analyze vegetation changes over time (Fu and Burgher 2015; Hamdan and Schmeeckle 2016). Because vegetation cover in the southwestern United States is highest near stream channels (Cadol and Wine 2017; Stromberg et al. 2017), NDVI can be a useful tool in analyzing vegetation patterns along a stream network (Manning, Julian, and Doyle 2020).

The purpose of this research is to identify the relationship between hydro-geomorphic connectivity and connectivity of the riparian vegetation to answer the overarching research question: Can riparian vegetation of arid, ephemeral streams be used to assess hydro-geomorphic connectivity throughout stream networks and inform water quality management in the southwestern United States? First, I analyze the ongoing CWA policy changes and their impact on the protection of ephemeral streams (Chapter II). Then, I identify the environmental controls on the spatial patterns of the riparian

community along ephemeral streams in the southwestern United States (Chapter III). Finally, I determine how the connectivity of the vegetation is related to hydro-geomorphic connectivity (Chapter IV).

II. POLICY REVIEW OF WATERS OF THE UNITED STATES AND ITS APPLICATION TO EPHEMERAL STREAMS IN THE ARID SOUTHWESTERN UNITED STATES

Introduction

The Clean Water Act (CWA) was enacted in 1972 as an amendment to the Federal Water Pollution Control Act with the intent to regulate pollution in surface waters of the United States. As a federal law, the CWA provides minimum water quality standards which must be met in all states. States and tribal entities are permitted to extend protections or raise standards, but they are not permitted to have standards lower than those set forth by the CWA.

CWA regulations apply only to certain jurisdictional waterways, which are known as the Waters of the United States (WOTUS). The definition of WOTUS has changed numerous times since the CWA was enacted, and how WOTUS is defined greatly affects whether a given stream is determined to be jurisdictional. At the time of this writing, the current definition states that traditionally navigable waterways (TNW), territorial seas, and interstate waterways are jurisdictional, as are adjacent wetlands, tributaries, and impoundments to the aforementioned waterways when used for interstate commerce (Environmental Protection Agency 2021). In December of 2021, the U.S. Environmental Protection Agency (EPA) and the U.S. Army Corps of Engineers (USACE), the two federal agencies that regulate the CWA, proposed a revised definition of WOTUS which includes traditionally navigable waterways, interstate waterways, and territorial seas and their impoundments and adjacent wetlands. Tributaries to such waterways are also considered WOTUS if they meet one of two standards: the relatively permanent standard and the significant nexus standard. To meet the relatively permanent standard, a given

waterway must maintain a continuous connection to a downstream jurisdictional waterway via surface water for a significant duration (> 3 months). To meet the significant nexus standard, a given waterway must, alone or with other nearby waterways, impact the water quality of a downstream jurisdictional waterway.

Jurisdictional determinations regarding which tributaries are protected by the CWA have serious implications for the quality of drinking water, especially in areas where water availability is limited. In the southwestern United States, the vast majority of streams are either intermittent (flowing seasonally) or ephemeral (flowing only in direct response to precipitation) (Goodrich et al. 2018), meaning they likely do not meet the relatively permanent standard. Whether or not such streams qualify for CWA regulation by the significant nexus standard has historically been difficult to determine.

The protection of ephemeral streams in particular has been a hotly-debated issue in American politics for the last few decades that has not yet been resolved. With a new definition currently in the process of being enacted, a review of WOTUS policy and how it is applied to ephemeral streams in the southwestern United States is in order. This chapter focuses on the history of the definition of WOTUS, its application to streams in the southwestern United States, and what knowledge gaps exist that serve as barriers to water quality protections in the southwestern United States.

History of the definition of the Waters of the United States

"Waters of the United States" is not defined within the CWA, rather, the definition is left to the discretion of the EPA and USACE. Before 1985, WOTUS was generally defined as waterways which were considered to be navigable or used for interstate commerce, with a broad interpretation (U. S. Library of Congress,

Congressional Research Service 2019). Since 1980, the Supreme Court of the United States has ruled on WOTUS three times. The first was in 1985 in *United States v. Riverside Bayview Homes, Inc.* (hereafter, *Riverside Bayview*). In that case, the Court determined that wetlands adjacent to navigable and/or interstate waterways significantly impact the water quality of those waterways and are subject to CWA jurisdiction (United States v. Riverside Bayview Homes, Inc. 1985). The *Riverside Bayview* case surrounded a housing development that was to be built on wetlands adjacent to Lake St. Clair (Michigan, USA). The decision clarified that because the adjacent wetlands contained vegetation that required regularly saturated soils, they were sufficiently connected to nearby Lake St. Clair to impact water quality and provide nursery habitat for aquatic lake-dwelling fauna.

The second Supreme Court case regarding WOTUS was in 2001 in the case of *Solid Waste Agency of Northern Cook County v. United States Army Corps of Engineers et al.* (hereafter, *SWANCC*). SWANCC was not granted a Section 404 permit to dispose of nonhazardous solid waste product into permanent and seasonal ponds that had developed in an abandoned gravel pit. USACE denied the permit because the ponds were used by migratory birds, which USACE claimed qualified the ponds as interstate waterways, which are regulated by the CWA. The *SWANCC* decision limited the scope of the CWA to WOTUS and waters with a "significant nexus" (Solid Waste Agency of Northern Cook County v. United States Army Corps of Engineers et al. 2001, p. 161) to downstream WOTUS. Chief Justice William Rehnquist wrote in the majority opinion that although the Supreme Court had decided wetlands qualified as WOTUS in the *Riverside Bayview* decision, the wetlands in that case had a significant nexus to a navigable

waterway, whereas the ponds in the *SWANCC* case did not (Solid Waste Agency of Northern Cook County v. United States Army Corps of Engineers et al. 2001).

In 2005, the definition of WOTUS was presented to the Supreme Court for a third time in the cases *John Rapanos et ux., et al. v. United States* and *June Carabell et al. v. United States Army Corps of Engineers et al.* John Rapanos was sued in civil court after backfilling wetlands on his property that were adjacent to ditches that eventually drain into a navigable waterway, but the ditches were separated from the waterway by several miles. In a similar case, June Carabell was denied a permit to fill a wetland which was in close proximity to a ditch, but was isolated from the ditch by a berm. The two cases were consolidated and are referred to here jointly as *Rapanos*.

Though the cases were vacated, the Court did not reach a majority opinion on the *Rapanos* case. Rather, the Court produced two tests for jurisdictional determinations. Justice Antonin Scalia, who wrote a plurality opinion in the *Rapanos* case, stated the opinion that ephemeral streams and other temporary water features should not be protected by the CWA, and that the Act only applies to "relatively *permanent* bodies of water" (Rapanos et ux., et al. v. United States 2006, p. 15). This opinion was based on language within the CWA stating that streams and rivers are within CWA jurisdiction. The plurality's opinion is referred to as the "Bright-Line Rule."

Justice Anthony Kennedy wrote his own opinion and disagreed that water bodies must be "relatively permanent" in order to qualify for protections. In his concurring opinion, he cited the example of the Los Angeles River, noting that many streams in the western United States, though dry much of the time, can discharge considerable amounts of water in times of flood (Rapanos et ux., et al. v. United States 2006). He suggested that

the significant nexus test be evaluated on a case-by-case basis (U.S. Library of Congress, Congressional Research Service 2019). Following the *Rapanos* decision, the EPA issued updated guidance excluding ditches and erosional features (i.e., rills and gullies) and stating that the significant nexus test will be applied based on assessments of hydrological and ecological function of tributaries and their biological, chemical, and physical connections to downstream waterways (Environmental Protection Agency and U.S. Army Corps of Engineers 2008).

In 2015, the EPA and the USACE issued new guidance which was known as the Clean Water Rule. This Rule defined WOTUS as TNWs, interstate waters, territorial seas, tributaries and adjacent waters, and waters found to have a significant nexus to downstream protected waterways (U S. Army Corps of Engineers and Environmental Protection Agency 2015). The Rule defined a tributary as "waters that are characterized by the presence of physical indicators of flow—bed and banks and ordinary high water mark—and that contribute flow directly or indirectly to a traditional navigable water, an interstate water, or the territorial seas" (U.S. Army Corps of Engineers and Environmental Protection Agency 2015, p. 37058). This language expanded protections to many stream channels throughout the nation. Though the Clean Water Rule was informed by science on watershed connectivity (Alexander et al. 2015), it was met with some criticism from the scientific community regarding its resilience in the context of climate change (Faust et al. 2016) and its exclusion of certain human-made water features (Gannon, Kinner, and Lord 2016).

The Clean Water Rule received harsh criticism from industry and Congress, resulting in swift motivations for legislative action to block the Rule (Hopkinson 2015).

Multiple lawsuits challenging the scope of CWA jurisdiction were filed, which resulted in the Rule being valid only in certain states (U. S. Library of Congress, Congressional Research Service 2019). The EPA and USACE issued a repeal of the Clean Water Rule in September 2019 (U. S. Army Corps of Engineers and Environmental Protection Agency 2019), and it was replaced by the Navigable Waters Protection Rule (NWPR) in January of 2020. The NWPR explicitly excluded ephemeral streams from CWA jurisdiction; intermittent streams were jurisdictional, but only if their flow derived from snowmelt or from seasonal high water tables (U. S. Army Corps of Engineers and Environmental Protection Agency 2020).

The NWPR came under fire from the scientific community and other stakeholders. Scientists asserted that the Rule ignored science in favor of loose regulations that would damage ecologically-important wetlands and headwaters streams (Sullivan, Rains, and Rodewald 2019; Sullivan et al. 2020). The Rule was vacated in September of 2021 by an Arizona federal court (*Pascua Yaqui Tribe et al. v. Environmental Protection Agency, et al.*). In that case, six Native American tribes (Pascua Yaqui Tribe, Quinault Indian Nation, Fond Du Lac Band of Lake Superior Chippewa, Menominee Indian Tribe of Wisconsin, Tohono O'odham Nation, Bad River Band of Lake Superior Chippewa) filed a complaint challenging the repeal of the Clean Water Rule and the issuance of the NWPR on the grounds that they were "arbitrary and capricious" (*Pascua Yaqui Tribe et al.*, p. 3) and represent an overstep of authority by the agencies. The Tribes all live in and depend on areas with wetlands and/or ephemeral streams that lost protections when the Clean Water Rule was repealed. The ancestral lands of the Pascua Yaqui Tribe and the Tohono O'odham Nation were threatened by

proposed mining operations, which would damage the ephemeral streams in the area. The Arizona District Court denied a hold on the case, vacating the NWPR.

Per an Executive Order from the Oval Office on January 21, 2021, the EPA was asked to review all regulations issued between January 2017 and January 2021, including the NWPR (Biden, Executive Order 13990). Thus, after the NWPR was vacated, the EPA and USACE announced a proposed rule to replace it. It is fundamentally similar to the post-*Riverside Bayview* guidance with some amendments, most of which are related to the post-*Rapanos* guidance on the relatively permanent and significant nexus standards (Table 1).

Table 1. Proposed amendments to the post-*Rapanos* Waters of the United States guidance. Adapted from Department of Defense and Environmental Protection Agency (2021).

Jurisdictional waters (post-*Riverside Bayview* regulations)	Proposed amendment(s)
Traditionally navigable waters	No amendments proposed.
Interstate waters	The agencies are considering including waters crossing tribal boundaries as "interstate."
Territorial seas	No amendments proposed.
Other waters (i.e., wetlands)	The relatively permanent test and the significant nexus test would replace the 1986 language regarding commerce. The language may be simplified to avoid naming specific waterbody types (i.e., wetlands).
Impoundments	Waterbodies that fall into the "other waters" category would be excluded from this category.
Tributaries	The relatively permanent test and the significant nexus test would be applied to tributaries of TNWs, territorial seas, interstate waters, and impoundments to any of the above. References to "other waters" from the 1986 guidance would be deleted.
Adjacent wetlands	The proposed rule would explicitly include wetlands adjacent to TNWs, interstate waters, territorial seas, wetlands adjacent to impoundments to the above, and jurisdictional tributaries to the above providing they meet the same standard used to determine jurisdiction to the impoundment or tributary (i.e., relatively permanent or significant nexus). Adjacent wetlands to "other waters" would be assessed in the "other waters" category.

The Supreme Court will hear a case regarding WOTUS for a fourth time in the summer of 2022. In 2007, Chantell and Michael Sackett filled wetlands on their property

adjacent to Priest Lake (Idaho, USA) to build a home. The EPA brought a civil case against them, and in 2008 they sued the EPA on the grounds that their due process rights were violated (Smith and Holden 2013). That case was eventually heard by the Supreme Court, and the Sacketts were granted judicial review. This time, the Sacketts are asking the Supreme Court to overturn the *Rapanos* decision and require that regulated waterways have a continuous surface water connection to downstream protected waterways (Howe 2022). This case may again alter the definition of WOTUS. If the Sacketts win their case and the significant nexus standard is thrown out, ephemeral streams and some intermittent streams may be unprotected by the CWA.

How WOTUS is applied to ephemeral streams

The protection of ephemeral streams under the CWA has been a point of contention for decades. A clarification of WOTUS by USACE in 2000 stated that ephemeral streams that have an ordinary high water mark (OHWM) were considered WOTUS (U.S. Army Corps of Engineers 2000). An OHWM is a line along the streambank characterized by changes in vegetation, soil, or bank morphology that indicates the vertical and lateral extent of the dominant discharge (Lichvar and McColley 2008). The 2000 guidance extended protections to many ephemeral streams in the arid southwestern United States, provided they had physical evidence of flowing water. Soon after, the protection of ephemeral streams came under fire in the *Rapanos* case, where in the plurality opinion, Justice Antonin Scalia derided the USACE for overstepping their regulatory authority by placing CWA protections on

> any parcel of land containing a channel or conduit—whether man-made or natural, broad or narrow, permanent or ephemeral—through which rainwater or drainage may occasionally or intermittently flow. On this view, the federally regulated "waters of the United States" include storm drains, road-side ditches, ripples of sand in the desert that

may contain water once a year, and lands that are covered by floodwaters once every 100 years (*Rapanos et ux., et al. v. United States* 2006).

Scalia's opinion specifically stated that desert ephemeral streams (and even intermittent streams) were not meant to be protected by the CWA, and that extending protections to such waterways was embarrassing (*Rapanos et ux., et al. v. United States* 2006).

The NWPR heavily cited the plurality opinion in the *Rapanos* case in its justification for the exclusion of ephemeral streams from CWA jurisdiction (U.S. Army Corps of Engineers and Environmental Protection Agency 2020). The NWPR placed the protection of ephemeral streams in the hands of states and tribes (U.S. Army Corps of Engineers and Environmental Protection Agency 2020). The promulgation of this Rule had serious implications in watersheds that bordered two or more entities. In the *Pascua Yaqui Tribe* case, the plaintiffs stated that the need to rely on cooperation from state agencies to protect ephemeral streams and other waterbodies on their lands created an undue hardship for the Tribes. Indigenous tribes have historically been subjected to difficulties and environmental racism when attempting to cooperate with federal and state government agencies regarding environmental management (Vickery and Hunter 2016). Thus, the NWPR created a social justice issue.

Prior to the promulgation of the NWPR, the 2015 Clean Water Rule significantly increased protections for ephemeral streams, mostly based on a scientific report on stream connectivity (see Alexander et al. 2015). The Rule required that ephemeral tributaries have a bed, bank, and OHWM to meet the significant nexus test (U.S. Army Corps of Engineers and Environmental Protection Agency 2015). Ordinary high water mark assessments are complicated for ephemeral streams in arid regions. Lichvar and

McColley (2008) provided some guidance in identifying OHWM in the arid West and acknowledged that field identification can be arduous. Hydrogeomorphic processes in ephemeral channels in arid and semi-arid regions are understudied, but differences from perennial channel processes have been identified for ephemeral streams in the southwestern United States. (Schumer, Knust, and Boyle 2014) and other parts of the world (Hooke 2016; Billi et al. 2018). Such processes also differ from one stream to another because of differences in variables such as underlying geology (Hooke 2016) and confinement (Merritt and Wohl 2003). High-water marks can persist for long periods of time in arid regions, further confounding OHWM field assessments. For example, at Yuma Proving Ground in southern Arizona, Merritt and Wohl (2003) identified a high-water mark estimated to be ~26 years old. Evidence of such extraordinary high flows can confound field-based OHWM delineations (Lichvar and McColley 2008), making it difficult to assess connectivity.

An understanding of connectivity in ephemeral watersheds is crucial for assessing whether a given ephemeral stream meets the significant nexus standard. Unfortunately, the knowledge of connectivity in arid ephemeral watersheds is incomplete. During flow events in arid ephemeral streams, flow volume generally decreases along a downstream gradient because some of the flow is lost to groundwater or hyporheic flow (Alexander et al. 2015). However, ephemeral channels in arid regions are quite diverse, with hydrogeomorphic processes varying greatly among them (Shaw and Cooper 2008). Relationships between streamflow and transmission losses are not always straightforward, even among flow events in the same channel (Mujere et al. 2020).

Additionally, when such channels are ungaged, determinations of flow frequencies and/or durations can be challenging.

Water quality protection for ephemeral streams is especially important in the context of climate change. Much of the precipitation in the arid and semi-arid southwestern United States is dependent on the North American Monsoon (NAM). Though most climate projections indicate that the climate of the region will become hotter and drier over time (Seager et al. 2013), predicted changes to the NAM are less clear (Colorado-Ruiz et al. 2018), but there is evidence that the monsoon season may be extended in a warming climate (Torres-Alvarez, Cavazos, and Turrent 2014). If the predictions for warmer and drier conditions and less frequent and more intense precipitation occurs, the connectivity of ephemeral streams to downstream waterways will be altered. Flow initiation in ephemeral streams is dependent on watershed size and the spatial and temporal extent of the precipitation event (with geologic setting and bed material interacting with this relationship) (Kampf et al. 2018). Fewer, lower-magnitude, and/or localized precipitation events reduce hydrologic connectivity throughout watersheds (Ye and Grimm 2013), while larger and/or more intense rain events increase connectivity during episodic floods.

Prolonged drought and increased evapotranspiration, along with increased withdrawal of surface water and groundwater for human use, have already led to reductions in overall streamflow and changes to the flow regime of streams in the southwestern United States (Graf 1988; Seager et al. 2013; Allen et al. 2019). The dependence of ephemeral streams on precipitation means that they are less resilient to climate changes than intermittent or perennial streams (Stromberg et al. 2015), and the

number of ephemeral streams is expected to increase (Seager et al. 2013). These changes will necessitate a review of policies governing the protection of water quality of ephemeral streams.

Knowledge gaps and future research questions

There is an urgent need for additional knowledge on the processes associated with ephemeral streams in arid and semi-arid regions. Knowledge derived from studies on perennial or intermittent streams or streams in humid regions do not necessarily apply (Chapter III), and policy decisions would be improved with more applicable information. Further study is needed on the geomorphic, biologic, and hydrologic processes of ephemeral streams in arid regions. Because many streams in the arid western United States lack stream gages, differentiating ephemeral streams from intermittent streams can be challenging. Though there has been some progress in this area (Mazor et al. 2021), current methods for determining streamflow permanence would be greatly improved with additional empirical data. Additionally, though transmission losses from flow events can be accurately predicted on gaged ephemeral streams (Cataldo et al. 2010), the amount and rate of precipitation that prompts flow events is more difficult to predict (Kampf et al. 2018). This dilemma raises the question: Is it possible to predict flow regime based on field assessments of environmental variables (i.e., geomorphology, presence/absence of biota)? Answering this question requires further study of ephemeral streams in a variety of geologic and mesoscale climatic settings in the southwestern United States.

Another important question for future research is: How should connectivity of ephemeral streams be addressed in a changing climate? Several authors have already identified a decrease in overall streamflows caused by groundwater and surface water

withdrawals and a warmer, drier climate (Graf 1988; Seager et al. 2013; Jaeger, Olden, and Pelland 2014; Allen et al. 2019). As populations in the southwestern United States increase and the area's climate continues to warm, the number of ephemeral reaches will rise and no-flow periods for intermittent rivers will be of longer duration (Jaeger, Olden, and Pelland 2014). Most stream reaches in the southwestern United States are intermittent or ephemeral already (Goodrich et al. 2018). The conversion of perennial and intermittent streams to ephemeral streams would necessitate a new approach to identifying a significant nexus. Furthermore, climate changes to the precipitation regime will affect the flow regime of ephemeral streams, possibly resulting in higher-magnitude floods capable of transporting materials long distances (Camarasa-Belmonte 2021).

The proposed definition of WOTUS states that a significant nexus can be established by one stream or by a combination of nearby streams, but it does not provide specific guidance on determining an appropriate number of streams to meet the standard, nor does it provide guidance on how to determine whether a given stream or set of streams alters downstream water quality in a significant way (Department of Defense and Environmental Protection Agency 2021). Both questions should be addressed empirically for the proposed rule to be effective.

Conclusions

The summarized definitions of WOTUS throughout its history as it applies to ephemeral streams in the southwestern United States provides important context for understanding the application of its regulatory power over time. Ephemeral streams fulfill important ecological functions (Levick et al. 2008) and potentially impact water quality downstream. The latest proposed definition (and interpretation) of WOTUS allows for

protections for ephemeral streams to some extent, but is somewhat vague. I proposed several questions which could alleviate some of the issues with interpretations of the proposed rule.

III. CONTROLS ON THE SPATIAL DISTRIBUTION OF RIPARIAN VEGETATION ALONG EPHEMERAL STREAMS IN THE ARID SOUTHWESTERN UNITED STATES

Introduction

Ephemeral streams, which flow only following precipitation events, comprise much of the total stream length in the arid southwestern United States (Goodrich et al. 2018). Their processes, form dynamics, and complex ecosystems provide considerable diversity to desert landscapes. Riparian vegetation along arid ephemeral streams generally consists of xeric (drought-adapted) trees and shrubs which commonly occur in upland environments and can withstand water-limited environments (Beauchamp and Shafroth 2011). Such species include mesquite (*Prosopis* spp.), catclaw acacia (*Acacia greggii*), desert willow (*Chilopsis linearis*), and paloverde (*Parkinsonia* spp.) (Lowe 1961; Johnson and Lowe 1985; Johnson, Bennett, and Haight 1989).

Riparian vegetation in arid environments generally increases in size and density along a transverse gradient from the uplands to well-defined channels and from xeric to hydric sites (Stromberg, Wilkins, and Tress 1993; Sponseller and Fisher 2006). Prior studies have found longitudinal patterns in riparian community composition related to water availability; wetter reaches support mesic species and xeric species occur in drier reaches (Hupp and Osterkamp 1996; Lite, Bagstad, and Stromberg 2005) and on abandoned floodplains (Reynolds, Shafroth, and House 2014). Less is known about the longitudinal patterns of the riparian community along ephemeral streams. Some patterns are obvious when viewed from above via satellite or aerial imagery (Malanson 1993;

Manning, Julian, and Doyle 2020), yet the environmental controls on such patterns are rarely examined in the literature.

The relationships between surface run-off, surface-groundwater connections, flow regime, and xeric vegetation is less clear than it is for mesic and hydric communities. However, there is some evidence that fluvial processes play a role in xeroriparian community composition, even on ephemeral streams (Douglas et al. 2018), and these processes may also influence community spatial patterns to some degree. Flow regime is considered a "master variable" in fluvial processes, including those governing vegetation dynamics (Poff et al. 1997). For ephemeral streams in arid regions, high flow pulses and floods recharge soil water and alluvial groundwater (Graf 1988; Stromberg, Tiller, and Richter 1996), providing water access for riparian trees. Timing of higher flows is important for some mesic and hydric riparian species, as germination of many native species coincides with spring floods (Stromberg, Patten, and Richter 1991). The relationship between flood timing and germination, however, is altered in a warming climate (Perry et al. 2020). Long-duration floods provide moisture for seedings during the dormant season (McBride and Strahan 1984) and recharge alluvial groundwater (Auble, Friedman, and Scott 1994). Low-frequency, high-magnitude floods are extremely important in the southwestern United States because they govern geomorphic processes including erosion and deposition (Graf 1988), which provide space for seedlings to germinate. High-magnitude floods also provide short-term alluvial groundwater recharge, which supports plant growth.

Though riparian vegetation responds to flow regime, the data used to analyze it are based on past conditions, so other hydrogeomorphic variables must be considered

(Poff 2017). Factors relating to fluvial processes and geomorphology are interdependent with riparian vegetation (Osterkamp, Hupp, and Stoffel 2012). Channel width and gradient influence vegetation by controlling sedimentation and disturbance patterns (Baker 1989; Shaw, Cooper, and Sutfin 2018) and seed dispersal (Cunnings, Johnson, and Martin 2016). Large floods scour away some vegetation, widening the channel. Sediment deposits serve as recruitment sites, which then results in channel narrowing as plants trap sediments and debris (Dean and Topping 2019). Though these processes are well-documented in large, alluvial rivers, they occur even in small, bedrock streams (Jerin 2019) and are likely important in ephemeral streams. Like flow regime, sediment regimes are altered by climate changes and anthropogenic alteration of waterways (Wohl et al. 2015), which contributes to changes in vegetation over time.

Position along the stream network influences vegetation by controlling erosion and deposition patterns, streamflow, and disturbance regimes (Shaw and Cooper 2008). Thus, riparian community composition at the headwaters typically differs from downstream community composition (Shaw, Cooper, and Sutfin 2018). Drainage basin area and drainage density increase downstream, controlling sediment transport, flood regime, and bankfull discharge (Baker 1989), which can in turn influence riparian vegetation. In addition to variability in hydrogeomorphic processes along the stream network, there are also variations in land-use that affect vegetation patterns. Anthropogenic influences such as impoundments and diversions are less likely to occur at the headwaters than at locations further downstream (Bruno et al. 2014). These types of alterations affect riparian vegetation patterns in part by altering water storage (Hamdan and Schmeekle 2016). In the southwestern United States, agriculture, urbanization, and

mining occur in floodplains, typically at low elevations, which results in the removal of riparian vegetation.

Topographic setting influences riparian vegetation by controlling light, flow regimes, and certain climate-related variables. For example, valley shape controls insolation and shade (Bendix 1994; Bruno et al. 2014) and to some extent flood regime and water table depth (Baker 1989; Cadol and Wine 2017). Elevation, one of the strongest topographic variables influencing riparian vegetation (Baker 1989) controls temperature and precipitation, though the effect of orographic precipitation is minimal in very dry areas of the southwestern United States (Graf 1988). Elevation also affects depth to alluvial groundwater (Stromberg, Tiller, and Richter 1996) and substrate size and alluvial thickness (Shaw, Cooper, and Sutfin 2018), controlling water availability to plants.

Climate variables, including temperature and precipitation, strongly influence riparian vegetation. Average annual precipitation is <350mm in much of the southwestern United States, much of which results from the North American Monsoon (Sheppard et al. 2002) and midlatitude cyclones originating over the Pacific Ocean. Convective thunderstorms, such as those occurring under monsoon conditions, produce highly localized rainfall that affects surface flow more so in small basins than in large basins (Graf 1988). In ephemeral and/or losing streams, precipitation controls soil moisture in the alluvium (Cadol and Wine 2017) because such streams do not typically have connections to high water tables.

Underlying geology provides some control on the riparian community by restricting or enabling access to groundwater. Deep water tables can be accessed by

phreatophytes (Graf 1988), but seedlings and young trees are more likely to survive in areas with shallow alluvial aquifers (McBride and Strahan 1984; Shaw and Cooper 2008). Because of these differences in water storage, riparian community composition varies between bedrock streams and alluvial streams (Shaw, Cooper, and Sutfin 2018) and spatial distribution may vary as well.

Variation of the riparian community along a lateral gradient from the channel across the floodplain has been well-documented in the literature. Riparian plants closer to the channel tend to grow larger than plants further from the channel (Balding and Cunningham 1974), likely because they have better access to soil moisture (Smith et al. 1995). Variation along a longitudinal gradient in arid and semi-arid regions is less well-studied, though some work has been done in this area. Hupp and Osterkamp (1996) observed vegetation patterns related to stream flow variations at Alamo Wash in southern Arizona. The authors noted an increase in mesic species with distance downstream, except on losing reaches, where xeric species increased. Van Coller, Rogers, and Heritage (2000) examined influences on riparian tree species at three environmental gradients (longitudinal, lateral, and vertical) in South Africa and found that the vertical gradient (i.e., elevation above the channel) was more important in determining species dominance. Both of these works focused on species composition rather than spatial patterns and distribution of riparian trees.

An understanding of longitudinal vegetation patterns with respect to environmental conditions, especially along ephemeral streams in arid and semi-arid regions, would guide policy and land management decisions related to water quality. In the United States, ephemeral streams are usually excluded from water policy legislation.

However, their collective contributions to downstream water quality could be considerable given that flow events in ephemeral streams mobilize sediments and transport pollutants downstream (Larned et al. 2010). In much of the southwestern United States, small and/or remote streams are rarely gaged, so the actual discharge, including the magnitude, frequency, and duration of flow events is unknown. Current methods for determining temporal streamflow patterns are somewhat subjective and are based on incomplete knowledge of flow regimes in arid ephemeral streams. Given the relationships among riparian vegetation, hydrology, and environmental factors, a better understanding of the controls on the structure (canopy size/density) and spatial distribution of riparian vegetation would benefit jurisdictional determinations.

This chapter aims to identify longitudinal patterns and environmental controls on riparian vegetation along ephemeral streams in the southwestern United States using Normalized Difference Vegetation Index (NDVI). NDVI is a ratio of red and infrared light reflectance (infrared – red/infrared + red) that is commonly used to analyze vegetation changes over time (Fu and Burgher 2015; Hamdan and Schmeeckle 2016). Because vegetation cover in the southwestern United States is highest near stream channels (Cadol and Wine 2017; Stromberg et al. 2017), NDVI can be a useful tool in analyzing vegetation patterns along a stream network (Manning, Julian, and Doyle 2020; Assal et al. 2021).

My research questions are: (1) How does the size of riparian vegetation clusters vary along ephemeral streams from channel head to tributary junction?, and (2) how is riparian vegetation distribution related to hydrologic and geomorphic conditions along ephemeral streams in the arid southwestern United States? To answer these questions, I

used PlanetScope imagery to identify riparian vegetation along ephemeral tributaries to six USGS-gaged streams in New Mexico and Arizona.

Methods

Study sites

Study streams were located on public lands in Arizona and New Mexico in a variety of physiographic and climatic settings (Figure 1; Table 2). All study streams are ephemeral tributaries to United States Geological Survey (USGS)-gaged streams. For this study, sites are named for the stream gage because many of the ephemeral tributaries are unnamed.

The Verde River site is comprised of two forks of an ephemeral tributary to the Verde River that flows through McDowell Mountain Regional Park north of Scottsdale, Arizona. McDowell Mountain Regional Park is a recreational area used for hiking, mountain biking, and other activities. The watershed consists mostly of parallel streams on an alluvial fan.

The Rio Ojo Caliente site consists of two branches of an ephemeral tributary to Rio Tusas. The tributary rises from the Mesa de la Jarita in Carson National Forest in north-central New Mexico and flows into Rio Tusas, which joins Rio Ojo Caliente at La Medera, New Mexico, and then flows into the Rio Grande. Grazing and recreation, including off-road vehicle touring, are common in the area. Tilled agriculture is conducted in the floodplains of both Rio Ojo Caliente and the Rio Grande. The site consists of a dendritic network of meandering streams at high elevations and a parallel network of braided streams at low elevations near Rio Tusas's confluence with Rio Ojo Caliente.

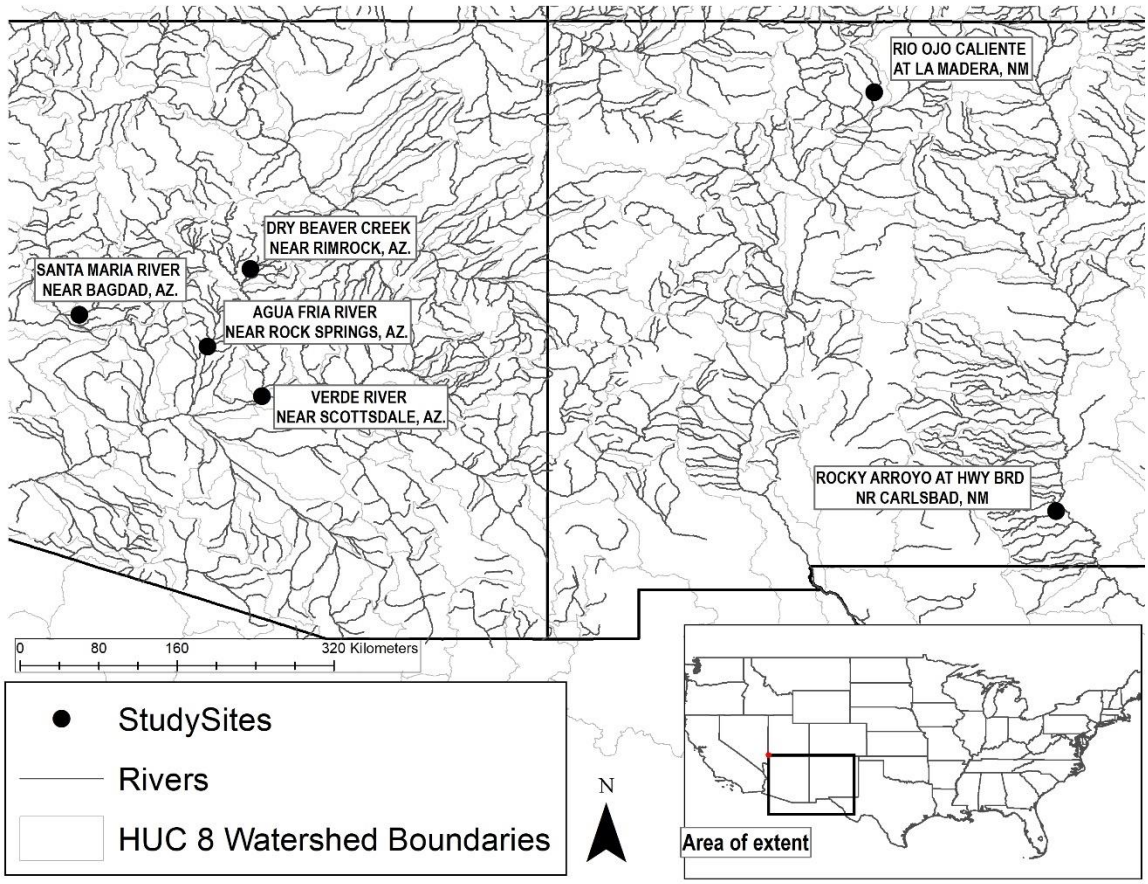

Figure 1. Study area.

The Agua Fria River site is along Slate Creek, an ephemeral tributary to the Agua Fria River that flows through land managed by the Bureau of Land Management (BLM). The area is used for recreation including off-road vehicles. The Agua Fria is an intermittent river dammed at Lake Pleasant north of Phoenix, and it ultimately flows into the Gila River at Goodyear, Arizona. Upstream of Lake Pleasant, the watershed consists of dendritic, meandering and braided channels. Between Lake Pleasant and its confluence with the Gila, the Agua Fria River flows through western Phoenix, where urbanization, agriculture, and mining have significantly impacted the channel and the floodplain.

Table 2. Study sites. The topography and management columns are based on the area where the ephemeral study tributary is located. Elevation and average annual precipitation are for the nearest USGS stream gage.

Site Name	Topography	Elevation (m)	Ecoregion	Management	Average annual precipitation (mm)	Land use
Verde River	flat	497	Sonoran Desert	Maricopa County McDowell Mountain Regional Park	273	Recreation
Rio Ojo Caliente	foothills	1991	Chihuahuan Desert	USFS	318	Recreation, grazing
Agua Fria River	foothills	549	Sonoran Desert	BLM	347	Recreation, urban, mining
Dry Beaver Creek	montane	1126	Colorado Plateau	USFS	371	Recreation
Rocky Arroyo	foothills	991	Chihuahuan Desert	BLM	343	Grazing, oil and gas production
Santa Maria River	foothills/flat	415	Sonoran/ Mojave Desert	BLM	148	Recreation, grazing

The tributaries to Dry Beaver Creek rise from House Mountain in Coconino National Forest in northern Arizona near Sedona. Dry Beaver Creek flows into Wet Beaver Creek near Montezuma Castle National Monument, and Wet Beaver Creek joins the Verde River at Camp Verde, Arizona. The headwaters are parallel, slightly meandering streams. Dry Beaver Creek, Wet Beaver Creek, and the Verde River are meandering streams with broad floodplains.

Rocky Arroyo is located near Carlsbad, New Mexico and is an ephemeral tributary to the Pecos River that flows through BLM land used for grazing and oil and gas production. The stream is in a dendritic network consisting of incised headwater channels. At lower elevations, the stream becomes a wide, braided channel.

The Santa Maria River site is an ephemeral tributary to Date Creek in western Arizona. The tributary rises in the Black Mountains and flows into Date Creek, an ephemeral tributary to the Santa Maria River. Date Creek and the tributary are situated on an alluvial fan and the area is managed by the BLM. The Santa Maria River is an intermittent stream that joins the Big Sandy River to form the Bill Williams River at Alamo Lake. The Bill Williams then flows into the Colorado River at Lake Havasu. Streams in the basin are parallel, braided, and mostly straight.

Data collection and processing

Cloud-free, high-resolution images (resolution = 3-5m) from late June to early July 2019 were obtained from Planet (www.planet.com) and processed to create polygons of individual trees and/or clusters of trees in the riparian zone of each study stream. Adjacent images were mosaicked and all images were classified in Erdas Imagine v.

16.6.0 (Hexagon AB, Norcross, GA) using Normalized Difference Vegetation Index (NDVI) unsupervised classification.

The response variable for this analysis is the size (area in m^2) of tree clusters in the riparian area of the study streams. The classified imagery was used to create polygons of trees and/or clusters of trees (Figure 2) in ArcMap v. 10.7.1 (ESRI, Redlands, CA). I identified the channel head of each study stream by identifying the point where flow becomes concentrated (Julian 2018). Using the original (unclassified) imagery as a guide, I created outlines of the floodplain of each of the study streams, excluding any tilled agriculture, which would alter the results. Along several reaches of the main stems of the Agua Fria River, Rio Ojo Caliente, and one segment of the Bill Williams River, most of the floodplain is tilled and had to be excluded from the analysis.

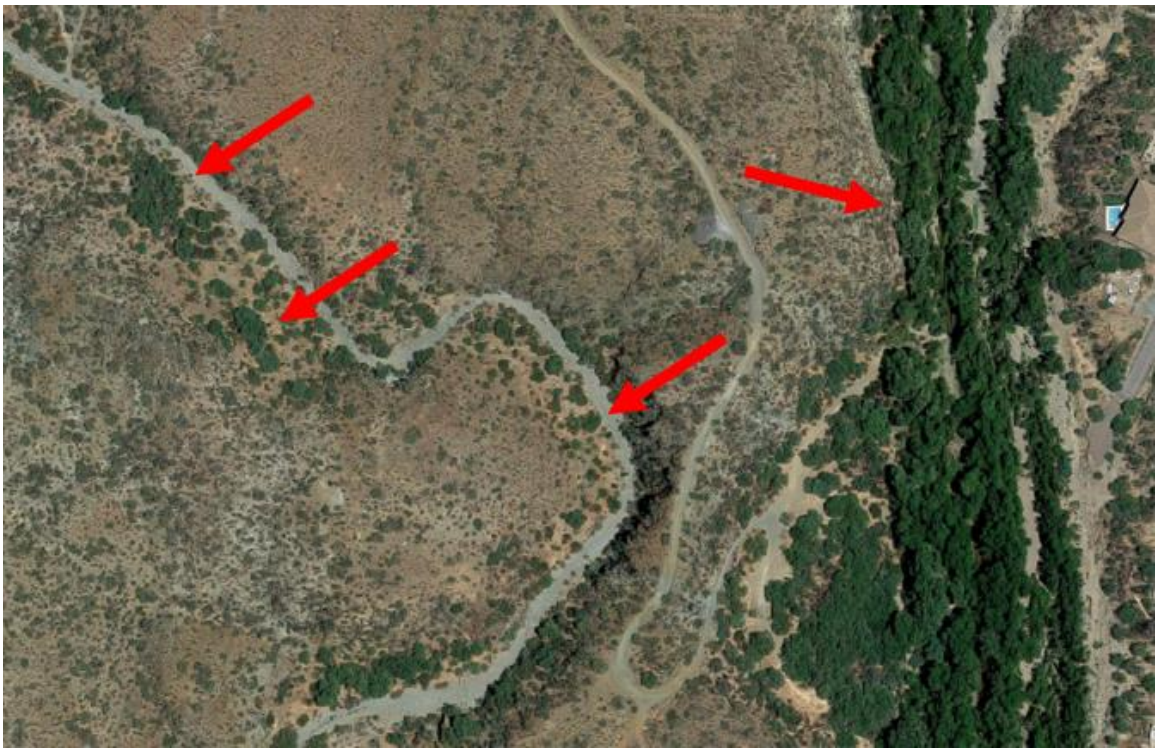

Figure 2. Examples of tree clusters. This example is from Slate Creek (left), the ephemeral study tributary to the Agua Fria River, just upstream of the tributary junction. The arrows point to tree clusters which vary in size along Slate Creek, but are generally much smaller than the clusters along the Agua Fria (right).

All floodplain outlines include the channel head and any upstream rill, gully, or wash. I clipped and masked the classified images to the study areas, then reclassified the images based on a threshold value of NDVI (0.25), with values ≥0.25 representing vegetation and values <0.25 representing non-vegetated surface features. This threshold was identified as the best representation of vegetation in the area during a previous case study (Manning, Julian, and Doyle 2020). The reclassified rasters were then converted to polygons using the Raster to Polygons conversion tool. Polygons classified as vegetation were selected and exported to a shapefile.

Climate data (mean annual maximum temperature (T_{max}), mean annual minimum temperature (T_{min}), daily mean temperature (T_{mean}), mean daily precipitation (P), maximum vapor pressure deficit (VPD_{max}), and minimum vapor pressure deficit (VPD_{min})) were based on 30-year averages derived from the PRISM dataset (PRISM Climate Group, Oregon State University, https:\\prism.oregonstate.edu, 2004)(Table 3). Using the Raster Calculator tool in ArcMap, I calculated the mean vapor pressure deficit (VPD_{mean}) by averaging the VPD_{min} and the VPD_{max}.

Because the vegetation polygons were quite small relative to the spatial resolution of the climate rasters (resolution = 1 degree), I converted the climate rasters to points and the points to Theissen polygons in order to join the data to the vegetation layer. I created a point layer composed of the centroids of each vegetation polygon, then used the Spatial Join tool in ArcMap to join the climate data. I then joined the vegetation centroid layer with the climate data to the vegetation polygon layer.

Digital Elevation Models (DEMs; 1/3 arc-second resolution) were obtained from USGS.gov (Table 3). I used the DEMs to create rasters for slope and aspect using the

Slope and Aspect surface analysis tools. I converted all three terrain rasters to points and Theissen polygons and joined the data using the same methods used for the climate data above.

Underlying bedrock type was derived from state geology datasets obtained from USGS.gov. Because the machine-learning algorithm I used to classify the data does not handle categorical variables well, I divided the bedrock type into four presence/absence variables: igneous, metamorphic, sedimentary, and other. Other rock types consisted of various unconsolidated material (i.e., sand, gravel, alluvium).

To measure the reach-scale variables, I divided each floodplain outline into reaches based on planform and/or channel width, such that each reach was relatively uniform in width and channel pattern. Reaches that were part of a reservoir were excluded from the analysis. I created a set of three random points within each reach and measured channel and floodplain width at each point using the original imagery. The values from all points within a single reach were averaged to define the channel and floodplain width. For braided reaches, I measured the width across all channels. I measured the length of the reach in the channel and in the floodplain, and the length of the reach along the channel. I calculated the sinuosity of the stream using the channel length measurements. Using the DEMs, I collected the elevation of the reach at upstream and downstream points in the channel and in the floodplain to calculate the channel and floodplain slope. To quantify confinement, I used a ratio of the channel width to the floodplain width.

I qualitatively identified the channel pattern, land use, and alteration using the original unclassified imagery and Google Earth, which is at a higher resolution. I

categorized the channel pattern into straight, meandering, and braided categories following Leopold and Wolman (1957) with additional categories to distinguish heavily altered channels, erosional features, and minimally braided channels (Table 2).

Table 3. Channel patterns. Follows Leopold and Wolman (1957) with exceptions for the minimally braided, rill, and open pit mine categories.

Pattern	Description
Straight	No evidence of migrating within the floodplain. Although few channels are actually straight for very long distances (Leopold and Wolman 1970), these types of streams do not meet the criteria for meandering, likely due to low discharge and high confinement, even though they may be slightly sinuous.
Minimally braided	Multi-threaded but narrow channel with few individual threads
Braided	Multi-threaded channel with wide width relative to reach length
Meandering	Single-threaded channel with have obvious point bars, cutbanks, in-channel islands, and/or a sinuous channel pattern within a less sinuous floodplain.
Rill	Rills, washes, or gullies upstream of the channel head. These are erosional features with no evidence of deposition.
Open pit mine	Channels completely altered by mining processes, where planform is not distinguishable using satellite imagery.

Some parts of the channel were altered in ways that were visible on the satellite imagery (i.e., obvious erosion, channelization). These alterations impact riparian plant growth, which affects tree cluster size. I categorized alteration according to the amount of obvious alteration present at each reach (heavy, medium, light, none). This variable captures the intensity of land use changes that is not captured by land use categories (Table 4).

Because the ephemeral study streams are ungaged, streamflow data from the nearest USGS stream gage was used as a proxy. Flow regime variables were calculated in Indicators of Hydrologic Alteration v7.1 (IHA) (The Nature Conservancy) and HEC-SSP v2.2 (U.S. Army Corps of Engineers) using data from the nearest USGS stream gage to each ephemeral study stream (Appendix A). Drainage areas above the stream gages ranged from 367 to 17,133 km^2, and multiple tributaries other than the study streams

contribute flow. IHA uses daily streamflow data to calculate 67 flow parameters. The variables I selected (Table 5) comprise the five components of the natural flow regime (magnitude, frequency, timing, rate, and duration) (Poff et al. 1997). These variables were calculated for a single time period (the period of record) for each stream.

Table 4. Channel alteration categories.

Category	Description	Application
Heavy	Channel pattern and/or flow has been completely and obviously changed from its natural state; planform is not recognizable as a natural stream	Channels that are completely mined (no channel visible), channelized (changing the pattern from braided or meandering to straight)
Medium	Channel pattern and/or flow has been altered, but extent of the alterations is not necessarily visible from satellite imagery	Channels that are obviously constricted in width; floodplains with significant agriculture, mining, or urbanization; flow that is increased or initiated by agriculture runoff, mining, or water treatment; or flow that has been altered by the presence of a dam
Light	Erosion from land use is visible, but natural hydrogeomorphic processes are not effected in a way that is obvious from satellite imagery	Channels with evidence of erosion resulting from recreation or grazing, which causes channel widening
None	No alterations are visible from satellite imagery. This does not mean that alterations have not occurred along such channels.	All other channels

HEC-SSP software was used to calculate flow duration curve analysis and flood frequency analysis. Flow duration curve analysis uses daily mean flow to determine the percent of time certain flow magnitudes are exceeded. I used the Rank/N+1 method with a linear y-axis and a normal probability x-axis. The flood frequency analysis uses peak annual data to calculate flood return intervals. I used 17B methods (Interagency Advisory Committee on Water Data 1982), Weibull plotting position, and the station skew to calculate these statistics.

Table 5. Variables used in random forest analysis.

Scale	Variable	Units	Source
Basin	Mean annual flow	m^3/s	USGS; calculated with IHA
	90-day minimum flow	m^3/s	USGS; calculated with IHA
	Low pulse count	count	USGS; calculated with IHA
	Low pulse duration	days	USGS; calculated with IHA
	High pulse count	count	USGS; calculated with IHA
	High pulse duration	days	USGS; calculated with IHA
	Extreme low flow timing	Julian day	USGS; calculated with IHA
	High flow timing	Julian day	USGS; calculated with IHA
	Time exceedance 10%	m^3/s	USGS; calculated with HEC-SSP
	Time exceedance 25%	m^3/s	USGS; calculated with HEC-SSP
	Time exceedance 50%	m^3/s	USGS; calculated with HEC-SSP
	Time exceedance 75%	m^3/s	USGS; calculated with HEC-SSP
	Time exceedance 90%	m^3/s	USGS; calculated with HEC-SSP
	1-year return interval	m^3/s	USGS; calculated with HEC-SSP
	5-yeat return interval	m^3/s	USGS; calculated with HEC-SSP
	10-year return interval	m^3/s	USGS; calculated with HEC-SSP
	50-year return interval	m^3/s	USGS; calculated with HEC-SSP
	100-year return interval	m^3/s	USGS; calculated with HEC-SSP
Reach	Channel to valley ratio	m	Channel width \ valley width
	Channel gradient	ratio	Calculated in ArcMap
	Channel width	m	Calculated in ArcMap
	Channel pattern: meandering		Calculated in ArcMap
	Channel pattern: braided		Calculated in ArcMap
	Channel pattern: straight		Calculated in ArcMap
	Channel pattern: rill		Calculated in ArcMap
	Channel pattern: open pit mine		Calculated in ArcMap
	Sinuosity	ratio	Calculated in ArcMap
	Valley gradient	ratio	Calculated in ArcMap
	Valley width	m	Calculated in ArcMap
	Alteration		Planet Imagery
	Land use: recreation		Planet imagery
	Land use: grazing		Planet imagery
	Land use: mining		Planet imagery
	Land use: urban		Planet imagery
	Land use: water treatment plant		Planet imagery
	Land use: highway		Planet imagery
	Land use: oil and gas		Planet imagery
Tree cluster	Distance from channel head	m	Calculated in ArcMap
	Mean daily precipitation	mm	PRISM dataset
	Mean daily maximum temperature	°C	PRISM dataset
	Mean daily minimum temperature	°C	PRISM dataset
	Mean daily mean temperature	°C	PRISM dataset
	Mean daily vapor pressure deficit	hPa	PRISM dataset
	Aspect	°	Derived from DEM
	Elevation	m	Derived from DEM
	Bedrock- igneous		USGS
	Bedrock- metamorphic		USGS
	Bedrock- sedimentary		USGS
	Bedrock- other		USGS

Analysis

Tree cluster size is the dependent variable for this analysis. Tree clusters along all streams ranged in size from 5.8 m^2 to 4,955,558.1 m^2 (mean = 1,857.6 m^2)(Figure 3). Along the ephemeral portions of the streams, clusters ranged from 5.8 m^2 to 3,197,083.4 m^2 (mean = 1054.8 m^2)(Figure 3). I classified the tree clusters into four categories by area using half of the standard deviation as the break points (class 1 = small, class 2 = moderately small, class 3 = moderately large, class 4 = large). The majority (26,141 of 26,194) of the tree clusters belonged to class 1.

The ephemeral portions of the study streams were dominated by class 1 tree clusters, especially along the tributaries to the Santa Maria River, Rocky Arroyo, and the Verde River. Large clusters (class 4) were present at the headwaters along the tributaries to Rio Ojo Caliente and Dry Beaver Creek. Along the tributary to the Agua Fria River, class 1 and 2 clusters were present.

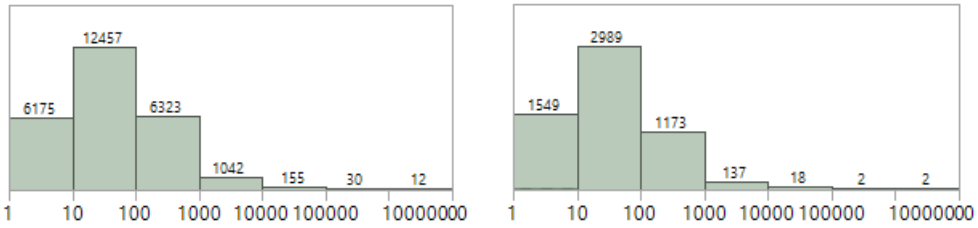

Figure 3. Histograms of tree cluster area (m^2) along the study streams from ephemeral headwaters to the nearest traditionally navigable waterway (left) and the ephemeral portions of the study streams (right).

To determine the environmental controls on tree cluster size, I ran two analyses using random forest and conditional inference trees. Both methods are decision tree classifiers. Random forest is a supervised machine learning classifier that uses random subsamples of the dataset to train the algorithm to determine the importance of each variable in the classification. Like random forest, conditional inference trees is a decision

tree model, however rather than explaining the importance of each variable, it explains the value of each variable that cause branching. This combination of methods more fully explains the spatial distribution of tree clusters by determining the importance of each variable and the values of each variable that govern cluster size.

I ran each model on the entire dataset (all ephemeral streams to the nearest TNW), and again on the ephemeral portions of all streams. To determine the best fitting random forest model, I used an iterative process using RandomForestClassifier from SciKit Learn in Python, altering the parameters (maximum depth, minimum leaf samples, and number of estimators (trees)) slightly with each iteration. I then did a final run using the parameters from the best fitting model. The validity of the model is measured with an out-of-bag (OOB) score, which is calculated using the data that were not used for training. The resulting trees from the best model was used to run a conditional inference tree model to determine the values of each variable that divided each tree. The conditional inference trees were created using the Tree package from SciKit Learn in Python.

Results

The random forest analysis of the entire dataset resulted in an OOB score of 0.998 (Table 6). For all ephemeral streams the OOB score was 0.999. These OOB scores are higher than expected, and may be evidence of overfitting caused by the dominance of class 1 tree clusters. For the full dataset, the best-fitting model used a maximum depth of 4, 15 minimum leaf samples, and 90 n estimators. For the ephemeral dataset, the best model used a maximum depth of 4, 20 minimum leaf samples, and 120 estimators (trees). Variables that were measured at the scale of tree clusters were most important across all

analyses. Elevation, distance from the channel head, and climate variables were most important (importance values >0.1). Climate variables are highly correlated with elevation (Appendix B). Random forest performs well with highly intercorrelated data sets, however correlations among variables must be considered when interpreting results. Reach-scale variables describing channel and/or valley morphology were moderately important, with the exception of valley width, which was the second-most important variable in both analyses. Geology, land-use and basin scale streamflow variables were ranged in importance from minimal to not at all (importance values <0.01).

Table 6. Results of random forest classifier. Cells with no value or rank were not found to be important. ** indicates very important variables. * indicates variables of moderate importance.

Variable	**Full dataset**		**All ephemeral streams**	
Number of tree clusters	26,194		6,040	
Training OOB score	0.998		0.999	
OOB score	0.998		0.999	
	Rank	**Importance**	**Rank**	**Importance**
Distance from channel head	3	0.131**	7	0.063*
Precipitation	9	0.040	13	0.005
Maximum temperature	4	0.084*	--	--
Minimum temperature	10	0.038	3	0.110**
Mean temperature	5	0.066*	--	--
Mean vapor pressure deficit	7	0.044	1	0.218**
Aspect	8	0.042	4	0.096*
Elevation	1	0.145**	--	--
Bedrock – igneous	18	0.007	--	--
Bedrock – metamorphic	--	--	9	0.044
Bedrock – sedimentary	24	0.004	--	--
Bedrock – other	27	0.003	5	0.089*
Channel-valley ratio	13	0.034	8	0.048
Channel gradient	11	0.038	--	--
Channel width	14	0.022	12	0.019
Sinuosity	6	0.054*	11	0.024
Valley gradient	12	0.035	10	0.043
Valley width	2	0.139**	2	0.165**
Meandering	32	0.002	--	--
Braided	--	--	--	--
Straight	31	0.002	--	--
Lake	--	--	--	--
Rill	--	--	--	--
Minimally braided	25	0.004	--	--
Alteration	36	0.001	--	--
Recreation	23	0.004	--	--

	Full dataset		All ephemeral streams	
	Rank	**Importance**	**Rank**	**Importance**
Grazing	34	0.001	--	--
Tilled agriculture	39	0.001	--	--
Mining	--	--	--	--
Urban	28	0.003	--	--
Water treatment plant	33	0.002	--	--
Oil and gas	--	--	--	--
Mean annual flow	19	0.006	--	--
90-day minimum	--	--	--	--
Low pulse count	35	0.001	--	--
Low pulse duration	26	0.004	--	--
High pulse count	20	0.005	--	--
High pulse duration	22	0.004	--	--
Extreme low flow timing	21	0.004	--	--
High flow timing	--	--	--	--
Time exceedance 10%	30	0.002	--	--
Time exceedance 25%	--	--	--	--
Time exceedance 50%	37	0.001	--	--
Time exceedance 75%	38	0.001	--	--
Time exceedance 90%	--	--	--	--
1-year return interval	17	0.007	6	0.076
5-year return interval	29	0.003	--	--
10-year return interval	16	0.008	--	--
50-year return interval	15	0.008	--	--
100-year return interval	40	0.001	--	--

The conditional inference trees (Appendix C) analysis for all tree clusters (the full dataset including ephemeral, intermittent, and perennial reaches) indicated that valley widths > 2959 m produced the largest tree clusters (> 1 sd; class 4). Elevation was an important variable at sites where the elevation was very high (> 2323 m) or low (< 142 m), while values in the middle of the range for temperature divided trees. Tree clusters at low elevations (< 142 m) with very high mean daily temperatures (≤ 23.4°C) fell into either the largest or the smallest (≤ -0.5 sd; class 1) class, as did tree clusters found at high elevations (> 2323 m) and moderate temperatures. Aspect was a dividing variable only for clusters on north-facing slopes. At high temperature sites, tree clusters occurring on a slope with a northern aspect were in class 1. At other high temperature sites, elevation and distance from the channel head determined whether a cluster fell into class

1 or class 2 (> -0.5 sd and < 0 sd). Tree clusters at north-facing sites with moderate elevations (> 142 m) were either class 1 or 4, while other tree clusters at moderate elevations were in classes 1-3.

Ephemeral tributaries were dominated by class 1 clusters. Conditional inference trees indicated that on north-facing slopes, clusters occurring in narrow valleys (valley width ≤ 68 m) were class 3, while those occurring in wider valleys were class 1. At other sites, very high elevation and low maximum temperature (≤ 15.7°C) determined that clusters were class one. Very high elevation, higher maximum temperature sites were dominated by class 4 clusters. Lower elevation sites were dominated by class 1.

Discussion

In the present study, a qualitative assessment of tree cluster size along the six study streams shows differences in cluster size related to the position along the stream network, but the influence of the environmental factors varied with the scale of analysis. Tree clusters along ephemeral streams responded to environmental variables differently than tree clusters along the entire stream network. In general, the smallest clusters occurred along ephemeral tributaries, whereas larger clusters were found downstream closer to TNWs, where valleys were generally wider. This pattern may be the result of more frequent flow events and larger, deeper alluvium deposits near and along the main-stem rivers. Flow frequency variables were not determined to be a primary control on tree cluster size in the present study, however flow regime variables for this study were based on the nearest stream gage. The lack of accurate streamflow data for the entire river length likely impacts the results.

Flow frequency influences vegetation because it controls access to soil moisture, and differences in flow frequency likely affect the size and density of riparian plants. Previous studies have demonstrated that xeric plants with smaller leaves and shrubby growth form occur at sites that flow infrequently, while mesic plants with larger leaves occupy wetter sites (Hupp and Osterkamp 1996; Lite, Bagstad, and Stromberg 2005). Species composition likely affects cluster size to some extent by controlling leaf size, growth habit, and canopy size, however, the same variables that govern species composition probably also play a role in determining tree cluster size. Flow permanence and the dry period duration influences infiltration and transpiration (Schilling et al. 2021), which further control water access and may affect the composition, size, and/or density of riparian vegetation. In the present study, cluster size is determined by canopy size and density of vegetation, but inferences about species composition can not be made using the methods employed here.

Hydrogeomorphic variables including geologic setting greatly influence riparian vegetation by controlling storage of alluvium and water. Along ephemeral streams, geomorphic factors controlling depth of alluvium have a strong influence on vegetation (Shaw, Cooper, and Sutfin 2018). In the present study, reach-scale hydrogeomorphic variables were moderately important, especially for ephemeral tributaries, though valley width was the second-most important variable overall for both analyses. Valley widths were fairly uniform along the ephemeral tributaries compared to those along main-stem reaches, but there was some variance, especially on the downstream ends. Channel width differed between main-stem rivers and tributaries as well. The larger main-stem rivers generally have much wider channels than their ephemeral tributaries; wide, braided

channels serve as floodplains in dryland regions (Graf 1988) and provide additional habitat for riparian trees. Except for Date Creek in the Santa Maria watershed, the channel widths of ephemeral streams were quite narrow and mostly uniform.

High elevation sites along ephemeral tributaries, especially those in the Rio Ojo Caliente (approximate elevation = 2333 m) and Dry Beaver Creek (approximate elevation = 1352 m) basins included exceptions to the pattern of smaller clusters (Figure 4). Both sites supported large, continuous clusters of vegetation near the channel heads, likely because of elevation. In general, low elevation sites in the southwestern U. S. are warmer and drier than higher elevation sites, and above-ground biomass is positively correlated with elevation (Pelletier et al. 2013). Thus, high elevation sites likely support larger riparian tree clusters because they support more vegetation in general. Lower-elevation ephemeral tributaries to the Santa Maria and Verde Rivers and Rocky Arroyo had small clusters along their entire stream length (Figures 4 and 5). The Santa Maria and Verde sites receive the least annual rainfall of all the sites (Table 2), which likely limits the density of vegetation. The vegetation at the Rocky Arroyo site is limited by shallow soils and steep slopes.

Composition also varies from high to low elevation sites; differences in canopy size among species affects cluster size. At the highest elevations on the Colorado Plateau, such as the headwaters of the tributaries to Dry Beaver Creek and Rio Ojo Caliente, the vegetation likely consists of conifers including firs, pines, and junipers. Lower elevations at all sites are likely dominated by cacti and desert shrubs with small leaves, such as creosote (*Larrea tridentata*), which is more common on the uplands, and mesquite (*Prosopis* spp.), which grows along ephemeral streams (Bailey 1995). Some of the

ephemeral streams may be occupied by mesquites or similar species, but other sites are likely dominated by creosote. Differences in root structure among species controls tree cluster size by limiting the spatial distribution of individuals. Creosotes, which have wide-spreading roots, grow farther apart than phreatophytes (deeply-rooted) such as mesquites and are less likely to form tight clusters.

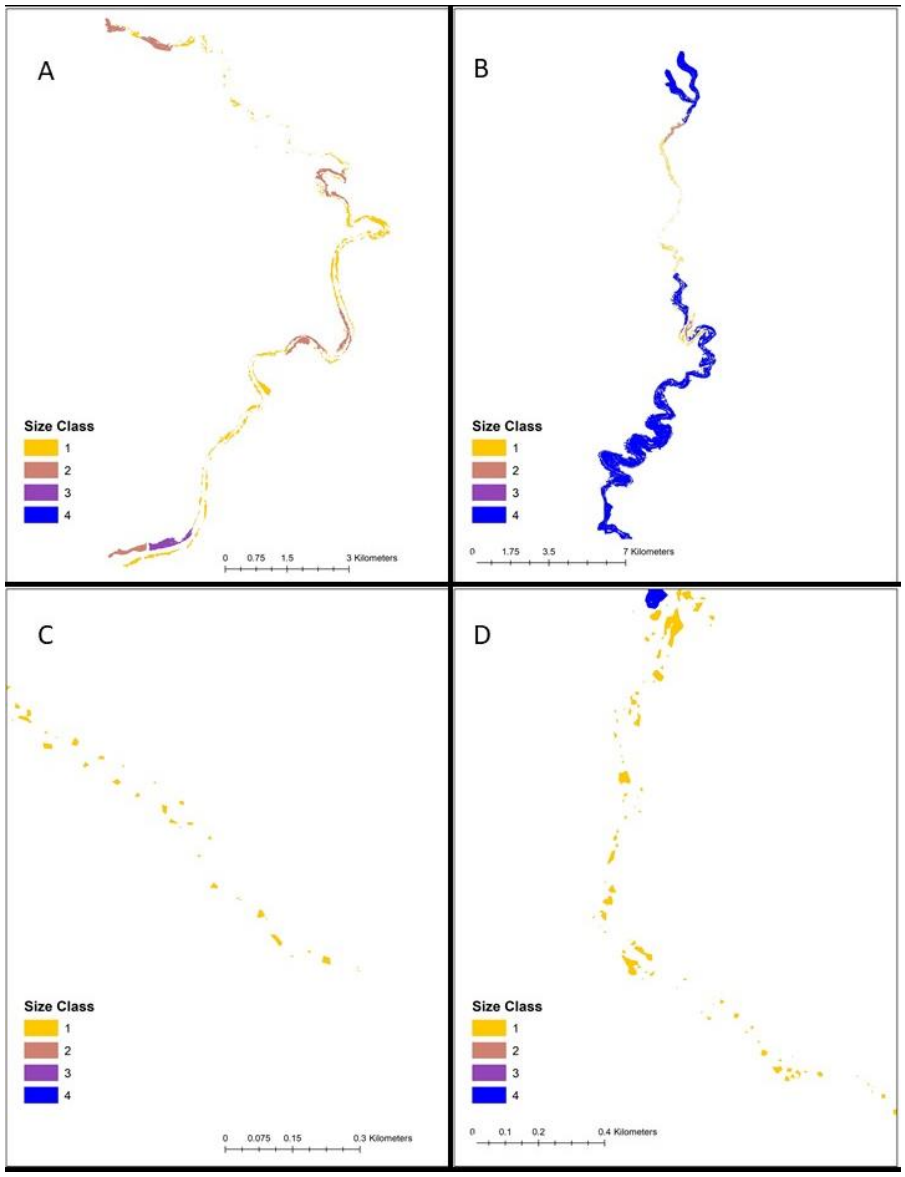

Figure 4. Differences in tree cluster size among Arizona ephemeral study sites. The tributary to the Agua Fria River (A) had tree clusters from classes 1-3. Much larger classes were found along Dry Beaver Creek and its tributaries (B). The tributaries to the Verde River (C) and the Santa Maria River (D) were dominated by class 1 clusters. Tree clusters along these two tributaries are so small they are only visible on a very large-scale map. Maps C and D are only displaying part of the tributaries.

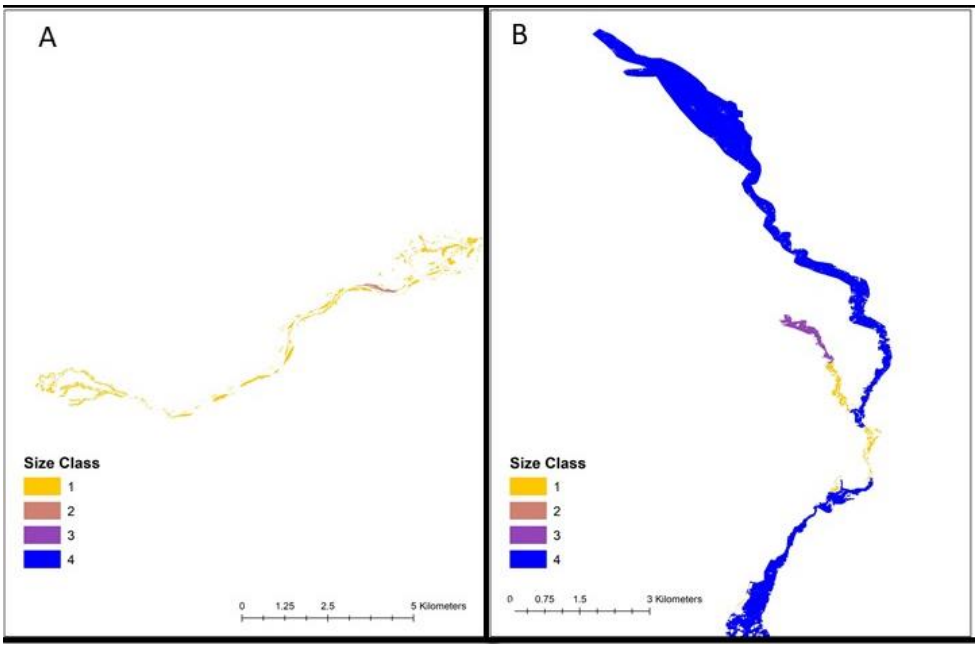

Figure 5. Tree clusters at New Mexico study sites. Rocky Arroyo (A) was dominated by class 1 clusters, while Rio Ojo Caliente (B) had much larger clusters.

Distance from the channel head was the third-most important variable for the entire dataset. Along the full length of the rivers, clusters far from the channel head were generally larger than clusters on the ephemeral tributaries. On the ephemeral streams, the largest clusters were found at the tributary junction with the main stem. Channel heads for all study streams occur at moderate to high elevation in mountains, foothills, or alluvial fans, and the confluences with the TNWs downstream occur at lower elevations. This gradient influences the size of tree clusters indirectly by influencing climate (temperature and precipitation) and sediment transport and deposition.

Variations in longitudinal vegetation patterns have been determined to be primarily controlled by climate and elevation in two studies in South Africa (van Coller, Rogers, and Heritage 2000; Sieban, Mucina, and Boucher 2009). For the six streams examined here, both channels and valleys were generally wider along the main stems

than on the tributaries, which likely also results in larger tree clusters downstream.

Elevation was less important when only the ephemeral tributaries were considered. Along ephemeral streams the largest clusters were closer to the channel head at high elevation, resulting in an inverse relationship between downstream distance and tree cluster size. This is likely due to the elevational differences in composition outlined above.

Climate variables, specifically VPD_{mean} and T_{min}, were more important along ephemeral streams than they were for the entire dataset. Plants are highly sensitive to vapor pressure deficit (Grossiord et al. 2020), and vapor pressure deficit is dependent on temperature. Both VPD_{mean} and T_{min} vary with elevation, which may also partially explain their importance. Precipitation varies along elevational and longitudinal gradients in the southwestern United States, with less precipitation at western sites and at lower elevations. Higher precipitation occurs at eastern sites and at higher elevation.

Limitations

The scale and/or resolution of the available data likely had a strong effect on the importance of each variable. The most important variables in determining size class were those measured at the tree cluster scale, including elevation, aspect, distance from the channel head and climate variables. Data at higher resolutions would likely yield more accurate results, but the use of publicly-available data for this study provided limited options. Finer-scale elevation data would allow for more accurate measurement of valley and channel slopes, confinement, and elevation, and would also provide data on the vertical gradient (elevation of each cluster relative to channel elevation). For example, in-channel geomorphic features result from interactions among sediment transport,

hydrology, and riparian vegetation even in ephemeral streams (Tooth and Nanson 2000; Jerin 2019). Higher-resolution data would allow for further exploration of these microtopographic features and their relationships to riparian plants.

Finer-scale climate and flow regime data would greatly improve these results, especially given the importance of climate (temperature and humidity) data for the ephemeral study streams. Streamflow variables were minimally important for all streams, however because the data is so coarse and is derived from nearby gages rather than the ephemeral study streams, conclusions about their actual importance cannot be drawn from this study.

Finally, the method of classification used for this study influenced the results to some extent. The size of the tree clusters varied widely, but the vast majority were very small. A classification scheme that better captures variations in both very small and very large clusters would likely provide better data on the spatial distribution of the riparian community.

Conclusions

The findings presented here demonstrate the importance of scale and geographic context in determining the environmental controls on ephemeral streams. Riparian vegetation responses to environmental variables differs among streams depending on stream permanence. Climate variables such as humidity and temperature were important for determining tree cluster size for ephemeral streams, and hydrogeomorphic variables determined tree cluster size along intermittent and perennial streams. Observations on riparian vegetation of large rivers cannot be applied to ephemeral streams without further

evaluation. These differences should be considered when making jurisdictional determinations and land management decisions.

IV. THE RIPARIAN VEGETATION OF ARID EPHEMERAL STREAMS AND ITS RESPONSE TO HYDROLOGIC AND TOPOGRAPHIC VARIABLES

Introduction

The Clean Water Act regulates water quality in the United States, but it only applies to certain waterways, recognized as Waters of the United States (WOTUS). These waters consist of traditionally navigable waterways (TNW), territorial seas, interstate waterways, and some of their tributaries and adjacent wetlands. The definition of WOTUS is in the process of being revised, but under current legislation, tributaries to TNWs may be protected under the CWA as long as they meet one of two standards: the "relatively permanent standard," meaning the waterway is continuously connected to a downstream WOTUS through surface water; or the "significant nexus standard," vaguely defined by the alteration of the water quality of a downstream waterway by the stream in question (Environmental Protection Agency 2021)[1]. The 'significant nexus standard' may be met by a single waterway or a combination of similar waterways in proximity.

Applying these standards in the same way across the United States is problematic because hydrogeomorphic processes differ across regions, which leads to unequal protection throughout the country. Indicators of permanence or connectivity that are based on scientific conclusions derived from humid regions or perennial streams do not necessarily apply to streams in arid regions or to non-perennial streams (Manning, Julian, and Doyle 2020). In arid and semi-arid regions of the US, few streams meet the 'relatively permanent standard' because the majority of streams in such areas are either

[1] The public comment period for this proposed definition ended on February 7, 2022, however the final definition will not be made official until after this dissertation is completed. Additionally, the Supreme Court is to hear a case on the Clean Water Act in summer 2022 which may further alter the definition. See Chapter II for further explanation.

intermittent (flowing seasonally) or ephemeral (flowing only in direct response to precipitation in the basin) (Goodrich et al. 2018), and thus do not have continuous surface water connections to downstream waterways. Since some intermittent and all ephemeral streams do not meet the 'relatively permanent standard,' the 'significant nexus standard' must be applied for a given intermittent or ephemeral stream to qualify for Clean Water Act protections.

Determining whether a significant nexus exists for a given ephemeral stream is challenging, especially in arid and semi-arid regions. Such a connection, if present at all, would be episodic and occur only during infrequent flow pulses (Nadeau and Rains 2007). The use of hydrogeomorphic indicators to establish a significant nexus is complicated in arid and semi-arid regions for several reasons. In the southwestern United States, climate regime, geologic setting, and land cover result in increased stream intermittence compared to more humid regions (Costigan et al. 2016). Topography and climate alter runoff patterns and processes (Doyle and Bernhardt 2011). This alteration results in runoff dominated by Hortonian overland flow from steep slopes, rather than by the saturation flow that occurs in humid environments (Bracken and Croke 2007). Sediment transport in arid regions is associated with higher-magnitude, lower-frequency floods than those responsible for sediment transport in humid climates (Graf 1988). This combination of factors complicates hydrogeomorphic assessments in the southwestern United States and highlights the need for a suite of biological indicators.

The dependence of riparian vegetation on hydrologic processes is well-documented for many species occurring along perennial and/or intermittent rivers. In the southwestern United States, species such as Fremont cottonwood (*Populus fremontii*) and

tamarisk (*Tamarix* sp.) require regular flooding throughout their life cycles (Lytle and Merritt 2004). However, these species are not likely to occur in great numbers along ephemeral streams. Many of the trees that occur along ephemeral streams in arid and semi-arid regions are drought-tolerant species that are also found in upland environments, often called xeroriparian communities (Lowe 1961).

Xeroriparian communities are composed primarily of terrestrial, upland plants, but their relationship to streamflow is understudied. Though the species composition is often similar to that of the surrounding terrestrial landscape, there are differences in volume and species richness (Stromberg et al. 2017). Xeroriparian species are not dependent on streamflow or floods for reproduction, however they may depend on streamflow for other processes. For example, some xeroriparian species have stomatal and photosynthetic responses to streamflow that do not necessarily occur with rainfall (de Soyza, Killingbeck, and Whitford 2004).

Because water is a primary limiting factor for vegetation, there may be some relationship between longitudinal surface hydrologic connectivity and vegetation growth patterns in regions where groundwater access and precipitation are limited. Covino (2017) describes longitudinal hydrologic connectivity as a spectrum with channelized rivers and human-made canals on one end (high connectivity) and dammed rivers on the other (low connectivity). Ephemeral streams in the southwestern United States likely fall toward the low connectivity end of the spectrum because much of their flow is lost to the alluvium, especially during peak flows (Lange 2005) and extreme rain events (Schreiner-McGraw, Ajami, and Vivoni 2019) when transmission losses increase.

Determining flow characteristics for ungaged streams in desert regions is complicated. Unlike streams in humid regions, desert streams lose flow along a downstream gradient as water infiltrates into the alluvium. This is true for small streams and large rivers alike. Even on a well-monitored river such as the Colorado River, flow can be difficult to model and predict because streamflow from the river and its tributaries often goes subsurface. For the Minute 319 environmental flow on the Colorado in 2014, analysis of tree rings, topography, soils, and other variables to determine the appropriate discharge for the environmental flow (Ramírez-Hernández et al. 2015). For small streams including ephemeral streams, such data may not be available.

Because the study streams for this research are ungaged, information on the flow regime is unavailable, however, unit stream power can be approximated using data derived from digital elevation models (DEM). Stream power, the energy exerted on a stream channel by flowing water, has been found to control riparian vegetation to an extent. Stream power can influence the composition and spatial patterns of the riparian forest (Bendix 1994, 1999; Shaw, Cooper, and Sutfin 2018). Stream power is calculated using the equation

$$\omega = \rho g Q S$$

where ρ is the density of water, g is acceleration due to gravity, Q is the discharge of the stream, and S is the slope of the channel. Because discharge is a component of stream power, it is difficult to approximate for arid streams because much of the flow of arid streams is lost to evapotranspiration and subsurface flow.

In this study, I use drainage basin area as a proxy for discharge and standardize my calculations by channel width to approximate specific unit stream power using the

equation (referred to from here on as ‘stream power proxy’:

$$\omega = (\frac{AS}{b})$$

where *A* is the area of the basin that drains into the channel, or drainage area. Drainage area increases in a linear fashion with distance downstream, but channel slope and width are only moderately correlated with distance downstream (Appendix B). Because the width of the channels varies greatly along the study streams, standardizing the equation by channel width allows for comparison among reaches with varying widths.

This proxy is used to determine the extent to which hydrologic connectivity controls xeroriparian vegetation. The stream power proxy can be used to approximate connectivity in this case because it is an estimate of the potential for a stream to transport sediments, including pollutants. Reaches with higher stream power proxy values could potentially transport greater amounts of sediments and pollutants than reaches with lower stream power proxy values. Because connectivity along arid ephemeral streams may be low, I also evaluate the effects of both elevation and distance downstream on vegetation. I aim to answer two questions: (1) Is riparian vegetation dependent on hydrology along arid, ephemeral streams?, and (2) Can remote sensing be used to evaluate connectivity along arid, ephemeral streams?

Methods

Study sites

The analysis was conducted on six ephemeral streams in New Mexico and Arizona (Figure 6). All study streams are located completely or partially on public land managed by the US Forest Service (Rio Ojo Caliente, Dry Beaver Creek), the Bureau of Land Management (Agua Fria River, Rocky Arroyo, Santa Maria River) and Maricopa

County Parks and Recreation (Verde River). All study streams eventually drain into a TNW. Elevations ranges from 379 m to 2541 m above sea level and most channel heads occur at high elevations (714m at Rocky Arroyo, NM - 2405m at Rio Ojo Caliente, NM). Channel slopes vary from 0.001 to 0.2 and channel widths range from 1 to 948 m.

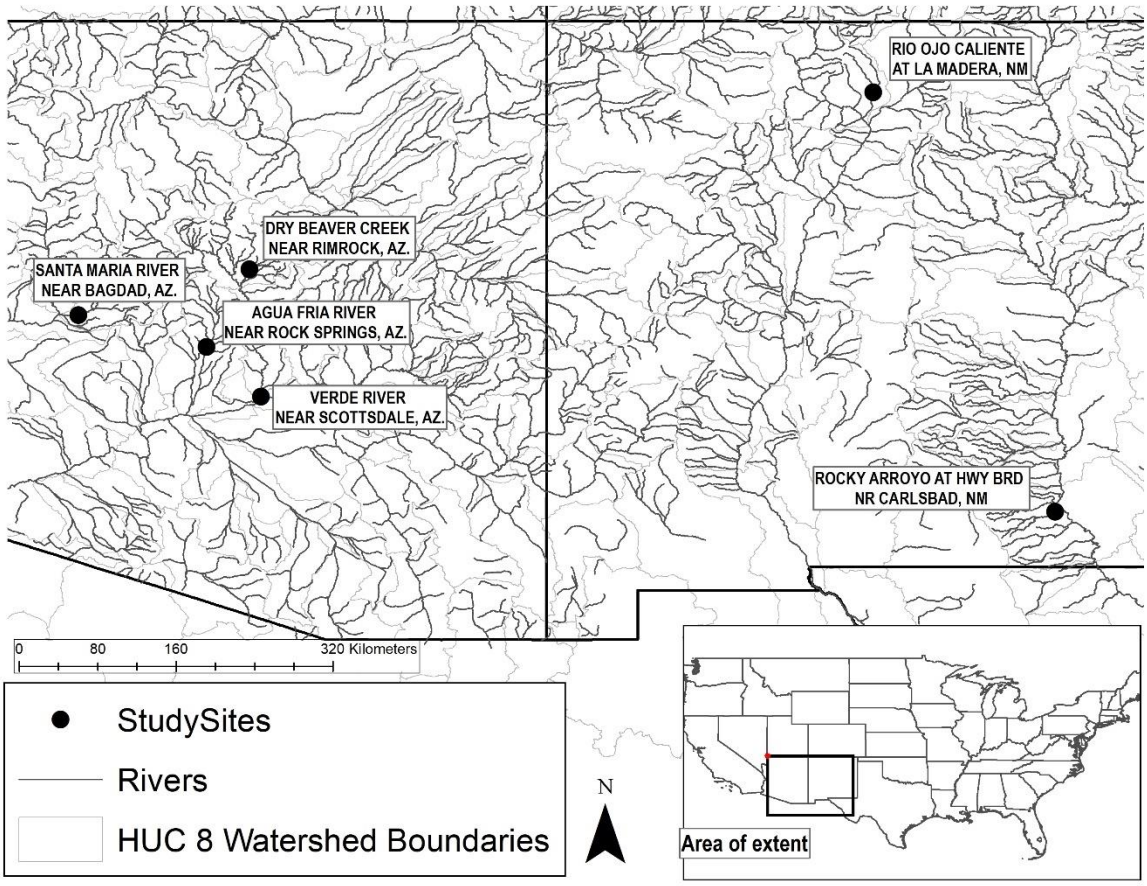

Figure 6. Study area. Study sites are named for the nearest downstream USGS stream gage because many of them are unnamed.

Climate in the region ranges from semi-arid to arid, with average annual precipitation values ranging from 148-371 mm at the nearest United States Geological Survey (USGS) stream gage. The study streams are single-threaded and straight to slightly sinuous at the headwaters. In the middle reaches, the streams widen and become

minimally to considerably braided at lower elevations close to their outlets. Land use along the streams consists of grazing (i.e., cattle, goats), recreation (i.e., hiking, off-highway vehicle use), and natural resource extraction (i.e., timber harvesting, oil and gas production).

Data collection and statistical analysis

Satellite imagery (resolution = 3-5m^2) was obtained from PlanetScope (www.planet.com). All imagery was classified using Normalized Difference Vegetation Index (NDVI) in Erdas Imagine v. 16.6.0 (Hexagon AB, Norcross, GA). Each ephemeral stream was divided into reaches based on channel planform and geomorphology so that each reach was consistent in width and channel pattern. The mean NDVI (range = 0.08-0.50) for each reach was calculated in ArcMap 10.7.1 (ESRI, Redlands, CA). All other variables were calculated or measured in ArcMap (Table 7).

Table 7. Calculation of independent variables and their components.

Variable	Measurement and/or calculation
Elevation	Upstream elevation (m) of each reach derived from USGS DEM (1/3 arc-second resolution)
Distance downstream	Distance in meters from the channel head
Channel head	The point where rills or gullies converge (Julian 2018). True-color satellite imagery (PlanetScope) was used as a guide.
Stream power proxy	$\frac{drainage\ area\ (km^2) * channel\ slope\ (m)}{channel\ width\ (m)}$
Drainage area	Calculated in ArcMap using Fill and Flow Accumulation tools
Channel slope	Change in elevation for each reach (upstream – downstream) divided by the reach length.
Channel width	Measured using PlanetScope imagery as a guide

Bivariate linear regression was used in JMP 14.0.0 (SAS institute, Cary, NC) to establish correlations between mean NDVI and three independent variables: stream power proxy, elevation, and distance downstream. Because stream power proxy is a variable created from multiple components (i.e., channel slope, channel width, drainage area) I also conducted linear regression to determine the effect of each component on

mean NDVI. There were a few extreme outliers present in the dataset, including reaches with zero stream power proxy values and reaches with abnormally high stream power proxy values. These outliers were caused by errors in the DEMs and were removed. These relationships were tested with the entire dataset (all ephemeral streams) and with each individual ephemeral stream.

Results

For the entire dataset, a weak, positive relationship was observed between mean reach NDVI and the stream power proxy (correlation = 0.346, r^2 = 0.120, p-value <0.001)(Figure 7, Table 8). However, the high NDVI values at the Rio Ojo Caliente site altered this result. With Rio Ojo Caliente excluded, the relationship becomes negative (correlation = -0.146, r^2 = 0.021, p-value = 0.03). Strong, significant (p-value <0.001) relationships were identified between mean NDVI and elevation (positive correlation = 0.906, r^2 = 0.821)(Figure 8) and mean NDVI and distance downstream (negative correlation = -0.538, r^2 = 0.289)(Figure 8).

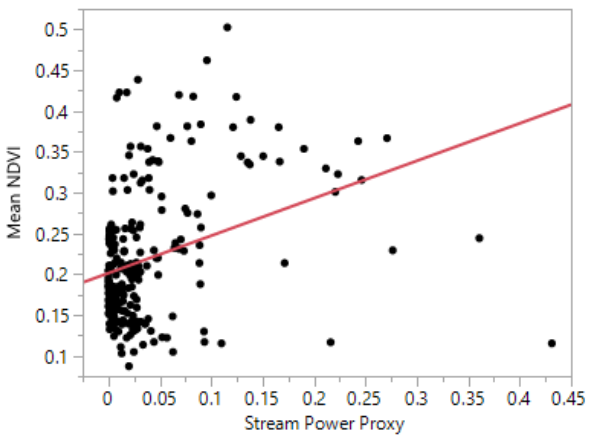

Figure 7. Relationship between stream power proxy and mean NDVI. R^2 = 0.120, p-value <0.001.

Relationships between mean NDVI for each individual stream and stream power proxy were all weak and negative (Table 8; Appendix D). Relationships between mean NDVI and elevation (positive) along individual streams were all significant and ranged from moderate to strong (Table 8; Figure 7). Relationships between mean NDVI and distance downstream were all negative and varied in strength (Table 8; Figure 7). All were significant except along the tributary to the Agua Fria River.

Table 8. Correlations with stream power proxy, elevation, and distance downstream. **indicates relationship is significant at 0.99 confidence. *indicates relationship is significant at 95% confidence.

	n	**Stream Power Proxy**		**Elevation**		**Distance Downstream**	
		Correlation	R^2	*Correlation*	R^2	*Correlation*	R^2
All streams	269	0.346**	0.120	0.906**	0.821	-0.538**	0.289
Agua Fria River	26	-0.044	0.002	0.675*	0.456	-0.254	0.064
Verde River	63	-0.087	0.008	0.672**	0.452	-0.447*	0.200
Dry Beaver Creek	34	-0.210	0.044	0.882**	0.779	-0.832**	0.692
Santa Maria River	67	-0.394*	0.155	0.927**	0.860	-0.945**	0.893
Rio Ojo Caliente	48	-0.257	0.066	0.833**	0.694	-0.401**	0.161
Rocky Arroyo	30	-0.219	0.048	0.549*	0.301	-0.723**	0.523

Analysis of the stream power proxy components (slope, width, and drainage area) found that all relationships were significant but weak for the entire dataset (Table 9). For individual streams, the relationships between mean reach NDVI and slope varied in strength and significance, with the only strong relationship occurring at the Dry Beaver Creek site. Relationships between mean reach NDVI and channel width also varied in significance and all relationships were weak except at the Rocky Arroyo site. The relationships between mean reach NDVI and drainage area were all negative with all but one (the Agua Fria River tributary) being significant. Strong negative relationships were

found between mean reach NDVI and drainage area at the Dry Beaver Creek, Santa Maria River, and Rocky Arroyo sites.

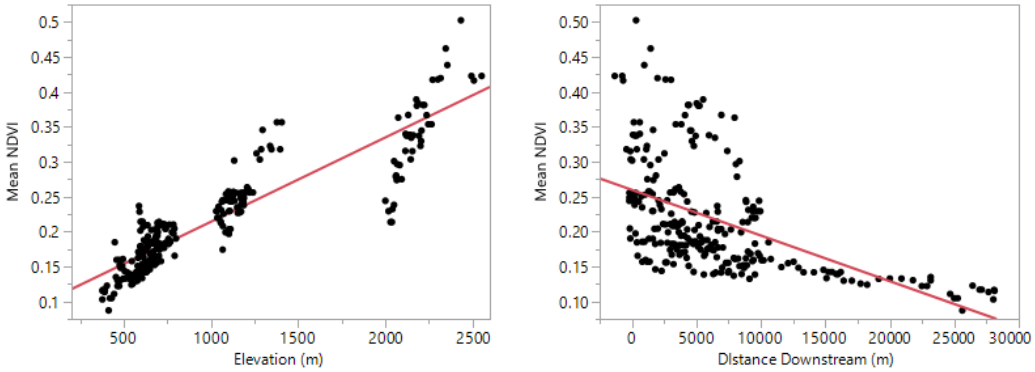

Figure 8. Relationships between mean NDVI and (a) elevation and (b) distance downstream. Both relationships are significant at 99% confidence.

Table 9. Correlations with components of stream power proxy. **indicates relationship is significant at 0.99 confidence. *indicates relationship is significant at 95% confidence.

	n	**Channel slope**		**Channel width**		**Drainage area**	
		Correlation	R^2	*Correlation*	R^2	*Correlation*	R^2
All streams	269	0.349**	0.121	-0.312**	0.098	-0.341**	0.116
Agua Fria River	26	0.010	<0.001	0.383	0.147	-0.185	0.034
Verde River	65	-0.008	<0.001	-0.226	0.051	-0.449*	0.201
Dry Beaver Creek	34	0.733**	0.537	-0.329	0.108	-0.743**	0.552
Santa Maria River	67	-0.049	0.002	-0.375**	0.141	-0.840**	0.705
Rio Ojo Caliente	48	0.273	0.075	-0.054	0.003	-0.495**	0.245
Rocky Arroyo	30	0.342	0.117	-0.704**	0.495	-0.753**	0.567

Discussion

Though the relationship between mean NDVI and stream power proxy was significant for the full dataset, it was weak and somewhat contradicted by the strong relationships with elevation and distance downstream. The relationships of mean NDVI to both distance downstream and elevation demonstrate that mean NDVI is highest at the

channel heads and decreases downstream along the stream's course. Qualitative observations of vegetation patterns support this result, as do the random forest results from the previous chapter.

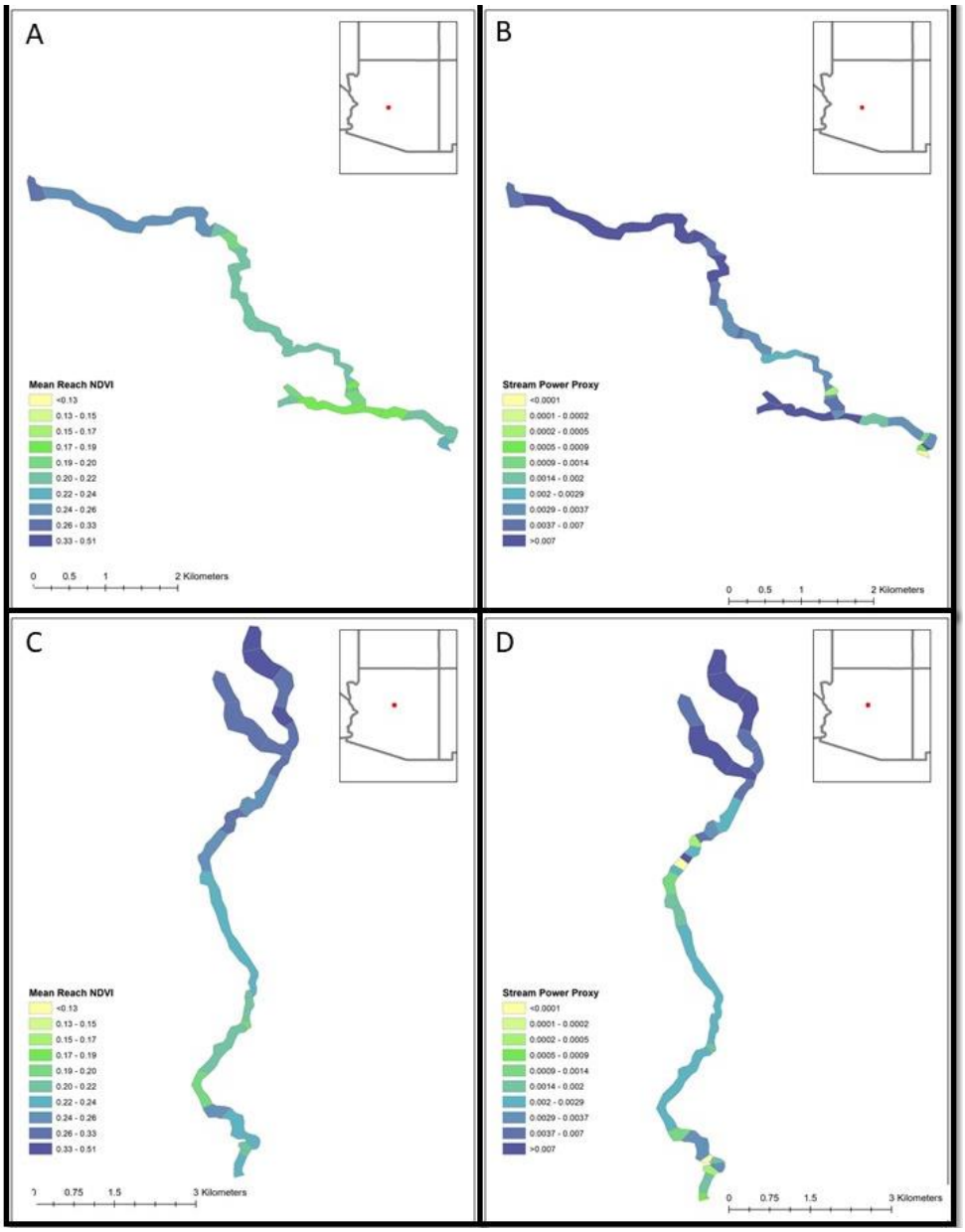

Figure 9. Mean NDVI and stream power proxy along the tributaries to the Agua Fria River and Dry Beaver Creek. Mean NDVI generally decreased from headwaters to mouth along the tributaries to the Agua Fria River (A) and Dry Beaver Creek (C). Stream power proxy was high at the headwaters of the tributaries to the Agua Fria (B) and Dry Beaver Creek (D), but did not decrease downstream in a linear fashion.

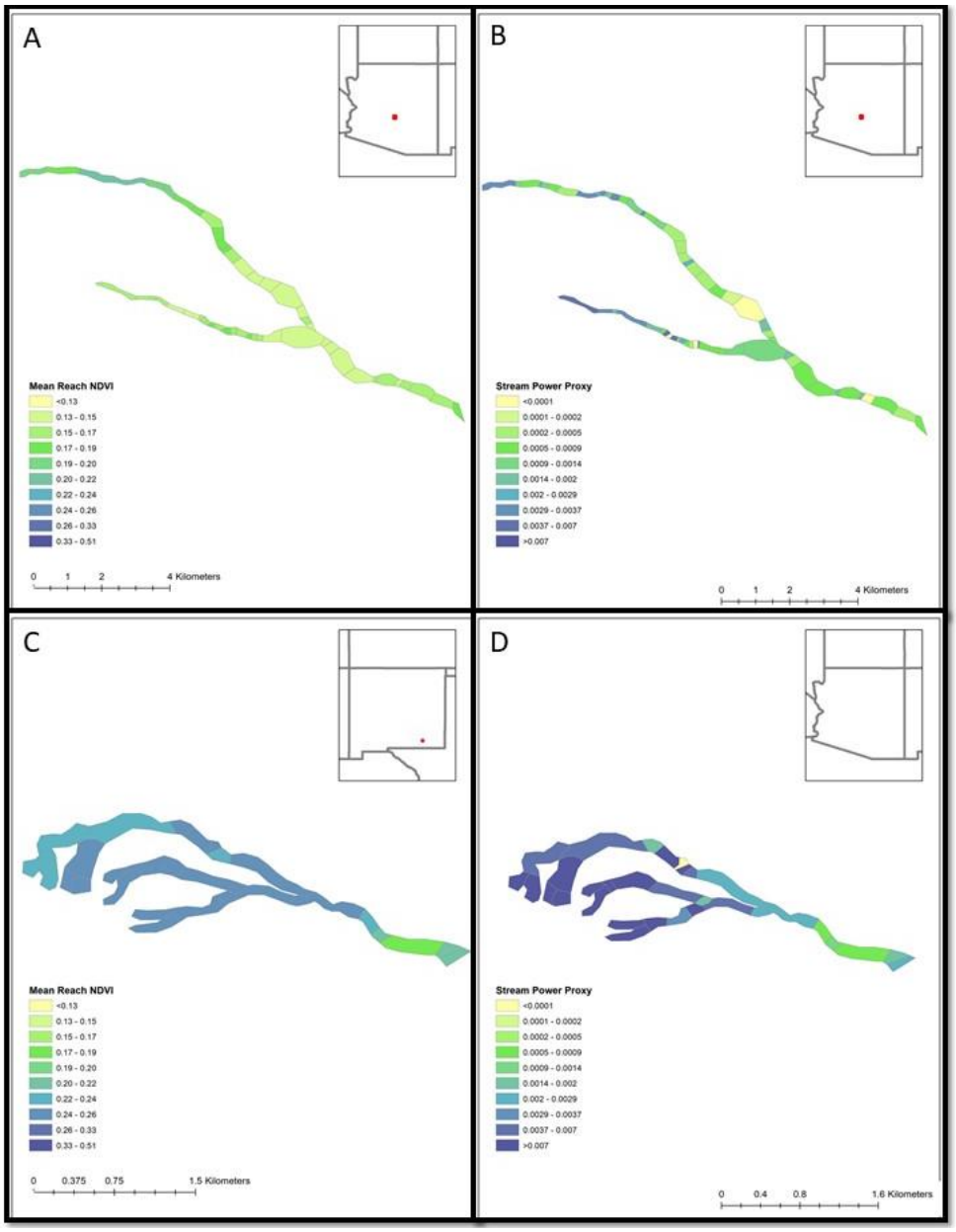

Figure 10. Mean NDVI and stream power proxy along the tributaries to the Verde River and Rocky Arroyo. Mean reach NDVI (A) was low along the tributary to the Verde River, but stream power proxy values (B) were much more erratic. Along the tributary to Rocky Arroyo, mean reach NDVI (C) and stream power proxy (D) were highest at the headwaters and generally decreased downstream.

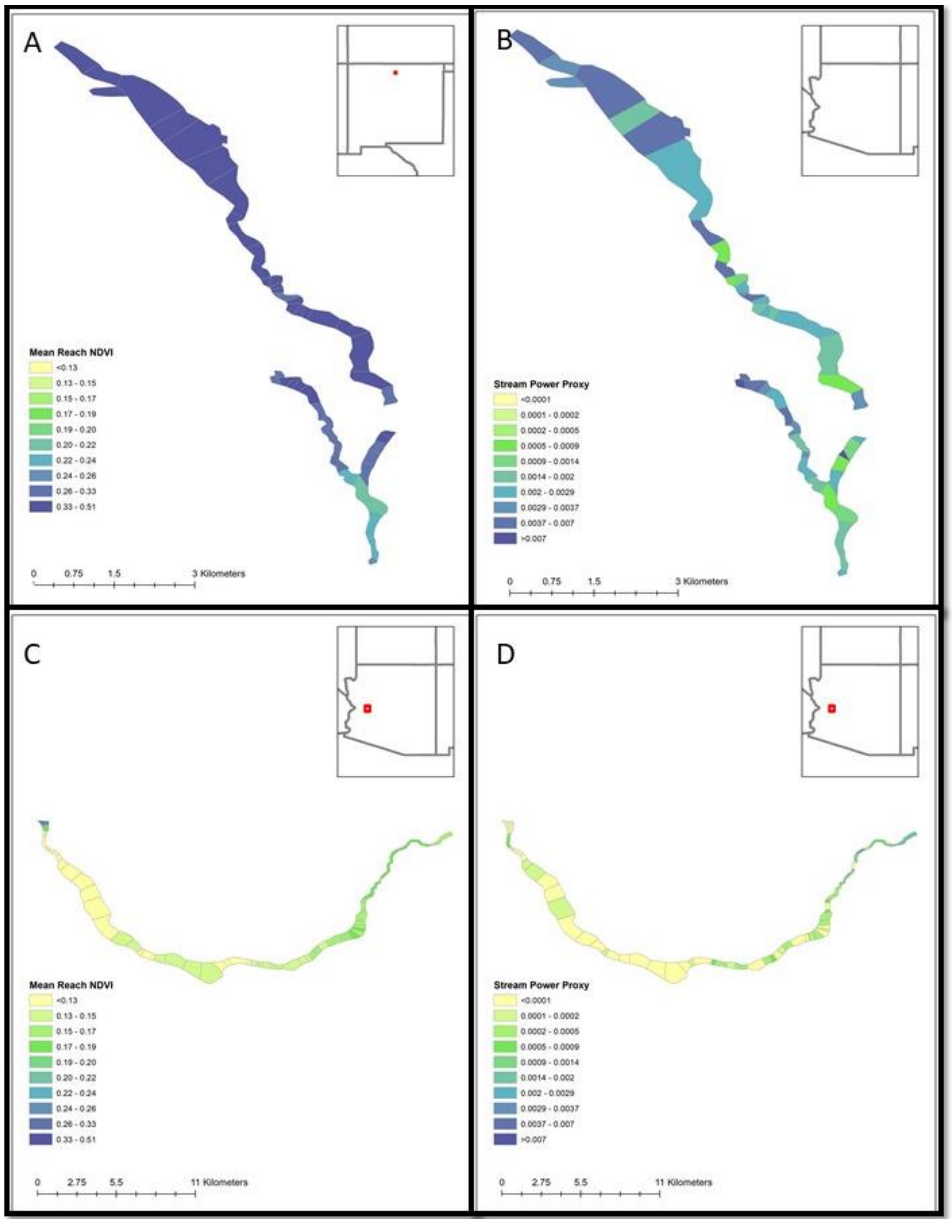

Figure 11. Mean NDVI and stream power proxy along the tributaries to Rio Ojo Caliente and the Santa Maria River. Mean NDVI was much higher at the Rio Ojo Caliente site (A) than the Santa Maria site (C), but mean NDVI generally decreased at both sites from headwaters to mouth. Stream power proxy was erratic at both sites, but was much higher at the Rio Ojo Caliente site (B) than the Santa Maria site (D).

The relationships between mean NDVI and stream power proxy varied were negative and weak for the individual study streams (Table 8, Figures 8-10). However, there were strong relationships between mean NDVI and individual components at some streams. Mean NDVI at the Dry Beaver Creek sites was strongly correlated with channel

slope and drainage area. Channel slope at Dry Beaver Creek is highest near the channel head and decreases downstream, which explains why high mean NDVI values were identified at high slope sites there. Drainage area increases downstream at all sites, and mean NDVI at the Dry Beaver Creek site likely decreases sharply with increasing drainage area because of the effect of elevation. Strong negative relationships between mean NDVI and both channel width and drainage area along the tributary to Rocky Arroyo may be partially explained by the presence of a mine near the tributary junction where the highest values of both drainage area and channel width were found. The strength of the relationship between mean NDVI and distance downstream at the Santa Maria River site can be explained by the presence of a large tributary near the mouth of the study tributary, which greatly increases drainage area. Vegetation is very sparse at both locations.

One possible reason that stream power proxy was not determined to be a strong control on vegetation is that the study streams may be disconnected networks where connectivity of streamflow is not a primary control on vegetation spatial patterns. Stream power proxy patterns vary considerably among the study streams, and many reaches appear to be disconnected (Figure 12). On poorly connected streams, the general vegetation pattern is likely related to the climatological effects of elevation and variations in geologic setting rather than any hydrogeomorphic characteristics of the stream. Most of the headwaters for the study streams were at relatively high elevation where the temperatures are slightly lower and the precipitation and humidity are slightly higher than at lower elevation sites. Vegetation occurs along an elevational gradient in the southwestern United States (Poulos, Taylor, and Beaty 2007) and throughout the

Madrean Sky Island region (Bataineh et al. 2007), even along ephemeral stream channels (McClaran and Brady 1994). The riparian communities along the ephemeral study streams are more strongly affected by climate variables than intermittent or perennial streams (see Chapter III).

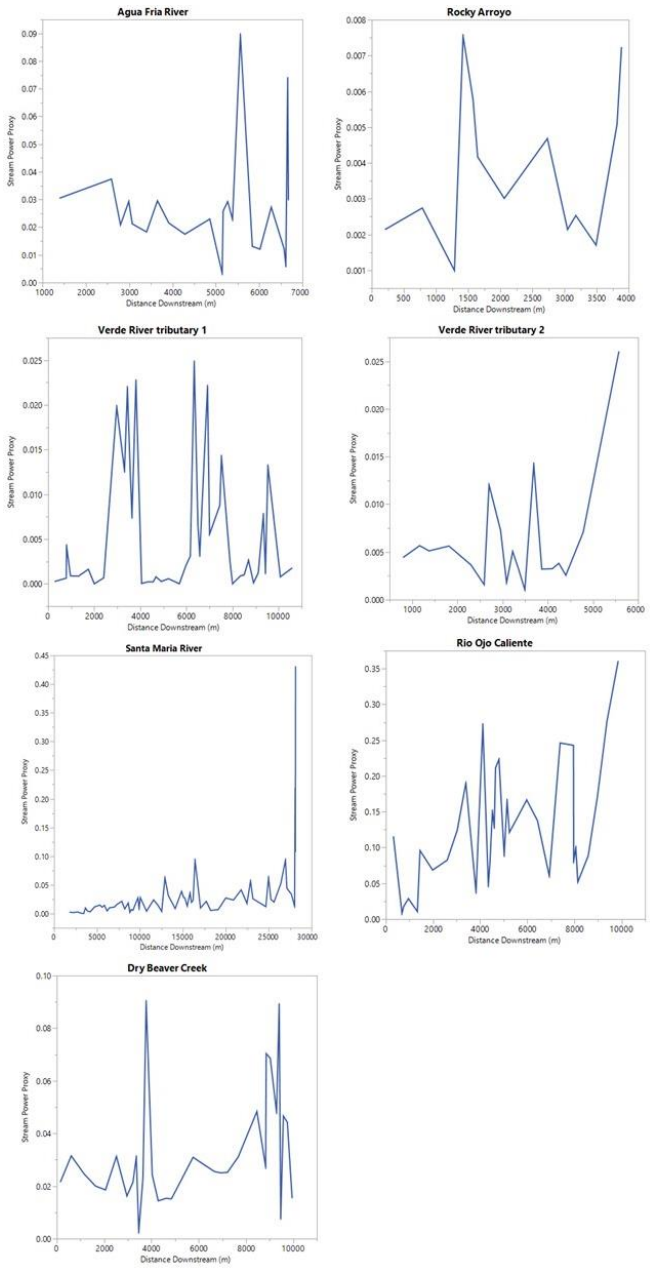

Figure 12. Stream power proxy variance along study stream courses.

If the study streams are disconnected from downstream waterways, then they would not meet the significant nexus standard individually. They may, however, meet the standard in combination with other “similarly situated” streams (Department of Defense and Environmental Protection Agency 2021). Alternatively, individual segments within each stream, especially those near the outlets, may meet the significant nexus standard. The literature on water quality of ephemeral streams and other small rivers and streams is quite limited, and there is some evidence that water quality modelling techniques used for larger rivers are not valid for ephemeral streams (Mannina and Viviani 2010). When an ephemeral stream flows into a larger river, much of the inputs are likely diluted (Patz, Reddy, and Skinner 2004), but less is known about the impacts of inputs from multiple ephemeral streams in the same watershed. The amount and rate of rainfall needed to initiate flow, as well as the frequency of flow in ephemeral streams, varies based on topography, substrate, vegetation cover, and other factors (Kampf et al. 2018). Thus, the combined impact of inputs from ephemeral streams throughout a watershed can be difficult to predict, especially in areas where streamflow and precipitation gages are not present. That said, in a region where the majority of streams do not have perennial flow, such as the southwestern United States (Goodrich et al. 2018), a better understanding of the water quality contributions of ephemeral streams is needed.

Another possible conclusion from this study is that remote sensing alone is not enough to evaluate connectivity along arid ephemeral streams. Weak and inconsistent correlations between mean NDVI and stream power proxy support this conclusion. Stream power proxyvaries widely along the study streams, and mean reach NDVI responds to it in differing ways. Along most of the streams, stream power proxy is high at

the mouth, where drainage areas are largest (Figure 13). Middle reaches along these streams typically have the lowest stream power proxy values because the shallow gradients are balanced by the wide channels. However, there is strong variability in stream power proxy on all streams, especially in the middle reaches. This may partially be explained by errors in the DEMs (i.e., incorrect elevation pixel values) or by the presence of short reaches. Because the reaches were segmented based on planform, some reaches were very short, especially those along braided rivers with high width variability.

The variance in stream power proxy along some of the study streams is quite large (Figure 12) for several reasons. At the Rocky Arroyo site, rapid changes in channel slope result in rapid changes to stream power proxy. At one of the Santa Maria reaches and a few of the Rio Ojo Caliente reaches, a dirt path crosses the channel, which causes an increase in reach slope and an increase in stream power proxy. Many of the reaches with abrupt changes in stream power proxy do not have a corresponding change in NDVI; that might be caused by the length of the reaches. Channel centerline reach length varied widely (range = 43.9-2755.8 m; mean = 383.1) and shorter reaches may be skewing the data.

The response of vegetation to stream power proxy varied considerably among study streams. This result was somewhat expected, as Bendix (1999) observed a similar phenomenon in the Transverse Ranges of Southern California. There, differences in substrate lithology controlled access to water tables, which complicated the response of vegetation to stream power proxy. In the present study, underlying geology was quite diverse, which may, at least in part, explain the differing responses among study streams. The response of mean NDVI to elevation and distance downstream along individual

streams was consistent with the response to the same variables among all streams, which further supports the idea that interactions with substrate complicate the relationship.

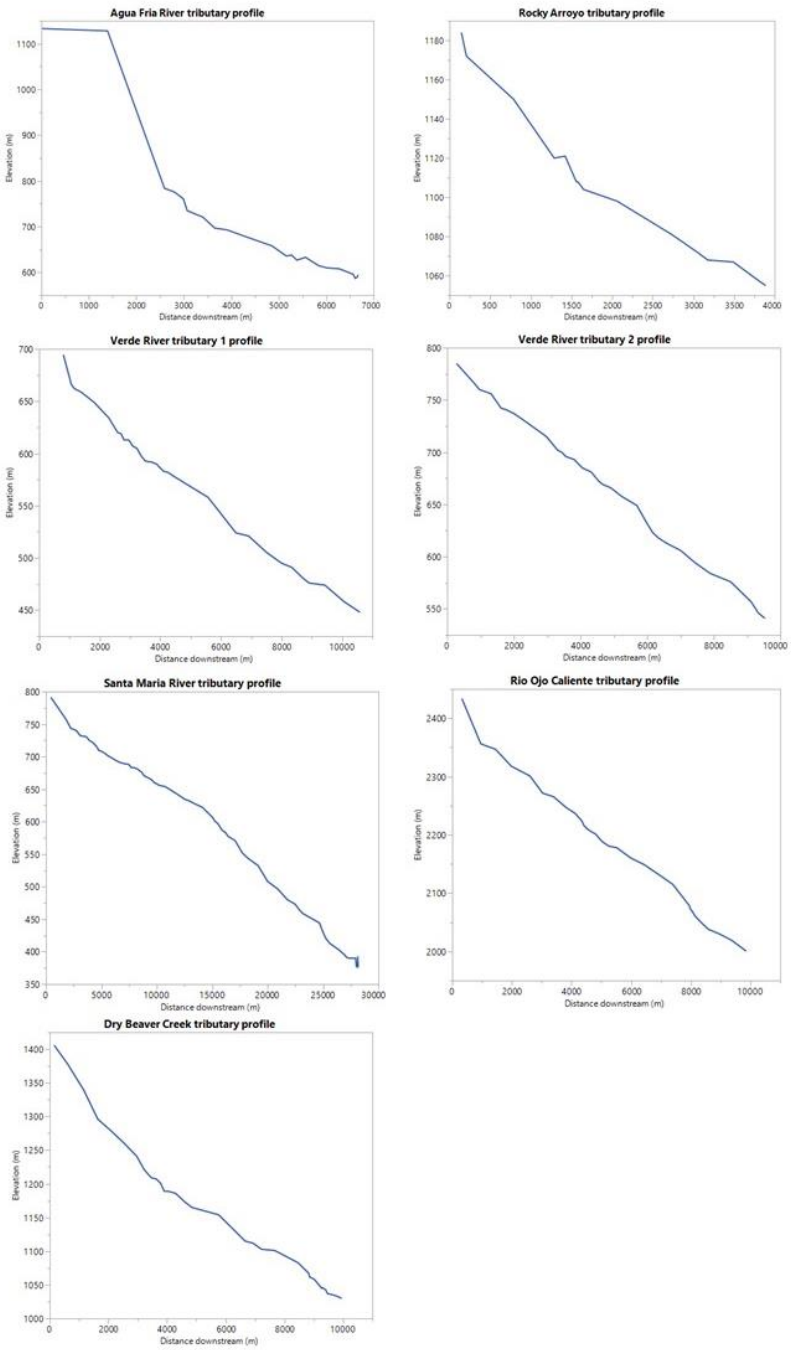

Figure 13. Elevational profiles of study streams. All units are in meters. The Verde River tributary has two profiles because it has two distinct branches. Increases in elevation in profiles is likely due to disagreements between the DEM and the imagery, causing elevation to be measured in the wrong place.

The varying relationships between mean NDVI and stream power proxy may also be caused by differences in planform and erosion/deposition patterns among stream reaches. In areas where there was a change in stream power proxy and a corresponding change in mean NDVI, the change in NDVI was not predictable. In some areas, an increase in stream power proxy corresponded to a decrease in mean NDVI, whereas in other areas, mean NDVI increased with increasing stream power proxy.

Because stream power proxy is a variable with multiple parts (i.e., channel gradient, channel width, drainage area), the response to it is highly varied. An abrupt change in stream power proxy may be caused by an abrupt change in only one of its components, but vegetation responds differently to each component. In areas where stream power proxy increases because of a sharp increase in gradient, the channel and valley may be too narrow to allow for much vegetation growth. Conversely, along sprawling braided channels with shallow gradients, transmission losses to the alluvium may create conditions that are too dry for dense vegetation, even if the width is balanced by a large drainage area. At the Verde River site, the channel planform varies widely from wide, braided channels to narrow, single thread channels. This results in high stream power proxy where the channel narrows and lower stream power proxy where it widens. This pattern likely indicates low connectivity because narrow reaches can transport sediments more efficiently, whereas deposition occurs in the wider, braided reaches. Similar patterns occur at the Santa Maria River site and parts of the Dry Beaver Creek site. Further complicating this relationship is the fact that stream power proxy is not static. It changes during flow events as the processes of deposition and erosion alter the channel (Graf 1983).

In addition, intense scouring associated with high stream power proxy may inhibit vegetation growth. On the other hand, on reaches where stream power proxy is high because the channel is wide and braided, vegetation may capitalize on the expansive alluvium. For example, along the tributary to the Santa Maria River (Figure 10) stream power proxy increased at low elevations because the width of the braided channel increased. In such locations, NDVI decreased, possibly in response to the change in elevation. At higher elevations near the headwaters, the relationship between NDVI and stream power proxy was positive, possibly because of patterns of deposition related to high elevations (i.e., step pools). Bed-steps in ephemeral streams are not necessarily submerged during high-magnitude flow events like they are for perennial streams (Wohl and Grodek 1994), making them effective at trapping sediments. These deposits may encourage plant growth.

Limitations

The present study has certain limitations. Some important data was not available for analysis, including variations in size (height and/or canopy width), species composition, and/or diversity, which may be more closely related to connectivity or other hydrologic variables. Such variables are best observed via ground surveys, which I was not able to do for the present study. Xeroriparian vegetation is generally varied in some way (i.e., diversity, density, size) from terrestrial vegetation even if there are few differences in species composition. For example, Schwinning et al. (2011) found that creosote (*Larrea tridentata*) was larger along small streams than on terrestrial surfaces on an alluvial fan. Stromberg et al. (2017) noted that along arid ephemeral streams, though composition is similar among riparian and upland sites, species richness was greater at

riparian sites than upland sites and varied along an aridity gradient. Data regarding the composition of riparian communities and the size of plants would complement and provide context to the remote sensing data analyzed in the present study.

Soil and substrate variables were not considered in the present study because county soil surveys were not available for all study streams, but they may be more important than some of the variables that were assessed. Some of the variations in mean NDVI may be directly related to alluvial soils associated with stream channels. Because substrate controls the porosity and permeability of the surface, it also controls access to water by plants. Previous studies in Arizona have noted changes in composition and size related to the presence of alluvium. Shaw, Cooper, and Sutfin (2018) found that variation in floristic composition among ephemeral channels was related to whether the bed material was composed of bedrock or alluvium. Stromberg et al. (2017) also questioned whether the presence of alluvium may be responsible for the presence of riparian communities along ephemeral streams.

The DEMs used for this study were very coarse (1/3 arc-second (~10m) resolution), which can create problems when analyzing small features such as ephemeral streams. In areas where the channels were very narrow, meandering, and/or flowing through a canyon or past a recent deposition (e.g., point bar), the DEMs did not always match up to the imagery. This created issues in measuring channel slope that may impact the results. Elevation data with higher resolution may have provided more accurate measurements and would have allowed for better floodplain outlines created with GIS tools such as the Valley Bottom Extraction Tool (V-BET)(Gilbert, Mcfarlane, and Wheaton 2016).

There were a few reaches where land use impacted the channel and/or floodplain, which skewed the results. As mentioned above, there were several locations where dirt paths crossing the channel caused strong variability in slope. However, since the paths impede plant growth, the variance in NDVI and slope did not necessarily relate to one another in an expected way. Along the tributary to Rocky Arroyo, a mine in the floodplain along four reaches caused NDVI to be lower than expected. The channel is used as a driveway for the mine, which compacts the soil and inhibits plant growth. Dust from the mine is also a likely factor in the lower NDVI.

The use of additional or different data in this study may have yielded more accurate results. For example, median NDVI rather than mean NDVI may provide a more clear picture of the vegetation conditions at each reach. The use of stream morphometrics such as drainage density and relief area ratio in addition to the stream power proxy may also improve these results. In addition, stream power proxy values for each reach and its upstream and downstream reaches may produce stronger results. The stream power proxy variable itself is a proxy of a proxy for connectivity, and the other connectivity metrics may have stronger relationships to vegetation.

Conclusions

Overall, the results here point to two possible conclusions: (1) that remote sensing of riparian vegetation is not an effective method for determining connectivity along arid ephemeral streams without additional boots-on-the-ground study, or (2) that the study streams analyzed here are disconnected from downstream waterways. Further study is needed to effectively determine whether a significant nexus is present for a given stream or group of similar streams. Riparian communities along individual streams vary in their

responses to hydrologic variables, making comparisons among streams challenging and emphasizing the need for rigorous case-by-case analysis.

V. CONCLUSION

This dissertation attempts to identify relationships between stream characteristics and the spatial patterns of riparian communities along ephemeral streams in the arid southwestern United States. My results indicate that riparian vegetation responds much more strongly to topographic and climatic variables than it does to streamflow. This is somewhat expected because the trees found along the study streams are likely upland trees. These results give rise to further questions about the riparian communities along the study streams.

Future study on the riparian communities of arid ephemeral streams should focus on finer-scale variations in tree size, species, and substrate characteristics. Because of travel restrictions during the Covid-19 pandemic, I was not able to conduct field work for this dissertation and therefore was not able to include these variables. More information about the riparian communities and their environments may have altered the results of this study. It may have also provided insight into whether the riparian communities along the study streams are present because of the infrequent flows or because of the presence of alluvium.

Water quality management considerations for ephemeral streams

Lack of data and research on ephemeral streams creates challenges for watershed management of ephemeral streams for downstream water quality, however the limited information that is available can provide some guidance.

Because connectivity throughout ephemeral watersheds is likely low, water quality protections for entire watersheds or even entire streams may not be necessary in every case. TNWs are typically protected at the segment scale, wherein only certain

portions of some rivers qualify as a TNW. For example, there are two segments of the Gila River that are deemed navigable in Arizona; one segment stretches from Coolidge Dam to Winkleman and the other from Powers Butte to Gillespie Dam (U.S. Army Corps of Engineers 2022). Both segments are perennial because of water management infrastructure (i.e., dam outflow and irrigation runoff), whereas most of the rest of the Gila River in Arizona is intermittent or ephemeral.

Varying protections for ephemeral streams based on segment connectivity may be appropriate as well. For example, in the previous chapters, I noted that vegetation clusters were larger and mean NDVI was higher at the headwaters. Because the vegetation cover on headwaters reaches is high, they may be target areas for conservation of biodiversity, but because stream power proxy (and probably connectivity) varies considerably in the middle reaches, headwaters streams may be less of a priority for water quality protection. Lower reaches may be more likely to influence water quality in the streams they drain into, so it may be better to focus on water quality protections there. This is not to say that middle reaches are not worthy of protection – they have ecological and cultural value – but based on this study they are not likely to significantly alter downstream water quality.

Since many ephemeral streams lose streamflow to the alluvium on a downstream gradient, assessing the volume of water contributed by a given watershed during a high flow pulse is challenging. Scientists and stakeholders addressed a similar problem when determining the appropriate environmental flow pulses to meet the agreements for Minute 319 amendment to the U.S. – Mexico Water Treaty of 1944. In 2014, an environmental flow was planned to reconnect the Colorado River from Morelos Dam near Yuma, AZ to the Colorado Delta in Mexico. Because of the water management

practices upstream including irrigation and large dams, much of the river below the dam is ephemeral. Scientists were tasked with determining the appropriate volume of water that would reconnect the river through the losing reaches (Pitt and Kendy 2017).

Though the Minute 319 project was a binational experiment on a large river, there are some aspects of it that could be applied in local-scale management of smaller ephemeral watersheds. The researchers working to determine the minimum flow that would reconnect the river had little data to work with regarding historic flow regimes (Pitt and Kendy 2017). Instead, they relied on ecological evaluations of the hydro- and meso- riparian vegetation to estimate flow requirements at each section of the river (Nelson et al. 2017; Pitt and Kendy 2017). Better information on how xeroriparian vegetation interacts with streamflow could allow for similar assessments to determine connectivity on ephemeral streams.

This dissertation is one of the first studies analyzing the longitudinal (rather than lateral or vertical) structure (i.e., canopy size, density) of riparian communities along ephemeral streams in arid and semi-arid regions. While questions about these communities remain to be answered, this work has contributed empirical data that can be used to inform policy and management decisions.

APPENDIX SECTION

APPENDIX A: STREAMFLOW METRICS

*****All values are in m^3/s unless otherwise stated.

Metric	**Agua Fria River**	**Dry Beaver Creek**	**Rocky Arroyo**	**Rio Ojo Caliente**	**Santa Maria River**	**Verde River**
USGS gage number	09512800	09505350	08401900	08289000	09424900	09511300
Contributing area (km^2)	2877.49	367.78	738.15	1085.21	2924.11	17132.85
Period of record	1970-present	1960-present	1963-present	1932-present	1966-present	1961-present
Percent time exceeded=10	0.45	2.25	0	4.37	2.23	28.28
Percent time exceeded=25	0.14	0.01	0	0.55	0.01	15.63
Percent time exceeded=50	0.05	0	0	0.42	0	7.50
Percent time exceeded=75	0.02	0	0	0.28	0	2.63
Percent time exceeded=90	0.01	0	0	0.15	0	1.42
1-year return interval	16.87	4.07	0.06	6.17	1.91	8.29
5-year return interval	304.76	229.78	254.53	51.27	279.26	342.00
10-year return interval	446.69	382.08	531.92	66.38	476.76	768.98
50-year return interval	838.95	924.05	1610.83	102.38	1149.54	4050.03
100-year return interval	1050.98	1260.76	2259.20	118.59	1543.52	7912.94
Mean annual flow	0.60	1.11	0.16	1.83	1.98	16.79
90-day minimum flow	0.02	0	0	0.24	0	4.10
Low pulse count (count)	5	0	0	5	0	6
Low pulse duration (days)	7	0	0	9	0	6.25
High pulse count (count)	9	6	2	4	2.5	8
High pulse duration (days)	2.5	3	2	3	5	6.25
Extreme low flow timing (Julian date)	207	225	218.5	211	208.8	286
High flow timing (Julian date)	220	231	223	173	21.5	321

APPENDIX B: CORRELATION MATRIX FOR CHAPTER II VARIABLES

	Precipitation	Max temperature	Min temperature	Mean temperature	Vapor pressure	Aspect	Elevation	Confinement	Channel slope	Sinuosity	Valley slope	Valley width	Channel width	Distance from channel head
Distance from channel head	0.23	0.34	0.33	0.34	0.26	0.05	-0.40	0.07	-0.04	0.02	-0.01	0.55	0.44	1.00
Channel width	0.39	0.50	0.48	0.49	0.45	-0.03	-0.51	0.61	-0.01	0.05	0.02	0.56	1.00	0.44
Valley width	0.06	0.16	0.12	0.14	0.16	0.00	-0.18	-0.06	-0.04	-0.01	0.00	1.00	0.56	0.55
Valley slope	0.05	0.05	0.05	0.05	0.04	-0.01	-0.05	0.04	0.00	0.01	1.00	0.00	0.02	-0.01
Sinuosity	-0.04	-0.01	-0.02	-0.02	0.00	0.01	0.01	0.04	0.01	1.00	0.01	-0.01	0.05	0.02
Channel slope	-0.03	-0.04	-0.03	-0.03	-0.03	-0.01	0.04	0.04	1.00	0.01	0.00	-0.04	-0.01	-0.04
Confinement	0.50	0.53	0.57	0.56	0.47	-0.09	-0.53	1.00	0.04	0.04	0.04	-0.06	0.61	0.07
Elevation	-0.88	-1.00	-0.99	-1.00	-0.85	0.14	1.00	-0.53	0.04	0.01	-0.05	-0.18	-0.51	-0.40
Aspect	-0.16	-0.15	-0.15	-0.15	-0.13	1.00	0.14	-0.09	-0.01	0.01	-0.01	0.00	-0.03	0.05
Vapor pressure deficit	0.65	0.85	0.84	0.85	1.00	-0.13	-0.85	0.47	-0.03	0.00	0.04	0.16	0.45	0.26
Mean temperature	0.89	0.99	1.00	1.00	0.85	-0.15	-1.00	0.56	-0.03	-0.02	0.05	0.14	0.49	0.34
Min temperature	0.89	0.98	1.00	1.00	0.84	-0.15	-0.99	0.57	-0.03	-0.02	0.05	0.12	0.48	0.33
Max temperature	0.88	1.00	0.98	0.99	0.85	-0.15	-1.00	0.53	-0.04	-0.01	0.05	0.16	0.50	0.34
Precipitation	1.00	0.88	0.89	0.89	0.65	-0.16	-0.88	0.50	-0.03	-0.04	0.05	0.06	0.39	0.23

APPENDIX C: CONDITIONAL INFERENCE TREE FIGURES

```
|--- valley width (m) <= 2959.23
|  |--- Mean daily temperature (°C) <= 23.41
|  |  |--- valley width (m) <= 2834.06
|  |  |  |--- Elevation <= 2323.33
|  |  |  |  |--- Elevation <= 142.57
|  |  |  |  |  |--- Elevation <= 142.39
|  |  |  |  |  |  |--- class: 1
|  |  |  |  |  |--- Elevation >  142.39
|  |  |  |  |  |  |--- class: 4
|  |  |  |  |--- Elevation >  142.57
|  |  |  |  |  |--- Aspect <= -0.50
|  |  |  |  |  |  |--- High flow timing <= 276.00
|  |  |  |  |  |  |  |--- Mean daily temperature (°C)  <= 23.39
|  |  |  |  |  |  |  |  |--- Mean daily vapor pressure deficit <= 25.26
|  |  |  |  |  |  |  |  |  |--- Maximum daily temperature (°C) <= 30.59
|  |  |  |  |  |  |  |  |  |  |--- class: 1
|  |  |  |  |  |  |  |  |  |--- Maximum daily temperature (°C)  >  30.59
|  |  |  |  |  |  |  |  |  |  |--- Distance from channel head <= 29876.72
|  |  |  |  |  |  |  |  |  |  |  |--- truncated branch of depth 7
|  |  |  |  |  |  |  |  |  |  |--- Distance from channel head >  29876.72
|  |  |  |  |  |  |  |  |  |  |  |--- truncated branch of depth 2
|  |  |  |  |  |  |  |  |--- Mean daily vapor pressure deficit >  25.26
|  |  |  |  |  |  |  |  |  |--- valley width <= 1481.94
|  |  |  |  |  |  |  |  |  |  |--- Distance from channel head <= 26862.63
|  |  |  |  |  |  |  |  |  |  |  |--- truncated branch of depth 2
|  |  |  |  |  |  |  |  |  |  |--- Distance from channel head >  26862.63
|  |  |  |  |  |  |  |  |  |  |  |--- class: 1
|  |  |  |  |  |  |  |  |  |--- valley width >  1481.94
|  |  |  |  |  |  |  |  |  |  |--- Distance from channel head <= 30399.80
|  |  |  |  |  |  |  |  |  |  |  |--- class: 1
|  |  |  |  |  |  |  |  |  |  |--- Distance from channel head >  30399.80
|  |  |  |  |  |  |  |  |  |  |  |--- truncated branch of depth 2
|  |  |  |  |  |  |  |  |--- Mean daily temperature (°C)   >  23.39
|  |  |  |  |  |  |  |  |--- Distance from channel head <= 78047.20
|  |  |  |  |  |  |  |  |  |--- Distance from channel head <= 77725.72
|  |  |  |  |  |  |  |  |  |  |--- class: 1
|  |  |  |  |  |  |  |  |  |--- Distance from channel head >  77725.72
|  |  |  |  |  |  |  |  |  |  |--- class: 4
|  |  |  |  |  |  |  |  |--- Distance from channel head >  78047.20
|  |  |  |  |  |  |  |  |  |--- class: 1
|  |  |  |  |  |  |  |--- High flow timing >  276.00
|  |  |  |  |  |  |  |  |--- Precipitation (mm) <= 3.75
|  |  |  |  |  |  |  |  |--- class: 1
|  |  |  |  |  |  |  |  |--- Precipitation (mm) >  3.75
|  |  |  |  |  |  |  |  |--- Elevation <= 439.00
|  |  |  |  |  |  |  |  |  |--- class: 1
|  |  |  |  |  |  |  |  |--- Elevation >  439.00
|  |  |  |  |  |  |  |  |  |--- class: 4
|  |  |  |  |  |--- Aspect >  -0.50
|  |  |  |  |  |  |--- valley width <= 2683.95
|  |  |  |  |  |  |  |--- High pulse count <= 1.00
|  |  |  |  |  |  |  |  |--- Distance from channel head <= 76.94
|  |  |  |  |  |  |  |  |  |--- class: 2
|  |  |  |  |  |  |  |  |--- Distance from channel head >  76.94
|  |  |  |  |  |  |  |  |  |--- Distance from channel head <= 78668.95
|  |  |  |  |  |  |  |  |  |  |--- Distance from channel head <= 28633.99
|  |  |  |  |  |  |  |  |  |  |  |--- truncated branch of depth 9
|  |  |  |  |  |  |  |  |  |  |--- Distance from channel head >  28633.99
```

Figure 1. Conditional inference trees – full dataset

```
|--- Distance from channel head (m) <= 27123.24
|   |--- Elevation <= 2323.33
|   |   |--- Sinuosity <= 1.68
|   |   |   |--- Distance from channel head (m)  <= 26859.09
|   |   |   |   |--- channel slope <= 0.06
|   |   |   |   |   |--- Maximum daily temperature (°C) <= 16.93
|   |   |   |   |   |   |--- Aspect <= 35.51
|   |   |   |   |   |   |   |--- class: 3
|   |   |   |   |   |   |--- Aspect >  35.51
|   |   |   |   |   |   |   |--- class: 1
|   |   |   |   |   |--- Maximum daily temperature (°C)  >  16.93
|   |   |   |   |   |   |--- class: 1
|   |   |   |   |--- channel slope >  0.06
|   |   |   |   |   |--- channel slope <= 0.06
|   |   |   |   |   |   |--- Aspect <= 50.48
|   |   |   |   |   |   |   |--- class: 2
|   |   |   |   |   |   |--- Aspect >  50.48
|   |   |   |   |   |   |   |--- class: 1
|   |   |   |   |   |--- channel slope >  0.06
|   |   |   |   |   |   |--- Max daily temperature (°C)  <= 23.19
|   |   |   |   |   |   |   |--- Elevation <= 1377.07
|   |   |   |   |   |   |   |   |--- class: 4
|   |   |   |   |   |   |   |--- Elevation >  1377.07
|   |   |   |   |   |   |   |   |--- class: 1
|   |   |   |   |   |   |--- Max daily temperature (°C)  >  23.19
|   |   |   |   |   |   |   |--- Mean daily vapor pressure deficit <= 21.10
|   |   |   |   |   |   |   |   |--- class: 1
|   |   |   |   |   |   |   |--- Mean daily vapor pressure deficit >  21.10
|   |   |   |   |   |   |   |   |--- Max daily temperature(°C)<= 26.59
|   |   |   |   |   |   |   |   |   |--- class: 2
|   |   |   |   |   |   |   |   |--- Max daily temperature (°C >  26.59
|   |   |   |   |   |   |   |   |   |--- class: 1
|   |   |   |--- Distance from channel head (m)  >  26859.09
|   |   |   |   |--- Distance from channel head (m)  <= 26862.63
|   |   |   |   |   |--- class: 4
|   |   |   |   |--- Distance from channel head (m)  >  26862.63
|   |   |   |   |   |--- class: 1
|   |   |--- Sinuosity >  1.68
|   |   |   |--- Distance from channel head (m)  <= 3528.31
|   |   |   |   |--- class: 1
|   |   |   |--- Distance from channel head (m)  >  3528.31
|   |   |   |   |--- class: 2
|   |--- Elevation >  2323.33
|   |   |--- Min daily temperature (°C) <= 0.66
|   |   |   |--- class: 1
|   |   |--- Min daily temperature (°C)  >  0.66
|   |   |   |--- class: 4
|--- Distance from channel head (m)  >  27123.24
|   |--- class: 4
```

Figure 2. Conditional inference trees – ephemeral streams.

APPENDIX D: LINEAR REGRESSION GRAPHS

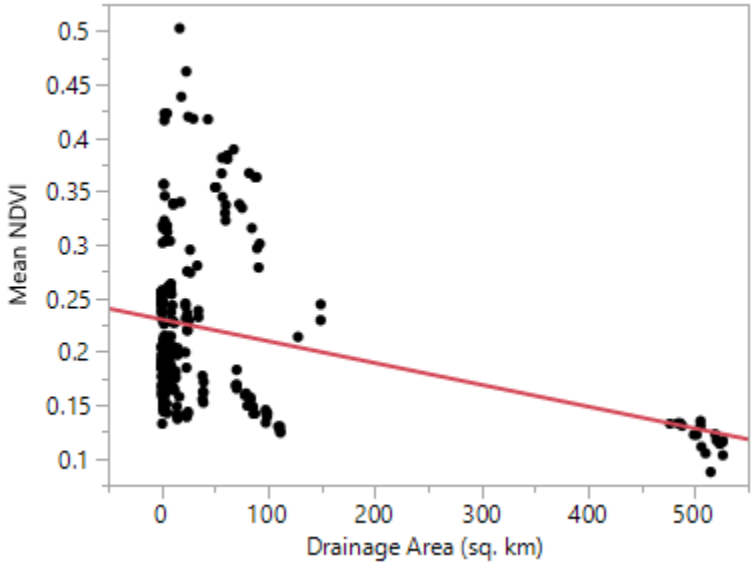

Figure 3. Relationship between mean reach NDVI and drainage area (km^2) for all ephemeral streams.

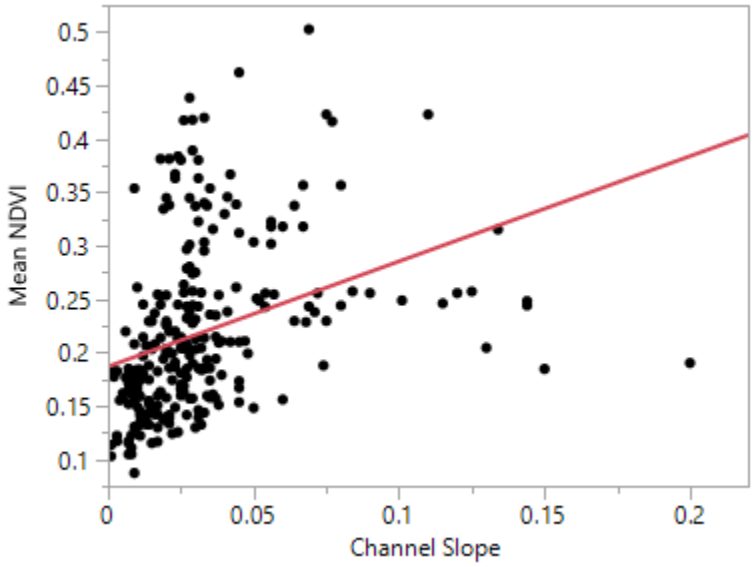

Figure 4. Relationship between mean reach NDVI and channel slope for all ephemeral streams.

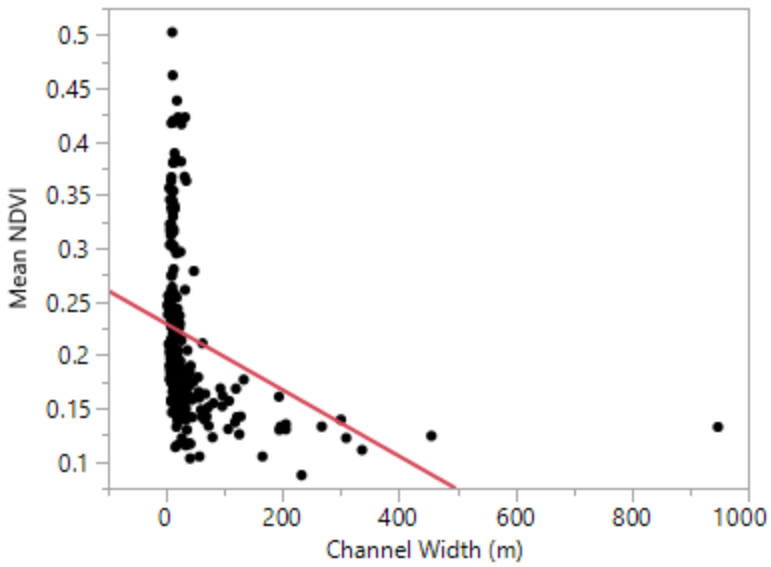

Figure 5. Relationship between mean reach NDVI and channel width (m) for all ephemeral streams.

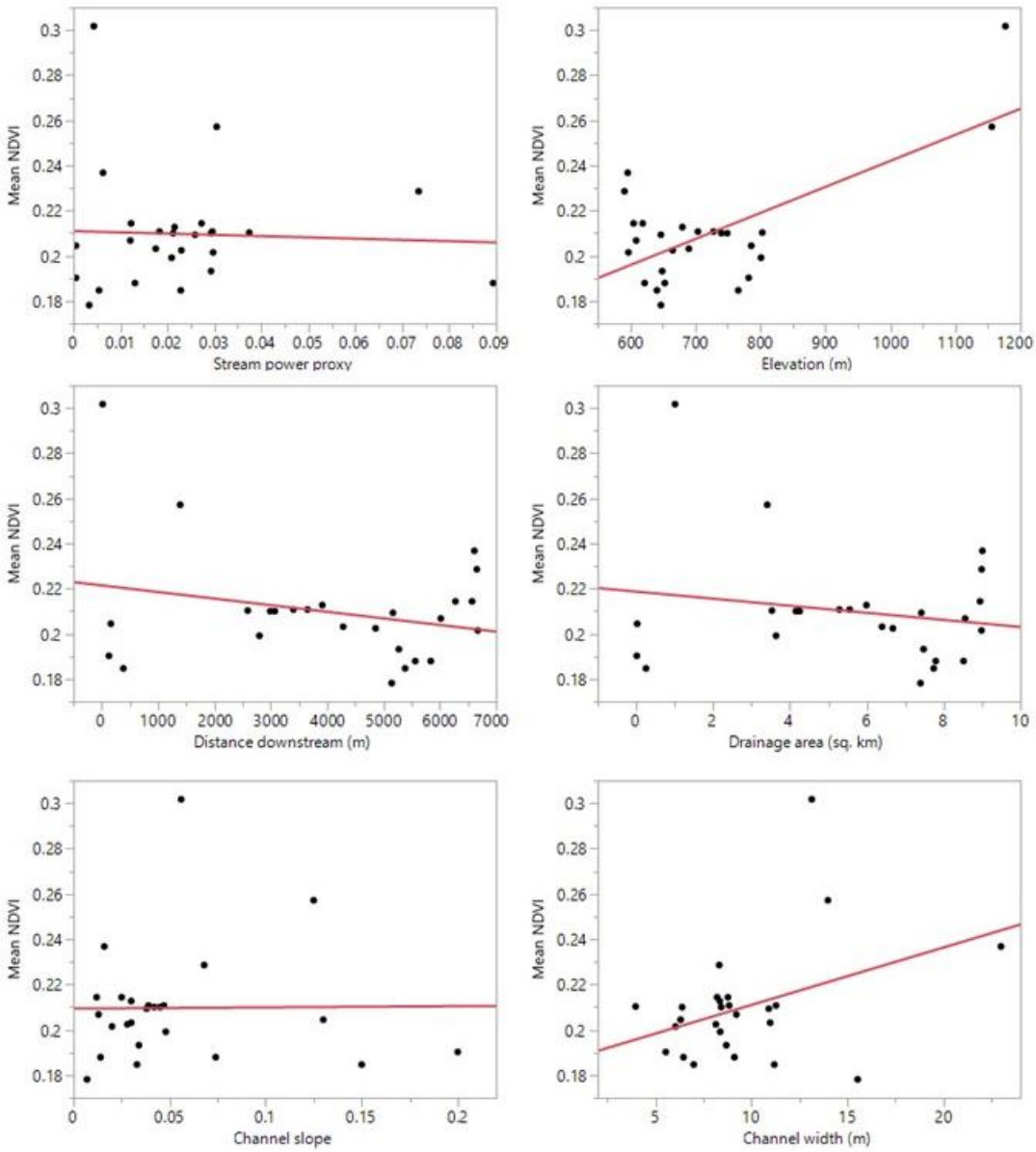

Figure 5. Linear regression results for the tributary to the Agua Fria River.

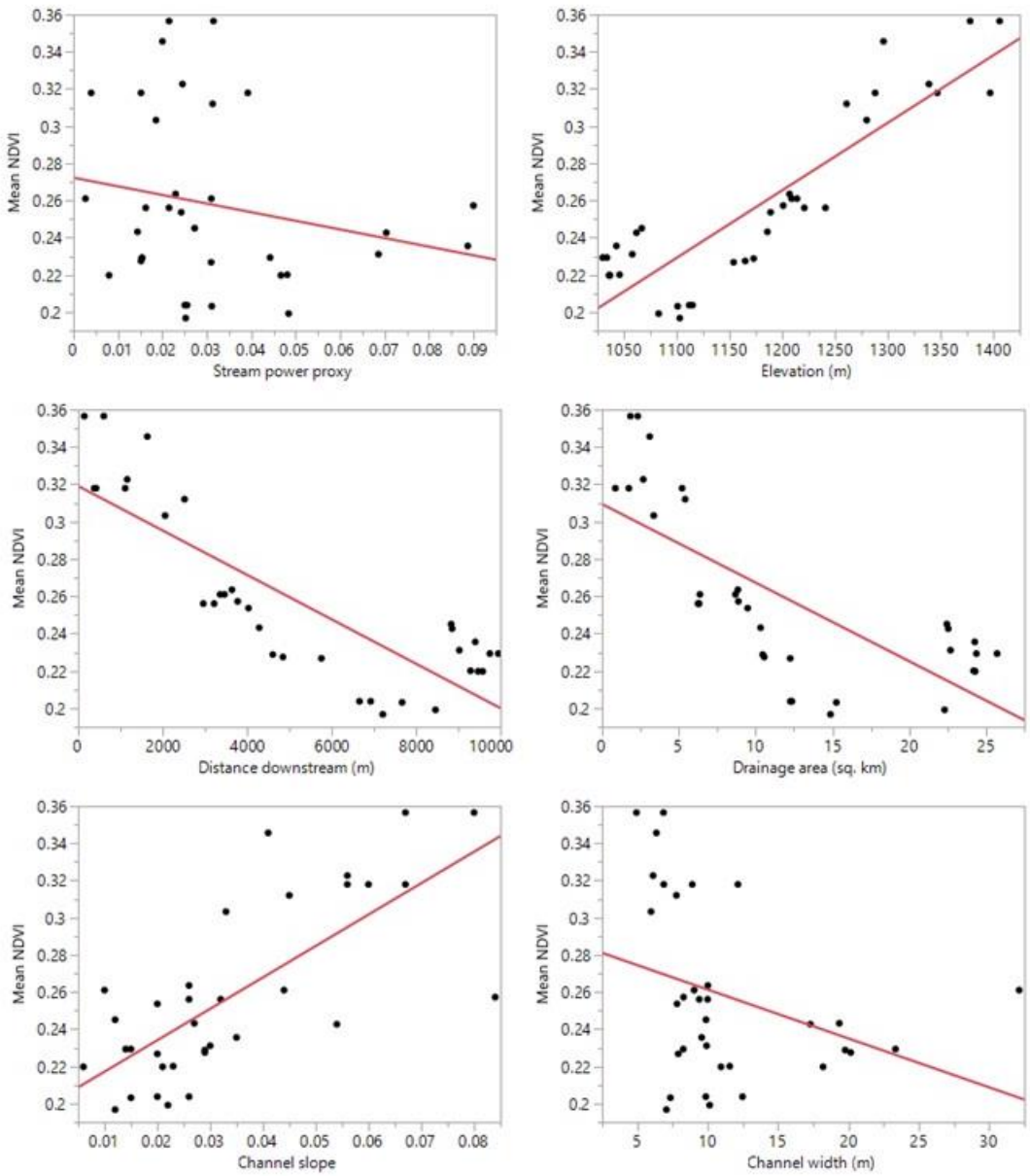

Figure 6. Linear regression results for the tributary to Dry Beaver Creek.

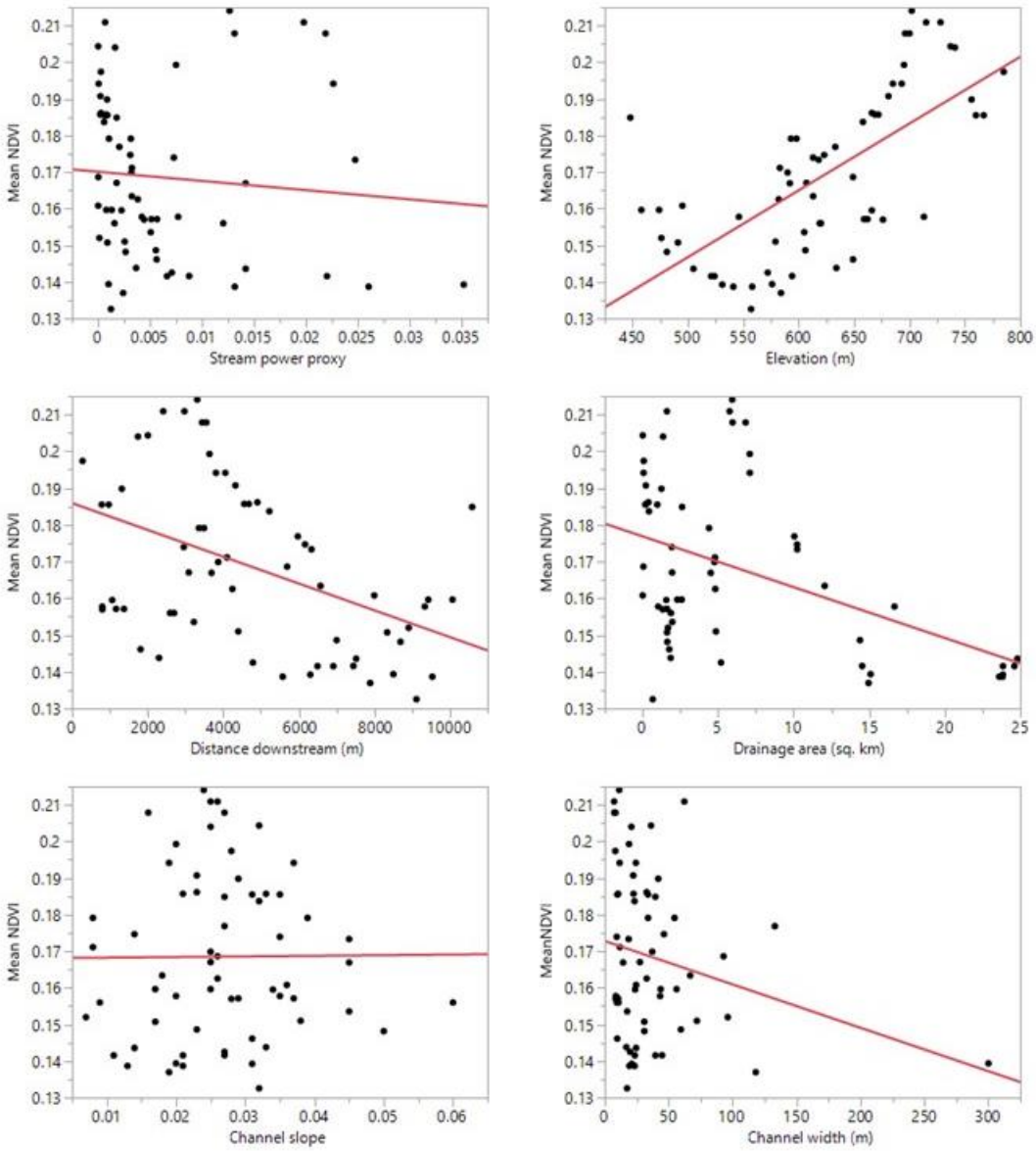

Figure 7. Linear regression results for the tributary to the Verde River.

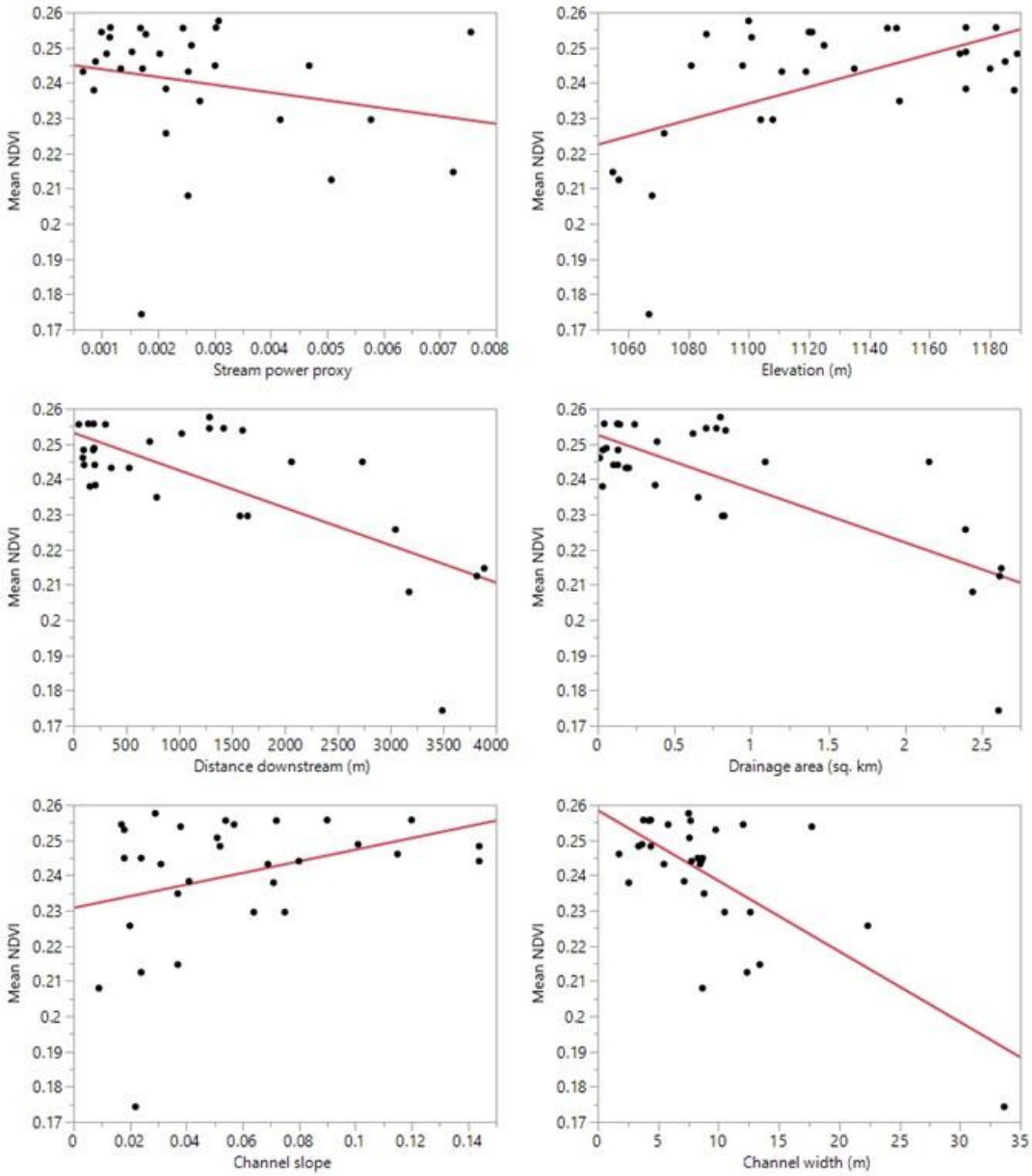

Figure 8. Linear regression results for the tributary to Rocky Arroyo.

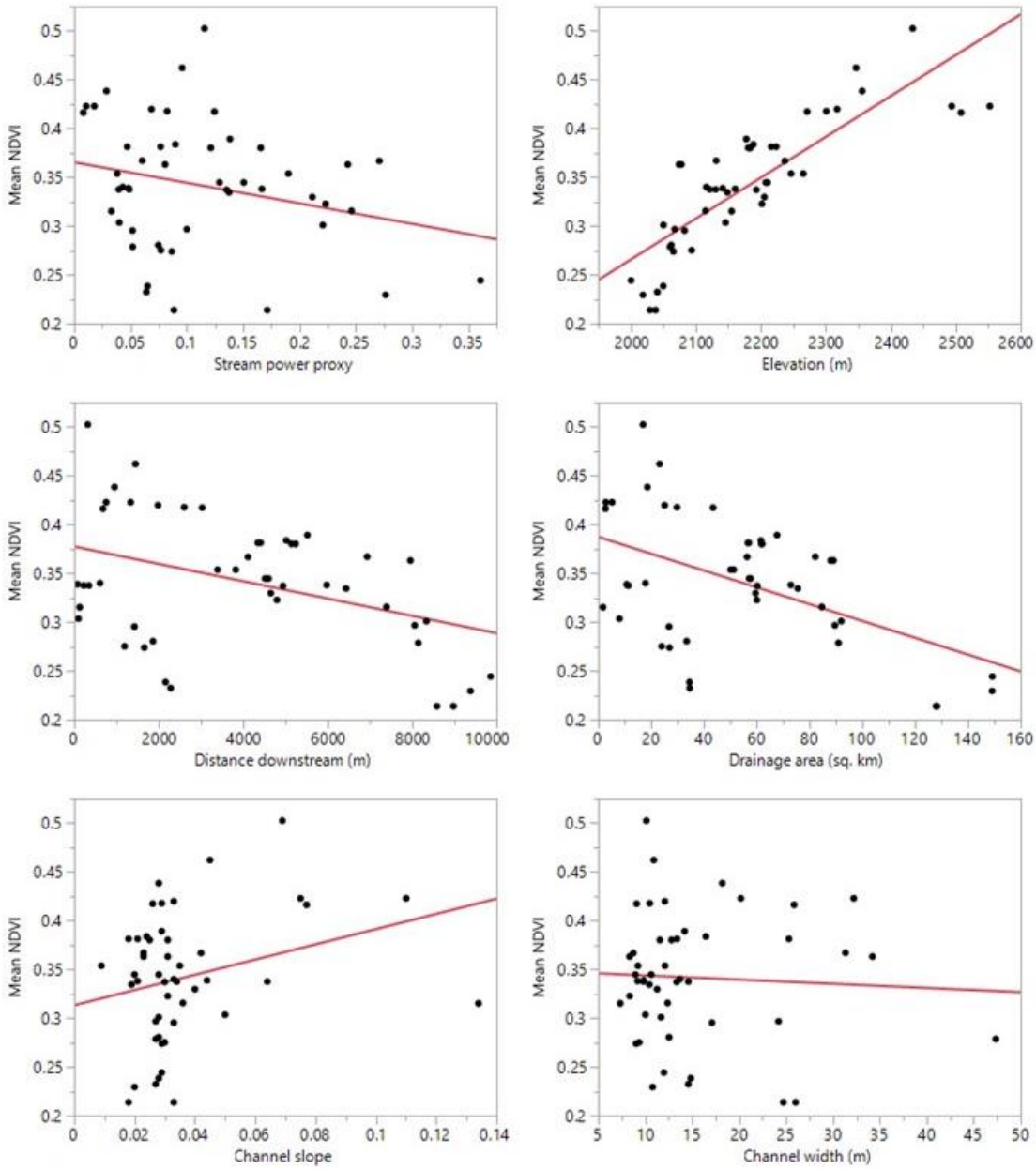

Figure 9. Linear regression results for the tributary to Rio Ojo Caliente.

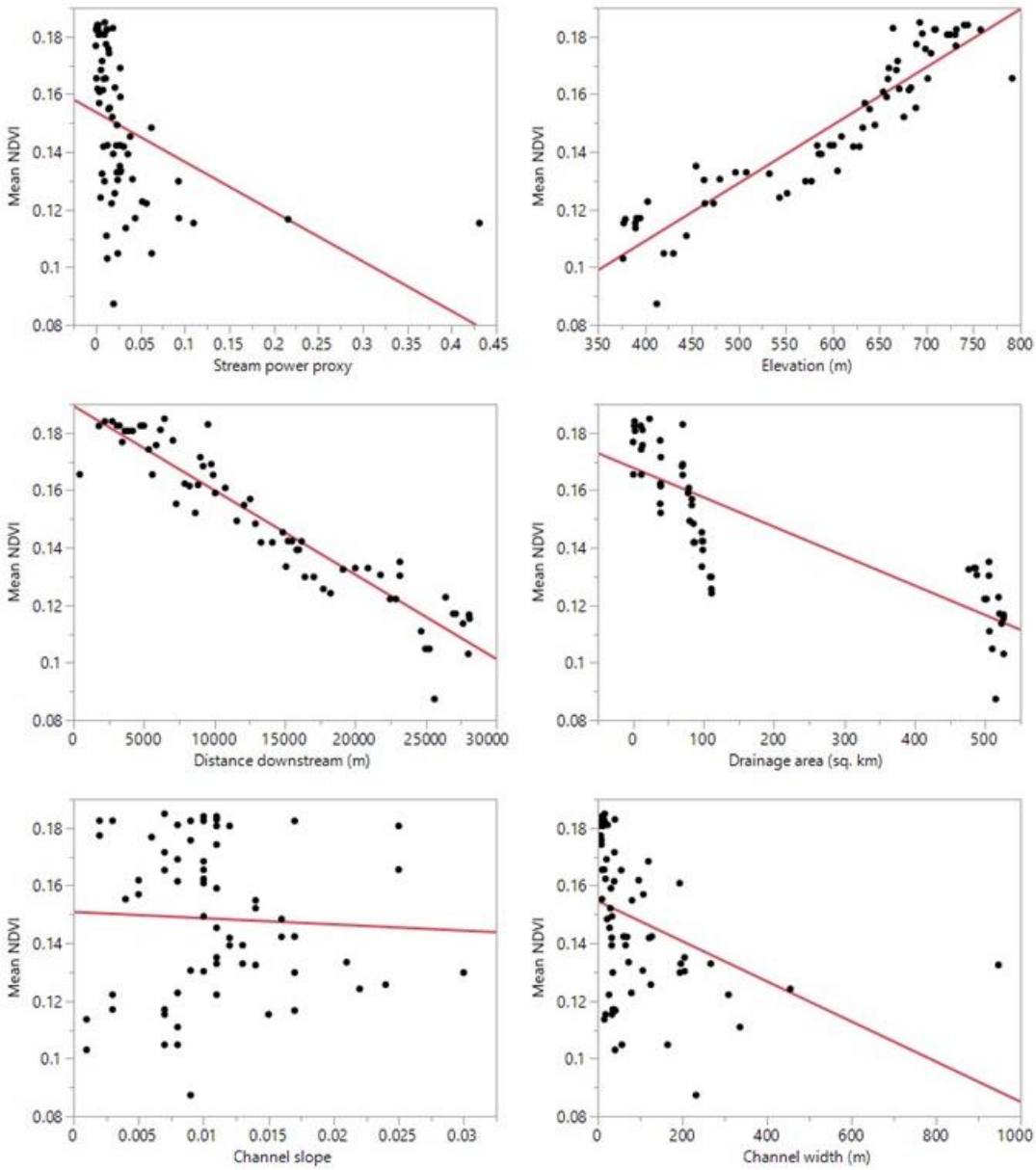

Figure 10. Linear regression results for the tributary to the Santa Maria River.

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