Assessing variations in water availability to vegetation and its consequences on the riparian forest of the arid southwestern USA in service of ecosystem conservation

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Abstract

As Earth's climate changes, a solid understanding of ecosystems' sensitivity and reactivity to climatic and environmental controls is critical. Vegetation is considered a key structural element of habitats and ecosystems. By studying changes in vegetation communities' (specific species assemblages) distribution, health, and timing of main life events, compared with potential controls, such as water availability, it is possible to infer important information on the interactions between vegetation communities (an indicator of habitat integrity) and a changing climate. This comprehension of vegetation dynamics is also crucial for the conservation of species vulnerable to changes and their habitat, even more so in the case of species that are already considered endangered. To fulfil their conservation mission, natural resources managers of lands, regardless of ownership, need a regional-scale understanding of climatic and environmental controls on habitat distribution, condition and vulnerability to climate change. They also need to be able to monitor habitat condition and distribution efficiently and accurately with limited resources.

This thesis examines the case of riparian habitats in drylands. Intermittent streams and their streamside vegetation act as moist and cool refuges, support a high species richness and buffer sensitive populations from drought. But they are also rare, isolated, and highly sensitive to changes in water availability. My goal is to reconstruct the complex links between water distribution, water availability to vegetation, and vegetation distribution across a diverse landscape covering a range of elevation, topography and geology, supporting diverse ecosystems. To achieve this landscape-scale overview, I use remote sensing to map the main vegetation communities distribution, their long-term evolution, health, and sensitivity to drought, and conductivity sensors to detect flow and open water presence in-non perennial streams. These methods provide high spatiotemporal resolution datasets that can cover wide areas. The resulting information on vegetation and flow is then compared to climatic and environmental factors, including local geology and rainfall, to better understand their potential vulnerability to drought. This work is focused on applied research and is set in a wider context of providing knowledge and tools for natural resources managers to keep track of rare and sensitive habitats' condition and extent. The methods and tools used were chosen to test their suitability as accessible and efficient monitoring tools. This thesis focuses on an area in the Southwest USA, covering the upper basin of the San Pedro River (an intermittent river of the Colorado basin), the Huachuca Mountains (an isolated mountain range) and the semi-arid valley in between. My research was conducted on military lands and, as such, the natural resources management side of this work is seen through the lens of military installations and their specific approach to ecosystem conservation. However, the findings of my thesis, both on the ecosystem functioning and the management implications sides, can be more widely applied to monitoring of isolated ecosystems in drylands.

This work provides an overall understanding of water availability to vegetation across a diverse landscape, and how this availability controls vegetation distribution and health, from rainfall-supported grassland to groundwater-fed dense riparian forests. I study the differences in long-term, interannual vegetation density variations between ephemeral and perennial reaches along a dryland river, showing how local geology and groundwater levels can buffer riparian ecosystems against drought. I also describe the spatiotemporal distribution of flow in mountain ephemeral streams, from perennial springs to rainfall-fed washes. I link flow permanence to underlying geology, consider how different reaches might be impacted by prolonged drought, and the consequences on local flora and fauna. Finally, I look at short-term, intra-annual changes in vegetation phenology along mountain streams and show how droughts, with higher temperatures and lower precipitation, might shorten the growing season and negatively impact vegetation leaf density. Throughout the thesis, I relate my findings to natural resources management questions and needs, with the goal of providing tools and conclusions useful for endangered and critical habitat monitoring.

Acknowledgments

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List of abbreviations

DoD: Department of Defence
EC: electrical conductivity
ENRD: Environmental and Natural Resources Division
ESA: Endangered Species Act
IDW: inverse distance weighting
INRMP: Natural Resources Management Plans
IPCC: Intergovernmental Panel on Climate Change
m.a.s.l.: meters above sea level
NAIP: National Agriculture Imagery Program
NAM: North American Monsoon
NASA: National Aeronautics and Space Administration
NDVI: Normalized Difference Vegetation Index
NOAA: National Oceanic and Atmospheric Administration
PET: potential evapotranspiration
REPI: Readiness and Environmental Protection Integration
SERDP: Strategic Environmental Research and Development Program
SPRNCA: San Pedro Riparian National Conservation Area
T&E species: threatened and endangered species
USAIC: United States Army Intelligence Center
USGS: United States Geological Survey

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Chapter 1
General introduction

1.1 Thesis aims

The goal of this thesis is to provide an overarching understanding of water and vegetation distribution across a diversity of ecosystems, to assess resilience of these ecosystems to climate change, and to develop monitoring tools and methods for natural resources managers, with a focus on riparian vegetation in drylands, as they represent critical habitats and refuge for an important biodiversity but are highly sensitive to perturbations in water availability. This work aims at providing keys to natural resources managers to better predict and plan for the impacts of climate change on riparian habitats and present metrics that could be implemented for continuous monitoring of these ecosystems. As such, this thesis is a case study in applied research, and the decisions taken when choosing research aims, methods and analysis were guided by the focus on applications to natural resources management practices.

I use analysis of remote sensing data and in situ conductivity sensors to study water and vegetation distribution over space and time in arid southern Arizona, USA, more specifically in the San Pedro River basin. The study site has been chosen for its diversity of vegetation communities and riparian systems, from mountain ephemeral streams to perennial valley river, covering a small area and sharing similar climatic forcings dominated by the North-American Monsoon. I linked spatiotemporal distribution of water availability in the form of streamflow, soil moisture and groundwater to vegetation metrics from remote sensing. Then I compare these relationships with climatic variables and local geology to better understand environmental controls and potential impacts of climate change on water distribution and availability to vegetation. Rainfall, streamflow and groundwater are then compared to vegetation greenness, used as a proxy for health, and life cycle to measure vegetation sensitivity to variations in water availability and potential resilience to climate change-induced drought. I conduct a quantitative analysis of forest health and functioning over recent decades as a means to detect and assess the response of this ecosystem to drought stress (as well as other potential threats and stressors). I also compare different plant functional types to investigate the variability of responses to water stress.

While this project is primarily aimed towards military lands due to the main land manager in my study area being the US military, its conclusions and the methods used are applicable more globally to any natural resources program that needs precise monitoring of riparian vegetation, land cover and flow. To assist natural resources managers in monitoring changes in habitat conditions and distribution, I use methods that can be scaled up for monitoring over large and remote areas, yielding results that provide information that is spatially and temporally explicit, presented through maps and time series. These methods can be adapted to the local management questions and for the specific habitats of interest. In this thesis, I develop metrics that give information on potential habitats for species of interest, ecosystem condition and climate change vulnerability. I work with freely available satellite imagery to map distribution and health of vegetation, and employ in situ conductivity sensors that provide useful information on streamflow presence within ephemeral streams.

The main research questions I aim to answer are:

- How are water and vegetation distributed over the study site (Fort Huachuca and San Pedro River), and what are the environmental controls on this distribution?

- How can a better understanding of environmental controls on vegetation health improve our understanding of ecosystems' resilience to climate change, and how can drought impact vegetation health and ecosystem condition?

- How can a better understanding of water availability to vegetation and flow permanence in ephemeral streams translate to monitoring and management recommendations for natural resources managers?

- How can long time series (Landsat) and high temporal resolution (Sentinel-2) be used together to map vegetation communities and keep track of inter and intra-annual variations in greenness to learn more about the external controls on vegetation distribution and health?

- How can conductivity sensors be deployed to measure flow permanence in non-perennial streams, and which metrics can be used to assess stream reach suitability for species conservation and climate change resilience?

1.2 Ecology of riparian vegetation in drylands

1.2.1 Drylands and non-perennial streams

1.2.1.1 Drylands

Drylands are defined as areas where evapotranspiration exceeds precipitation (Levick et al., 2008; Millennium Ecosystem Assessment (MEA), 2005). They are also characterized by low annual rainfall and high intra-annual variability (D’Odorico et al., 2019; Levick et al., 2008). Dry regions cover about 41 % of the land surface (MEA, 2005) and include a wide range of ecosystems, each with their own characteristics, flora and fauna (Figure 1.1). Dry areas usually have rain seasons, during which most of the rainfall occurs (D’Odorico et al., 2019).

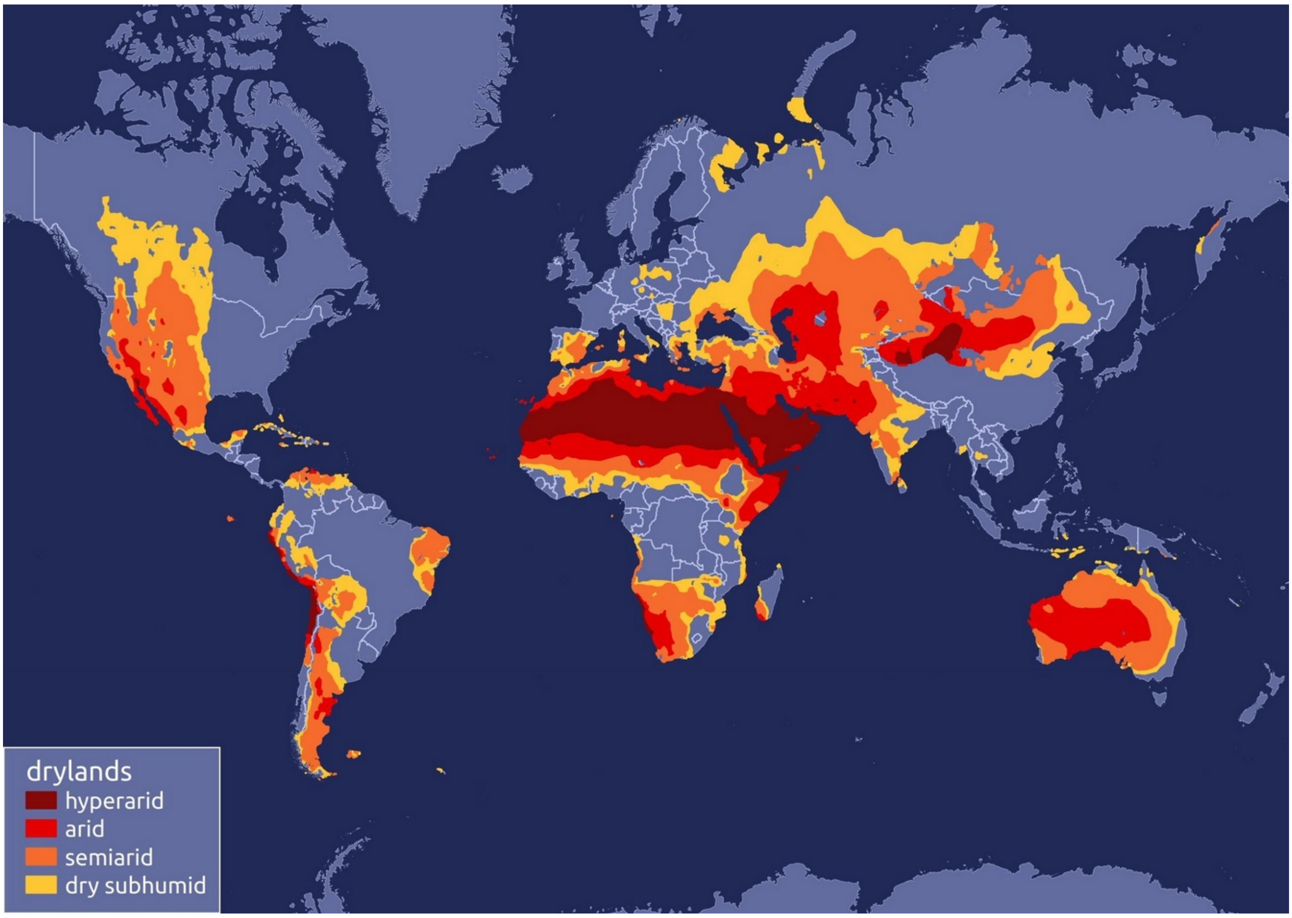

Figure 1.1: Map of world drylands, data produced for the Millennium Ecosystem Assessment (2005). Aridity zones were derived from an Aridity Index (AI) which is calculated as the ratio of precipitation (P) to potential evapotranspiration (PET). Zones included in this dataset are: hyperarid (P/PET < 0.05), arid (0.05 < P/PET < 0.2), semiarid (0.2 < P/PET < 0.5), and dry subhumid (0.5 < P/PET < 0.65) (UNEP-WCMC, 2007).

In drylands, there is a strong coupling between hydrological and ecological processes. Plant productivity is limited by water availability, which makes them sensitive to any changes in spatiotemporal water distribution and fluctuations (D’Odorico et al., 2019). Drought results in stressed vegetation with a lower canopy density, but vegetation is able to recover quickly as soon as a good rain season occurs (Gómez-Mendoza et al., 2008; Méndez-Barroso et al., 2009; Pennington & Collins, 2007). While dryland vegetation is mostly well adapted to water stress, some species are more vulnerable and might be living at the limit of their tolerance range. For these species, any decrease in rainfall distribution or amount will have consequences and drought can directly kill trees, or weaken them, making them more vulnerable to pest attacks (C. D. Allen et al., 2010; Breshears et al., 2005).

Vegetation phenology (the timing of life events, such as flowering, leaf-on or senescence) is driven by environmental controls, including soil moisture availability, temperature and length of day (Cleland et al., 2007; Smith et al., 2019). In drylands, phenology is under the influence of water availability, including rainfall timing and amount (D’Odorico et al., 2019; Smith et al., 2019). For grasses, both onset and end of the growing season are controlled by rainfall-recharged soil moisture, while temperature only has a modest effect on green-up (Currier & Sala, 2022). High precipitation in the season preceding green-up or senescence lead to an early green-up and a delayed senescence, but drought corresponded with a shorter growing season due to delayed green-up and early senescence. During the dry season, deciduous plants remain dormant or show a slowed increase in leaf density during leaf-on, while evergreen vegetation can reduce their growth rate (Littell et al., 2008). In some cases, a winter rain season can recharge the water table thanks to low evaporation (low temperatures and low intake by vegetation), and this water is then used during the day length or temperature-triggered growing season, even if that season is dry (D’Odorico et al., 2019). Several rain seasons a year can lead to several growth cycles for vegetation (Smith et al., 2019; Weiss et al., 2004).

Droughts are periods of prolonged decreased water availability (below long-term averages) which cause stress to ecosystems and societies. Droughts can be defined according to their consequences on surface water and groundwater (hydrological droughts; measured with streamflow and lake levels), precipitation (meteorological drought; measured with rainfall), vegetation (agricultural drought, measured with soil moisture) and society (societal drought) (D’Odorico et al., 2019; Van Loon, 2015). Several drought indices have been developed. For example, the U.S. Drought Monitor (https://droughtmonitor.unl.edu/CurrentMap.aspx) maps drought across the United States every week, relying on a panel of indices integrating soil moisture, streamflow, precipitation, temperature and other relevant metrics. These indices help define the extent and severity of the drought (Figure 1.2).

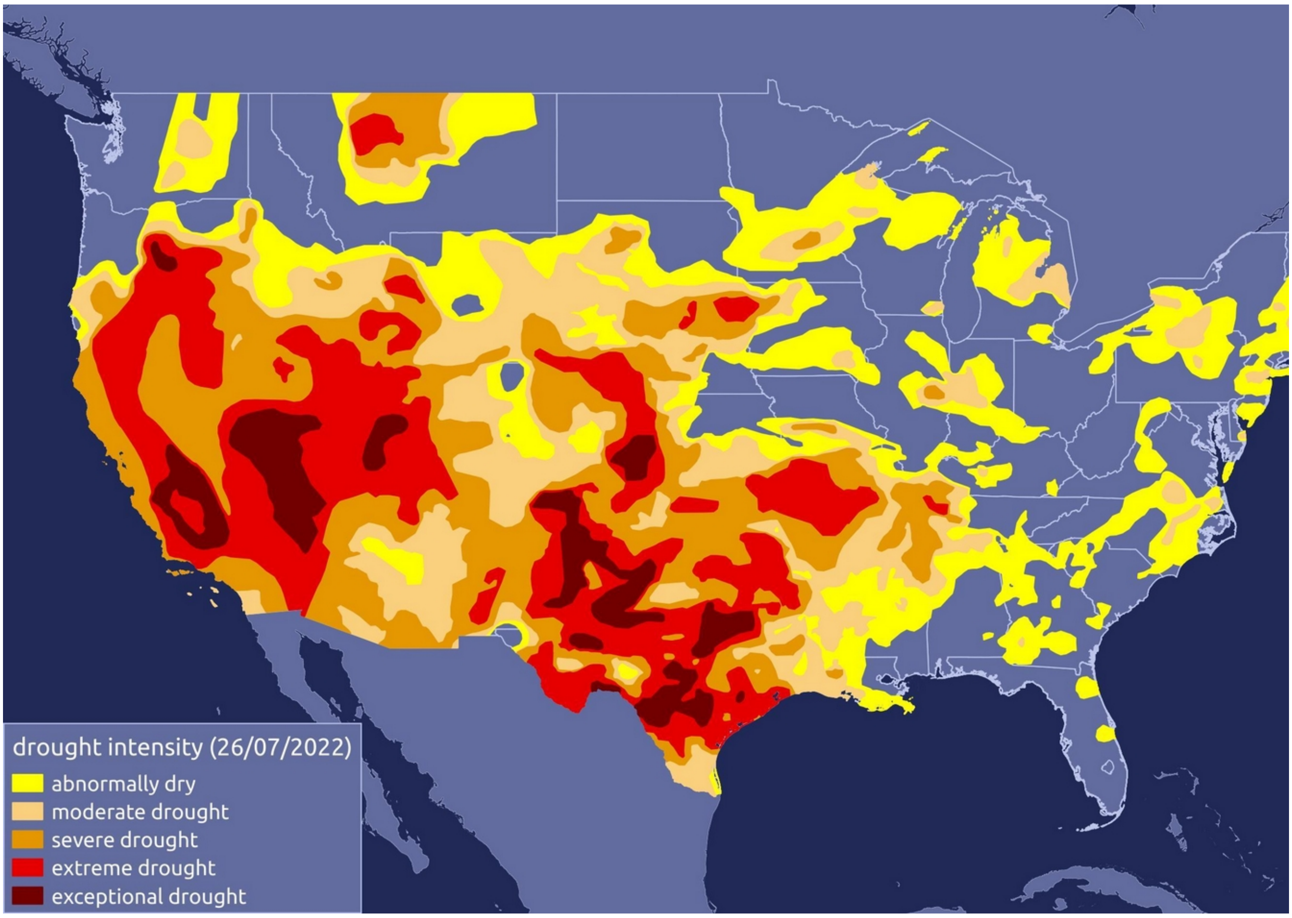

Figure 1.2: Drought monitor map for the conterminous U.S. produced on June 26, 2022.

In drylands, water is available through soil moisture, groundwater and limited surface water. Water is often stored and carried in temporary streams that periodically dry out, or the water tables that run beneath them (Levick et al., 2008; Messager et al., 2021a).

1.2.1.2 Non-perennial streams

Water from rainfall and snowmelt can either move into the ground as infiltration or stay at the surface and travel downhill as surface runoff. Runoff water gathers in streams and rivers, whose path is controlled by slope and topography. Water that infiltrated can also resurface in springs and add to streamflow. Streams can lose water through evaporation, intake from vegetation or infiltration losses. Anthropogenic actions can also alter streamflow amount and distribution. In areas where water gain is higher than losses, streams are perennial and flow all year round. But if losses are higher than gains, streams can dry out at some point in time and/or space (M. H. Busch et al., 2020; Levick et al., 2008).

Non-perennial streams are sorted further into ephemeral or intermittent depending on their connectivity with groundwater and responsiveness to rainfall. Ephemeral streams and rivers only flow in direct response to rainfall; they are losing streams. The water infiltrates into the sediment and percolates to a water table below the streambed (Levick et al., 2008; Quichimbo et al., 2020). Intermittent streams are defined by regular cycles of drying and wetting with water coming from melting snow, groundwater or springs. Depending on the time of year, they can be losing (water infiltration to the groundwater below the streambed) or gaining (the water table is above the streambed and provides flow) (M. H. Busch et al., 2020; Levick et al., 2008; Sophocleous, 2002). In this thesis, I will use the terms “non-perennial” and “intermittent” in reference to streams that dry out at least once every year.

Non-perennial streams can be spatially intermittent. They are characterized by an alternation of ephemeral, intermittent and perennial reaches (Figure 1.3) (Levick et al., 2008), with high variability of longitudinal (upstream-downstream) connectivity, but also lateral (streambed to floodplain) and vertical (surface to water table) exchanges of water, organisms, nutrients and energy (Nadeau & Rains, 2007). Even in dry reaches, water can keep travelling downstream in the hyporheic zone (underneath the stream) and can provide moisture to streamside vegetation before resurfacing downstream (Levick et al., 2008). Rapid and high variability of flow is also a characteristic of non-perennial streams in drylands (Figure 1.4). They can rapidly go from a dry streambed to flood (Levick et al., 2008). Floods bring water to the whole stream network, refill pools and restore longitudinal connectivity between perennial reaches or perennial ponds (Arthington & Balcombe, 2011; Levick et al., 2008). Floods also bring water to floodplains and river banks, which turns into soil moisture (Sophocleous, 2002). Large flood events can also temporarily change the spatial pattern of water availability by storing water in the stream banks and alluvial aquifer. The water stored is then slowly released over months and helps maintain the surface flow in a stream that would usually dry out and decrease the seasonal variability of the water table level. After the flood, ephemeral reaches dry out again and exchanges between the stream, the floodplain and the underlying water table are interrupted (Boulton et al., 2017). This high spatiotemporal variability in flow induced by the spatial alternation of perennial and intermittent reaches and the temporal alternation between a dry streambed and floods creates physical, chemical, and biological diversity at multiple scales (Boulton et al., 2017).

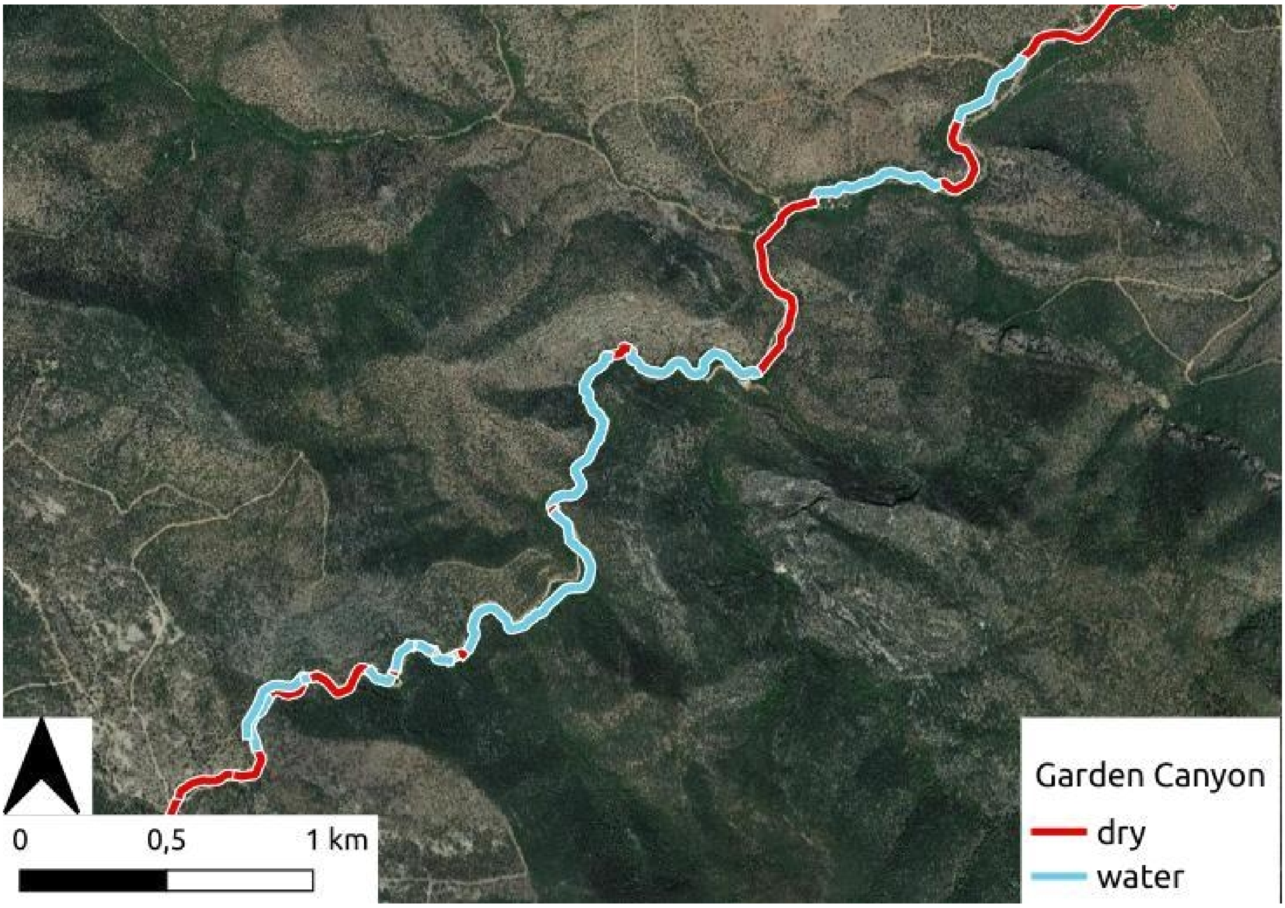

Figure 1.3: Spatial variability of flow: flow presence in Garden Canyon, mapped by walking down the streambed on the 19th of July 2019, at the end of the dry season. In red are the dry reaches and in blue are the reaches with surface water (flowing or isolated pools).

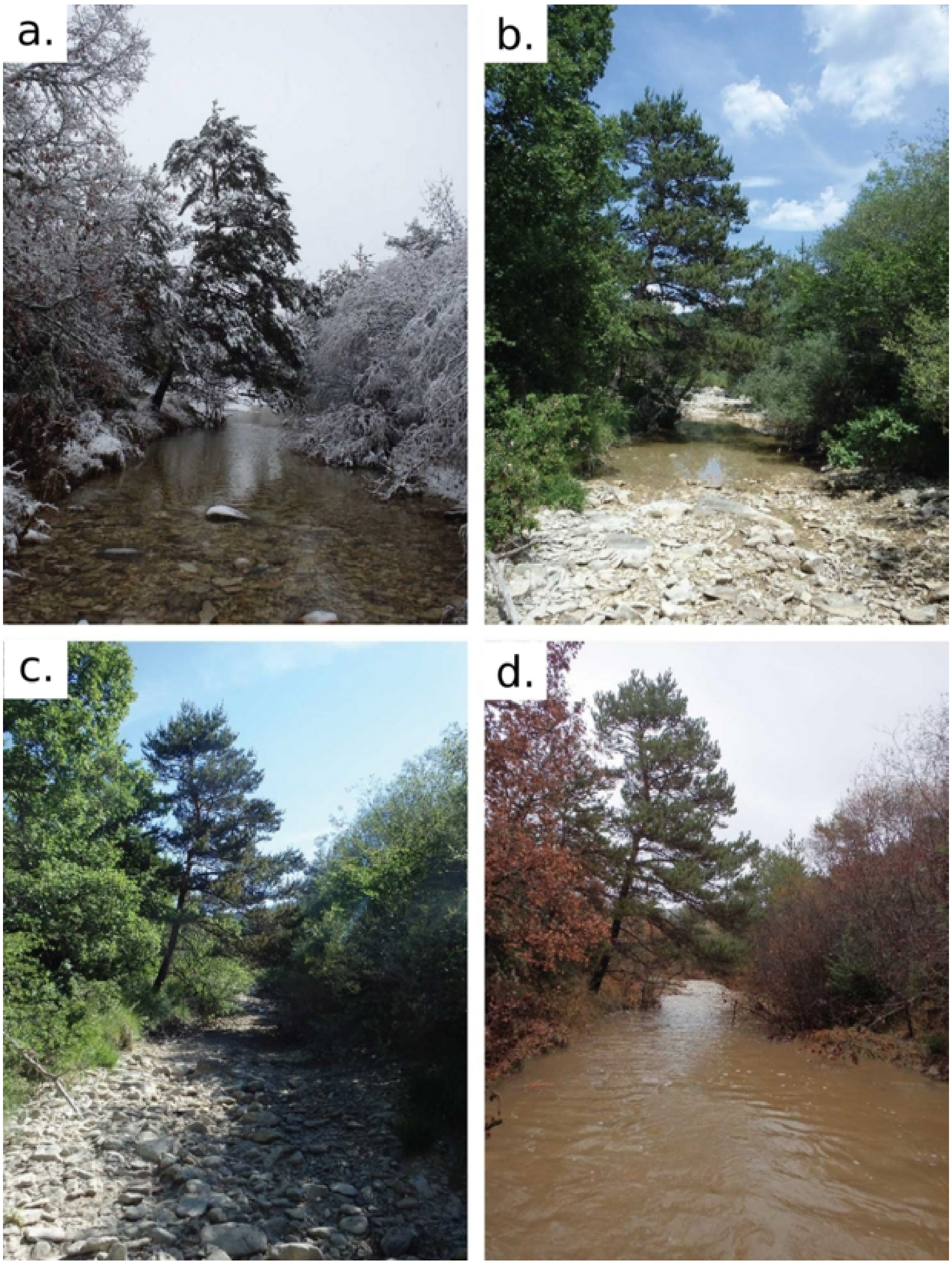

Figure 1.4: Temporal variability of flow: four different hydrological phases in the Calavon River, southeastern France: (a) flowing, (b) nonflowing, (c) dry, and (d) flooding (Datry et al., 2017). Photos: courtesy B. Launay.

Water in non-perennial streams can come from rainfall, snowmelt or springs, and water can travel downstream as overland flow (either from groundwater reaching the surface or from rainfall unable to infiltrate in the soil fast enough) or underground flow (either in the water table or in the permeable sediment of the streambed) (Levick et al., 2008; Nadeau & Rains, 2007). Streams dry out through infiltration of water in the sediment or through fractures, and evaporative demand (amount of water taken in the atmosphere through evaporation and vegetation transpiration, it depends of land cover, temperature, humidity, wind speed, and incoming solar radiation) that takes water from the stream to the atmosphere. Several environmental factors govern the presence of overland flow in the stream, and climate-induced aridity (balance between rainfall and evaporative demand) is considered an overarching key driver of flow permanence (Hammond et al., 2021; Messager et al., 2021a; Sauquet et al., 2021). Climate, geology, topography and land cover are also considered key controls on flow permanence (how often water is present in a stream reach) in intermittent streams (Costigan et al., 2016; Kennard et al., 2010). Topography, permeability of underlying geology, channel morphology and streambed composition regulates the speed of water infiltration in the water table and moisture of the streambed (Bourke et al., 2021; Levick et al., 2008; Sophocleous, 2002). Faults and fractures in weathered bedrock are where groundwater can reach the surface and springs can be found, providing flow to intermittent and perennial reaches (Lovill et al., 2018; Martin et al., 2021). Areas of high permeability act as sinks, and areas of low permeability keep water at the surface in pools and perennial reaches (Bourke et al., 2021; Lovill et al., 2018; Sophocleous, 2002). Land cover also controls water infiltration by being more or less impervious, and vegetation can pump enough water to temporarily dry a stream or alter stream morphology (Tabacchi et al., 2000). Land cover and streambed morphology can be altered by human activity, and flow can also be directly modified by pumping and flow diversion.

1.2.1.3 Biodiversity in non-perennial streams

In drylands, non-perennials streams support the majority of biodiversity and woody vegetation thanks to their high moisture content and shade provided by streamside vegetation (McDonald et al., 2004). High soil moisture and a shallow water table provides water for a dense and diversified vegetation, which in turn provides refuge and food for wildlife (Levick et al., 2008; Section 1.2.2). They often are the only available surface water during dry periods, which is needed by most mammals (Goodrich et al., 2018; Levick et al., 2008). Many invertebrates also need water for part of their life cycle, and are a food source for birds or bats (Levick et al., 2008; Sánchez-Montoya et al., 2017). These streams are an essential driver of biodiversity and ecosystem function (such as carbon and nutrient cycles) (Bogan et al., 2015; Datry et al., 2014; Larned et al., 2010; Leigh et al., 2016; Poff & Ward, 1989). The diversity of flow patterns along a stream provide a wide range of conditions for a mosaic of habitats (Datry et al., 2014; Larned et al., 2010). A single location can host various species (both for vegetation and wildlife) during different seasons depending on the variability of flow in time (Bogan et al., 2015; Jaeger & Olden, 2012; Snelder et al., 2013). The spatial and temporal variations in habitat patch distribution and composition lead to a high watershed-scale biodiversity, and can make them more diverse than perennial streams at longer time scales (annual and inter-annual) due to the turnover in species when local flow conditions change (Katz et al., 2012; Stromberg et al., 2005). Drying and flooding control the spatiotemporal distribution of flow, which impact the connectivity, extent and quality of habitat available for wildlife, especially for aquatic species (Datry et al., 2014; Jaeger et al., 2014).

A wide range of species depend on both the surface water and shallow groundwater linked to non-perennial streams. During phases of flood, ephemeral and intermittent streams move water, nutrients, organisms and sediments downstream (Acuña et al., 2014; Jaeger & Olden, 2012).

Connectivity is continuous through the watershed, which is useful for seed dispersal and movement of animals closely linked to moisture and water such as amphibians (Levick et al., 2008). Drought will induce habitat fragmentation and species relying on permanent pools, which might remain the only surface water available, will be isolated during the dry season with no means to move up or downstream (Bogan et al., 2015; Nadeau & Rains, 2007; Sheldon et al., 2010).

Even dry riverbeds host their own species, keeping soil moisture longer than the surrounding uplands and with riparian vegetation providing shade and protection, allowing animals to move easily between water holes (Levick et al., 2008; Steward et al., 2012). Some species are also found specifically in areas with high perturbation rates such as reaches with fast drying/wetting cycles (Datry et al., 2014). The drying cycles lead to habitat and population fragmentation and only species adapted to this fragmentation and drought stress are able to survive in ephemeral and intermittent streams (Boulton et al., 2017). Native species are adapted to the variable flow regimes, and intermittent streams offer protection against predators and invasive species that might alter flow regimes, water availability to native vegetation and reshape local ecosystems by replacing native species (Bogan, Boersma, et al., 2013; Goodrich et al., 2018; Le Maitre et al., 2015; Schmitt et al., 2019). Moody et al. (2017) studied riparian ground-dwelling arthropods and found that species assemblages in perennial, intermittent and ephemeral reaches had little overlap, and non-perennial reaches hosted more unique species not found in perennial reaches.

Non-perennial streams in drylands provide a cool and moist refuge for vegetation and wildlife. This cooler, shadier and wetter environment serves as shelter for species seeking refuge from thermal and moisture stress (Albright et al., 2017; Seavy et al., 2009; Stromberg et al., 1996) and is a rest stop for migrating animals (Mills et al., 1991; Patten, 1998). It also acts as habitat corridors along streams for the dispersal of plants and movement of animals, such as birds (Knopf & Samson, 1994). During the dry season, perennial pools might remain the only available water in the area (Levick et al., 2008). They host fishes and aquatic or semi-aquatic wildlife and allow recolonization throughout the watershed during floods (Figure 1.5) (Arthington & Balcombe, 2011).

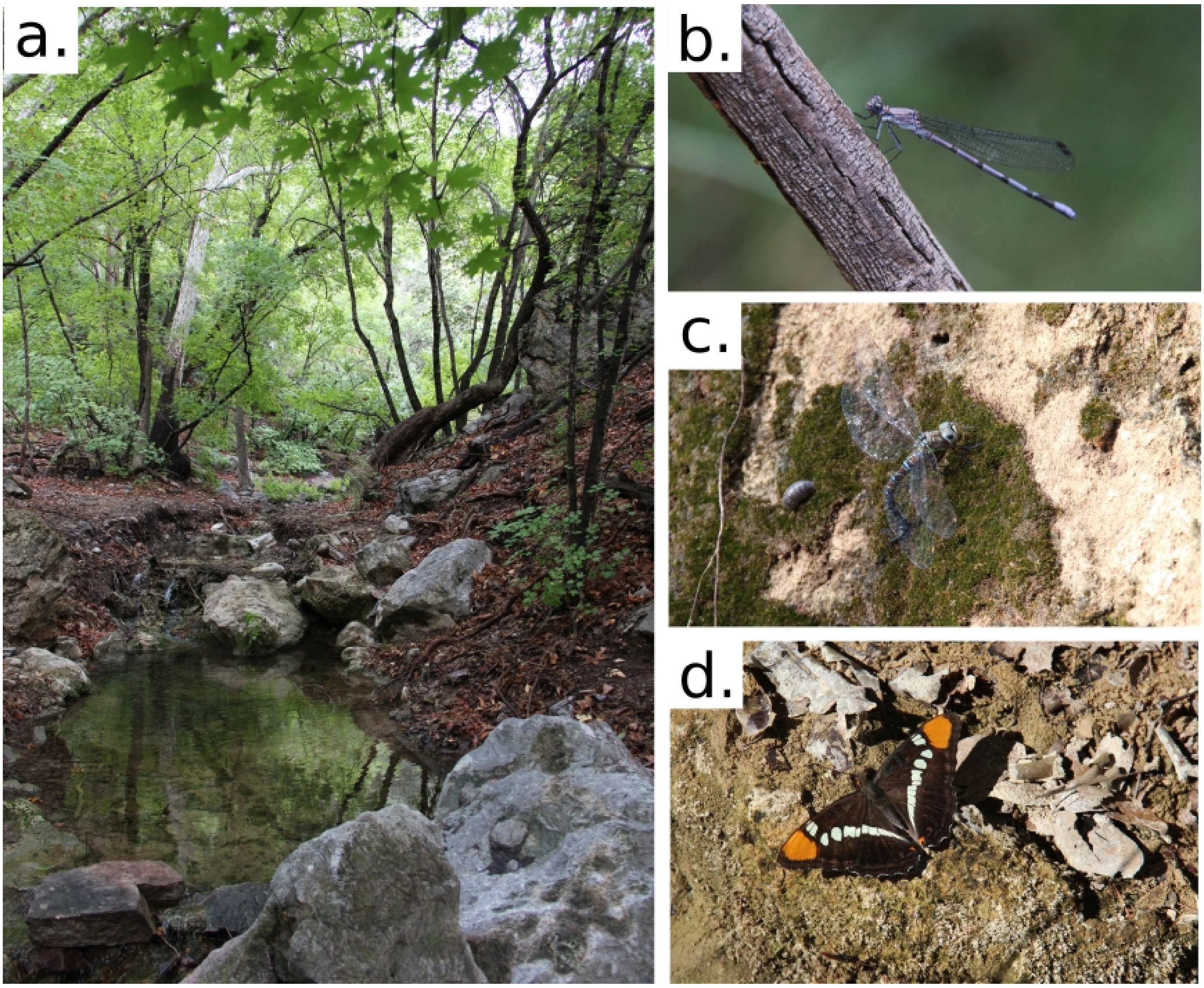

Figure 1.5: Refuge pool along Garden Canyon (a). Wildlife dependent on moisture found in ephemeral streams during the dry season: tonto dancer (*Argia tonto*) (b), riffle darner (*Oplonaeschna armata*) laying eggs on wet moss (c) and Arizona sister (*Adelpha eulalia*) on wet streambed sediment.

1.2.2 Riparian vegetation

1.2.2.1 Overview

Riparian zones are ecotones, a transition area between two biomes and communities. They are the interface between aquatic and terrestrial ecosystems where vegetation may be influenced by elevated water tables or extreme flooding and by the ability of the soils to hold water (Leenhouts et al., 2006; Naiman et al., 1993; Patten, 1998) and usually consist of specific plant communities along the banks of streams and water bodies of distinct composition from the surrounding uplands (Bren, 2014, p. 217; Naiman & Décamps, 1997; Sabater et al., 2017). Vegetation structure, composition, distribution and evolution within riparian corridors depends on water availability (Nilsson & Berggren, 2000; Sabater et al., 2017). However, riparian vegetation is also affected by soil properties, land surface topography, climatic forcing and disturbances (such as flooding, meander migration, fire or pest attacks), making them dynamic environments (Jones et al., 2008; Naiman & Décamps, 1997; Palmquist et al., 2018; Patten, 1998; Schilling et al., 2021).

The water sources to trees vary by their proximity to the stream, their rooting depth, floodplain topography, seasonal precipitation/evaporation and temporal variation in river flows and associated groundwater depth. Different species within the same forest stand will have different strategies to avoid competition and ensure their access to water (Singer et al., 2013). Some species might be very sensitive to even a small groundwater decline and to the switch from a perennial to an ephemeral stream (annual plants and vegetation growing directly in the stream, for example), which might lead to a sequential loss of species and diversity of the riparian zone, even if more resistant species remain (Nagler et al., 2021; Stromberg et al., 1996). Mature streamside trees may use several water sources such as groundwater, stream water and soil water, which vary in proportion over space and time, while streamside trees with shallow roots are more likely to use infiltrated rainfall as their main water source (Dawson & Ehleringer, 1991; Singer et al., 2014).

Riparian areas encompass a complex mosaic of habitats often following upstream-to-downstream or stream-to-upland gradients. This high diversity of habitats, over a range of scales (from the region to the watershed) is caused by a diversity of conditions (such as water availability or slope) and a high-frequency disturbance regime (including floods and river meandering) (Naiman & Décamps, 1997). This mosaic of potential habitats within riparian zones promotes high species richness (Moody & Sabo, 2017; Naiman et al., 1993; Patten, 1998; Schriever et al., 2015).

Riparian ecosystems are intensively studied as they provide critical ecosystem services. Roots provide bank stabilization and resistance to erosion (Easson & Yarbrough, 2002; Simon & Collison, 2002), vegetation and organic debris will slow down the flow during floods, retaining material in transport (Gregory et al., 1991; Tabacchi et al., 2000). Bank vegetation, trees most of all, also have an influence on the local micro-climate: under the canopy of trees, water is cooler, the amount of sunlight that reaches the stream is decreased and humidity is higher (Rykken et al., 2007; Trimmel et al., 2018; Wondzell et al., 2019). Another alteration to the stream by the riparian vegetation is by trapping nutrients, sediments and matter coming from upslope (Butterfield et al., 2020; Ensign & Mallin, 2001).

Riparian vegetation is also important when it comes to ecology and biodiversity. This mosaic of habitats provide food and shelter for terrestrial animals that use trees, forest understory or litter as well as aquatic species that favour submerged tree roots, fallen branches and debris (Gregory et al., 1991; Hamilton, 2008).

1.2.2.2 Riparian vegetation in drylands

In drylands, the variations in stream size, patterns of flow, and flood frequency and duration lead to spatial variations in the structure and composition of riparian vegetation (Levick et al., 2008; Pettit et al., 2001; Stromberg & Merritt, 2016). Species are adapted to environmental stressors, such as flooding and scouring, seasonal water shortage and cyclical droughts (Pettit et al., 2001; Stella, Rodríguez-González, et al., 2013). Reaches with less disturbance and more soil moisture are more favourable to obligate riparian species with denser canopy cover and a more diverse forest structure, while more disturbances and drier reaches are home to more drought-tolerant species adapted to drier environments, with less differences between streamside and upland vegetation and more low species such as grasses and shrubs, with sparse canopy (Figure 1.6d, e) (Leenhouts et al., 2006; J. R. Shaw & Cooper, 2008; Stromberg et al., 2007). Flood duration, frequency and amplitude are key elements to replenish soil moisture in the floodplain, are necessary for seedlings of some riparian species and control distribution of species (Friedman et al., 2006; Pettit et al., 2001; Stromberg, 2013). Flood pulses can lead to a temporary increase of annual species needing high soil moisture to germinate and able to do so thanks to the greater light reaching the ground, and the species diversity of intermittent reaches can temporarily surpass perennial reaches (Levick et al., 2008; Stromberg et al., 2008).

The close link between riparian forests and the specific hydrology of streams and surface water means that they are isolated and unevenly distributed in the landscape, and they are surrounded by a much drier environment hosting a completely different set of species (Malagnoux et al., 2007; Sabater et al., 2017). The contrast between streamside and upland vegetation can be very stark, with riparian vegetation being more dense and composed of completely different species thanks to access to soil moisture (Figure 1.6a-c) (Levick et al., 2008, 2015; Stromberg et al., 2015). This lateral gradient is mainly determined by soil moisture availability, depth to the water table and sediment deposition (Leenhouts et al., 2006; Lite & Stromberg, 2005; Patten, 1998; Shafroth et al., 2000; Stromberg et al., 1996). Riparian forests are very vulnerable to any change in climate and hydrology that affect root zone water availability because when the climate changes and the area is no longer favourable for the species, they cannot change their range to adapt, being surrounded by a land too dry for them (Malagnoux et al., 2007; Reidmiller et al., 2018).

Riparian forests cover a small area in drylands but support very high biodiversity (Levick et al., 2008; Powell & Steidl, 2015; Stromberg et al., 2005). The denser and more structurally complex vegetation of riparian forests is used by wildlife for foraging, nesting, as cover or as migration corridors and stopovers and act as a seed dispersal corridor (Levick et al., 2008). Birds are especially attracted to riparian forests in drylands thanks to the higher diversity in vegetation structure (being composed of grasses, shrubs and trees), which determines the distribution and availability of perching, foraging, and nesting sites (Levick et al., 2015). Their importance as migration corridors is due to the fact that they provide continuous chains of vegetation that wildlife can utilize for cover and food. They sustain water-dependent plants and animals that cannot survive in the dry uplands thanks to higher moisture and cooler soil, air and water temperatures thanks to vegetation cover providing shade and limiting transpiration (Lite & Stromberg, 2005; Nimmo et al., 2016).

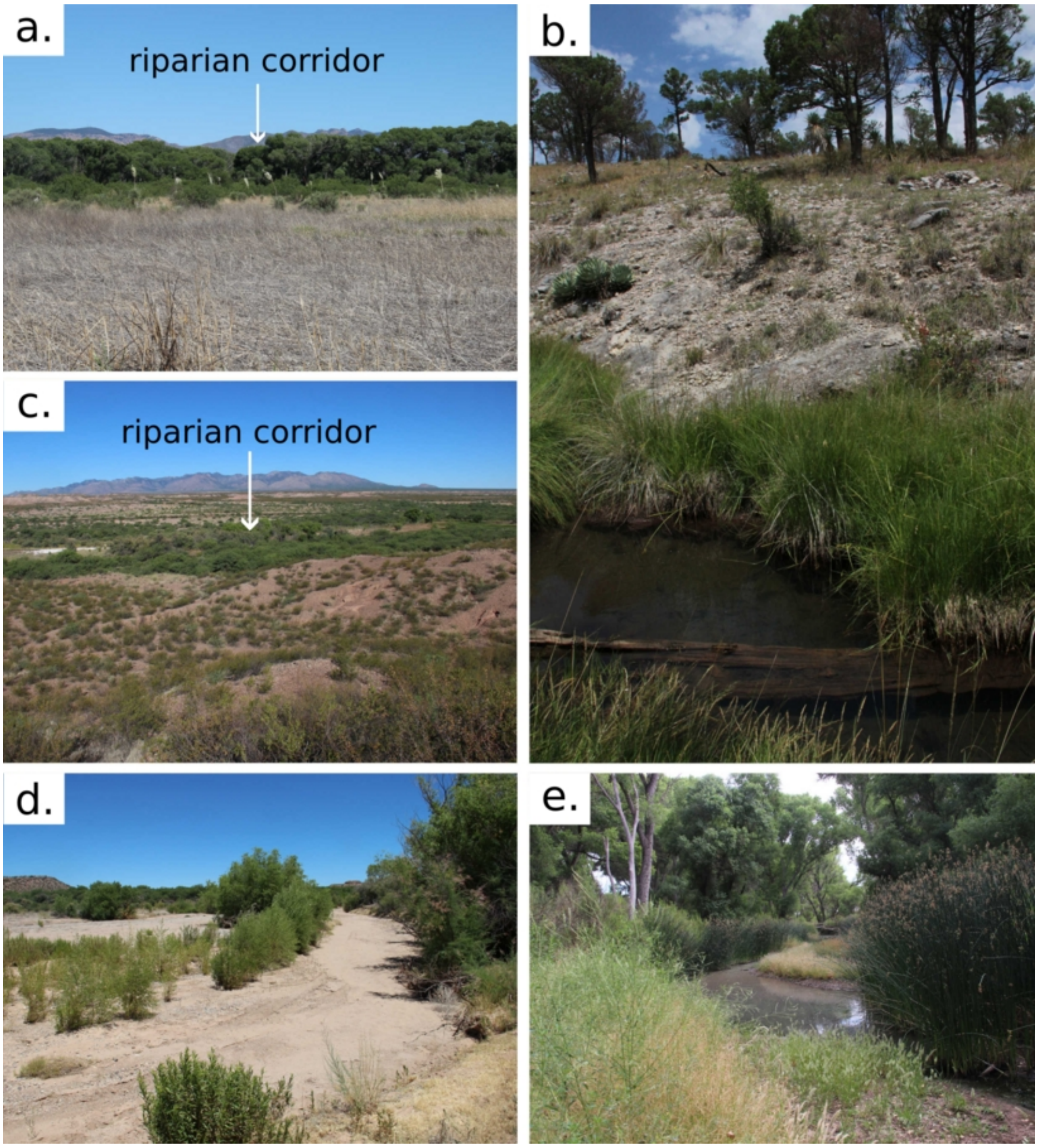

Figure 1.6: Contrast in vegetation density, structure and species between riparian and upland/arid vegetation (a, b, c), and between highly ephemeral and perennial stream reaches (d, e).

1.2.3 Tree-water interactions

Health, composition and distribution of vegetation are dependent on water availability in the rooting zone, as well as the processes that control the distribution of water in time and space. The soil-plant-atmosphere continuum is a key concept in studies of plant water use. Water moves from soil into roots, through stem and leaves and into the atmosphere, following a decreasing gradient of water potential (Cowan, 1965; Steudle, 2001). This transfer of water from soil to atmosphere through vegetation is called transpiration.

There are mechanisms and constraints regulating water uptake, transport and evaporation in roots, stem and leaves. The ability of roots to supply water to the plant depends on the hydraulic conductance of the root system, the distribution and depth of roots and mycorrhizal fungi within the soil profile, along with the ability to produce new roots as soil water is used and replenished (Ding et al., 2021; Passioura, 1988; Ward & Trimble, 2004, p. 85). Soil structure also plays a role in how much water is available and how easy it is to take up through roots (Passioura, 1991). Root distribution is species-specific but it seems that the bulk of the roots are located in the upper layers, where they can access nutrients from the litter and soil moisture coming from rainfall (Dhyani et al., 1990; February & Higgins, 2010), with plants being able to adapt their potential to remove water from the upper to the lower layers (Hainsworth & Aylmore, 1989). But the upper soil is also the first to dry out during periods without rain and is prone to stronger oscillations than deeper layers (Ehleringer & Dawson, 1992; L. Yang et al., 2012). Trees with dimorphic root systems can adapt to these patterns by using shallow lateral roots to get water from the upper soil layers during wet months and relying on deep tap roots for their water intake during dry spells (David et al., 2013; Dawson & Pate, 1996). When the roots of an individual plant are in contact with soils of spatially variable moisture content, they can be used for water redistribution in the soil, carrying water from saturated to depleted areas, following the soil water potential gradient. This process, which can occur horizontally as well as vertically, is called hydraulic lift or hydraulic redistribution (Burgess et al., 2001; Caldwell et al., 1998; Yu & D’Odorico, 2014). This delay in upper soil layers drying can provide water to the plants around the tree during droughts (Dawson, 1993) and preserve root function (Domec et al., 2004).

In the stem, water moves through hollow xylem cells that operate via cohesion-tension mechanics. This means of water transport is potentially vulnerable to disruptions by cavitation (interruption of the water column by a vapour-filled bubble) caused by water stress or freezing. This phenomenon blocks water transport in the affected cells (Tyree & Sperry, 1989). Xylem vulnerabilities vary among tree species growing under different ecological conditions (Cochard, 2006; Ogasa et al., 2013; Tyree & Sperry, 1989). Tyree et al. (1994) studied xylem vulnerability to loss of hydraulic conductivity for three cottonwood species and found them very vulnerable to drought-induced cavitation, with a loss of hydraulic conductivity in young stems that could lead to branch die-back (Nolan et al., 2021; Rood et al., 2000).

Water loss, as vapour, occurs through the leaf stomata, along with carbon absorption. Transpiration is regulated to ensure that water loss does not exceed the supply capacity (Benecke, 1980). The atmospheric drivers of transpiration include humidity, solar radiation and temperature: they control the air saturation in water and the energy gradient between the leaves and the atmosphere (Benecke, 1980; Schulze et al., 1972). Wind will also increase transpiration rates by dissipating the saturated layer of air close to the leaf. The ability of the tree to draw in water through its roots, water availability and hydraulic conductivity of the soil also play a role in transpiration (Benecke, 1980; Bréda et al., 2006; Wood et al., 2007, p. 9).

Plants control transpiration by regulating the size of their stomatal pores (Bond & Kavanagh, 1999; N. G. McDowell et al., 2002). Stomatal pore size responds rapidly to various environmental and physiological variables, including soil water deficit. As the soil dries, hydraulic (increase in xylem tension) or chemical signal can be generated and transmitted to the leaves to trigger stomatal closure. A change in the levels of abscisic acid, a growth regulator able to travel between the roots and the leaves, can act as a signal for changes in stomatal conductance (Hsu et al., 2021; Munemasa et al., 2015).

Drought stress within plants occurs when soil water drops below a threshold inducing restrictions to growth and transpiration. To maintain the integrity of their hydraulic system, trees will reduce their transpiration by closing their stomata, which leads to a decrease in carbon and nutrient assimilation and induces limited tree growth (Benecke, 1980; N. McDowell et al., 2008). If the drought persists, the tree might need to use its reserves and shed its leaves as a means to reduce transpiration. Cavitation reduces water delivery to the canopy and causes branch death and leaf loss (Choat et al., 2018). Carbon starvation (carbon absorption and storage failing to meet consumption) and hydraulic failure due to cavitation are the main cause of tree death during droughts (N. McDowell et al., 2008; N. G. McDowell, 2011) As water is also the carrier of nutrients and minerals, water stress disturbs nutrient availability, transport, and partitioning through water deficit (Hu & Schmidhalter, 2005). Drought will also have consequences on the population as a whole by limiting bud production and will increase the tree’s vulnerability to insect damages, diseases, competition, fire or other droughts (Bréda et al., 2006; N. McDowell et al., 2008). Species have developed different responses to droughts, from drought avoidance (isohydry) to drought tolerance (anisohydry). Isohydric vegetation close their stomata early to minimize transpiration and protect xylem tissue from cavitation, but this interrupts carbon absorption, which can lead to carbon starvation as respiration continues. Anisohydric vegetation maintains its stomata open and does not interrupt transpiration, allowing it to keep absorbing carbon, albeit with a higher risk of cavitation (C. D. Allen et al., 2010; N. McDowell et al., 2008).

1.2.4 Climate change

By the end of the century, anthropogenic climate change and its consequences may be the dominant direct driver of biodiversity loss and changes in ecosystem services globally (Millennium Ecosystem Assessment, 2005). By changing temperature extremes and rainfall intensity and distribution, climate change affects water distribution and availability to vegetation across the world. The main consequences of these changes on vegetation are shifts in geographic ranges and timing of key life events, as well as increased risk of extinction for some species.

1.2.4.1 Climate and water distribution

We are currently going through an unprecedented warming of the global climate system of anthropogenic causes. The increase of greenhouse gas emissions has lead to the warming of air and oceans, an increase in snow and ice melt, and sea level rise. Global average temperature shows a warming of 1.09 °C from the period 1850–1900 to 2011–2020 (Intergovernmental Panel on Climate Change (IPCC), 2022). This global warming leads to longer and more frequent droughts, and is also paired with a higher frequency and severity of extreme weather events such as hurricanes and an increase in rainfall variability (IPCC, 2022). Due to the many local factors influencing small-scale climate, the consequences of climate change are variable locally. The highest temperature increase is projected to be over the Arctic Ocean (Figure 1.7a). Shifts in precipitation are expected to occur with increasing temperatures, but the direction and intensity of these shifts will vary regionally (Ault et al., 2016; Dwire et al., 2018), with some areas receiving more precipitation than during a

1850-1900 baseline, while other regions will receive 30% less rainfall than the baseline and will see an increase in consecutive dry days (Figure 1.7b, c). Current projections indicate that climate will continue to change over the 21st century (IPCC, 2022).

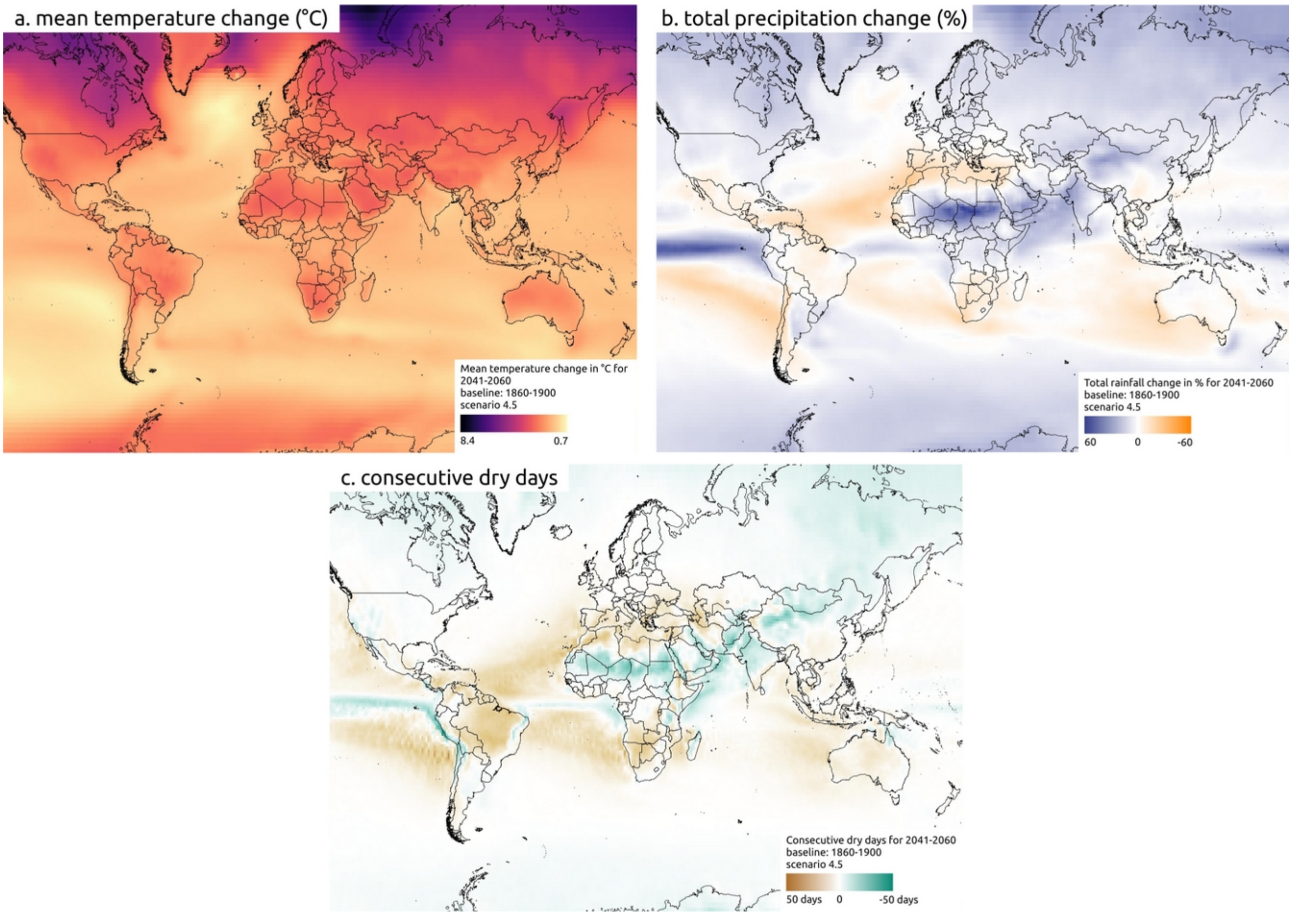

Figure 1.7: Model simulation for change in annual mean temperature, in °C (a), total precipitation, in % (b) and consecutive dry days, in number of days for the period 2041-2060 compared to the baseline 1860-1900 (“pre-industrial” period). The model is CMIP6 (Coupled Model Intercomparison Project 6) and the climate change scenario is SSP2-4.5 (CO_2 emissions around current levels until 2050, estimated global warming of 2°C for the period 2041-2060)-. Data from the IPCC interactive atlas: https://interactive-atlas.ipcc.ch/.

At the Earth’s surface, modifications of rainfall and temperature lead to changes in water distribution in space and time. Globally, climate warming is anticipated to increase the frequency and severity of droughts (Samaniego et al., 2018). Areas with less total annual rainfall, clustered in less frequent but more violent events might experience droughts followed by floods. A reduced snowpack can lead to lower streamflow and groundwater levels, and an increase in temperature can induce an early snow melt, altering the temporal distribution of water (IPCC, 2022). Groundwater can act as a buffer to climate change impacts on water resources in many regions due to long recharge and travel time underground (Cuthbert et al., 2019), but this will also translate to delayed impacts on streamflow and groundwater-dependent ecosystems that are harder to plan for.

Non-perennial streams depend on groundwater discharge, rainfall and snowmelt to maintain surface flow. An increase in dry days (Figure 1.7c) will have a consequence, especially on ephemeral streams that only flow in response to rainfall. Stronger precipitation events will also result in larger floods. In the conterminous USA (continental USA without Alaska), Zipper et al. (2021) looked at metrics of flow permanence (number of days with no flow, dry-down period and day of the first dry-out event) in non-perennial streams across the country and half of the studied streamflow gauges showed a significant trend through time in at least one metric, mostly shifting towards drier conditions. Longer dry periods were usually due to an earlier date for the first drying of the stream controlled by aridity (ratio between precipitation and evaporative demand). These authors found that changes in date and length of dry streambeds were organized in distinct regional patterns. The south was dominated by drying while streams in the north were mostly getting wetter.

1.2.4.2 Consequences of climate change on vegetation

Climate change has already impacted and will keep impacting vegetation, wildlife and humans. Temperature is increasing, and the amount and distribution of available water is changing and causing vegetation to shift their geographic range toward colder areas (at higher elevations for example) to remain within their climate tolerance (Dwire et al., 2018; Visser & Both, 2005). Kelly and Goulden (2008) found that dominant plant species climbed 65 m between surveys done in 1977 and 2007 along an elevation gradient in Southern California, where temperature and precipitation variability increased, while the amount of snow decreased.

Species that cannot shift their range, either due to an inability to shift their range fast enough or because they have no new land to move to, face physiological stress and an increased risk of extinction, especially when climate change is paired with other stressors, such as habitat loss due to human impacts or invasive species (C. D. Allen et al., 2010; IPCC, 2022; Walther et al., 2002). Decrease in annual rainfall is projected to increase the frequency of drought-induced tree mortality (Choat et al., 2018; Parolari et al., 2014). Physiological stress caused by drought can result in tree death through xylem cavitation (Section 1.2.3), or through stomatal closure causing carbon starvation and a weakened tree vulnerable to extreme climatic events and pest, with pest density potentially increased by the higher temperatures (Jump & Peñuelas, 2005; N. McDowell et al., 2008). As species can be more or less tolerant to climate change, and can shift their geographic range faster or slower, species assemblages structuring ecosystems, species interactions and ecosystem dynamics are at risk because of shifts in species composition (IPCC, 2022; Walther et al., 2002). Increase in CO_2 concentration in the atmosphere can alleviate the stress caused by drought by increasing the water use efficiency of plants (more CO_2 absorbed by the stomata) (C. D. Allen et al., 2010; Tietjen et al., 2009) but large-scale tree death triggered by drought has been observed, for example in the Southwestern USA by Breshears et al. (2005) and Goulden and Bales (2019).

Water availability and temperature are also key phenological controls. Increase in temperature, especially during the spring, can lead to an early green-up allowing a longer growing season for vegetation, early breeding season for birds or amphibians or early arrival of migrant birds for example (Cleland et al., 2007; Walther et al., 2002). Parmesan and Yohe (2003) documented a significant mean advancement of spring phenology by 2.3 days per decade, and Root et al. (2003) gathered information from 143 studies and established that spring phenology occurs on average 5.1 days earlier per decade for species showing changes in phenology, with a range going from 24 days earlier to 6.3 days later. The end of the growing season, in autumn, is also changing but does not seem to show a clear trend (Cleland et al., 2007). Phenology cycles of species are often synced to events important to their survival. Several riparian trees seed dispersal coincide with flood timing, necessary for dispersal and establishment of seedlings (Stella et al., 2006). With changes in rainfall distribution and storm intensity, shifts in flood timing could mean that riparian tree seedlings no longer have access to ideal conditions. Species often depend on the resources produced by another. If offsets in phenological events between species relying on each other do not follow the same trends and rate of change, organisms risks missing on important resources such as food or shelter during key moments, including migration or reproduction (Kellermann & van Riper, 2015; Kharouba et al., 2018; Visser & Gienapp, 2019). For example, species who synchronize egg-hatching season with peak availability in food resources have been shown to lag behind as warm temperatures are reached earlier in the season, as shown in great tits (*Parus major*) (Visser et al., 1998) and winter moth (*Operophtera brumata*) (Visser & Holleman, 2001). Variations in climatic conditions can also impact inter-species competition (Stenseth et al., 2015).

1.2.4.3 Impacts of climate change on drylands vegetation

Drylands have already been affected by climate change, with areas experiencing increases in aridity, though there is no global trend for changes in vegetation density (Mirzabaev et al., 2022). Some drylands are increasing and other are shrinking as rainfall changes are uneven worldwide (Mirzabaev et al., 2022) (Figure 1.7b). Several drylands are seeing a change in vegetation distribution, with grasslands being overtaken by shrubs due to climate change paired with fire suppression and land use change (Mirzabaev et al., 2022; Munson et al., 2011). Dryland vegetation is adapted to low and highly variable water availability, but some species are already at the edge of the geographic range, or located in isolated areas of higher and steadier water availability (such as mountain peaks and riparian corridors) and are more vulnerable to an increase in aridity, especially since they are often already at risk due to cattle grazing and water pumping (Dwire et al., 2018).

Along streams, loss of streamflow and streamside soil moisture will trigger shifts in vegetation and communities composition will be closer to highly ephemeral streams and uplands (Section 1.2.2.). More perennial reaches are supported by groundwater and any changes in regional rainfall that can affect groundwater levels are a risk for riparian vegetation and wildlife (Dwire et al., 2018). Since the high biodiversity of these streams is due to the high spatiotemporal variability of flow across the landscape, any change in water distribution, due to climate change or human intervention, leading toward uniform flow or complete drought, will lead to a reduced species richness. Jaeger et al. (2014) modelled streamflow responses to forecasted climate change and the resulting flow connectivity in the Verde River Basin (Arizona, USA). Model simulations project an increase in zero-flow days in a year, in frequency of drying events and in duration of these events. Perennial reaches will be shorter and further away from each other, isolating species relying on perennial water in smaller refuges.

1.3 Management of military lands

This project is a case study in applied research to support natural resources management. A key land manager of the Huachuca Mountains and San Pedro valley is Fort Huachuca, a US Army installation. Since most of my research was conducted on Fort Huachuca (notably Chapters 3 and 4), this thesis' conclusions and management recommendations are turned toward military lands management, although they can applied to a wide diversity of organizations and situations.

1.3.1 Natural resources management

Natural resources management and conservation are essential considerations for land owners. Maintaining or restoring ecosystems allows for the protection of key resources (such as water), protection against disasters, and safekeeping of diverse ecosystems supporting high biodiversity, including endangered species under protection. It can be the main mission of land managers, for example in the case of national parks, or a secondary duty, for example on military lands. Even when conservation is not the main purpose of a land owner, it is necessary for maintaining ecosystem services provided and complying with legislation.

The United States Department of Defense (DoD) lands cover approximately 109 000 km^2 of land across a wide diversity of landscapes and natural habitats (Ripley et al., 2021; Stein et al., 2008). The military's need for natural landscapes used for realistic training and acting as buffers between the installation and its surroundings makes military lands important refuges for functioning ecosystems and habitats. Some installations have become the last refuges of endangered species habitat in rapidly urbanizing landscapes (NatureServe, 2004). As such, they are responsible for natural resources management of these lands, both to follow current regulations and to fulfil the military mission.

1.3.1.1 Need for natural resources management

The goal of natural resources management and biodiversity conservation on military installations is to maintain and improve the sustainability and biological diversity of ecosystems while supporting human use, and the military mission. Natural landscapes and wilderness constitute a high-quality environment for in-situation training and testing. Maintaining healthy ecosystems is an essential component of maintaining military readiness and ensuring that the installation fulfils its role in the long term (DoD, 2014; Ripley et al., 2021). Good stewardship of natural resources and maintaining functional ecosystems is also critical for mission sustainability, ensuring the region can properly support the people living both on the installation and in its surroundings. The military has to play its part in the conservation of resources shared with the surrounding population, such as groundwater, and management of risks, such as fires, floods and erosion (Li & Male, 2020). For example water conservation is a region-wide issue in the southwest USA, and military installations have to take this issue into account in their natural resources management plan (Section 1.4.5). Nature areas are also important for installation personnel well being, they provide opportunities for outdoor recreation, including hiking, fishing or hunting (Ripley et al., 2021). Several natural resource conservation regulations at all levels of government (federal, state) are also applicable to military lands.

The DoD formally established a policy for an ecosystem approach to natural resources management and for the conservation of biological diversity. The goal is to maintain or restore native ecosystem and populations integrity and enhance their resilience. The ecosystem approach is focused on a multi-species conservation planning at the landscape scale, with the goal of sustaining or restoring key ecological processes (such as disturbance regimes, hydrological processes, and nutrient cycles) and habitat connectivity (Ripley et al., 2021). Proactive management is the method of taking action preemptively before an issue arises. It can support more efficient measures and prevent losses that might need more costly and complex interventions (for example, it is safer and more efficient to organize prescribed burns than needing to extinguish an uncontrolled fire). It is also a means to take action in advance of environmental impact assessment processes under the National Environmental Policy Act as well as consultation processes under the Endangered Species Act and potentially make the approval of future actions easier (Ripley et al., 2021). Natural resources management focused on whole ecosystems is a data-intensive approach that needs adaptive long-term monitoring to assess ecosystem dynamics and improve conservation actions through a continuous feedback cycle. Monitoring is the collection and analysis of repeated observations, or measurements, compared against a baseline and organized around a conservation goal. It allows conservation managers to follow changes to the ecosystem in response to change, such as monitoring practices or climate change, by keeping track of key metrics such as population numbers or flow permanence in intermittent streams. The set of tools used in monitoring helps identify problems early, such as population crash, decrease in streamflow or exotic species invasion, before it is too late or very costly to act.

DoD installations collaborate with natural resources agencies, mainly the U.S. Fish and Wildlife Service and state fish and game agencies. They are also establishing partnerships to develop conservation actions beyond installation boundaries. There is an important focus on protecting buffer lands (areas around the installation) from development to reduce the risk of disruption. As weapon systems become bigger and louder, buffers are needed to shield urban areas from disturbances, and to isolate training areas from external disruptions. For example, lighting from cities impedes nighttime training and residential development near installations results in noise and smoke restrictions (Li &

Male, 2020). Cooperative agreements are being established with federal organizations, states, local governments, nongovernmental organizations, and individuals to help maintain and improve off-base habitat and to ease encroachment problems around military installations. Installations can provide expertise and funds for management actions on other protected lands, and buy land with the goal of maintaining or restoring high quality habitats (Li & Male, 2020).

1.3.1.2 Regulations, funding and supporting programs

As federal land managers, DoD installations must comply with several natural resources conservation and protected species federal laws, including the Endangered Species Act (ESA), Clean Water Act, Migratory Bird Treaty Act, Marine Mammal Protection Act, Bald and Golden Eagle Protection Act or National Environmental Policy Act (Table 1.1) (Li & Male, 2020; Ripley et al., 2021). In addition to these federal laws, many states have enacted their own endangered species and wildlife protection laws. The main text regulating natural resources management specifically on DoD lands is the Sikes Act, passed in 1960. It has been amended several times, and currently provides the legal basis for natural resources conservation and public access for recreation on military land, authorize the development of cooperative plans by the military, the U.S. Fish and Wildlife Service, and state fish and game agencies (Li & Male, 2020). It includes ESA-listed threatened and endangered species (T&E species) and requires that installations employ professionally trained natural resource professionals. Each of the military branches (Army, Navy, Air Force and Marines Crops being the main ones) has incorporated policies regarding ecosystem management and biodiversity conservation into their natural resources directives, and each installation has its own set of local regulations.

Table 1.1: Federal legislation for protected species (Ripley et al., 2021).

Law	**Administering agencies**	**Covered species**	**Key protections**
Endangered Species Act of 1973	U.S. Fish and Wildlife Service, National Oceanic and Atmospheric Administration (NOAA) Fisheries Service	Listed Endangered and Threatened species. Species listed under the ESA include vertebrate animals, invertebrate animals, and plants	Prohibits the take of listed species, which includes to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect. Prohibits federal actions that are likely to jeopardize the continued existence of listed species or adversely modify designated critical habitats.
Marine Mammal Protection Act of 1972	NOAA Fisheries Service, U.S. Fish and Wildlife Service, U.S. Marine Mammal Commission	Whales, dolphins, porpoises, seals, sea lions, walrus, polar bears, sea and marine otters, dugongs, and manatees	Prohibits the take of marine mammals, which includes harassment, hunting, capturing, collecting, or killing.
Migratory Bird Treaty Act	U.S. Fish and Wildlife Service	Migratory bird species that naturally occur in the United States or U.S. territories	Prohibits the take of protected migratory bird species, which includes killing, capturing, selling, trading, and transport.

Law	**Administering agencies**	**Covered species**	**Key protections**
Bald and Golden Eagle Protection Act	U.S. Fish and Wildlife Service	Bald eagles and golden eagles	Prohibits the take of bald or golden eagles, including their parts, nests, or eggs, which includes to possess, sell, purchase, barter, export or import.

The Sikes Act requires the development and implementation of Integrated Natural Resources Management Plans (INRMPs) for all U.S. installations with significant natural resources, which are required on about 340 installations. The INRMP outlines the specific natural resource management goals for an installation, and the steps for achieving those goals. It also defines public access to the installation, outdoor recreation, and integrates other information and plans such as installation master plans and range and training land plans; recreation; natural resources compliance commitments; and partnerships. It is cooperatively developed with the U.S. Fish and Wildlife Service, and the relevant state wildlife agencies. The INRMP is reviewed at least every five years and is subject to public review and comment (Stein et al., 2019).

Many installations attract funding from numerous sources. DoD funds, as administered through the military services, are the primary sources of conservation funding. Research and conservation innovations are supported by several DoD environmental funding programs, including the Legacy Resource Management Program, Readiness and Environmental Protection Initiative (REPI), Strategic Research and Development Program (SERDP), and Environmental Security Technology Certification Program (ESTCP) (Li & Male, 2020). The REPI program’s goal is to facilitate a landscape-scale management perspective in and around DoD lands, mainly for combating encroachment that can impact military training, testing, and operations by facilitating buffer partnerships among the military services, private conservation groups, and state and local governments (Li & Male, 2020). State and federal conservation programs can also be an important source of funding (Ripley et al., 2021).

1.3.1.3 Threatened and endangered species

The first endangered species protection act was adopted by Congress in 1966, and later replaced by the more expansive Endangered Species Act (ESA) of 1973. The purpose of the law is to avoid the loss of species and encourage recovery (Stein et al., 2008). The U.S. Fish and Wildlife Service is in charge of assessing the condition of plants and wildlife to identify those species considered endangered, defined in the Act as “an animal or plant species in danger of extinction throughout all or a significant portion of its range,” and those considered threatened, defined as “an animal or plant species likely to become endangered within the foreseeable future throughout all or a significant portion of its range.” As of November 2022, there are 1269 endangered and 402 threatened species in the U.S. for a total of 1671 species, including 942 plants and 732 animals (U.S. Fish and Wildlife Service, 2022). There are also candidate species, which are under consideration for official listing. Critical habitats are designated, which are considered essential for the conservation of T&E species. The law requires federal agencies, in consultation with the U.S. Fish and Wildlife Service and/or the National Oceanic and Atmospheric Administration (NOAA) Fisheries Service, to ensure that actions they authorize, fund, or carry out are not likely to jeopardize the existence of any listed species or result in the destruction or adverse modification of designated critical habitat of such species (U.S. Environmental Protection Agency, 2013). The law also prohibits any action that causes the taking (defined as pursue, hunt, shoot, wound, kill, trap, capture, or collect) of any listed species (Ripley et al., 2021). The ESA also aims to provide a means for conserving the ecosystems that T&E species depend upon for survival and recovery.

With the highest density of T&E and imperilled species of any other federal agency and hosting 487 T&E species and 550 at-risk species, DoD installations must comply with the ESA (Li & Male, 2020; Stein et al., 2008). Their T&E species are monitored and actions are taken to maintain populations. If an installation has to take an action that might affect a listed species, it must send a “biological assessment” to the U.S. Fish and Wildlife Service or the NOAA. If the proposed actions are considered a risk of extinction for a listed species, the actions can be vetoed. In 2004, an amendment to the ESA exempted the DoD from critical habitat designations so long as a comprehensive and approved INRMP is in effect and specifically address the conservation of the listed species (Ripley et al., 2021). The INRMP Implementation Manual specify that the DoD must consult with the U.S. Fish and Wildlife Service and NOAA Fisheries when T&E species or designated critical habitats are in question, to ensure no action will jeopardize the existence of listed species, or destroy or modify designated critical habitats. The Secretary of the Interior is also allowed to make critical habitat exclusions for economic or national security considerations.

DoD-specific directives have also been established to maintain T&E species on installations. On top of management for T&E species, a high priority of natural resources management is the proactive conservation of at-risk species (candidate and proposed species for listing under the ESA and imperilled or critically imperilled species according to the NatureServe conservation status rank; https://www.natureserve.org/conservation-status-assessment) on and around DoD installations to prevent the need for federal listing under the ESA (NatureServe, 2004). There is also a will to restore current T&E species populations to achieve recovery goals and ultimately delist these species (Ripley et al., 2021). Every year, the DoD and other federal agencies have to report their spending related to T&E species conservation. In 2018,

$72,821,686 were reported to the U.S. Fish and Wildlife Service (Table 1.2) (U.S. Fish and Wildlife Service, 2018).

Table 1.2: Expenses reported by the DoD for T&E species management in 2018. “Species total” is for direct species conservation and “land total” is for land acquisitions (U.S. Fish and Wildlife Service, 2018).

DoD branch	**Species total**	**Land total**
Army	$41,263,489	$0
Marine Corps	$13,407,400	$0
Navy	$18,150,797	$0
Air Force	No expenses submitted to the U.S. Fish and Wildlife Service for 2018	

1.3.2 Climate change integration

1.3.2.1 Risks

Climate change and its consequences are an important concern for the DoD as they pose a threat to their mission and operations (Li & Male, 2020). DoD lands are considered mainly vulnerable to rising global temperatures, changes in precipitation patterns, the increase in frequency or intensity of extreme weather events and rising sea levels ((DoD, 2014). The associated risks identified for installations are recurrent flooding, drought, desertification, wildfires, and thawing permafrost (DoD, 2019). A 2019 DoD report assessed that about two-thirds out of 79 priority installations are vulnerable to current or future flooding, more than one-half are vulnerable to drought and one-half are vulnerable to wildfires (DoD, 2019). The Climate-Related Risk to DoD Infrastructure Initial Vulnerability Assessment Survey report (DoD, 2018) concludes that half of the DoD installations are vulnerable to extreme weather events (flooding, extreme temperatures, wind, drought and wildfire) (Figure 1.8), with drought being the risk reported most often. Airfield operations, transportation infrastructure, energy infrastructure, training/range facilities, and water/wastewater systems were considered the most impacted. Climate change will have cascading effects on the landscape, with shifts in vegetation and animal species, ecosystem structure and in processes, such as water and nutrient cycling (Section 1.2.4).

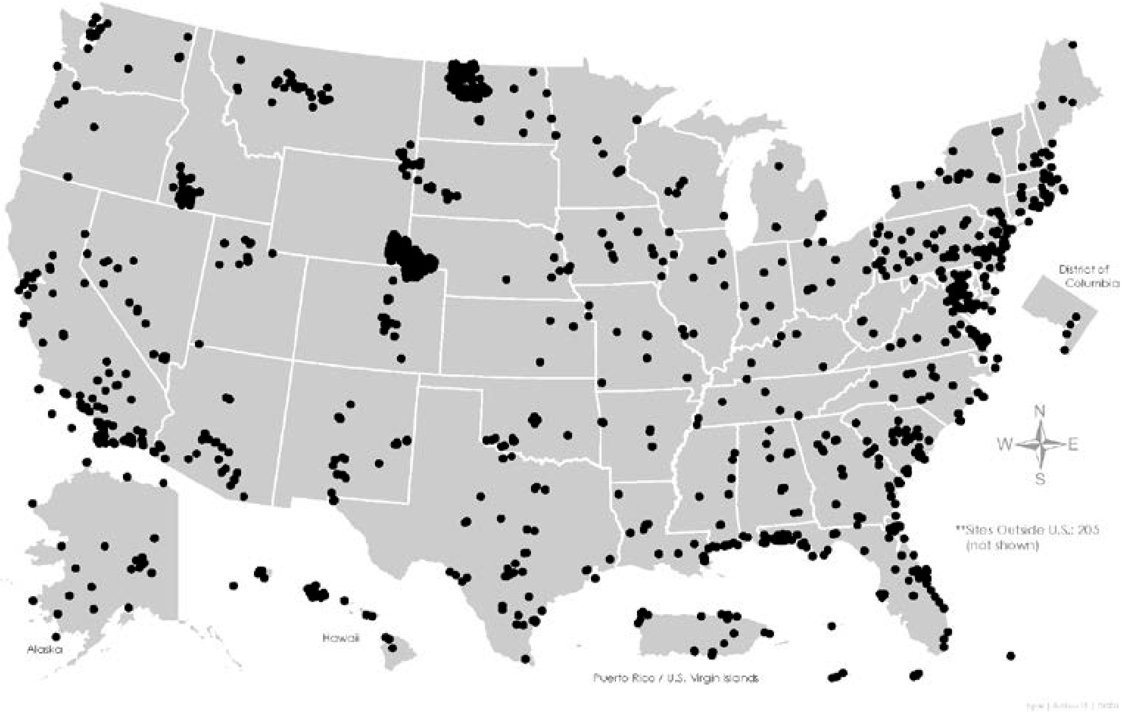

Figure 1.8: Sites that indicated effects from multiple vulnerability areas (flooding, extreme temperatures, wind, drought, wildfire) (DoD, 2018).

Climate change will engender new natural resource management challenges and priorities, on top of complicating current management practices, and possibly diminishing their effectiveness (Stein et al., 2019). An increase in the strength and magnitude of extreme events such as droughts, floods or wildfires will decrease the suitability of installations for their current mission, can expose facilities and operational assets to significant damage, and threaten personnel protection and safety (DoD, 2014; Garfin et al., 2021; Ripley et al., 2021). Climate change can jeopardize military preparedness by affecting the suitability of training and testing sites due to land loss or alteration of ecosystems; limiting the timing of training or testing due to wildfires, floods, or other extreme events; and increasing damage to buildings and equipment due to extreme weather and loss of protection from wetlands, dunes, and other natural systems (DoD, 2014, 2021; Stein et al., 2019). Drought and aridification will threaten water supply, especially to installations already in arid areas, and increase the energetic demand for cooling (DoD, 2021). The new climatic conditions could also be the ideal conditions for new exotic species invasion, which will increase the cost of management and can make areas unsuitable for training (Stein et al., 2019). These climate-related impacts are already being observed at installations throughout the U.S. and overseas and affect many of the Department’s activities and decisions related to future operations (DoD, 2014).

Resources regulated under laws are at risk and may stronger impacts under changing climatic conditions. Climate change will impede regulatory compliance, and might result in increased costs or training restrictions (Li & Male, 2020). Drought and shifts in precipitation distribution increase the risk of wetlands and open waters drying up, which may increase the costs and difficulties of complying with the Clean Water Act (Stein et al., 2019). Installation managers will need to work with regulatory organizations such as the U.S. Fish and Wildlife Service to identify innovative approaches that could accommodate the increasing disconnect between climate-driven ecological changes and the persistence-oriented regulations (Stein et al., 2019).

1.3.2.2 Regulations

In 2014, the *Department of Defense Climate Adaptation Roadmap* established three broad adaptation goals: assessing the effects of climate change on the DoD; integrating climate change considerations across the DoD and managing associated risks; and collaborating with internal and external stakeholders on climate change challenges (DoD, 2014). In 2016, the DoD issued directive *4715.21 Climate Change Adaptation and Resilience*. This policy outlines the preparation, cooperation, and coordinated planning by the DoD to adapt to climate change, plan for its impact, and to ensure the continuity of DoD operations (Garfin et al., 2021). The need to address climate change is also now incorporated into the DoD’s Natural Resources Conservation Program Instruction as well as the INRMP Implementation Manual (Ripley et al., 2021). The updated INRMP Implementation Manual calls for installations to address climate considerations when updating their INRMPs (Stein et al., 2019). These instructions are complemented by branch-specific guidance on climate change integration in INRMPs. The DoD Climate Adaptation Plan (2021) sets the goals of climate change adaptation on military lands. These goals include the monitoring of the relevant environmental conditions on operations and installations, the impacts of installations’ actions to guide climate-informed management and resource allocation decisions. The main objective is to ensure the DoD can operate under changing climate conditions and maintain operational capability (DoD, 2021).

The supporting programs funding natural resources management on military installations (SERDP, ESTCP, Legacy Resource Management Program) have also adapted their goals to include climate change adaptation and resilience. They are funding studies directly relevant to DoD installations for improving understanding of climate change risks to natural resources and the continuity of the military mission (Garfin et al., 2021; Stein et al., 2019). They support the development of the science, technologies, and methods needed to manage and enhance the resilience of DoD installation infrastructure and natural spaces (DoD, 2019). The REPI program has also been adapted, providing funds that can also be leveraged by the installation and its partners to enhance state, local, and non-governmental initiatives for climate resilience efforts (DoD, 2021).

1.3.2.3 Adaptations

Climate change-focused natural resources management is built around resilience and adaptation, management is designed to reduce climate-related vulnerabilities and enhance resilience to climate impacts (DoD, 2019; Stein et al., 2019). Steps are taken to prepare for, manage, and reduce the impacts and risks of a changing climate (Ripley et al., 2021). Based on observed and projected climate-related changes, managers can assess climate vulnerabilities of infrastructure, environment and natural resources and implement appropriate strategies and actions to maintain natural resources on installations and maintain base operations, with monitoring changes. This loop of assessments and actions has to be repeated to keep track of changes caused by ongoing climate change and management actions (DoD, 2014; Ripley et al., 2021; Stein et al., 2019). Efficient assessments comprise a set of metrics that provide key information on ecosystem status that can be repeated regularly to track changes (DoD, 2014). Adaptation plans can then be updated on a periodic basis. Information on climate change can be integrated in INRMPs in the form of historical regional trends and projections of future climate, information developed for other purposes (e.g., facilities risk assessments) that can be used to assess climate change impacts or adaptation strategies, a discussion of sustainability in the context of climate change supporting the development and updating of vulnerability assessments, information from regional collaboration to develop vulnerability assessments and adaptation strategies, and comprehensive incorporation of training and test vulnerabilities related to climate change (Ripley et al., 2021).

Due to the complexity of local climate change and the intricate chain of reactions in the ecosystem, reactive actions are the method used more often. With planning, adaptation actions can be taken proactively to prepare for change, thanks to assessments including modelling of future climate trends paired with known reactions of species and ecosystems (Stein et al., 2019). Either way, effective climate adaptation needs a solid understanding and incorporating of current and potential future climatic conditions and associated impacts and uncertainties into relevant decision-making processes (Stein et al., 2019). The goal of climate change-oriented management is to increase the resilience of ecosystems, their ability to resist, withstand, adapt or recover from change (Stein et al., 2019). Depending on the resilience of species and ecosystems, actions can be taken to increase this resilience, but with continuing changes, permanent shifts in natural systems will be triggered. In some cases, management goals need to be reconsidered, as managing for the persistence of current conditions will not be possible. In those cases, managers and regulators that shifts are inevitable (Stein et al., 2019).

1.3.2.4 Threatened and endangered species

Climate change impacts distribution, health and life cycles of vegetation and wildlife (Section 1.2.4). Several species have already been listed under the Endangered Species Act primarily as a result of climate change-related threats (Ripley et al., 2021). Other federally listed species are regarded as sensitive to climate change (Delach et al., 2019) and more species are considered climate-vulnerable and could experience declines to a level where ESA protections may be needed, adding to DoD regulatory responsibilities (Ripley et al., 2021).

As climate changes, maintaining populations of vulnerable species will become more complex and costly (Stein et al., 2019). T&E species management is a balance between regulatory compliance and supporting the mission. If species need more drastic measures, it might become progressively difficult to maintain this balance (Stein et al., 2019). Different speed and amplitude in species responses to climate change can also result in shifting conservation priorities (Stein et al., 2019). Some species will expand their range into military lands, while others will shrink. Population declines and local extinctions are important concerns for DoD installations with T&E species, who might have to increase species and land management requirements (DoD, 2014; Stein et al., 2019). As the population declines, protecting the individuals left becomes increasingly important. Climatic refuges, areas where conditions remain relatively stable into the future, are a key areas that can increase the resilience of species of interest that need to be identified and protected (Stein et al., 2019). They can also be areas that need less management efforts to remain stable than their surroundings. Actions that can benefit T&E species can be the conservation of climate refuges, habitat-based management, restoration or enhancement of connectivity and corridors, and monitoring in relation to key climate metrics for the species affected (Stein et al., 2019).

1.4 Study site: the San Pedro basin

1.4.1 Southwest USA

The Southwest USA includes Arizona, New Mexico, California and Nevada (with Colorado, Utah and Texas sometimes included) (Figure 1.9). This region displays a wide range of ecosystems with their own specifications and climatic characteristics. It includes the hottest and driest regions of the country, covered in deserts and grasslands. In these arid areas, already scarce water resources are used for agriculture, energy production and in cities (Meixner et al., 2016; Reidmiller et al., 2018). This thesis was conducted in the Sonoran Desert in Southeast Arizona, in an area where the low and flat stretches of semi-arid scrubland of the San Pedro valley are interrupted by isolated small mountain ranges called the Sky Islands, such as the Huachuca Mountains. The study site is located in the Sierra Vista Subwatershed, and covers the Huachuca Mountains, the San Pedro River and the valley in between (Figure 1.10). I selected this study site for its diverse mosaic of habitats, including several riparian systems distinctive in drylands, including mountain non-perennial streams, grassland washes, and valley river with perennial and intermittent reaches. This complex patchwork of ecosystems, all under the influence of the North-American Monsoon, allowed me to look at how environmental factors such as climate and local geology can impact water, vegetation and habitat distribution, even at a small scale.

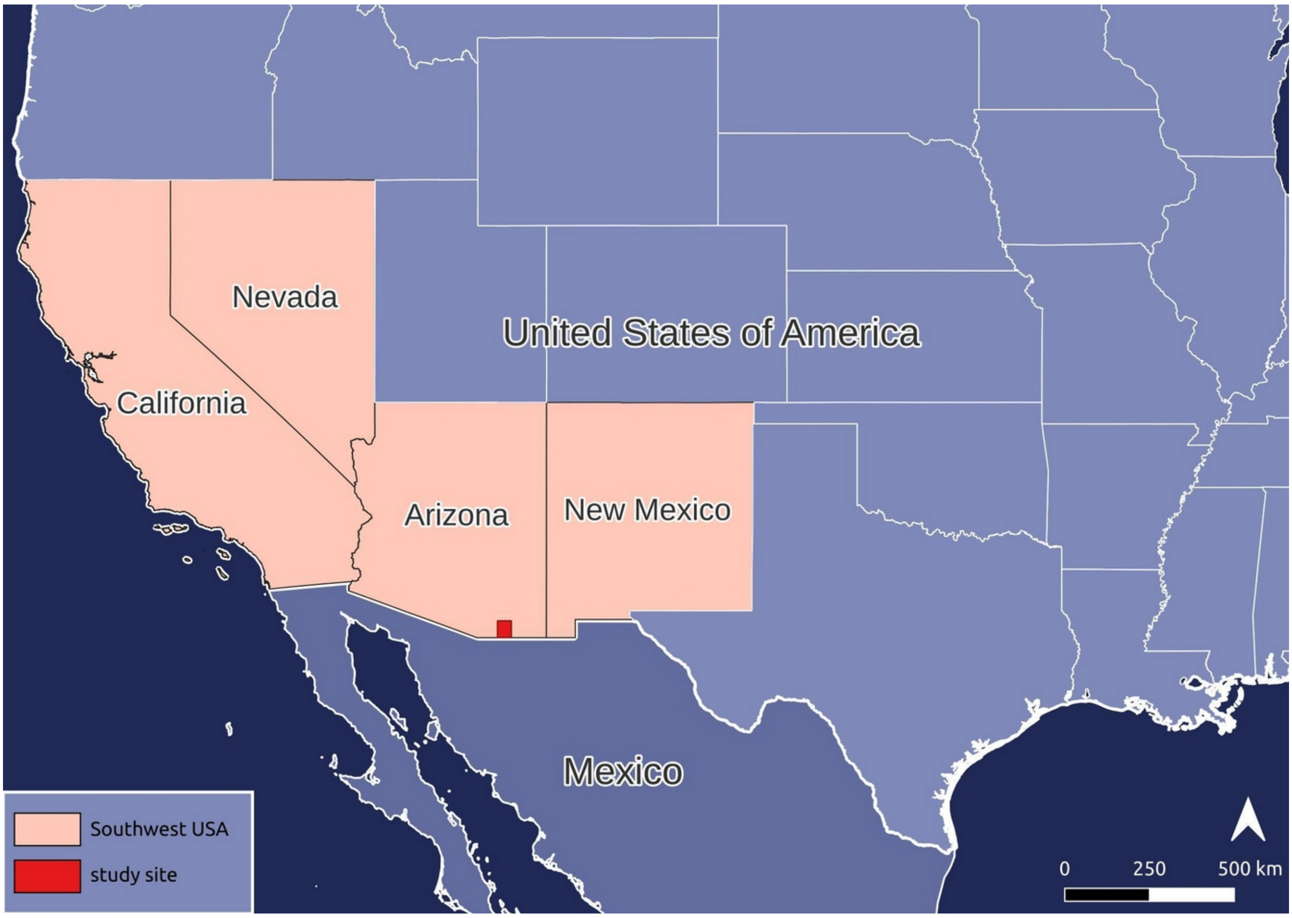

Figure 1.9: States of the Southwest USA and location of the study site in south-east Arizona.

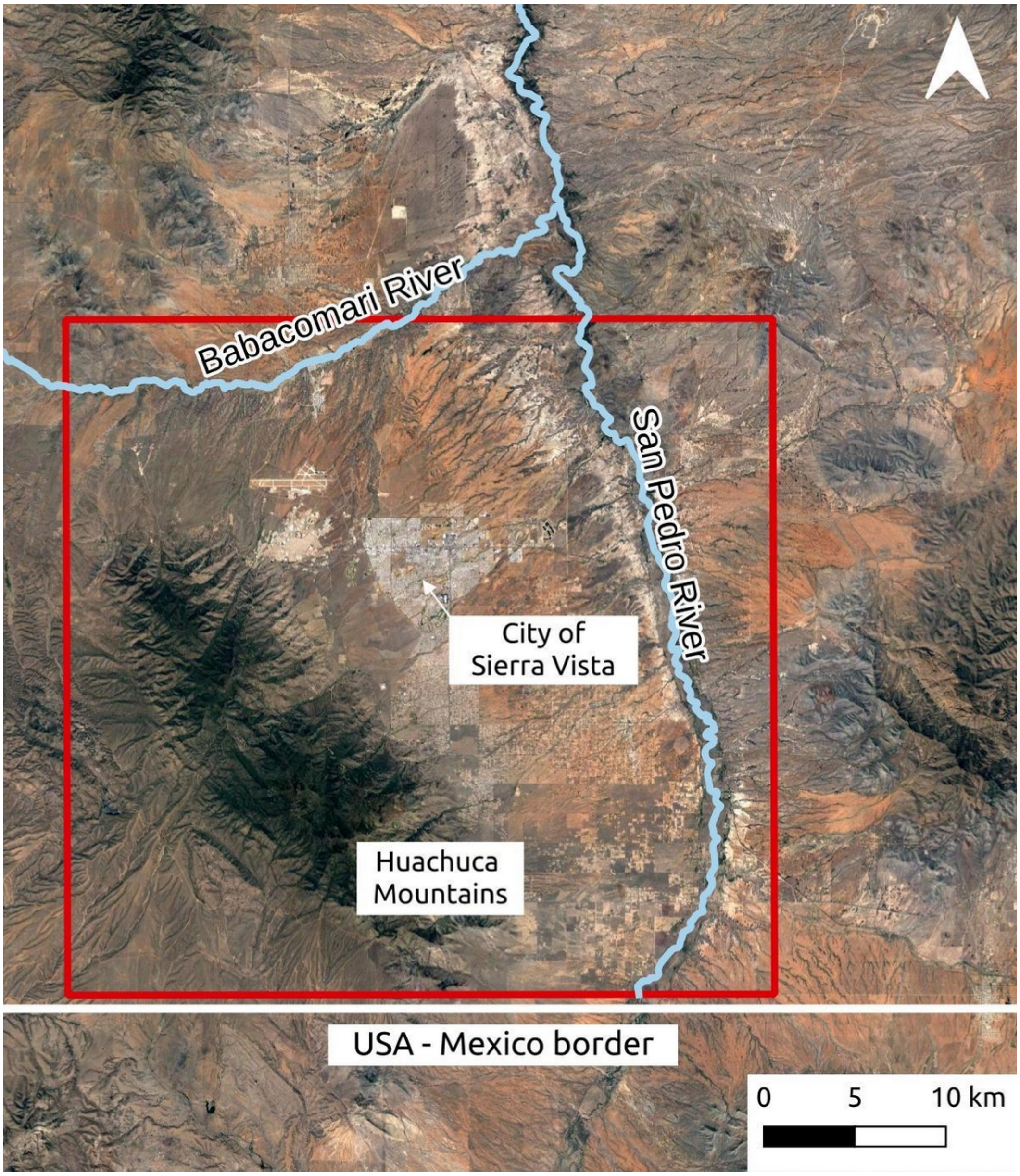

Figure 1.10: Study site (red outline), which covers the Huachuca Mountains, the San Pedro River and the Sierra Vista valley.

1.4.1.1 Climate and water distribution

Inland regions of the southwest USA, which includes my area of interest, are characterized by a continental climate with an important seasonal variation in temperature and rainfall (Figure 1.11). Annual precipitation distribution is focused during the North American Monsoon (NAM) and in winter, whereas spring and autumn are largely dry (Figure 1.11) (Sheppard et al., 2002).

Between low to no precipitation and high temperatures, spring (May-June) is the driest season. Precipitation is greater at higher elevations (Figure 1.12 & 1.13) and winter rain can be stored as snow on mountain peaks (Sheppard et al., 2002).The NAM is the main rain season and occurs during the summer (in July, August and September). Short but strong monsoon storms comprise ~60% of annual rainfall, with less intensive but longer and more widespread winter precipitation providing the remainder (Levick et al., 2008). Monsoon precipitation is delivered in short, localized and intense storms coming from the Pacific Ocean (Gulf of Mexico and Gulf of California) (Webb & Betancourt, 1992). On top of seasonal variability, precipitation in the region also follows decadal-to-multidecadal cycles, alternating drought and wetter-than-average periods (Levick et al., 2015). Rainfall distribution and amounts are partially controlled by cyclical temperature variations in the Pacific Ocean. During El Niño events (increase in sea-surface temperature of the eastern Pacific Ocean), winters are wetter, while La Niña (the opposite of El Niño) results in drier winter for the Southwest USA (Sheppard et al., 2002). The Pacific Decadal Oscillation (temporal variation in sea-surface temperatures over the Northern Pacific Ocean) also influences precipitation variability (Sheppard et al., 2002). Temperature follows the typical seasonal cycle with a maximum in mid-summer and a minimum in mid-winter (Figure 1.11).

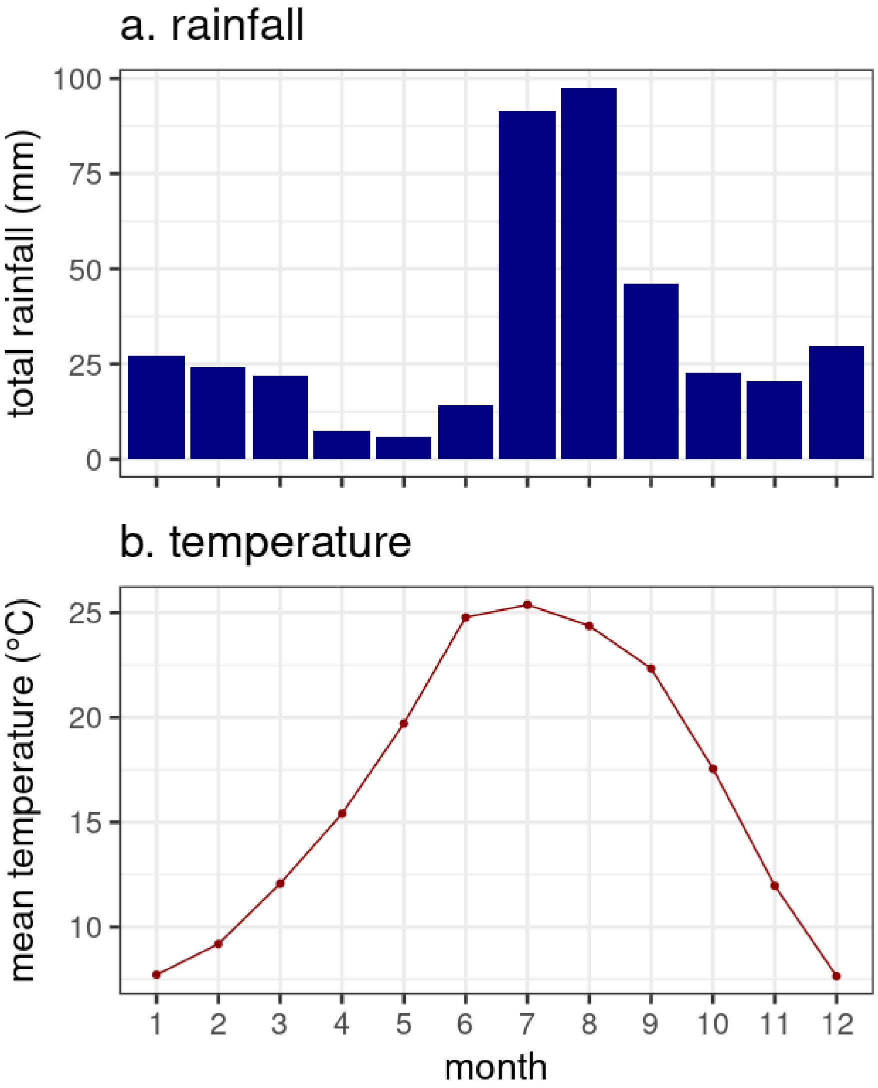

Figure 1.11: 1991-2020 average monthly total rainfall (a) and temperature (b) over the study area. PRISM data, https://www.prism.oregonstate.edu/.

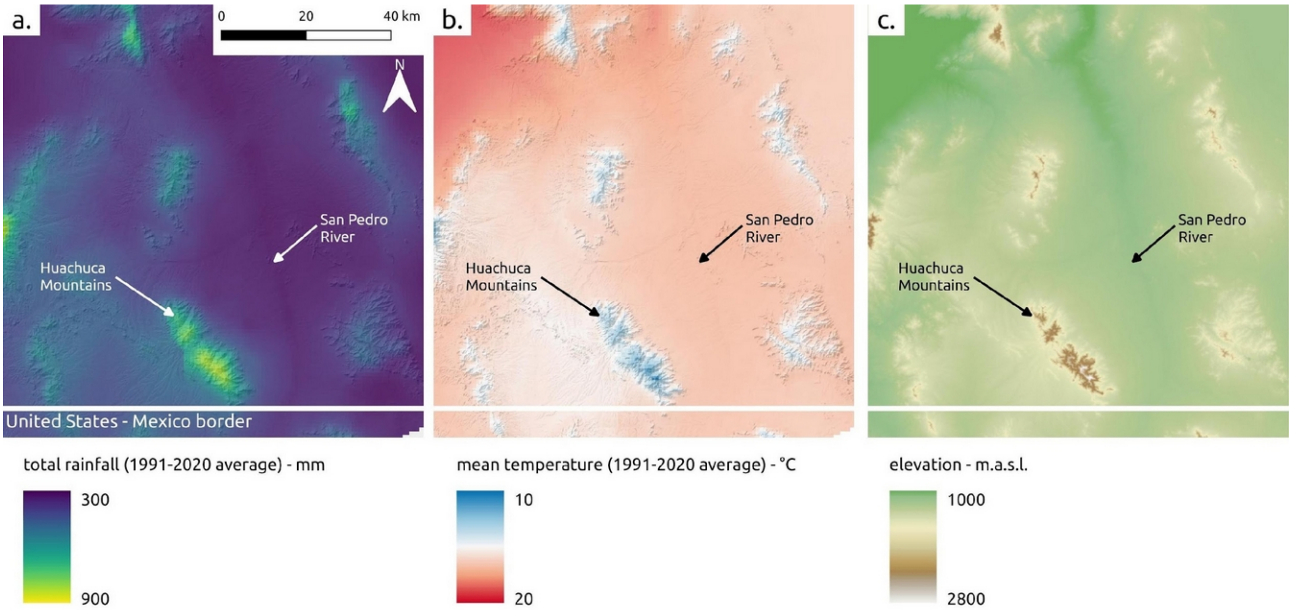

Figure 1.12: Average annual rainfall (1991-2020 average) (a), mean temperature (1991-2020 average) (b) and elevation (c) over the study area. PRISM data.

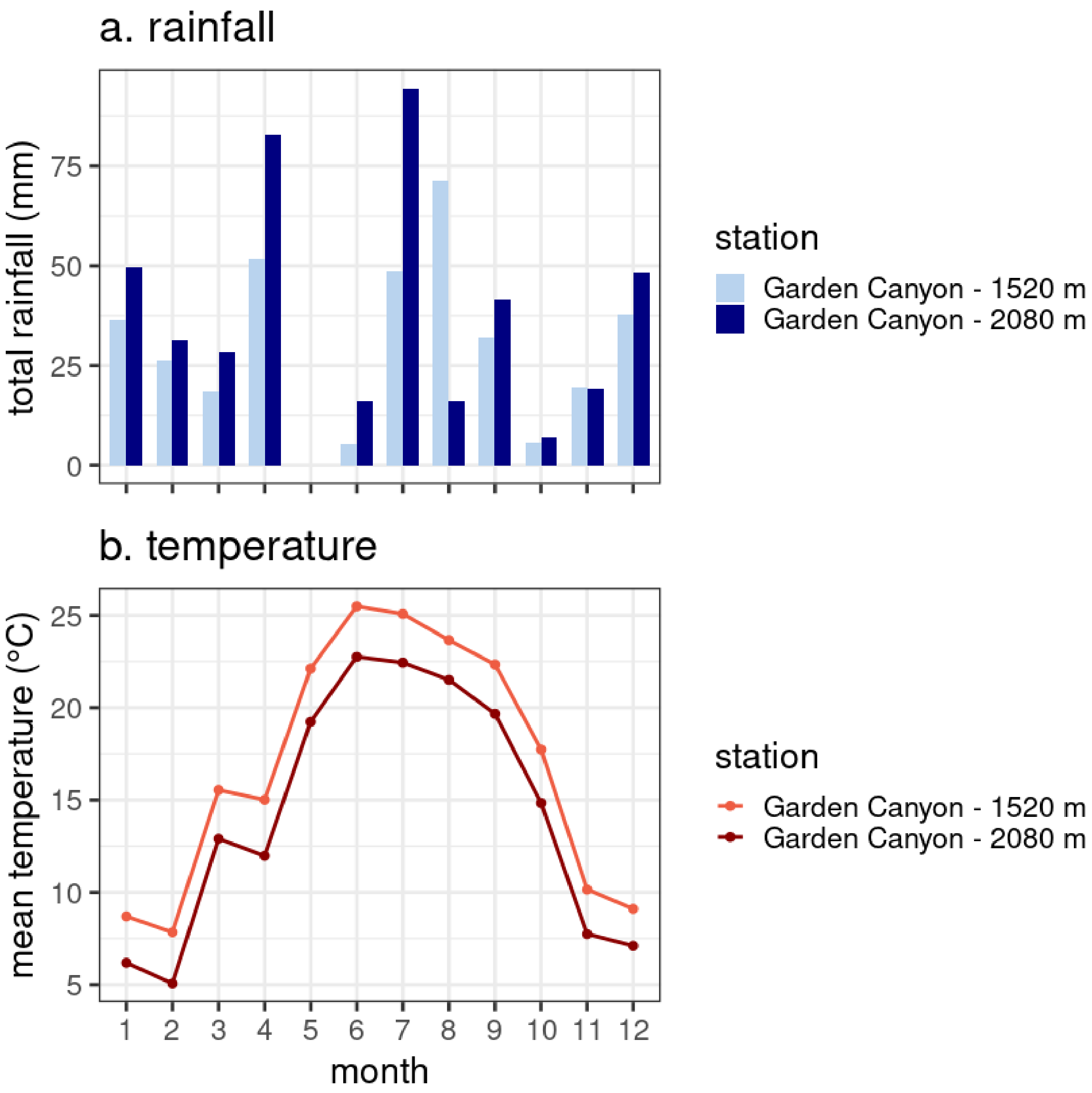

Figure 1.13: Climatic variations between lower Garden Canyon (in the valley, 1520 m.a.s.l.) and upper Garden Canyon (in the mountain, 2080 m.a.s.l) for monthly total rainfall (a) and monthly mean temperature (b). Data from Fort Huachuca weather stations, 2004.

Nearly 60% of streams in the contiguous USA are classified as ephemeral or intermittent, a proportion which rises to 81% in the arid southwest, and 94% in Arizona (Levick et al., 2008; Nadeau & Rains, 2007). Streamflow is fed by rainfall and to a lesser extent by snowmelt and the local water table. Intra-annual variability of streamflow and groundwater level are largely controlled by seasonal cycles of precipitation and evapotranspiration (Patten, 1998; Shafroth et al., 2000; Stromberg et al., 1996). Rainfall from the monsoon intense thunderstorms turn into runoff and flash floods, while milder winter rains and snowmelt more readily infiltrate, provide soil moisture and recharge the water table that support baseflow in perennial reaches (Loik et al., 2004; Simpson et al., 2013; Singer & Michaelides, 2017; Vera et al., 2006). This water follows paths along canyons where it forms non-perennial streams, sometimes travelling underground along fault lines. The high seasonal variability of precipitation leads to a high variability of flow, with most reaches drying out in spring and flooding during the monsoon. Perennial rivers tend to be gaining reaches and be linked to high and stable water tables, whereas intermittent streams tend to lose water and have a greater and more seasonally variable depth to groundwater (Section 1.2.1) (Leenhouts et al., 2006).

1.4.1.2 Vegetation

In the southwest USA, upland vegetation only access to water is rainfall-derived soil moisture, while riparian vegetation has access to soil moisture derived from rainfall, bank infiltration from the stream, and the capillary fringe associated with a shallow water table (Lite & Stromberg, 2005; Patten, 1998). Ehleringer et al. (1991) studied water use by desert plants in southern Utah and found out that, while all species used winter-spring recharge precipitation for spring growth, summer rains were not used by all species. If annuals and succulent perennials depended heavily on summer precipitation, herbaceous and woody perennial species used both summer and winter rains with several of the woody perennials showing no response to summer precipitation. In an Arizona floodplain, the cottonwood-willow forests (the main riparian vegetation community of this region) decline in abundance and age-class diversity along spatial gradients of decreasing streamflow permanence and groundwater depth (Leenhouts et al., 2006; Snyder & Williams, 2000). Stromberg (2001b) showed that frequent summer floods increased the growth of the riparian tree Arizona sycamore (*Platanus wrightii*) in both perennial and non-perennial reaches by replenishing limiting nutrients and soil moisture. Antecedent season moisture condition is an important control on phenology, especially in lowland vegetation communities. Precipitation of the previous autumn is an important control on spring phenology at all elevations, and a decrease in autumn rainfall leads to a later onset of the growing season (Crimmins et al., 2010). Seasonal rainfall distribution is especially important for annual grasses that cannot rely on groundwater. Drought before the start of the growing season resulted in a delayed onset and an earlier senescence, shortening the length of the growing season (Currier & Sala, 2022).

Riparian forests of the arid Southwest fulfil their role as biodiversity hotspots (Section 1.2.2). Arizona riparian forests support more plant species than the surrounding area, including species only found in the riparian corridor, and total species richness decreases as aridity increases (Stromberg et al., 2017). The cottonwood-willow gallery forests found along non-perennial streams of the Southwest are one of the rarest habitats in North America, but are used by at least 80% of all animals, with more than half of these species considered riparian obligates. They are critical stopover areas for migrating birds by providing cover, food, and water. Riparian forests were found to shelter 10 times the number of migrating birds per hectare than adjacent non-riparian habitats and twice as many breeding individuals (Krueper, 1993). Riparian ecosystems within the Southwest USA have been greatly altered by management of land and water resources, including grazing and groundwater pumping in rivers’ floodplains (D. E. Busch & Smith, 1995; Patten, 1998). Arizona has lost more than 90% of its historic gallery cottonwood/willow forests, and populations of riparian-dependent species have seen important declines due to habitat loss or modification (Krueper, 1993).

1.4.1.3 Climate change

Even if projected climate change varies across the Southwest USA, there has been a detectable warming of the region since 1901, with the average annual temperature having increased by 0.9ºC between 1901 and 2016, and temperatures are expected to keep increasing throughout the area (Figure 1.14) (Ault et al., 2016; Meixner et al., 2016; Reidmiller et al., 2018). Precipitation predictions have a larger degree of uncertainty than temperature (Archer, 2008). In the past, rainfall has fluctuated over short and longer timescales, and precipitation total has not shown an obvious trend in direction (Goodrich et al., 2008; Thomas & Pool, 2006). But climate is projected to become more extreme, with longer dry periods interrupted by more violent storms translating to intensified floods and droughts (Archer, 2008; Ault et al., 2016; Coe et al., 2012).

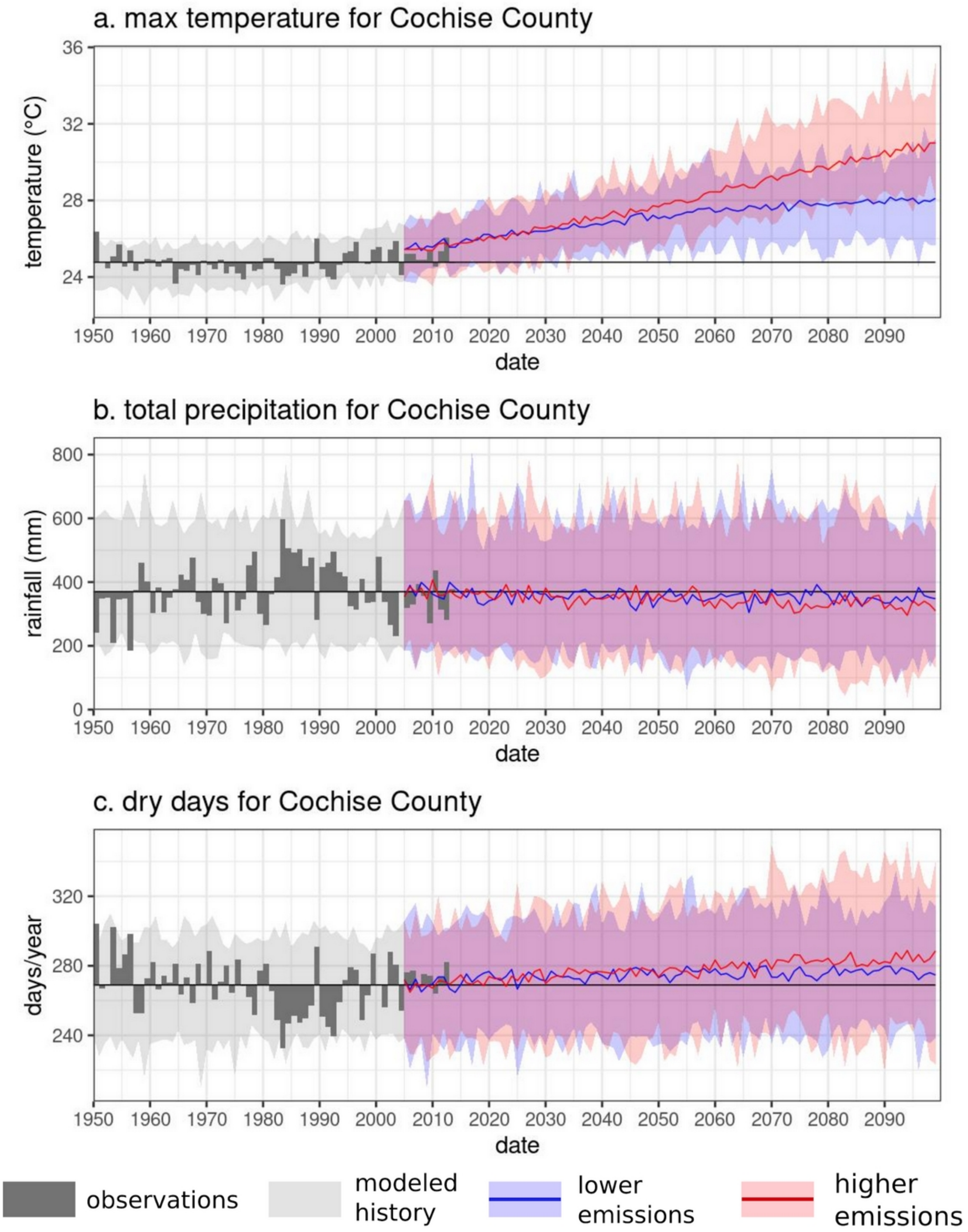

Figure 1.14: Historical observations and projection of climate variables for Cochise County, where the study site is located, with the changes in maximum temperature (a), total precipitation (b) and number of dry days (c). The black horizontal line represents the 1961-1990 observed average. Data from the Climate Explorer, https://crt-climate-explorer.nemac.org/.

Water resources have declined in the Southwest USA as a consequence of rising temperatures, changes in rainfall distribution and an increased demand for agriculture and population use. As spring temperatures peak earlier in the year, snow melts earlier and the fraction of total annual precipitation falling as snow is also decreasing, which means that less water is stored in mountains for gradual release. A smaller snowpack leads to a drop in mountain recharge, the main water source for several aquifers, especially in the south (Archer, 2008). Higher temperatures and reduction of snow intensified the recent severe drought in California and are amplifying drought in the Colorado River Basin (Meixner et al., 2016; Reidmiller et al., 2018; Wuebbles et al., 2017). Simulations of drought duration show depleted soil moisture lasting for 12 years or more, compared to historical duration between 4 and 10 years (Cayan et al., 2010).

As for groundwater, the available estimates show average declines of 10–20% in total recharge across the southern aquifers due to reduced snowpack but with a wide range of uncertainty. These aquifers are already concerned by a significant overdraft under current climate conditions (Meixner et al., 2016; Reidmiller et al., 2018; Wuebbles et al., 2017).

More frequent and severe droughts linked to climate change are set to significantly alter flow intermittence patterns and hydrologic connectivity in dryland streams of the Southwest by increasing the number of zero-flow days, the length of dry periods and their frequency (Jaeger et al., 2014). A general decline in streamflow, surface-water availability and soil moisture is expected across the region (Archer, 2008; Reidmiller et al., 2018; Seager et al., 2013).

The Southwest USA is affected by a regional-scale mortality of large overstory trees, that seems mainly caused by an increase in temperature leading to droughts and increased fire risk, as well as severe pest infestations induced by weakened trees and warmer winters. Breshears et al. (2005) quantified that, after 15 months of depleted soil water content, over 90% of the dominant overstory tree species (*Pinus edulis*) of their study site died, across a wide range of size and age classes. These changes alter ecosystem types, their associated properties and land surface conditions. Continued climate change would cause north–south or upslope shifts of ecosystems as vegetation follows the changes in temperature and water availability (Section 1.2.4) (Archer, 2008). Areas highly vulnerable to such changes include the Arizona Sky Islands and the riparian forests in arid areas (Reidmiller et al., 2018).

Shafroth et al. (2000) studied the reaction of *Populus sp.* (cottonwood) and *Salix sp.* (willow), two common genii found in riparian forests in Arizona, and they linked the mortality of the individuals to droughts that resulted in a severe decline of the water table. In sites with important fluctuations in groundwater levels, *Populus* saplings presented a broad depth distribution of coarse roots and were rooted relatively deep, whereas in sites with a more stable water table, trees showed roots distribute in a flare directly above the water table. The results of their work suggest the importance of change in groundwater depth relative to a previous condition or pattern as opposed to the absolute depth to the water table. The trees will adapt the shape and distribution of their root system to the depth of the water table (Shafroth et al., 2000; Sprackling & Read, 1979; Stromberg, 2013). But it’s when the drawdown rate of the aquifer is too fast and too deep that the riparian trees, unable to grow roots fast enough, will loose their main water source and start to suffer from water stress, as shown in Colorado (Scott et al., 1999; Segelquist et al., 1993) and California (Scott et al., 2000).

Dryland vegetation distribution and health depends on the timing and amount of rainfall. An increase in temperature will increase evapotranspiration rates, and rainfall will occur with increasing intensity, turning to rapid runoff which will reduce water infiltration in the soil and water table, which results in reduced water availability for vegetation (Archer, 2008). Annual grasses depend on the summer monsoon and winter rainfall is an important recharge for groundwater providing water to riparian forests and baseflow to streams, so climate change-induced shifts in seasonal rainfall and temperature patterns will have consequences on vegetation and wildlife composition, distribution, and abundance (Archer, 2008). Reduced water permanence in streams, groundwater decline and changes in flood timing and amplitude are leading to a reduction of the cottonwood-willow gallery forest, a mismatch between flood and seed dispersal, and invasion by exotic species such as saltcedar (*Tamarix ramosissima*) (Coe et al., 2012; Lite & Stromberg, 2005; Rood et al., 2008; Stromberg, 1998). Higher temperatures also increase the risk of lethal dehydration for smaller birds, which might have to limit foraging to avoid heating up (Albright et al., 2017). For wildlife already at the limit of their temperature range, riparian forests' role as a cooler refuge will be exacerbated (Stromberg, 2013). Reduction of refuge pools during the dry season will have a negative effect on all species depending on them, including endangered fishes (Jaeger et al., 2014).

1.4.2 San Pedro River

Crossing the semi-arid scrubland of southeast Arizona is the San Pedro River. This river originates in Sonora, Mexico, and flows from south to north into the Gila River (Arizona), covering a distance of 279 km. The San Pedro is the last significant free-flowing river in Arizona with long perennial reaches and following natural cycles of floods, critical for the rare riparian cottonwood-willow corridor forest (Goodrich et al., 2008). Between the late 19th and early 20th century, the San Pedro underwent entrenchment, followed by channel widening (Hereford & Betancourt, 2009), which led to shifts in vegetation in some reaches. The upper San Pedro basin between the Mexican border and Saint David is part of the San Pedro Riparian National Conservation Area (SPRNCA), created in 1988, which resulted in removal of cattle grazing and agriculture from the riparian zone (Stromberg & Tellman, 2012, p. 371). The SPRNCA is managed by the Bureau of Land Management, a federal organization. The primary tributary to the San Pedro River in the Sierra Vista Subwatershed is the Babocomari River.

The San Pedro is characterized by a series of perennial and intermittent reaches (Leenhouts et al., 2006; Stromberg & Tellman, 2012, p. 377). Longitudinal organization of intermittent and perennial reaches along the San Pedro River are controlled by the underlying geology. Perennial reaches are mainly located in areas of impervious consolidated volcanic and sedimentary rocks cropping which push groundwater to the surface. In some areas, the permeable alluvium that forms the streambed and constitutes a productive aquifer is interlayed by silt and clay layers maintaining a perched water table under the stream (Brown et al., 1966; Stromberg & Tellman, 2012, p. 287). Flow is derived from summer monsoon precipitation, and, for perennial reaches, from winter baseflow supported by local groundwater discharge from the regional aquifer and storm runoff stored as alluvial groundwater (Kennedy & Gungle, 2010; Stromberg & Tellman, 2012, p. 292; Thomas & Pool, 2006). Dry sections of the river reach their maximum extent during the dry season (May-June), due to low precipitation, an increase in temperatures and high transpiration by deciduous trees that have started their growing season (Kennedy & Gungle, 2010). Peak streamflow occurs during the monsoon thanks to infiltration-excess overland flow. During wetter years, streamflow in intermittent reaches can be maintained throughout the fall. In winter, streamflow levels rise again due to rainfall and a decrease in vegetation transpiration after senescence.

The San Pedro River is home to various riparian vegetation communities dominated by gallery forest of Fremont cottonwood (*Populus fremontii*) and Goodding's willow (*Salix gooddingii*) near the river, with mesquite (*Prosopis velutina*) forest and grasslands on an elevated former floodplain (Figure 1.15) (Gungle, 2006; Leenhouts et al., 2006; Stromberg et al., 2005). Streamflow permanence, and groundwater seasonal fluctuations and depths are an important control on riparian vegetation distribution (Leenhouts et al., 2006). Perennial reaches (fed by shallow groundwater) are the main areas with cottonwood-willow gallery forest, while more intermittent and ephemeral reaches are occupied by shrubs and saltcedar (*Tamarix ramosissima*) (Lite & Stromberg, 2005). The riparian ecosystem along the San Pedro provides habitat for over 350 species of birds, 80 species of mammals, and 40 species of reptiles and amphibians, including several federally listed endangered species (Kennedy & Gungle, 2010; Levick et al., 2015).

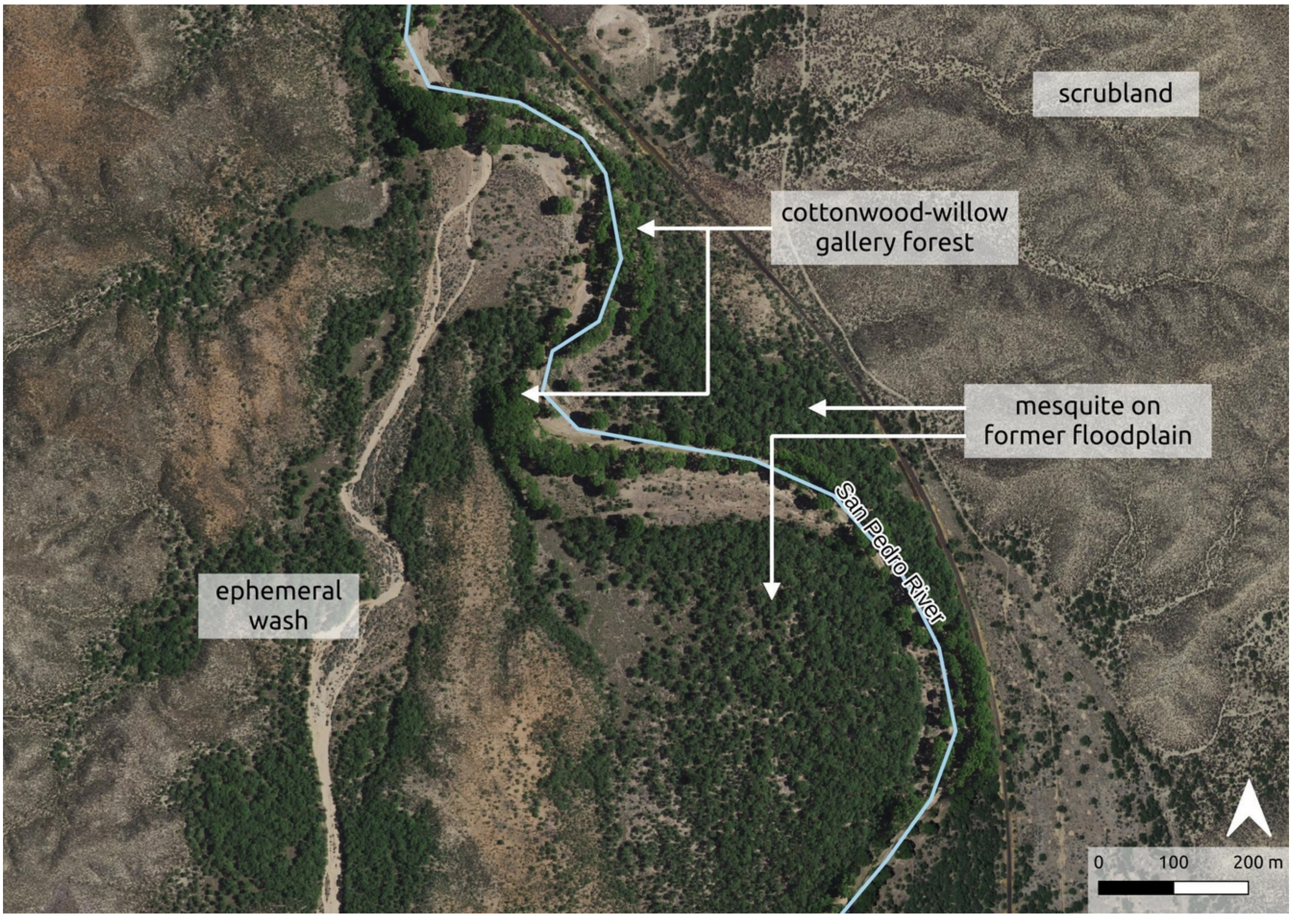

Figure 1.15: National Agriculture Imagery Program (NAIP) imagery of the San Pedro River with the main vegetation communities labeled.

The work of dividing the San Pedro River and its riparian corridor into homogeneous reaches and sorting each reach in a riparian condition class (dry, intermediate or wet) was conducted by Stromberg et al. (2006). The length of the river included in the SPRNCA was split in reaches based on two physical controls on the structure of riparian vegetation: streamflow hydrology and geomorphology. Flow permanence was used as a measure for streamflow hydrology. Every year, the Upper San Pedro Community Monitoring Network, led by The Nature Conservancy, monitors the spatial extent of perennial flow in the river right before the beginning of the monsoon and generates a flow permanence map. The 2002 map was used to estimate the extents of surface flow in preliminary 3 km segments of the river. As for geomorphology, it entails both channel sinuosity and floodplain width. The river was split into 3 km segments and dividing stream length by straight length gave channel sinuosity.

Floodplain width was measured in the field and the final extent of each reach was determined by visually examining overlays for streamflow intermittency, sinuosity, and floodplain width and maximizing within-reach homogeneity. A riparian condition index based on nine field-measured vegetation traits (indicators) sensitive to changes in streamflow permanence and/or groundwater levels was developed, with the goal to link vegetation and ecosystem conditions to changes in water availability. The riparian assessment model underlying this index was built by using data collected during 2000–2002 at 17 sites and validated at 10 additional sites. This model was then applied to the SPRNCA by using indicator data collected at 26 sites from 2002 through 2004. The field data collected at each site are used to determine a scoring value for each indicator. These values are then averaged to get a site-specific score before scaling the result at the reach level. The score for each reach reflects the spatial variation in water availability along the river. Each riparian condition class (wet, intermediate, dry) relates to a specific range for site hydrology, vegetation structure, and ecosystem functional capacity (Figure 1.16 to 1.18).

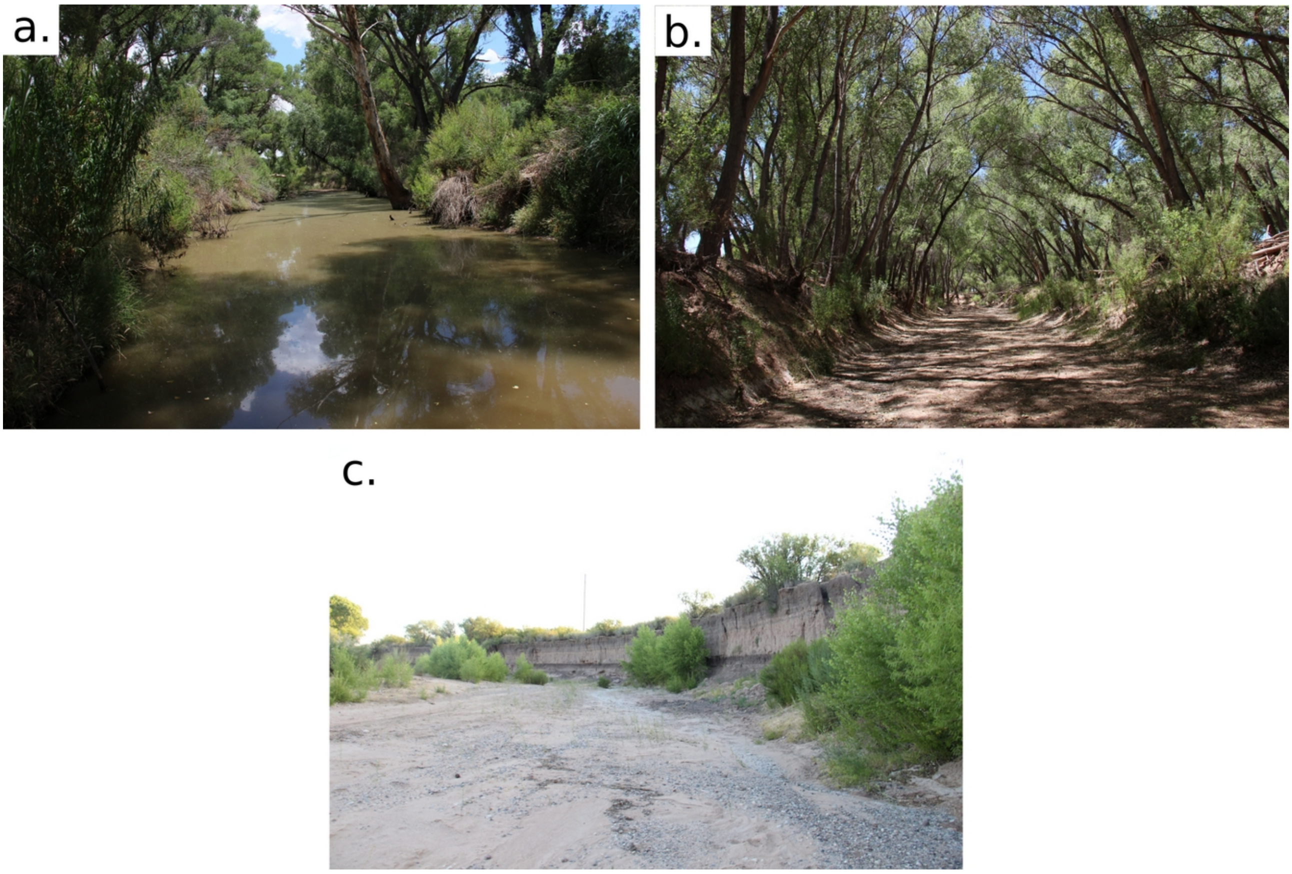

Figure 1.16: Pictures of wet (a), intermediate (b) and dry (c) reaches of the San Pedro River (June 2019). The entrenchment, with the former floodplain on a terrace, is visible in picture c.

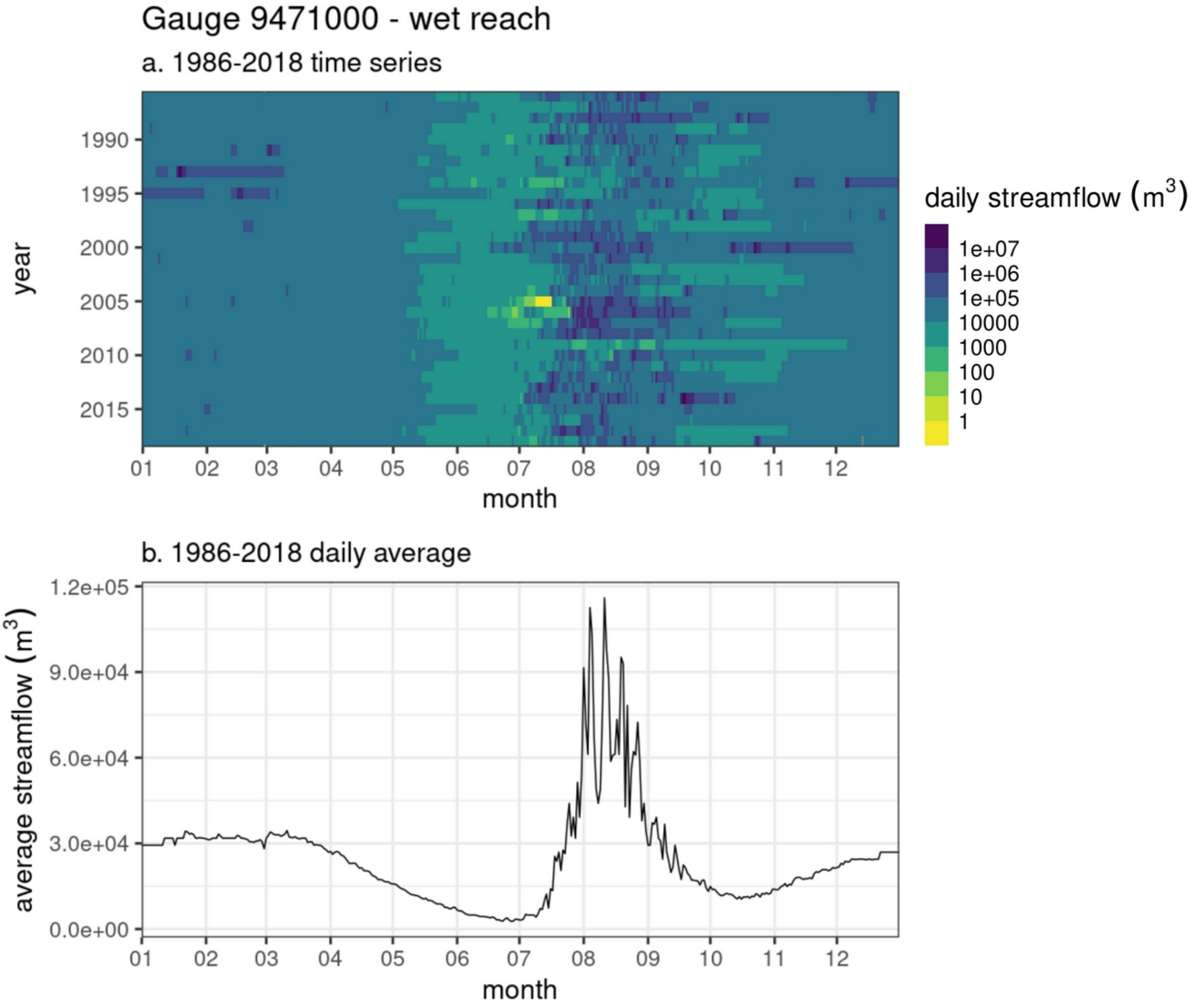

Figure 1.17: Daily streamflow for gauge 9471000, on a wet reach. Heatmap of daily time series from 1986 to 2018 (a) and daily average for the 1986-2018 period (b). Gauge location is shown in Figure 2.2.

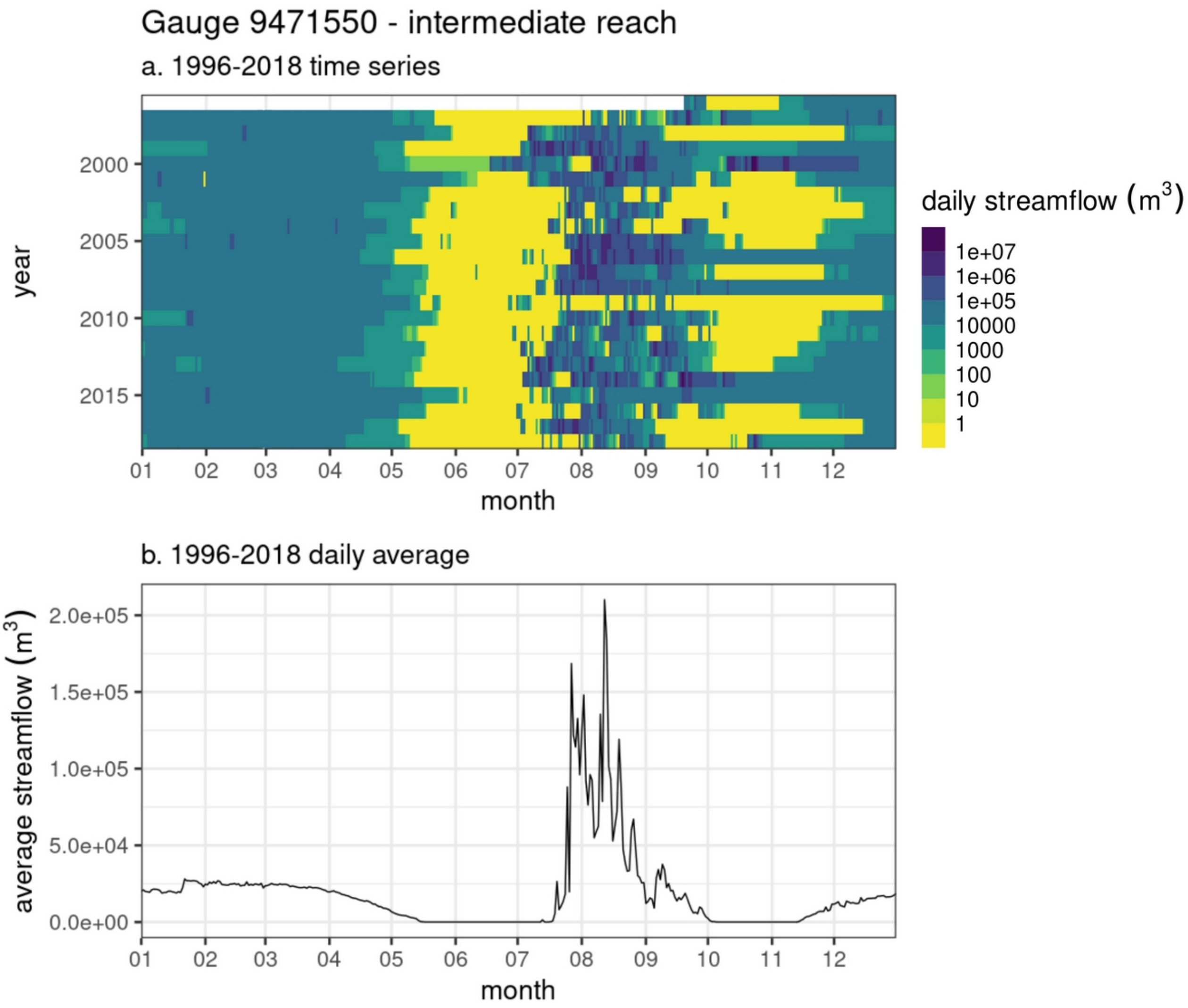

Figure 1.18: Daily streamflow for gauge 9471550, on an intermediate reach. Heatmap of daily time series from 1996 to 2018 (a) and daily average for the 1996-2018 period (b). Gauge location is shown in Figure 2.2.

1.4.3 Huachuca Mountains

The Huachuca Mountains are part of the Madrean Sky Islands (or Madrean Archipelago). Sky Islands are small and scattered mountain ranges separated by low valleys. The Madrean Sky Islands consist of approximately 40 mountain ranges between the Colorado Plateau (northern Arizona) and the Sierra Madre (Mexico), separated by 15 to 25 kilometre-wide arid valleys (Figure 1.19) (López-Hoffman & Quijada-Mascareñas, 2012). They were created by geological extension splitting a bedrock plateau into a series of mountain ranges separated by basins filled with alluvium (Eastoe & Wright, 2019). They cover small areas but they are hotspots of biodiversity due to their location between several biogeographic regions, their steep elevation gradient and their diverse topography (Devender et al., 2013; Poulos & Camp, 2010). They are at the crossroads between the Rocky Mountains and the Sierra Madre, between temperate and subtropical climates, and between the Sonoran and Chihuahuan desert. The range of species living in each of these areas overlap here (López-Hoffman & Quijada-Mascareñas, 2012). The climate gradient along elevation, with temperature decreasing and precipitation increasing with elevation (Figure 1.9 and 1.19), forms a succession of ecological niches, each with their own vegetation and wildlife. Aspect, slope and water permanence in the canyons also diversify the general biodiversity by providing a mosaic of environmental conditions (Burnett et al., 1998).

Figure 1.19: Madrean Archipelago, a scattering of isolated mountain ranges between the Colorado Plateau and the Sierra Madre.

The Huachuca Mountains are located at the Mexican border, west of the San Pedro River and south of the Babocomari River (Figure 1.10). Miller Peak, the highest summit, stands at 2884 m.a.s.l. and ~1,500 m above the surrounding valley. The mountain range is structured in a single southeast-northwest ridge approximately 40 km long and 7 km wide, altered by faulting and erosion

(Brady & Bonham, 1976; Wallmo, 1955). The Huachuca Mountains are made of bedrock of various permeability, with weathered mudstone and limestone intersected with fissures at the top, cliffs of quartzite, and impervious granite at the bottom (Figure 1.20). Several faults are also dividing the mountain range (Wallmo, 1955).

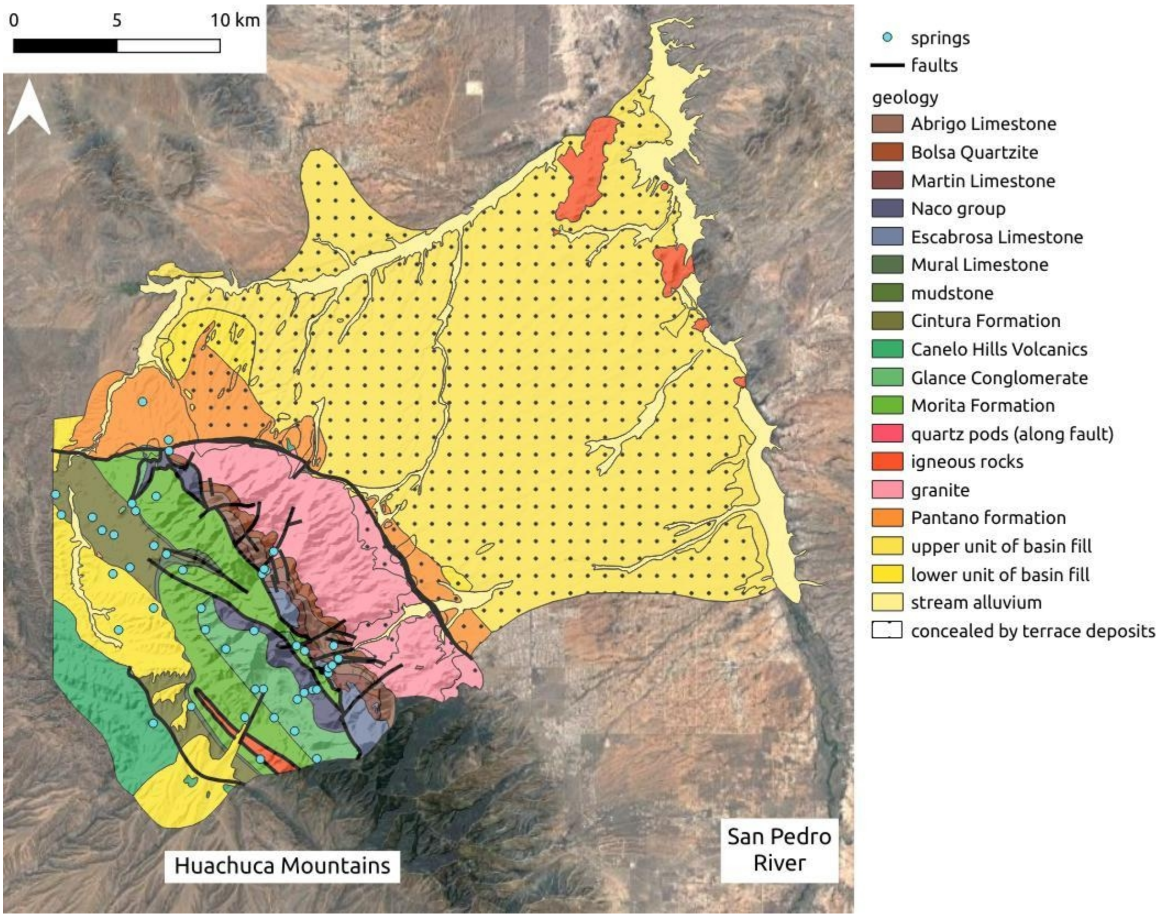

Figure 1.20: Geology under the Huachuca Mountains and the San Pedro valley, springs and faults. From Brown, 1966.

Streams of the Huachuca Mountains are non-perennial (Figure 1.21) and spatially intermittent and characterized by an alternation of wet and dry reaches, and surface water is mainly found in scattered perennial pools for most of the year (Bogan, Boersma, et al., 2013; Bogan et al., 2015; J. Olden & Lytle, 2015; Stromberg et al., 2013). At high elevation, streams are on top of weathered bedrock and water infiltrates in cracks and fissures, filling a shallow water table, and flowing downstream along a network of interstices in the rock. This underground water is pushed to the surface again and forms springs when its path is blocked by impervious rock formations (Brown et al., 1966; Levick et al., 2015). These springs can be permanent or temporary, maintaining intermittent or perennial pools during the dry spring and autumn (Brown et al., 1966). Water from springs form short flowing reaches until evaporation, infiltration losses and vegetation transpiration result in the streambed drying downstream. Reaches not supported by springs are ephemeral and flow only in response to rainfall and snowmelt, often flooding during the monsoon storms (Bogan et al., 2015; Stromberg et al., 2015). Stream channel morphology is characterized by cascade and bedrock reaches, morphing into step-pool, plane bed and pool-riffle in flatter channels downstream. Upstream, streambeds mainly comprise large cobbles and boulders, transitioning to primarily sand and gravel downstream (Jaeger & Olden, 2012). In several areas, calcium carbonate dissolved in the water when flowing in limestone underground is deposited once the water is at the surface again. Calcium carbonate has armoured the channel bed and contributed to a travertine step-pool morphology in some reaches (Jaeger & Olden, 2012). Several wetlands can be found along the canyons, when local geology and morphology allow the formation of a flatter area supplied by a spring (Figure 1.22).

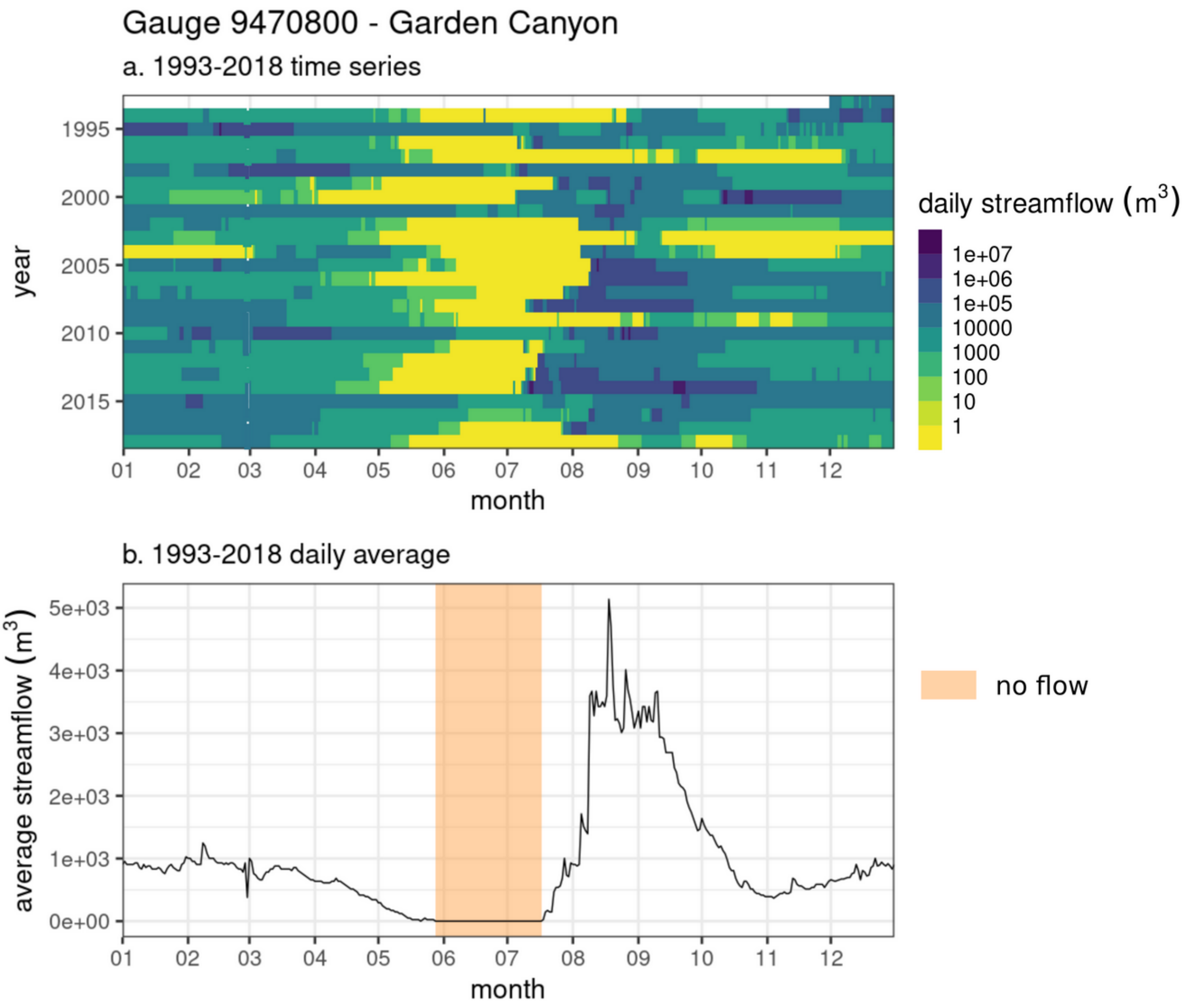

Figure 1.21: Daily streamflow for gauge 9470800, in Garden Canyon. Heatmap of daily time series from 1993 to 2018, with no flow periods in yellow (a) and daily average for the 1993-2018 period, no flow periods in orange (b). Gauge location is shown in Figure 3.1.

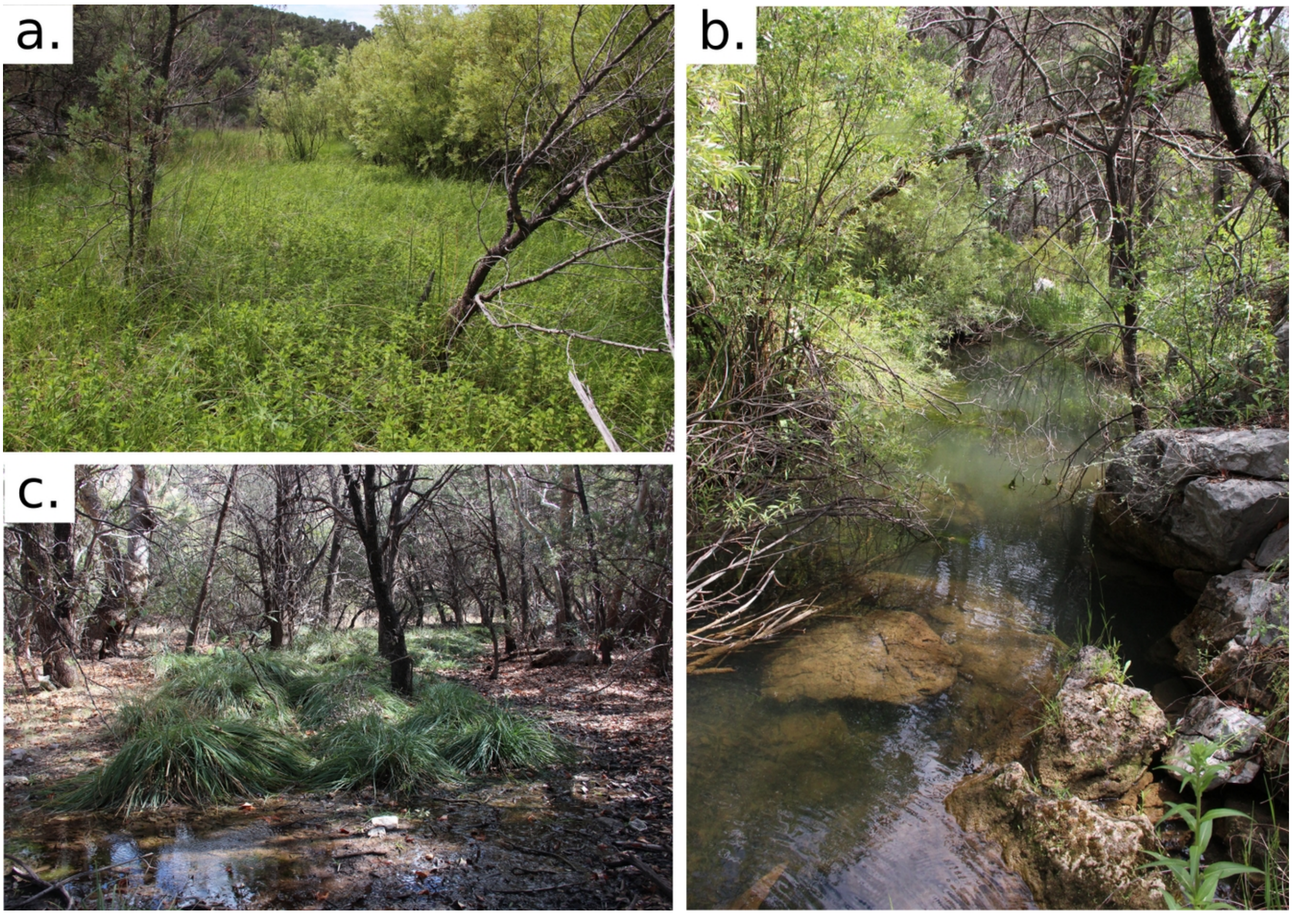

Figure 1.22: Wetlands along Garden and Huachuca Canyons. Headwaters (a), high elevation (b) and low elevation (c).

Each species is more abundant where environmental conditions are (theoretically) optimal for growth and reproduction (Brady & Bonham, 1976). Vegetation communities (a group of species occurring together) are layered along the elevation gradient, with species at higher elevation needing more water and colder temperatures. The dominant genus of trees is oak (*quercus sp.*) with Arizona white oak (*Quercus arizonica*), Silverleaf oak (*Quercus hypoleucoides*) and Emory oak (*Quercus emoryi*) as the main species (H. G. Shaw, 1999). These species can be found in the Huachuca Mountains spread across the elevation range. Likewise, alligator juniper (*Juniperus deppeana*) is also found across the Huachuca Mountains. The valley is covered in semi-arid mixed grassland/scrubland, which turns to grasslands on the foothills of the Huachuca Mountains. There is a succession of vegetation communities, usually with the dominance of one key species in each community. As elevation increases, vegetation goes from mixed grassland and evergreen trees (oaks, including Mexican blue oak, *Quercus oblongifolia*), followed by dense forests of evergreen oaks. Conifer forests with oaks mixed in cover the peaks (Arizona pine (*Pinus arizonica*), ponderosa pine (*pinus ponderosa*), pinyon pine (*Pinus discolor*) and douglas fir (*Pseudotsuga menziesii*)) (Figure 1.23) (H. G. Shaw, 1999; Wallmo, 1955). In areas impacted by fire, vegetation succession (the succession of species after a disturbance, from pioneering species to stable community) can temporarily alter the elevation-controlled organization of vegetation (Brady & Bonham, 1976). In the canyons, riparian forest species composition is largely dependent on elevation, the presence of perennial water (with higher flow permanence, water has more time to infiltrate in the sediment and be available for vegetation), soil moisture and local topography (which control drainage and, as a consequence, soil moisture) (Jaeger & Olden, 2012; Stromberg et al., 2015; Urban et al., 2000). In drier areas, streamside vegetation is similar to upland species, but highly diverse riparian vegetation (mainly Arizona sycamore (*Platanus wrightii*), bigtooth maple (*Acer grandidentatum*), velvet ash (*Fraxinus velutina*), Arizona walnut (*Juglans major*)) can be found along wetter reaches ((Levick et al., 2015; H. G. Shaw, 1999). Species richness of riparian communities is higher than upland areas (Poulos & Camp, 2010). In the foothills and valley, riparian vegetation is mainly mesquite (*Prosopis velutina*) and grasses.

Figure 1.23: Vegetation communities along the elevation gradient of the Huachuca Mountains.

Most wildlife relies on perennial pools found in the canyons of the Huachuca Mountains for water and moisture during the dry season. These streams are also home to many moisture-dependent invertebrates and amphibians. Bogan et al. (2015) recorded over 400 aquatic invertebrate taxa across the Madrean Sky Islands. Perennial and intermittent reaches support distinct aquatic insect communities, with species found in intermittent reaches adapted to regular cycles of wetting and drying and are able to recover quickly after a drought (Bogan, Boersma, et al., 2013; Bogan et al., 2015). Flow permanence is an important control on diversity, with high flow permanence is associated with increased species richness and functional diversity (Schriever et al., 2015). Birds also rely on riparian vegetation. Powell and Steidl (2015) found that species richness of songbirds increased with volume of understory vegetation and density of Arizona sycamore, which is key to maintaining biodiversity in these areas. Riparian vegetation covered <0.5% of the landscape, but supported the majority of the region's vertebrate diversity. Species with low dispersal abilities and those which are moisture dependent cannot cross between mountain ranges because they would not survive the hot and dry valleys, making small, isolated populations vulnerable to habitat destruction and modification (Coe et al., 2012).

With climate change, the Madrean Sky Islands are at risk of higher temperatures, increased fire activity and lower snowpack and runoff (Archer, 2008). Vegetation is expected to climb along the elevation gradient (Section 1.2.4), where possible. Semi-arid grassland cover will increase and the conifer forests that cover the mountain peaks will decline (Coe et al., 2012; Monroy-Gamboa et al., 2021). Barton and Poulos (2018) predict a transition from pine-oak forest to oak-dominated communities due to the increase in fire and drought frequency. Streams and riparian vegetation’s role as a cool refuge will become even more critical and might turn into the only favourable habitat to vegetation and wildlife needing a cooler climate (Monroy-Gamboa et al., 2021). An increase in evaporation due to higher temperatures and a decline in local groundwater recharge due to a diminished snowpack might also drive a loss of permanent pools along the canyons, putting all species using this habitat at risk, especially species with low dispersal ability (Coe et al., 2012).

1.4.4 San Pedro valley

In between the Huachuca Mountains and the San Pedro River lies a flat valley comprised of sedimentary terrace deposits, mainly composed of gravel, sand and clay, and lying on top of upper (sand and gravel) and lower (silts and clays) basin fill (Brown et al., 1966; Goodrich et al., 2018; Wahi et al., 2008). The regional water table is filled through mountain front recharge: water travels down the mountain canyons before reaching the permeable valley floor and percolates to reach the basin fill aquifer (Ajami et al., 2012; Eastoe & Wright, 2019). Water slowly moves in the aquifer, following the elevation gradient and reaches the San Pedro River (Brown et al., 1966; Wahi et al., 2008). Rainfall runoff is gathered in highly ephemeral washes over permeable alluvium that also act as recharge zones for the regional aquifer through transmission losses (Goodrich et al., 2018; Kennedy & Gungle, 2010). These washes are the continuity of canyon streams starting in the Huachuca Mountains, with water flowing from the mountains toward the San Pedro River or the Babocomari River. Continuous flow in these washes only reach the San Pedro River as overland flow during the strongest monsoon storms (Goodrich et al., 2018). The spatiotemporal distribution of flow over the valley is more uniform during the winter events, and more erratic during the monsoon storms, due to their localized and intense nature (Gungle, 2006). Vegetation cover mainly consists of a mix of grass and desert scrub, with mesquite being the main riparian vegetation along the ephemeral washes (Figure 1.24) (Devender et al., 2013; Gungle, 2006). Mesquite is well adapted to these ephemeral streams thanks to its high plasticity in canopy density, and root depth and distribution (Stromberg et al., 2013). The regional aquifer under the valley that provides water to the San Pedro River is declining due to heavy pumping from Fort Huachuca and Sierra Vista, and appears to induce decreased streamflow (Stromberg & Tellman, 2012, p. 295).

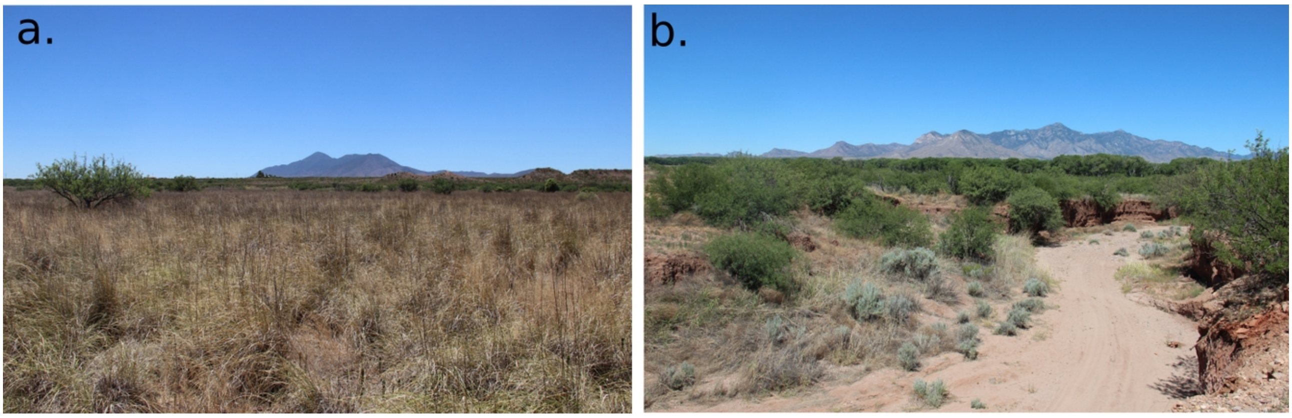

Figure 1.24: Vegetation communities in the San Pedro valley, with a mix of grasses and scrub~~s~~ (a) and mesquite along an ephemeral wash (b).

1.4.5 Fort Huachuca

Fort Huachuca is an U.S. Army installation established in 1877. It is now a research, development and testing ground for intelligence, electronic warfare, and information systems, as well as an unmanned aerial vehicle training facility. The installation covers 327 km^2, including 270 km^2 of natural land used for outdoor training and testing in wilderness. It is also an outdoor recreation area for hikers, hunters and birders. Fort Huachuca is one of the main land owners and managers of the Huachuca Mountains, the rest being included in the Coronado National Forest, managed by the U.S. Forest Service. It also covers part of the valley and reaches the SPRNCA (Figure 1.25) (Section 1.4.2). The Environmental and Natural Resources Division (ENRD) is the department in charge of natural resources management and conservation of protected species. Natural resources management includes groundwater, watersheds, fire, threatened and endangered species, nature-based recreation, and archaeological resources, with the goal of enhancing biodiversity, protecting listed species and rare habitats, improving quality of life, and supporting the military mission (USAIC & Fort Huachuca, 2002). Grassland with mesquite and oak forests are the main vegetation communities on the installation. There is also 0.3 km^2 of wetlands and 3 km^2 of riparian habitat (~ 1% of the installation's area). Riparian habitat is mostly found in Garden and Huachuca Canyons (ENRD Fort Huachuca, 2021).

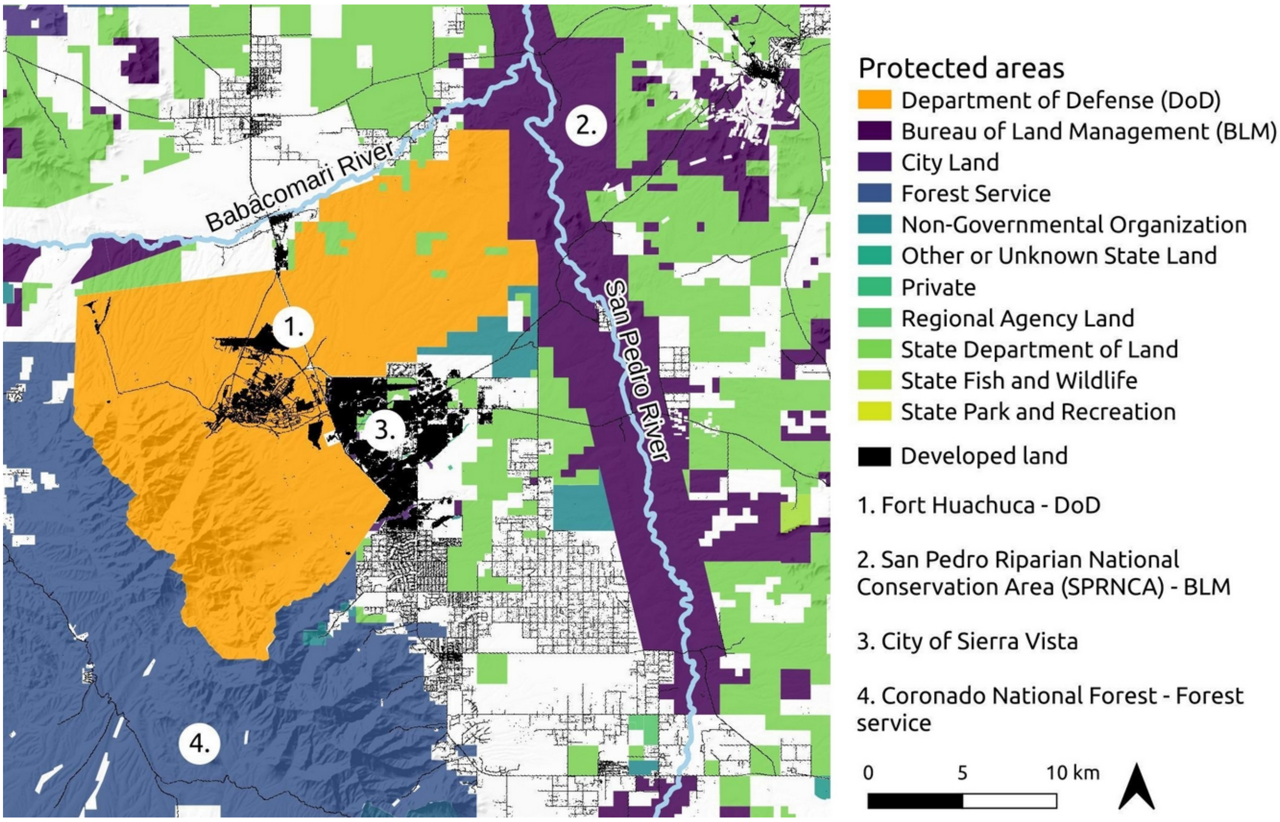

Figure 1.25: Protected areas, grouped by organization, and extent of developed land, with 1. Fort Huachuca, 2. SPRNCA, 3. city of Sierra Vista, 4. Coronado National Forest. Data from the U.S. Geological Survey, https://www.usgs.gov/programs/gap-analysis-project/science/pad-us-data-download.

Water is a key resource in the area, with a lot of stakeholders, including Fort Huachuca, Sierra Vista, Huachuca City and the SRPNCA. Many reports have studied water distribution in the area in the context of human use (Brown et al., 1966; Gungle et al., 2019; Pool & Coes, 1999). Fort Huachuca has about 7.2 km of perennial stream, mostly in Garden Canyon (5.6 km) and Huachuca Canyon (1.2 km). Small perennial reaches also occur in McClure and Blacktail canyons (McFarland, 2014). Historically, drinking water for Fort Huachuca was coming from Garden, McClure and Huachuca Canyons (H. G. Shaw, 1999). Pipes, spring boxes and concrete dams are still in place along these streams. Today, domestic water for the Fort comes from wells in the San Pedro valley (McFarland, 2014; H. G. Shaw, 1999). Water conservation constitutes part of the Fort's compliance with the ESA, with the goal of limiting draw down of the regional aquifer providing water to the San Pedro River (Sections 1.3 & 1.4.2). Fort Huachuca is a member of the Upper San Pedro Partnership, a consortium of 21 local, state, and federal agencies and private organizations (Luster, 2002). Their goal is to establish a collaborative water use management program in the Sierra Vista watershed to protect the San Pedro River and the SPRNCA by reaching a sustainable yield of the regional aquifer. The installation has progressively reduced its water use over the last years, even though its population has increased, with a decrease in groundwater withdrawals of over 65 percent between 1993 and 2014 (Luster, 2002; McFarland, 2014). Actions have been taken to reduce water use across the installation, for example through communication to raising awareness, demolition of unused buildings, and a policy on water conservation which regulates water use by limiting watering of residential areas, landscaping with drought tolerant plants, using water-efficient fixtures and monitoring for leaks in plumbing (Department of the Army, 2021). The installation has also purchased land close to the San Pedro and Babocomari Rivers to protect them from agricultural water pumping (Luster, 2002). Treated effluents are used to water outdoor sports complex and parade fields. Effluent and stormwater recharge basins have also been built to support water infiltration and groundwater recharge (Luster, 2002).

Fort Huachuca is responsible for the conservation of the high biodiversity found across the Huachuca Mountains and the valley. Part of its mission is to safeguard federally-listed species and their critical habitat (Table 1.3), in compliance with the ESA (Section 1.3.1). Fort Huachuca has also developed conservation actions for species that are candidates or proposed for listing under the ESA, and State species of concern, as declared by the Arizona Game and Fish Department. Recently delisted species such as the lesser long-nosed bat (*Leptonycteris yerbabuenae*) are also still managed for conservation (ENRD

Fort Huachuca, 2021). The Fort must account for direct (within the installation’s boundaries) and indirect (repercussions of groundwater pumping on flow permanence in the San Pedro River) impacts on protected species (McFarland, 2014). Fort Huachuca and the SPRNCA are home to 20 species protected under the ESA, including 13 species using non-perennial streams or their riparian corridors (Table 1.3). Species such as the Huachuca water umbel, Huachuca springsnail or Sonora tiger salamander need moist environments and surface water for at least part of their life cycle. When present on Fort Huachuca, these species are mainly found in Garden and Huachuca Canyons. Other species, mainly birds such as the yellow-Billed cuckoo or the southwestern willow flycatcher, use the riparian forest for nesting and the invertebrates found along the stream as a food source (ENRD Fort Huachuca, 2021). The main climate change-related risks for these species are loss of habitat, an increase in occurrence of extreme events, decreased streamflow and change in flow timing, and an increase in the frequency and extent of wildfires (McFarland, 2014).

Table 1.3: special status species for Fort Huachuca and the SPRNCA, with the species dependent on streams and riparian vegetation in blue (adapted from ENRD Fort Huachuca, 2021). CCA: Arizona Wildlife Species of Greatest Conservation, occurs on the Fort and is managed under a Candidate Conservation Agreement. SPRNCA: San Pedro Riparian National Conservation Area. An horizontal dash (-) indicates that the species was not observed.

Species	**Status**	**Occurrence**	
		Fort Huachuca	**SPRNCA**
Plants			
Huachuca water umbel (*Lilaeopsis schaffneriana ssp. recurva*)	Endangered	Present	Present
Lemmon fleabane (*Erigeron lemmonii*)	CCA	Present	-
Beardless Chinchweed (*Pectis imberbis*)	Proposed Endangered	Potential habitat	Potential habitat
Bartram's stonecrop (*Graptopetalum bartramii*)	Proposed Threatened	Potential habitat	Potential habitat
Arizona Eryngo (*Eryngium sparganophyllum*)	Endangered	Potential habitat	Present
Invertebrates			
Huachuca springsnail (*Pyrgulopsis thompsoni*)	CCA	Present	Potential habitat
Birds			
Mexican spotted owl (*Strix occidentalis lucida*)	Threatened	Present	-
Southwestern willow flycatcher (*Empidonax traillii extimus*)	Endangered	-	Present

Species	**Status**	**Occurrence**	
		Fort Huachuca	**SPRNCA**
Yellow-billed cuckoo (*Coccyzus americanus*)	Threatened	Present	Present
Mammals			
Lesser long-nosed bat (*Leptonycteris yerbabuenae*)	Delisted	Present	-
Jaguar (*Panthera onca*)	Endangered	Present	Potential habitat
Ocelot (*Felis pardalis*)	Endangered	Present	Potential habitat
Amphibians			
Sonora tiger salamander (*Ambystoma mavortium stebbinsi*)	Endangered	Present	-
Chiricahua leopard frog (*Lithobates chiricahuensis*)	Threatened	Present	Potential habitat
Reptiles			
Northern Mexican gartersnake (*Thamnophis eques megalops*)	Threatened	Potential habitat	-
Fish			
Gila chub (*Gila intermedia*)	Endangered	-	Potential habitat
Spikedace (*Meda fulgida*)	Endangered	-	Potential habitat

Species	**Status**	**Occurrence**	
		Fort Huachuca	**SPRNCA**
Gila topminnow (*Poeciliopsis occidentalis occidentalis*)	Endangered	-	Potential habitat
Desert pupfish (*Cyprinodon macularius*)	Endangered	-	Potential habitat
Loach minnow (*Tiaroga cobitis*)	Endangered	-	Potential habitat

Management for endangered species involves habitat protection, surveying and monitoring, research, and integration with the military mission and recreation (species management can put restrictions on training, testing and recreation activities) (USAIC & Fort Huachuca, 2002). Conservation actions have included removing old fences to improve wildlife corridors, preventing access to caves, old mines and critical habitats, controlling invasive species (such as salt cedar and watercress), protecting critical habitat from human impact with barriers and limitations to training and educating personnel and visitors (ENRD Fort Huachuca, 2021). Information signs are set up at canyon trailheads, caves and riparian habitats. Training limitations have also been implemented: vehicles have to stay on roads and trails and any alterations of the streambed in mountain streams is prohibited. In agave management areas, delineated for the conservation of the lesser long-nosed bat for which agave is an important food source, night operations, and firing of blanks or pyrotechnics are prohibited when the bat is present (McFarland, 2014). Fort Huachuca also partners with conservation organizations such as the Arizona Game and Fish Department, U.S. Fish and Wildlife Service, U.S. Forest Service or The Nature Conservancy. These organizations provide support for monitoring populations and planning of conservation measures. Fort Huachuca is also the first Army installation to use conservation easements to protect T&E species of the San

Pedro River. The Nature Conservancy was in charge of identifying suitable land and acquiring it. Suitable land included areas that could connect the SPRNCA to other protected habitats, and land that might be used for activities contributing to groundwater deficit, such as agriculture and urban development. Fort Huachuca provided the funds for the purchase and the land was transferred to the Bureau of Land Management for management (Luster, 2002). Fort Huachuca spends approximately $1.5M annually to implement proactive conservation measures for protected species (USAIC & Fort Huachuca, 2002).

Climate change trends for Fort Huachuca are in line with the rest of the Southwest (Section 1.4.1), with risks of increased temperatures and rainfall distribution variability, leading to land cover changes (loss of pine-oak forest and gain of grassland in the Huachuca Mountains), fires and floods (Table 1.4) (Garfin et al., 2021). Post-fire flooding is an important concern due to the increased danger of erosion and sediment transport. Fuel reduction programs that are already in place might need to be extended (Garfin et al., 2021). Water supply to the Fort and the surrounding cities is also at risk if evapotranspiration increases and water evaporates before having a chance to infiltrate and recharge the regional water table. Climate change can also lead to habitat loss for T&E species, for example through reduction of perennial reaches due to increased evaporation and changes in rainfall distribution, sediment filling wetlands due to post-fires floods, or disappearance of pine-oak forests. Many species, such as the Huachuca Water Umbel or the Sonora tiger salamander, are also vulnerable to the introduction of invasive species more competitive in the new climate conditions and new predators (ENRD Fort Huachuca, 2021).

Table 1.4: Risks and consequences of climate change for Fort Huachuca (Garfin et al., 2021).

Climate drivers	**Processes affected**	**Operational and mission consequences**
Warmer and drier conditions Climate-driven vegetation change, altered landscape fuels	Water scarcity Increased risk of wildfires	Competition for resources increases with neighbouring communities and private sector water users; long-term water management challenges increase; Endangered Species Act-related impacts Training grounds and buildings damaged or destroyed; training readiness affected; potential short and long-term impacts on mission and budget
Warmer conditions and more variable precipitation	Increased ecosystem stress, including for threatened and endangered species Potentially increased invasion of non-native species, altering ecosystem function and fire regimes	Land management requirements and constraints are more complex Potential reductions in biodiversity, increased cost, and damages associated with wildfire
More intense precipitation	Inundation and erosion of transportation routes; severe post-fire flooding	Emergency preparedness and evacuation procedures compromised; property damage
More frequent extreme high temperatures and extended heat waves	Heat stress for personnel engaged in outdoor activities	Increased rotation of personnel required; lost training time

1.5 Main methods

Riparian ecosystems within the Southwestern USA have been greatly altered by management of land and water resources (D. E. Busch & Smith, 1995; Patten, 1998). They are also sensitive to climate change as climate is one of the factors controlling water availability and transpiration demand (Shafroth et al., 2000). Their monitoring allows us to understand how this ecosystem evolves and why. Several methods and approaches can be used to study non-perennial streams and their riparian forests. For this project, I purposefully selected methods that could be applied to a wide diversity of contexts and needs while being relatively easy to implement, understand and interpret. The tools and equipment used needed to be accessible (free satellite imagery available worldwide, cheaper sensors easy to deploy) and the methods needed to be replicable in other locations and landscapes.

1.5.1 Flow permanence

In non-perennial streams, knowing where and when surface water and soil moisture are is critical for conservation, as vegetation and wildlife distribution and density are tightly linked to water availability (Section 1.2.2). There are several ways to measure the spatiotemporal variability of flow presence, and several metrics and classification systems for categorizing flow permanence (Levick et al., 2008).

1.5.1.1 Overall information

Flow permanence, or intermittency, can be measured as the number of dry days, or number of flow days, in a set time period (Leenhouts et al., 2006). Other metrics of non-perennial flow are the length of dry periods, the date of the first drying event, daily, seasonal, and annual flow variability, and rates of change (J. D. Olden et al., 2012; Zipper et al., 2021). Flood frequency, duration and magnitude are also important metrics of flow regime as floodplain inundation provides soil moisture to vegetation. Stream reaches can then be sorted into classes to identify streams or reaches with similar characteristics that might react to the same environmental controls or serve as habitats to similar species assemblages (Levick et al., 2015). Classification can be established by focusing on hydrologic variables that describes the flow regime such as number of dry days or seasonal distribution of flow (summer or winter flow for example), or by focusing on environmental controls and water source (snowmelt, rain or groundwater for example) (Eng et al., 2015; Kennard et al., 2010; J. D. Olden et al., 2012). For example, Poff and Ward (1989) used overall flow variability, flood regime patterns, and extent of intermittency to sort streams into nine types, depending on their temporal patterns and water source.

Given the high variability of flow in non-perennial streams, time-series of streamflow at multiple locations along a stream are imperative to understand longitudinal flow continuity along the stream, and how connectivity evolves through time. Due to the high variability, and sometimes short duration, of rainfall in drylands, paired with high transmission losses and atmospheric demand, ephemeral streams can flow for very short times (Section 1.2.1). The temporal resolution of any flow recording method must be able to keep pace with these fast variations of water presence in the streambed.

In the U.S., streamflow gauges record 15-minute streamflow and some gauges can be found in non-perennial streams such as in Garden or Huachuca Canyons (Section 1.4.3). They have the advantage of measuring the frequency, duration and magnitude of flow events. But smaller streams may be ungauged, or there will only be one gauge, which is not enough to reconstruct dynamics of flow connectivity (Adams et al., 2006). Streamflow gauges are also mainly set in the more perennial reaches. Satellite or aerial photography analysis can also be used (Galea et al., 2019), but non-perennial streams, especially headwater streams, are often too small and hidden in vegetation for flow to be visible from above. Vegetation mapping can also be used to identify the streamflow patterns needed to support specific communities (Manning et al., 2020), but, for this to work, it is necessary to establish a strong relationship between vegetation distribution and flow, as done in Lite (2003) and Stromberg et al (2006). Human observation can be carried out (for example along the San Pedro for mapping of surface flow during the dry season, led by The Nature Conservancy) but these surveys are labour intensive and as such, are usually only carried periodically and over smaller extents (Larned et al., 2010). Their accuracy is limited by subjectivity of interpretation and the frequency of site visits (Costigan et al., 2016). Temperature sensors can also be used, paired with air temperature sensors at the same location, as water temperature is usually lower than air temperature, but these sensors require a complex interpretation (Blasch et al., 2002; Goulsbra et al., 2009).

1.5.1.2 Electrical conductivity sensors

Electrical conductivity (EC) sensors are increasingly used to accurately map spatiotemporal distribution and dynamics of flow in non-perennial streams (Assendelft & van Meerveld, 2019; Blasch et al., 2002; Costigan et al., 2016). By measuring the variations of electrical conductivity (or the opposite, electrical resistivity) between two electrodes in the stream, they are able to detect onset and cessation of flow, and measure timing, duration and frequency of flow (Blasch et al., 2002; Goulsbra et al., 2009). Setting several of these sensors along a stream grants information on advancing and retreating water fronts and longitudinal connectivity (Costigan et al., 2016; Jaeger & Olden, 2012).

When placed above the streambed, the sensor is able to distinguish between the presence or absence of water, and when shallowly buried in the streambed, the sensor is able to record the difference in conductivity between surface water, soil moisture in the sediment and dry sediment (Blasch et al., 2002; Goulsbra et al., 2009; Jaeger & Olden, 2012). The records are easy to interpret, especially after calibration, as water temperature and mineral concentration can have an impact on water conductivity (Blasch et al., 2002; Chapin et al., 2014). Electrical conductivity of air is almost zero. Recorded value increases with water saturation and decreases when the sediment or air dries up (Adams et al., 2006; Assendelft & van Meerveld, 2019). The rate of conductivity change is used to identify flow events, as conductivity is steady and low in a dry river, and increases abruptly during onset of flow (Blasch et al., 2002; Jaeger & Olden, 2012) (Figure 1.26). Once the conductivity threshold between dry and wet state, these records provide time series of water presence, which can be translated to flow permanence metrics, such as the number of dry days, the length of dry events and the date of streamflow onset.

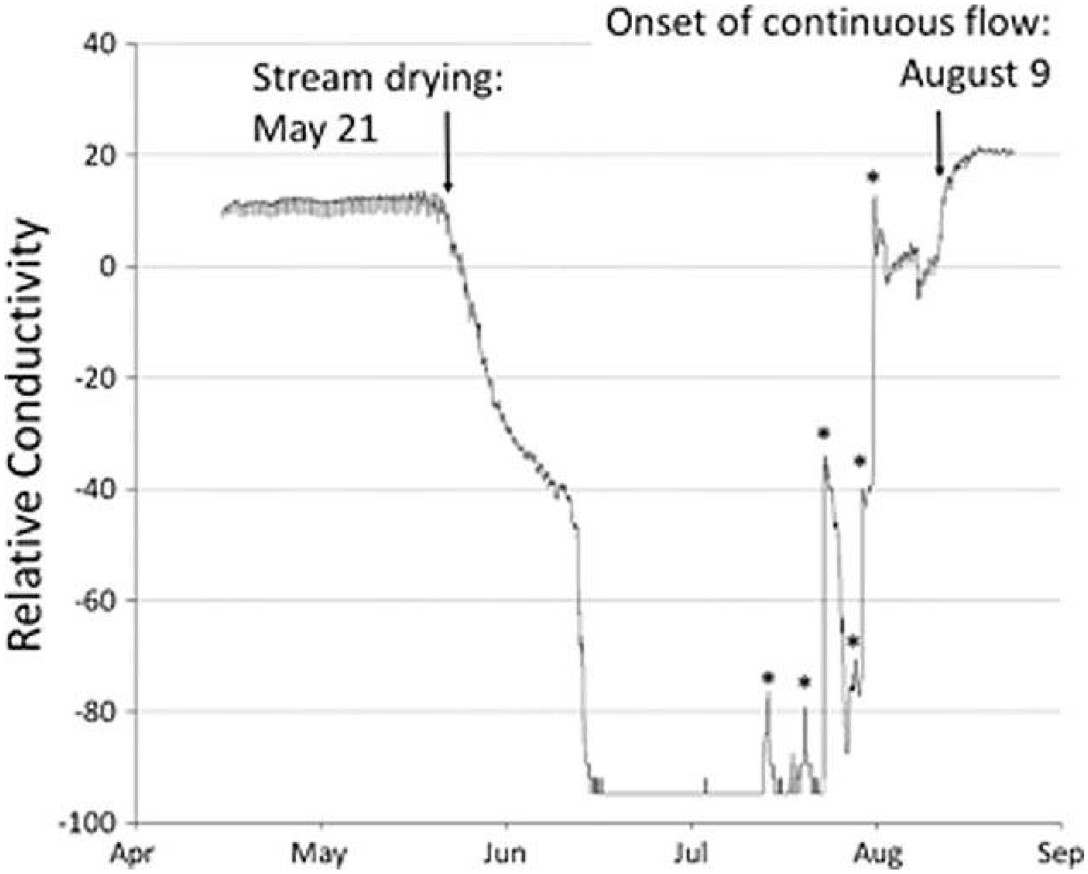

Figure 1.26: Electrical resistance sensor record in an intermittent reach in upper Huachuca Canyon (H2). Abrupt increases in relative conductivity values indicate onset of surface flow. Sustained declines in relative conductivity values indicate stream drying. * represent periods of short duration <24 h streamflow from convective thunderstorm precipitation during the North American Monsoon (Jaeger et al., 2012).

EC sensors are able to record flow permanence at high accuracy and high spatiotemporal resolution (Assendelft & van Meerveld, 2019; Blasch et al., 2002). Due to their low cost and small size, they can be set up anywhere along the stream. Adams et al (2006) used EC sensors to measure flow along nearly vertical rock faces with diffuse flow (depths between 0.3 and 1.0 cm). They can record data every 10 or 15-minutes during several months or years and require limited fieldwork as they simply need to be set up and collected at the end of the battery life, though regular site visits are usually done to control the integrity of the sensors (Adams et al., 2006; Chapin et al., 2014; Jaeger & Olden, 2012).

Sensors can be set up on top or in the streambed. Sensor location must be chosen depending on the study's goal. Non-perennial streams are composed of a succession of pools and riffles, sensors in pools will record the evolution of refuge pools, while sensors on riffles are more likely to record streamflow onset and cessation. If the sensor is buried in the sediment, the lag of the sediment slowly getting wet and drying up can skew results for the timing of onset and cessation of flow, and this time lag increases with depth. The lag at onset of flow is a function of the hydraulic properties of the sediments and the antecedent water content. Lag at flow cessation is controlled by hydraulic conductivity, and evapotranspiration demand (Blasch et al., 2002). The offset depends on the sediment, with clay and fine-grained mediums drying slower and keeping high conductivity value for longer (Blasch et al., 2002). In high energy streams, sensors are protected in housing (often a piece of pipe) and are secured to boulders, tree roots, concrete structures or metal poles so they won't be carried downstream during floods (Kaplan et al., 2019). If they are simply fixed on top of the streambed, sensors must be at the deepest point of the channel to avoid missing flow events by being too high, but they run the risk of being buried in sediments during floods (Blasch et al., 2002; Kaplan et al., 2019).

A common method to record EC in non-perennial streams is to use temperature or light sensors modified with electrodes. The details for these modifications are in Blasch et al. (2002), who used temperature sensors, and Chapin et al. (2014) who used light-temperature sensors. These sensors are cheap, small and they are built to be deployed in streams and left in place with an onboard datalogger and battery in a waterproof casing. The temperature or light circuitry is repurposed to record EC response. This method has been adapted in the following studies on flow permanence (Adams et al., 2006; Goulsbra et al., 2009; Jaeger & Olden, 2012; Warix et al., 2021).

The flow permanence dataset used in Chapter 3 of this thesis comes from a total of 37 electrical conductivity (EC) sensors that were installed along 10 streams of the Huachuca Mountains and operated between 2010 and 2014 for a project on hydrologic regimes of non-perennial streams and their ecological integrity led by Levick et al. (2015). Part of the dataset from these sensors was analysed by Jaeger and Olden (2012), and the raw data used in this thesis was provided by Kristen Jaeger. The loggers used are the TidBit v2 temperature loggers (Onset Computer Corporation). They have been modified after the methods described by Blasch et al. (2002) and Gungle (2006). The original thermistor has been replaced by copper wires soldered to the sensor circuit board, the sensors have been tied inside a perforated PVC tube, buried in the streambed (<10 cm) and secured to a nearby tree or rock. The sensors measured relative conductivity as a proxy for streamflow presence every 15 minutes (Jaeger and Olden, 2012).

1.5.2 Remote sensing for vegetation

1.5.2.1 Earth observation satellites

Remote sensing is the use of sensors that are not in direct contact with the object they are measuring. Remote sensing is an important and widely used tool for studying riparian vegetation. The most common data used include multispectral imagery from planes and low spatial resolution satellite (pixel size over 10 m), while RADAR is the imagery least used (Huylenbroeck et al., 2020). More recently, LiDAR and drone imagery have been used as they can provide higher resolution images and additional information on forest structure. The choice of data depends on the project’s goals and the extent of the study site. Often, a higher spatial resolution means a smaller coverage, although very high spatial resolution satellites are getting more common. In this thesis, I chose to use medium-resolution satellite imagery from the Landsat and Sentinel-2 programs. These datasets cover the planet, are freely available and analysis-ready images can be downloaded online, which makes them particularly useful for projects aimed at being reproducible across the world and accessible to any natural resources management organization. In my case, they also had adequate spatial and temporal resolution to map riparian vegetation dynamics.

Earth observation satellites (EOS), which are used to measure physical characteristic of the Earth’s surface, and more precisely passive EOS, that use the light emitted by the sun and reflected by the ground (in contrast, active sensors such as radars emit their own light source). They are equipped with a sensor recording in different points of the electromagnetic spectrum. They take several pictures of the same area in different bands, each band covering a limited area of the spectrum. Each object on the planet’s surface absorbs and reflects specific wavelengths in the electromagnetic spectrum. For example, vegetation strongly reflects infrared (wavelength around 0.8 µm). The satellite records the intensity of reflected infrared light, measured as reflectance (value between 0 and 1), and in the image produced, vegetation will be characterized by a high reflectance (Figure 1.27). Objects that absorb infrared will have a low reflectance value in the infrared band.

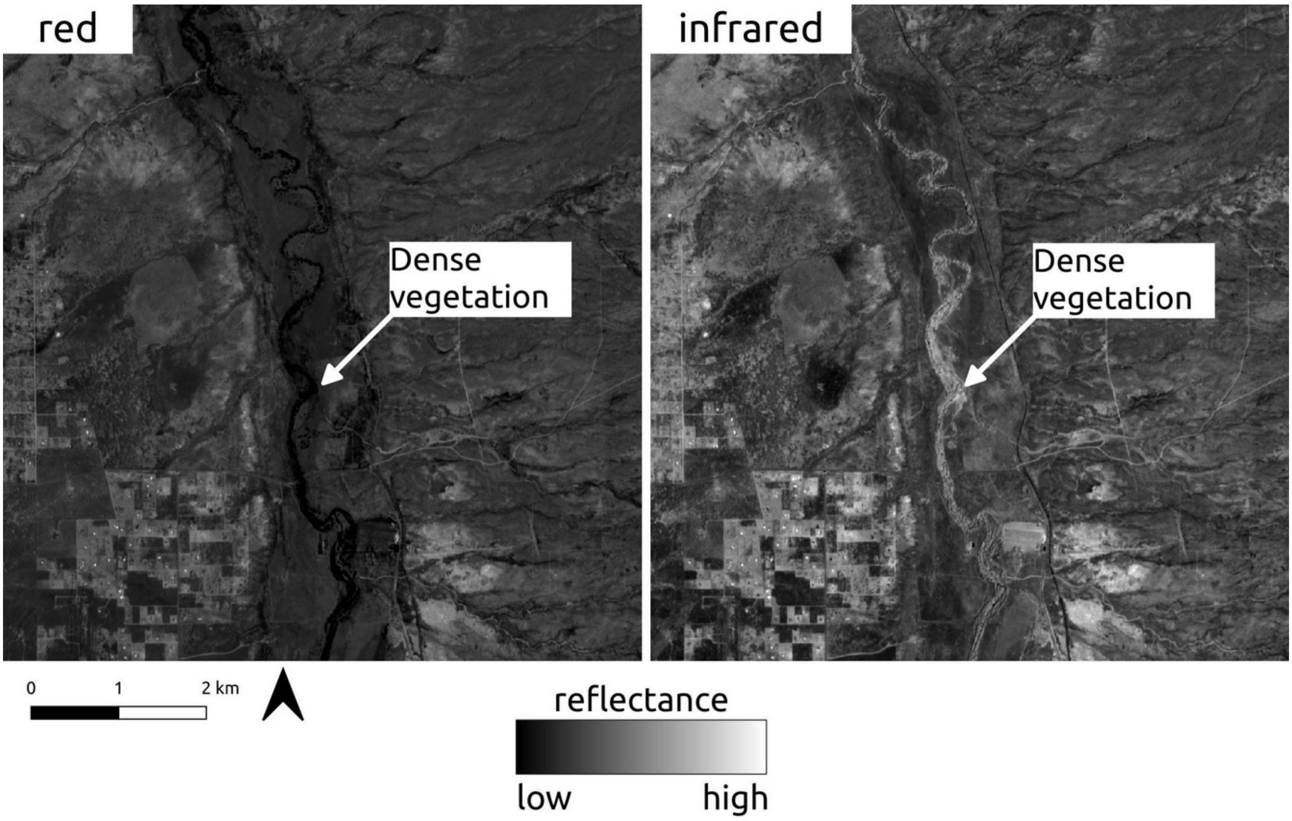

Figure 1.27: Reflectance of red and infrared bands (Sentinel-2) over the study site. Vegetation absorbs red (low values) and reflects infrared (higher values).

Earth observation satellites provide standardized data with a nearly global coverage of the planet. Satellite imagery is defined by its temporal, spectral and spatial resolution. The temporal resolution, or revisit time, is the frequency with which a same area is covered, the spectral resolution is how narrow the spectral bands are, and the spatial resolution is the size of a single pixel in the image (Kerr & Ostrovsky, 2003). For example, Sentinel-2 is characterized by a temporal resolution of 5 days, a spectral resolution between 15 and 175 nm, and a spatial resolution between 10 and 60 m depending on the band. Depending on the study's goal, a compromise has to be found between these three resolutions to find the satellite that is the best adapted. A high temporal resolution with a short revisit time is important for reconstructing time series, while a high spatial resolution with a small pixel size is essential to study small objects or ecosystems. The Landsat program, managed by the National Aeronautics and Space Administration (NASA) and the U.S. Geological Survey has been keeping a consistent record since 1972 and is still ongoing, with the recent launch of Landsat-9 in September 2021.

They are used to study a wide array of subjects, including environmental monitoring. They can help detect, identify, map and monitor ecosystem changes (Coppin et al., 2004). Long time series are critical to study long-term changes in land cover for example by permitting a consistent assessment with comparable data (figure 1.28). Analysis can cover wide extents, including areas inaccessible for fieldwork, and with regular new data (as often as every day for MODIS for example) (Kerr & Ostrovsky, 2003).

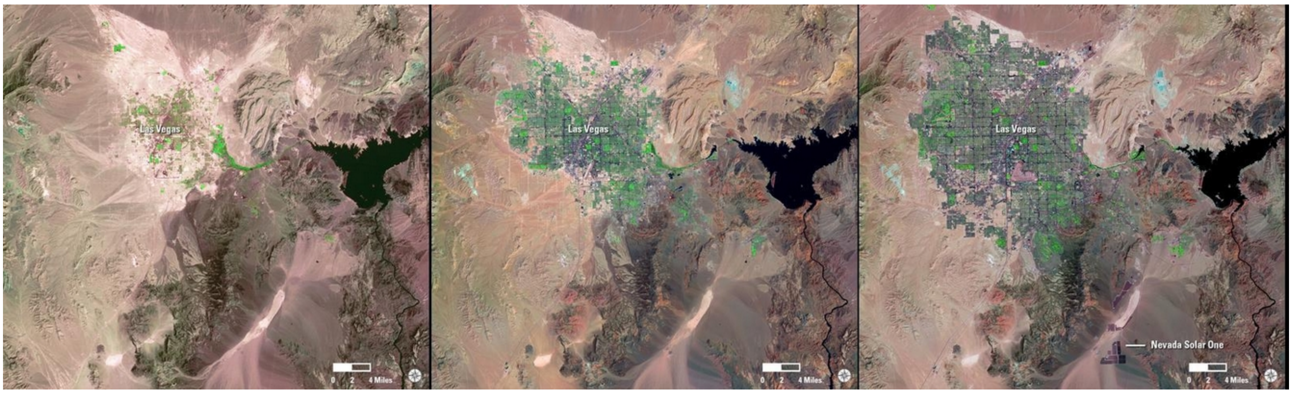

Figure 1.28: Urban growth in Las Vegas, Nevada, USA (Landsat). From left to right: 1972 – 1997 – 2018. Maps produced by the U.S. Geological Survey, available at: https://www.usgs.gov/media/images/urban-growth-las-vegas-nevada-usa.

1.5.2.2 Landsat and Sentinel

The Landsat program is a joint initiative of NASA and the U.S. Geological Survey (USGS) for Earth observation (Goetz, 2006). Started with the launch of Landsat-1 in 1972, it is still ongoing with Landsat-8 and 9 today. This long time series is often used to study land cover changes (Kerr & Ostrovsky, 2003; Klemas, 2014). The goal of this program is to provide a continuous consistent record, so the spectral, temporal and spatial resolutions have been kept the same across satellites, though recent sensors record in 11 bands, compared to the former seven (Table 1.5). The revisit time is 16 days and the pixel size is 30 m (except for thermal and panchromatic). I have used images from Landsat-5 (between 1984 and 2011) and Landsat-8 (from 2013 and still active). Landsat-6 never reached orbit, and Landsat-7 (1999-2022) was not used due to a defect in its images resulting in missing data over my study site.

Table 1.5: Spectral bands of Landsat 5 and Landsat 8. Bands highlighted in blue are used in this thesis.

Bands	**Wavelength (µm)**	**Spatial resolution (m)**
Landsat-5		
Band 1 – Blue	0.45-0.52	30
Band 2 – Green	0.52-0.6	30
Band 3 – Red	0.63-0.69	30
Band 4 – Near infrared (NIR)	0.76-0.9	30
Band 5 – Shortwave infrared (SWIR) 1	1.55-1.75	30
Band 6 – Thermal	10.40-12.50	120
Band 7 – SWIR 2	2.08-2.35	30
Landsat-8		
Band 1 – Coastal aerosol	0.43–0.45	30
Band 2 – Blue	0.45–0.51	30
Band 3 – Green	0.52–0.60	30
Band 4 – Red	0.63–0.68	30
Band 5 – NIR	0.84–0.88	30
Band 6 – SWIR 1	1.56–1.66	30

Bands	**Wavelength (µm)**	**Spatial resolution (m)**
Landsat-8		
Band 7 – SWIR 2	2.10–2.30	30
Band 8 – Panchromatic	0.50–0.68	15
Band 9 – Cirrus	1.36–1.39	30
Band 10 – thermal 1	10.6-11.2	100
Band 11 – Thermal 2	11.5-12.5	100

Sentinel-2 is a constellation of two identical satellites (Sentinel-2A, launched in 2015 and Sentinel-2B, launched in 2017) part of the European Space Agency earth observation program (Copernicus). The two satellites are on the same orbit and phased at 180° to each other, which brings the temporal resolution to five days (European Space Agency, 2015). Their 13 bands have a spatial resolution between 10 and 60 m (Table 1.6). Sentinel-2's high spatial and temporal resolutions are useful to study phenomenons changing rapidly and covering small extents, such as emergency management, for tracking phenology over limited habitats, or mapping vegetation at the species level (European Space Agency, 2015; Misra et al., 2020; Persson et al., 2018).

Table 1.6: Spectral bands of Sentinel-2. NIR: near infrared, SWIR: shortwave infrared. Bands highlighted are used in this thesis.

Sentinel-2		
Bands	**Wavelength (μm)**	**Spatial resolution (m)**
Band 1 – Coastal aerosol	0.42-0.46	60
Band 2 – Blue	0.43-0.56	10
Band 3 – Green	0.52-0.6	10
Band 4 – Red	0.63-0.7	10
Band 5 – Vegetation red edge	0.69-0.72	20
Band 6 – Vegetation red edge	0.73-0.75	20
Band 7 – Vegetation red edge	0.76-0.8	20
Band 8 – Near infrared (NIR)	0.73-0.94	10
Band 8A – Narrow NIR	0.84-0.88	20
Band 9 – Water vapour	0.92-0.96	60
Band 10 – SWIR – Cirrus	1.34-1.4	60
Band 11 – SWIR	1.52-1.7	20
Band 12 – SWIR	2.03-2.38	20

1.5.2.3 Remote sensing for vegetation

Remote sensing can be used to follow the distribution and dynamics of vegetation, get information on health, species and density. Band combinations used for vegetation studies are called vegetation indices. They rely on the way solar radiation interacts with healthy leaves and which sections of the electromagnetic spectrum are reflected (high reflectance) or absorbed/transmitted (low reflectance) (Figure 1.29), which are related to the leaf surface, thickness, water content, biochemical composition, and pigment concentration, as well as canopy structure (Adam et al., 2010; Glenn et al., 2008; Zhang et al., 2012). They are used for a qualitative and quantitative evaluation of vegetation cover. During droughts, the stress caused by unfavourable conditions may cause plants to reduce their leaf area. The foliage density, which can be estimated by the vegetation indices, is a useful parameter to gather information on plant health and physiological status (Bannari et al., 1995; Glenn et al., 2008). However, the low tree cover and density make detection of vegetation in arid areas more difficult and the spatial resolution used can become an important error source (Lawley et al., 2016; Pettorelli et al., 2005; J. Yang et al., 2012).

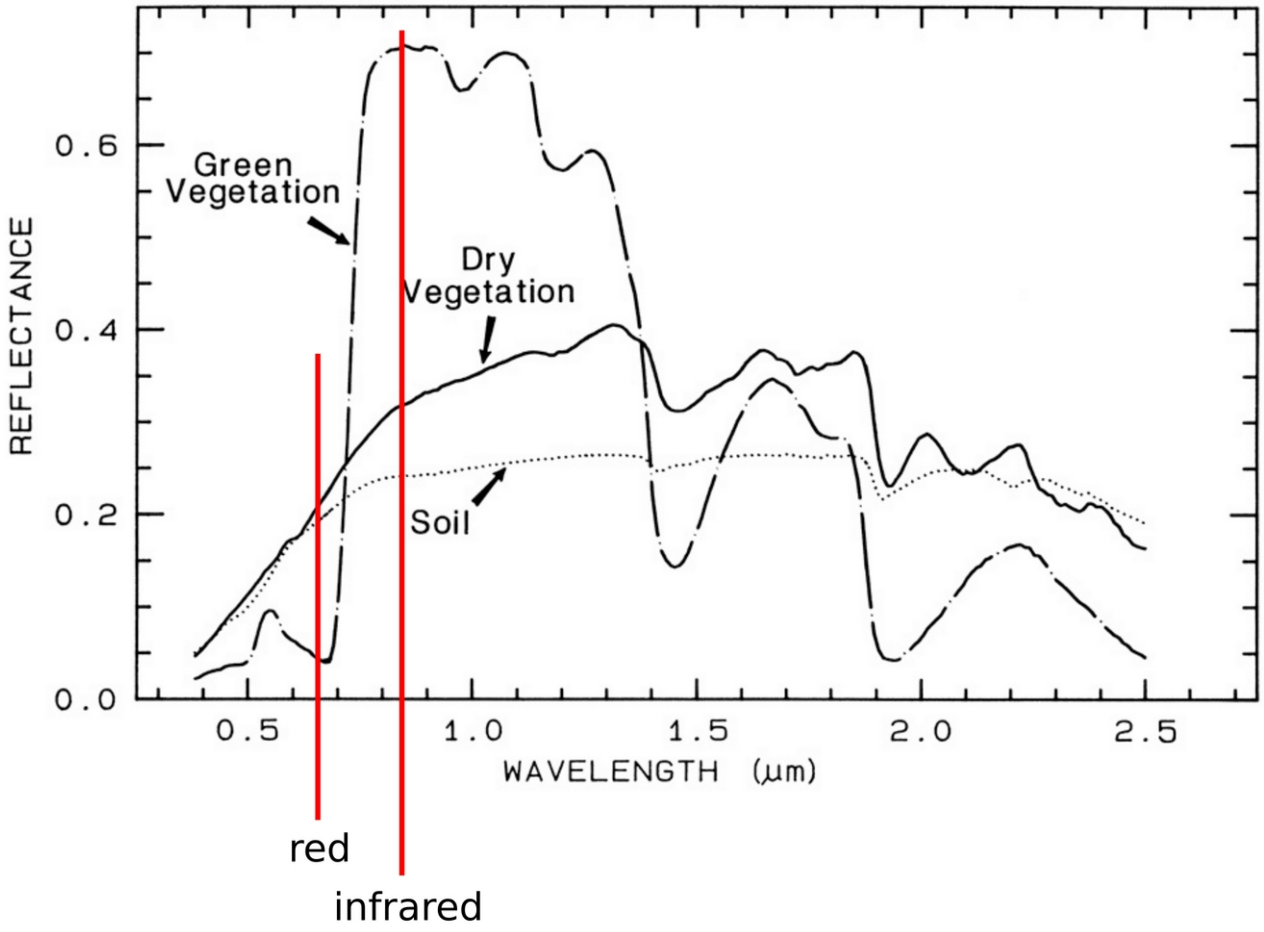

Figure 1.29: Reflectance spectra of photosynthetic (green) vegetation, non-photosynthetic (dry) vegetation and soil (adapted from Verhoeven & Jo, 2006). Red is absorbed and infrared is reflected by green vegetation.

Among the many vegetation indices available, the Normalized Difference Vegetation Index (NDVI) is the one used most often (Bannari et al., 1995; Kerr & Ostrovsky, 2003; Lawley et al., 2016; Sonnenschein et al., 2011; X. Yang, 2007) and it has been shown to strongly correlate with photosynthesis and primary production of vegetation (Kerr & Ostrovsky, 2003). It is a ratio using the red and infrared bands, as red is strongly absorbed by chlorophyll (low reflectance) and infrared is reflected by leaf cellular structure (high reflectance) (Figure 1.27 & 1.29) (Glenn et al., 2008). This index can be used to detect sparse vegetation and to follow tree cover dynamics (J. Yang et al., 2012). When used to predict percent vegetation cover in a riparian forest of the Colorado River delta, NDVI was the index which gave results closest to the result of ground surveys (Nagler et al., 2001). NDVI ranges between +1 and -1, where negative values represent water, low but positive values represent bare soil and higher positive values represent high vegetation greenness (also referred to as vegetation density or vegetation cover) (Figure 1.30) (Glenn et al., 2008).

The formula for NDVI (Rouse et al., 1973) is:

$$\mathrm{NDVI}=\frac{\mathrm{NIR}-\mathrm{Red}}{\mathrm{NIR}+\mathrm{Red}}$$

With NIR: near infrared

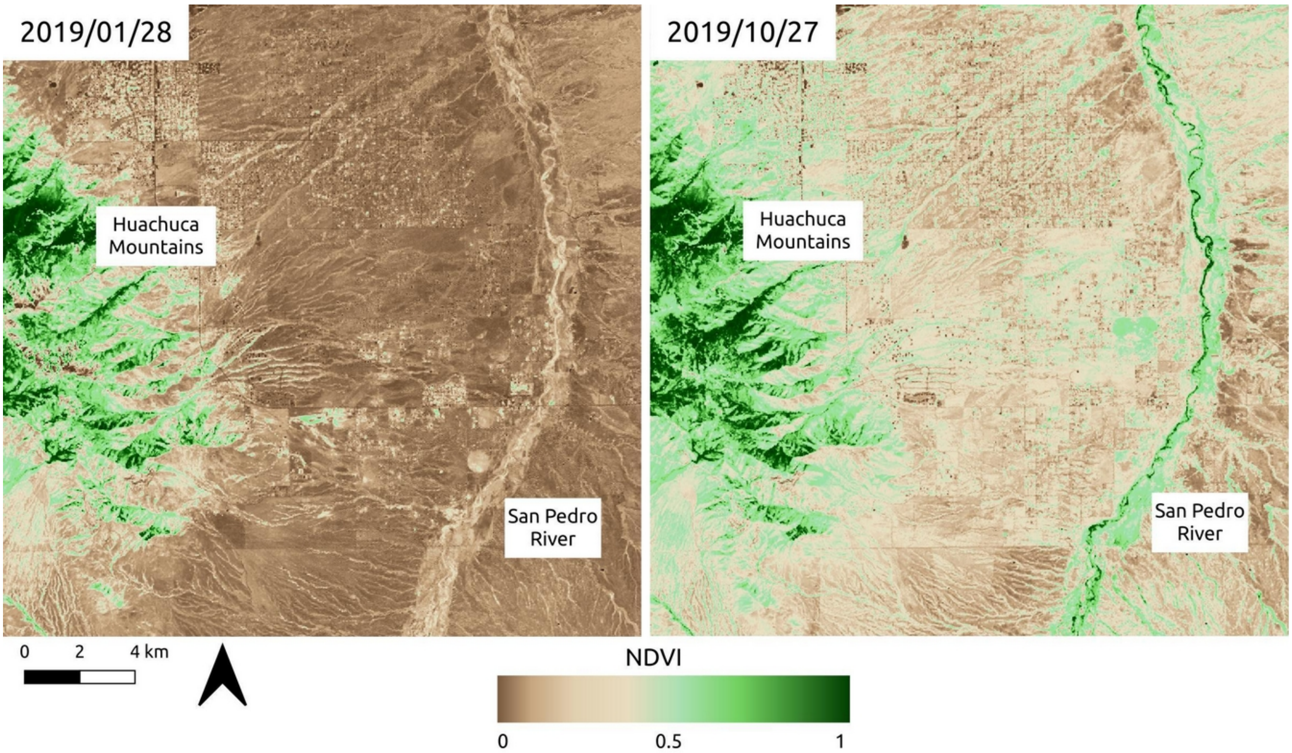

Figure 1.30: NDVI values over the study site for January (left) and October (right) 2019. Vegetation in the Huachuca Mountains is evergreen pine-oak, vegetation along the San Pedro River is cottonwood and the valley in-between is semi-arid scrubland/grassland.

Many other vegetation indices have been developed, often adapted from the NDVI. The Soil Adjusted Vegetation Index (SAVI) was designed to be used for vegetation detection in arid areas with a sparse vegetation cover and highly reflective soils. Its formula is the same as the NDVI but with a correction factor for soil brightness (Nagler et al., 2001). Another version of this index is the Optimized Soil Adjusted Vegetation Index (OSAVI). It is used with the same goals as the SAVI but uses a different correction constant. The Enhanced Vegetation Index (EVI) uses the red, near infrared and blue bands, as well as a canopy background correction constant. It was designed to correct for smoke or haze in the atmosphere. Each of these indices perform differently depending on the study site. The suitability of a specific index is related to the goal of the study, experimental conditions and field applications (Bannari et al., 1995; Lawley et al., 2016; X. Yang, 2007).

I decided to use NDVI after comparing it with other indices such as SAVI and EVI by comparing how well they could differentiate between the main land covers found in my study site (Figure 1.31). If SAVI cover a wider range of values (from 0 to 1.2 against 0 to 0.8 for NDVI), class separability between the different land covers is similar for both indices with little to no overlap between classes (Figure 1.31a, b). EVI was characterized by more overlap, especially for the denser trees (cottonwood, evergreen trees and mesquite), but also between urban areas and valley scrubland (Figure 1.31c). NDVI was found to be efficient to separate the main vegetation communities and the fact that the value is normalized between -1 and 1 made comparing trends and rates of change easier. Available metrics were also limited by band availability. The index chosen had to be obtainable with Landsat and Sentinel-2 10 m bands (red, green, blue and near infrared). Sentinel-2 was chosen specifically for its 10 m resolution to map narrow riparian corridors in mountain canyons so using other data, such as the 20 m resolution red-edge bands did not fit the goals of the project.

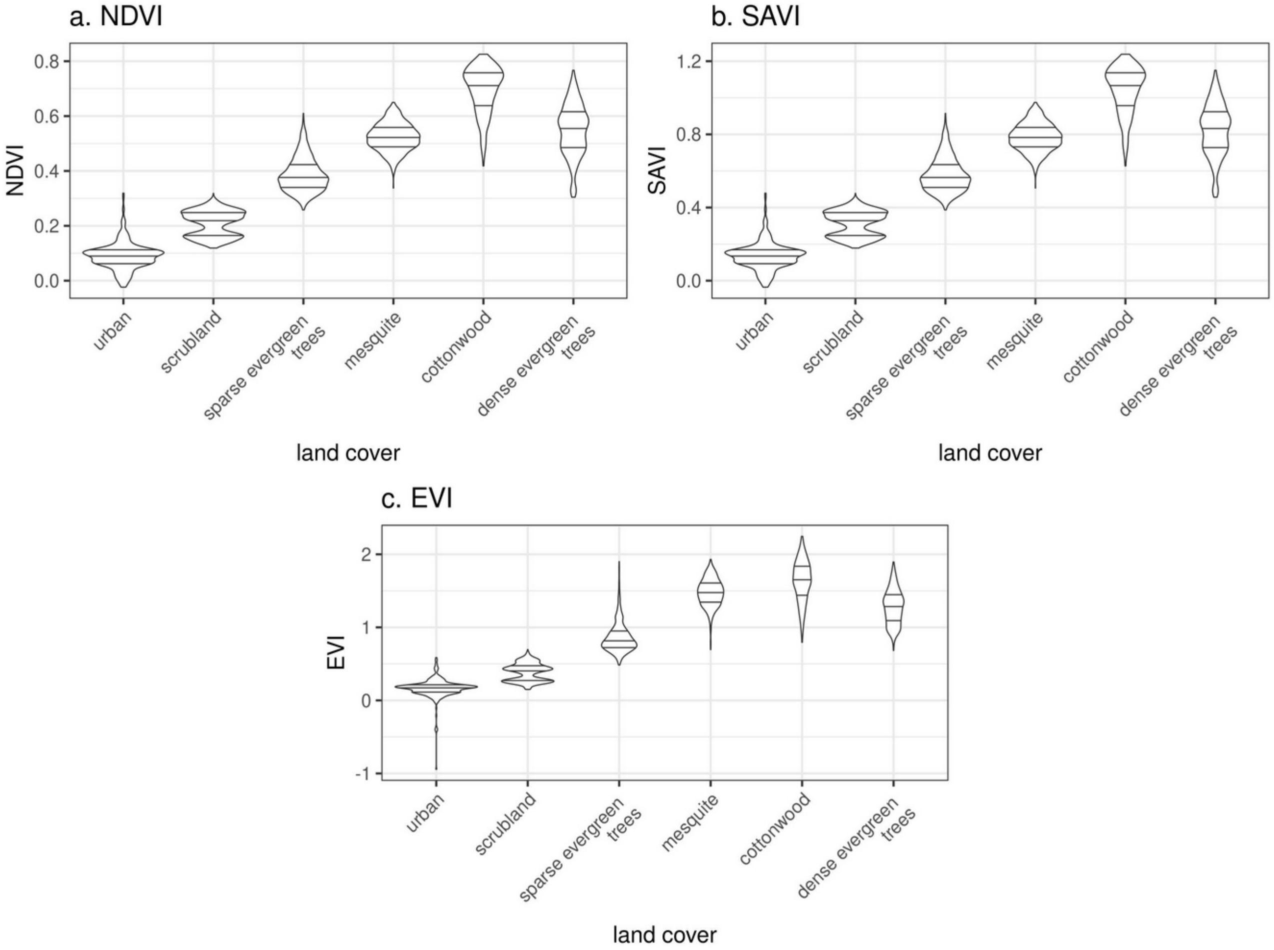

Figure 1.31: Value distribution of various vegetation indices for the main land cover classes over the study site: NDVI (a), SAVI (b) and EVI (c). Landsat-8 image for June 26 2015.

Vegetation indices, by measuring vegetation greenness, can be used to create vegetation maps by classification, sorting all the pixels over a specific threshold as vegetation and all the pixels under the threshold as non-vegetation, or mapping vegetation of different density (grasses and riparian trees for example) (Johansen & Phinn, 2006; Manning et al., 2020; Senay & Elliott, 2002; X. Yang, 2007). This allows us to have a clear look at the extent of a riparian corridor and can also be used to show differences in spatial patterns of change between areas or through time by computing statistics on average patch size and connectivity (Jones et al., 2008; Lawley et al., 2016). More detailed classifications can be done, using the satellite bands directly, vegetation indices or other metrics, to create land cover maps with varying levels of precision, from a simple vegetation/non-vegetation map, to distribution of individual species. There are several algorithms available to regroup pixels sharing similar spectral characteristics, and classifications can be supervised (comparing each pixel spectral signature to a training dataset) or unsupervised (pixels are grouped by their spectral similarity) (Adam et al., 2010). The goal of the study and the satellite imagery used will govern which metrics for the classification and which classes are chosen. Classification results can be validated by comparing how a subset of pixels was sorted in classes to the observed class of this subset during fieldwork or by looking at very high resolution imagery (Kerr & Ostrovsky, 2003).

Remote sensing can be used to keep track of vegetation health by measuring variations in vegetation greenness and phenology. Vegetation indices are a measure of vegetation health, since stressed vegetation sheds leaves and branches (Section 1.2.3), lowering the vegetation greenness, which translates to a lower value of the vegetation index. We can then compare variations in vegetation greenness against potential environmental controls on vegetation health such as rainfall distribution to try and understand the link between vegetation health (as in leaf density and chlorophyll content) and these variables, but also to measure anthropogenic impacts (a dam construction or removal for example) or the success of management practices (Nguyen et al., 2015). Satellites with high temporal resolution are used to study phenology and characterize seasonal vegetation density variations (Cleland et al., 2007; Kato et al., 2021; Pastick et al., 2018). Leaf-out and senescence are the stages exhibiting the strongest variations in vegetation greenness (Figure 1.30), and for some species, flowering can also be visible in satellite imagery (Kato et al., 2021; Paz-Kagan et al., 2019). Start, length and end of the growing season can be measured and, like vegetation greenness, be compared to potential controls such as temperature, length of day or precipitation (Elmore et al., 2005; Gómez-Mendoza et al., 2008; Walker et al., 2015; Zhang et al., 2012). Inter-species differences in phenology timing and amplitude are also a means to discriminate between different vegetation communities and species, and is used for land cover classification (Pastick et al., 2020).

Satellite imagery continuously record long-term measurements of key ecological parameters, which makes it a great fit for monitoring of land cover change, vegetation health or fire recovery (Glenn et al., 2008; Kerr & Ostrovsky, 2003; Klemas, 2014). Images used in a time series are well suited for following trends in vegetation cover as they allow to study large areas with consistencies in spatial and spectral resolution over time. Studying an area over several years is also useful to highlight the differences between regular seasonal variation and significant transition, as well as assess if the change is gradual and long-term (invasive species arrival for example) or abrupt and short-term (fire) (Coppin et al., 2004; Pettorelli et al., 2005; J. Yang et al., 2012). Pixel-based linear trend analysis can be used to keep track of gradual change (Roder et al., 2008). It can also be useful to look at vegetation changes in steps. By subtracting the NDVI values pixel by pixel between two dates (image differencing), the resulting difference map can help to highlight the main changes in vegetation health and extent. This method is used to get an overview of the evolution of riparian corridors and point out the main areas of change where the index values have increased or decreased by a set proportion or value (Jones et al., 2008; Nguyen et al., 2015; Sheffield et al., 2012). A similar method can be used between two land cover classification of the same area (Figure 1.32) (Coppin et al., 2004). Vogelmann et al. (2009) also used a specific set of rules allowing to discriminate between pixels showing an increase five times in a row, four times, three times, or only two times over a time series. This allowed them to understand if the change was slow (decreasing over several years) or fast (decreasing over a few years before increasing again or levelling off). Important parameters to take into account for change detection is the date chosen for the comparison (especially when studying long-term changes on objects following regular short-term seasonal variations, such as vegetation phenology), the temporal resolution of the sensor used, which will influence the change interval length, and the classes used when comparing two land cover classification at different times (Coppin et al., 2004; Sonnenschein et al., 2011).

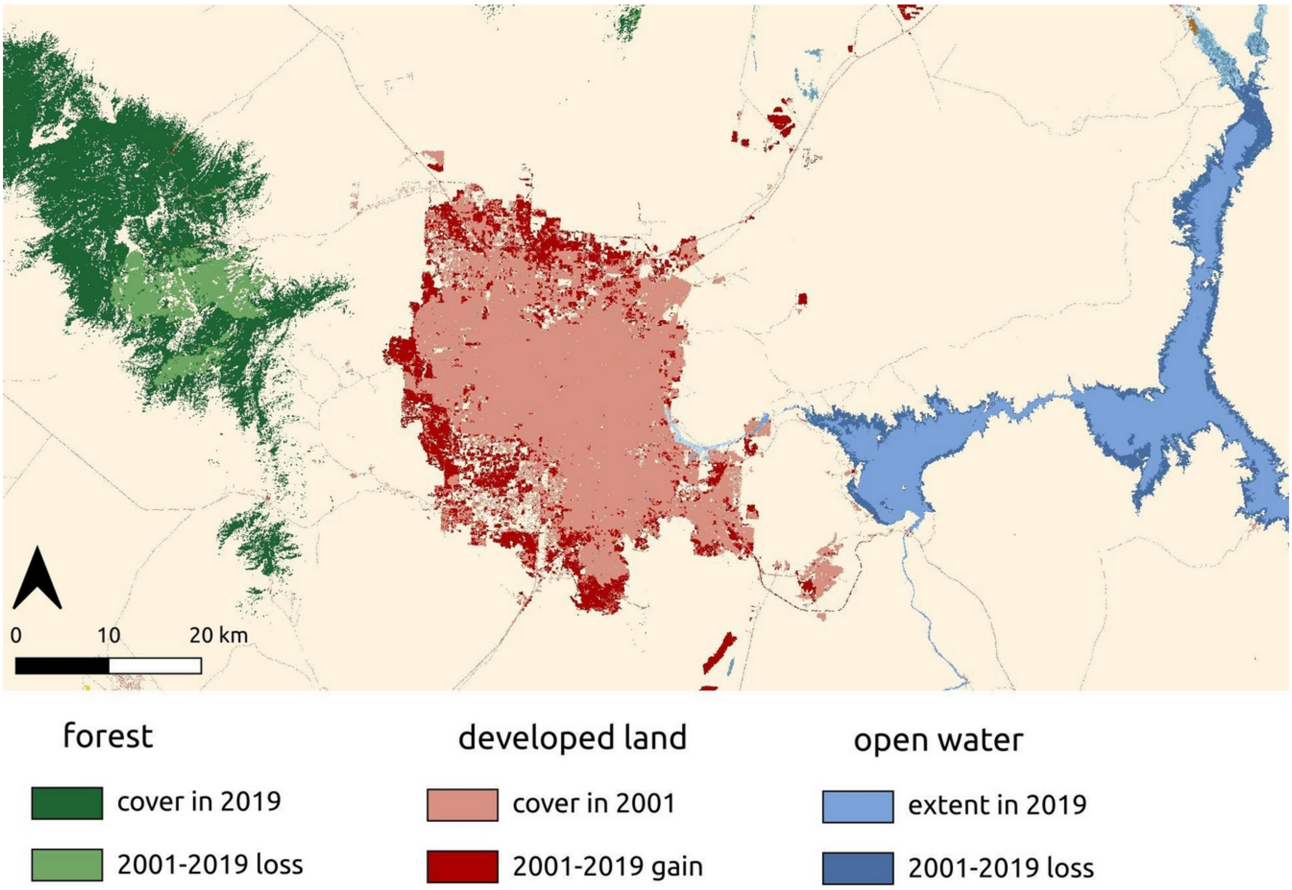

Figure 1.32: Land cover changes around Las Vegas, Nevada, USA and urban growth between 2001 and 2019. Data from the National Land Cover Database, https://www.mrlc.gov/data.

1.6 Thesis outline

This thesis is organized in three analysis chapters and a general discussion. Chapter 2 is focused on time series of vegetation greenness for the main vegetation communities across the study site, Chapter 3 is a study of the spatiotemporal distribution of flow permanence in the Huachuca Mountains streams, and Chapter 4 is an analysis of environmental controls on vegetation distribution, health and life cycle in the Huachuca Mountains.

Chapter 2 is organized around the analysis of long-term time series of water availability and vegetation. This chapter’s goal is to understand how and when water enters and moves through a diverse arid landscape, as well as how local vegetation cover changes in response to water distribution and availability. I have used a wide array of data: streamflow, precipitation and groundwater levels daily datasets, local geology, flow permanence surveys and Landsat images to measure vegetation greenness. The study is conducted over a 30 years time series to try and understand the dynamics at play between water availability and vegetation. The results highlight the differences between different vegetation types but also between riparian vegetation growing in drier or wetter reaches along the San Pedro River in the Lower Colorado River basin. This chapter is focused on inter-annual variations in vegetation greenness and long-term changes in riparian forest health, as well as general water distribution information including rainfall, streamflow and groundwater.

The purpose of Chapter 3 is to understand spatiotemporal flow permanence distribution across the Huachuca Mountains, and the environmental controls that regulate streamflow. I investigate how precipitation translates into streamflow in the various canyons scattered through the mountains. This chapter showcases geology, precipitation and flow permanence data derived from conductivity sensors. This chapter examines seasonal classification of stream reaches, the mechanisms behind the spatiopemporal distribution of flow, and the potential vulnerability of flow permanence to drought.

In Chapter 4, the focus is on vegetation distribution in the Huachuca Mountains, how water permanence and elevation shifts influence vegetation distribution, and how drought influences density, distribution and phenology of riparian vegetation. A supervised classification provides a land cover map of the study site, including the main vegetation communities. I analyse the influence of flow permanence and elevation on riparian vegetation distribution along the mountain canyons, and compare timing and length of the growing season to seasonal rainfall distribution. This chapter is focused on intra-annual changes in vegetation greenness and short-term variations in phenology timing. The main environmental control studied in this chapter is drought, with data on rainfall and temperature.

In the general discussion (Chapter 5), I present an overview of water availability to vegetation over the study site, and how variations in water availability can impact vegetation distribution, health and vulnerability to climate change. I also go into more details on how this work can be used in a management context. I describe a set of metrics that can be derived from the tools and methods I use in this thesis and can be useful for the continuous monitoring of potential critical habitats and their sensitivity to stressors, such as drought, at the landscape scale. I describe the strengths of each method and the questions they can answer before providing implementation suggestions.

Chapter 2
Vegetation responses to climatic and geologic controls on water availability

An edited version of this chapter was published as: Sabathier, R., Singer, M. B., Stella, J. C., Roberts, D. A., & Caylor, K. K. (2021). Vegetation responses to climatic and geologic controls on water availability in southeastern Arizona. *Environmental Research Letters*, *16*(6), 064029. https://doi.org/10.1088/1748-9326/abfe8c

2.1 Abstract

Vegetation distribution, composition and health in arid regions are largely dependent on water availability controlled by climate, local topography and geology. Despite a general understanding of climatic and geologic drivers in plant communities, trends in plant responses to water distribution and storage across areas under different local controls are poorly understood. The multi-decadal interactions between spatial heterogeneity of geologic controls and temporal variation of climate was investigated, as well as their impacts on water availability to vegetation and plant responses (via Normalized Difference Vegetation Index, NDVI) in a monsoon-driven arid region of southeastern Arizona. The results show that grasslands display low NDVI and respond directly to monsoonal rainfall. In the uplands, vegetation on west-facing slopes and in canyons share similar NDVI averages and variability, suggesting that they both use water from surface-groundwater flow paths through fractured rocks. Along the San Pedro River, streamflow, groundwater, and NDVI in deciduous riparian woodlands are strongly responsive to monsoonal rainfall, but water availability stratifies between wet (perennial), intermediate, and dry

reaches, underlain by different local geologic controls that affect water table elevation. These controls interact with the driving climate to affect water availability in the shallow alluvial aquifer of the riparian zone, a primary water source to the gallery phreatophytes. A recent shift toward a strengthened monsoon in the region has led to an increase in water availability for grasslands and for dry reaches of the San Pedro, while the benefit is more muted along wetter reaches, where the riparian forest shows signs of having reached it’s maturity, with diminished trends in NDVI. These results have implications for the future vulnerability of dryland vegetation to climate change, which may be either dampened or intensified by local controls such as geology.

2.2 Introduction

Plants in dryland ecosystems may experience differential seasonal access to water and distinct long-term trends in their responses to water availability changes, based on rooting depth, as well as the local expression of hydrology and water storage at their rooting location. Thus, changes to the climatically controlled water cycle in an arid region, where water availability is the main limiting factor to plant growth, can have important consequences for vegetation distribution, health and functioning (Caylor et al., 2005; Loik et al., 2004; Shafroth et al., 2000; Singer et al., 2014; Stella, Riddle, et al., 2013; Tietjen et al., 2009). Precipitation brings water to the land surface where it may become available to vegetation as a function of local storage, yet the amount and distribution of water (in streamflow, soil moisture and groundwater) depends on the rainfall intensity, duration, location and seasonal distribution throughout the year, as well as the fluxes in the hydrological cycle including evapotranspiration, infiltration of rainfall into the soil, runoff generation, and percolation to aquifers.

A key unknown is how spatial variations in subsurface geology along dryland riparian systems affects plant-water interactions and corresponding ecosystem responses to climate-controlled variations and shifts in water availability. In arid environments, where evapotranspiration exceeds precipitation, vegetation is typically concentrated at locations in the landscape where runoff accumulates and/or where the water table is close to the surface (Dawson & Ehleringer, 1991; Lite & Stromberg, 2005; Patten, 1998; Rodriguez-Iturbe et al., 2007), yielding potentially strong differences in vegetation types and density across a region with the same driving climate depending on the local geologic controls and geomorphology (Caylor et al., 2005, 2009; Franz et al., 2010). Lowland riparian forests in arid regions, for example, may have frequent access to water from multiple, seasonally mixed water sources (Singer et al., 2014), in contrast to open grasslands, shrubs, and trees growing on slopes, which are prone to more seasonal dryness and susceptible to drought (C. D. Allen & Breshears, 1998; Breshears et al., 2005). Although lowland riparian forests have a small footprint in arid landscapes, they represent critical moisture and thermal refuge for a range of species, many of which may be considered threatened or endangered (Albright et al., 2017; Seavy et al., 2009; Stromberg et al., 1996). However, dryland riparian forests are vulnerable to shifts in climate that affect root zone water availability to the key plant species because they cannot expand their range (Loarie et al., 2009a; Malagnoux et al., 2007; Reidmiller et al., 2018), making them sensitive to climate change.

The goal of this chapter is to address how climate variation through time and/or geologic controls in space affect water availability to vegetation growing across a diverse landscape under the same climate regime by using various time series datasets including satellite-derived vegetation greenness, groundwater wells, as well as streamflow and rainfall gauges. These datasets were used to provide a general understanding of the controls on water availability and vegetation community responses across a dryland region. The premise is that if the climatic forcing on water availability to vegetation in the recent past can be better understood, then the vegetation response to climate change in the coming years could be better predicted.

Conceptually, I consider how water from rainfall travels through the landscape in a mountain front recharge system, and how groundwater and surface flow (and soil moisture) affect vegetation communities distribution and composition (Figure 2.1).

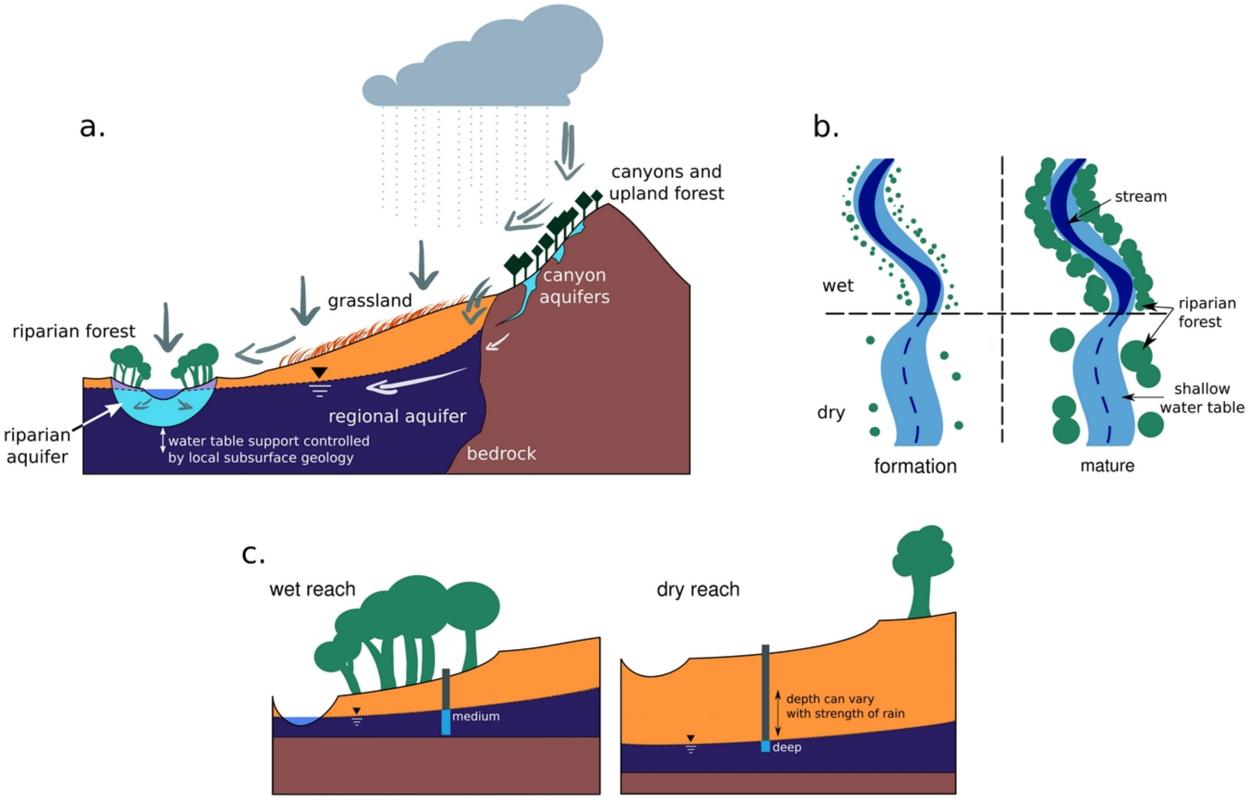

Figure 2.1: Conceptual model showing the path of water through the landscape and the location of each vegetation community studied (rainfall water going down the canyons and seeping in fractures to be used by upland and canyon vegetation, then reaching the deep regional aquifer before reaching the river and interacting with streamflow, providing a stable water source for the riparian trees while the grasslands have to rely on rainfall) (a), a view of the river showing the temporal changes experienced within the riparian forest along the stream (the growth of a riparian corridor for the wet reach, and scattered trees in the dry reach (b), and a view showing the underlying geology of the wet and dry reaches, as well as the response of water table depth to precipitation (depth reached during the drier month of the year) (c).

2.3 Data and methods

Temporal changes in vegetation greenness were investigated by measuring trends in NDVI, using Landsat images taken between 1986 and 2017. Three vegetation communities of the San Pedro basin were compared, as well as seasonal differences (before and after the monsoon) and period differences (before and after 2006). These results were compared to time series of water availability through a rainfall model, streamflow gauges and groundwater wells.

2.3.1 Study area

The study area covers the Huachuca Mountains, the semi-arid grassland north of Fort Huachuca, and a 60 km stretch of the San Pedro River between the Mexican border and Saint David. The study area vegetation was mapped using the U.S. Department of Agriculture’s National Agriculture Imagery Program (NAIP) imagery (https://www.fsa.usda.gov/programs-and-services/aerial-photography/imagery-programs/naip-imagery/), classified by communities that occur in different parts of the landscape and have potentially different access to water: (1) cottonwoods (*Populus fremontii*) of the San Pedro River riparian forest, (2) open grasslands (mix of grasses and scrubs), (3) fir-oak forest in the Huachuca Mountains and (4) riparian forest along the canyons of the Huachuca Mountains, specifically Huachuca Canyon and Garden Canyon (Figure 2.2). For the trees of the San Pedro corridor, separate masks were created to stratify the vegetation responses within different categories of flow permanence (wet, intermediate and dry reaches), based on riparian condition classes, based on the riparian condition classes mapped by Stromberg et al. (2006) from vegetation traits sensitive to changes in streamflow permanence and/or groundwater levels (Chapter 1).

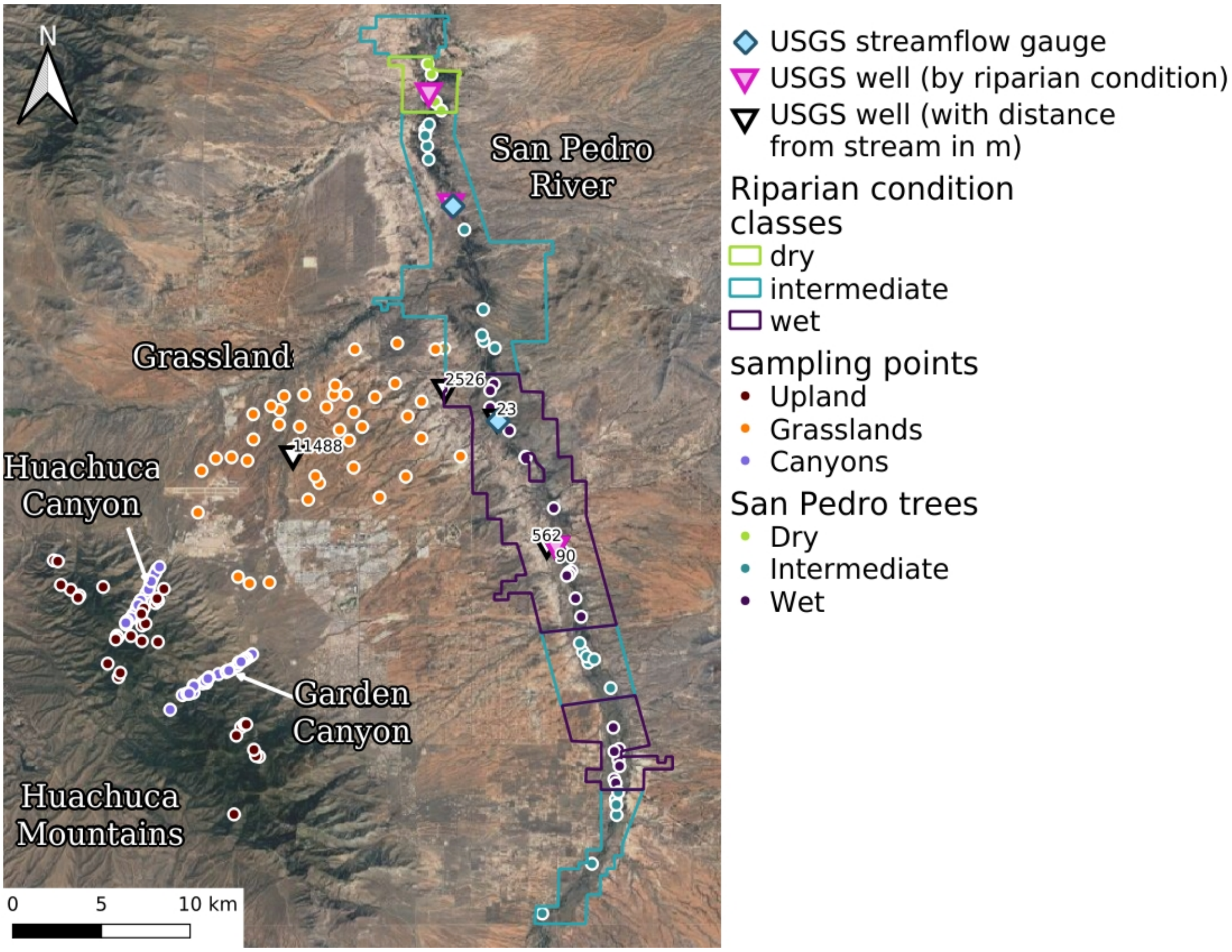

Figure 2.2: Study area (National Agriculture Imagery Program image), hydrologic stations, and sampling points used for the NDVI analysis, split by vegetation community. Riparian condition classes are from Stromberg et al. (2006).

The geology under the San Pedro River is comprised of various units of varying permeability to water (Figure 2.3). This dataset was provided by the U.S. Geological Survey (USGS) and downloaded from https://mrdata.usgs.gov/geology/state/. The years of permanent flow was derived from annual flow permanence data provided by Lisa McCauley from The Nature Conservancy, which leads the project monitoring the spatial extent of perennial flow in the river right before the beginning of the monsoon (more information available here: https://azconservation.org/projects/water/wet_dry_mapping).

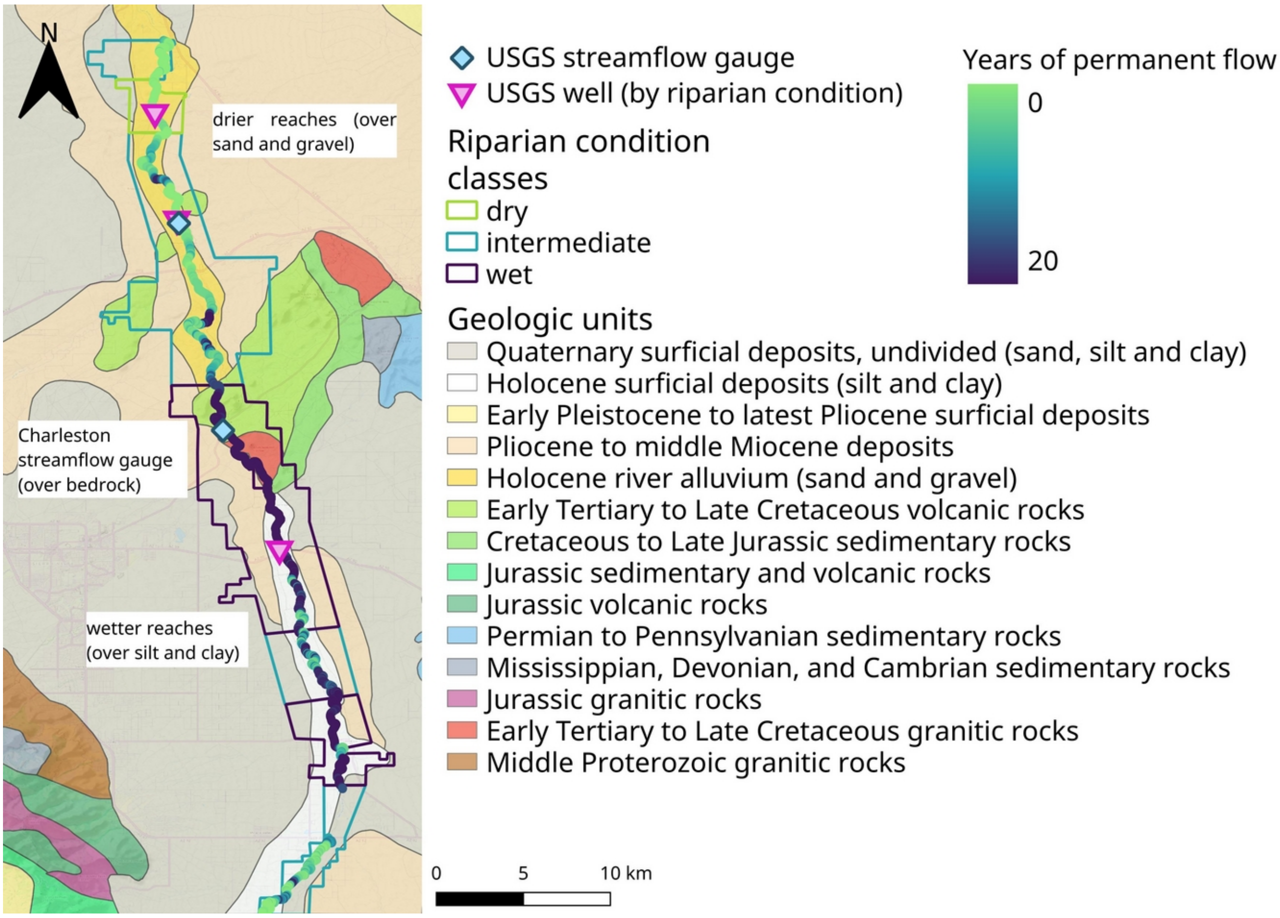

Figure 2.3: Geologic map of the San Pedro River and its surroundings showing the geologic units, the riparian condition (Stromberg et al., 2006) and flow permanence.

The southern reaches are underlain by deposits of sand with layers of silt and clay that act as confining beds, holding the water close to the surface and allowing for an upward flow along gaining reaches (Blakemore, 2006; Pool & Coes, 1999). Around Charleston, an outcrop of low-permeability granitic and volcanic bedrock keeps water at the surface and the river flows year-round reliably. In contrast, the north half of the study site is underlain by sand and gravel, enabling high transmission losses under the stream, resulting in losing (intermittent/ephemeral) reaches (Blakemore, 2006; Pool & Coes, 1999; Quichimbo et al., 2020).

This region is under the influence of the North American Monsoon, which leads to large seasonal variations in precipitation, and rainfall falling mainly during the summer monsoon (July to September) and in winter (Loik et al., 2004; Vera et al., 2006; Chapter 1). At a more local scale, topography and orography also affect precipitation patterns (Loik et al., 2004).

2.3.2 Datasets and methods

Cloud-free images were selected from the Landsat Earth Observation Satellite Program, contrasting those from summer (June) and autumn (October), to understand the impact of the summer monsoon and winter rains on vegetation. June images were selected for the pre-monsoon, and October images were selected for the post-monsoon. If there was no clear image for the months of interest, the closest clear date was selected. Pre- and post-monsoon imagery time series were thus built from 1986 to 2017 (Figure 2.4).

To measure how vegetation changed over time (both as long-term changes and inter-annual responses to climate variability), I used the Normalized Difference Vegetation Index (NDVI; Rouse et al., 1973), which provides information on vegetation greenness (Bannari et al., 1995; Kerr & Ostrovsky, 2003; Lawley et al., 2016; Yang et al., 2012; Chapter 1).

$$\mathrm{NDVI}=\frac{\mathrm{NIR}-\mathrm{Red}}{\mathrm{NIR}+\mathrm{Red}}$$

with NIR: near infrared

Analysis-ready NDVI Landsat images are provided by the U.S. Geological Survey and were downloaded from the Earth Explorer (https://earthexplorer.usgs.gov/). Images before 2012 are from Landsat 5, and images after 2012 are from Landsat 8.

All cloudy images were removed from the analysis, and data from Landsat-5 and Landsat-8 were homogenized after Goulden and Bales (2019) with the following formula:

$$L5\,NDVI = \frac{L8\,NDVI * 0.9553 - 0.0136}{1.0630}$$

With L5 NDVI: NDVI value from Landsat 5, and L8 NDVI: NDVI value from Landsat 8. Gouden and Bales (2019) used regressions by comparing consecutive observations and established formulas linking Landsat 7 NDVI to Landsat 5 NDVI, and Landsat 7 NDVI to Landsat 5 NDVI, which allowed me to determine the relation between Landsat 5 and Landsat 8 directly.

For each vegetation class (cottonwoods, open grasslands, fir-oak forest), I defined a mask covering an area with a homogeneous land cover based on NAIP images visual analysis and a cloud of random points was drawn in this area (Table 2.1). Vegetation indices values for the points were then extracted for each year (both pre- and post-monsoon).

Table 2.1: Vegetation communities sampled

Class		**Number of sampling points**
San Pedro riparian forest	Wet	23
	Intermediate	23
	Dry	6
Grassland		41
Canyons		50
Upland forest		31

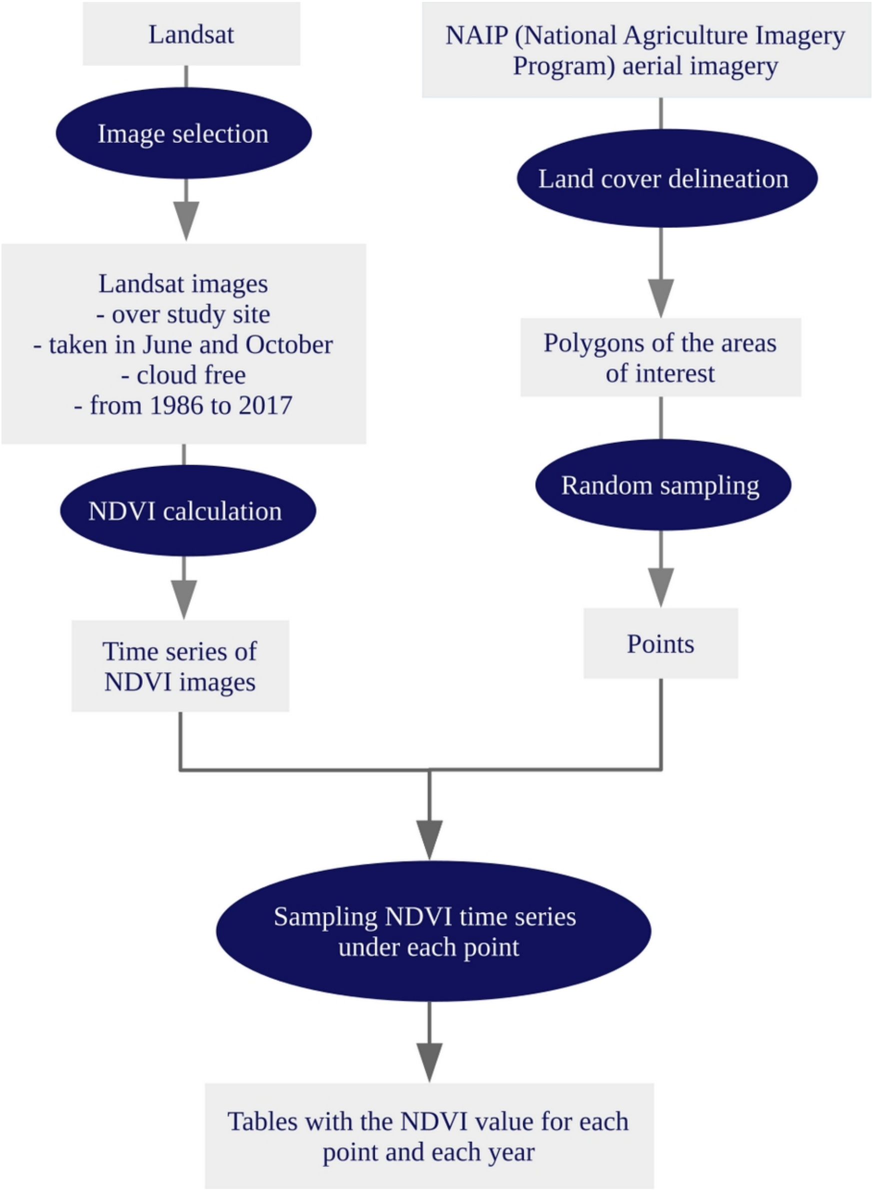

Figure 2.4: Steps of the Landsat images sampling

Local hydrology data came from two different datasets. Daily rainfall data came from the National Oceanic and Atmospheric Administration (NOAA)'s Climate Prediction Center's Unified Gauge-Based Analysis of Daily Precipitation (https://psl.noaa.gov/data/gridded/data.cpc.globalprecip.html). This dataset is provided on a 0.5°x 0.5° grid and the grid cell covering both the Huachuca Mountains and the San Pedro River was used. This dataset uses quality controlled daily precipitation data from gauge data all over the world interpolated using the optimal Interpolation method with orographic consideration. The cross validation performed by Chen et al. (2008) showed a correlation of 0.811 and a bias under 1%. Daily streamflow and water table depth data for the time period overlapping with the NDVI dataset were acquired for selected locations in the study area from the U.S. Geological Survey (USGS) National Water Information System (U. S. Geological Survey, 2022) (Figure 2.2 and see tables 2.2 and 2.3 for gauges and wells numbers). Interpolated monthly potential evapotranspiration (PET) provided by the Climatic Research Unit (https://catalogue.ceda.ac.uk/uuid/89e1e34ec3554dc98594a5732622bce9) was also used. PET takes into account atmospheric parameters such as temperature and wind and their effect on water demand by plants. This dataset shows no trend in PET over the 1986-2017 time period for our study (Figure 2.5), meaning that the variations in NDVI are not caused by a change in atmospheric demand. To identify changes in mean precipitation through time, I used the At Most One Change method, meant to find one main shift in mean value along a time series. I chose this method as I wanted to highlight only the main change to compare with the shift in seasonal rainfall distribution pinpointed in Figure 2.6.

Table 2.2: U.S. Geological Survey streamflow gauges used

gauge number	**gauge name**
USGS 9471000	San Pedro River at Charleston
USGS 9471550	San Pedro River near Tombstone

Table 2.3: U.S. Geological Survey wells used

well number
USGS 313309110094301
USGS 313312110082301
USGS 313615110175601
USGS 313738110102901
USGS 313841110122201
USGS 314511110120601
USGS 314901110125401

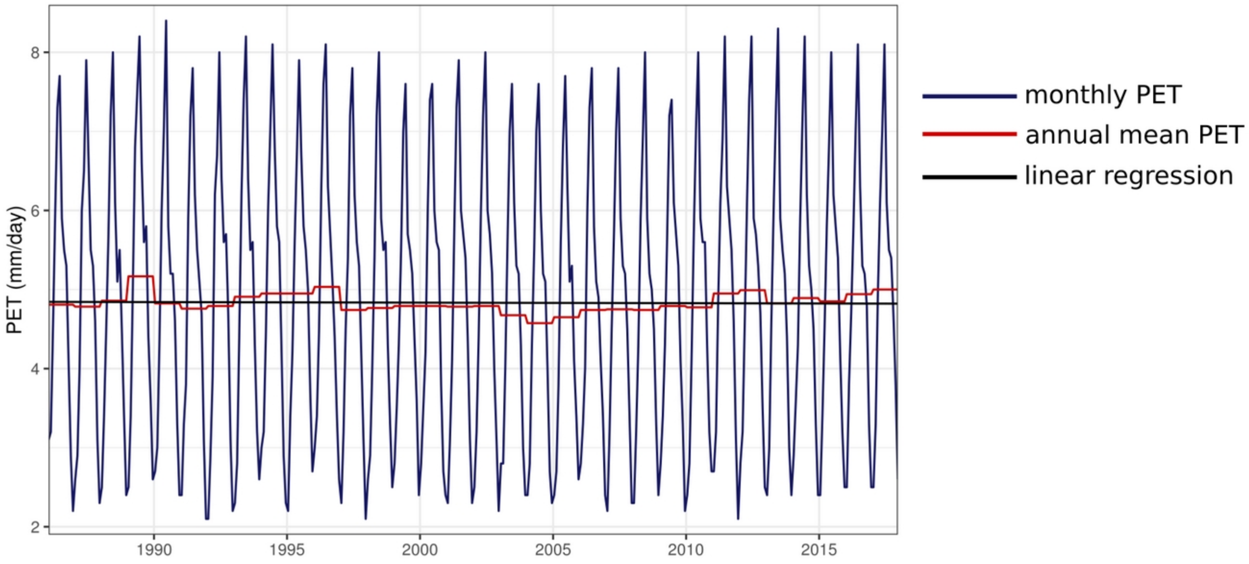

Figure 2.5: PET (potential evapotranspiration) from the Climatic Research Unit (1986-2017). Peak PET is reached right before the monsoon and the lowest values are during winter.

Overall steps for image selection and sampling are detailed in Figure 2.4. Trends were quantified by linear regression of NDVI over time. NDVI data were used to produce violin plots comparing NDVI values and trends across vegetation communities, flow permanence and season. Wilcoxon-Mann-Whitney rank sum tests were performed between seasons and vegetation communities to highlight significant differences. Hydrological time series were plotted to identify the relative contributions of monsoon and non-monsoon precipitation, and groundwater wells data were analysed to explore spatial patterns in water table elevation, variability, and temporal trends.

2.4 Results

The results are presented in two sections: the first one presents time series of rainfall and streamflow and the changes in seasonal water distribution, with a focus on the monsoon. Groundwater median depths and temporal variations are also compared depending on the water table distance with the San Pedro River. Differences in streamflow and water table depths of the San Pedro River between a weak and strong monsoon are also displayed. The second section is focused on vegetation, comparing greenness and greenness trends against communities and seasonal rainfall. Median NDVI values and linear trends are compared between vegetation communities, between the months before and after the monsoon, and before and after 2006.

2.4.1 Hydrology

Total rainfall in the study area does not show a trend ($r = 0.005$; $p = 0.239$), but the seasonal distribution of rainfall during the year appears to be shifting (Figure 2.6a, c). Monsoon rains are slightly increasing ($r = 0.005$; $p = 0.08$), while winter precipitation is significantly decreasing ($r = -0.01$; $p = 0.004$). Furthermore, monsoon precipitation has exceeded winter precipitation since 2005 (Figure 2.6a, c), based on means of 250 mm and 109 mm, respectively. This precipitation shift can also be seen in the San Pedro River discharge

(Figure 2.6b, c). Before 2004, high volumetric discharge was recorded both during the monsoon (mean of 7.15 × 10^6 m^3/year) and during winter (10.52 × 10^6 m^3/year). Since 2004, however, streamflow volume is higher during the monsoon (16.0 × 10^6 m^3/year), than during winter (5.9 × 10^6 m^3/year). These results are confirmed by a changepoint analysis, using the At Most One Change method and run on annual monsoon rainfall for the 1986-2017 time period showing a shift in precipitation distribution in 2006 (Figure 2.7), which in turn has impacted runoff and streamflow generation.

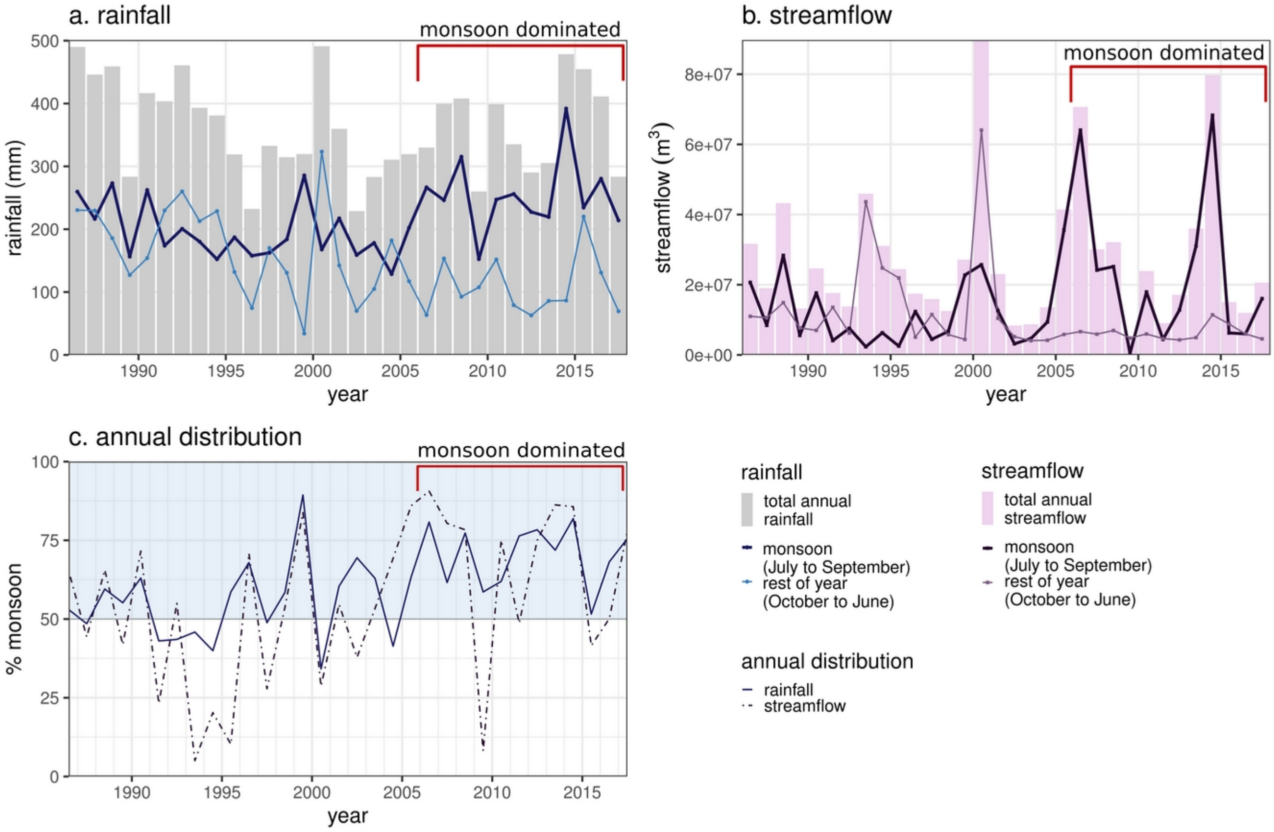

Figure 2.6: 1986-2017 time series of precipitation and streamflow. Precipitation panel includes total annual rainfall, total annual monsoon rainfall (July to September) and total rainfall for the rest of the year (October to June) (a). Streamflow includes total volumetric annual streamflow, total monsoonal streamflow and total streamflow for the rest of the year (b). Panel c shows the annual proportion of rainfall and streamflow measured during the monsoon. When above 50 %, this means that more than half of the annual precipitation or streamflow has been measured during the monsoon season.

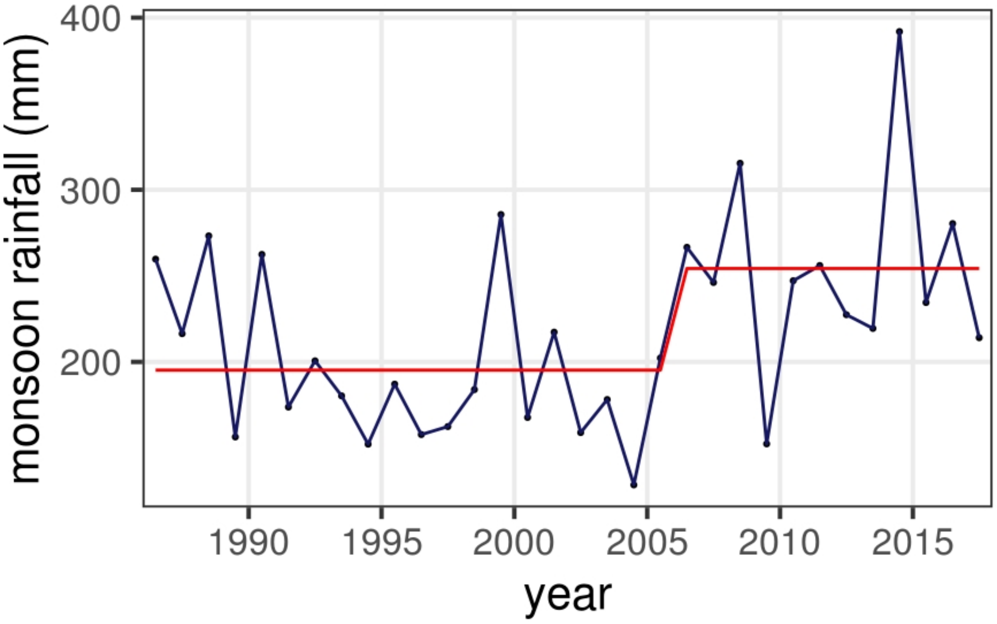

Figure 2.7: Total annual monsoon rainfall (July to September) (in blue) and result of a changepoint analysis using the At Most One Change method (in red) for single point detection.

In terms of groundwater, there is a shallow water table (<2 m deep) directly under the river (Figure 2.8a) with brief and substantial rises (to <1 m deep) during monsoon months and more prolonged but lesser increases during the winter, expressing strong streamflow-groundwater interactions under and around the streambed (Figure 2.8b). However, the water level in these near-stream wells has been steady through the years of this analysis (Figure 2.8b), suggesting consistent support by a deeper groundwater system across the study area (Ajami et al., 2012; Meixner et al., 2016). The water table becomes progressively deeper with much lower seasonal variability (down to 90 m deep in the farthest well with no seasonal variations) with distance from the San Pedro River, reflecting less streamflow-groundwater interactions (Figure 2.8a). Interestingly, the wells located >2000 m from the San Pedro under the grasslands display a slow and steady decline in water table depth (Figure 2.8b). Therefore, although there may be good support for the shallow alluvial aquifer along the San Pedro from mountain front recharge and streamflow, this benefit seems to bypass the aquifer below the grasslands.

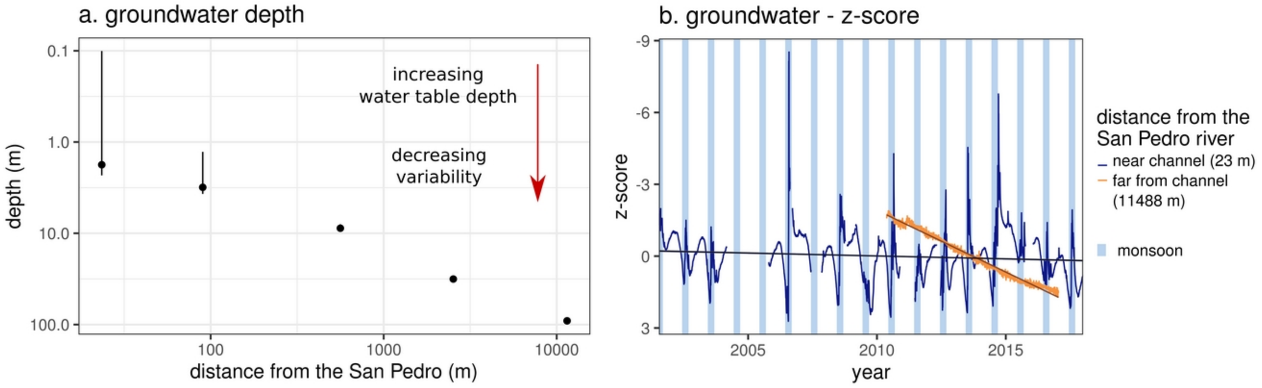

Figure 2.8: Distribution of water table depth over recent decades for several wells at various distances from the San Pedro River, showing median depth, as well as minimum and maximum (a), and time series of daily groundwater z-score for two of these wells: the closest to the channel and the farthest, with the corresponding linear regression (b). Z-score = (daily value – overall average) / overall standard deviation. Both figures use a reversed y axis to show changes in depth.

Streamflow and water table depth variations within flow permanence classes (wet, intermediate, dry) along the San Pedro River were further investigated, focusing on the cases of a strong monsoon in 2008 (positive 98 mm anomaly from 1986-2017 average) versus a weak monsoon in 2009 (negative 65 mm anomaly). The rainfall anomaly for the whole time series is shown in Figure 2.9. In the wet reach, the river is flowing all year round (Figure 2.10a), the minimum value is reached right before the monsoon while the highest peaks are usually during the monsoon. The water table in the wet reach rises by ~1.5 m during the monsoon, but it is otherwise nearly constant at ~3 m below the surface, even during weak monsoon years (Figure 2.10b). In the intermediate reach, streamflow variability is high, as there is generally only flow during monsoon and winter rains (in years with a very strong monsoon, both streamflow and groundwater stay high all year round), when discharge approaches values of the wet reach (Figure 2.10c). Groundwater has a flashy response to streamflow (Figure 2.10d) with similarly high variability (2-3 m fluctuations) and lengthy dry periods of very low water table depths (below the sensor). During a weak monsoon, levels remain low for the whole summer.

Streamflow from the USGS gauge in the dry reach was only recorded from July 2001 to June 2002, so it is challenging to draw conclusions. The water table here again has no consistent elevation, but instead rises and falls with the streamflow regime during monsoon and winter rains (Figure 2.10e), apparently supported by a geologic control at a minimum value of 4 m below the surface. A strong monsoon keeps water table depth above 3 m all year round, with a high at 1 m during the summer, but a year with a weak monsoon will see the water table drop under 3 m, end even the winter rains will not be able to bring the water back up.

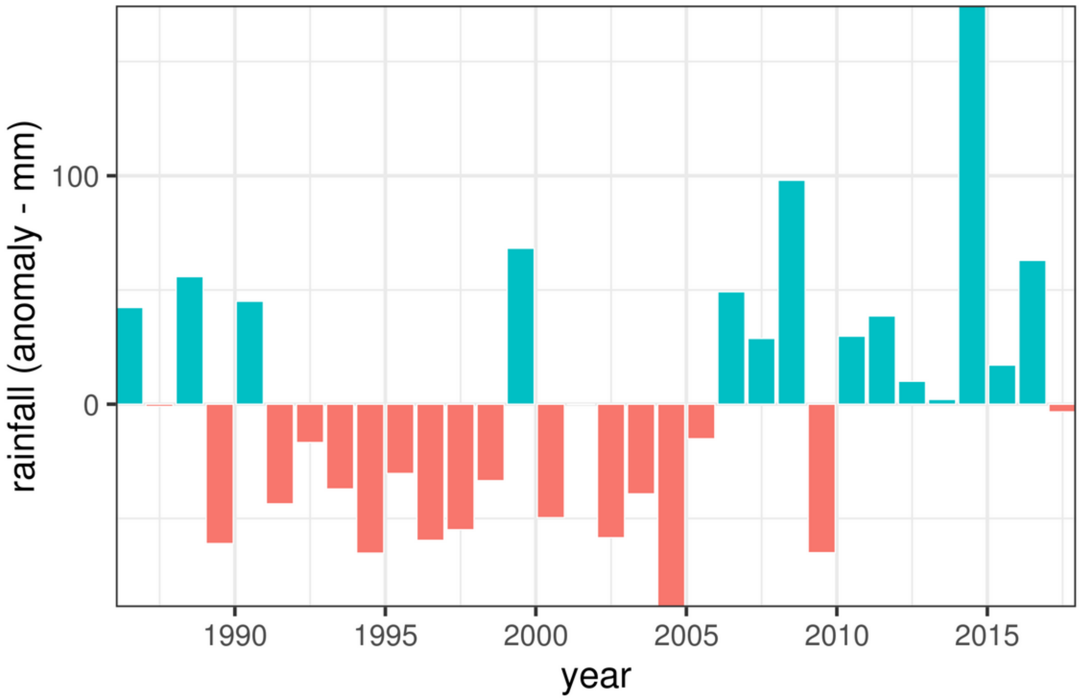

Figure 2.9: Monsoon precipitation anomaly (anomaly = annual total monsoon – average for the time series). In blue are the stronger than average years, in red are the weaker than average years.

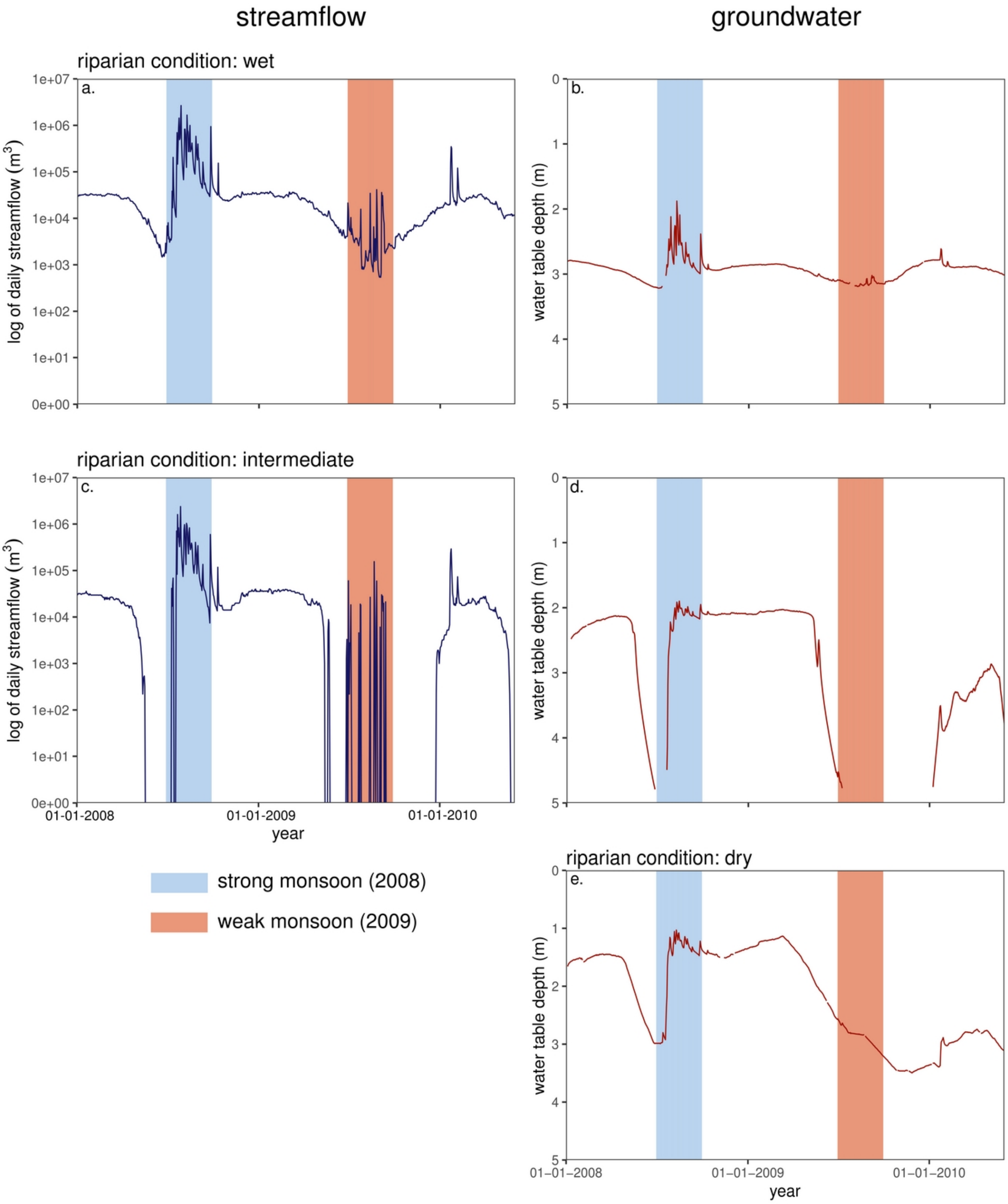

Figure 2.10: Time series (2008-2010) for streamflow and groundwater under the San Pedro for each flow permanence class: wet (a, b), intermediate (c, d), dry (e). Strong and weak monsoons are shown in blue and red, respectively.

2.4.2 Vegetation

The distribution of NDVI values for each vegetation class highlights the difference between sparse, small vegetation in the grasslands (NDVI median under 0.2) and trees (median always above 0.4) (Figure 2.11a). Some vegetation communities display a strong response to the monsoon rains with an important green-up mostly noticeable for the grassland, the upland forests and the canyons ($P < 2 \times 10^{-16}$), while the San Pedro riparian forest is less responsive ($P = 8.6 \times 10^{-5}$) (Figure 2.11a). Examining the San Pedro riparian cottonwoods more closely in relation to flow permanence, trees in all reaches show a significant increase in NDVI values after the monsoon, though the wet reach display the smallest increase ($P = 6.8 \times 10^{-6}$) while the intermediate and dry reaches show a stronger increase in vegetation greenness ($P = 2.7 \times 10^{-8}$ and $P = 6.2 \times 10^{-8}$, respectively) (Figure 2.11b).

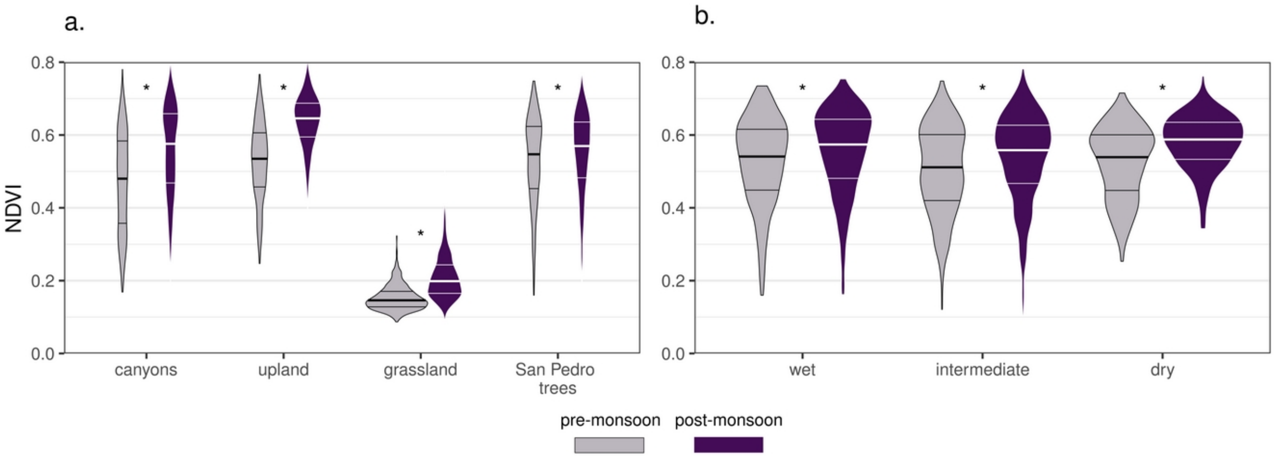

Figure 2.11: Distribution of pre-monsoon and post-monsoon NDVI values over time (1986-2017) for the Huachuca Mountains canyon riparian vegetation, mountain forest, riparian trees along the San Pedro River and grasslands (a) and flow permanence (b). Violin plots showing median and interquartile range, asterisks indicate significant differences between pre- and post-monsoon ($P < 0.05$).

A pixel-wise trend analysis was performed over the 30 years of the time series as a means to look at the long term changes in vegetation over the whole study area. For each pixel, the value is the slope of a linear model applied to the values taken by this pixel from 1986 to 2017. San Pedro trees (both pre- and post-monsoon) follow a significant positive trend, while pre-monsoon upland vegetation shows a significant negative trend (over 50% of pixels in each of these classes show a significant trend according to a Mann-Kendall trend test). Likewise, trees in wet and intermediate reaches (both pre- and post-monsoon) are characterized by a significant positive trend.

The linear trend analysis highlights multi-decadal differences between the various vegetation communities (Figure 2.12). Riparian areas along the San Pedro River tend to have increasingly dense vegetation through time, with rising NDVI values (median annual trend of $\sim 10^{-5}$) in both seasons. In contrast, grasslands and mountain vegetation (both upland and canyons trees) show no annual trend, or a negative trend for upland forests (median of -6×10^{-6} for pre-monsoon upland). Upland and grassland vegetation are the only groups with a significant difference between pre- and post-monsoon, with a stronger negative trend before the monsoon (Figure 2.12a). Along the San Pedro, the NDVI trends depend on flow permanence, with wet and intermediate reaches characterized by a wide distribution but an overall increase (median $> 10^{-5}$), and dry reaches displaying a narrower range of values and a median $< 10^{-5}$ (Figure 2.12b). The wet and intermediate pre-monsoon values are also significantly different from the dry reach values.

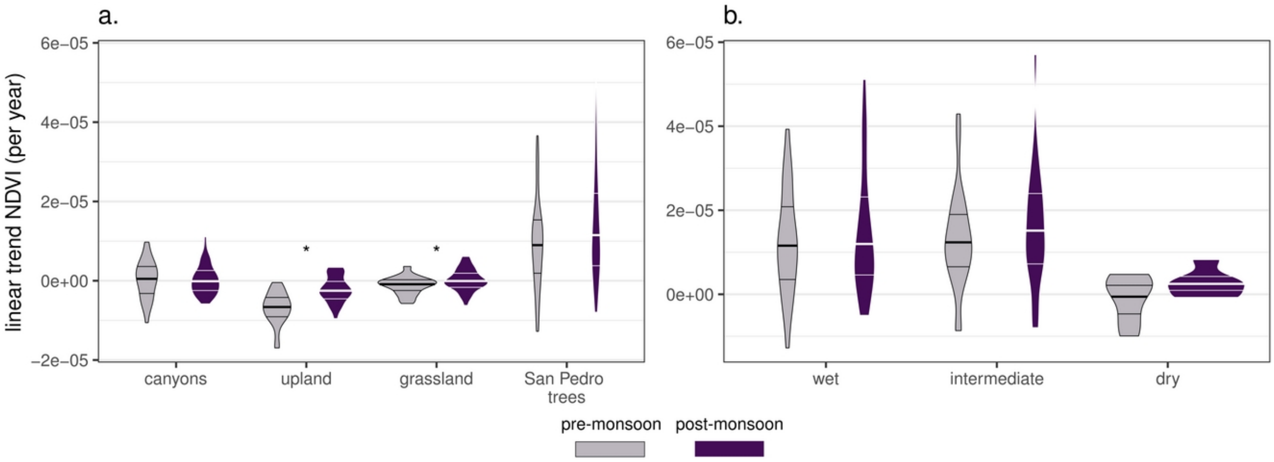

Figure 2.12: Distribution of time series trends. The values used are the slope of a linear model fitted to the 1986-2017 time series for each sampling point in the following classes: Huachuca Mountains canyon riparian vegetation, mountain forest, riparian trees along the San Pedro River and grasslands (a) and flow permanence (b). Violin plots showing median and interquartile range, asterisks indicate significant differences between pre- and post-monsoon ($P < 0.05$).

The years since 2006 are dominated by the monsoon (Figure 2.6). The annual amount of water has not changed but at least half of this water is falling/flowing during the monsoon, meaning that vegetation receives more water in a smaller time step and water distribution is shifting, with potential consequences on water storage and accessibility to vegetation. Riparian trees along the San Pedro River display a different trend through time than other vegetation communities. Grassland and trees in the Huachuca Mountains (in canyons and uplands) followed a slight decrease before 2006, and either a slight increase (upland and canyons) or no change (grassland) from 2006 to 2017) (Figure 2.13a, b, d). The riparian trees along the San Pedro show an opposite pattern of NDVI increasing before 2006 and slightly decreasing after 2006 (Figure 2.13c), which is mainly due to trees in the intermediate and wet reaches increasing from 0.4 to 0.6 (Figure 2.13f, g) while the trees in the dry reaches remain around 0.6 for the whole period (Figure 2.13e).

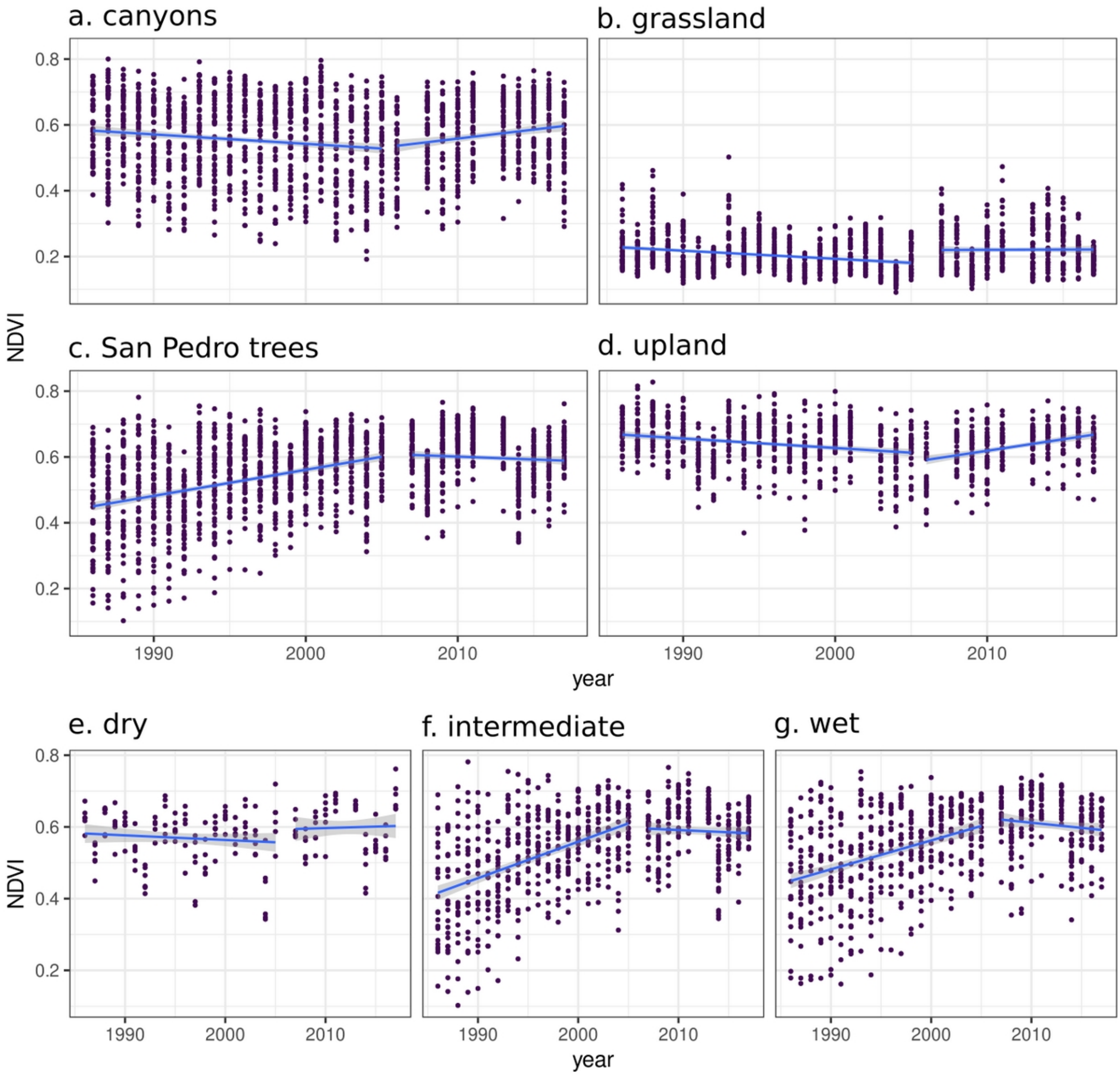

Figure 2.13: Time series of NDVI values of all sampling points (post-monsoon average for each year) and linear trends before and after 2006 for the canyons (a), grassland (b), all San Pedro trees (c), upland trees (d), and each flow permanence class along the San Pedro (e to g).

Grasslands and the San Pedro riparian forest show a significant increase in NDVI values, with the San Pedro trees displaying the strongest increase (Figure 2.14a, median value goes from 0.55 to 0.61), mostly due to an increase in vegetation greenness in the wet and intermediate ranges (Figure 2.14b). When looking at trends, the opposite patterns are apparent, with a significant shift from weak negative trends to strong positive trends for all the vegetation communities except the San Pedro trees (Figure 2.14c). Along the San Pedro, wet and intermediate reaches went from strong positive trends before 2006 to negative trends after 2006, while the dry reaches showed no significant change in a trend that stays negative before and after 2006 (Figure 2.14d).

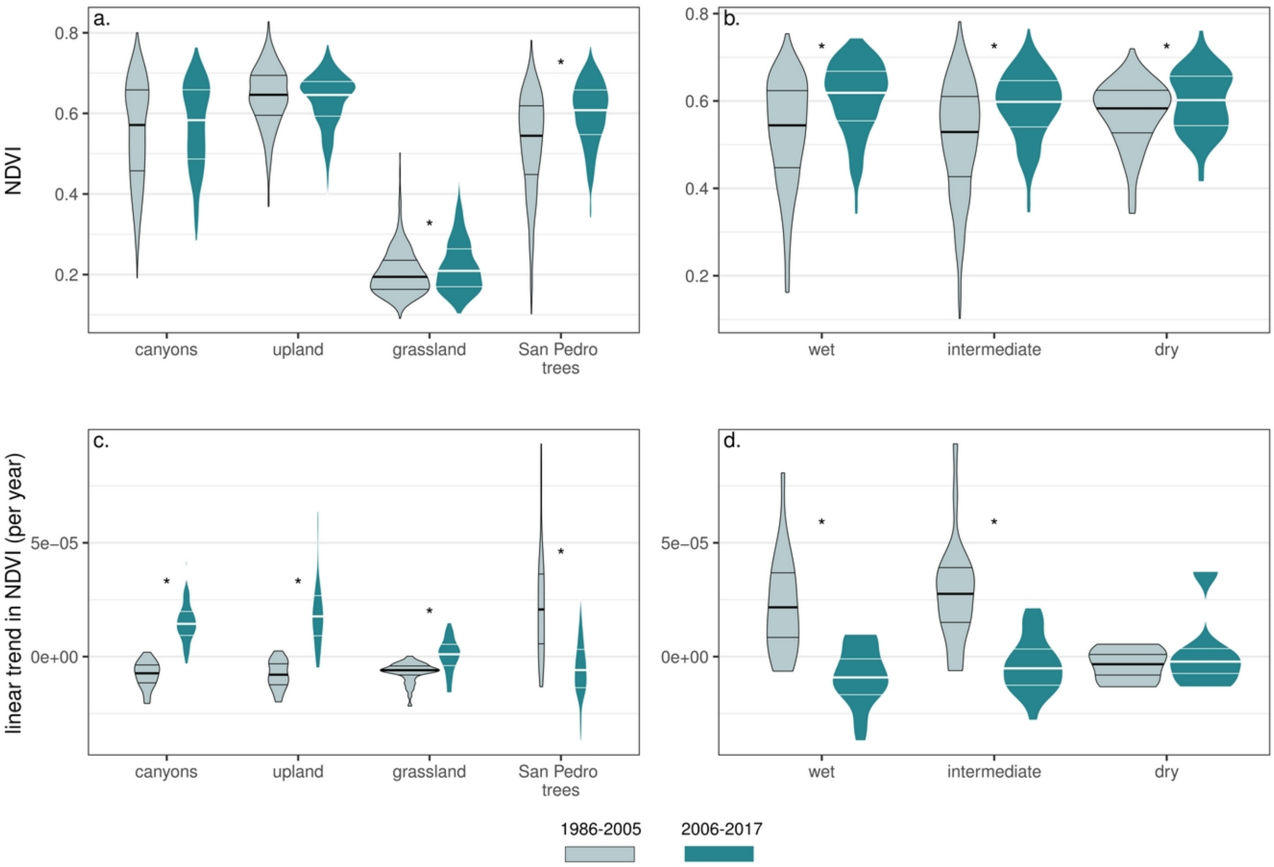

Figure 2.14: Distribution of pre-2006 and post-2006 NDVI values for the canyon riparian vegetation, mountain forest, San Pedro riparian trees and grasslands (a) and flow permanence (b). Distribution of time series trends before and after 2006 for canyon riparian vegetation, mountain forest, San Pedro riparian trees and grasslands (c) and flow permanence (d). Violin plots showing median and interquartile range, asterisks indicate significant differences between pre- and post-monsoon ($P < 0.05$).

2.5 Discussion

I set out to explore variations in water availability and vegetation responses within this subregion of the San Pedro River basin based on diversity in landscape attributes (e.g., topography, subsurface geology). This analysis is particularly important considering climate change projections for the Southwest USA that portend more prolonged and extreme drought conditions, which may affect vegetation in profound ways (Asner et al., 2016; Ault et al., 2016; Cayan et al., 2010; Choat et al., 2018; Cook et al., 2015; Parolari et al., 2014).

2.5.1 Hydrology and water availability

Most of the annual precipitation in this region of south-east Arizona is partitioned by season and elevation into rainfall or snow accumulation on the Huachuca Mountains, but it is only the intense monsoonal rains that generate significant runoff in ephemeral streams, which deliver flow to the San Pedro River (Goodrich et al., 2008; Singer & Michaelides, 2017; Thomas & Pool, 2006). Rainfall in the mountains infiltrates into fractured rocks as temporary aquifer storage (Coes & Pool, 1984) before periodically emerging at the surface in ephemeral and intermittent canyon channels, which are bounded by canyon riparian forests, on their downslope journey to the lowlands (Jaeger & Olden, 2012; Meixner et al., 2016; H. G. Shaw, 1999). At lower elevations, the water table dips far below the surface in the grasslands, before becoming shallow again near the San Pedro River, where there are strong streamflow-groundwater interactions (Coes & Pool, 1984). These interactions, expressed largely during the summer monsoon, recharge the shallow water table around the channel, briefly raising the water table level and saturating the soil close to the surface. Once the water table declines again, the residual moisture in the soil pores is available to vegetation for at least part of the remaining growing season. The reasons for the slow water table decline over time under the grasslands are unclear, although it could have to do with pumping for water supply to Fort Huachuca and Sierra Vista (Gungle et al., 2019; Stromberg & Tellman, 2012, p. 299) and/or a deeper plunging of the water table below the grasslands, for example due to lower mountain front recharge under declining snowpack.

The other major factor controlling the distribution of water along the San Pedro River is flow permanence along the channel, which reflects the presence of subsurface geologic layers (bedrock and clay layers) that support a locally perched alluvial aquifer in some reaches. However, depth to bedrock is not the main factor governing water table characteristics in this and many other riparian systems (except around Charleston); the spatial distribution of alluvial deposits play a dominant role. There is evidence for a diversity of such sedimentary controls along the SPRNCA, which essentially stratify this area into wet, intermittent, and dry reaches. Wet reaches are over bedrock or river alluvium layered with clay and silt that maintains the alluvial aquifer close to the surface, sustains perennial flow, and enables flashy responses of streamflow-groundwater interactions to monsoonal rainfall. Relatively dry (intermediate and dry) are more over stretches of sand and gravel alluvium, providing limited benefits to moisture retention, so the flow series at these locations only responds to significant rainfall events, dropping back to zero flow for extended periods. The water table responses in these drier reaches are also flashy with several meter variations depending on the driving flow. When the flow is low or zero, the water table drops down to its minimum, again supported by deeper geologic controls. There might also be a difference in lateral underground flow from the surrounding mountains, with wet reaches receiving more water than dry reaches.

In wet years, high monsoonal rainfall may minimize the importance of geologic controls by, for example, creating higher sustained flows, strong streamflow-groundwater interactions, and a shallower water table, even in the drier reaches. These would generate high water storage in the riparian zone similar to that in wet reaches. In years of very low monsoonal rainfall, however, the apparently strong geologic support to the water table in wet reaches creates large differences in water availability compared to drier reaches, where the streamflow is low or zero and water table drops substantially. Thus, if climate change trends toward a stronger monsoon, I would expect an equalizing in moisture availability across all reach types in the riparian zone. However, if the monsoon becomes weaker, it could exacerbate the moisture storage differences between reaches.

2.5.2 Vegetation responses to water availability

With differences in water distribution and storage in the landscape, there are also differences in vegetation communities, species as well as vegetation density. At high elevation, oak-sycamore forests of the canyon riparian corridor and the oak-fir forests on slopes share similar seasonal NDVI distributions, suggesting that they use similar sources of water (seasonal flow passing through fractured rock), though differences in trends might be explained by their difference in position in the landscape (bottom of canyons vs steep slopes). In lower elevation grasslands, vegetation relies on rainfall-derived soil moisture, as the water table is below the rooting zone, producing notable green-up after monsoon rains.

Along the San Pedro, the deciduous trees of the riparian forest green-up earlier than other vegetation communities in the study area, perhaps because they are phreatophytes that have access to groundwater before the monsoon starts and grow their leaves early with little change in leaf area or chlorophyll content over the growing season (Brock, 1994). The monsoon rains generate significant streamflow, which recharges the shallow water table, thus increasing hyporheic soil moisture within the riparian corridor. In perennial flow reaches, the riparian corridor is dense, with a closed canopy of cottonwoods and scattered willows, so the vegetation water requirements are high (Leenhouts et al., 2006). This community is supported by a sustained, shallow water table and strong interactions with streamflow, even during years of low monsoon (Leenhouts et al., 2006). In drier stretches of the river, sparse patches of old cottonwoods, mesquite (*Prosopis velutina*) and tamarisk (*Tamarix* sp.) (Stromberg et al., 2006, 2010), subsist on moisture in the unsaturated zone generated during brief flow events and water table rises, which appear to be favourable to adult trees in small numbers, but apparently limit the establishment of a denser forest.

2.5.3 Trends

The relationships between water fluxes and vegetation responses provide insight into how dryland vegetation communities have responded to climate over recent decades, as well as their likely response to different scenarios of climate change. Both upland forests and grasslands show a significant positive change in trend after the monsoon which could be driven by the fact that these vegetation communities are more reliant on monsoon rain, and the shift to a monsoon-dominated precipitation regime. In the grasslands, an increase in monsoon rainfall paired with no change in PET brings an increase in soil moisture available to plants during the growing season. The riparian trees of both canyons and San Pedro River might not be as sensitive to the monsoon, thanks to the support of a shallow water table, and their NDVI trends pre- and post-monsoon show no significant change.

The partitioning of rainfall and associated seasonal water availability has changed in this region of the southwest USA, which has impacted all vegetation communities. With the monsoon rains becoming the predominant water source after 2006, canyons, upland forests and grasslands have shown a shift from negative NDVI trends to positive trends, which suggests that the annual distribution of rainfall and the strength of the monsoon are important factors for the evolution of these communities. The San Pedro riparian forest shows the opposite pattern and the flow permanence classes did not react in the same way to these changes in rainfall distribution. Both wet and intermediate reaches have seen an increase in NDVI values over time but a shift from positive to negative trend, which might indicate a forest that has grown, has reached maturity and is now declining (high NDVI values but a slightly negative trend). Looking at NDVI trends of the last few years can be used as an early warning system by highlighting recent changes in a vegetation community. In the dry reaches, the median NDVI was high before 2006 and remained high afterward, with no change in trends. This suggests that the scattered trees of the dry sites had reached maturity before 2006. Since 2006, the monsoon rains increased and overtook winter rains in terms of annual moisture contribution, apparently providing vegetation in the dry reaches a new source of moisture to exploit, providing a relative advantage compared to wet and intermediate reaches, and allowing them to maintain a trend close to zero.

The strength and trends of the North American Monsoon is a subject of debate, probably because of near decadal cycles of strength and weakness based on ocean-atmosphere connections and opposing trends of annual precipitation and precipitation intensity (Luong et al., 2017; Pascale et al., 2017; Singer & Michaelides, 2017). Regardless, our results suggest periods of stronger monsoons will maintain a more consistent source of moisture for riparian forests along all reaches of the San Pedro, overprinting the effects of subsurface geology. Strong monsoon rains would maintain a shallower water table and replenish soil moisture every summer. However, a prolonged period of weaker monsoons may result in reduced water storage and moisture availability in intermediate and dry reaches, which might make them even less favourable for riparian vegetation, even for older trees with deeper roots. Thus, strengthened decadal cycles of strong and weak monsoons in the Southwest

USA may result in prolonged periods of moisture stress followed by rapid greening for dryland riparian forests, especially for forests with no benefits from regional groundwater drainage and subsurface geology. When the currently mature trees of the San Pedro riparian forest start dying and leaving room for the establishment of younger trees, climate-controlled water distribution will affect the composition, density, and health of the successional riparian forest community. Future climate projections call for a weakened monsoon due to more stable air masses across the region under global warming (Pascale et al., 2017). If this comes to pass, I suspect that this may ultimately lead to significant die back of dryland riparian forests across the region. Whereas, if there is a consistent intensification of the monsoon (Luong et al., 2017), there is great potential to create a greener and more continuous riparian gallery forest. To complete this overview of water distribution and availability to vegetation in the landscape and through time, flow distribution in the canyons of the Huachuca Mountains and in the valley floor washes also needs to be taken into account.

2.6 Conclusion

In this chapter, I analysed long-term changes in water fluxes and vegetation greenness across a range of vegetation communities over a broad dryland region of the Southwest USA. I showed the importance of the driving climate in controlling water availability to dryland vegetation. I also illustrated the importance of subsurface geology, with its role in controlling water availability and vegetation distribution along the San Pedro River. Additionally, I identified distinct monsoonal cycles over a multi-decadal time series, which have affected subsurface water availability to a range of vegetation communities. This regional expression of the climate system is strong enough to overprint the effects of local geology in the strong monsoon phase, allowing trees in the dry reaches to maintain their leaf density, while trees of the wet and intermediate reaches show a decline in greenness trends, suggesting that they are reaching end of life. The renewal of water-limited riparian forest communities that have reached their maturity is strongly dependent on future shifts in water distribution and the availability of new surfaces for phreatophyte recruitment and establishment. Our results suggest that climate-controlled water availability is a first-order control on vegetation distribution and health in different vegetation communities within arid regions, subject to spatially varying constraints on water table support.

Chapter 3
High resolution spatiotemporal patterns of flow at the landscape scale in montane non-perennial streams

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3.1 Abstract

Intermittent and ephemeral streams in dryland environments support diverse assemblages of aquatic and terrestrial life. Understanding when and where water flows provide insights into the availability of water, its response to external controlling factors, and potential sensitivity to climate change and a host of human activities. Knowledge regarding the timing of drying/wetting cycles can also be useful to map critical habitats for species and ecosystems that rely on these temporary water sources. However, identifying locations and monitoring the timing of streamflow and channel sediment moisture remains a challenging endeavour. In this chapter, I analysed daily conductivity from 37 sensors distributed along 10 streams across an arid mountain front in Arizona (United States) to assess spatiotemporal patterns in flow permanence, defined as the timing and extent of water in streams. Conductivity sensors provide information on surface flow and sediment moisture, supporting a stream classification based on seasonal flow dynamics. The results provide insight into

flow responses to seasonal rainfall, highlighting stream reaches very reactive to rainfall versus those demonstrating more stable streamflow. The strength of stream responses to precipitation are explored in the context of surficial geology. In summary, conductivity data can be used to map potential stream habitat for water-dependent species in both space and time, while also providing the basis upon which sensitivity to ongoing climate change can be evaluated.

3.2 Introduction

Intermittent and ephemeral streams are widely distributed across the globe and are particularly prevalent in drylands where there is strong coupling between climate, streamflow, and shallow groundwater over multiple timescales (S.-A. Chen et al., 2019; Messager et al., 2021a; Quichimbo et al., 2020). These streams often are the main source of moisture in otherwise dry landscapes, making them important hotspots of biodiversity (Bogan et al., 2015; Datry et al., 2014; Larned et al., 2010). As such, understanding the timing and controls of flow in ephemeral and intermittent streams is needed to map potential habitats in drylands, and how the distribution of these habitats might shift with climate change (Hammond et al., 2021; Zipper et al., 2021). In this chapter, I leverage a conductivity dataset from a network of sensors deployed in streams along an arid mountain front to investigate the spatiotemporal distribution of flow, map flow condition, identify potential controls on surface flow and sediment moisture, and establish a seasonal classification of flow for dryland streams. The resulting spatial and temporal maps of temporary flow can provide useful information for assessment of habitat suitability for a wide range of species, and to support improved interpretations of the linkages between climate forcing and mountain front hydrology.

Dryland regions, defined as areas where plant productivity is limited by water availability, cover about 41% of the land surface (Millennium Ecosystem Assessment, 2005) and are dominated by temporary streams that dry at least once per year (Messager et al., 2021a). Significant drylands include the southwestern region of the United States, where ~81% of streams are classified as non-perennial, a proportion which rises to 94% in Arizona (Levick et al., 2008; Nadeau & Rains, 2007). Non-perennial streams occasionally dry out (fully dry streambed), and can be classified as ephemeral or intermittent, with ephemeral reaches reaching surface flow only in response to rainfall, while intermittent reaches display cycles of drying and wetting (M. H. Busch et al., 2020; Gallo et al., 2020a; Levick et al., 2008, 2015). Perennial streams flow year-round supplied by groundwater discharge to the stream bed. Streams that alternate between perennial, ephemeral, and intermittent reaches are considered interrupted or spatially intermittent (Levick et al., 2008). Streamflow permanence is controlled by various environmental factors such as rainfall distribution, evaporative demand, topography, underlying geology, streambed composition, channel morphology and vegetation (Costigan et al., 2016; Goodrich et al., 2018; Levick et al., 2018; Shanafield et al., 2021; Singer & Michaelides, 2014), but climate-induced aridity (balance between rainfall and evapotranspiration) is considered an overarching key driver (Hammond et al., 2021; Sauquet et al., 2021). In areas with a seasonal distribution of precipitation, such as the region of the Southwest USA, affected by the North American Monsoon, flow permanence can follow this highly uneven temporal distribution (Eng et al., 2015; Singer & Michaelides, 2017).

High variability in upstream-downstream arrangement of perennial and non-perennial streams support a mosaic of habitats for plant and animal life (Boulton et al., 2017; Datry et al., 2014; Larned et al., 2010). Spatial and temporal variations in habitat patch distribution and composition lead to high watershed-scale species diversity (Burnett et al., 1998; Larned et al., 2010; Stromberg et al., 2015). In drylands specifically, the presence of these wet reaches contributes to the strong contrast in water availability between riparian areas and the surrounding arid landscape (Levick et al., 2015; Stromberg et al., 2015), leading to contrasts in flora and fauna both in terms of species composition and density (Chapter 2) (Goodrich et al., 2018; Levick et al., 2008). The denser vegetation of riparian forests and wetlands is used for foraging, nesting or as migration corridors and stopovers, cool and humid refuges, and seed dispersal corridors (Datry et al., 2014; Levick et al., 2008).

More frequent and severe droughts linked to climate change are projected to significantly alter flow intermittence patterns and hydrologic connectivity in dryland streams by increasing the number of zero-flow days and the length and frequency of dry channel reaches (Jaeger et al., 2014; Sauquet et al., 2021; Zipper et al., 2021). In the United States, a general decline in surface-water availability and soil moisture is expected across the southwestern region (Seager et al., 2013), which would dramatically impact ephemeral and intermittent channels. This water-availability decline, added to other stressors such as water pumping and other flow diversions, lead to loss of wetlands and the species they host (Hendrickson & Minckley, 1985). Knowing precisely where and when there is surface flow is essential to map the distribution of potential streamside habitats, but also to anticipate habitat distribution shifts induced by climate change (D. C. Allen et al., 2019; Jaeger et al., 2014; Sauquet et al., 2021).

To understand how flow permanence varies along streams across a mountain front within a dry climatic region, I use electrical conductivity sensors to detect dryness and wetness of the streambed (Blasch et al., 2002; Chapin et al., 2014; Jaeger & Olden, 2012). These sensors can be used in ephemeral headwaters to map perennial and intermittent flow (Adams et al., 2006; Assendelft & van

Meerveld, 2019). The fine spatial resolution and high temporal frequency of observations are capable of capturing flow variability (Arismendi et al., 2017; Larned et al., 2011) to support classification of ephemeral and intermittent streams and better understand the environmental factors governing water distribution (Jensen et al., 2019). A similar method was used by Gallo et al. (2020a) across the same mountain front with a limited number of sensors across three canyons, and focusing on rainfall and sediment hydraulic conductivity. I use daily conductivity from 37 sensors across 10 canyons to compare seasonal flow timing to precipitation and underlying geology. This high spatial and temporal resolution dataset, which provide daily information for all the main headwater streams on the north-eastern slopes of the mountain range, allows for an understanding of landscape-level flow patterns and helps decipher regional (rainfall) and local (geology) environmental controls on flow permanence.

3.3 Methods

I investigated spatial and temporal variability of flow by mapping daily electrical conductivity (EC) values across my study site and compared these values to daily rainfall. I then sorted each sensor in a seasonal classification to link flow condition to seasonal rainfall. This response is evaluated further by comparing rainfall and EC values over several years of variable precipitation distribution. Lastly, I compared stream reaches and their seasonal classes to permeability of the underlying geology to examine the role of geology as a potential factor to flow patterns in non-perennial streams.

3.3.1 Study site

The study site spans 10 non-perennial streams spread across the eastern side of the Huachuca Mountains, a mountain range in southeastern Arizona that is part of the Madrean Sky Islands (Figure 3.1). The Madrean Sky Islands are scattered mountain ranges covered by oak-pine forests surrounded by low and flat valleys of semi-arid grasslands and desert scrub (Levick et al., 2018; López-Hoffman & Quijada-Mascareñas, 2012). The stream network consists of a series of intermittent and ephemeral reaches connecting scattered perennial reaches. The streams of interest are named for their canyons of drainage: Ramsey (R), Brown (B), Tinker (T), Garden (G), Woodcutters (W), Rock Spring (RS), Huachuca (H), Split Rock (SPR), Slaughter House (SL) and Blacktail (BT).

Streamflow is fed by rainfall and to a lesser extent by snowmelt and the local water table. Short but strong monsoon storms that occur from July to September comprise ~60% of annual rainfall, with less intensive winter precipitation providing the remainder. The driest season occurs before the monsoon, from May to June. Precipitation is greater at higher elevations (Section 1.4.1). The monsoon brings intense thunderstorms that turn into runoff and floods, while milder winter rains and snowmelt more readily infiltrate and provide soil moisture (Loik et al., 2004; Vera et al., 2006).

The streams of the Huachuca Mountains cross over a diversity of geologic units (mudstone, limestone, quartzite, and granite), as well as several faults (Brown et al., 1966), before reaching the lowlands. Channels have cascade and step pool morphology at the upper extents typical of steep headwater streams and transition to pool riffle morphology in the downstream valley (Wohl & Pearthree, 1991). The valley surrounding the mountains is composed of permeable basin fill, terrace deposits, and stream alluvium. Water crosses the valley underground within the basin fill (or in washes during the strongest monsoon events) to reach the two main intermittent rivers draining the area: the San Pedro River to the east, and the Babocomari River to the north (Figure 3.1) (Gungle, 2006; Levick et al., 2008).

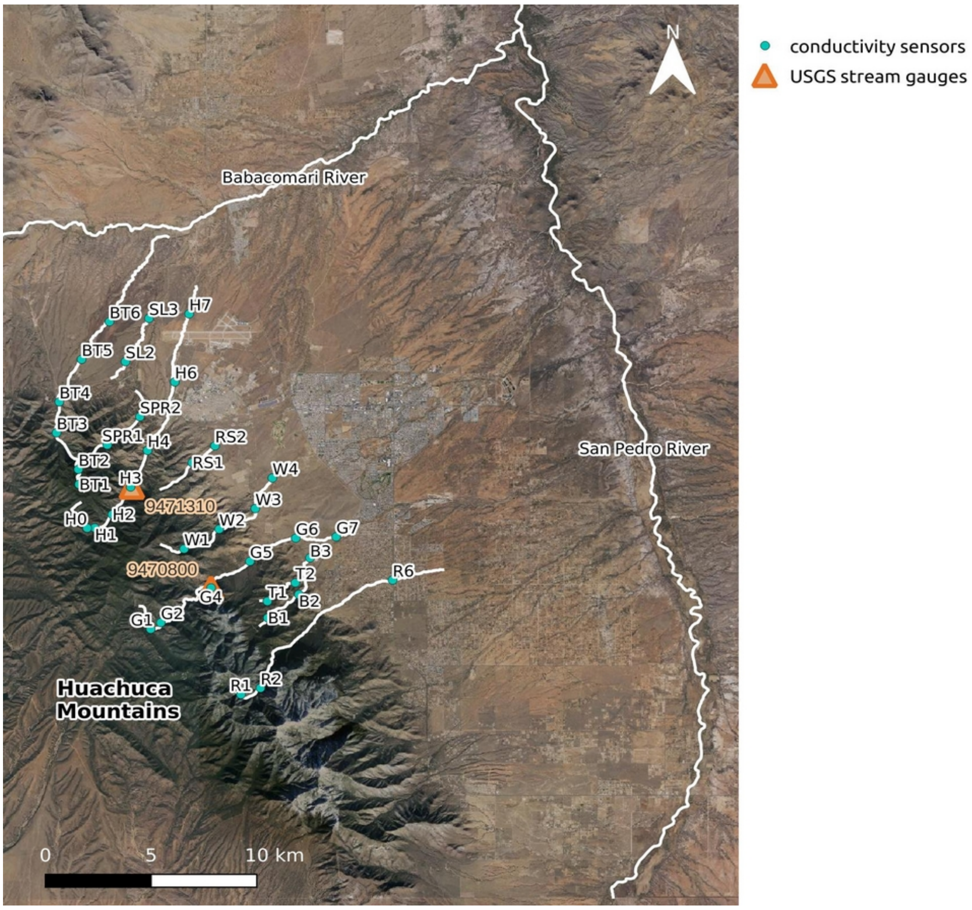

Figure 3.1: Study area with streams, location of sensors (BT: Blacktail; SL: Slaughter House; SPR: Split Rock; H: Huachuca; RS: Rock Spring; W: Woodcutters; G: Garden; T: Tinker; B: Brown; R: Ramsey canyons) and location of USGS stream gauges along Garden Canyon and Huachuca Canyon.

3.3.2 Sensor array

A total of 37 electrical conductivity (EC) sensors were installed along 10 streams of the Huachuca Mountains and operated between 2010 and 2014 (Chapter 1). Originally, 44 sensors were installed, but seven were omitted from this study because of short recording periods or quality issues (Figures 3.1 & 3.2) (Sabathier & Jaeger, 2022). These sensors were initially installed to quantify flow condition (flow, wet, or dry) through both time and space, including longitudinal flow connectivity (Jaeger & Olden, 2012). Their high spatial and temporal resolution is useful for capturing responses to local and short-term climatic events over wide areas (Adams et al., 2006; Assendelft & van Meerveld, 2019; Jensen et al., 2019). These EC sensors recorded relative conductivity every 15 minutes, with large values reflecting surface water presence, and smaller values reflecting dry channel conditions. Conductivity values are considered relative to each other consistent with other studies (Jensen et al., 2019; Warix et al., 2021) as sensor values were not calibrated with a solution of known conductivity. As such, the measurement uncertainty is unknown. Analysis was based on relative changes in value as compared with supporting data (streamflow gauges) and known response of conductivity to changes in water presence (Figure 1.26). These sensors having all been built with the same methods and devices, and having been deployed at the same time, I considered that they would all react alike to changes in soil moisture and surface water. The data collected can be used to detect onset and end of flow in non-perennial streams that are too small or too dry to be equipped with streamflow gauges (Blasch et al., 2002; Chapin et al., 2014; Goulsbra et al., 2009; Stromberg et al., 2015).

Figure 3.2: Conductivity sensor being put in place in 2010 (a) and close-up of the sensor in its protective white casing (b). Photos courtesy of Meryl Mims.

I used daily average values of relative conductivity from June 1, 2010 to May 31, 2011 to analyse flow permanence. This time frame was chosen because it covers a full year during which all 37 sensors operated without gaps. To investigate inter-annual variability of flow, I used four sensors (G2, H7, T1 and T2) that recorded EC for three years. Electrical conductivity records a low and constant value in dry sediment and progressively increases in wet sediments, finally exhibiting an abrupt increase at the onset of streamflow (Blasch et al., 2002; Goulsbra et al., 2009). Because the sensors are buried to a depth of <10 cm in the channel bed, sediment type or grain size distribution can affect the recorded values (Blasch et al., 2002), and may also cause a delay between the onset or cessation of flow and observed electrical conductivity (Adams et al., 2006; Blasch et al., 2002).

Sensors G4 (Garden Canyon) and H3 (Huachuca Canyon) were located close to U.S. Geological Survey (USGS) stream gauges 9470800 and 9471310, respectively. Daily streamflow data were downloaded from the USGS database (U. S. Geological Survey, 2022). The co-location of conductivity and flow data allowed me to directly classify conductivity in terms of flow permanence. I compared gauged stream discharge with adjacent EC sensor values for the same time-step and period (Figure 3.3) (Adams et al., 2006; Blasch et al., 2002; Stromberg et al., 2015). Some discrepancies between these datasets are expected, due to mismatches in precise location and measurement resolution, but their comparison provides an indication of how EC sensors react to flow conditions.

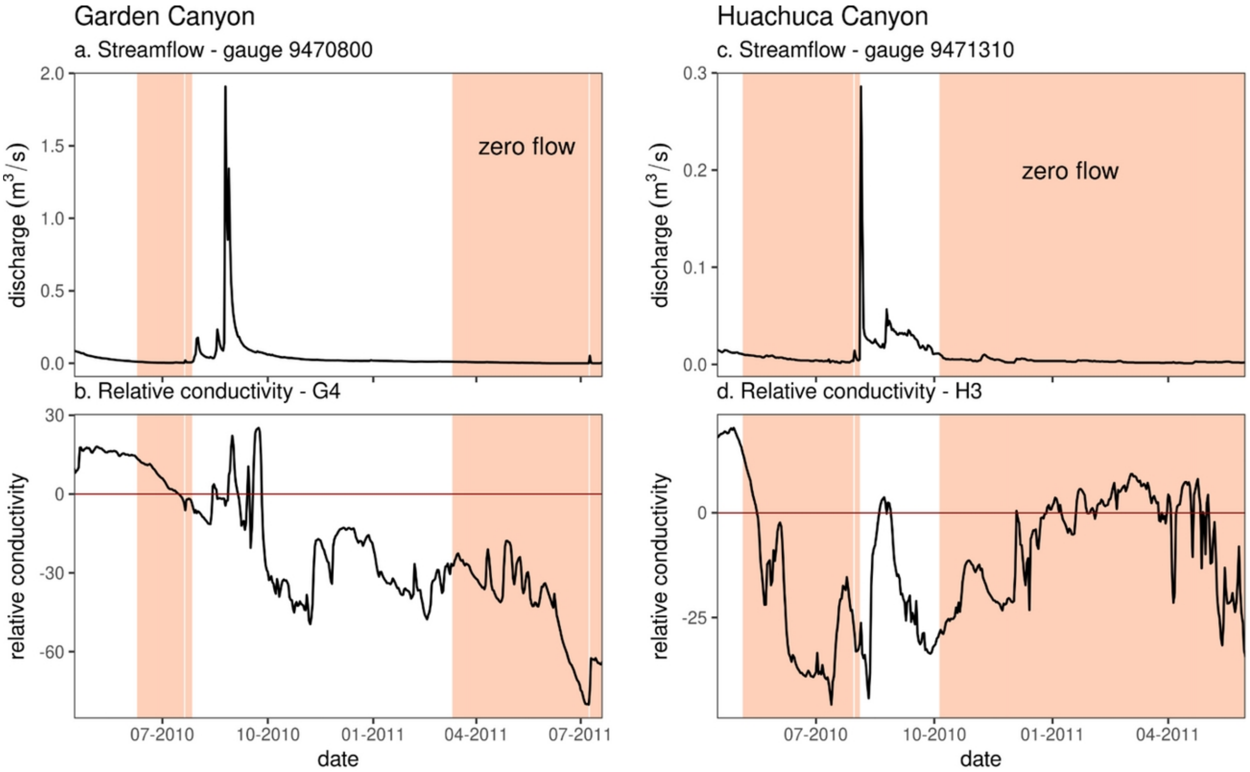

Figure 3.3: Discharge from USGS gauge (U.S. Geological Survey, 2022) and relative conductivity from sensors installed near gauge locations for streams in Garden Canyon (a, c), Huachuca Canyon (c, d) from June 2010 to May 2011. Shading highlights the periods of zero flow recorded at the USGS gauge (<0.01 m3/s). Conductivity values above 0 (red horizontal line) indicate surface water presence.

Acknowledging the uncertainty of this method and the potential influence of spatial variability in stream bed substrate, I focused on general categories covering a range of values. Thus, I built a scale between relative conductivity values and flow state (dry, wet sediment, and flow) (Table 3.1). A relative conductivity value of –90 is considered to represent dry sediment, as it is the lowest values reach by the sensors, and it is the only value that remains constant with no variations for days or weeks at a time. The threshold for water in the stream is 0 as this is the value reached during the sharpest conductivity peaks, following the strongest rainfall events (Figure 3.3).

Table 3.1: Relative conductivity measured by the sensors and its translation to flow condition and simplified state used for seasonal classification. Flow condition is represented by a set colour ramp through this chapter.

Relative conductivity	**Flow condition**	**Conor ramp**	**State for seasonal classification**
≤ -90	dry	🟥	dry
> -90 and < 0	wet	🟧🟩	wet
≥ 0	flow (or standing water)	🟦	

3.3.3 Flow condition classification

Relative conductivity was further classified into two classes: dry and wet, with the wet class including wet sediment and flow (or standing water) (Table 3.1). The seasonal classification of each sensor was established using data from June 1, 2010 to May 31, 2011. In July 2011, an apparent battery issue caused values for all sensors to drop ("dry sediment" baseline dropped from -97 to -138). An offset was applied for all values recorded after July 24, 2011 to bring the values back in line to pre-July 2011 levels (Figure 3.4) and used for the analysis of inter-annual variability.

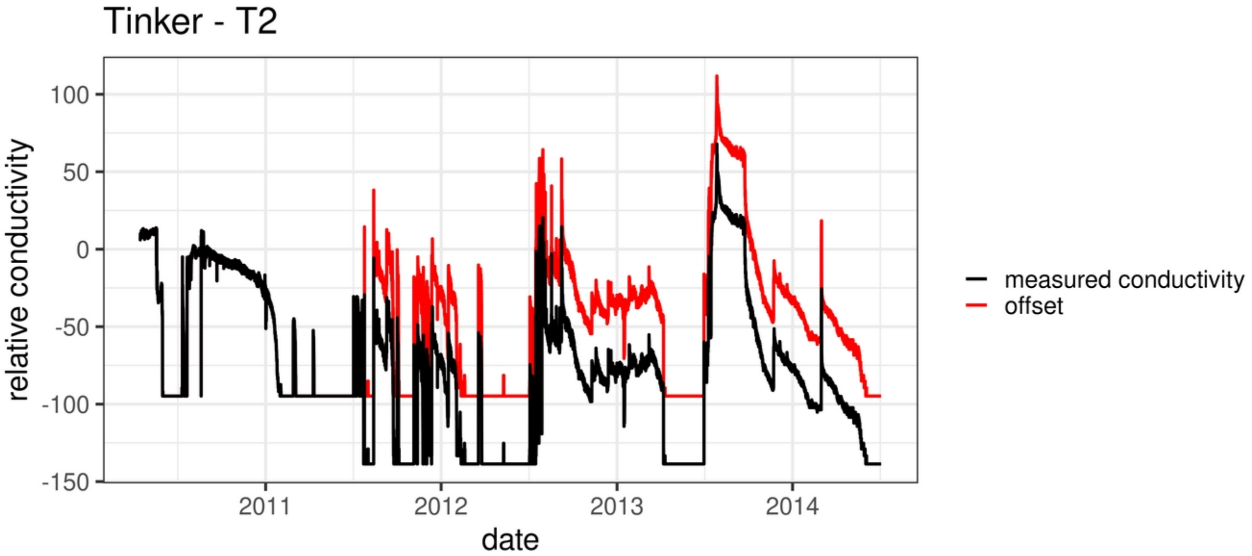

Figure 3.4: Daily relative conductivity measured by sensor T2 along Tinker Canyon, with the measured conductivity in black and the offset applied after July 24, 2011 in red.

Flow condition was reported as continuous (and represented by a continuous colour ramp, Table 3.1), with no hard limits between flow, wet sediment, and dry streambed classes in order to accommodate the potential uncertainties in EC values as a metric of flow. In maps displaying daily flow condition across sensors throughout the stream network, inverse distance weighting (IDW) interpolation in the longitudinal direction was used to reconstruct a continuous flow condition record at all points along the channel of each stream. While I do not know precisely how flow condition changes between sensors, the IDW interpolation provides a visualization tool to represent the dynamics of flow connectivity along each stream.

To understand the impact of rainfall on flow permanence, I used daily rainfall from PRISM (https://prism.oregonstate.edu/recent/), a gridded dataset at 4-km resolution, modelled by interpolation from ground stations, climate data and elevation (Daly et al., 2008). PRISM’s accuracy at a specific location is dependent on local ground station density, on top of the quality of the data received and errors introduced during the spatial interpolation. Comparison between PRISM data and the rainfall gauges of the U.S. Climate Reference Network shows a good accuracy of PRISM across the country (most mean daily differences <1 mm between datasets) (Buban et al., 2020), although inaccuracies increase in small mountain ranges with low gauge density (McEvoy et al., 2014). Rainfall from June 2010 to May 2011 was only 351 mm, which is a characteristically dry year compared to the 30-years (1991-2020) average precipitation from PRISM of 409 mm. Rainfall distribution across the year was also slightly unusual, with a stronger monsoon in 2010-2011 (308 mm versus 235 mm for the 30-y average), but a drier winter and dry season.

I defined a classification based on temporal distribution of flow condition throughout the year, a common way to classify ephemeral streams (Costigan et al., 2016; Eng et al., 2015; Sauquet et al., 2021). Daily rainfall was used to define the seasons based on precipitation distribution. I divided the year into four seasons, based on rainfall temporal distribution: dry spring (May-June), the summer monsoon (July to September), dry autumn (October-November) and wet winter (December to April). For each sensor, I counted the number of “wet” days (wet sediment or flow, relative conductivity above -90, Table 3.1) in each season. If the sensor measured flow or wet sediment for more than 50% of the season, then the whole season is considered “wet” for this sensor. All 37 sensors could then be assigned to one of six classes depending on when the stream reach is wet (always dry, wet during monsoon, wet during monsoon and autumn, wet during monsoon and winter, wet from monsoon to winter, always wet).

These classes were compared with ephemerality (frequency of dry days in a year) (Figure 3.5). I computed ephemerality at each sensor location as the frequency of dry days (conductivity under -90) in a year. This more general intermittency metric is useful to support the robustness of the EC classification system (Jaeger et al., 2014; Leigh et al., 2010). This frequency is computed for each sensor, and sensors are then grouped by their seasonal class.

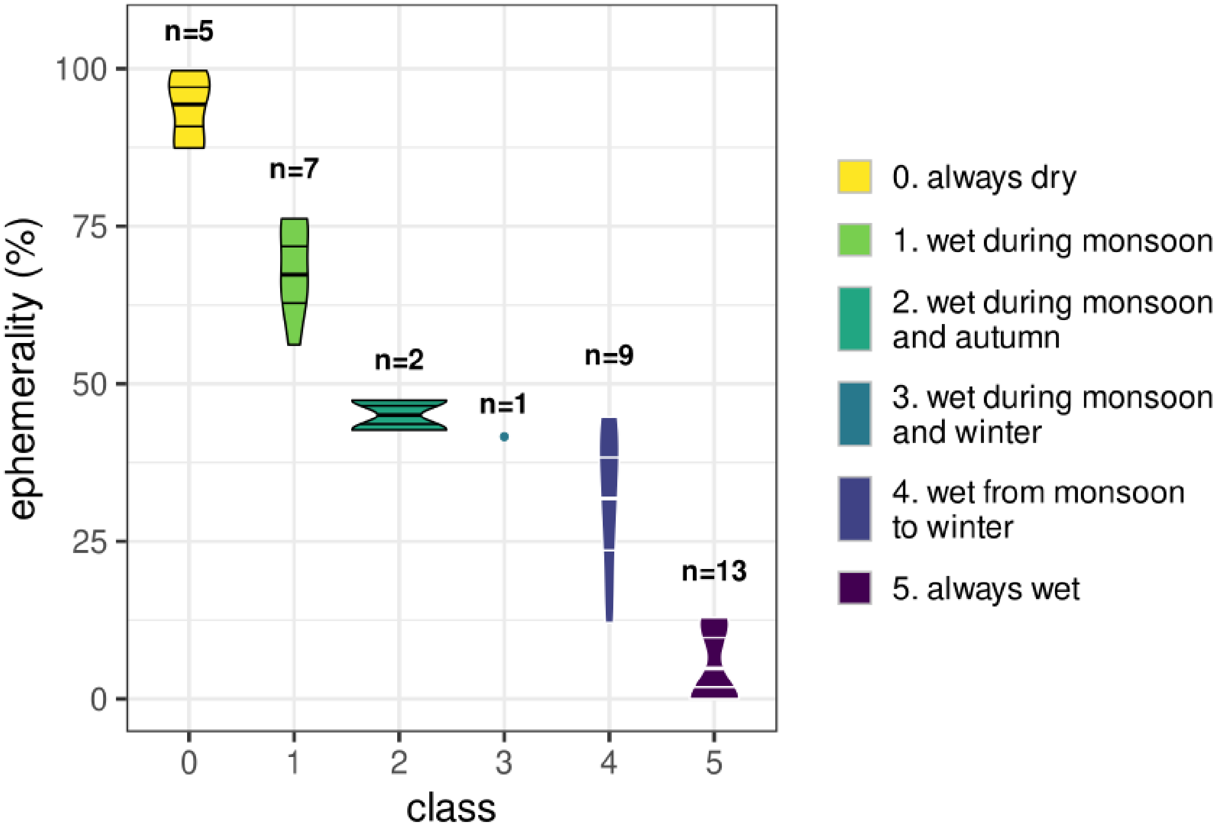

Figure 3.5: Violin plot of EC sensors classes by ephemerality (percentage of days with conductivity values below -90 in a year) grouped by seasonal class.

Underlying geology was also investigated for its association with local flow permanence; this was made possible based on the location of units with different permeability and fracturing (Goodrich et al., 2018; Larned et al., 2011; Levick et al., 2008). I used the hydrogeologic map and report from Brown et al. (1966), which provides information on geologic units, springs, and faults across the Huachuca Mountains to conduct a qualitative interpretation of the links between geology and flow condition. The hydrogeologic map, covering the north-east section of the Huachuca Mountains and the plain between the mountain front, the San Pedro River, and the Babocomari River, was digitized by hand in QGIS and augmented with information on lithologic unit permeability (Sabathier & Jaeger, 2022).

3.4 Results

Canyons of the Huachuca Mountains display a diversity of flow patterns within and among streams, as well as a variable responsiveness to rainfall. ER sensor arrays provide opportunities to quantify streamflow variability in both space and time. Here I present the results as maps of flow condition distribution across the mountain front, distribution of sensors by ephemerality and daily time series of flow condition for individual sensors.

3.4.1 Spatial and temporal variability of flow

Daily flow condition classification maps show aspects of spatial and temporal variability of flow condition across the landscape (Figure 3.6). The 2010 spring dry season (May to June) received its first rain on June 29. By June 13, the number of sensors registering dry conditions increased from to 76% from 70% on June 1 (28 sensors out of 37, Figures 3.6a; 3.7a); I attribute the increase in dry sensors to moist sediments drying out. The sensors remained dry until June 30, the day following the first monsoon rain. I note that the five sensors that recorded surface flow remained steady throughout the season. During the three months of the monsoon (July to September), average rainfall increased to 3.3 mm/day, mostly falling in July and August with the strongest event registering 36.6 mm in a day (August 25, 2010) over the Huachuca Mountains (Figure 3.7a); all sensors registered flow or wet sediment on that day (Figure 3.6c), including 54% of sensors recording surface water. The sensors responded quickly to the August 25 event (Figure 3.7a), transitioning from 22% dry to all wet during the event. One stream never registered flow regardless of the volume of precipitation, and one sensor in Huachuca Canyon recorded surface water 97% of the year.

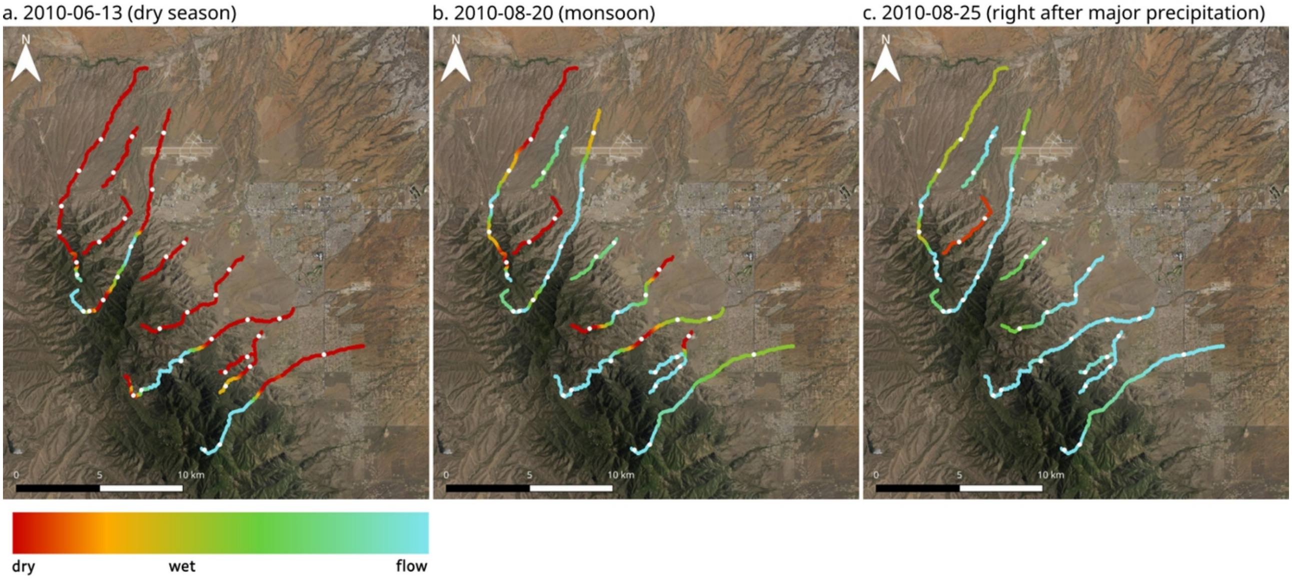

Figure 3.6: Maps of flow condition in the Huachuca Mountains based on inverse distance weighting interpolation from electrical conductivity sensors for three periods in time in 2010: dry season (a), monsoon (b) and monsoon right after a major storm (c). White dots represent sensor location. The interpolation between sensors is a visualization tool and does not represent the reality of flow between sensors. Flow lines are from the National Hydrograpahy Dataset (https://www.usgs.gov/national-hydrography/national-hydrography-dataset).

Flow permanence was patchy in streams of the Huachuca Mountains, with alternating drier and wetter reaches (Figure 3.7). For example, in Blacktail Wash, the upstream sensor BT1 was never dry during the study period, while BT2 was fully dry for 97% of the year although they are less than 1 km apart. Blacktail Wash stays wet during the monsoon and winter at the base of the mountain range (BT4 and BT5) but is mostly dry just 4 km downstream in the valley (BT6). In Huachuca and Garden Canyons, the driest reaches were located in downstream sections. The stream along Huachuca Canyon was wet close to the mountain top (H0, 1900 m.a.s.l.) and then showed patterns of drying downstream (H1 dried out 47% of the year and H2 dried out 56% of the year).

At ~1650 m.a.s.l., H3 and H4 are nearly perennial (no dry days) and the valley wash part of the stream heading toward the Babocomari River was the driest (H6 and H7). In Garden Canyon, most reaches responded directly to the monsoon, both at high and low elevation (G1, G5, G6, G7), while the mid-elevation (1700 m.a.s.l.) reaches are perennial (G2) or always remain wet (G4).

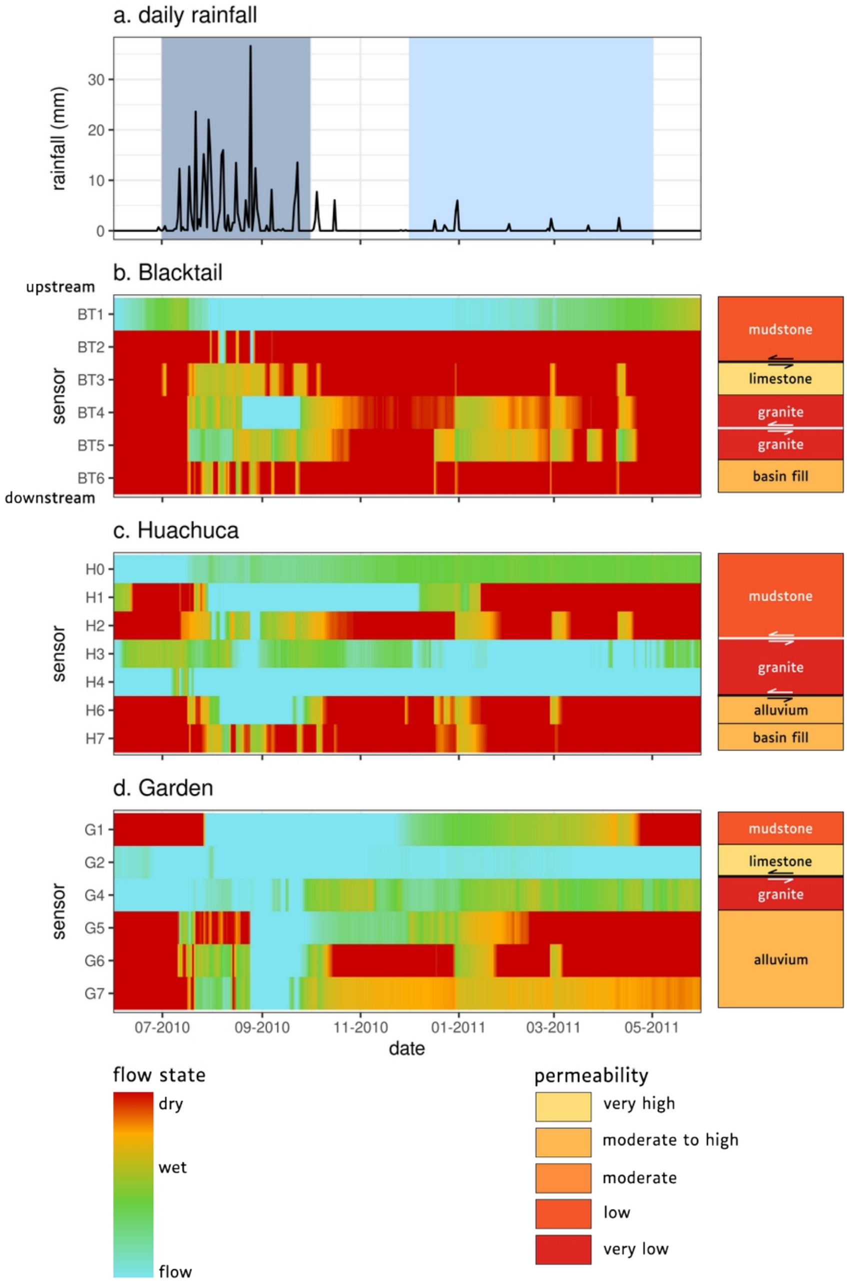

Figure 3.7 : Daily rainfall (in mm) averaged over the Huachuca Mountains with the monsoon season shown as a dark gray rectangle and the winter shown as a light blue rectangle (a) and heatmaps of daily flow condition based on classified electrical conductivity measurements along Blacktail Canyon (b), Huachuca Canyon (c), and Garden Canyon (d). For each sensor, stacked along the y-axis from upstream (top of plot) to downstream (bottom of plot), flow condition is represented by color. On the right is the geology under each sensor (permeability to water represented by color).

Stream reaches can be sorted by the temporal distribution of flow. Comparing the spatial distribution of flow against rainfall can also highlight the responsiveness of flow to precipitation. Figure 3.8 displays mean daily conductivity for each ephemerality class. There is no rainfall over the spring dry season (May-June). Precipitation increased to a total of 308 mm during the monsoon (July to September, dark grey rectangle) before declining to 19 mm during the dry autumn (October to November) and to 23 mm for the winter (December to April, light blue rectangle). The wide range of seasonal distribution of dry/wet cycles is shown in Figures 3.8a to 3.8f. The “always dry” (14% of sensors) and “always wet” (35% of sensors) classes are the most disconnected from rainfall, while other classes follow rainfall distribution patterns (get wetter during rainy seasons), though the monsoon is always the period with highest daily conductivity for all classes, and the only period where values above 0 (surface water) were reached. Even the “always dry” reaches responded to the strongest monsoon events. On the opposite end of flow permanence, the spring season is dry for all classes except the “always wet” reaches, which still drops to their lowest conductivity before the first monsoon event.

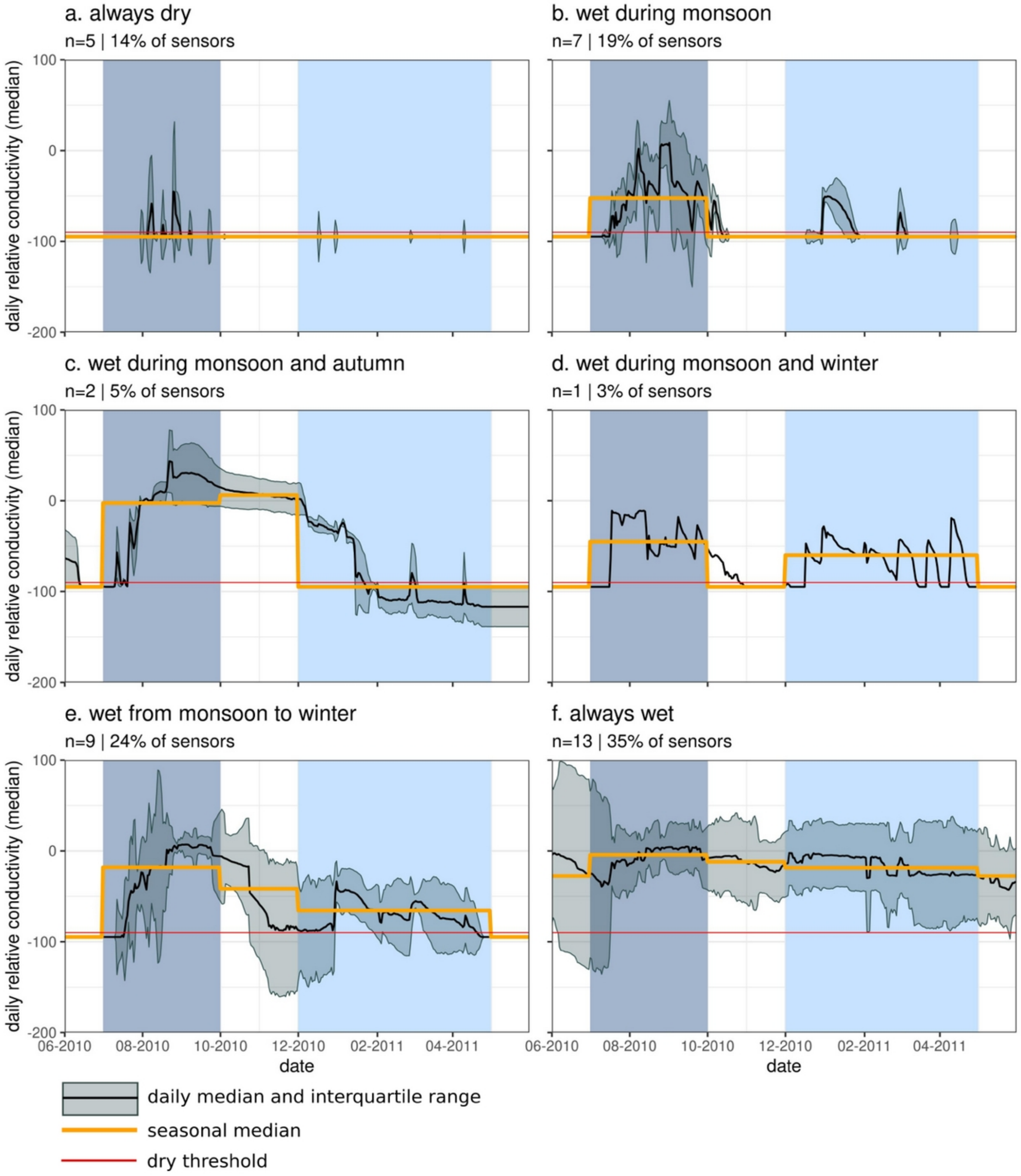

Figure 3.8: Time series of relative conductivity (daily median with interquartile range, seasonal median) for each seasonal class: always dry (a), wet only during the monsoon (b), wet during the monsoon and in autumn (c), wet during the monsoon and winter (only one sensor, d), wet from monsoon to winter (e), always wet (f). Dark gray shading highlights the monsoon season and light blue shading highlights the winter season. Conductivity values greater than -90 (red horizontal line) indicate wet flow state.

Daily records of surface water presence recorded by the EC sensors were used to categorize each reach into a seasonal classification (Figure 3.9, Table 3.2). While 35% of the reaches (13 out of 37) remained wet year long, only one of them flowed the entire year, and five reaches were dry all year. Seven reaches wetted up only during the monsoon and remained dry for the rest of the year, while only one became wet during both the monsoon and winter rain season. Nine reaches were wet during the monsoon and remained so until the end of winter, even during the autumn dry season, while only two remained wet only during the monsoon and autumn dry season.

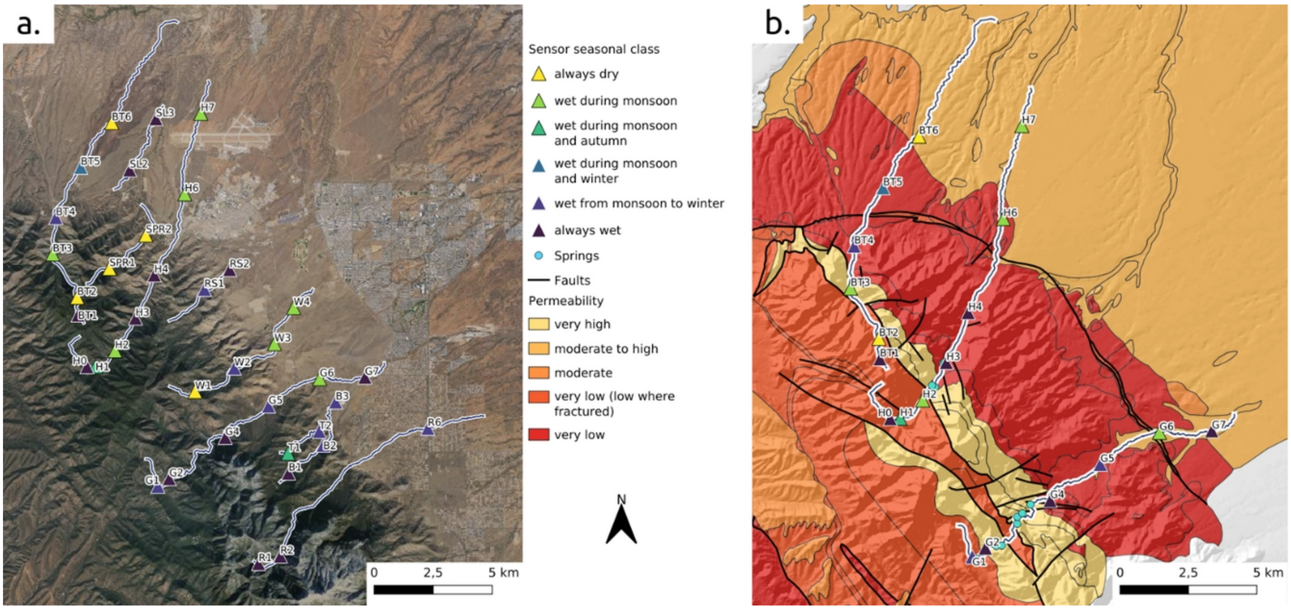

Figure 3.9: Map with location of conductivity sensors and their seasonal class (a). Hydrogeologic map with location of conductivity sensors and their seasonal class (for Blacktail (BT), Huachuca (H) and Garden (G) Canyons), location of springs and faults, and geologic units permeability to water (b). Seasonal class was established from sensors' measurements from June 1, 2010 to May 31, 2011.

Table 3.2: Flow permanence (in proportion of days) for each season (dry spring, summer monsoon, dry autumn and winter), seasonal class and underlying geology each sensor.

Stream	**Sensor**	**Class**	**Flow Permanence (% of days)**				**Geology**
			spring	**monsoon**	**autumn**	**winter**	
Brown	B1	always wet	100	100	100	100	granite
	B2	monsoon to winter	0	78	100	57	granite
	B3	monsoon to winter	0	60	100	93	stream alluvium
BlackTail	BT1	always wet	100	100	100	100	mudstone
	BT2	always dry	0	13	0	0	mudstone
	BT3	monsoon	0	78	5	8	limestone
	BT4	monsoon to winter	0	83	82	80	granite
	BT5	monsoon and winter	0	83	43	74	conglomerate + terrace deposits
	BT6	always dry	0	39	0	5	basin fill
Garden	G1	monsoon to winter	0	72	100	95	conglomerate
	G2	always wet	100	100	100	100	limestone
	G4	always wet	100	100	100	100	quartzite
	G5	monsoon to winter	0	71	100	50	stream alluvium
	G6	monsoon	0	86	25	25	stream alluvium
	G7	always wet	51	82	100	100	stream alluvium
Huachuca	H0	always wet	100	100	100	100	mudstone
	H1	monsoon and autumn	20	81	100	30	mudstone
	H2	monsoon	0	88	46	34	mudstone
	H3	always wet	100	100	100	100	granite
	H4	always wet	100	100	100	100	granite

Stream	**Sensor**	**Class**	**Flow Permanence (% of days)**				**Geology**
			spring	**monsoon**	**autumn**	**winter**	
	H6	monsoon	0	83	21	25	stream alluvium
	H7	monsoon	0	53	12	24	basin fill + terrace deposits
Ramsey	R1	always wet	100	100	100	100	limestone
	R2	always wet	77	100	100	85	limestone
	R6	monsoon to winter	0	75	90	56	basin fill + terrace deposits
Rock Spring	RS1	monsoon to winter	0	77	64	70	granite
	RS2	always wet	61	80	100	100	granite + terrace deposits
Slaughter House	SL2	always wet	51	83	100	100	conglomerate
	SL3	always wet	51	85	100	100	basin fill + terrace deposits
Split Rock	SPR1	always dry	0	1	0	0	granite
	SPR2	always dry	0	23	7	14	granite
Tinker	T1	monsoon and autumn	0	86	100	46	granite
	T2	monsoon to winter	16	88	100	100	granite
Woodcutters	W1	always dry	0	13	0	0	granite
	W2	monsoon to winter	41	91	100	100	granite
	W3	monsoon	0	85	20	21	granite + terrace deposits
	W4	monsoon	0	55	13	20	basin fill

3.4.2 Geology

While geology does not fully explain flow patterns, changes in subsurface formations translate into changes of surface flow. In Figures 3.7 and 3.9b, sensor locations and their seasonal ephemerality class are overlaid on the local geology, where each geological unit is characterized by its approximate relative permeability. Ephemerality classes do not appear to be organized along an elevation gradient or a north/south gradient. All three canyons displayed in Figures 3.7 and 3.9b start in mudstone, where permeability is low except along fractures, and the variability of flow patterns observed is high, from wet all year round (H0 and BT1) to always dry (BT2) (Figure 3.7a, 3.7c), even over short spatial scales. Limestone also displays this variability in water permeability, being highly permeable but also speckled with springs, especially along Garden Canyon (Figure 3.9b). As a result, G2 is always flowing while BT3 only flows during the monsoon (Figure 3.7a, 3.7c). Farther downstream, the impervious granite increases surface flow permanence, which is visible for sensors BT4 (wet in monsoon, autumn winter) and BT5 (winter in monsoon and winter) for Blacktail Canyon, as well as H3 and H4 (both wet all year) for Huachuca Canyon (Figure 3.9b). At the bottom of the mountain front, all three streams display a decrease in flow permanence with drier reaches. Streams reach the permeable basin fill that constitutes most of the valley and sensors are either dry all year long (BT6) or wet only during the monsoon (H7) (Figure 3.9b). They also never reach surface flow. Sensors H6, G5, G6 and G7 are on top of alluvium and only manage to reach surface water during the monsoon even if the sediment can stay wet longer (Figure 3.7).

3.4.3 Interannual variability

Stream reaches react differently to interannual variations in rainfall, with some areas showing a steady behaviour every year while others are more variable. Sensors G2, T1, and T2 recorded EC for three years (Figure 3.10) and exhibit the inter-annual variability of flow condition (from June 1, 2010 to May 31, 2013). The 2010-2011 period had a stronger monsoon and a drier winter than the following years. Rainfall total for the 2010 monsoon was 308 mm, against 236 mm for 2011 and 243 mm for 2012, while rainfall for the 2011 winter was 23 mm against 65 mm in 2012 and 75 mm in 2013 (Figure 3.10a). Sensors G2 (Figure 3.10b) and T2 (Figure 3.10d) recorded a steady pattern across all three years of record despite interannual variability in precipitation. G2 (located at the mountain top) kept flowing through the whole period (Figure 3.10b) and T2 (mountain front) maintained a “wet from monsoon to winter” pattern, only drying up during the spring dry season, although this reach slowly dried up over the 2011 winter, while it maintained surface water in 2012 and 2013 (Figure 3.10d). Sensor T1 (mountain front) displayed more variability year to year without following a specific trend. It sustained flow through the 2010 monsoon and autumn, then shifted to a flashier pattern in 2011 with cycles of drying/re-wetting, before going back to remaining wet from monsoon to winter in 2012 (Figure 3.10c).

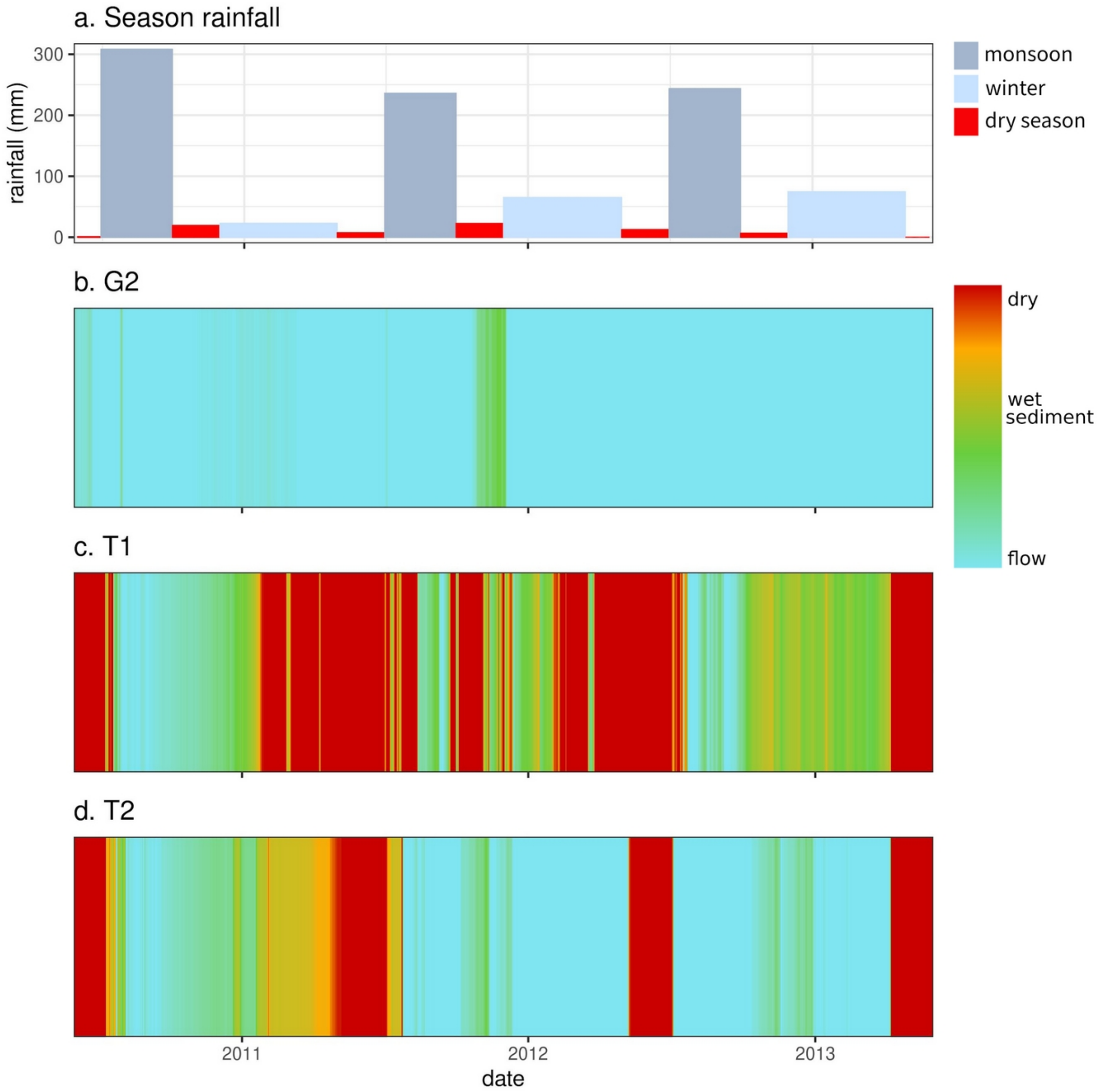

Figure 3.10: Seasonal rainfall (in mm) averaged over the Huachuca Mountains with the monsoon season shown as in dark blue, winter in light blue rectangle, spring and autumn dry seasons in red (a) and heatmaps of daily flow condition based on classified electrical conductivity measurement for G2 (b), T1 (c) and T2 (d), from June 1, 2010 to May 31, 2013.

3.5 Discussion

Non-perennial streams in drylands are important sources of moisture and hotspots of biodiversity (Bogan et al., 2015; Datry et al., 2014; Larned et al., 2010). As such, understanding the timing and distribution of flow is critical for mapping habitats and their potential climate change vulnerability (Price et al., 2021). In this chapter, I demonstrate how electrical conductivity sensor data can be used to map distribution of surface water and channel sediment moisture at high spatiotemporal resolution in small non-perennial streams. This information can then be used to classify stream reaches by seasonal patterns, a useful metric to summarize the temporal variability of flow in a way that can be compared to climate patterns and related to wildlife and vegetation dynamics. The uncertainty caused by the lack of calibration and the use of a relative conductivity prevents me from determining the exact timing and period of flow, but the use of a continuous scale and a general classification based on seasons, allows me to identify broad differences in timing of flow between sensors. In future studies, sensors should be calibrated by being put in wet and dry environments in controlled conditions to establish which values are linked to a particular moisture state before being deployed in the field (Adams et al., 2006).

3.5.1 Spatial and temporal variability of flow

Streams of the Huachuca Mountains display high variability of flow, both through time and space, with alternating wet and dry reaches. Most reaches are very sensitive to rainfall and only flow during the monsoon and/or the winter rain season, while others remain constant (always dry or always wet) no matter the precipitation input (Gallo et al., 2020a). The light winter precipitation and melting snow (low intensity and long duration) travels more slowly and has greater potential to infiltrate into the ground and feed the many springs that supply the perennial reaches (Stromberg et al., 2015), while the intense monsoon storms (high intensity, high frequency, and short duration) are more likely to initiate overland flow in the canyons (Levick et al., 2008; Stromberg et al., 2015). Non-perennial reaches can be more or less responsive to rainfall. Some reaches get wet both during monsoon and winter, responding to the smallest precipitation events, and others that need significant rain falling in a short period only flow during the monsoon.

Despite the small sample size and the limited number of parameters investigated in this study, I can still combine my findings and the literature to identify the potential controls on flow permanence in these canyons. The reaches that exhibit wet sediment or flow during the monsoon and stay wet through the dry autumn and to the winter are likely fed by local aquifers that manage to fill up during the monsoon. For example, flatter areas can allow for seepage into the local aquifer to feed the stream downstream, and faults form preferential paths for groundwater drainage to springs (Lovill et al., 2018; Martin et al., 2021). Areas sheltered by vegetation or the surrounding topography might also stay wet longer, as evaporation is reduced. As for the reaches that remain dry, I noticed that they were either on top of permeable sand and gravel layers or colluvium. A wider and denser sensor network would allow to build a more robust picture of small temporal and spatial scales changes in water distribution across the mountain range.

3.5.2 Geology and additional controls

Streams in the Huachuca Mountains, as is true in other ephemeral streams of the Southwest USA, show abrupt longitudinal changes in flow permanence influenced by geomorphological processes and discontinuities (Goodrich et al., 2018; Larned et al., 2011; Lovill et al., 2018). An example of how water moves downstream along Huachuca Canyon is shown in Figure 3.11. Geology can alter surface hydrology through permeability of underlying formation, spring location, perched aquifers, faults, fractures or sediment deposits (Levick et al., 2008). The headwaters of the streams studied here are mainly located on top of mudstone and sandstone, before meeting limestone. Flow permanence on top of these layers is variable, going from reaches that are always dry to always flowing. This behaviour could be explained by the fact that the mudstone units of the Huachuca Mountains are impervious layers but intersected by small fractures that collect water, which is then released to springs and streams, while the very high permeability of limestone, due to a high density of fractures and solution channels, is interrupted by impervious siltstone beds (Brown et al., 1966). This upper area of the mountains is also dissected by faults that form preferential flow paths for water. The diversity of structures, each with their own permeability to water, in part, leads to the diversity of flow permanence patterns we see along the canyons. The lower half of the mountain range is underlain by quartzite and granite and it is on top of these impermeable bedrock units that I observed an increase in flow permanence in these canyons and where most of the perennial flow occurs. Down in the San Pedro River basin, water travels over the low permeability conglomerate (Brown et al., 1966) before reaching the sand and gravel of the sedimentary basin fill that form a highly permeable fan around the Huachuca Mountains. This is the area with some of the driest reaches in this study. Rainfall distribution is also highly dependent on elevation, with higher areas receiving more rainfall and lowland stream reaches receiving lower precipitation. Monsoon storms can also cover small extents and might cover only one watershed, bringing water to one canyon while its neighbours remain dry.

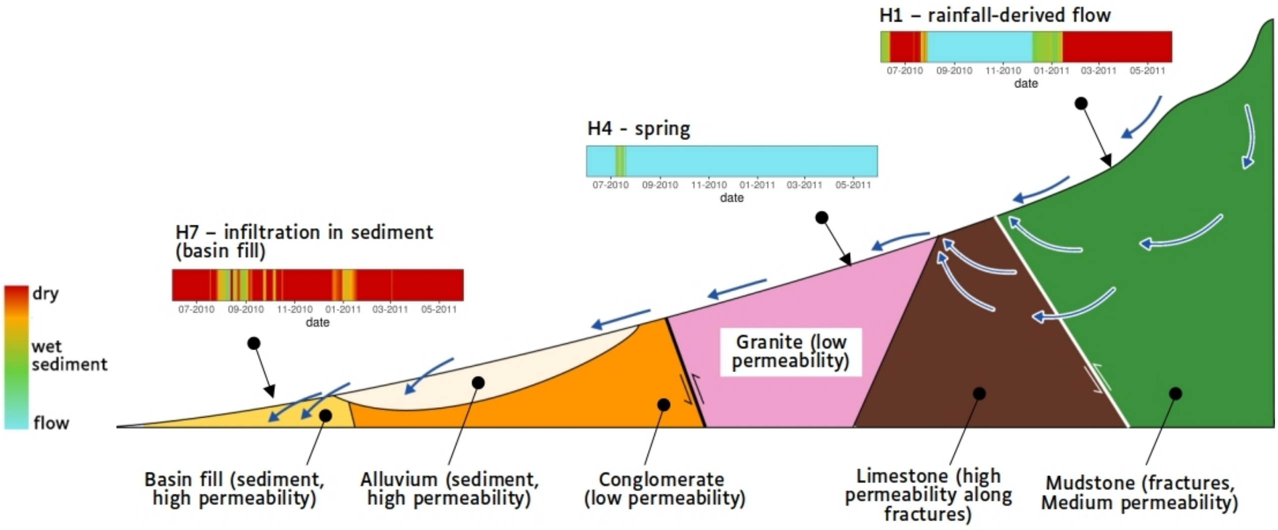

Figure 3.11: Conceptual model of flow distribution along Huachuca Canyon and water travel downstream (overground and underground), with the location and daily records of three conductivity sensors from the 1st of June 2010 to the 31st of May 2011 (from upstream to downstream: H1, H4, H7). Water (blue arrows) seeps in fractured mudstone and limestone before reaching the surface when encountering impervious units (granite) and faults. At the bottom of the mountain front, water travels down in permeable sediment layers to reach the regional water table.

Local channel conditions and human activity can override expected geologic response at local scale. Channel geometry and stream channel density, itself dependent on grain size and sediment composition, are important reach-scale controls on flow permanence and streambed sediment moisture (Gallo et al., 2020a; Larned et al., 2010; Pate et al., 2020; Whiting & Godsey, 2016). Some sensors, such as BT2, were dry no matter the underlying geology; a result likely due to a thick and very permeable sediment layer in the streambed. There are also anthropogenic controls on flow permanence in the Huachuca Mountains. The streams of Garden and Huachuca Canyons have historically been used as a water source for the U.S. Army Installation Fort Huachuca. Spring boxes and pipes are still redirecting water down to the fort (Brown et al., 1966). Some downstream reaches are in urban areas, which can also affect flow regimes. Artificial impervious surfaces prevent rainwater from seeping through the sediment and redirect it instead to the non-perennial washes, which leads to flow being present more often and for longer periods (Gungle, 2006). Additional metrics, such as groundwater and streamflow depth, information on flowing water versus pools or rainfall and temperature measured by local gauges, could provide a better understanding of the water’s travel along the stream and below it, local geology permeability, and reactivity to climate variables.

3.5.3 Interannual variability

Response to inter-annual variations in rainfall was also variable, with some stream reaches demonstrating consistent flow patterns every year while others fluctuating more. Reaches likely fed by springs, such as G2, show little variation and remained flowing through the dry seasons and weak winter rains. Sensor T2 also recorded a regular pattern, only fully drying up during the spring dry season, but the weak winter rain of 2011 led to a progressive dry up while stronger precipitation in winter for 2012 and 2013 seems to have managed to keep that reach flowing until the spring dry season. The flow pattern for T1 is less regular but a weak winter rain season led to an early dry-up in 2011 and weaker monsoons in 2011 and 2012 might be the cause of the shorter period of surface flow. Due to the uncertainty in the link between conductivity value and conductivity, precise timing of shifts between dry and wet sediment, or wet sediment and flow is imprecise and might explain some of the interannual variability.

3.5.4 Implications for conservation

Conductivity analysis demonstrated in this chapter could be an important tool for mapping potential habitats for species of conservation interest. The key elements that make this work useful are its high temporal resolution (daily data in remote areas with complex topography, which makes fieldwork time-consuming), high spatial resolution (I am able to measure flow state at a precise location), and the fact that the high number of sensors spread across and along streams provide a landscape-scale overview of temporal and spatial distribution of moisture and flow across a whole mountain front. The dataset provides information on sediment moisture and surface flow, but also on which state is reached when, and how often. This record of spatiotemporal distribution of flow and soil moisture supports efforts to pinpoint reaches of perennial flow (such as H4 or G2) in an otherwise dry region. Reaches that manage to remain wet during the dry spring (all sensors in the "always wet" class) can play a critical role as moist and cool refuges. Once flow resumes in the drier reaches and the stream network connects, animals that had found shelter in the perennial reaches can re-colonize the whole network (Bunn et al., 2006; Larned et al., 2010). They can also be favourable habitats for species such as the Huachuca water umbel (*Lilaeopsis schaffneriana ssp. Recurva*), an herbaceous, semi-aquatic perennial plant, which needs a permanently wet environment (Bagne & Finch, 2013). For non-perennial reaches, I am able to compare the limited periods of sediment moisture or surface water to the phenology of species of interest. The Chiricahua Leopard Frog (*Lithobates chiricahuensis*) can be found in temporary streams that dry periodically to discourage non-native predators and competitors while still staying moist enough for the frogs and with surface water for breeding, the breeding period depending on elevation (Bagne & Finch, 2013). A host of other amphibian species in the region are dependent on the patchwork of water availability (Mims et al., 2015), as are aquatic invertebrates (Phillipsen et al., 2015). Being able to map the distribution of flow across the landscape could also be used to highlight potential wet corridors for allowing species dependent on sediment moisture and surface water to travel between favourable habitats and breeding locations. Data on flow permanence can be paired with wildlife and vegetation surveys (through camera traps, bioacoustics, or remote sensing for example) to study flow permanence as a parameter in habitat mapping and species distribution. This method can be upscaled across a wider region, keeping in mind that the original setup and the regular inspections necessary to collect data and control the sensors' condition are time consuming. These sensors also don't provide information on flow intensity or water depth. Lastly, sensors' location has to be decided depending on the parameters studied. For example, a sensor buried in the streambed will only start recording once water reaches the sensor's depth. For wider streams, flow can be detected on very high spatial and temporal resolution imagery from satellites, drones or aircrafts (Wang & Vivoni, 2022).

Riparian and semi-aquatic species in the study region are considered highly vulnerable to climate change (Bagne & Finch, 2013), so recording the flow condition in streams for several years could be a useful tool for detecting areas that are particularly sensitive to variations in rainfall and/or moisture. A shift in rainfall distribution, timing and intensity could, for example, change the distribution of seasonal water patterns across the landscape and create ecological shifts for communities depending on specific flow regimes (Bogan et al., 2015; Jaeger et al., 2014; Stromberg, 2013). Depending on the controls governing the presence of water in a reach, areas might be more or less sensitive to climate change. In reaches that are more sensitive to precipitation, a dry winter might lead to an earlier drying of a stream that usually flows until spring. A non-perennial stream that only responds to monsoon rains (wet during monsoon class, sensors H6 and G6 for example) might be very responsive to a stronger or weaker monsoon, while reaches sustained by groundwater inputs (always wet class, sensor BT1 or G2 for example) might be better buffered and could remain wet, affirming their critical status as refuge for drought-sensitive species (Gallo et al., 2020a; Stromberg et al., 2015). Springs are the areas most likely to provide a steady water source to the surface, but they are reliant on sufficient water inputs in upstream locations that replenish the local aquifers. Thus, severe changes in rainfall regimes could also lead to shifts even in the wettest perennial reaches (Van Loon, 2015). The vegetation and wildlife in these always wet reaches might also be more severely impacted, as they are adapted to a perennial water source, and a temporary dry-up could lead to changes in riparian forest extent and shift in species. Increases in dryness could also lead to a loss of connectivity, with flowing reaches becoming less frequent and more isolated (Jaeger & Olden, 2012; Seager et al., 2013).

3.6 Conclusions

I documented the spatiotemporal variations in flow permanence and channel substrate moisture in the temporary streams of the Huachuca Mountains, southeastern Arizona, USA. I distinguished between reaches highly responsive to local climate and those with more stable flow patterns. Although climate is the first control on water distribution at the regional scale, I revealed that underlying geology, as well as other localized factors such as stream bed composition and landscape topology, affect flow permanence locally. This work shows how the high spatial and temporal resolution provided by electrical conductivity sensors can be used to build a local, reach-scale understanding of surface flow permanence and distribution by using a seasonal classification of flow patterns, and how the resulting. The resulting local, reach-scale understanding of surface water distribution can then provide critical information on potential habitat for riparian species and these habitats' sensitivity to climate change.

Chapter 4
Vegetation distribution along montane non-perennial streams and the impact of drought on vegetation dynamics.

4.1 Abstract

Understanding vegetation spatial distribution and temporal dynamics across complex topography where water availability is widely variable is important, as it can be altered by climate change through its interaction with the water cycle. In drylands, riparian forests and isolated mountain ranges are of high interest for conservation due to high biodiversity and their role as a cool and moist refuge for many species. But forests along channels are sparse, isolated, and sensitive to changes in temperature and precipitation, which can affect their access to water. In this chapter, my goal was to map vegetation distribution along the intermittent streams of the Huachuca Mountains, in southeast Arizona. I explored the influence of elevation and flow permanence on vegetation distribution, and influence of rainfall temporal distribution on vegetation greenness and phenology, through drought and recovery. I used Normalized Difference Vegetation Index (NDVI) time series derived from Sentinel-2 images for 2018-2021 to map vegetation communities and establish timing of the growing season. I found that vegetation communities (grassland, deciduous trees, sparse evergreen mixed with grassland, and dense evergreen trees) are structured along the elevation gradient, but temporal distribution of flow does not seem to play an important role on vegetation distribution along

the stream. Phenology and peak vegetation greenness was impacted by rainfall distribution for all communities, with deciduous trees and grassland being more sensitive, while dense evergreen trees showed smaller variations. Through the use of straightforward and adaptable tools and methods, I mapped vegetation distribution, highlighted key environmental controls and gained insight into the impact of drought on vegetation greenness and timing of the growing season. This work's aim is to test monitoring tools' ability to provide information on a habitat's distribution and dynamics so natural resources managers can keep track of key habitat evolution with climate change and adapt management practices.

4.2 Introduction

As climate changes, an increase in the risk of severe drought could threaten riparian vegetation in drylands. Vegetation is a key element in several aspects of conservation and natural resources management, notably as a structural element of habitat for threatened and endangered (T&E) species (Odom & Ford, 2020; Powell & Steidl, 2015), and managers need to be able to better understand and monitor vegetation extent and health in relation to environmental controls and drought as climate continues to shift. While Chapter 2 was centered on long-term changes in vegetation greenness along the San Pedro River (30 years time series with only NDVI values right before and right after the monsoon), this chapter looks at riparian vegetation along mountain canyons and variations in short term phenology (4 years time series with daily NDVI values), supported by the findings of Chapter 3. Together, they provide a comprehensive look at the effects of drought on riparian vegetation in different settings (valley river against mountain canyon) and at different time scales. In this chapter, my goal is to map the main vegetation communities over the Huachuca Mountains and the surrounding valley. I focus on vegetation along the Huachuca Mountains canyons to evaluate if elevation is the only main controls on vegetation distribution, or if flow permanence can also play a role on vegetation distribution along canyons. I also assess the consequences of a short-term drought on vegetation greenness and on the timing of key life events, such as leaf-on and senescence for the main vegetation communities of the study area. The aim of this chapter is to better understand the environmental controls behind vegetation distribution along montane streams, how shifts in rainfall distribution might impact phenology, and test monitoring methods that could be useful for natural resources management.

While the effects of climate change are variable across the globe, an increase in warm temperature extremes and in intensity and frequency of heavy precipitation events is expected in a number of regions (Intergovernmental Panel on Climate Change (IPCC), 2022). By the end of the century, climate change and its consequences may be the dominant direct driver of biodiversity loss globally (Millennium Ecosystem Assessment, 2005). Shifts in the distribution, geographic ranges, density and seasonal activities have been observed for many species, leading to changes in ecosystem structure and dynamics (IPCC, 2022; Walther et al., 2002). Species with limited ranges and low mobility (such as trees on mountain tops) are at risk of dying-off due to drought and being replaced by species more adapted to the new climate conditions (Breshears et al., 2005; Lenoir et al., 2008; Munson et al., 2011).

Climate change creates new natural resources management challenges and priorities. Ecosystem conservation is often focused on maintaining ecosystems or restoring ecosystems to some baseline condition. These goals become challenging when the baseline condition shift due to climate change, leaving some ecosystems no longer viable in future climates (Seavy et al., 2009). The protection of climate refuges, such as the cool and moist streamside vegetation in drylands, and their restoration can help mitigate species loss but there is also a need to accept and adapt management practices to cope with unavoidable changes (Seavy et al., 2009; Stein et al., 2019). Another difficulty is the uncertainty of change, with specific impacts of climate change depending on regions, and the degree of change is uncertain (Odom & Ford, 2020).

The United States Department of Defense (DoD) is one of the main land managers in the USA (Ripley et al., 2021). Military installations cover high quality natural habitats and host many protected species (NatureServe, 2004; Ripley et al., 2021). The management of these spaces is important for conservation, long-term sustainability of natural resources, natural hazards risk management and for supporting the military mission (Garfin et al., 2021; Stein et al., 2019). As with other federal land managers, natural resources managers of military lands are also concerned by statutes for protected species, water quality, or wetlands through several federal regulations (Chapter 1) (Ripley et al., 2021; Stein et al., 2019). Climate change adaptation and resilience has been an important point of interest, with several climate adaptation plans and guides published to assist natural resources managers (Odom & Ford, 2020; Stein et al., 2019) with drought, desertification and wildfires being highlighted as some of the main climate risks for installations (Odom & Ford, 2020; Stein et al., 2019)

Vegetation is a key element for conservation and natural resources management, as a structural element for habitat and as a food source for wildlife (Odom & Ford, 2020; Powell & Steidl, 2015). In drylands, plant productivity is limited by water availability (Chapter 1) (Millennium Ecosystem Assessment, 2005; Mirzabaev et al., 2022). Water is stored in groundwater and streams that often dry at least once per year (Messager et al., 2021b) and these streams and their riparian vegetation often are the main source of moisture in otherwise dry landscapes, making them important hotspots of biodiversity (Bogan et al., 2015; Datry et al., 2014; Larned et al., 2010), including protected species. Since access to water is one of the key factor that controls habitat and species distribution, riparian habitats are vulnerable to changes in water availability (e.g. due to climate change) (Tietjen et al., 2009).They are also highly vulnerable as they cover a limited area, are isolated and cannot expand their range as the surrounding landscape is too dry for them ((Bertrand et al., 2011; Loarie et al., 2009b; Malagnoux et al., 2007; Reidmiller et al., 2018)). Vegetation can respond to climate change in several ways: shift in range and distribution of species (notably with distribution ranges shifting north or to higher elevation), change in structure of ecosystems (loss of trees, transition from grassland to shrubs or non-native species invasion) and fluctuations in phenology (delays or advance in key life cycle events) (C. D. Allen et al., 2010; Cleland et al., 2007; IPCC, 2014; Walther et al., 2002).

Phenology is the study of periodic events in wildlife and flora life cycles of animals or plants, as influenced by environmental conditions (Cleland et al., 2007). It can be used to study the influence of external factors such as rainfall on duration of the growing season, or offsets in season start and end (Broich et al., 2014; Vitasse et al., 2009; Weiss et al., 2004). Phenology, density and species shifts can be studied with remote sensing, which provides global, high resolution images. These images can be used to generate vegetation indices such as the Normalized Difference Vegetation Index (NDVI), which measures vegetation greenness and is linked to leaf density (Chapter 1) (Kerr & Ostrovsky, 2003; Lawley et al., 2016). NDVI can be used to look at trends of vegetation greenness to measure how vegetation health is changing (Pettorelli et al., 2005; J. Yang et al., 2012), or seasonal variations of density to measure timing of phenological events such as green-up and leaf loss. Changes in vegetation greenness can also help to identify shifts in species (from dense green trees to sparse grassland for example, (Coppin et al., 2004) or annual time series to compare phenology (between deciduous and evergreen trees for example, (Weiss et al., 2004).

The study site covers the Huachuca Mountains and the valley (Chapter 1). The mountain range is a hotspot of biodiversity due to a high topographic diversity (Devender et al., 2013; Poulos & Camp, 2010) and much like dryland riparian forests, is an isolated colder and wetter refuge for wildlife, vulnerable to climate change due to the inability to migrate and follow changes in temperatures and rainfall (Bogan et al., 2015; Monroy-Gamboa et al., 2021). Fort Huachuca is one of the main managers of this small mountain range, which is home to many endangered species, making habitat and species vulnerability to climate change critical (Bagne & Finch, 2013).

The main manifestation of climate change in this area of the Southwest USA is warmer and drier conditions, as well as more variable and intense precipitation (Chapter 1) (Garfin et al., 2021). Several consequences have already been observed, including shifts in vegetation distribution (Barton & Poulos, 2018; Breshears et al., 2005) and loss of hydrologic connectivity in ephemeral streams, with consequences for riparian vegetation and all species whose life cycle is timed to predictable flow patterns (Jaeger et al., 2014). Increased water stress is inducing community and range shifts (Walther et al., 2002). In the mountain ranges, the vegetation of the semi-arid valley is encroaching on more temperate species and climbing higher along slopes, with cooler canyons acting as a last refuge (Kelly & Goulden, 2008; Monroy-Gamboa et al., 2021), and habitat loss leading to increased vulnerability for wildlife (Coe et al., 2012). Even in communities that manage to maintain themselves, shifts in phenology can translate to an increased vulnerability and offsets between vegetation and wildlife life cycles (Walther et al., 2002). Many species dependent on riparian habitats are considered as vulnerable due to habitat loss, added to low mobility for amphibians or event timing (offset between vegetation and wildlife phenology) for birds (Coe et al., 2012).

Climate is currently undergoing changes of unprecedented magnitude, and we are only starting to see the consequences on vegetation and wildlife. Understanding how environmental parameters, such as climate or topography, control vegetation, and how vegetation reacts to changes in distribution or intensity of climate, is critical as it can help establish habitat sensitivity to climate change (how they might change and how fast). To be able to carry out their mission, natural resource managers need to be able to monitor current changes and plan for the future. They need to be proactive and plan for expected shifts in habitat quality and extent, especially for endangered species vulnerable due to their low numbers, and especially for habitats that are sparse, isolated and sensitive to shifts in water availability and temperatures, such as riparian vegetation in drylands.

This chapter is a case study of riparian vegetation across the Huachuca Mountains and the San Pedro valley, Arizona, with the goal to understand the environmental controls behind vegetation distribution, how shifts in rainfall might impact phenology, and to test monitoring methods that could be useful for natural resources management. I compared riparian vegetation greenness and distribution along two parameters affected by climate change: elevation (as representation of rainfall and temperatures, with rainfall increasing and temperature decreasing with elevation) and flow permanence in ephemeral streams (number of days with flow or soil moisture in a year). I used remote sensing to map the main vegetation communities (unique species assemblage) representing different habitats and investigated if and how the key environmental parameters that are elevation and flow permanence control vegetation distribution along the canyons. NDVI time series were also reconstructed to establish key phenological events and vegetation health, which are then compared to seasonal rainfall distribution.

My goal is to better understand how the various vegetation communities of the Huachuca Mountains are distributed and their sensitivity to drought, while using tools and methods that are efficient while staying relatively straightforward. These tools used to map and track changes in vegetation distribution can be adapted to other locations and specific management and conservation questions. The methods are applied here to riparian vegetation in drylands, but any habitat of interest used by endangered species that can be mapped with satellite imagery could be studied in relation to climate change to better predict and plan for potential shifts.

4.3 Study site and methods

Vegetation communities were mapped using a supervised classification and monthly Sentinel-2 images. This classification map was then used to compare vegetation distribution along non-perennial streams to elevation and seasonal distribution of flow to better understand how these environmental factors might control vegetation organization in the landscape. We then measured highest (peak) and lowest (trough) vegetation greenness through NDVI, as well as growing period length and timing, to compare vegetation dynamics to rainfall distribution, highlighting the difference between years from 2018 to 2021.

4.3.1 Study site

The study site for this chapter covers the Huachuca Mountains, the San Pedro River and the valley in-between (Chapter 1) with a focus on the vegetation along the canyons studied in Chapter 3. Miller peak is the highest summit of the Huachuca Mountains and stands at 2884 m.a.s.l, while the San Pedro River is at 1250 m.a.s.l. Dominant tree species in the narrow riparian corridor along the canyons also follow an elevation gradient, from mesquite (*Prosopis velutina*) in the lowlands, to interior riparian trees (proportions of dominant trees vary but include Arizona sycamore (*Platanus wrightii*), Arizona white oak (*Quercus arizonica*) and silverleaf oak (*Quercus hypoleucoides*), alligator juniper (*Juniperus deppeana*) and bigtooth maple (*Acer grandidentatum*)) and Madrean montane riparian vegetation, represented by a mix of deciduous and evergreen trees such as Douglas-fir (*Pseudotsuga menziesii*), ponderosa pine (*Pinus ponderosa*) or Apache pine (*Pinus engelmannii*) mixed with the interior riparian trees (Figure 4.1) (Brady & Bonham, 1976; H. G. Shaw, 1999).

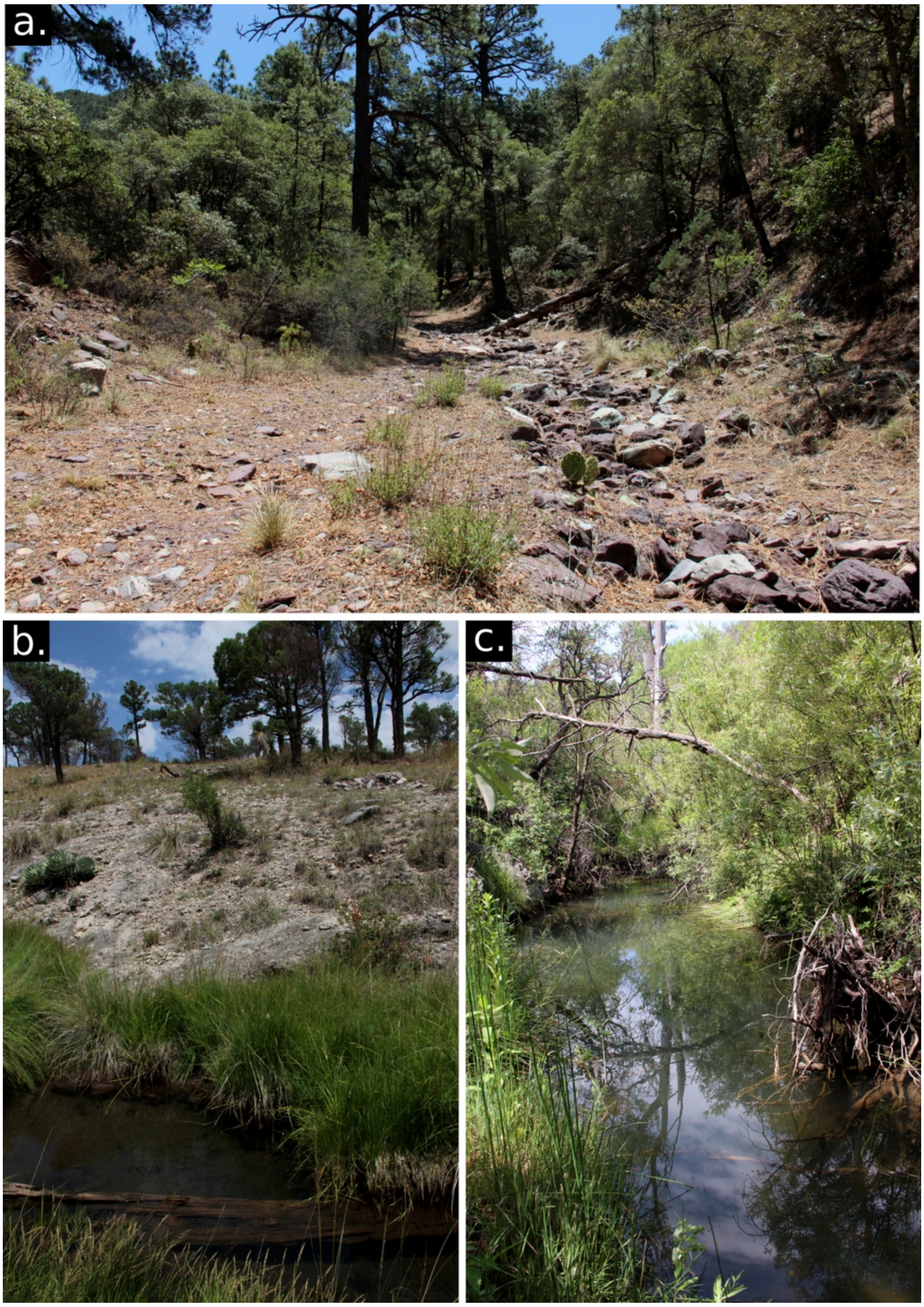

Figure 4.1: Examples of a dry reach in a dense evergreen trees area (Huachuca Canyon, a) and two perennial reaches (Garden Canyon) with riparian vegetation (b, c). Pictures taken in June 2019.

As seen in Chapter 3, canyon streams are non-perennial, they dry out for at least part of the year. They are characterized by a high spatiotemporal variability, with alternating wet and dry reaches. Out of the 37 sensors spread across 10 streams, 5 remained dry, 13 recorded either flow or a wet sediment for the whole year, and the rest (19 sensors) were in reaches that reacted to rainfall and got wet during the monsoon, and for the most reactive of them, during both monsoon and winter rain season (Table 4.1).

Table 4.1: For each conductivity sensor, seasonal flow permanence, distance along channel (distance from most upstream sensor), elevation and streambed morphology.

stream	**sensor**	**flow permanence**	**distance along channel (m)**	**elevation (m.a.s.l)**	**morphology**
Brown	B1	always wet	0	1662	step-pool
	B2	monsoon to winter	2034	1555	shallow step-pool
	B3	monsoon to winter	2262	1478	sand bed wash
BlackTail	BT1	always wet	0	2228	shallow pool-riffle
	BT2	always dry	840	2074	colluvium/ cascade
	BT3	monsoon	2324	1709	step-pool
	BT4	monsoon to winter	1604	1604	plane bed
	BT5	monsoon and winter	2530	1511	plane bed
	BT6	always dry	2476	1445	sand bed wash

stream	**sensor**	**flow permanence**	**distance along channel (m)**	**elevation (m.a.s.l)**	**morphology**
Garden	G1	monsoon to winter	0	1893	shallow step-pool
	G2	always wet	755	1883	pool-riffle
	G4	always wet	1210	1646	step-pool
	G5	monsoon to winter	3600	1546	pool-riffle
	G6	monsoon	2549	1484	pool-riffle
	G7	always wet	2104	1444	sand bed wash
Huachuca	H0	always wet	399	1891	shallow step-pool
	H1	monsoon and autumn	0	1910	cascade
	H2	monsoon	1197	1807	step-pool
	H3	always wet	1707	1723	step-pool/ travertine
	H4	always wet	2019	1605	step-pool/ travertine
	H6	monsoon	2318	1487	pool-riffle
	H7	monsoon	3797	1406	plane bed
Ramsey	R1	always wet	0	2204	step-pool/ cascade
	R2	always wet	1209	2030	step-pool
	R6	monsoon to winter	4304	1441	sand bed wash

stream	**sensor**	**flow permanence**	**distance along channel (m)**	**elevation (m.a.s.l)**	**morphology**
Rock Spring	RS1	monsoon to winter	0	1566	cascade
	RS2	always wet	1518	1486	pool-riffle
Slaughter House	SL2	always wet	2235	1454	plane bed
	SL3	always wet	2992	1402	sand bed wash
Split Rock	SPR1	always dry	0	1770	cascade
	SPR2	always dry	2154	1611	pool-riffle
Tinker	T1	monsoon and autumn	0	1637	step-pool
	T2	monsoon to winter	1846	1524	shallow step-pool
Woodcutters	W1	always dry	0	1887	colluvium/ cascade
	W2	monsoon to winter	2222	1624	pool-riffle
	W3	monsoon	2336	1520	pool-riffle
	W4	monsoon	1881	1470	sand bed wash

The study area is under the influence of the North American Monsoon (Chapter 1). This study focused on the period from 2018 to 2021, with each year split into four seasons: dry spring (May-June), the summer monsoon (July to September), dry autumn (October-November) and wet winter (December to April) (Figure 4.2a, b, c, data is daily rainfall and temperature from the PRISM Climate Group, https://www.prism.oregonstate.edu/). The years 2018 to 2021 were chosen as this is the period with Sentinel-2 imagery (used to map and monitor vegetation in this chapter) with enough data available over our study site to reconstruct time series of vegetation greenness. The summer monsoon and winter are the two main rain seasons, bringing water to vegetation through soil moisture, groundwater recharge and snow. Vegetation in the area uses soil moisture, groundwater where accessible (along streams) or a mix of both (Sabathier et al., 2021; Snyder & Williams, 2000; Stromberg & Tellman, 2012, p. 42). According to the U.S. drought monitor (https://droughtmonitor.unl.edu/DmData/TimeSeries.aspx), 2018 was mostly considered as in moderate or severe drought, while 2019 and the first half of 2020 were generally considered free of drought. The second half of 2020 saw a steady increase in drought severity until 100% of the county was classified as in exceptional drought from April to July 2021, before the strong 2021 monsoon (Figure 4.2d).

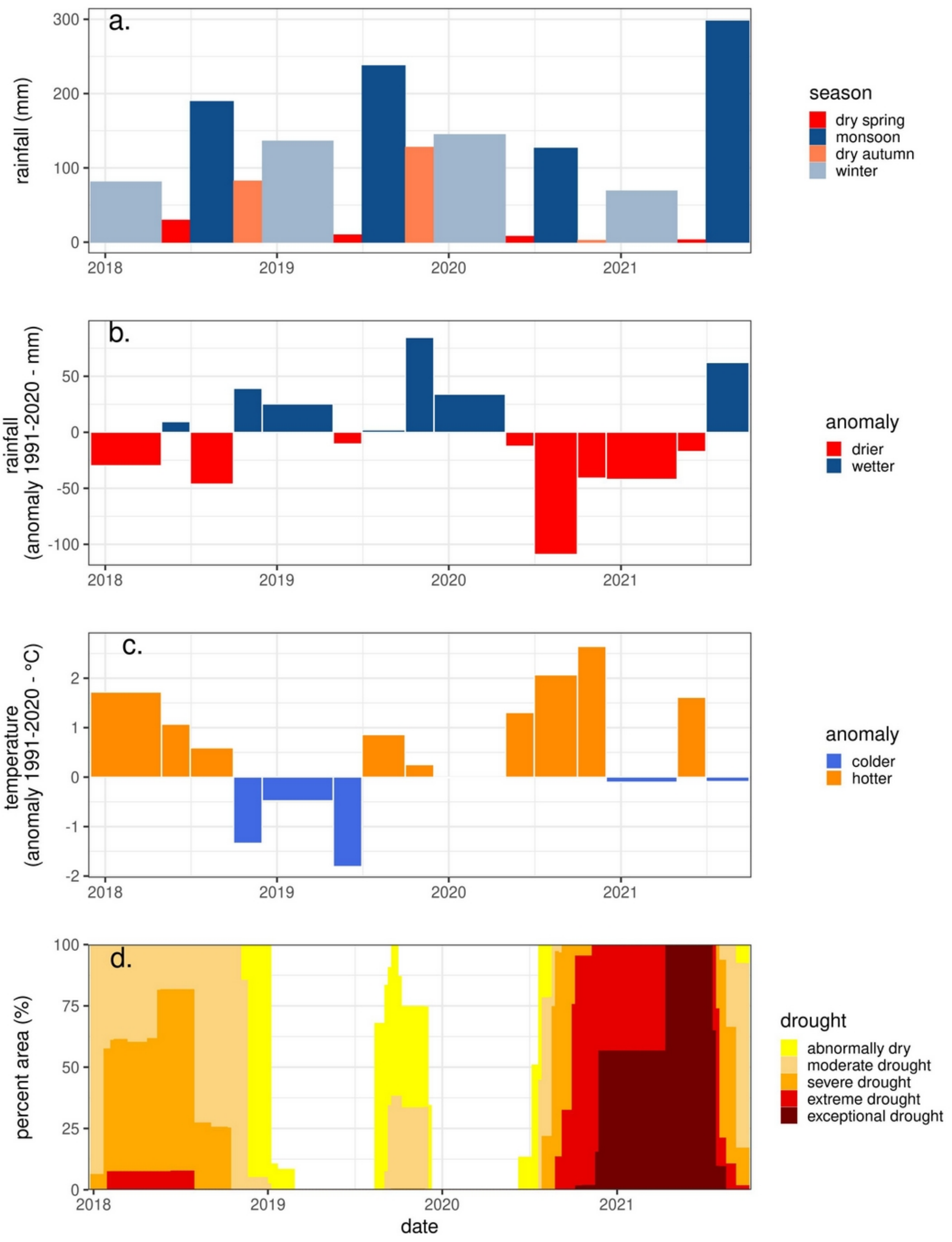

Figure 4.2: Seasonal total rainfall (mm) over the study site for the years 2018 to 2021, with each year split into 4 seasons: dry spring (May-June), monsoon (July to September), dry autumn (October-November) and winter (December to April) (a), seasonal rainfall anomaly (against seasonal 1991-2020 mean) (b), seasonal temperature anomaly (against seasonal 1991-2020 mean) (c) and percent area in drought for Cochise County, where the study site is located, weekly data downloaded from the U.S. Drought Monitor (d).

4.3.2 Method

vegetation greenness

We use NDVI (Rouse et al., 1973) as an indicator of leaf density in vegetation (Bannari et al., 1995; Kerr & Ostrovsky, 2003; Lawley et al., 2016). It can be used to map vegetation type (Senay & Elliott, 2002; Van Wagtendonk & Root, 2003), phenology (Weiss et al., 2004; Zhang et al., 2012) and vegetation dynamics (Pennington & Collins, 2007; Roder et al., 2008; Smith et al., 2019). We estimated NDVI from the red and infrared bands from Sentinel-2 ground reflectance.

$$NDVI = \frac{NIR - Red}{NIR + Red}$$

with NIR: near infrared

Sentinel-2 was chosen to map land cover over the area and build time series of vegetation greenness along the Huachuca Mountains canyons (Figure 4.3). Its high spatial resolution (10 m) is essential to distinguish potential riparian vegetation along the narrow canyons. With a revisit time of five days, the high temporal resolution was used to gather the most complete time series possible to capture key stages of the annual vegetation growth cycle and to reconstruct phenology patterns. This temporal resolution of 5 days was only achieved once Sentinel-2B joined Sentinel-2A in orbit and started sending images in July 2017. Due to the significantly lower number of images available before July 2017 (between 11 and 55 images available a year before 2018 against 75 images/year from 2018 onward), the time series used in this chapter starts in 2018. Sentinel-2 images were collected for the years 2018 to 2021 and manually checked for clouds over the study site. 135 clear images were kept (29 in 2018, 30 in 2019, 39 in 2020, 37 in 2021). Top-of-atmosphere images were downloaded from the Onda repository (https://www.onda-dias.eu/cms/data/catalogue/sentinel-2/). Atmospheric corrections were applied and the NDVI was calculated for each image using SNAP, the toolbox provided by the European Space Agency (https://step.esa.int/main/toolboxes/snap/).

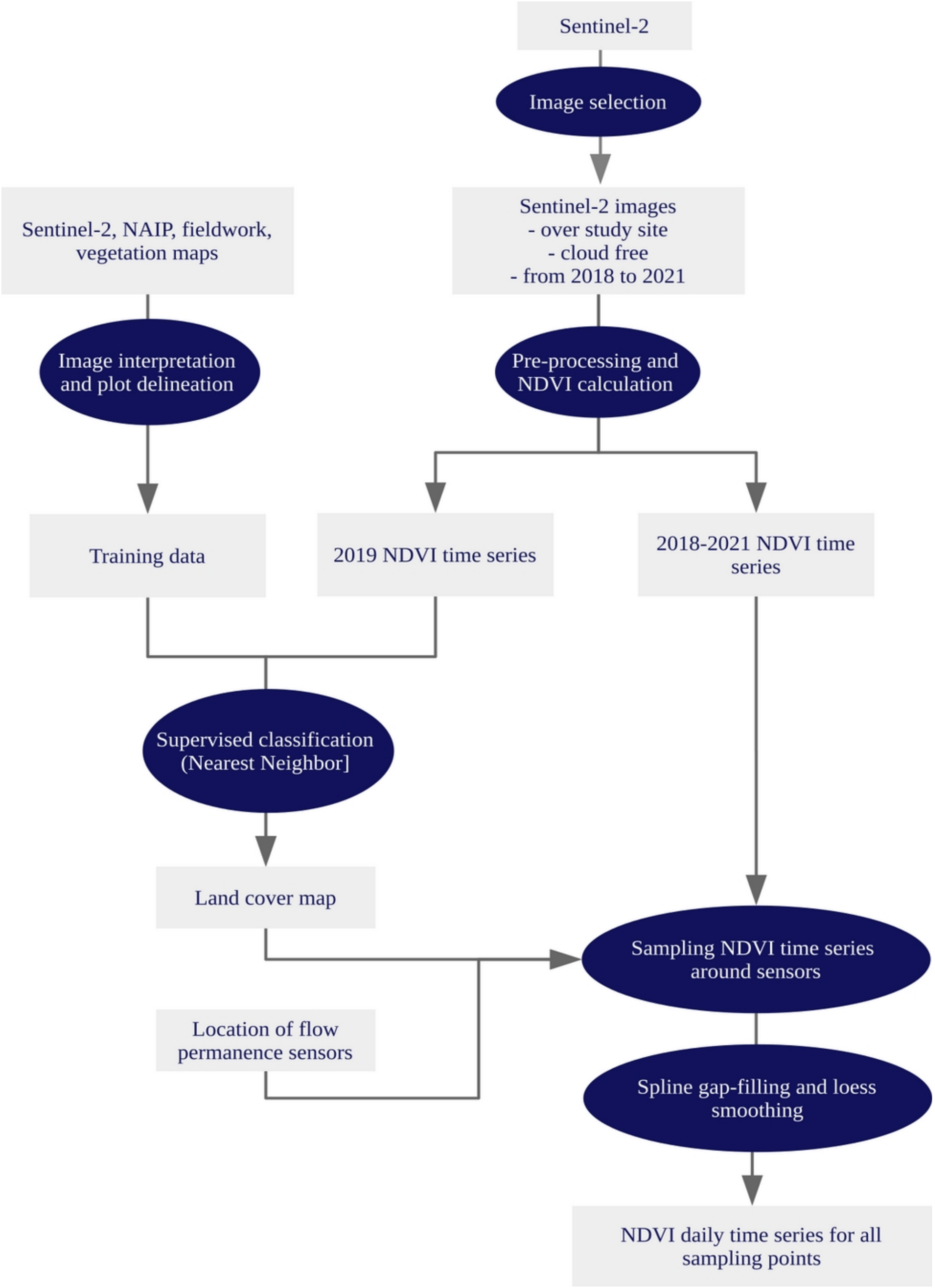

Figure 4.3: Steps of the satellite imagery analysis and sampling (datasets in grey rectangles and analysis in blue ovals). NDVI time series are derived from cloudless Sentinel-2 images. Paired with training data, the 2019 NDVI dataset is used to run a land cover classification over the study site. The result of this classification is paired to the whole 2018-2021 NDVI dataset for a subset of points around flow permanence sensors to build daily NDVI time series for each land cover type.

NDVI Sampling

To study the detailed distribution of vegetation and change through time, I looked at NDVI values for all the Sentinel-2 pixels along 10 streams scattered across the Huachuca Mountains. These streams are the same as those investigated in Chapter 2. A theoretical floodplain was defined using a HAND (Height Above Nearest Drainage) model (Nobre et al., 2011), with the area under 10 m above the stream considered as the floodplain and all pixels in this floodplain were selected (Figure 4.4). This value of 10 m was chosen as it was the best fit to cover riparian vegetation visible in satellite and aerial imagery (Levick et al., 2015). A subset of points in this theoretical floodplain around the flow permanence sensors used in Chapter 2 was defined for more detailed study, going 100 m upstream and 100 m downstream (Figure 4.4, 4.5b). This length was selected to have enough points for the analysis without straying too far away from the sensor (as flow is highly variable along the stream) and without the risk of sampling around two sensors overlapping. The narrowing of the floodplain at higher elevations leads to a decrease in sampling points around the flow sensors. Sampling points over man-made infrastructure (roads and trails) were manually removed. Sampling for Figures 4.8 and 4.9 include all points in the canyons’ floodplain, while all other figures (except maps) only include points around the flow permanence sensors (Figure 4.5b). All vegetation communities along the canyons are sampled.

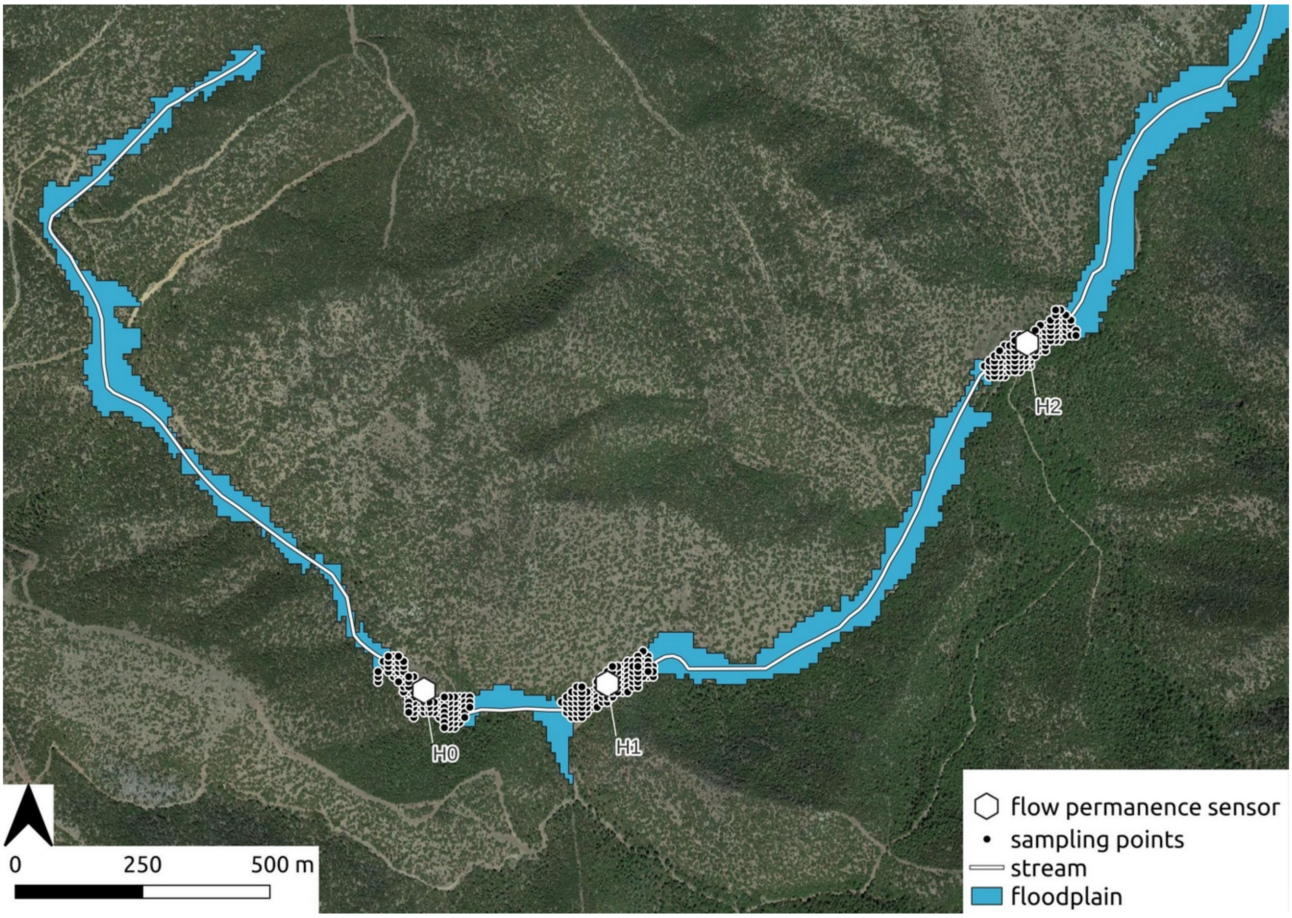

Figure 4.4: Close-up of Huachuca Canyon showing the theoretical floodplain in blue, the location of the flow permanence sensors, and the sampling points around each sensor.

Comparing NDVI to elevation and flow permanence

The first section of the results compares vegetation greenness and distribution to elevation and flow permanence. Vegetation greenness is measured using NDVI median for all years of record (2018 to 2021) over the growing season (March to August) for each sampling point. Elevation data comes from the Shuttle Radar Topography Mission data provided by the National Aeronautics and Space Administration (NASA). Flow permanence is measured in number of dry days (days with no surface water) in a year (June 1, 2010 to May 31, 2011) and comes from the flow permanence sensors presented in Chapter 2. Flow permanence was measured in 2010-2011, but flow patterns in these canyons are unlikely to have gone through big changes, as the management of flow does not seem to have changed (ENRD Fort Huachuca, 2021). Nine canyons are on Fort Huachuca, and the last one (Ramsey Canyon) is part of the Ramsey Canyon Preserve. Results are presented as violin plots for NDVI values, and bar plots for land cover distribution, with median and the interquartile range (IQR) used as measures of central tendency and dispersion of NDVI values. Medians were compared using a Wilcoxon rank sum test and a two-sample Kolmogorov-Smirnov Test was run to compare distributions.

Land cover classification

Sentinel-2 NDVI images from 2019 were used for vegetation mapping across Fort Huachuca. 2019 was selected since it is the only year available from Sentinel-2 that is not considered a drought year in Cochise County, with no area considered under extreme or exceptional drought (Figure 4.2c). For automatic mapping of vegetation and land cover over the study site, the classes chosen must fit three requirements: 1) the average patch size must be bigger than the spatial resolution of Sentinel-2 (10 m); 2) the variations of NDVI in a year must allow to separate the classes, as we’re using NDVI time series; 3) The classes chosen must fit with the goals of the study. Here, the classification must be able to map the main vegetation functional groups that constitute distinct habitats. First, an unsupervised classification (K-means cluster analysis performed with SNAP) was run with 10 spectral classes, based on images from February, August and December 2019. This clustering method groups all pixels with similar spectral characteristics into the number of classes defined by the user and is used to test for separability. I started with this unsupervised classification to test for general spectral signature separability. By comparing the classes created and the general cover of vegetation communities as identified on aerial imagery, I was able to determine which communities could be efficiently mapped (low spectral separability inside a class and high separability between classes). The final classes used for this study are: water, urban, bare soil, grassland, deciduous trees, sparse evergreen (mix of evergreen trees and grassland), and dense evergreen trees. The final classification was run with a supervised K-nearest neighbour (KNN) classifier, using five dates spread along the year (March 6, June 14, August 13, October 12 and December 16, 2019). Training and validation polygons were drawn from fieldwork, photo-interpretation of aerial images from the National Agriculture Imagery Program (NAIP) and Sentinel-2, and Fort Huachuca vegetation map.

The separate validation dataset was used to check the quality of the classification. The confusion matrix counts the number of correctly classified pixels in each class and is used to compute classification performance metrics. Producer’s accuracy is the number of accurately classified pixels against the observed total. It corresponds to how often observed land cover on the ground is correctly classified on the map. User’s accuracy is the number of accurately classified pixels against the predicted total and represents how often the class on the map will actually be present on the ground. User’s accuracy corresponds to the reliability of the map. Overall accuracy is the total number of accurately classified pixels against the total number of validation pixels. The Kappa coefficient (Cohen, 1960) is the result of a statistical test and compares the classification to a random classification. Kappa values are between -1 and 1. A value of zero indicates that the classification is not better than a random classification. Positive and negative values mean that the classification is better or worse than a random classification, respectively.

Variability in vegetation greenness and phenology

The second section of the results is focused on change in vegetation through time, focusing on greenness and phenology. Variability of post-monsoon (August-September) NDVI values for 2020 (weak monsoon) and 2021 (strong monsoon after drought) are compared to the 2018-2021 average. Then phenology metrics (peak, trough, timing and duration of the growing season) are compared for sampling points around each flow permanence sensor, by land cover and by year.

For this study, The growing season is defined as the time between the lowest NDVI value of the year (trough) and the highest value (peak). For each year, the five metrics measured are: trough (lowest NDVI value), peak (highest NDVI value), start of season (date when NDVI value starts to increase), end of season (date when peak NDVI value is reached) and length of season. To get these metrics, a daily time series was reconstructed from the Sentinel-2 NDVI images using a spline interpolation, and smoothed with a loess regression (span of 0.1). Even though we can only guess at probable NDVI values through Sentinel-2 data gaps, I chose a smoothing method and parameters that fit the closest with what is known of the phenology cycles of the vegetation communities studied and fit with the NDVI values measured by Sentinel-2. While this analysis is limited to four years, it is still useful to study vegetation dynamics during dry and wet years.

Simple linear regressions between phenology metrics (start and end of season, peak and trough) and climatic variables (rainfall and temperature) were run for data across the period 2018-2021 and for each vegetation community. I have tested various ways to aggregate climatic information into meaningful metrics. Rainfall influence has been tested with 1-month, 2-months, 3-months, 4-months and 5-months antecedent accumulated rainfall to take into account the potential effect of the monsoon and winter rain season and the storage of water in the soil. The impact of temperature has been summarized by looking at average temperature in the previous 15 and 30 days. Rainfall and temperature data are average daily values over the study area from the PRISM dataset. For each linear regression, the coefficient of determination (R^2) was used to determine which climate metric was the most tightly linked to phenology and only the strongest results (high R^2) are displayed in this thesis. This is meant as a very first step in exploring the relationship between climate and phenology, and a more detailed analysis would include looking at growing degree-days and developing models to test this relationship. A more advanced study of phenology and climatic controls in this region can be found in Warter et al. (2023).

4.4 Results

The first section of the results focuses on the land cover map obtained from the supervised classification, followed by analysis of the environmental controls on vegetation greenness and community distribution. Median NDVI values over the growing season, representing vegetation greenness, and the number of pixels of each land cover classes are compared to elevation. Then the same metrics are compared to flow permanence (measured in proportion of dry days in a year). In the second section, key phenology (start, end and length of growing season) and vegetation health (peak and trough NDVI) metrics are measured for the years 2018 to 2021, and their variations are linked to seasonal precipitation.

4.4.1 Land cover map and vegetation communities distribution

The supervised classification map from Sentinel-2 imagery shows the distribution of the main land cover types over the area. Bare ground is likely representative of sparse scrubland and roads, as seen during fieldwork, and bare plots in developed areas (south of Sierra Vista) (Figure 4.5). Bare ground covers the central part of the valley, with grassland mainly distributed along the San Pedro River and in the Huachuca Mountains foothills. The mountains are mainly covered in sparse evergreen trees (with grasses as the understory). At high elevation and on the north-facing slopes, dense evergreen trees are the dominant land cover, while deciduous trees can be found along ephemeral washes, in the canyons and in the San Pedro River floodplain. Deciduous trees patches are also scattered in the mountains, with an important area in the south above 2300 m.a.s.l. (Figure 4.5).

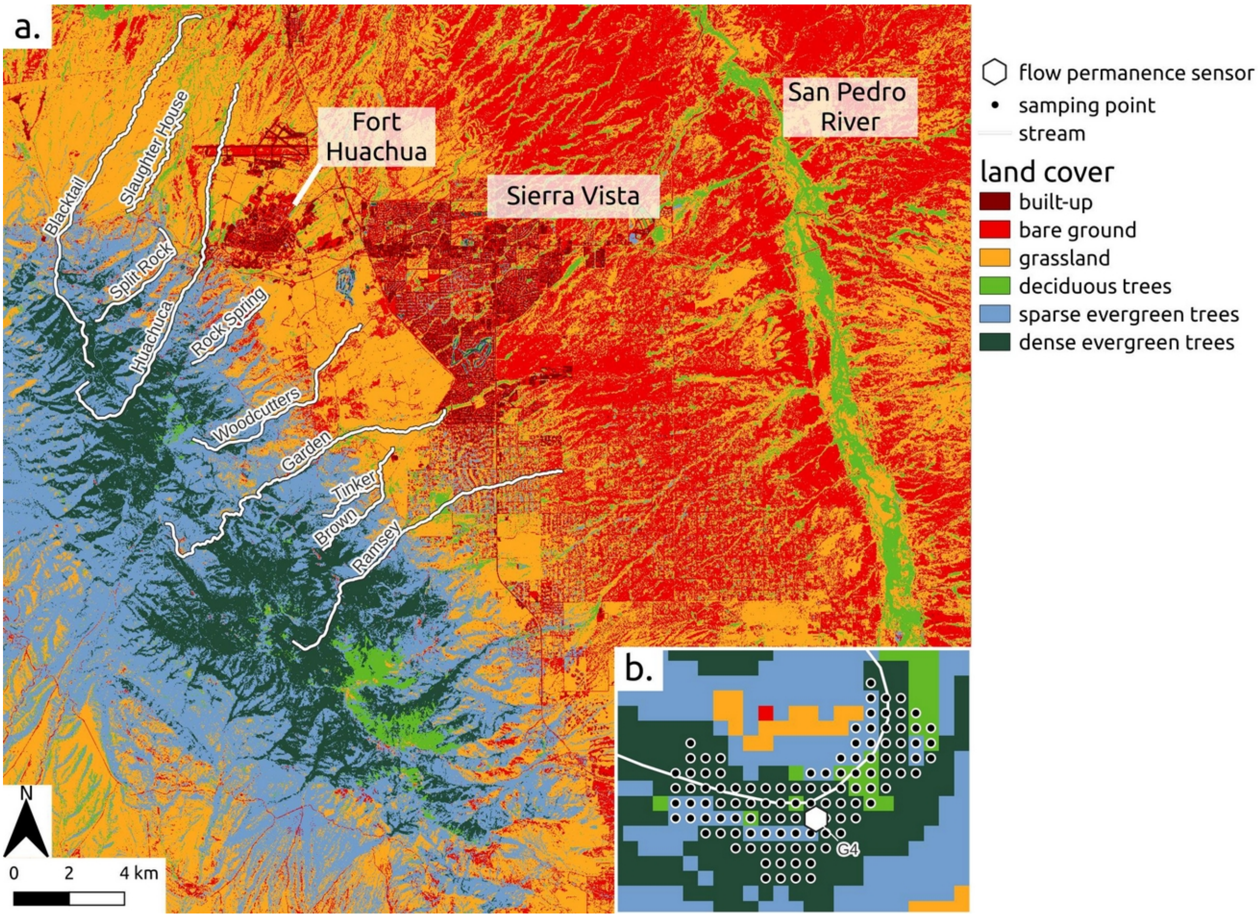

Figure 4.5: Land cover over Fort Huachuca and location of the flow permanence sensors (a) and close-up of the vegetation sampling points around sensor G4 (b).

The supervised classification accuracy was checked with a set of validation polygons, comparing observed and predicted classes for each pixel included in the validation dataset. The confusion matrix (Table 4.2) shows that all classes are properly mapped; with confusions mainly happening between classes of similar spectral signatures (for example urban and bare ground, grassland and bare ground). Confusion also happens between all classes of high vegetation greenness (all trees: deciduous, sparse evergreen and dense evergreen) since the classification is based on NDVI, which measures greenness. Accuracy is above 90% for all classes except urban built-up which shows a producer's accuracy of 88% (512 accurately classified pixels for 582 validation pixels) (Table 4.3). An overall accuracy of 0.98 and a Kappa of 0.97 suggest that this method is efficient for mapping the distribution of the land cover classes chosen in this study. While land cover classes where carefully selected to be most accurately mapped with the data available and classification method used, it is also important to note that there are limitations to the validation data. These validation polygons were drawn by looking at true-color imagery from the NAIP program and, as such, only includes areas where land cover could be easily identified. To improve the quality of the validation dataset, polygons could be drawn during fieldwork in areas where the dominant land cover is less uniform. Even with this caveat, the classification results are consistent with observations done in the field and previous studies in the area (Brady & Bonham, 1976; H. G. Shaw, 1999).

Table 4.2: Confusion matrix. For each validation pixel, the validation class (observed) is compared to the class assigned during the classification (predicted). Cells in light blue correspond to the number of accurately classified pixels.

		observed							
		water	**urban**	**bare ground**	**grassland**	**deciduous**	**sparse evergreen**	**dense evergreen**	**total**
predicted	**water**	684	0	0	0	0	0	0	684
	urban	0	512	31	0	0	0	0	543
	bare ground	0	68	10374	61	0	0	0	10503
	grassland	0	1	107	4842	74	166	0	5190
	deciduous	0	0	0	17	2596	33	1	2647
	sparse evergreen	0	1	0	1	21	6315	42	6380
	dense evergreen	0	0	0	0	34	11	2971	3016
	total	684	582	10512	4921	2725	6525	3014	28963

Table 4.3: Performance metrics: producer's and user's accuracy for each class, overall accuracy and Kappa.

	water	**urban**	**bare ground**	**grassland**	**deciduous**	**sparse evergreen**	**dense evergreen**
producer's accuracy (%)	100	88	99	98	95	97	99
user's accuracy (%)	100	94	99	93	98	99	98

overall accuracy	0.98
Kappa	0.97

Reconstructed NDVI time series for each vegetation community display the seasonal variations in vegetation greenness (Figure 4.6). For all vegetation communities, the lowest NDVI is reached at the beginning of the year (February-May), while the peak is reached in the second half of the year (August-September). Grassland and deciduous trees follow a bimodal phenological cycle typical of the influence of the North American Monsoon (Figure 4.6a, b) (Pennington & Collins, 2007; Weiss et al., 2004). the growing season starts at the beginning of the year, greenness dips during the dry season before following a second increase during the monsoon. Peak NDVI is reached and greenness immediately starts to decrease again. The first increase before the dry season is relatively low in grassland, but can represent half of the total greenness increase for deciduous trees. Dense evergreen trees (Figure 4.6d) keep a high greenness all year long (above 0.5) and the bimodal distribution is not as distinct as grassland or deciduous trees. Peak NDVI is maintained for most of the year but there is still a brief decrease in greenness at the start of the year. The “sparse evergreen” class, being composed of a mix of evergreen trees and grassland, follows an annual cycle similar to dense evergreen trees, but with lower values overall, and a slow decrease in greenness after the peak is reached (Figure 4.6c).

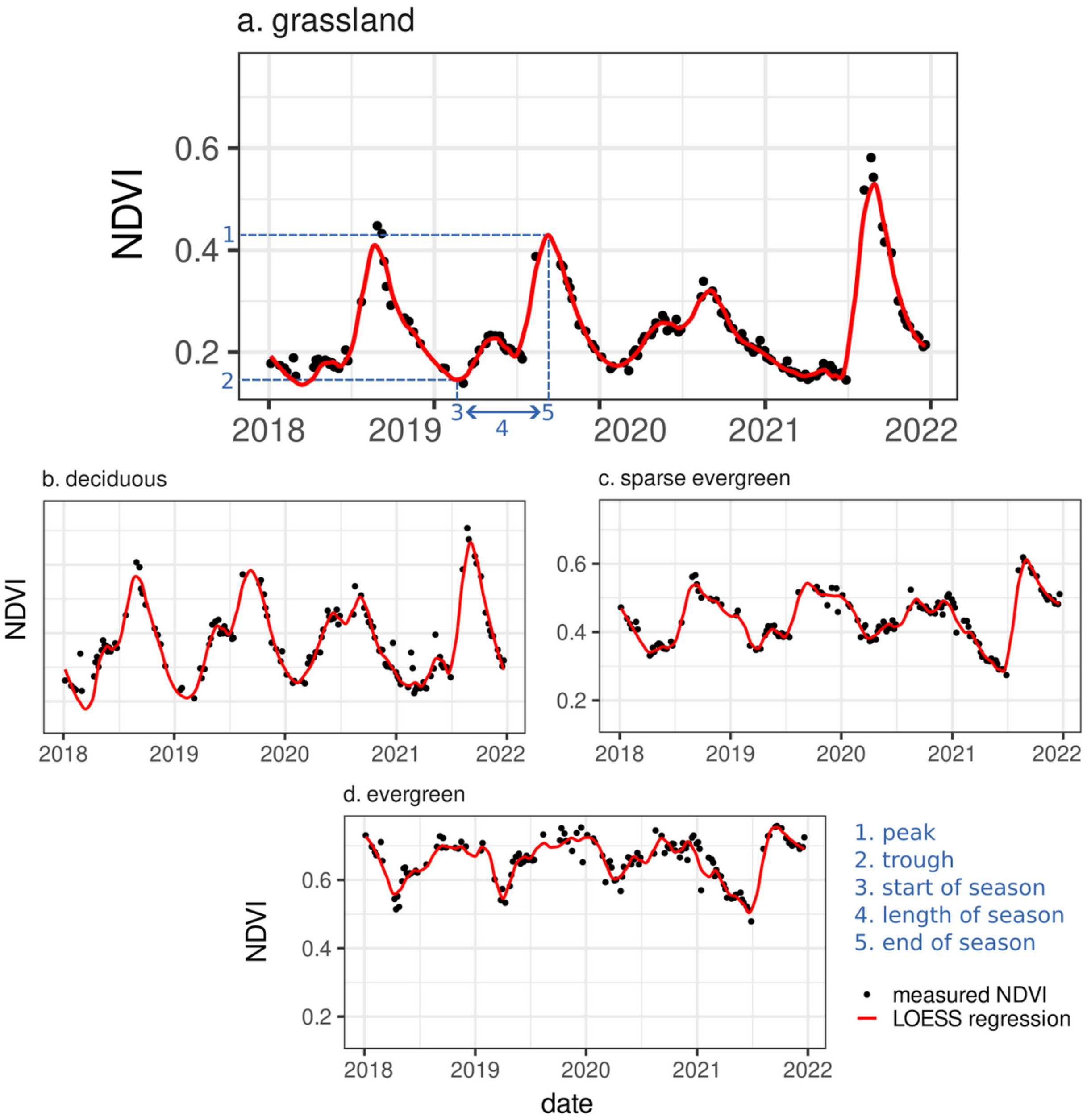

Figure 4.6: Measured NDVI (Sentinel-2, black dots) and interpolated daily NDVI values (red line) for grassland (a), deciduous trees (b), sparse evergreen trees (c) and dense evergreen trees (d) from 2018 to 2021, with the four main metrics used to characterize vegetation dynamics displayed: trough, peak, start, end and length of growing season.

4.4.2 Controls on vegetation along canyons

Elevation

Median NDVI over the growing season increases with elevation along the Huachuca Mountains canyons. NDVI median rises from 0.2 at 1400 m to 0.6-0.7 above 2000 m (Figure 4.7a). The highest NDVI values are reached at 2000 m, decreasing above 2000 m. All medians are significantly different except between 1700 and 1900 m ($p = 0.215$). The range of NDVI values is smaller at low (1400 m) and high (above 2000 m) elevations, and wider between 1400 m and 2000 m (Figure 4.7a), and all distributions are significantly different except between the 1700 and 1900 m groups, according to pairwise Kolmogorov-Smirnov test. A similar pattern is observed even when stratifying the data by flow permanence, with very dry reaches and very wet reaches showing an increase in NDVI values (from 0.2 to 0.7) with elevation and shifts in range as elevation increases (small range of values at 1400 and 2000 m but wider range of values at mid-elevation (1500 to 1900 m) (Figure 4.7b, c).

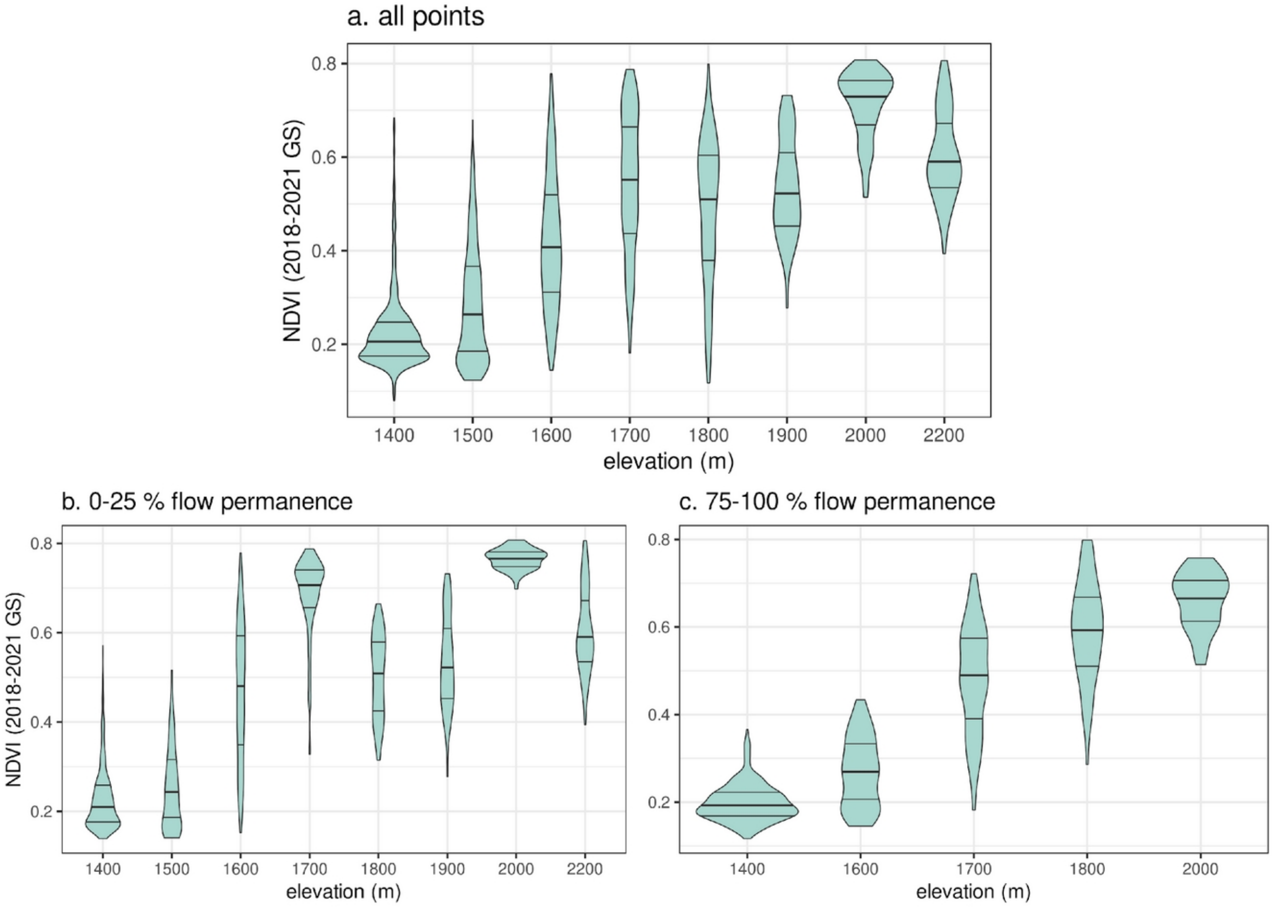

Figure 4.7: Violin plots of NDVI distributions for vegetation around sensors (2018-2021 growing season median), grouped by elevation (rounded to the lower thousand, e.g. all sensors between 1400 and 1499 m are in the 1400 m group). All points (a), only points around sensors with flow permanence under 25% (b) and over 75% (c).

Vegetation communities are stratified by elevation along the Huachuca Mountains, from grasslands in the valley to sparse evergreen mixed with grassland and dense evergreen forests at the summit (Figure 4.8, 4.9). The increase in NDVI (Figure 4.7) can be attributed to these shifts in vegetation, from the low NDVI of the grassland (under 0.25) to the high NDVI (above 0.5) of the evergreen trees. Ramsey and Huachuca Canyons, which cover the widest elevation range from ≈ 1400 m to ≈ 2200 m, demonstrate this vegetation shift (Figure 4.8). There’s a shift in community around 1600 m, from a landscape dominated by grassland and deciduous trees, to mostly evergreen and mixed evergreen trees. Deciduous trees are the only vegetation community present at most elevations. This terracing of vegetation is in line with numerous studies (Brady & Bonham, 1976; Poulos et al., 2007).

The wide range of NDVI at mid-elevation observed in Figure 4.7 appears to be caused by a mix of communities occupying the same elevation. Low and high elevations are less mixed, with a smaller range in NDVI values, dominated by grassland at low elevation (below 1500 m) and evergreen trees at high elevation (above 1750 m) (Figure 4.8 & 4.9). The mid range (above 1500 m and under 1750 m) is more mixed, with grassland, deciduous trees and sparse evergreen trees (Figure 4.8 & 4.9). Deciduous trees also show higher NDVI values over 1500 m, as we go from to all deciduous trees points under 0,5 to a wide spread of NDVI values around 1500 m (from 0.25 to 0.8) and mainly higher values (above 0.5) over 1600 m.

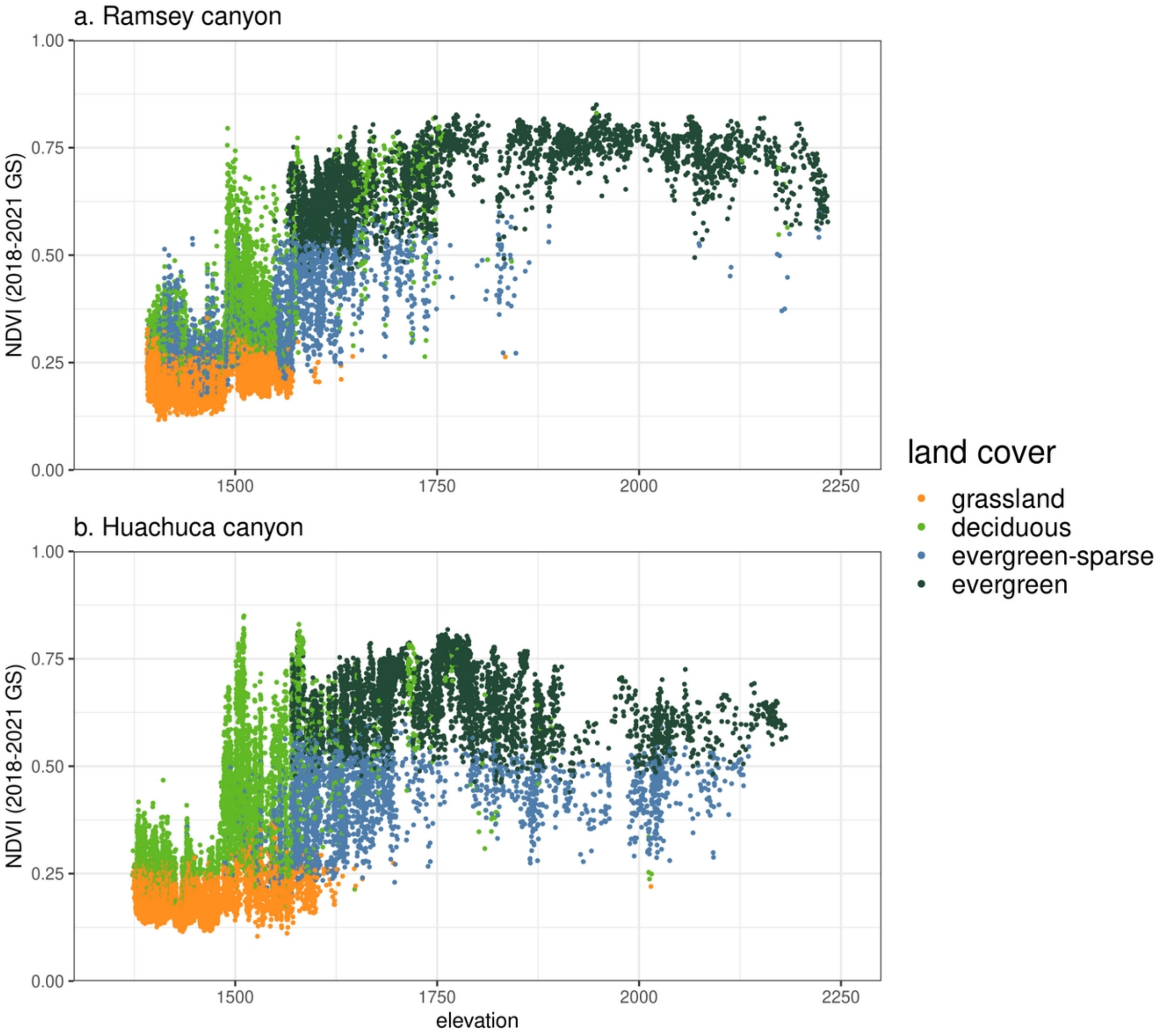

Figure 4.8: Plots of NDVI (2018-2021 growing season median) for Ramsey Canyon (a) and Huachuca Canyon (b), includes all points in the floodplain, and color-coded by vegetation class.

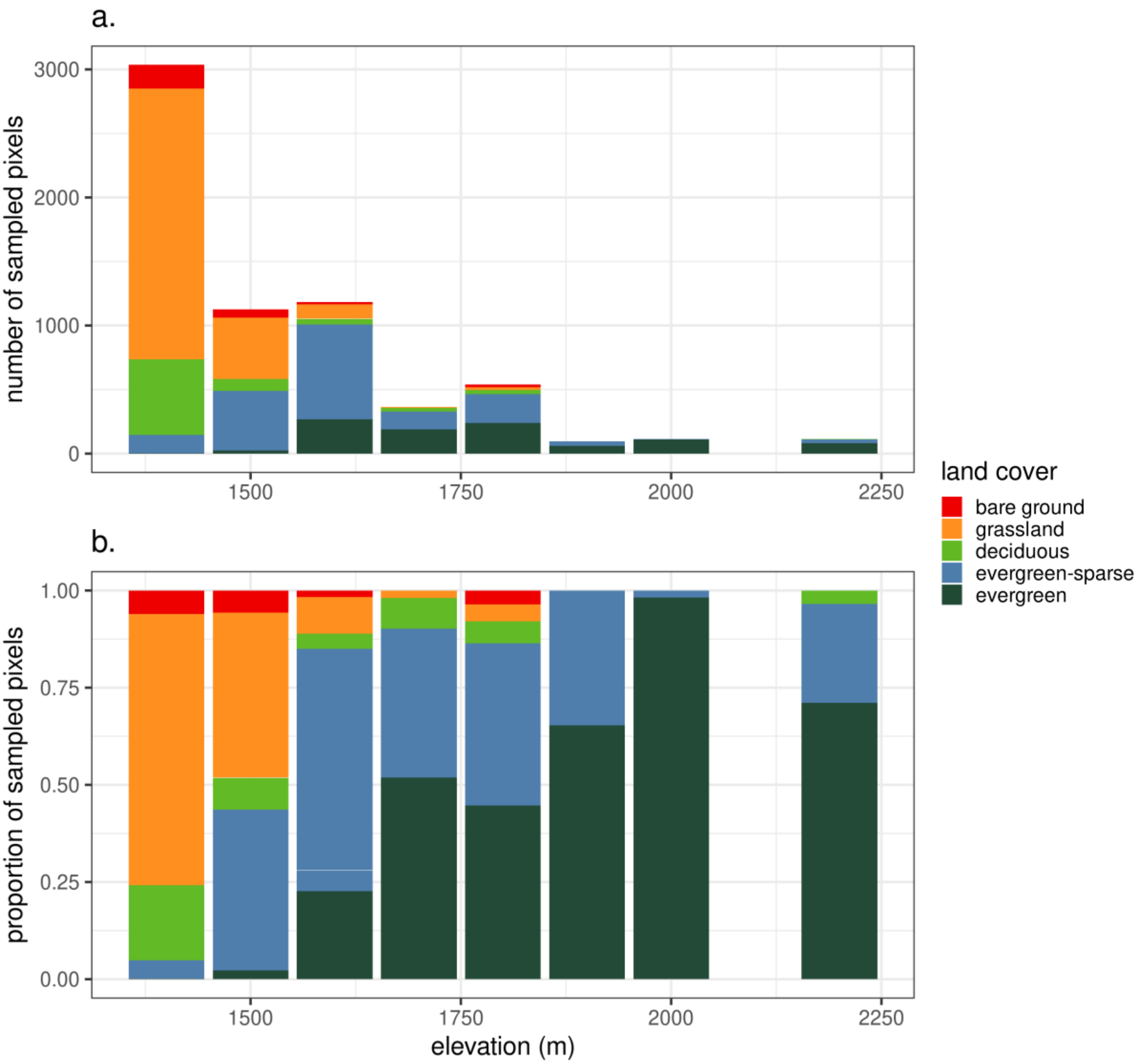

Figure 4.9: Distribution of land cover by elevation, as a function of the number of sampling points (a) and proportion of sampled pixels (b). The figure includes points around sensors for all canyons, grouped by elevation and color-coded by vegetation class.

flow permanence

There is no obvious trend in NDVI values or vegetation shifts between flow permanence groups. All land cover classes are represented across the range of flow permanence in the Huachuca canyons (Figure 4.10). Land cover classes are arranged along the NDVI values axis, with bare ground and grassland at low values and evergreen trees at high values. The main noticeable difference is that deciduous trees at very low flow permanence (75-100 % of dry days) cover a small range of NDVI values (0.2-0.35) compared to deciduous trees at higher flow permanence (0.2-0.7 range). Grasslands and bare ground represent the highest number of points with grouped low NDVI values, and are the reason the overall distribution show a concentration of lower values in each group. This is likely due to the floodplain being wider in the semi-arid valley, which means more sampling points are in this area.

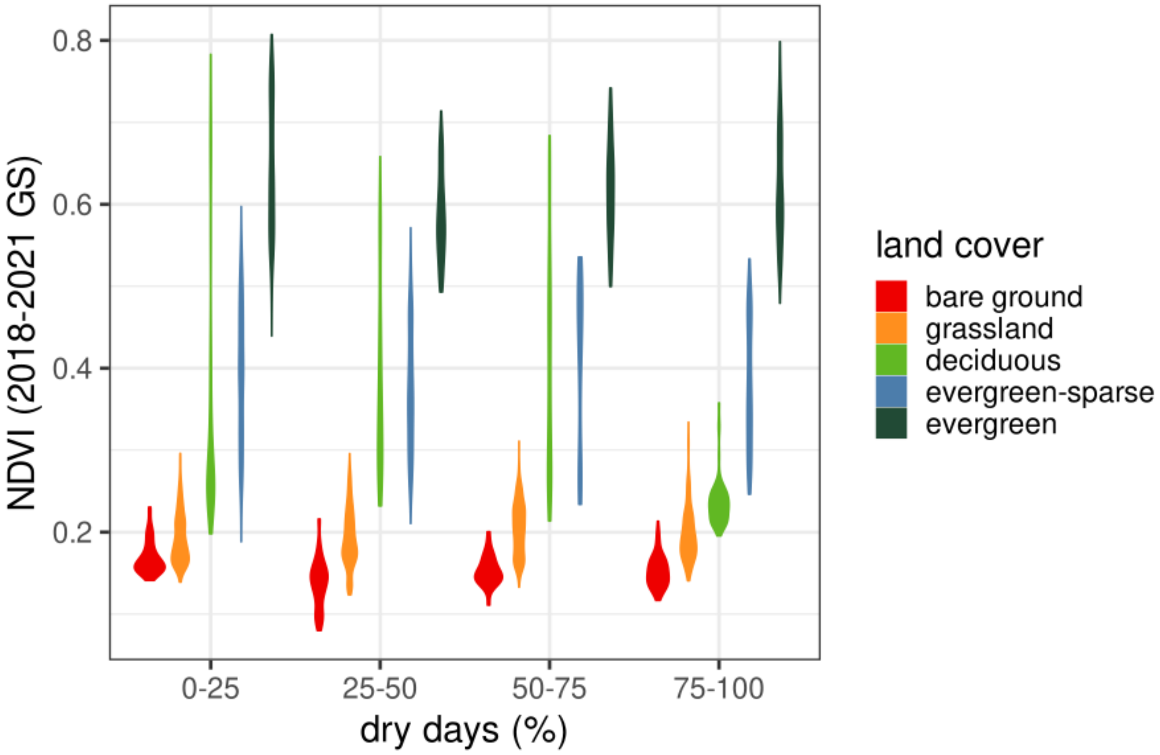

Figure 4.10: Violin plot of NDVI (2018-2021 growing season median), includes all points around sensors, split by flow permanence and land cover (% of dry days in a year).

There is no shift in vegetation distribution along increasing dryness when looking at elevation subsets (Figure 4.11). Under 1600 m, grassland is the dominant land cover class, except in flow permanence group 25-50 %, which is mainly covered in both grassland and sparse evergreen trees. Group 50-75 % has the highest cover of deciduous trees. At median and high elevation, dense evergreen forest is the main land cover, followed by sparse evergreen trees, with no visible influence of flow permanence on land cover distribution. Flow permanence 25-50 % at median elevation is the only group mainly covered in sparse evergreen trees.

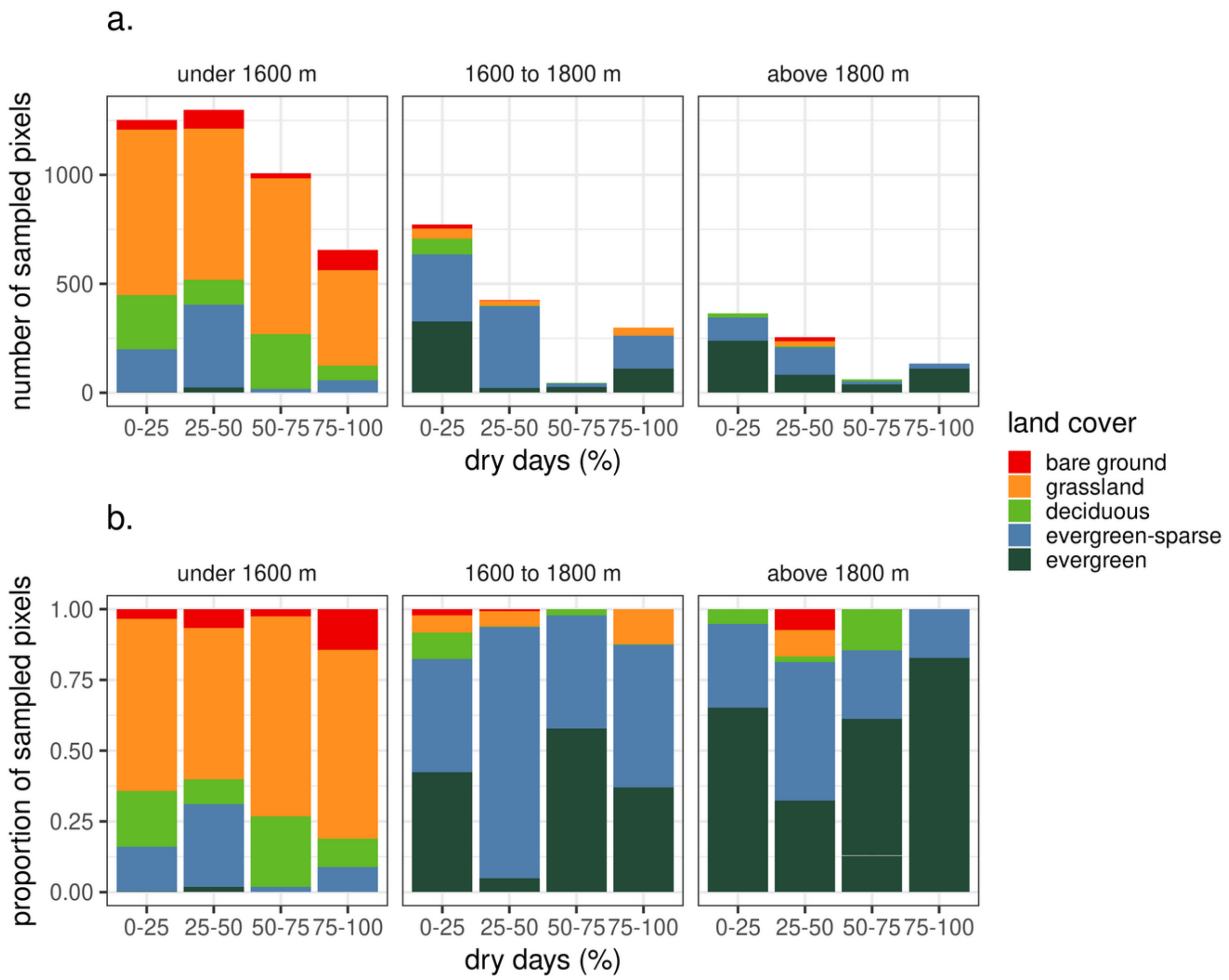

Figure 4.11: Distribution of land cover by flow permanence. Includes points around sensors for all canyons, grouped by elevation and color-coded by vegetation class. In number of points (a), in density (b).

4.4.3 Dynamics and phenology

Post-monsoon trends

When comparing each season’s rainfall and temperature to an average of that same season over 1991-2020 (Figure 4.2b, c), we notice that 2018 is variable but slightly drier during the monsoon and winter, and 2019 is wetter than average for all seasons. 2020 is characterized by a good winter (145 mm, +34 mm anomaly) followed by a rainfall deficit for the rest of the year, including a very dry and hot monsoon season (126 mm, -108 mm and +2°C anomaly). This deficit continued in 2021, with a hot autumn (+2.6°C anomaly) and a dry winter (69 mm, -42 anomaly) and spring, before a strong monsoon (297 mm, +62 mm anomaly). Drought patterns over Cochise County, which includes the study site, follows the rainfall distribution with an offset (Figure 4.2d).

The drought in 2020 and recovery in late 2021 have had consequences on vegetation across Fort Huachuca. When mapping the difference between the post-monsoon (August-September) average for 2018-2021 and the same period in 2020, following a very weak monsoon (Figure 4.2), there was an overall decrease in NDVI, mainly focused in specific spots in the grasslands, along valley washes and along Garden Canyon (Figure 4.12a), while evergreen forests in the mountains are less affected and show little change. In 2021, a strong monsoon (+62 mm anomaly compared to 1991-2020 median) came after a year of weaker-than-average rainfall. Grassland and washes reached NDVI values higher than the 2018-2021 average, while evergreen forests remained at average levels (Figure 4.12b).

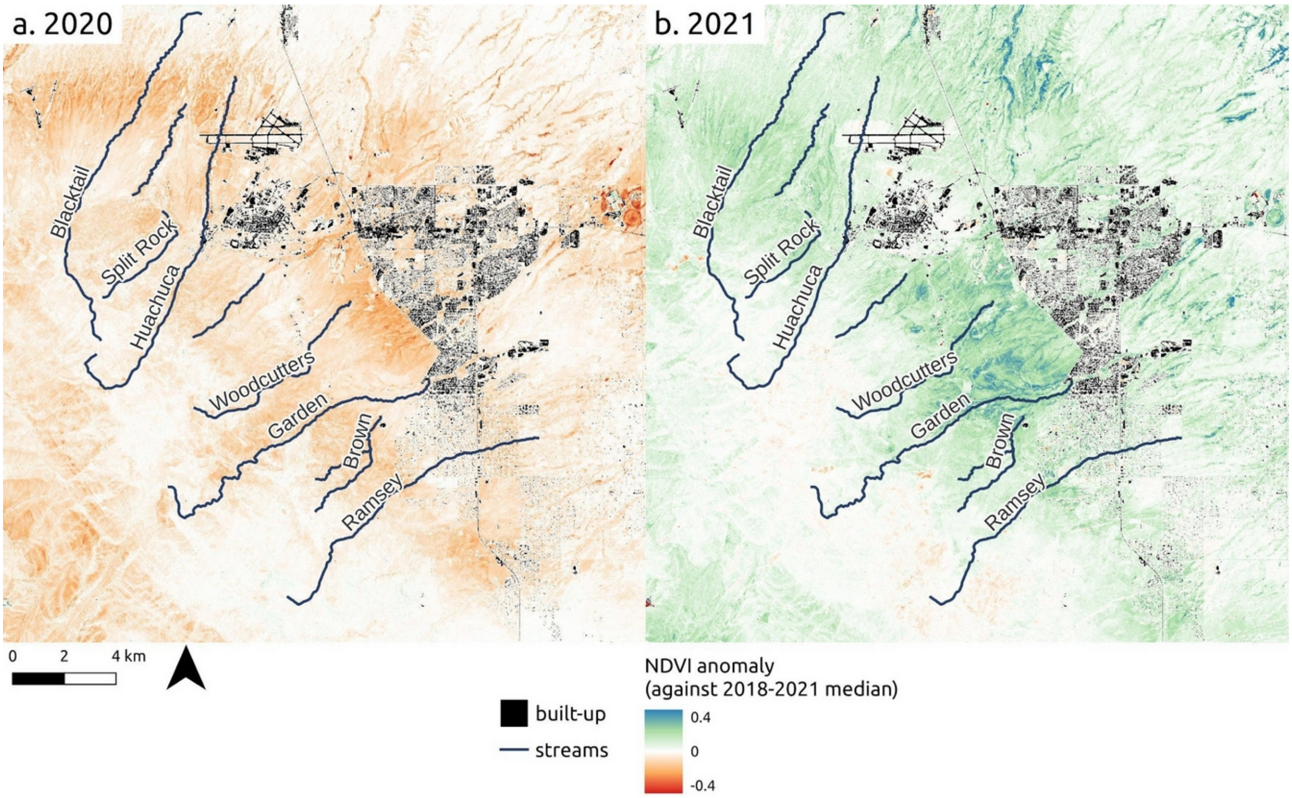

Figure 4.12: Maps of pixel-wise NDVI post-monsoon variability across the study site. Baseline used is the median NDVI of all August and September from 2018-2021 compared against the NDVI median of August-September images for 2020 (a) and August-September images for 2021 (b).

Sensitivity to monsoon strength differs with vegetation type. Grassland, deciduous trees and sparse evergreen trees all showed a similar decrease in NDVI for the August-September period in 2020 compared to the 2018-2021 average (Figure 4.13). They also recovered similarly, indicated by NDVI values above average levels. Grassland showed the highest increase (+0,11), followed by deciduous trees (+0,08) and sparse evergreen trees (+0,06). Dense evergreen trees are the one that showed the least amount of change (-0,01 in 2020 and +0,008 in 2021). All medians are significantly different from each other between land cover classes and between years.

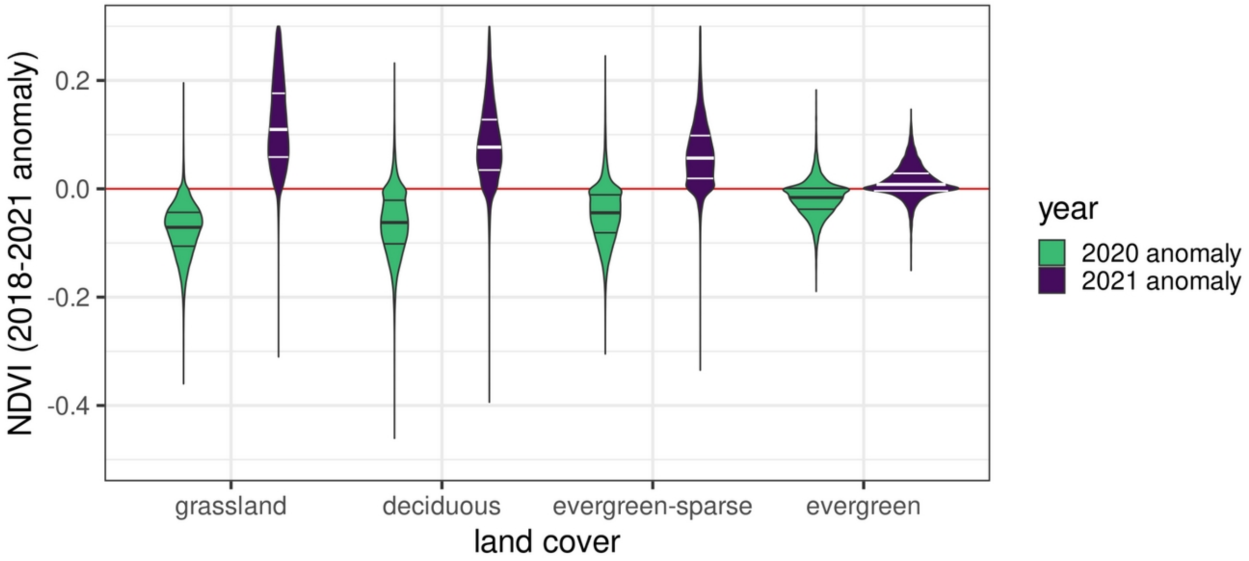

Figure 4.13: Violin plot of variability between median for August-September 2018-2021, compared against median of August-September images for 2020 and 2021. Groups by land cover, color for 2020 (green) and 2021 (purple).

Phenology variability

Trough and peak NDVI is highly variable for all vegetation communities. For all classes, peak and trough are relatively similar in 2018 and 2019. Then the 2020 growing season displayed the highest trough and the lowest peak, except for evergreen trees, which maintained the same peak as the previous year. In 2021, trough NDVI is slightly higher than the 2018-2021 median for deciduous trees (+0,02 anomaly), equal to period median for grassland, and lower than period median for dense and sparse evergreen trees (-0,02 and -0,05 respectively). Peak NDVI reached its highest value in 2021 for all vegetation classes. For all classes, 2020 is the year of the shortest value range (highest trough and lowest peak), while 2021 is the year of the widest range (lowest trough and highest peak) (Figure 4.14).

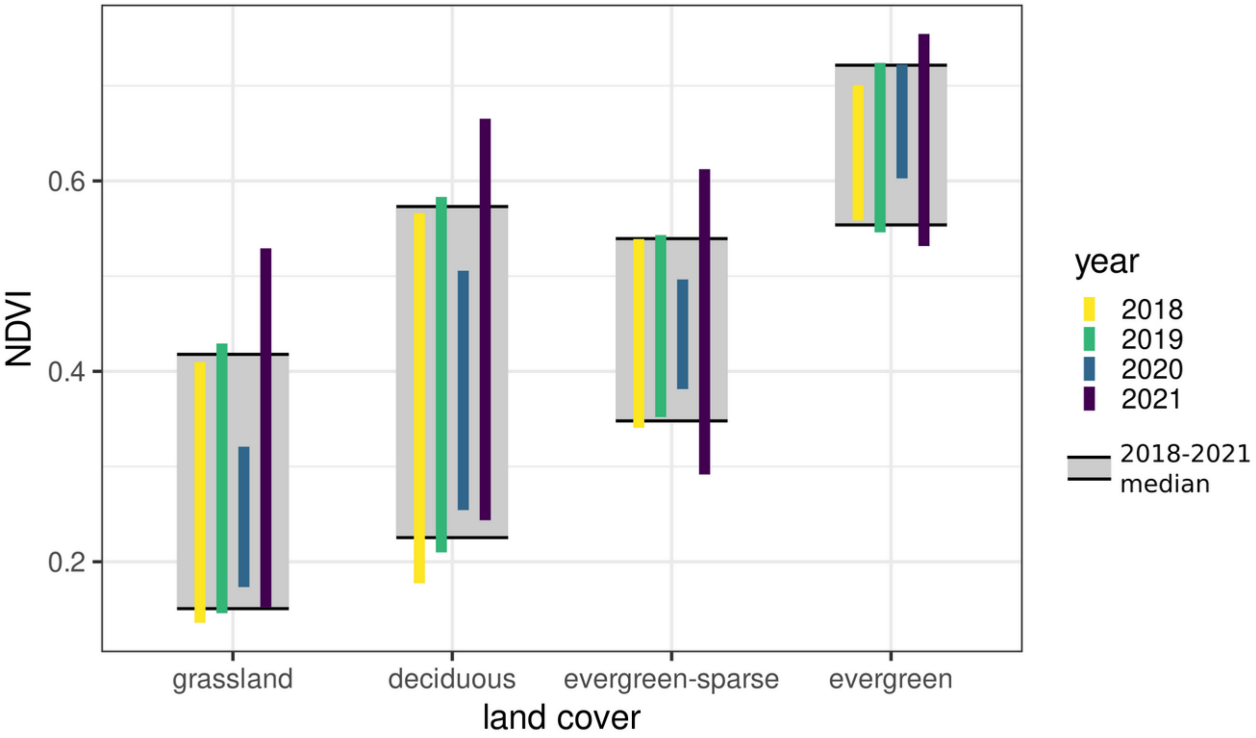

Figure 4.14: NDVI range covered during the growing season, from lowest value (bottom of bar) to highest (top of bar), by land cover (grassland, deciduous trees, sparse evergreen trees and dense evergreen trees) and by year (2018 to 2021). The horizontal black lines represent the 2018-2021 average minimum and maximum NDVI values for each land cover class.

Different land covers showed differences in growing season between years (Figure 4.15). For grassland, there was an increase in the length of the growing season between 2018 (160 days) and 2019-2020 (203 and 206 respectively) while the 2021 growing season was very short 2021 (93 days). Sparse evergreen trees displayed a similar pattern with 2019 and 2020 slightly longer than 2018, and 2021 much shorter than every other year (going from 157 days in 2020 to 94 days in 2021). Deciduous trees had an increase in growing season length in 2019 and 2020. Their 2021 season was similar to 2018, compared to all other vegetation classes, which had a much later start in 2021 compared to all other years. Overall, grassland and deciduous trees had a longer growing season and dense and sparse evergreen trees, though in 2021, grassland's growing season was the same length as sparse evergreen trees. All classes had a similar end of season for every year, which match to the end of the monsoon, except for evergreen trees that show a very long growing season in 2019 (273 days, against 159 days in 2018, 154 days in 2019 and 114 days in 2021). Grassland and deciduous trees had the earliest start of season and so the longest growing season. In 2021, all classes had a delayed start and the shortest growing season of the 2018-2021 period.

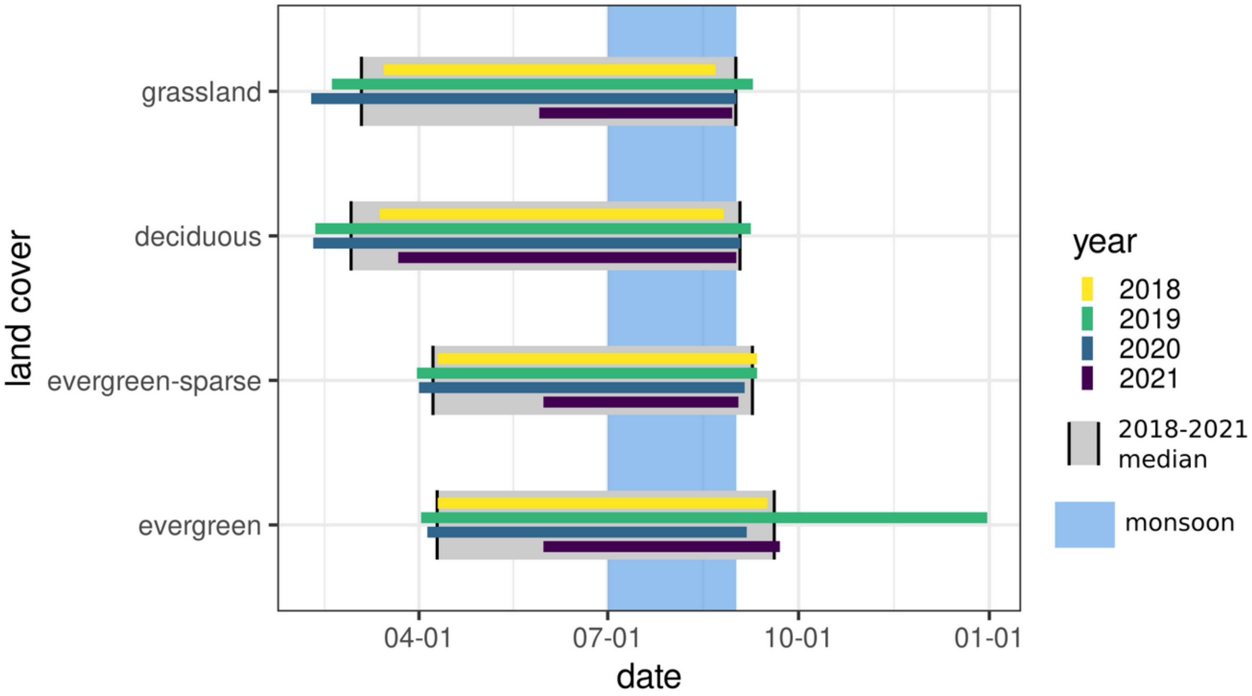

Figure 4.15: Period covered by the growing season by land cover (grassland, deciduous trees, sparse evergreen trees and dense evergreen trees) and by year (2018 to 2021). Light blue background is the monsoon. The vertical black lines represent the 2018-2021 average start and end of season for each land cover class.

Climate controls on phenology

Phenology metrics for each vegetation community were compared to several climate variable (antecedent accumulated rainfall for 1 to 5 months and antecedent average temperature for 15 and 30 days). Peak and trough, expressed as maximum and minimum NDVI values reached during a growing cycle, did not have a strong correlation with any of the climate metrics for any of the vegetation communities (R^2 always under 0.3 for rainfall and under 0.2 for temperature). The start of the growing season was closely correlated to both 30-days temperature and 4-months antecedent rainfall (Figure 4.16a, b), with higher temperatures and lower precipitation associated with a delayed start. Temperature R^2 was over 0.8 and rainfall R^2 over 0.6 for evergreen trees (both dense and sparse) and grassland. Deciduous trees showed the lowest fit (R^2=0.2 for 4-months rainfall, R^2=0.6 for 30-days temperature). The date of the end of the growing season for evergreen trees (dense and sparse) also showed a strong correlation to 15-days temperature (R^2>0.9) when again, deciduous trees are the community least correlated to the climate variables studied (Figure 4.16c). Dense evergreen trees was the only community with a coefficient of correlation over 0.5 (0.54) when comparing the end of season to antecedent rainfall.

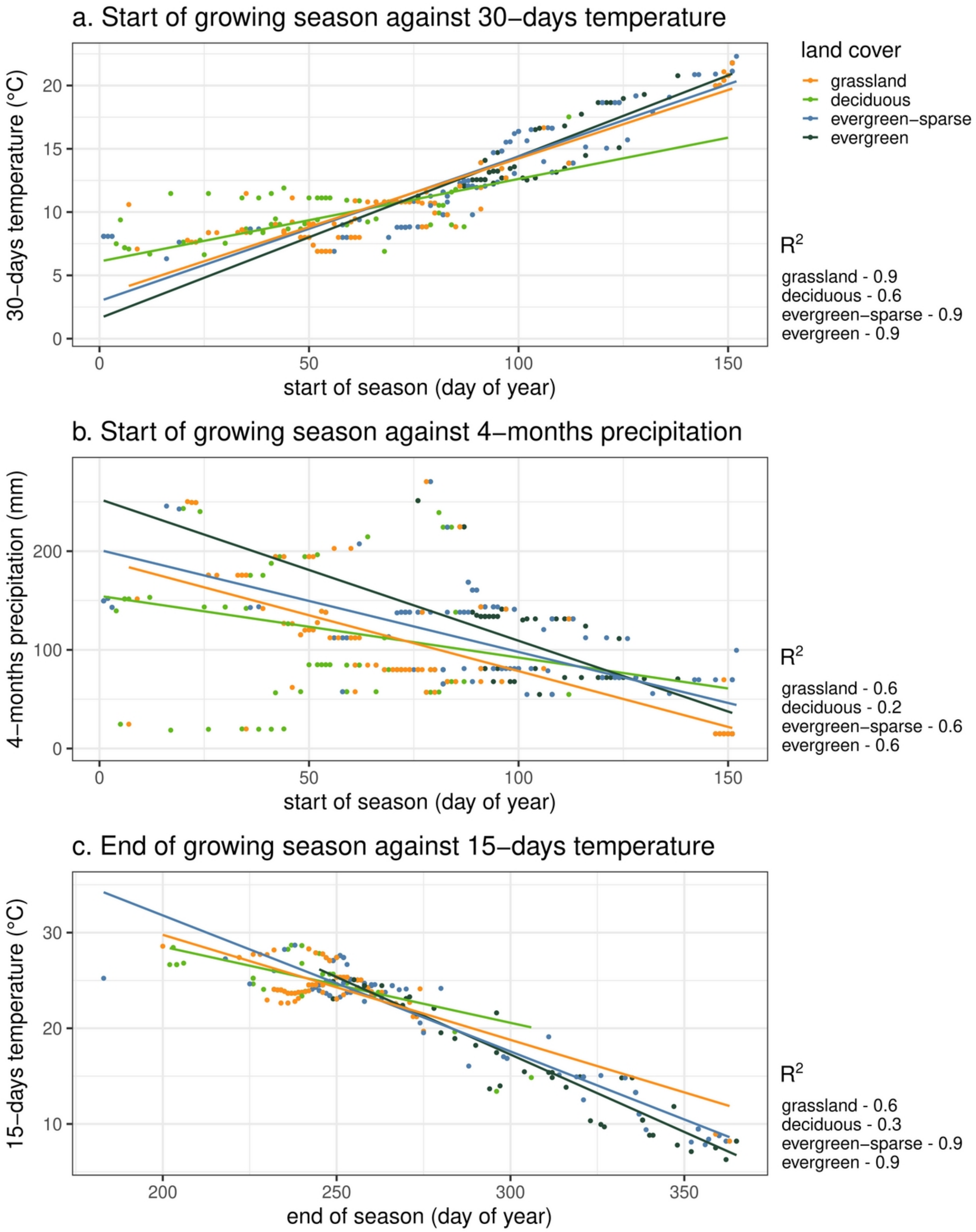

Figure 4.16: Linear regressions between start of growing season and 30-days temperature (a), start of growing season and 4-months precipitation (b), end of growing season and 15-days temperature (c).

4.5 Discussion

4.5.1 Controls on vegetation

Vegetation communities along the Huachuca mountains canyons follow the elevation gradient, from grassland (below 1500 m) to dense evergreen forest (above 1750 m), with sparse evergreen trees mixed with grassland in between and deciduous trees scattered throughout. This result concurs with previous vegetation survey in the Huachuca Mountains (Brady & Bonham, 1976; H. G. Shaw, 1999; Wallmo, 1955). From these works we know that the sparse evergreen trees mixed with grasslands are mainly oaks with alligator juniper (*Juniperus deppeana*) also present, while the dense evergreen forest are dominated by oaks and pines. The increase in NDVI with elevation is due to this shift in vegetation, from low NDVI of grassland to high NDVI of dense evergreen trees. Low and high elevations are more uniform (low grassland and high evergreen) with mid-elevations more diverse. Elevation is a strong factor on vegetation distribution (Poulos et al., 2007; H. G. Shaw, 1999), as elevation is a proxy for a climatic gradient of higher precipitation and lower temperature as elevation increases, which exerts a strong control on soil moisture deficit either by affecting water supply or water demand (Padien & Lajtha, 1992; Urban et al., 2000). There is also a gradient in NDVI values for deciduous trees, from low values at low elevation to a wider range at higher elevation, which might be due to the shift in species, from mesquite along the highly ephemeral washes in the valley grassland, to a mix of riparian trees, including Arizona sycamore (*Platanus wrightii*), Arizona walnut (*Juglans major*) or velvet ash (*Fraxinus velutina*) in the canyons (Shaw, 1999) (Figure 4.17). The high elevation patch of deciduous trees in the south of the Huachuca Mountains mapped in the land cover map might correspond to the Gambel oak (*Quercus gambelii*) community described by Brady & Bonham (1976). They noted that this deciduous species was present at 2600 m.a.s.l, which fits with the distribution observed in the land cover map.

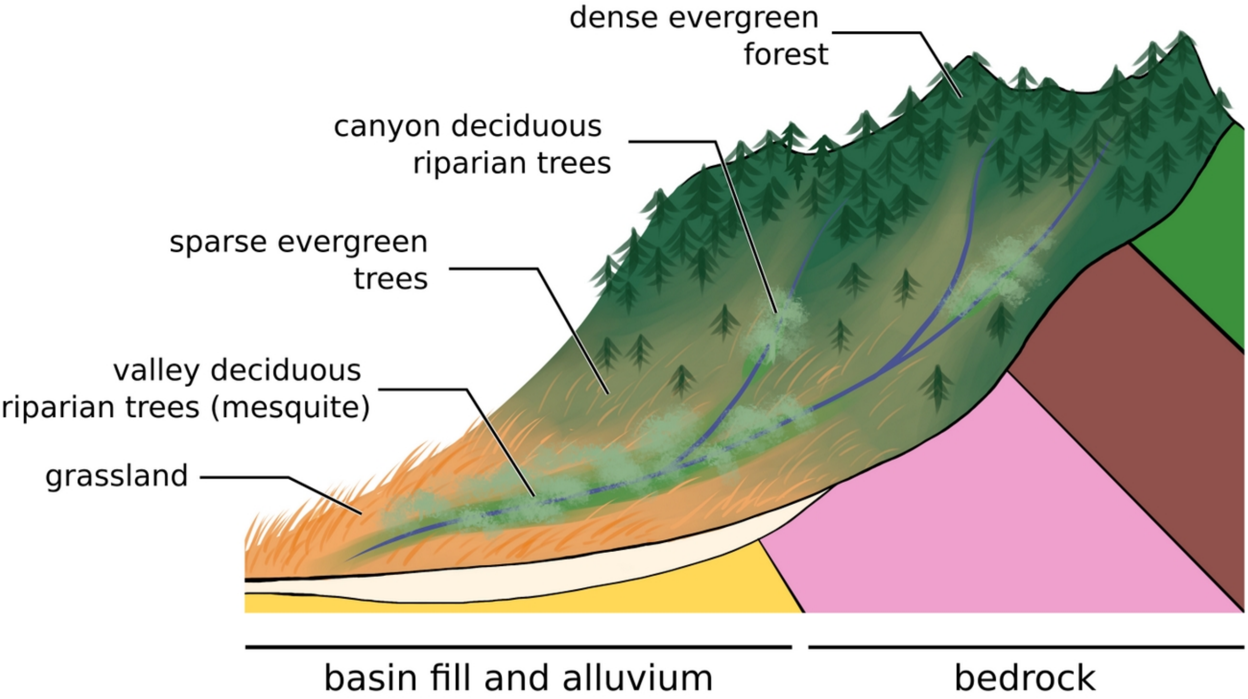

Figure 4.17: Vegetation communities distribution in the Huachuca Mountains. In the valley grasslands are the main vegetation community, with mesquite trees along intermittent streams. As elevation increases, grasslands are mixed with sparse evergreen trees, and dense evergreen forests cover the higher elevation. The underlying geology of the valley floor is comprised of permeable basin fill and alluvium, while the Huachuca Mountains are impervious bedrock with fractures and faults.

There does not appear to be a strong control of flow permanence on vegetation, at least at the scale we are working at. Along most reaches at higher elevation, vegetation was similar along the stream and upland, as is possible in first order streams (J. R. Shaw & Cooper, 2008), though there still were scattered deciduous riparian trees in small groups and mixed with evergreen trees, thanks to higher soil moisture due to runoff and shallow water table (H. G. Shaw, 1999). Previous studies (Katz et al., 2012; Stromberg et al., 2015) found a difference in riparian vegetation with flow permanence, and fieldwork showed small wetlands around springs (Figure 1.22) and riparian trees, such as Arizona sycamore (*Platanus wrightii*). Along the ephemeral channels in the valley, mesquite forms narrow riparian corridors. The shift in riparian species and density is due to the difference in elevation, slope and geology (Chapter 3). In the mountain, water is mainly stored in fissures in permeable units (such as limestone), with very little alluvium, while in the valley, there is a shallow water table in alluvium and basin fill (Chapter 3), at least close to the mountain front that provides enough water for the mesquite corridor.

4.5.2 Dynamics and phenology

2019 was a year with an above average monsoon and winter rain seasons and cooler temperatures, followed in 2020 by a good winter but a very weak monsoon with high temperatures, and in 2021 by a dry winter and strong monsoon. Comparing post-monsoon NDVI anomaly across the study site highlights areas of high or low sensitivity. Over Fort Huachuca, grasslands are more reactive to rainfall (sharpest decrease in NDVI during drought and highest increase during recovery) as they fully rely on soil moisture from rainfall. Vegetation along washes (highly ephemeral streams in the valley that only flow during rainfall events) also react strongly to rainfall variations, especially as we move away from the mountains. In this area, washes are on top of a highly permeable basin fill and alluvium (Chapter 3), water travels down and the water table can reach depths of 90 m (Chapter 2). As this regional water table is inaccessible to vegetation, riparian trees and shrubs along washes depend on rainwater concentration in the channel, which makes them highly sensitive to droughts. Evergreen trees in the mountains have access to shallow water storage in the fractures and faults recharged through snow melt and winter precipitation (Chapters 2 & 3) and are less sensitive to short term changes in rainfall distribution.

2020 was a year with good precipitation during the previous year, strong autumn and winter rains followed by a weak monsoon and no autumn rains. Most vegetation communities managed to remain greener than usual during the winter, but vegetation was less dense than usual at the end of the summer, except for dense evergreen trees who are less sensitive to monsoon rains thanks to access to groundwater recharged during the winter. An early start allowed for a longer growing season. In 2021, precipitation distribution was reversed, with a dry preceding year and dry winter, followed by a strong monsoon. The start of the growing season was delayed for all vegetation communities, which led to a shorter growing season. This delayed start might have been due to the lack of water storage following the weak monsoon, autumn and winter, which caused the exceptional drought recorded in 2021. Evergreen trees showed a lower trough, which can be interpreted by an abnormal loss of leaves. Grassland and deciduous trees, which lose their leaves every year, did not show this unusual drop. All vegetation classes were denser than usual at the end of the monsoon, even after a weak winter, with grassland and deciduous trees displaying the strongest increase.

The growing season tends to start earlier during years with good winter rains (2019, 2020), with grassland and deciduous trees showing the earliest start. Both higher temperatures and lower precipitation, which are important factors of drought, tend to delay the start of the growing season for all vegetation communities studied here. The end of the growing season appears to be more stable than the start, almost always happening around the end of the monsoon, no matter its strength. The end of the growing season seems partially driven by temperatures, with higher temperatures associated to an earlier end of season, and could be controlled by additional factors such as photo-period and seem less sensitive to water availability. Dense evergreen trees display the most variable end of season. In 2019, the highest NDVI value was reached at the very end of the year, though the main NDVI increase happened earlier in the year and was followed by a slow and steady increase. This more irregular pattern would fit with evergreen trees, that don't lose their leaves all at once and which phenology cycles are much less pronounced than grassland or deciduous trees. Interestingly, the U.S. Geological Survey phenology viewer, which provides timing of key phenology event over the USA every year, also warns of an abnormal delay in the end of season in 2019.

One year of drought was enough to have visible consequences on all vegetation communities of the Huachuca Mountains, but was followed by a very quick recovery following a strong monsoon, which shows the adaptability and resilience of these vegetation communities (Brown-Mitic et al., 2007; Knowles et al., 2020), but also how rainfall distribution and temperature are key phenology and vegetation health controls in drylands (Currier & Sala, 2022; Gómez-Mendoza et al., 2008; Jenerette et al., 2010). Several years with higher temperatures and low precipitation in both winter and during the monsoon could lead to a lower vegetation greenness after the monsoon, a shorter growing season due to a delayed start, and abnormal loss of leaves in winter for evergreen trees (Figure 4.18). These changes can be considered as a loss of habitat quality for species used to dense vegetation with regular life cycles providing food and shelter during critical times. A delayed start of the growing season might lead to a lack of food or suitable dense trees for nesting during the breeding season of endangered species (Coe et al., 2012; Kellermann & van Riper, 2015). Riparian vegetation serves as habitat, shelter and food source for endangered species such as the Northern buff-breasted flycatcher (*Empidonax fulvifrons pygmaeus*) or the Southwestern Willow Flycatcher (*Empidonax trailii extimus*) (Bagne & Finch, 2013). A delayed start and shortened length of the growing season could mean that shelter and food are no longer synchronous with the life cycle of the animals that need them.

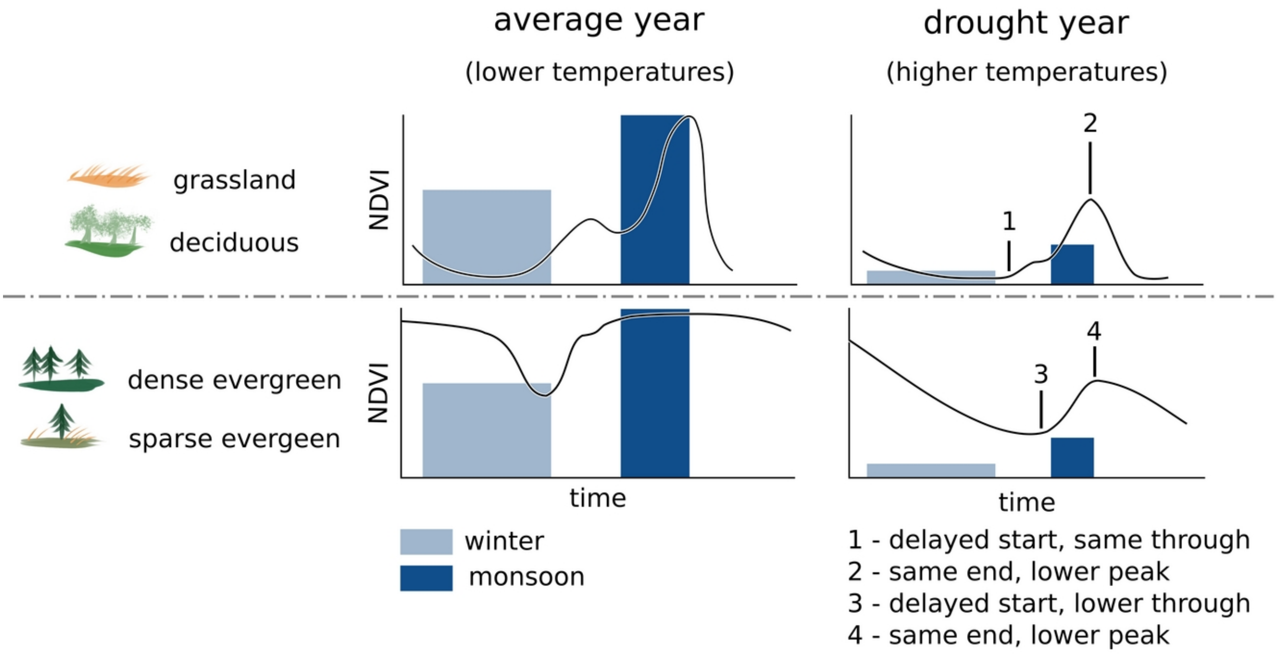

Figure 4.18: Theoretical effect of a drought year (high temperatures and low precipitation) on the phenology of the Huachuca Mountains vegetation communities. Grasslands and deciduous trees follow a bimodal NDVI distribution during an average year, but drought leads to a delayed start of the growing season and a lower peak. Dense and sparse evergreen trees keep a higher NDVI year round with a short period of lower NDVI at the end of the winter rains and a shorter growing season. During droughts, evergreen trees show a delayed start of the growing season, a lower trough and a lower peak.

4.5.3 Management and conservation

The methods used here can provide important insight to local vegetation communities. A trained vegetation mapping can be run every five or 10 years to provide an up-to-date vegetation map and keep track of shifts in general vegetation distribution. If a community is considered a critical habitat for endangered species, vegetation mapping can be applied to follow shrinking, expansion or fragmentation of this habitat. For example, the protected Florida scrub-jay (*Aphelocoma coerulescens*) found at Cape Canaveral Space Force Station is dependent on oak scrub and is victim of habitat loss. Remote sensing could be used to identify high quality habitat based on vegetation greenness and track how this vegetation community has been changing, building on the work by Breininger et al. (1991). Mapping NDVI values anomaly for drought years, or after a fire, can help locate and map areas that are more sensitive, or recover more slowly, and might need specific management interventions. For example, keeping track of NDVI anomalies along a stream during a drought can highlight which areas are losing leaves and showing a decrease in greenness, which is a starting point to then investigate the reasons of decline in these areas and take corrective measures if possible (such as reducing water pumping nearby, or removing weirs or pipes diverting streamflow). Anomaly maps can also be produced for any metric of interest, such as peak and trough NDVI, or timing and length of the growing season. These methods can be applied anywhere, and adapted to available data (for example using radar, which is able to see through clouds, LiDAR to map vegetation height and forest structure, or Landsat to look at longer time series). We focused on vegetation along canyons here, but the same methods can be applied on any location of interest, such as wetlands, habitats of endangered species, or to compare two areas under different management practices.

Mapping and monitoring vegetation communities distribution across the installation, their health, and how growing season is shifting in response to climate change is an important tool for natural resources management on base. If the focus of this chapter was on decrease in water availability through drought, as this is the main control on vegetation distribution and phenology in drylands, other controls, such as growing degree days or CO_2 concentration can be used and paired with additional controls on vegetation such as flow permanence. In further studies, a predictive model based on bioclimate envelopes could be developed (Pearson & Dawson, 2003). This comparison between vegetation and climatic controls could then be used to measure the consequences of climate change on key habitats used by endangered species, and how water availability in riparian forests could buffer habitats from change (for example here, the decrease in precipitation leading to delayed start of the growing season and lower vegetation greenness) (Seavy et al., 2009). We can then define thresholds that have been shown to trigger changes in vegetation distribution, density or phenology and compare with modelled climate change for the region to plan for the future and start taking proactive measures to buffer critical areas from change when possible, or adapt management expectations.

4.6 Conclusion

In conclusion, vegetation communities mapping showed how elevation controls vegetation distribution, and time series analysis revealed how drought and the following recovery can affect vegetation greenness and timing of the growing season in a dryland mountain range, a key ecosystem for conservation. We could start to understand how climate change could impact vegetation and the wildlife that depends on it. This work could be used for modelling of habitat and ecosystem shifts with climate change. This chapter highlights how we can create monitoring methods and data that can be useful to natural resources managers by answering specific conservation questions such as key vegetation communities distribution and sensitivity to climate variables.

Chapter 5
General discussion

5.1 Water availability and vegetation distribution in the upper San Pedro basin

This thesis is set in a topographically and ecologically diverse region of the Southwest USA. Through analysis of vegetation distribution and health, compared to water availability variations through space and time, I am able to apprehend the spatiotemporal organization of vegetation and water across the study site, how access to water govern vegetation resilience to drought, and the implication of these findings for natural resources management and conservation of the local biodiversity.

5.1.1 Organization of water and vegetation from mountains to river

The complex topography and geology of the upper San Pedro basin leads to strong variations of water distribution and availability for vegetation, which in turn induce a diversity of vegetation communities and ecosystems. In this thesis, I mapped the distribution of vegetation in the landscape, and showed that distribution is organized along gradients of water availability, which is in turn controlled by climate and geology.

In the Huachuca Mountains, vegetation communities are arranged along an elevation gradient, from semi-arid grassland to evergreen oaks at mid-elevation and dense pine-oak forests at the summit, as has been observed in

Chapter 4 and by Brady & Bonham (1976) or H. G. Shaw (1999). Upland vegetation uses soil moisture stored in bedrock fissures (Chapter 2). At low elevation, health and phenology are mainly controlled by water availability (Chapter 4). At high elevation, vegetation communities receive more water and temperature is a strong control on their phenology (Crimmins et al., 2010). In the canyons, non-perennial streams display high variability of flow, both through time and space, and are comprised of a succession of perennial, ephemeral and intermittent reaches (Chapter 3). Rainfall and geology are important controls on local flow permanence in each reach (Chapter 3), (Goodrich et al., 2018; Larned et al., 2011; Lovill et al., 2018). Ephemeral reaches only flow during rain events, while others are fed by springs (Gallo et al., 2020b). Cracks and fissures in weathered bedrock and faults carry water downstream, until impervious layers form an obstacle and water is brought to the surface in springs (Brown et al., 1966). In headwaters ephemeral streams, vegetation is often similar to upland communities (Chapter 4) (J. R. Shaw & Cooper, 2008), though in small wetlands, where a flatter topography and a perennial spring allow the formation of refuge pools, specific riparian vegetation and species can be found (Katz et al., 2012; Levick et al., 2008; Stromberg et al., 2015).

Grassland and scrubland cover the flat valley on top of permeable basin fill and have only access to shallow soil moisture derived from rainfall, as the regional water is too deep to be accessible (Chapter 2). This vegetation is sparse and highly sensitive to shifts in rainfall distribution, which controls their health and the length of the growing season (Chapter 4). They are characterized by a bimodal growing season, following the bimodal rain distribution between the winter rains and the monsoon. Ephemeral washes in this valley collect rainfall and can briefly flow during strong storms (Chapter 3). These washes are lined with sparse mesquite trees, a riparian species with deep roots adapted to high variability in flow and soil moisture. The same mesquite, this time as a dense forest, is also found on the former floodplain of the San Pedro River where the alluvial water table is deep but remains accessible (Stromberg & Tellman, 2012).

In the current, entrenched San Pedro River, flow permanence is also controlled by the underlying geology. A shallow water table on top of impervious silt-clay layer or bedrock maintains perennial flow, and intermittent reaches are located on top of permeable sand and gravel (Chapter 2) (Blakemore, 2006; Stromberg & Tellman, 2012). Groundwater levels follow the seasonal cycles of rainfall (peak during the monsoon and lowest levels during the dry season) with strong variations in dry reaches and more stable levels in wet reaches (Chapter 2). The streamflow-groundwater interactions provide moisture for riparian vegetation that is not found anywhere else in this landscape (Chapter 2). Density and composition of the San Pedro riparian corridor is organized along the gradients of flow permanence (Chapter 2). Perennial and intermittent reaches are occupied by dense corridors of cottonwood, with willow and a high diversity of grasses and wetland vegetation in the understory (Stromberg et al., 2008). These cottonwoods appear to have recently reached a stable density. In the drier reaches, cottonwoods are sparse but seem to have maintained a stable canopy cover for a longer time, and seem to be older individuals (Chapter 2). The main vegetation along these dry reaches are small willows, baccharis and the invasive saltcedar.

Over the region, the intense and localized monsoon storms are likely to initiate floods in the canyons and the San Pedro River, providing important moisture in the floodplain and essential for some riparian species, such as cottonwood, to sprout (Stromberg & Tellman, 2012). The monsoon also generates overland flow in ephemeral streams of the valley and bring soil moisture to the mesquite corridors. The less intense but longer lived winter precipitation and melting snow, paired with lower temperatures, has greater potential to infiltrate into the ground and recharge the springs in the mountains and the regional water table feeding the San Pedro River (Stromberg et al., 2015). This organization of more perennial (permanent springs) and intermittent streams (intermittent springs) in the mountain, and ephemeral washes in the valley is characteristic of mountain front recharge systems, where water from the mountain travels underground in the valley, here this water goes to the San Pedro River. This whole system is linked and impacts on regional water table recharge at the base of the Huachuca Mountains and in the valley affect flow in the San Pedro, which makes managers such as Fort Huachuca responsible for water management at the regional scale.

As a whole, the Huachuca Mountains and the San Pedro valley are a good representation of a diverse dryland landscape. This study site covers a wide range of typical dryland ecosystems, including mountain ephemeral streams along a strong elevation gradient, grassland washes, and a valley river with a typical riparian gallery-forest along variations in flow permanence. Each of these environments can be found in other drylands and the findings of this study contribute to the global understanding of vegetation health, phenology and distribution in these water-limited systems. Moreover, my work’s goal was to provide a exhaustive understanding of landscape-scale processes, how water travels from mountain peak to valley water table and the influence of its path on vegetation variations. Findings on underlying controls on flow, and water availability impact on vegetation distribution and phenology are in line with previous findings (Chapter 1) and come strengthen our overall understanding of streams and riparian vegetation in drylands.

5.1.2 Resilience to drought and climate change

Knowing ecosystems and species resilience to climate change and drought is essential for natural resources managers if they want to adapt their practices and methods to ongoing wide-scale changes (DoD, 2021). Climate projections for the Southwest USA trend toward an increase in temperature and a shift in precipitation distribution toward fewer but more intense events, which could lead to an increase in aridity, longer droughts and stronger floods (Ajami et al., 2012; Levick et al., 2015). These shifts in climate could be the source of lower recharge rates for the mountain and valley aquifers, decrease in flow permanence on the San Pedro, and extension of ephemeral reaches in the canyons of the Huachuca Mountains (Ajami et al., 2012; Gungle et al., 2019). A shift in rainfall distribution, timing and intensity toward longer droughts between stronger precipitation events might change the distribution of seasonal water patterns across the landscape and create ecological shifts for communities depending on specific flow regimes (Bogan, Gutierrez-Ruacho, et al., 2013; Jaeger et al., 2014; Stromberg, 2013; Williams et al., 2022). Wildlife and vegetation that need water at specific times of life cycle might see their range shift or their distribution shrink.

In the Huachuca Mountains uplands, the main risk of climate change is the loss of summit pine-oak forest due to the increase in temperature and the expansion of grassland (Reidmiller et al., 2018). High elevation species and communities are less resilient to drought (Poulos et al., 2007) and there might be a risk of massif die-off due to prolonged drought, as has happened elsewhere in the southwest (C. D. Allen et al., 2010; Breshears et al., 2005). In the canyons, habitat resilience will likely depend on flow permanence. Perennial reaches fed by spring might be more resilient to short-term drought thanks to storage in the aquifer, while ephemeral and intermittent reaches will more likely follow shifts in rainfall (Chapter 3). Weak winter rains can lead to an early dry-up of intermittent reaches that usually flow until spring (Chapter 3), but vegetation along ephemeral reaches is also adapted to variations in rainfall and more resilient to drought. Riparian vegetation in wetlands relies on permanent water availability along perennial reaches and is sensitive to springs drying up, which might happen with severe enough changes in rainfall and temperature (Van Loon, 2015). Between the expansion of semi-arid grassland in the mountains and the potential dry-up of ephemeral and intermittent streams, perennial reaches fed by springs, which might be better buffered from drought, might become even more critical as refuges for drought-sensitive species (Gallo et al., 2020b; Stromberg et al., 2015).

The valley grassland is more sensitive to short term changes in precipitation distribution as it is fully dependent on rainfall. Drought leads to a reduced greenness and a shorter growing season (Chapter 4), but it is also the community with the fastest recovery following a drought and the most adapted to high variations in water availability due to natural cycles of climate variability of the region (Bogan et al., 2015). As such, grassland might be more resilient to permanent change. Mesquite trees along valley washes are also adapted to high variability of water availability and highly ephemeral streams, so they are likely resilient to drought, and their role as dense and structurally complex corridors might remain.

Along the San Pedro River, the effect of drying up of perennial reaches is visible in current intermittent streams. The cottonwood-willow gallery forest is vulnerable to changes in groundwater table depth, which will also impact flow permanence and the extent of refuge perennial pools. In case of drought, the water table in intermittent reaches drops down and can no longer support dense cottonwood corridors (Chapter 2). Old trees with roots deep enough might remain, but young cottonwood might not have roots deep enough to access the water table during the dry season, which is when the water table is the deepest but also when the growing season starts (Chapter 2). In current intermittent and dry reaches, reduced water storage and moisture availability might make them even less favourable for riparian vegetation, though cycles of strong monsoon and winter rains might be enough to maintain the older cottonwood trees (Chapter 2). Loss of cottonwood and spread of saltcedar is a potential consequence of prolonged drought and overall increase in aridity (Chapter 2) (Stromberg, 1998) During droughts, the impervious geological layers under the wet reaches can support the shallow water table and buffer riparian vegetation, creating an even stronger disparity in water availability between these wet reaches and the drier areas, where the streamflow is low or zero and water table drops substantially (Chapter 2). Species dependent on perennial water in the San Pedro, such as Huachuca water umbel (*Lilaeopsis schaffneriana ssp. recurva*) or Arizona eryngo (*Eryngium sparganophyllum*), are the most at risk.

More generally, climate change is an important risk for natural resources and wildlife conservation as it will lead to habitat loss for many species and offsets between life cycles (Chapter 4) (Coe et al., 2012; Kellermann & van Riper, 2015). If rainfall is focused on stronger events, the amplitude of floods will increase, which might lead to shifts in vegetation species more resistant to scour (Stromberg et al., 2015). The combination of severe drought and floods could lead to geomorphic changes, including channel widening or river entrenchment. Long-term droughts could cause permanent transitions from perennial to intermittent flow and lead to a loss of connectivity, with shorter and more isolated perennial reaches both in the San Pedro River and in the canyons (Bogan et al., 2015; Jaeger & Olden, 2012; Seager et al., 2013). As aridity increases in the Southwest USA, non-perennial streams and their riparian habitats will become increasingly important for maintaining regional diversity. Species are not equal in the face of increased ephemerality. Some species are able to disperse easily and move between refuge pools and are more resilient to change (Yellow-billed cuckoo for example), while weak dispersers dependent on perennial reaches are not likely to survive dry-up of their wetland (Bogan et al., 2015). As habitat isolation increases along the San Pedro River or in the canyons of the Huachuca Mountains, the potential for community recovery from extreme drought decreases. Higher temperatures also increase the risk and frequency of lethal dehydration during heat waves for many species that will rely on the remaining perennial reaches and pools during the dry season (Albright et al., 2017). Groundwater pumping can also exacerbate drought risks and urban development increases the fragmentation of mesquite corridors in the valley.

5.1.3 Monitoring and management

Biodiversity conservation in the Southwest USA depends upon the protection of a diversity of habitats and vegetation communities, each with their water availability needs. Non-perennial streams and riparian corridors form a rare ecosystem, but host the highest diversity of species, including many endangered and federally protected species (Krueper, 1993; Stromberg et al., 2017). Their dependence on open water and shallow groundwater make them particularly sensitive to higher temperatures and shifts in rainfall distribution. My work gives an overview of the distribution of water and key habitats for threatened and endangered (T&E) species in the landscape, and of the main controls on this distribution is part of the monitoring and understanding necessary for efficient management and conservation.

The monitoring methods and tools used in this thesis can help map ecosystems distribution and measure ecosystem condition and health. Perennial reaches in the canyons and along the San Pedro are important refuges for wildlife and are the only open water in the landscape during the dry season. Monitoring flow permanence along intermittent streams of the Huachuca Mountains allowed me to map spatiotemporal flow across the landscape and identify these perennial reaches (Chapter 3). Seasonal classification of each reach provides information on habitat condition (from perennial to fully dry stream reach), and by recording flow permanence for several years, it is possible to keep track of resilience to stressors (such as drought or military training) or reaction to management actions. Mapping and monitoring flow permanence in streams is the first step to maintain this mosaic of habitats critical for T&E species, flood protection and good ecosystem function. Similarly, remote sensing was efficient to map vegetation communities distribution across the landscape and, by measuring vegetation greenness, to assess vegetation health and ecosystem condition (Chapters 2 & 4). As both conductivity sensors and satellite imagery analysis record time series of spatially explicit data, they provide information on how habitat conditions change depending on location and through time. They can also be used to identify controls on the measured variables. I was able to demonstrate that rainfall and geology are strong controls on flow permanence, water availability and vegetation health, and that establish that vegetation communities of the area show a stable health by looking at trends of vegetation greenness.

Mapping flow permanence and vegetation distribution and health can be used as a proxy to map potential habitat for species of interest. The methods used can help identify areas of dense deciduous riparian trees along perennial reaches, which are key refuge areas during the dry season or prolonged drought, and help prioritize conservation actions (Figure 5.1) (Powell & Steidl, 2015). Perennial reaches mapped based on conductivity sensor data might be used by T&E species such as the Huachuca water umbel or the Chiricahua leopard frog (*Lithobates chiricahuensis*). Deciduous trees identified by classification of satellite imagery are potential habitat for riparian birds such as the Yellow-Billed Cuckoo (*Coccyzus americanus*) or southwestern willow flycatcher (*Empidonax traillii extimus*). Arizona sycamore (*Platanus wrightii*), the dominant species in these deciduous tree patches, is a keystone species for local biodiversity (Stromberg, 2001a). Riparian areas are also a critical habitat for jaguars (*Panthera onca*), which need water and cover. Hatten et al. (2003) found that perennial and intermittent water sources within 20 km were considered important to dispersing jaguars. Streambeds are used as travel corridors, and the denser riparian vegetation host a denser prey abundance. In the valley, identification of mesquite corridors is easy with remote sensing due to the strong difference in vegetation greenness with the surrounding grassland. Since mesquite form important wildlife corridors, distribution mapping can be a tool to measure and control connectivity between important areas, for example between the mountain canyons and the riparian corridor of the San Pedro River (Figure 5.2). Managers can use the maps of critical habitats to adjust high-impact training away from critical areas and plan more precise future surveys for T&E species (Levick et al., 2015). Once potential habitats for T&E species have been identified, surveys can be conducted to confirm the presence of the targeted species and the methods presented in this thesis can help keep track of potential habitat extent and condition. Mapping critical habitat can also be a first step before additional monitoring, to make sure that this monitoring is as efficient and precise as possible; For example, conductivity sensors can be preferentially set in areas of the Huachuca Mountains canyons with riparian deciduous trees (Figure 5.2).

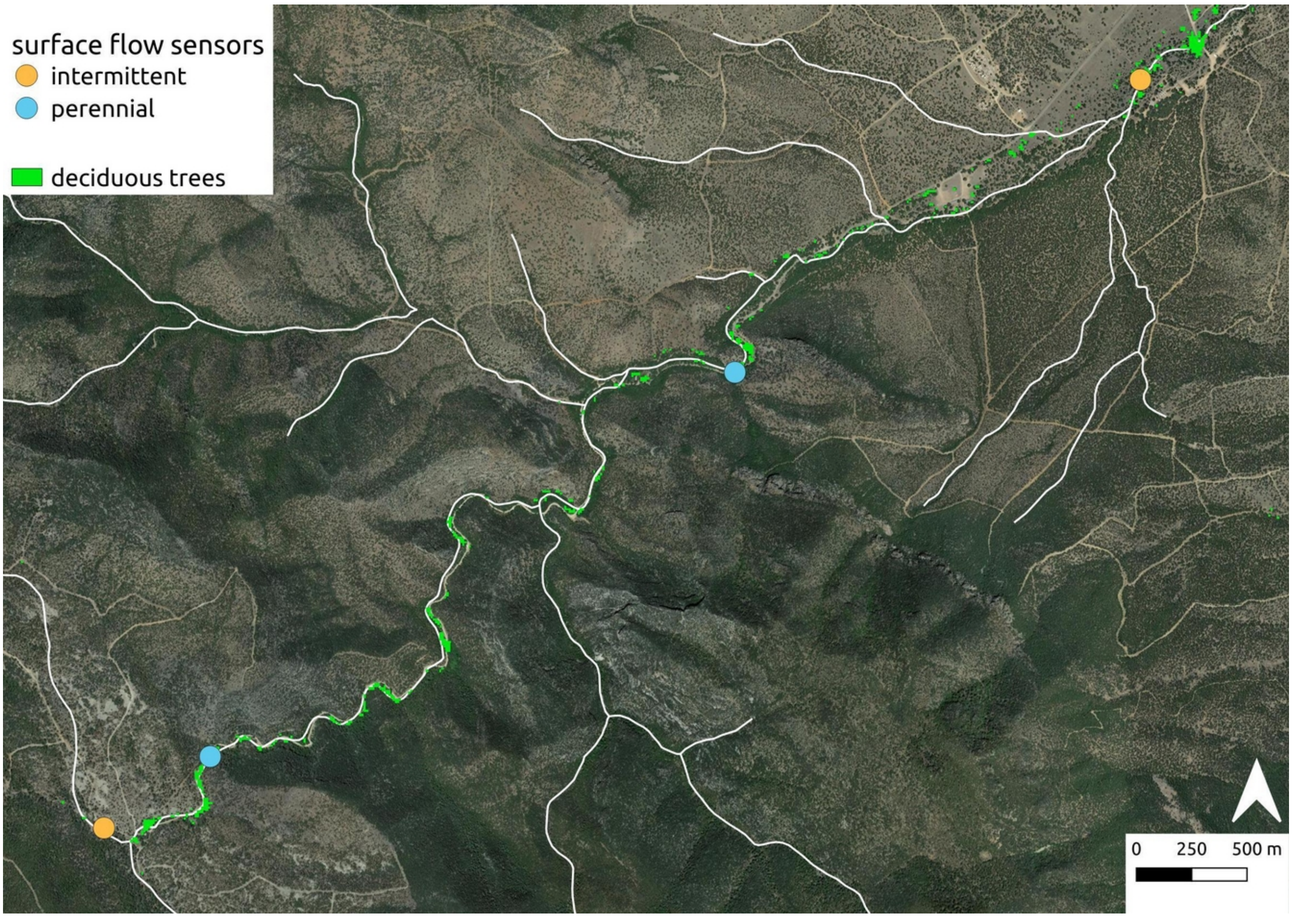

Figure 5.1: Distribution of deciduous trees along Garden Canyon and flow permanence sensors. Putting both information together allows us to find the location of deciduous trees close to perennial reaches, and the locations where there are deciduous trees but no information on flow permanence.

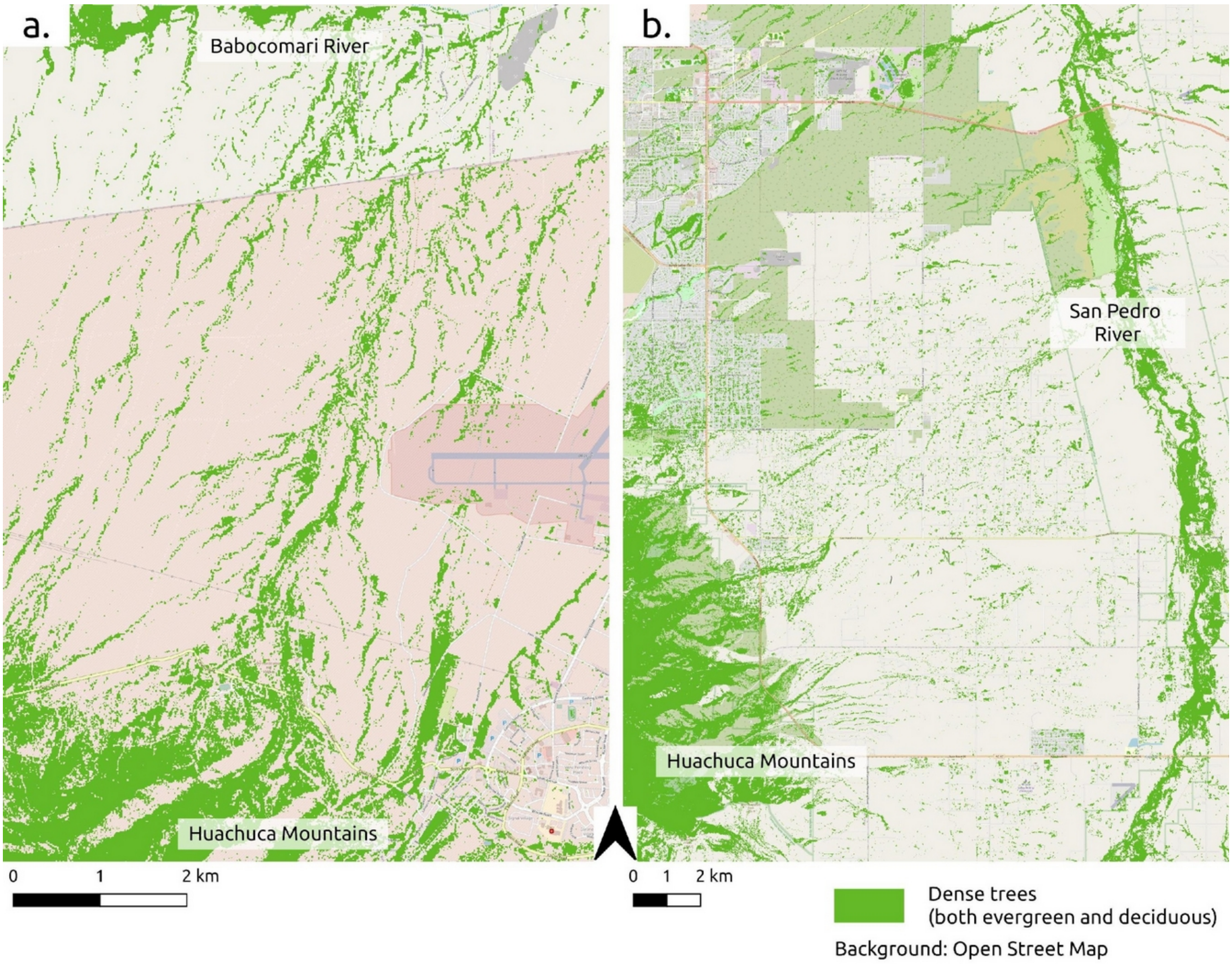

Figure 5.2: Dense vegetation distribution in the valley between the Huachuca Mountains and the lowland rivers (San Pedro River and Babocomari River). The map shows the good connectivity of the mesquite corridors between the mountains and the Babocomari River (a) and the fragmented patches of isolated mesquite between the mountains and the San Pedro River, mainly due to urban development (b).

In the context of climate change, flow and vegetation monitoring is an important means to keep track of impacts on critical habitats. Regular vegetation communities maps over the area can help follow the potential decrease in pine-oak forests, loss of riparian trees and expansion of semi-arid grassland. Fort Huachuca has implemented fuel and fire management practices with the goal of maintaining high elevation species on the short-term, and biodiversity by limiting competition and avoiding widespread fires that might speed up the conversion of evergreen forests to semi-arid grasslands (2021).

The loss of dense forests will also have an impact on flow in the canyons by altered surface runoff and sediment transport. Conductivity sensors in the known perennial pools and close to springs will record variations in flow permanence and inform managers on habitat resilience to climate change. As temperatures increase, accurate and systematic mapping of refuge pools and dense riparian vegetation will be even more important than today, as more species will become at risk of lethal dehydration and will need cool and moist habitats to survive the dry season (Albright et al., 2017; Stromberg et al., 2015). The need to ensure the availability and conservation of these habitats will only grow as climate changes. Knowing which changing environmental parameters control habitat integrity, can help managers take actions to work toward enhancing climate change resilience, buffering ecosystems from change or anticipating and facilitating ecological transitions (Coe et al., 2012).

5.2 Tools for continuous monitoring of critical habitats

One goal of this thesis is to develop and test potential tools and methods for continuous monitoring of ecosystems through vegetation communities distribution and health. In this section, I present remote sensing and flow permanence metrics that could be useful for natural resources managers, as well as implementation and improvements suggestions. While my study was mainly focused on a military installation, the methods presented here can be applied to most natural resources management settings.

5.2.1 Need and applications

Ecological monitoring is essential for adaptive management at the landscape scale (Ripley et al., 2021). In this thesis, I worked with various methods and tools that allow me to keep track of flow permanence, as well as vegetation distribution and health, keeping in mind that they needed to be relatively simple to understand, easy to implement and reliable. Being able to monitor habitats and species of interest, such as the collection and analysis of repeated observations, or measurements, to evaluate changes in condition, is essential for natural resources managers to keep track of habitat health, assess vulnerability, find potential habitats and spot trends (Ripley et al., 2021). Ecosystem monitoring is driven by objectives. In this project, the management goal is focused on critical habitats for T&E species, and their vulnerability to drought, but these methods can be applied to any management targets centered on land cover, vegetation and streamflow.

By monitoring and surveying the main elements structuring ecosystems, such as water availability and vegetation distribution, managers can map potential critical habitats for T&E species (Kerr & Ostrovsky, 2003). This can help target defined locations for fieldwork and management actions, identify favourable areas that may have been overlooked (Levick et al., 2015). Being able to map locations of individuals, habitat patches, nest sites, or other indication of the species’ presence is a critical tool for understanding potential conflict with other activities in the area, such as training and target practice, as well as for identifying management strategies (Levick et al., 2015; Ripley et al., 2021). Similarly, risks and threats can be assessed and monitored through time.

Monitoring is also necessary to keep track of population health and ecosystem conditions. It is an important tool to identify problems early, when cost-effective actions can still be taken and before population collapse. Surveys can be conducted before, during and after programs and interventions to monitor changes, measure the efficiency of conservation actions and adapt practices if necessary (Nadeau & Rains, 2007; Ripley et al., 2021). Likewise, they can help measure detrimental impacts of stressors, such as military training or urban development, and be used to support science-driven plans and objectives when talking to decision-makers, funding bodies and other users and stakeholders (Levick et al., 2015; Ripley et al., 2021). Monitoring metrics can also be compared to potential controls, such as rainfall, temperature or disturbance to better understand the causes of change, or what external factors need to be maintained or restored to reach conservation goals (Ripley).

To follow and predict the impacts of climate change on natural resources, habitats and species, managers need to be able to monitor rates of change with methods providing continuous measurements and indicators of changing climatic conditions (DoD, 2021; Pastick et al., 2018). Knowing the direction of change helps inform adaptations in management practices, and implementation of new actions, for example to avoid federal listing of a species sensitive to climate change. Knowing the current environmental limiting factors on habitat and species distribution (such as flow permanence) enables managers to understand how changing conditions can amplify existing stressors and threats to the installation’s T&E species (Stein et al., 2019).

Ecosystem monitoring needs to be cost and time-effective, due to limited funding and the tools used needs to cover wide areas, to limit intensive fieldwork due to time and effort spent in accessing areas in large installations (Pastick et al., 2018). For efficient long term and large-scale monitoring, natural resources managers need methods and tools that are repeatable and comparable through time and across space, and easy to interpret (Nadeau & Rains, 2007). They have to be able to provide repeated measurements of metrics and variables linked to specific management goals. The set of tools used must also cover a range of spatial and temporal scales. Remote sensing and conductivity sensors are two methods that can provide spatially and temporally explicit information on water distribution, land cover distribution and ecosystem condition.

These tools and methods can be used to develop key metrics related to the ecosystem or to the stressors that affect its integrity (Ripley et al., 2021; Stein et al., 2019). These metrics are key tools of monitoring: they give information on ecosystem state and health and can be compared regularly to track change. They can be part of the regular monitoring of T&E species habitats, or used before and after a disturbance (such as measuring vegetation greenness after a fire to track recovery) or management actions (monitoring groundwater depth after recharge ponds have been built for example). When compared to climate variables, such as rainfall or temperature, they can be used as vulnerability indices, taking into account sensitivity and exposure to climate change (Coe et al., 2012). Once the relationship between controls and habitat condition is determined, it is possible to establish threshold values (such as minimal amount of rain necessary in a day to trigger flow in a stream, or minimal monsoon to trigger grassland greenup and prevent early senescence). The threshold being reached is a warning sign and, in case where conservation actions can be taken, we know when to take them. For example, if we know the rainfall-temperature threshold necessary to maintain water in refuge pools, rainfall going below this threshold is likely to dry up pools, and can lead to action being taken, such as amphibian rescue, artificial recharge or set up of artificial water troughs.

5.2.2 Remote sensing

Remote sensing using satellite imagery provides systematic coverage over most of the world, with repeated observations at set intervals, and comparable information. In this project, I tested how free data from public-owned satellites (Landsat and Sentinel-2) could provide monitoring metrics for land resources managers. The data is freely available to anyone and can be accessed online. For Landsat, ready to use pre-processed vegetation indices such as Normalized Difference Vegetation Index (NDVI) are also available online (https://espa.cr.usgs.gov/). Landsat provides a long time series useful to look at historical trends, including changes in land cover distribution or vegetation health (Chapter 2). Landsat-9 was launched in 2021 and thus ensures the continuity of the dataset. Sentinel-2 is more recent, but provides high temporal and spatial resolution images useful for habitats covering limited extents and rapid changes (Chapter 4). This study showed that NDVI is effective to measure trends of vegetation greenness, map vegetation communities, and reconstruct phenological cycles.

Several metrics of ecosystem condition and vegetation health can be derived from NDVI times series. The first step is to pick stationary, representative sampling points over each habitat of interest. These points can serve as a representative subset of the habitat and their continuous monitoring can inform on their condition and send early warnings of habitat loss or degradation. They can be used to keep track of vegetation evolution through the years (Chapter 2) or as training and validation for supervised classification (Chapter 4). Sampling points can be located in areas easier to access but must be representative of the habitat or ecosystem distribution over its geographical and climatic range. Another strength of using satellite imagery is that a metric, such as maximum vegetation greenness can be measured for each pixel over the whole image, which is essential to spot spatial trends or specific areas of interest.

Time series of NDVI over the last 10 years, or over the whole time series available, can be used to measure the rate of change and show decrease (or increase) in vegetation greenness (Table 5.1). This metric can help identify a slow decrease which can be the sign of vegetation dying-off or being replaced by other species, or abrupt drops in vegetation greenness after a disturbance and the following recovery or permanent shift in community (after a fire for example) (Chapter 2). The time range used when looking at time series and trends must be adapted to answer the question of interest (long time series for long-terms and slow changes, or short time-series for abrupt modification of the environment) and adapted to the local climatic and phenological cycles. With Landsat long time series, a baseline can also be chosen (for example the year before a fire, or before a management practice is implemented) and compared against a second more recent date (Table 5.1). The result of rates of change can be represented as plotted values from sampling points or computed as a pixel-wise comparison over the whole area of interest on a map (Chapter 2). Mapping NDVI values anomaly (comparing NDVI for one year against a long-term median) during a drought or after a disturbance (fire, logging, military training) can help map areas that are more sensitive to stressors, or recover more slowly (Table 5.1). Anomaly maps can also be produced for any metric of interest, such as peak and trough NDVI, or timing and length of the growing season.

Time series of vegetation greenness at high temporal resolution, as can be obtained with Sentinel-2, can be used to reconstruct phenology cycles of vegetation communities (Chapter 4). Key phenological metrics are the start, end and length of the growing season. They can be computed in various ways depending on the goals and specific cycles of the local vegetation. Due to the bimodal cycle of vegetation under the influence of the North American Monsoon, I decided to use a simple method based on the dates of minimal and maximal NDVI. This method has the advantage of not confusing the dip in vegetation greenness during the dry season for the end of the growing season, and is easy to compute and implement. Changes in phenology have major implications for species population dynamics and ecosystem functioning (Crimmins et al., 2010). Length and timing of the growing season can be computed every year and compared to cycles of T&E species (arrival of migratory birds for example) relying on vegetation for food or cover to identify potential mismatches and offsets that might become more marked as climate changes (Table 5.1). These metrics can be compared between years and against climatic controls (Chapter 4).

Remote sensing can also be used to map land cover for an installation. Supervised classification, like the one presented in Chapter 4, can be run regularly to keep track of land cover changes, such as urban development or semi-arid grassland expansion. The same sampling points used for time series can be used as training points for the classification. Land cover of each point is confirmed with field work and the same classification is run, with images taken at the same time of year, every time the management plan is updated. These classifications can also be run on images from previous years, as long as there is a way to confirm the land cover of training points for these years, for example by using very high resolution aerial photography. Maps for different years can then be compared to map land cover changes and transitions (Table 5.1). While national land cover maps are completed by the USGS every few years, it can be worth it for installations to develop their own maps which can have a higher spatial resolution (10 m with Sentinel-2 images for example) and adapt the land cover classes to the installation's needs. Specific metrics derived from land cover maps includes the extent of each vegetation community, and extent of change when comparing two land cover maps from different years (in % of an area of interest, such as the total installation, or a river's floodplain).

Precise metrics measuring health, extent and life cycle timing of vegetation communities are important to investigate potential reasons of change or decline in these areas and target restoration activities, or assess the success of previous management actions (Goetz, 2006). These methods can be applied anywhere, and adapted to available data (for example using radar, which is able to see through clouds, or LiDAR to map vegetation height and forest structure). I focused this work on riparian vegetation, but these methods can be applied on any location of interest, or specific vegetation communities, as long as they can be differentiated from their surroundings by difference in vegetation density, leaf or flower colour or phenology timing.

Table 5.1: Main metrics for monitoring of vegetation greenness and distribution. A 10 years trend is used as an example, but the time range used should be adapted to the question that needs to be answered and the local climatic and vegetation growth cycles to provide meaningful information.

Metric	**Data used**	**Information provided**
NDVI trend – 10 years	Annual average NDVI values over the end of the growing season	Continuous times series of vegetation health
NDVI anomaly	NDVI of current date minus NDVI for long-term average	Above-average increase or decrease in vegetation greenness (due to fire for example)
NDVI 2 dates comparison	NDVI for most recent date minus NDVI for older date	Decrease or increase in vegetation greenness targeted between two specific dates.
Start of growing season	Date when minimum NDVI is reached	Phenology of the vegetation community, can be compared between years, and to timing of wildlife phenology
End of growing season	Date when maximum NDVI is reached	
Length of growing season	Number of days between minimum and maximum NDVI	
Current land cover	Supervised classification	Distribution of main vegetation communities and urban development
Land cover change	Difference between two land cover classifications	Extent and location of land cover changes between two dates

5.2.3 Conductivity sensors

Conductivity sensors can be used as a monitoring network to map flow permanence along a stream and across a watershed (Chapter 2). While ready-to-use conductivity data loggers are expensive, the method used here relies on adapted temperature sensors. Detailed information on fabrication and calibration of these sensors have been given in (Blasch et al., 2002; Chapin et al., 2014). They are cheaper than conductivity-specific data loggers, small, waterproof, easy to set up and can record data at high temporal resolution for several years. Low prices mean that they can be set up extensively and get data at high spatial resolution. Several sensors along a stream can record downward moving wetted fronts and dry-up patterns. If one sensor fails, an extensive network means there is still usable data available. Once properly calibrated, they provide precise information about flow state in the streambed. The dataset provides information on sediment moisture and surface flow, but also on which state is reached when, and how often. Validation can be done through calibration, fieldwork and citizen science projects (for example hikers and birders sharing pictures of a designated area of the stream). Recently, ready-to-use sensors have been made available (for example the HOBO TidbiT MX Temperature 400' Data Logger), which record water presence by measuring conductivity. Conductivity sensors can also be paired with cameras in key areas. These cameras can confirm if the water recorded by the conductivity sensor is due to flow or pools, and give information on water levels (Noto et al., 2022) and record use of refuge pools by wildlife.

Conductivity sensors can be used to measure key metrics of flow permanence, such as the number of dry days in a year or during a specific season, and the average length and frequency of dry periods (Table 5.1). Sensors can be set regularly along streams to get a global understanding of spatiotemporal flow patterns across a whole installation or watershed (Table 5.1). In case critical areas, such as springs and perennial pools, have been identified, sensors can be set preferentially in these areas to keep track of water presence and sensitivity to drought. Riparian and semi-aquatic species in the study region are considered highly vulnerable to climate change (Bagne & Finch, 2013), so recording the flow condition in streams for several years can be a useful tool for detecting areas that are particularly sensitive to variations in rainfall and/or moisture, and measuring the impact of perturbations (climate change or military activities for example) (Levick et al., 2015). The information gathered can be shared with partners working on flow modelling in non-perennial streams at bigger scales (as done by Levick et al., 2015).

Information on flow permanence can be used to sort similar stream reaches into classes (Table 5.1). The choice of classes depends on the management goals and is organized around temporal distribution of flow, sometimes paired with additional information on vegetation structure, soil type or topography. The seasonal classification presented here can be complemented with information gathered during fieldwork, including vegetation community and stream morphology to refine potential habitat mapping for T&E species. This classification provides a method of identifying stream reaches with similar characteristics, and can be used to plan sampling schemes or surveys for wildlife and land use management (Levick et al., 2018).

Sensors can be set in key areas of conservation interest, such as cottonwood corridors along the San Pedro River, deciduous trees along the Huachuca Mountains canyons or perennial pools. For T&E species, the number of dry days and seasonal distribution of flow are key metrics as different species need different spatiotemporal water distribution. Conductivity sensors can be used to map potential habitats, especially if flow permanence data are paired with information on land cover and vegetation structure. Being able to map the distribution of flow across the landscape could also be used to highlight potential wet corridors for allowing species dependent on sediment moisture and surface water to travel between favourable habitats and breeding locations (Costigan et al., 2016; Jaeger & Olden, 2012). For species whose habitat requirements are not well known, flow permanence data, vegetation community mapping and population surveys for these species can help improve the understanding of their habitat and management needs (Levick et al., 2015).

By providing a continuous monitoring of flow distribution, conductivity sensors can be an important tool to assess the resilience of stream reaches and the populations that depend on them to climate change. Knowing the locations, current condition and sensitivity of critical habitats such as perennial reaches and pools is an important tool to adapt location, type, and intensity of military use or management actions (Ripley et al., 2021). As with vegetation health and distribution, understanding the links between flow permanence and climatic controls such as rainfall allow managers to predict the potential changes in flow distribution by monitoring local rainfall and temperature. When compared with data on rainfall and drought (as measured by the drought monitor for example), it is possible to establish the resilience of flow permanence in these areas, and identify threshold of rainfall, or drought level, where these pools might dry up. Then, if critical climatic conditions are predicted or observed, natural resources managers can know which pools and stream reaches are at risk. If these areas host T&E species with low dispersion capabilities or if these species need moisture and water during the predicted drought (fish and amphibians for example), actions can be taken to safeguard these species, by bringing additional water or displacing the population to safer areas, when possible.

Table 5.2: Main metrics for spatiotemporal flow distribution in non-perennial streams.

Metric	**Data used**	**Information provided**
Number of dry days in a year	Daily flow permanence from conductivity sensors or other flow detection methods	Ephemerality at the stream reach scale: how often is the stream reach dry, and for how long
Number and average length of dry periods		
Seasonal classification	Daily flow permanence sorted by season	Temporal distribution of flow, which can be compared to needs of species of interest
Connectivity	Flow detection by successive sensors along a stream	Timing and extent of connected stream reaches
Reactivity to rainfall	Daily rainfall from local gauges and daily flow detection	How dependent a stream reach is on rainfall and which threshold of rainfall intensity leads to onset of flow

5.2.4 Implementation

Monitoring can be implemented in a variety of ways on military installations. Surveys can be led by external partners such as contractors, federal or state natural resources management organizations or universities, or directly by the natural resource management personnel on the installation. This thesis is part of a more global project focused on water availability to riparian vegetation on military installations of the Southwest US, and follows other academic projects focused on vegetation and water management on Fort Huachuca, including Levick et al. (2015) and Stromberg et al. (2015). The goal of my work was to provide a better understanding of vegetation and water distribution across a diverse landscape, but also to present metrics that can be included in monitoring plans. In this section, I introduce ways to implement these metrics so installation personnel can use them in the long term.

Remote sensing and satellite imagery analysis requires trained specialists, especially for land cover classification, which needs expert knowledge of the land cover and vegetation communities in the area of interest. For regular monitoring of vegetation distribution and health across the installation, specialists can be hired on temporary contracts, as long as they follow the same method every time the analysis is done. If natural resources managers are interested in remote sensing and execute vegetation monitoring themselves, training is available, including the free NASA Applied Remote Sensing Training Program (ARSET) (https://appliedsciences.nasa.gov/what-we-do/capacity-building/arset) which provides online courses, from the fundamentals of remote sensing to the methods presented in this thesis. ARSET includes training on free data sources, software, online analysis tools such as Google Earth Engine. Key metrics can also be delivered more globally for use by all stakeholders and land managers through online tools or publicly available scripts, though such tools need to be maintained and updated on the long term to be useful. Nation-wide examples include the National Land Cover Dataset, a land cover mapping effort by the USGS with land cover maps for 2001, 2006, 2011, 2016 and 2019 at 30 m of spatial resolution, and the land cover changes between these dates (https://www.usgs.gov/centers/eros/science/national-land-cover-database). Similarly, phenology metrics derived from NDVI, including start and end of the growing season, are computed for the conterminous USA every year (https://phenology.cr.usgs.gov/viewer/). More custom online tools could be developed where a location is selected and a list of metrics is available, including difference in NDVI values between two dates, long-term trends for a specific period and a set location, or pixel-wise maps of NDVI difference for a specific year against the long-term mean. The IPCC Interactive Atlas (https://interactive-atlas.ipcc.ch/) is an example of an online tool providing various metrics in map or graph form.

Flow permanence can be implemented directly by installation personnel with the help of key papers describing the methods in detail, such as Chapin et al. (2014) and Blasch et al. (2002), or through agencies and universities with expertise in this field. Proper calibration makes the link between conductivity values and flow state in the stream reach straightforward. Sensors can be set at the bottom of the streambed or in shallow sediment, and regular field visits are necessary to make sure the sensors have not been carried away or pushed out of the streambed. Locations of sensors also have to be carefully chosen, in a way that replies to management questions. For example, installation of sensors in pools during the dry season to follow the evolution of wet refuges, or sensors in riffles, where water does not form ponds, to follow wetting fronts down a stream after a rainfall event. Conductivity sensors can also be paired with flow-detection sensors, which are based on a flap that opens under water pressure, to distinguish between pools and flowing water. Geology, stream topography and observations can be used to develop online tools for modelling and predicting flow permanence. An existing example is the PROSPER tool, used to predict if and where streams in the Pacific Northwest will cease to flow in any given year (Jaeger et al., 2019). To provide observations for this tool, an online stream permanence reporter has been set up, and conductivity sensor networks could provide important observations for the potential development of a similar tool in the Southwest USA.

Technology transfer workshops and collaborations with conservation partners are key ways to implement new monitoring tools and methods. The National Military Fish & Wildlife Association hosts a training workshop every year and introductions to remote sensing, conductivity sensors and available free online tools and resources could be organized. Proper documentation has to be made available to ensure comparability of long-term, repeated monitoring in the form of accessible reports, step-by-step tutorials, and information on how to access more resources and trained people. Flow permanence and vegetation data collected in the field, paired with species presence, topography and climate can also be used by contractors or organizations to build models for identifying potential habitats of T&E species, or predicting how flow and vegetation will evolve with climate change (Garfin et al., 2017; Levick et al., 2015). The results can be highly informative for directing survey efforts, prioritizing conservation efforts or adapting management practices.

5.3 Study limitations and future research

Climate change in the Southwest USA is translating to changes in temperatures and precipitation. In this thesis, I focused on rainfall, as in this area, water availability is the main control on vegetation distribution. But temperature is also an important control, notably through evaporation. For a complete assessment of climatic controls on vegetation and flow permanence, I would work with both precipitation and evaporative demand, which gather information on temperature, wind and all climatic variables influencing evaporation, for example through a drought intensity metric, such as the ones used in the drought monitor. In addition to temperature, other stressors, including fire or invasive species, might accentuate, offset, or override climate effects and could also be added to the analysis to build a more complete understanding of environmental and climatic controls on water availability and vegetation distribution. Local rainfall gauges and temperature could provide more precise comparisons between these climatic controls and water availability.

I managed to map land cover over the study site as functional vegetation communities, but it could be possible to map more detailed communities dominated by a dominant species (for example separating oak forest from pine forest). A possible way of doing this would be to add the red edge bands of Sentinel-2, which cover the area of the electromagnetic spectrum between red and infrared, and can be used to differentiate between species. An even more precise classification could be obtained by using vegetation height and undergrowth structure provided by LiDAR data. This more precise map would help identify potential habitats for T&E species more precisely. Invasive species could also be mapped and monitored at the species level. For example, saltcedar (*Tamarix ramosissima*) has a unique spectral signature thanks to its pink flowers that can be visible on very high resolution imagery and aerial photography.

The methods used here could be applied to other military installations of the Southwest USA and adapted to each installation’s specific management needs to test their soundness and versatility. A more regional analysis of climatic and environmental controls on flow permanence and vegetation could be useful to establish threshold values that are shown to trigger critical changes in water availability and vegetation distribution, density or phenology. This understanding of relations between climate and flow or vegetation is also an important input for modelling, which can then be used for risk assessment and to predict potential changes as climate continues to shift and help installations and other land management entities adapt their management practices, to either prevent, slow down or adapt to change.

5.4 Final conclusions

In this thesis, I used remote sensing and conductivity sensors to gather information on spatiotemporal distribution of water and vegetation across a diverse landscape. While the case study presented in this thesis focuses on military lands, this work is meant to support natural resources management for a wide range of situations and ecological settings. I was able to map potential critical habitats, such perennial pools and deciduous riparian trees; measure and keep track of ecosystem condition, in the form of flow permanence in streams and vegetation health. Comparing key metrics of flow and vegetation greenness to geology and rainfall allowed me to establish climate change vulnerability of riparian ecosystems.

This case study of Fort Huachuca, in southern Arizona, provides a better understanding of water availability to vegetation in cottonwood-willow riparian corridors of semi-arid valleys and canyons of Sky Islands mountain ranges. It was found that grasslands are the most sensitive to variations in water availability, but are also used to strong variations and can recover quickly, while high elevation forests and riparian corridors are adapted to high water availability. These habitats are currently buffered from drought by high water tables and perennial flow supported by springs and impervious rock layers, making them critical refuges as climate changes. But as temperatures increase and rainfall distribution changes, even perennial stream reaches might dry up and groundwater levels decrease. These riparian forests, which are adapted to high and stable water availability making them the least resilient habitats to drought and with no way to move north or upland in an environment too dry for them, might then disappear, and with them the high biodiversity they support.

The tools and methods used in this study were chosen specifically for their accessibility and efficiency, covering wide areas with repeatable and comparable observations. I tested their ability to provide time series of spatially explicit key metrics of ecosystem condition and resilience to climate change. My goal was that the methods used in this thesis could be adapted to a wide array of landscapes and ecosystems, and be useful for natural resources managers to monitor their land and plan for future changes.

Chapter 6
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