IMPACTS OF MULTIPLE EXTREME DISTURBANCE EVENTS ON LANDSCAPE COVER: WILDFIRE AND FLOODING IN CANYON RIPARIAN ECOSYSTEMS OF THE JEMEZ MOUNTAINS, NEW MEXICO, USA

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ABSTRACT

Impact of Multiple Extreme Disturbance Events on Landscape Cover:
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Changes in regional climate and human land use in the Southwest United States have contributed to widespread tree mortality and provided the conditions for large-extent high-intensity fires to proliferate across the landscape. Increasingly extreme disturbance regimes initiated by widespread fires and high-volume flooding may cause irreversible changes to ecosystems, and lead to unsuitable conditions for long-standing forests to re-establish. On the land occupied by the Bandelier National Monument on the Pajarito Plateau of the Jemez Mountains in New Mexico, United States, three large-scale fires have occurred since 1996, resulting in intense flooding and sediment transport, and eliminating propagules and habitat connectivity of long-standing riparian forests. This study interpreted aerial photographs of the area of interest from 1991-1992, 2008, 2014, and 2018 to quantify and describe changes to the landscape and cover types prior to and following known disturbance events. In each canyon, four distinct segments were classified according to the types and degrees of disturbance and effects observed. Overall, dramatic decreases in forest cover following fires were observed and quantified. Forest cover in Frijoles Canyon decreased from over 70% in the early 1990s to under 20% in 2018, and in Capulin Canyon, from 57% to under 5% over the same time period. Understanding patterns of disturbance and vegetative re-establishment helps reveal opportunities and challenges for management directions to support the intrinsic resilience of the system, or intervene where necessary. If management goals seek to maintain critical ecosystem services such as water filtration, flood control, and carbon sequestration, in addition to preserving cultural and natural resources, comprehensive assessment of the changing landscape will be critical for determining current impacts and future trajectories following recent disturbance events.

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Acknowledgements

The land that is the focus of this study is the ancestral home of the Cohiti, San Felipe, San Ildefonso, Santa Clara, Santo Domingo, and Zuni Pueblo people. Several Indigenous communities and nations, including the San Juan Pueblo, Zia Pueblo, Hopi, and Navajo also assert traditional relationships with the area. My deepest thanks to the Indigenous people and their ancestors who have been the traditional stewards and inhabitants of the area that is now known as New Mexico in the American Southwest.

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Chapter One: Literature Review

Riparian ecosystems are characterized by regular disturbances, unique ecological processes, and heterogeneous composition, providing habitat to diverse aquatic and terrestrial plants and animals (Naiman, Decamps, et al., 2005). Increasingly intense and frequent disturbance events including wildfires and flooding are occurring in many riparian systems, and long-standing patterns of disturbance and recovery are derailed as some systems experience unprecedented stressors and permanently altered trajectories (Harmon et al., 1986). Riparian plants possess a range of specialized adaptations that allow them to be resistant and resilient to regular disturbances; however extraordinary disturbance events may stretch systems beyond their ability to resume functionality congruent to their pre-disturbance status (Allen, 2014). Categorizing stream segments based on disturbance can serve as an important first step to understanding local dynamics, and later serve as an aid to defining management goals and strategies for distinct areas (Quinn et al., 2001)

Riparian ecosystems

Riparian zones extend from the edges of water bodies into upland communities, and are characterized by the regular influence of fresh water, strong energy regimes, and biological and physical diversity (Naiman, Decamps, et al., 2005). They are home to unique environmental processes and communities of animal and plant assemblages, and connect ecological systems and physical processes from upland areas to the oceans (Naiman & Decamps, 1997). Due to their positioning as hydraulic conduits of biotic and abiotic elements and energy from their headwaters to mouths, rivers serve as “critical transition zones” connecting ecosystems together, with disturbances resulting in consequences at every level of interaction (Richardson et al., 2007).

Riparian ecosystems are often unique in their surroundings, integrating aquatic and terrestrial components of the landscape, and providing substantial habitat heterogeneity for native plants and animals (Naiman, Bechtold, et al., 2005). Riparian ecosystems are connected in three dimensions: laterally, longitudinally, and vertically, with hydrology influenced by both surface and groundwater, as well as local geology and topography (Naiman, Decamps, et al., 2005). Considering the fourth dimension of time adds complexity to understanding riparian ecological processes, as considerable changes in composition and structure from one time period to another can fundamentally alter the functionality and trajectory of the system (Reiners, 2005).

Disturbance in riparian systems

Understanding disturbance regimes in riparian ecosystems is of critical importance for maintaining ecosystem services and cultural resources in flood- and fire-prone zones (Naiman et al., 1998). While a wide range disturbances may occur naturally in many riparian systems, increased frequency of high-intensity and large-scale disruptions in areas not typically prone to such disturbances are presently producing historically unobserved and lasting consequences, where ecosystems may fail to recover to locally recognizable states (Odum et al., 1979; Rapport & Whitford, 1999; Sparks et al., 1990). Conceptual models of disturbance and recovery trajectories distinguish between different scales, intensity, and frequency of disturbance, with differing degrees of influence that shape the landscape and vegetation dynamics on seasonal, decadal, or larger scales (Brinson, 1990; Richardson et al., 2007). Such disturbances can include landslide, flood, wind, fire, drought, disease, litter accumulation, herbivory, drought, or other physical influences, and interact to exert short-term and lasting impacts on riparian landscapes and ecosystems (Naiman et al., 1998).

While rivers are subject to frequent and regular disturbances at multiple scales, especially extreme and successive disturbances can cause ecosystems and cover types to convert to historically unprecedented states (Didham et al., 2005). In most parts of the world, riparian vegetation is dominated by woody species. However, in places where climate is cold, hydrogeomorphology is waterlogged, or disturbances are too frequent for trees to establish, grass and shrublands dominate (Richardson et al., 2007). Shifts from forested ecosystems to grass- and shrub-dominated systems can have long-term consequences altering hydrologic regimes, biodiversity, and carbon sequestration, among other ecosystem and cultural services, as the landscape ceases to function as it did before (Allen, 2014; Hicke et al., 2012; Richardson et al., 2007).

Flood disturbance and fluvial have been shown to exert considerable influence on vegetation, and act as drivers of habitat patch dynamics in riparian systems, producing a mosaic of shifting habitat that is maintained in quasi-equilibrium by simultaneous cut and fill processes throughout the river channel (Mouw et al., 2009). However, in riparian systems that are also influenced by fire, fire has been shown to exert the strongest total effect on variability of floodplain habitat patch composition compared to stream power and geomorphic position (Kleindl et al., 2015). Regular seasonal or intermittent flooding and fire patterns may not be considered disturbances, since their occurrence facilitates habitat heterogeneity and species diversity; however when the extent and intensity of disturbances increase drastically, they can result in irreversible and fundamental shifts to community compositions and ecosystem functionality (Rapport & Whitford, 1999; Sparks et al., 1990).

In places such as the southwestern United States, climate and drought regimes that are unprecedented in at least the last 1000 years, in tandem with changing human land use and fire suppression, have provided the conditions for the most intense and large-scale fires of the historical era to occur (Allen, 2014). Because changing disturbance regimes have been relatively recent and extreme on the scale of forest development, the impacts and existence of *legacies*, persistent effects of past events, are poorly understood (Johnstone et al., 2016). Additionally, stabilizing interactions and resistance of mature vegetation to minor disturbances in long-standing intact forests can obscure reduced resilience of a system until more intense disturbance occurs (Ghazoul et al., 2015). A mature forest may not display signs of acute distress until a single fire or pest outbreak causes uncharacteristically major damage from which the ecosystem is unable to recover.

In many cases, a single disturbance may not be enough to initiate regime shift, but *linked disturbances*, such as the effects of wildfire after increased tree mortality following a bark-beetle outbreak, can be exponentially severe (Simard et al., 2011). Disturbances at various time scales and intensities select for specific life-history traits that are important to consider when assessing current ecosystem health and future trajectories. While regular flooding and fire regimes are normal in most river systems, more frequent larger-scale and higher-intensity disturbances that cause high mortality can reduce or eliminate the possibility that long-established communities are able to repopulate their historic ranges, and altered structural and functional states may emerge (Harmon et al., 1986).

In many cases, long-standing mature, diverse forest can be burned and repopulated by a more limited range of trees, grasses, and shrubs, with distinct ecosystem functions and services for animals and humans. This shift from one ecosystem type to another can be characterized as a regime shift or “landscape trap,” alluding to the less-desired or compromised functions offered by the new ecosystem (Lindenmayer et al., 2011). However, the question of ascribing value to specific landscapes and evaluating best management practices is one that is strongly site-dependent and defined in accordance the interests of decision-making stakeholders, and beyond the scope of the current study (Nassauer, 1997; Richardson et al., 2007). Assessing riparian ecosystem trajectories, recovery, and resilience following recent wildfires will be critical for addressing management needs and defining reasonable restoration goals and trajectories. Minimal tree regeneration, or regeneration of drastically altered communities, following recent disturbances from wildfires in the southwestern United States currently indicates that many forests may have already reached tipping points of regional-scale forest ecosystem change (Allen, 2014).

Fire and flood regimes and climate change

A prominent effect of the changing global climate has been wildfires of increased frequency, intensity, and extent (Sun et al., 2019). Changes in fire season lengths and wildfire frequencies are expected to increase in magnitude substantially over 74% of land worldwide, especially in the United States, Canada, Brazil, China, and Australia (Sun et al., 2019). Immediate effects of such fires include the destruction of mature established vegetation, animal habitats, release of CO_2 into the atmosphere, and threats to agriculture, human homes and air quality (Zybach et al., 2009). Although many forests are well-adapted to regular surface burns, intense large fires can fundamentally alter landscape cover composition when they occur at unprecedented scales (Harmon et al., 1986).

In riparian areas, additional impacts of extreme fire disturbance may include drastically increased stormflow, sediment transport, and flooding in areas where vegetation historically absorbed or slowed runoff of much of the water following seasonal rains (Bock & Bock, 2014; Dwire & Kauffman, 2003). The changed dynamics following the extirpation of existing vegetation contribute to changes in the geomorphology of river channels and floodplains, altering physical structure of established river systems, and fundamental characteristics of habitats that have been historically adapted to smaller-scale, intermittent disturbances. Depending on a river's natural geomorphology, native species, and disturbance legacies, river banks may be largely held together by vegetation that is rapidly established in the aftermath of intense disturbance. Subsequent destabilization of river banks during vegetative dormancy can then produce positive feedback loops of intensifying flood and fire disturbance regimes that prevent re-establishment of previously existing vegetation (Tickner et al., 2001).

Adaptations of riparian vegetation

Riparian ecosystems experience flooding, sediment deposition, and other physical disturbances requiring colonizing species to possess specialized adaptations that allow for their establishment and survival (Naiman, Decamps, et al., 2005). Specific adaptations include stem flexibility, root suckering, ability to reproduce from plant fragments, water resistance, rapid root growth, resprouting from hardy below-ground rhizomes, and seed dispersal methods that are aided by seasonal hydrology and intermittent floods (Allen, 2014; Naiman, Decamps, et al., 2005). Abiotic and biotic legacies left by disturbances such as avalanche, flood, wind, fire, drought, disease, litter accumulation, herbivory, and other physical influences, have enduring impacts on vegetation communities and cover types (Naiman et al., 1998).

As colonizers of naturally dynamic systems, riparian species and vegetation communities exhibit high *resistance* and *resilience* to disturbance. Holling (1973) describes *resilience* as "a measure of persistence of systems and of their ability to absorb change and disturbance *and still* *maintain the same relationships* between populations or state variables," contrasted with stability, or *resistance*, the "*ability of a system to return to an equilibrium state* after a temporary disturbance" (emphasis added). Specific areas of the riparian system may favor different colonizing vegetation more than others, and a more limited set of sites will be suitable for long-establishment of plant species (Harper, 1977).

In places where erosional processes dominate, plant communities tend to be resilient to disturbance; breakage and loss of biomass are likely, but relationships between species will persist and the community will re-establish over time. In contrast, where sediment deposition is the dominant process, plants may be resistant to anoxia and burial, but less likely to suffer breakage or other disturbance from high-velocity water flow that significantly alters community dynamics and relationships (Bornette et al., 2008). A resistant system, subject to unusual disturbance, may not possess the intrinsic resilience to return to its previous state.

Understanding what factors determine the structure and composition of riparian vegetation communities, and how they will respond to regular and extreme disturbances, will be critical to managing riparian zones for long-term resilience and maintenance of desirable ecosystem services. Merritt et al. (2010) propose a framework of organizing riparian plants into non-phylogenetic groupings, or *guilds* that share traits related to their role in the hydrologic regime: life history, reproductive strategy, morphology, adaptations to fluvial disturbance, and adaptations to water availability. Organizing vegetation types by their functions in the ecosystem requires a complex understanding of interactions and relationships between traits and environmental gradients (Merritt et al., 2010). However, such a framework may make clear functional possibilities that would not be apparent if managing to maintain or restore species composition.

Healthy coniferous forests generally exhibit some degree of fire resistance, and are able to re-establish following disturbance by dispersing seed from undisturbed patches to adjacent zones (Haffey et al., 2018). Forest understories in surface fire-prone landscapes often display high resilience, with previously existing communities returning to the understory after low-intensity burns (Naiman, Decamps, et al., 2005). However, greatly increased patch size of high-severity burns can reduce or eliminate the possibility of seed dispersal to open areas by previously prolific species (Johnstone et al., 2016). In contrast to surface fires, high-severity fires reach the forest crowns and cause mortality of trees that are well-adapted to surface burns.

In these situations, species known as "ruderal" or *'r'* strategists tend to dominate. Characteristics of *'r'* strategists include: small size and limited lateral spread, short life spans, high relative growth rates where nutrients are available, early maturation, frequent flowering, production of a large number of small seeds with wide dispersal potential, establishment of a large seed bank, and persistence of dormant seeds or zygotes (Bornette et al., 2008; Grime, 2006; Kautsky, 1988; Southwood, 1988). Both native and non-native plant species can exhibit '*r*' strategies and and fill important ecological niches. Less disturbed areas are likely to be re-colonized by seeds and propagules from adjacent patches, and may continue to resemble the vegetation communities in their immediate surroundings; whereas more highly disturbed areas over larger extents are more likely to be repopulated by extremely resourceful *'r'* strategists colonizing across great distance to the detriment of slower-growing species (Connell & Slatyer, 1977). Where *'r'* strategists are early colonizers, their high relative growth rates and prolific seed production in low-competition environments can then quickly exclude slower-growing vegetation, establish positive feedbacks mechanisms, and fundamentally alter the plant community in the medium- or long-term (Richardson et al., 2007). Various models of succession distinguish between different vegetation types and their propensity to colonize and establish in riparian zones, with competition and disturbance dynamics, and interactions between species, heavily influencing the trajectory of vegetative community development (Naiman et al., 1998; Oliver et al., 1996).

Implications for viability of forest recovery and management

Riparian vegetation communities respond to disturbances such that specific features of the system measured individually may be dynamic, however over scales of decades or centuries, ecosystem organizations tend to exhibit quasi-equilibrium in the absence of extraordinary disturbance (Naiman, Decamps, et al., 2005). Regular flood and fire pressures on riparian floodplains contribute to a *shifting habitat mosaic* where similar but heterogeneous habitats coexist and convert to different types in the context of disturbance (Kleindl et al., 2015; Mouw et al., 2009). Extreme disturbance events can contribute to widespread mortality of established trees and other species, fundamentally altering community composition and microclimates, and preventing the re-establishment of similar ecosystems (Benda et al., 1998, 2004; Harmon et al., 1986).

Many species are well-adapted to regenerating after fire by a variety of means; however, others are unable to re-establish, especially when burns are of larger extents and intensities than populations have historically experienced (Grime, 2006). When those opportunistic species rapidly re-establish following disturbance, they can exclude the conditions for populations of other slower-growing species to return, even if propagules or seeds are available (Richardson et al., 2007). Drastic and immediate shifts in species composition can confer changes in capacity of the system to filter stormwater and mitigate flooding, sequester carbon, and provide habitat to native animals (Sweeney et al., 2004).

The quaking aspen (*Populus tremuloides*), a tree species native to the American Southwest that resprouts from clonal root systems and long-distance seed dispersal, is facing regional decline linked to drought stress from warming temperatures, and declining precipitation that does not provide suitable habitat for the species in its historic range (Rehfeldt et al., 2009; Worrall et al., 2013). Historically, trees would resprout from below-ground tissue following surface fires, however the combination of a warming climate and drought do not allow for young trees to regrow. A better understanding of how to manage for resilient landscapes in the context of new climate conditions and disturbance regimes may be necessary to maintain biodiversity and healthy ecosystems.

With patterns of increased tree mortality from intensifying fire disturbances, disease, and pest outbreaks being seen across western North America, land managers need cohesive direction and robust information to revise goals and projected outcomes at local and regional levels (Meddens et al., 2012; Raffa et al., 2008; Westerling et al., 2006). As vegetation communities change in response to novel disturbance and climate, ecosystem services and functions, such as habitat, nutrient cycling, water filtration and distribution, and carbon sequestration, will be altered as well. Riparian systems consist of several vegetation and cover types, with no single site performing all desirable functions at once (Findlay et al., 2002). Restored habitat must similarly consist of a patchwork of ecosystem, vegetation, and cover types to perform a suite of functions. However, choosing which ecosystem services to invest in and ascribe value to are management decisions that must be made in consultation with and consideration of local communities and stakeholders (Nassauer, 1997; Richardson et al., 2007). While there may not be consensus on how and what to manage for, beginning to address these questions in the context of shifting regimes can help motivate effective long-term projects possessing social and political will for success.

Stream segment classification

Categorization of distinct stream *segments* that share characteristics such as geomorphic condition, recovery potential, and levels of management priority, may be one of the most important first steps in formulating robust network-based maps that assist in advancing management goals (Buffington & Montgomery, 2013; Naiman, Decamps, et al., 2005; O'Brien et al., 2017). Understanding local processes and linkages that connect rivers laterally, longitudinally, and vertically, can help managers identify *refugia*, healthy nodes of riparian vegetation that may act as seed and propagule sources, and legacies that support resilience in the context of subsequent disturbance, which may become apparent at finer scales of reaches, habitats, or microhabitats (Johnstone et al., 2016; Naiman, Decamps, et al., 2005; Sedell et al., 1990). Once specific segments can be identified as particularly susceptible or resilient to disturbance, opportunities and limitations of restoration efforts can be evaluated, and management can be maximized for successful enhancement of desirable services and functions (Naiman, Decamps, et al., 2005).

The loss of mature trees across widely disturbed areas, in tandem with present and projected increases in forest drought-stress index (FDSI), suggests that regeneration needs of specific species are less likely to be met. This includes factors such as availability of seed, which must coincide with favorable climatic conditions for successful germination. Management with attention to appropriate scales and intervals that can support regeneration of productive landscapes are needed (Jackson et al., 2009; Johnstone et al., 2016; A. P. Williams et al., 2013).

Bornette et al. (2008) suggest a model for predicting organization of plant communities in river floodplains that considers:

i. the nature of the physical constraints that affect plant communities (the scouring or depositing character of flood disturbances)
ii. the frequency and intensity of disturbances that limit competitive interactions and create gaps for recruitment for new individuals, and ultimately impede plant colonization
iii. the specific life-history traits that allow plant maintenance, recruitment and colonization in the variously disturbed riparian systems.

This model can help categorize segments by vegetative communities possessing species-specific traits that allow them to persist, given localized physical constraints and expectations of disturbance.

Given the context of changing climate and disturbance regimes, it should be expected that without intervention and with altered disturbance regimes and climatic conditions, forests will convert into different ecosystems, including grasslands and shrublands, or alternative forests with different dominant species (Jackson et al., 2009; J. W. Williams & Jackson, 2007). Understanding how specific segments and reaches have been disturbed, and what habitat potential they possess, will be critical to interpreting landscape change and developing effective management plans. Even in places where legacies such as *nurse structure* persist, which provide critical shade to vulnerable seedlings establishing in the semiarid landscape, changes to climate and disturbance regimes may lead to recruitment failures at otherwise suitable sites (Haffey et al., 2018; Johnstone et al., 2016). Broad changes in elevational distribution and dominance of many plant species have already been documented, and suggest that novel patterns will continue to emerge over the course of the 21st century with projected warming and drought conditions

(Allen, 2014). However, consequences ranging from loss of biodiversity, to reduced carbon sequestration and depletion of water supplies due to increased uptake from novel vegetation, must be considered when deciding where and how to intervene (Allen, 2014; Hicke et al., 2012).

Brierley and Fryirs (2013) propose an organizational framework of geomorphic assessment consisting of four steps: (1) river classification (i.e. segment classification), (2) geomorphic condition assessment, (3) recovery potential analysis, and (4) development of a management plan that strategically addresses restoration and rehabilitation goals. Performing a careful assessment in this manner can help inform effective restoration projects and avoid expensive mistakes; managers can attempt to work with, rather than against nature, to support resilient and diverse ecosystems that benefit people, communities, and native bioassemblages (Haffey et al., 2018).

Well-managed river floodplains should provide functions such as sediment retention, floodwater attenuation, nutrient absorption, erosion control, and biodiversity (Hughes et al., 2013; Meitzen et al., 2018). Rather than aiming to reconstruct an ecosystem state that existed at a specific point in time, management can seek to restore functional processes and provide habitat for threatened native flora and fauna (Richardson et al., 2007). Management of invasive species should respect the function that they now serve as part of a novel ecosystem, and avoid costly and short-sighted attempts at total eradication that could jeopardize the stability of riparian geomorphology and riverbanks (Tickner et al., 2001). Recognition that novel ecosystems in drastically altered riparian ecosystems will develop and can be managed to enhance biodiversity and functional ecosystem services will be critical for restoration success (Brooks et al., 2004; Sarr et al., 2005). However, many invasive plant species are early seral colonizers and thrive in low-competition environments. If native species fail to establish, positive feedback mechanisms may be triggered that support the continued dominance of invasive species to the exclusion of native vegetation (Richardson et al., 2007). In areas where forest regeneration is unlikely due to the limitations of changing environmental conditions, planting or seeding dense native herbaceous groundcover species in burned areas to promote the establishment of desired functional species rather than early colonizers that confer degraded environmental conditions, such as annual cheatgrass, which does little to prevent soil erosion or impede fire recurrence (Haffey et al., 2018). However, there are significant challenges to engineering novel landscapes that have no historical precedent, and where complex interactions between abiotic and biotic factors are poorly understood. Invasive species can serve critical functions within novel ecosystems, and management should not necessarily focus exclusively on extirpation of those species (Hobbs et al., 2009). Despite some challenges and limitations, directly researching a changing landscape may be the most effective means of understanding it, since suitable reference systems are globally rare (Richardson et al., 2007).

Restoration and management decisions require the definition of priorities in consultation with stakeholders, including traditionally-associated indigenous people, setting of multi-step goals, and continued monitoring and re-evaluation of priorities and progress (Richardson et al., 2007). Status quo management of fire suppression, and optimistic restoration projects involving the re-planting of disturbed species that are unable to effectively re-establish, are exceedingly expensive and reap limited rewards (Stephens et al., 2013). Managing for reduced flammability, as could be achieved through establishment of drought-tolerant deciduous-dominated landscapes, could help mitigate impacts of changing climate on fire frequency (Kelly et al., 2013). Controlled burning, mechanical tree harvesting, and extensive ground mulching could also reduce forest densities, competition for water, and fine fuel connectivity, in turn alleviating some of the forest drought stress and risk of high-intensity and wide-ranging fires (Ager et al., 2010; Finney et al., 2008). Additionally, in places where re-establishment or preservation of standing forests is still possible, pursuit of effective management strategies should be prioritized to reduce the likelihood of continued loss of threatened ecosystems, and the initiation of positive feedbacks that degrade landscape functionality (Haffey et al., 2018). Understanding the structural patchwork of vegetation, geomorphic conditions, and how both forest and non-forested landscapes can provide continued ecosystem services and fulfill management goals in both broad and hyper-local contexts will be critical as managers are forced to seek new solutions in a changing world.

History of land use and fires in Bandelier National Monument, New Mexico

Human land use and habitation has coincided with the stabilization of regional forest patterns over the last several thousand years on the Pajarito Plateau in North America, and corresponded with changes to fire regimes over at least the last several hundred years (A. P. Williams et al., 2013). Examination of fire scars around prehistoric Jemez village sites in the uplands of the study area show that few fires occurred in and around occupied village sites prior to European colonization, a pattern that is likely explained by continual fuel use by human populations that prevented the unimpeded spread of fire across the landscape (Swetnam et al., 2016). Evidence from tree-ring studies suggest that low-intensity surface fires that do not reach the crowns of ponderosa pine and mixed conifer forests were common throughout the Holocene. However, the size and intensity of recent stand-replacing fires is likely unprecedented since the establishment of regional climate, vegetation and fire regime patterns around 6,000-9,000 years before present (Allen, 2014; Anderson, Jass, et al., 2008; Fulé et al., 2014; Swetnam & Baisan, 2003).

Prior to contact with Spanish colonists, indigenous Puebloan people lived in and around the Pajarito Plateau for thousands of years. It is estimated that 5000-8000 people lived in an approximately 500 km^2 area from ca. 1300-1640 C.E., on par with the present-day definition of wildland-urban interface, where human land use interacts with and is strongly impacted by the natural environment and ecosystem (Radeloff et al., 2005). Unlike conventional wilderness management which seeks to eliminate human influence on the landscape, indigenous populations and cultures were integrated directly with the natural environment and ecosystems, and traditionally were participants in rather than disruptors of environmental regimes (Swetnam et al., 2016). Human habitation of upland village sites prior to 1600 corresponded with continuous clearing of fuelwood and reduced landscape fire connectivity. Weak associations between fires and interannual climate variations within the Jemez Mountains prior to 1680 indicate that fires were likely set by humans and kept in check by continuous land and fuel use (Swetnam et al., 2016). Small-scale, low-intensity fires were common and frequent, disturbing patches of forest that would leave mature trees standing, clear the understory, and open ecological niches for a diverse and resilient forest community to thrive (Swetnam et al., 2016). Indigenous history is corroborated by preserved pollen and tree-ring records that tell the same story of shifting land use and occupation patterns over the last millennium.

The earliest megadrought in the pollen and tree-ring record occurred during the 1200s, and corresponded with widespread tree mortality as well as changing land use practices by the San Ildefonso Pueblo people (Whitman & Whitman, 1947; A. P. Williams et al., 2013). Subsequently, several Pueblo villages were established on the Pajarito Plateau from the late 12th century until the 16th century, when drought pressures forced plateau-dwellers into the valley of the still-flowing Rio Grande River (Merlan & Levine, 2000; Whitman & Whitman, 1947;

Wilson, 2013). By the time the first Europeans arrived in the 16th century, the only permanent habitation on the plateau was the town of Tsirege. This name, meaning "bird-place" in Tewa, lent its meaning to the Spanish place name of the Pajarito, or "Little Bird" Plateau (Harrington, 1920; Merlan & Levine, 2000). Although the plateau might have been repopulated following the drought in the absence of colonization, European influence permanently altered historic patterns of habitation such that the uplands of the study area have not been home to any permanent settlements since approximately 1700 C.E. (Swetnam et al., 2016).

Depopulation, the introduction of livestock grazing to the region, and ensuing policy of fire suppression, contributed to the densification of mountainous forests in the region (Allen, 2014). Changes in land use that started in the dry period prior to Spanish arrival and continued through the colonial period known as *congregación* provided the conditions for forests to proliferate in and around former village sites at higher densities (Swetnam et al., 2016). Healthy ponderosa pine forests historically consisted of an open understory where low-intensity surface fires were common; as conditions changed, stem density increased in some places on the order of tenfold or more, from less than 100 to over 1000 trees per acre (Allen et al., 2002). The increased fuel load and development of a tall deciduous understory provided the conditions for several large scale stand-replacing fires to burn across tens of thousands of acres (Allen, 2014). Following the establishment of forests of unprecedented density, increasingly widespread and high-intensity fires have initiated drastic shifts in ecosystem organization (Allen, 2014). In the period from 1500 to 1680 C.E., only two widespread fires were recorded; in the following 180-year period ending in 1860, nine widespread fires occurred across the Jemez Mountains (Swetnam et al., 2016). This time period, following colonization but preceding any organized changes in management, was already seeing significant changes in the typical fire regime.

In the period following 1860 through the present day, cattle ranching came to dominate the landscape, and suppressing forest fires became a high priority for local and regional forest managers (Stephens et al., 2013; Swetnam et al., 2001; Swetnam & Baisan, 2003). Especially wet periods between 1905-1922 and 1978-1995 further promoted the establishment of dense woody vegetation (Allen et al., 2002). For most of the 20th century, fire suppression dominated, and the increasing standing biomass was not interpreted as a loss of resilience. Standing biomass likely reached unprecedented levels, with tree densities of ponderosa pine and mixed-conifer forests increasing ten-fold or more, and greater proportions of shade-tolerant and fire-sensitive species proliferating (Covington & Moore, 1994). This set the stage for more high-severity stand-replacing fires to occur throughout the region, including the La Mesa (1977), Dome (1996), Cerro Grande (2000), and Las Conchas (2011) fires that burned through the Bandelier National Monument (Allen, 2014; Veenhuis, 2002). Evidence from preserved bog records dating back over 15,000 years suggests that the sizes of patches recently disturbed by high-severity fires across southwestern ponderosa pine forests is likely unprecedented in the last 6,000-9,000 years, around the time that regional climate, vegetation, and fire regime patterns had stabilized up until Western colonization (Anderson, Jass, et al., 2008; Fulé et al., 2014).

Contextualized regionally within the southwestern United States, the Pajarito Plateau and Bandelier National Monument have well-documented records of forest drought-stress index (FDSI) from comprehensive tree-ring data sets preserved across the region (A. P. Williams et al., 2013). The reconstructed FDSI is critical for our understanding of how climatic trends influence tree growth and mortality, and for making predictions about how future changes will influence forest health and resilience. Examining how observed climate data from 1896-2007 is associated with recent tree-ring-derived FDSI values, it appears that warm-season vapor-pressure differential, which is largely a factor of temperature and cold-season precipitation, are the strongest predictors of FDSI (A. P. Williams et al., 2013). This information, taken with predictions of decreased cold-season precipitation and increased temperatures over the course of the 21st century, suggests that FDSI is on track to reach its most severe in at least a millennia by 2050 (Breshears et al., 2005; A. P. Williams et al., 2013). Additional observed correlations derived from climate records and satellite measurements indicate an exponential correspondence between FDSI and areas burned by recent wildfires, compounding the effects of changing land use on fire regimes (A. P. Williams et al., 2013).

Over the course of the 19th and 20th centuries, conflicting interests between archaeologists, homesteaders, and livestock owners on the Pajarito Plateau called for the establishment of more comprehensive and centralized management to address different stakeholders' needs (National Park Service, 2015). Following years of contention and discussion, Bandelier National Monument was designated by Woodrow Wilson in 1916, with the stated purpose of "reserving these relics of a vanished people," making explicit the assumption that the indigenous people and cultures traditionally associated with the land had been extirpated (Merlan & Levine, 2000). In reality, Pueblo and Navajo communities and governments persist as living cultures and autonomous nations within the borders of the United States, with their own traditional uses and relationship with the area. Through review of literature and primary consultation with tribes, Merlan and Levine (2000) determined that six Pueblos: the Cochiti, San Felipe, San Ildefonso, Santa Clara, Santo Domingo, and Zuni, are traditionally associated with the land that has been designated as Bandelier National Monument. Three other communities, the San Juan Pueblo, Zia Pueblo, and Hopi Tribe, also assert historic or traditional relationships with the area, and the Navajo Nation additionally has noted that at least four Navajo clans likely have origins in Puebloan communities of the Rio Grande (Merlan & Levine, 2000).

Defining appropriate management goals necessarily requires consultation with indigenous communities and understanding the cultural context and history of land use in the area. After beginning the process of performing an ethnographic investigation of the area in the late 20th century, anthropologists Merlan and Levine soon changed their focus from conducting empirical research to establishing partnerships. By the second phase of their work, in consultation with the traditionally associated tribes, they expanded their project to primarily focus on the formation of a consultation committee, with representatives of the six aforementioned Pueblos. The tribes "wanted to emphasize government-to-government consultation on management issues rather than ethnographic research" (Merlan & Levine, 2000). This trajectory is reflective of the historic relationship between colonial establishments and indigenous culture, wherein Native American communities of practice and bodies of knowledge are perceived as extinct and something to be studied and evaluated through an external lens, rather than living through active cultural practice that continues to be passed from generation to generation. Management today is under the primary jurisdiction of the National Park Service, and is performed in consultation with indigenous governments. However, United States federal agencies continue to serve as primary decision makers in management decisions.

Within Bandelier National Monument, stated goals by the National Park Service include protecting and preserving ancestral Pueblo archaeological sites, as well as cultural and natural resources for the living Pueblo cultures traditionally associated with the Monument (National Park Service, 2015). With fundamentally altered native plant communities, disturbance regimes, and geomorphological and hydrologic conditions within the Bandelier National Monument, management must recognize that riparian systems are open and dynamic, and historically and presently influenced by human practices and use. Today, it seems that unprecedented conditions and ensuing drought have set the stage for several of the most severe and widespread wildfires on record to occur throughout the region, converting conifer-dominated landscapes to other cover types for likely the first time in the Holocene Era (Anderson, Jass, et al., 2008; Fulé et al., 2014).

Chapter Two: Research Manuscript

Introduction

River systems and floodplains are resilient to seasonal and intermittent flooding and depend on them to distribute nutrients, organic matter, sediment, and organisms. However, dramatic changes to flood regimes in either direction can produce novel reinforcing feedback loops and result in establishment of a degraded ecosystem that is resilient to newly intensified disturbance regimes (Sparks et al., 1990). Significant changes to the hydrology of riparian systems and geomorphic conditions can alter vegetative communities that are specifically associated with distinct fluvial landforms, and prevent them from reestablishing due to lack of recruitment and unsuitability of altered habitat (Hupp & Osterkamp, 1985). A thorough analysis of changes to vegetative cover over time can help extrapolate the effects of fire disturbance in analogous situations as unprecedented disturbances continue to occur worldwide.

The Bandelier National Monument in New Mexico, USA, exhibits diverse vegetation types and habitats at different elevations, including juniper savannas, piñon-juniper woodlands, canyon-wall shrublands, ponderosa pine (*Pinus ponderosa*) forests, mixed conifer and riparian forests, and montane grasslands (National Park Service, 2015). From the onset, federal protection of the land occupied by the Bandelier National Monument has been motivated by the presence of rich archaeological and cultural artifacts belonging to the Pueblo people. These artifacts and historical sites are the legacy of the living Pueblo cultures that have been displaced by drought and colonization. Today's management incorporates the goals both of preserving archaeological remains and ecological functions and value within the Monument (National Park Service, 2015).

The present study focuses on the Frijoles and Capulin canyons of the Pajarito Plateau, from their headwaters in the Eastern Jemez Mountains to their confluence with the Rio Grande in Northeastern New Mexico, USA, and the effects of wildfire and flood disturbance on vegetative cover since the early 1990s. Large, high-intensity stand replacing fires have taken place over great extents in the American Southwest, in an area with well-preserved bog and tree ring records indicating that present climactic and disturbance conditions are unprecedented in at least the last 6,000-9,000 years (Anderson, Allen, et al., 2008; Anderson, Jass, et al., 2008). Quantifying how these disturbances have affected the landscape in the immediate aftermath and years following disturbances will be critical for addressing management needs and defining reasonable restoration goals and trajectories.

How have recent wildfires and floods have altered the composition of riparian canyon vegetation in Bandelier National Monument? To answer this question and quantify changes to landscape cover, remotely-sensed imagery was manually reviewed to determine proportions of cover type occupied by forests, smaller vegetation, and bare ground. The results were compiled by stream segment and categorized by type and degree of disturbance. Classifying riparian zones contributes to a framework for riparian management that recognizes the unique internal dynamics as well as costs and outcomes associated with managing distinct segments (Quinn et al., 2001). This inventory of riparian vegetation will be used by the National Park Service to evaluate management practices and make determinations for how to adapt to meet evolving goals.

History of fire on Pajarito Plateau

Prior to the arrival of Europeans to the Pajarito Plateau in the 16th century, humans coexisted with fire in the semiarid landscape, using fire for active management of the landscape, and keeping their communities and villages safe by clearing and burning fuel wood around inhabited sites (Swetnam et al., 2016). Small-scale, low-intensity, surface fires that did not reach the crowns of mature forests were common and frequent during pre-colonial times, and would disturb patches of the forest, clearing the understory, and opening ecological niches for a diverse community to establish (Swetnam et al., 2016). The estimated density of the Plateau from 1300-1640 C.E is on par with the present definition of wildland-urban interface, which today are managed for their high risk of environmental consequences on human populations, however weak associations between fires and interannual climate variations during precolonial times indicates that fires that would be destructive to human habitations were likely kept in check by continuous land use and effective management (Haight et al., 2004; Swetnam et al., 2016). During periods of drought, indigenous populations would move to areas where water was more plentiful, however regular human migrations provided for long-term large-scale management of fire (Merlan & Levine, 2000).

After colonization, changing land use and fire suppression prevented regular surface fires, and standing forest biomass multiplied by as much as ten-fold, with increased proportions of fire-sensitive species proliferating (Covington & Moore, 1994). Several high-intensity stand-replacing fires have recently occurred throughout the region, including the La Mesa (1977), Dome (1996), Cerro Grande (2000), and Las Conchas (2011) fires that burned areas of Bandelier National Monument (Allen, 2014; Veenhuis, 2002). Today, intensifying forest drought-stress index and an exponential correspondence between FDSI and recent severe wildfires, indicates that the conditions that have given rise to the current fire regime and challenged the re-establishment of native forest vegetation are set to continue (Haffey et al., 2018; A. P. Williams et al., 2013). For what is probably the first time since humans inhabited the region, conifer-dominated landscapes characterized by frequent surface fires across the American southwest are being replaced by novel cover types and disturbance regimes (Anderson, Jass, et al., 2008).

Direct effects of recent fires on Capulin and Frijoles Canyons

While the present study will serve to formally describe and quantify changes in vegetative cover types, the immediate impacts of recent severe wildfires on the vegetative communities and geomorphological conditions of the Capulin and Frijoles Canyons are already clear. Mortality of mature trees was obvious in large areas of burned forest, and significant changes to streamflow volume and sediment transport were immediately observed at gaging stations as well (Veenhuis, 2002).

In the first season following the 1977 La Mesa and 1996 Dome fires, peak flow at most downstream streamflow-gaging stations in the Frijoles and Capulin Canyons increased to about 160 times the maximum flow recorded before the fire (Veenhuis, 2002). As vegetation re-established two seasons following the fires, maximum peak flow decreased to 10-15 times pre-fire maximums in the second year, and in the third year it decreased to 3-5 times pre-fire maximums. In addition to maximum peak flow volumes being dramatically affected, the frequency of large stormflow events increased markedly in the three years following the La Mesa and Dome fires (Veenhuis, 2002). The flooding caused by high volume stormflow events had drastic effects on vegetative cover composition even in areas that did not directly burn.

The previously existing riparian ecosystem provided habitat to native flora and fauna, as well as ecosystem services that may not be fulfilled in the context of altered dynamics following disturbance. Drastically altered vegetation dynamics may introduce positive feedback loops encouraging the long-term dominance of shrub and herbaceous species, and in turn prohibiting riparian forests from reestablishing and providing the same habitat and services that they did before the disturbance (Richardson et al., 2007).

Changes in climate conditions and land use practices in the southwestern United States may be preventing the re-establishment of some native vegetation following fires, which require specific structural and environmental conditions, as well as availability of viable propagules to re-establish following disturbance (Haffey et al., 2018). In a study of regeneration patterns in ponderosa pine forest landscapes following eight fires in Arizona and New Mexico during an 18-year regional drought from 1996 to 2013, researchers found evidence indicating that low-elevation, dryer areas, and areas further from conifer seed sources, were less likely to regenerate with similar vegetation communities following disturbance (Haffey et al., 2018). Warmer temperatures have generally been associated with increased tree mortality, and reduced success for establishment of new seedlings, likely due to water stress from increased atmospheric vapor pressure deficits (Allen, 2014; McDowell et al., 2011; A. P. Williams et al., 2013). Recent large-extent high-severity wildfires have caused mortality of virtually all tree seed sources across tens of thousands of acres, with the consequence of grasses and shrubs achieving dominance without competition (Allen, 2014). Similar patterns are likely to be observed in similar forests such as those of the Frijoles and Capulin canyons, unless there is significantly more availability of seed in the riparian systems, or human intervention. Quantifying changes to the canyons and distinguishing between segment categories by disturbance will help inform effective management that heavily weighs localized factors and characteristics.

Methods

Study area

The area of interest consists of the Frijoles and Capulin Canyon riparian ecosystems in the Eastern Jemez Mountains of New Mexico, USA. It includes the areas from their headwaters at 2825 m above sea level to their confluence with the Rio Grande at 1630 m. The study encompasses the entirety of the riparian canyon bottoms. The Frijoles Canyon bottom consists of 154 hectares, including a main channel that runs about 19 km from the Rio Grande towards its uplands before splitting into two: a northern fork extends four additional kilometers, and the southern fork extends three kilometers from the junction. The Capulin Canyon bottom is 141 hectares, and consists of a main channel that splits approximately 17 km upstream of the Rio Grande to a northern fork extending an additional 5 km and the southern fork just 2 km past the junction (Figure 1).

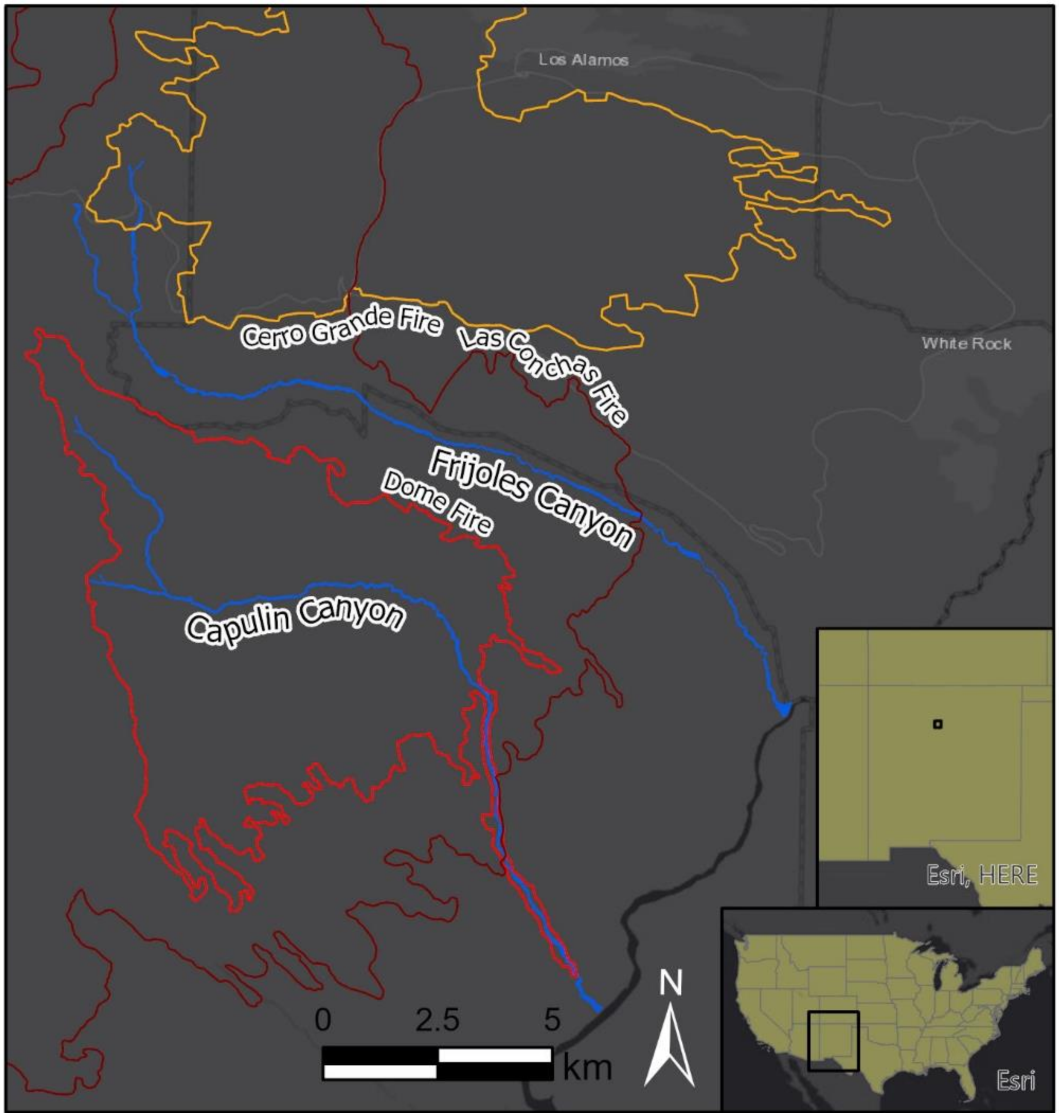

Figure 1. Study area map showing Frijoles and Capulin Canyons, and three large recent fire extents.

Sample design

Canyon bottom areas were designated using a bottomland boundary developed by the United States Geological Survey (USGS). The procedure calculated the topographical position index from a 10 foot DEM using R to identify connected raster cells that were associated with a highly negative TPI (Assal et al., 2015). The polygons were then manually reviewed to fill in holes and remove islands that appeared outside the canyon bottom so that each canyon bottom was confined by a single contiguous border. This layer was divided into 1 km stream reaches for exploratory purposes and was used as the focus layer for ArcGIS interpretation. A grid of approximately 30 m x 30 m squares was created over the bottomland layer using the "Grid Index Features" geoprocessing tool (Figure 2). We then selected only the squares that had their center in the bottomland layer, excluding those whose centers did not fall within the area of interest. We created points at the centers of the squares of the grid polygon layer, and buffered the points to generate circles with a radius of 3 m. The sample design was developed to provide a thorough representation of all stream reaches, with at least 40 buffered sample points per 1 km reach in each river. The final sample consisted of 2094 points in Frijoles Canyon and 1911 points in Capulin Canyon.

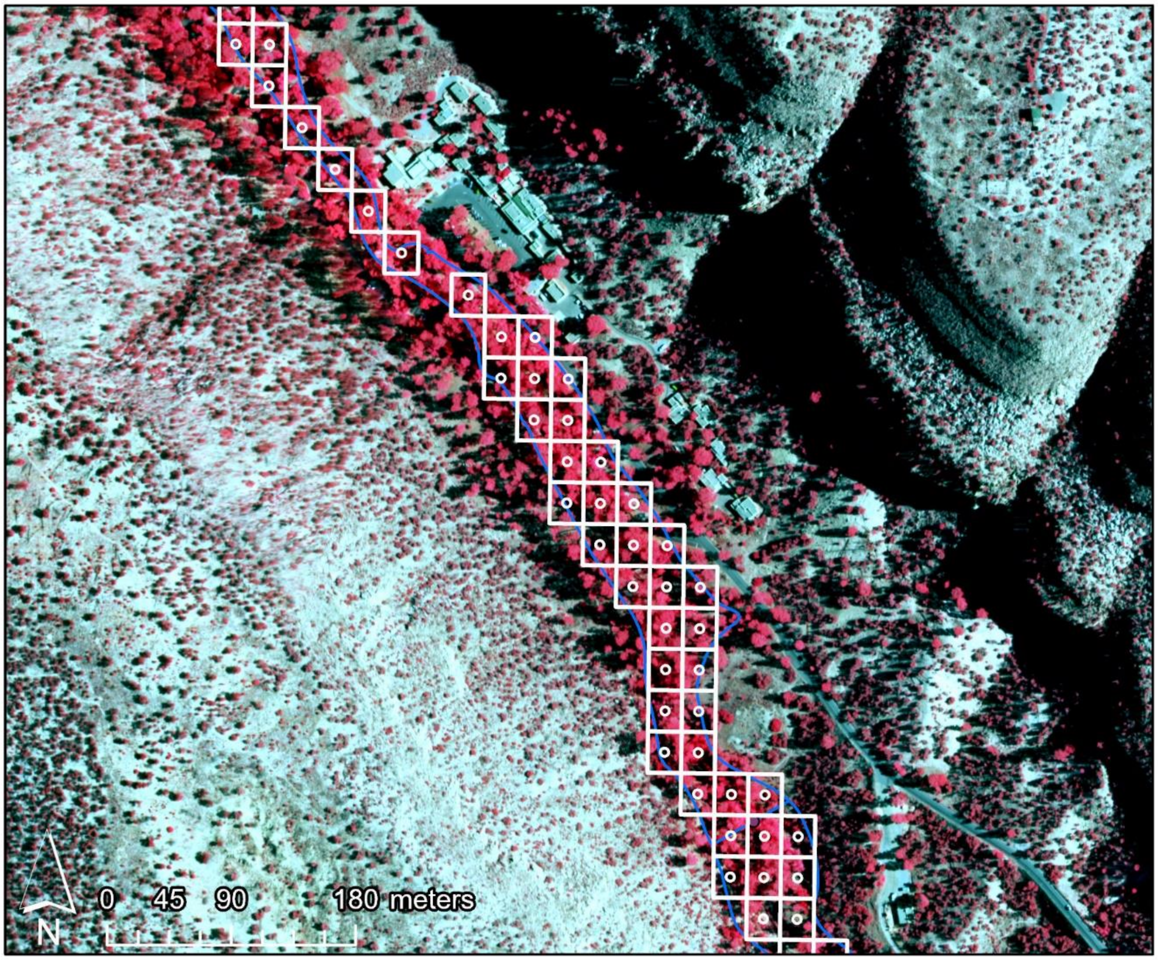

Figure 2. Buffered centers of grid squares in the Frijoles Canyon bottom alongside the Bandelier National Monument Visitor Center.

Satellite interpretation

Aerial photographs and satellite imagery were obtained from the Los Alamos National Lab (LANL) and United States Forest Service (USFS) for the years 1991 and 1992, respectively, and from the National Agriculture Imagery Project (NAIP) for the years 2011, 2014, and 2018.

LANL and USFS imagery included 3 spectral bands that were symbolized with a standard deviation clip to highlight extreme values for visual clarity. NAIP imagery was symbolized using the color infrared filter and a standard deviation clip to highlight the presence or absence of vegetation (Figure 3).

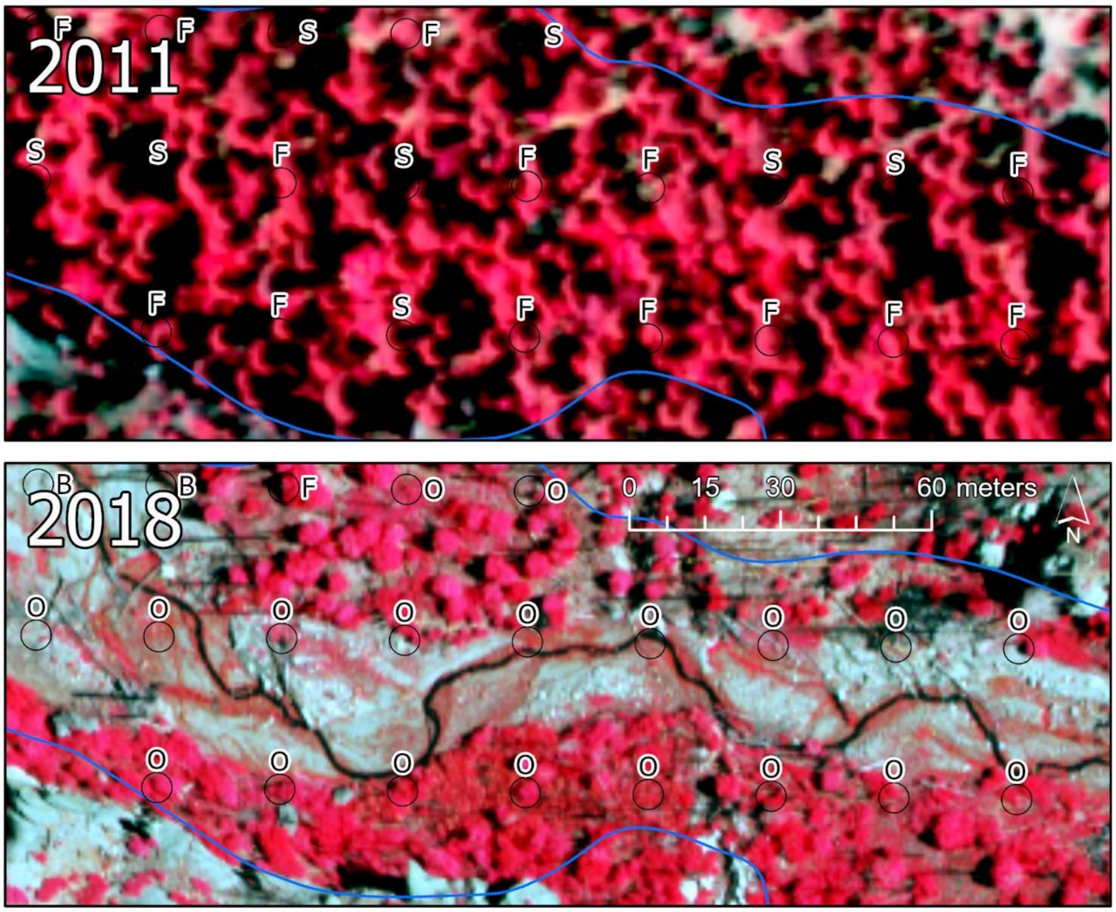

Figure 3. NAIP Imagery from 2011 and 2018 showing examples of sample point classified as forest (F), other vegetation (O), wetted channel (W), bare ground (B), and shadow (S).

The sample array was overlaid on each set of imagery using ArcGIS Pro. Each point was classified as one of seven cover types by visually interpreting the imagery (Table 1). The three most abundant cover types were: large arboreal riparian vegetation (F), other, smaller woody or herbaceous vegetation (O), and dry channels or bare floodplain sediment, which may include dormant herbaceous vegetation (B). A small number of sample points landed on semi-permanent reservoirs and were designated as wetted land (W). Additionally, some sample points were shadowed (S), or did not have imagery available (U), and were excluded from analysis of relative proportions of cover types.

A bare channel category (C) was designated using a spatial analysis. The locations of the river channels in Frijoles and Capulin canyons were designated as a polygon layer using the post-Las Conchas Fire 2014 imagery, when the channels were most disturbed. The channel was determined by the boundary of the centrally scoured area, with no channel designated where vegetation obscured the ground. A spatial analysis was performed on the data to designate a bare channel category – these were sample cells that were both manually designated as bare ground, and spatially coincided with the channel layer. Changes in this category may reveal patterns of geomorphology affecting legacies and recovery of riparian vegetation. These cover types were selected to accurately illustrate functional changes to the landscape at the highest possible level of detail (Table 1).

Table 1. Criteria used to designate different cover types for NAIP imagery.

Cover Type	Criteria
Forest	Bright red spectral signature Crown diameter >6 m
Other vegetation	Bright red spectral signature Scattered or uniform- may be small deciduous, coniferous, or herbaceous Crown diameter <6 m
Bare ground	Infrared or green light not reflected in spectral signature
Bare Channel	Bare ground cover that spatially coincides with the designated channel
Wetted ground	Ground is covered by water
Shadow	50% or more of cell is completely black
Unidentifiable	No imagery for sampled cell

Spectral signatures including color, texture, and shadow from satellite imagery were used to manually distinguish each cover type. Each point was visually inspected at a scale of 1 to 1000 using the visible signature within each buffered circle to determine which cover type was present at the sample. For each cell, the cover type that comprised the majority of the visible (non-shadowed) area of the buffered sample point was designated as the dominant cover type for the sample. Points that were 50% or more shadowed were designated as shadow, however if at least 50% of the image within circle was visible, then it was interpreted according to the preceding rule. Where the sample intersected the wetted channel, the wetted area was treated as shadow and the rest of the circle was analyzed for interpretation. For points where the majority cover type was not immediately obvious, the imagery was inspected at a larger scale; where uncertainty remained, the point was flagged and revisited with a second reviewer, and a decision was taken by consensus.

Data analysis

Data were summarized using R (R Core Team 2020) to explore the overall composition and changes to cover type over time for each of the canyons. Initial analysis consisted of calculating the proportional composition of each canyon for each year analyzed. Charts were produced to visualize the cover proportions by 1 km reaches, and were used as the basis for exploring changes to the landscape detectable through the remotely-sensed imagery. Snapshots in time, as determined by aerial photographs (1991/92, 2011, 2014, and 2018), and disturbance type were considered as variables that had an influence on cover proportions. The Dome (1996), Cerro Grande (2000), and Las Conchas (2011) fires all occurred within the time period over which the analysis spans and had direct and indirect effects on vegetative cover. Areas within and downstream of the fires also experienced increased sediment discharge, stormflow, and flooding in the years following burns, causing additional mortality of mature forest, and limiting the reestablishment of vegetation.

Stream segment types were classified by disturbance and include (1) areas that were directly impacted by burns, (2) areas flooded by high volume stormflow downstream of burned areas and (3) areas that both burned and flooded. A permutation test was used to assess the independence of land cover counts with disturbance type (i.e. burned and/or flooded areas), stratified by year, in each canyon separately. This test was used to assess any changes to vegetative cover that have occurred in the context of these disturbances, in contrast to the 1 km reaches. The R package 'coin' was used to calculate an asymptotic general independence test using χ^2 values, which is a permutation-based calculation of the classical Cochran-Mantel-Haenszel test of independence (Hothorn et al., 2008).

Results

Initial analysis consisted of calculating the proportional composition of each canyon for each year analyzed. The proportion designated as shadow ranged from 5% in Capulin 2018, to 25% in Frijoles 2014 (see Appendix). Shadowed sample points and those for which imagery was not available were excluded from analyses to obtain weighted proportions of Forest, Other vegetation, Bare ground, and Bare channel. Wetted ground consisted of less than 1% of sampled points and was excluded from the analysis.

Frijoles Canyon

Parts of Frijoles Canyon burned in the 2000 Cerro Grande Fire, and 2011 Las Conchas Fire. Less than 10% of the canyon burned in the Cerro Grande Fire, and over 75% burned in the Las Conchas Fire, according to fire perimeters published by the Monitoring Trends in Burn Severity (MTBS) Program. While analysis of additional snapshots in time, especially in the time periods immediately following each fire, may further enhance our understanding of how cover changes over time, looking at the points directly and indirectly affected by fires even years later can reveal the types of lasting effects that stand-replacing fires incur.

Although a small decline in forest cover was observed between 1992 and 2011 in forest cover, the cover change to the burned area was likely much greater in the years more immediately following the 2000 Cerro Grande Fire, which is not reflected in the data due to the 19-year gap in measurement, and lack of data for the early 2000s. Likewise, the significant drop in forest cover, and increases in other vegetation and bare ground cover, between 2011 and 2014 show changes that occurred as a result of the 2011 Las Conchas Fire (Table 2 and Figure 4).

Table 2. Proportions of vegetative cover type observed in Frijoles Canyon overall.

Cover type	Year1992	Year2011	Year2014	Year2018
Forest	73%	69%	17%	18%
Other vegetation	17%	21%	45%	71%
Bare ground	4%	5%	14%	4%
Bare channel	6%	6%	25%	8%

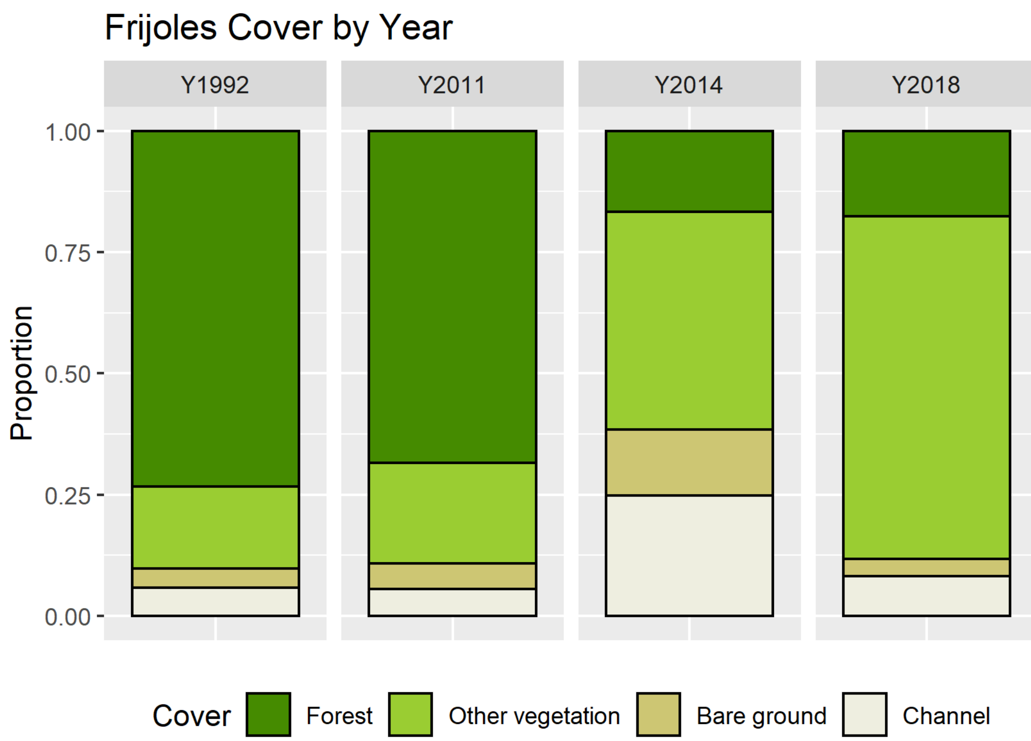

Figure 4. Proportions of vegetative cover observed in Frijoles Canyon overall.

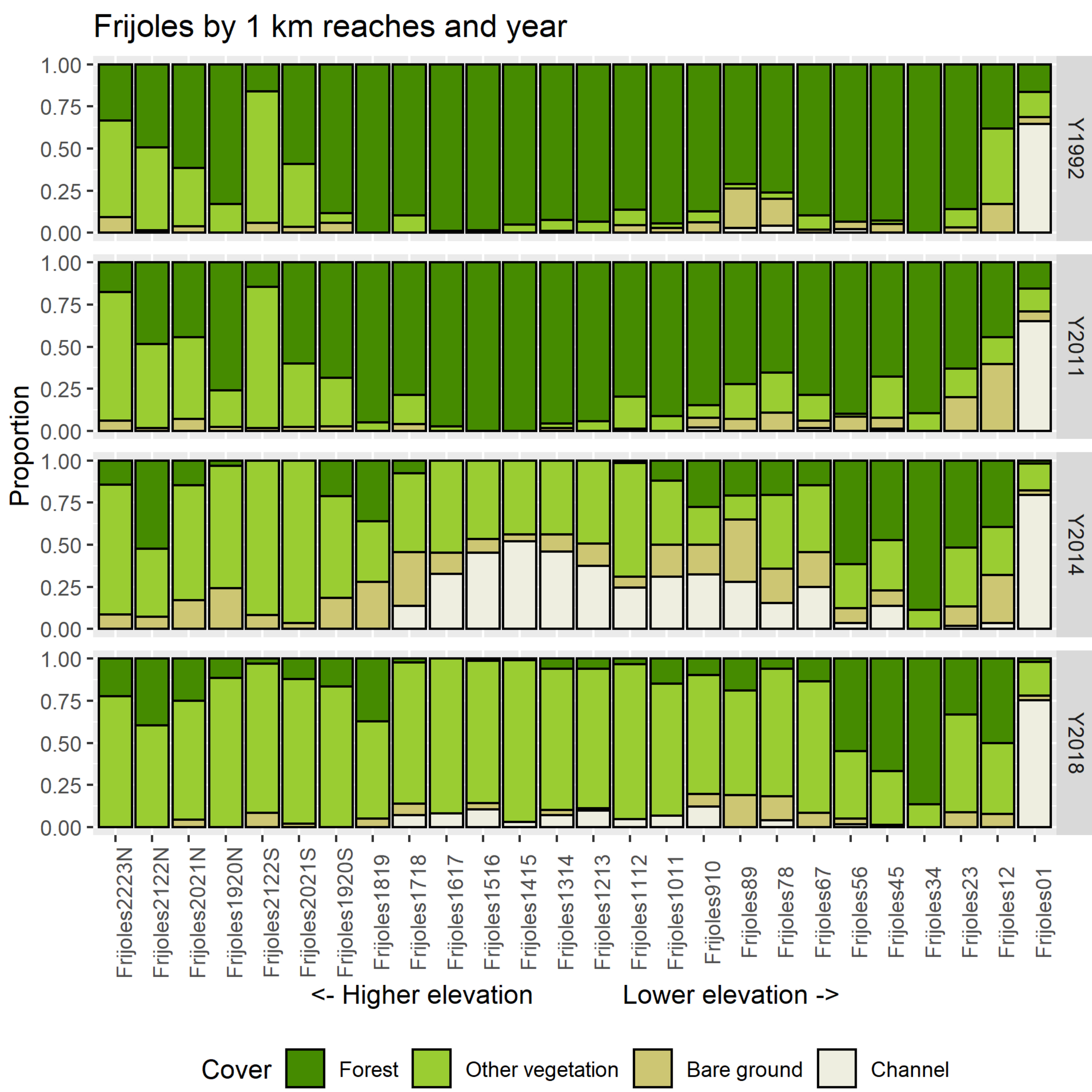

Figure 5. Proportions of vegetative cover type observed by 1 km reaches in Frijoles Canyon.

Further analysis of the proportions of cover type by 1 km stream reaches shows differences between the lower, middle, and upper segments, which have been variably affected by both fire and flood. Labels indicate the kilometer reach represented, with “Frijoles01” being the lowermost reach between kilometer markers zero and one, and so on (Figure 5). Frijoles

Canyon was not affected by the Dome Fire, and only the northernmost 1 km of Frijoles was burned by a large fire in 2000. Any changes, apart from those located in the highest reach of the northern branch of Frijoles, between 1992 and 2011 cannot be directly attributed to large wildfires.

Following the Las Conchas Fire, in 2014, the effects of wildfire disturbance throughout Frijoles Canyon are clear. A tributary at km 17.5 brought a large volume of water, sediment, and debris to the main channel, and resulted in significant scouring between km 6 and 17.5 represented by the bare channel. From km 6 to km 12, there is some retention of forest cover following the Las Conchas Fire, and from km 12 to 17.5 there is virtually no forest. In 2018, much of that area had been revegetated by pioneer species, however reforestation has not yet occurred (Table 3, Figure 6). The lowermost kilometer of Frijoles contains a wide delta that was mostly categorized as bare ground, and anomalous compared to the rest of the system. Vegetative cover counts in Frijoles Canyon were not independent of disturbance type while stratified by year (χ^2 = 843.6 and p < 0.0001, Figure 6).

Table 3. Classification and descriptions of segments by disturbance in Frijoles Canyon.

Frijoles km markers	**Segment**	**Description of disturbance**
km 0-0.3	Y	Discarded from analysis; river delta, limited established vegetation observed across time periods
km 0.3-1.5	Z	Discarded from analysis due to extensive dark shadows in 2014 and 2018 imagery
km 1.5-6	A	Flooding downstream of fire, some disturbance of forest vegetation with significant retention of forest cover
km 6-12	B	Direct burn and heavy flooding, significant loss of forest cover following Las Conchas Fire
km 12-17.5	C	Direct burn and heavy flooding, nearly complete loss of forest cover following Las Conchas Fire
km 17.5-headwaters	D	Directly impacted by Las Conchas Fire, some retention of forest cover

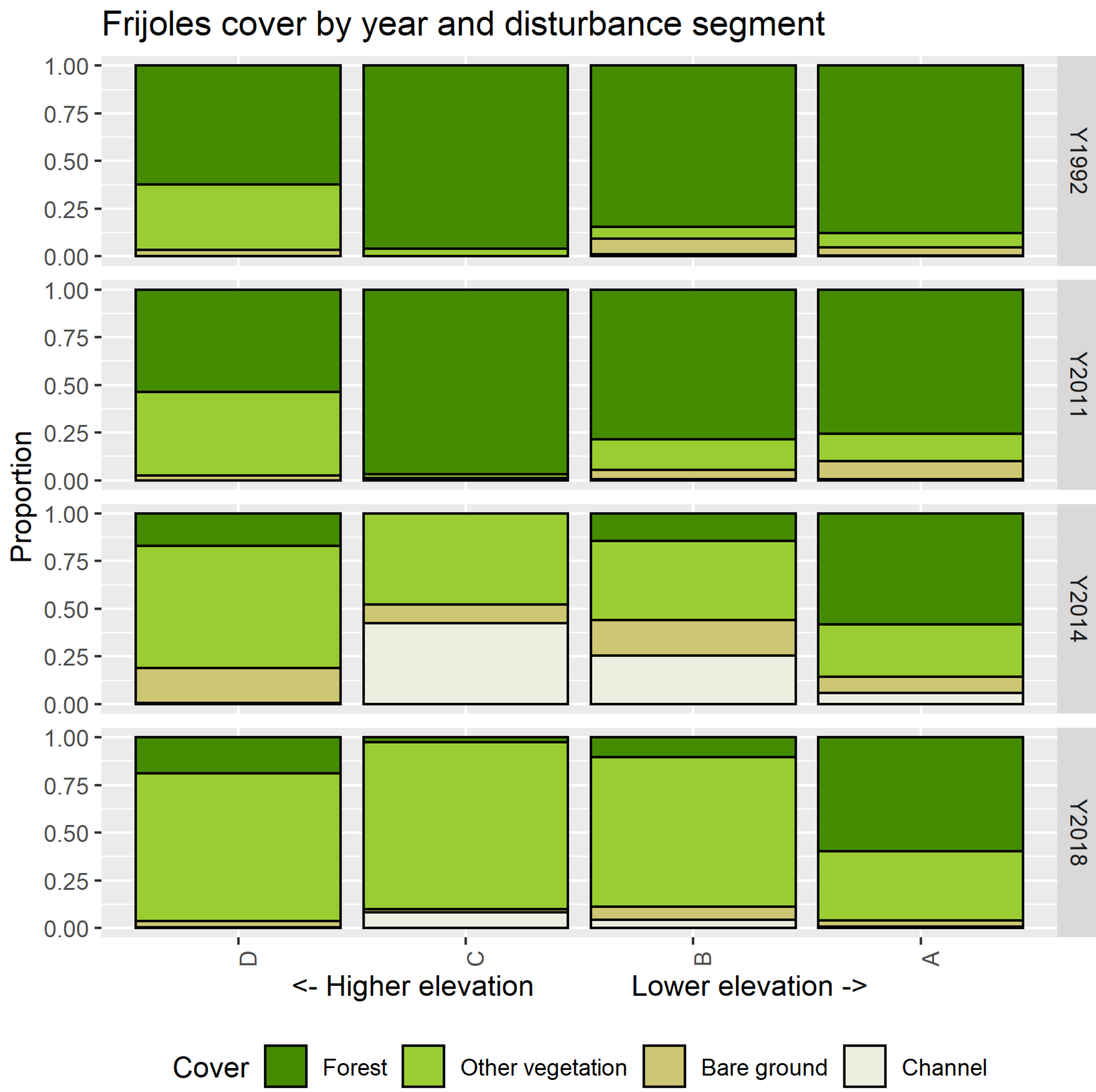

Figure 6. Proportions of cover types observed by disturbance segment and year in Frijoles Canyon.

Capulin Canyon

Capulin Canyon was unaffected by the 2000 Cerro Grande Fire, however experienced nearly complete burning in the 1996 Dome Fire and 2011 Las Conchas Fire. A drastic decline in forest cover is apparent between 1992 and 2011, and 2011 and 2014, during which these two fires occurred. While the canyon was not as densely forested as Frijoles at the beginning of the study, its decline from 57% to 7% cover overall is drastic (Table 4 and Figure 7).

Table 4. Proportions of vegetative cover type observed in Capulin Canyon overall by year.

Cover type	Year1992	Year2011	Year2014	Year2018
Forest	57%	29%	7%	3%
Other vegetation	32%	52%	61%	75%
Bare ground	6%	13%	8%	6%
Bare channel	5%	6%	25%	17%

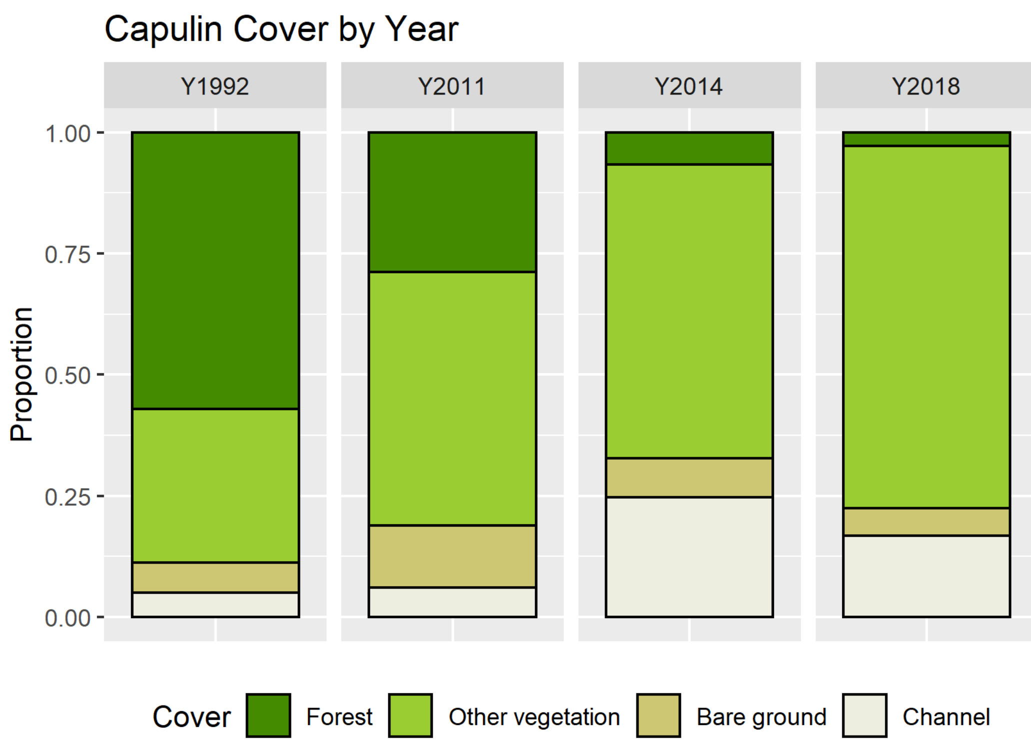

Figure 7. Proportions of vegetative cover observed in Capulin Canyon overall by year.

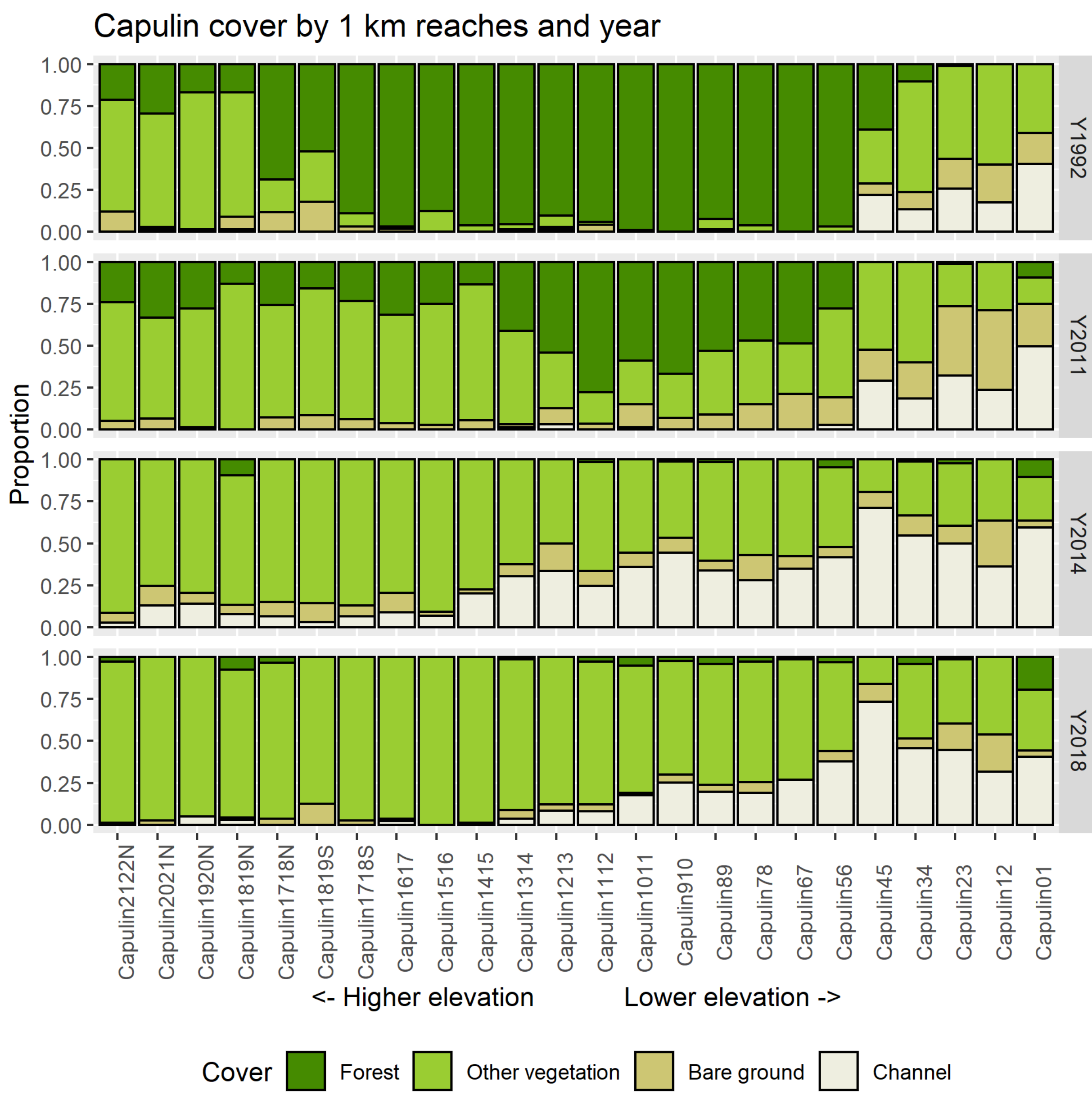

Figure 8. Proportions of vegetative cover observed by 1 km reaches in Capulin Canyon.

The Dome Fire in 1996 burned most of the canyon north of km 4.5. The drastic change from near-complete forest in the middle reaches to significantly less in 2014, and nearly none in 2014 and 2018 is clear. The bare channel is persistent in the lower reaches from 1992 with the proportion varying little between years. Following the 2011 Las Conchas Fire, the channel becomes apparently scoured from km 15 down in 2014, with some recovery of vegetation by 2018. The forest fails to recover to the proportion it represented in the early 1990s. Labels indicate the kilometer reach represented, with “Capulin01” being the lowermost reach between kilometer markers zero and one, and so on (Figure 8). Categorizing the canyon by reaches show four distinct categories of disturbance (Table 5, Figure 9). Vegetative cover counts in Capulin Canyon were not independent of disturbance type while stratified by year (χ^2 = 2099.7 and $p <$ 0.0001, Figure 9).

Table 5. Classification and descriptions of segments by disturbance in Capulin Canyon.

Capulin km markers	**Segment**	**Description of disturbance**
km 0-4.5	E	Multiple direct burns from km 1-4.5; area includes the river delta, heavy flooding and regular scouring of vegetation; absence of surface water for much of the year
km 4.5-11	F	Multiple direct burns, heavy flooding, nearly complete disturbance of forest vegetation and limited reestablishment of herbaceous vegetation; perennial flow ends upstream of km 4.5
km 11-13.8	G	Multiple direct burns and heavy flooding; significant reestablishment of vegetation following disturbance
km 13.8-headwaters	H	Multiple direct burns and heavy flooding, nearly complete loss of forest cover following Las Conchas Fire, with substantial reestablishment of smaller vegetation

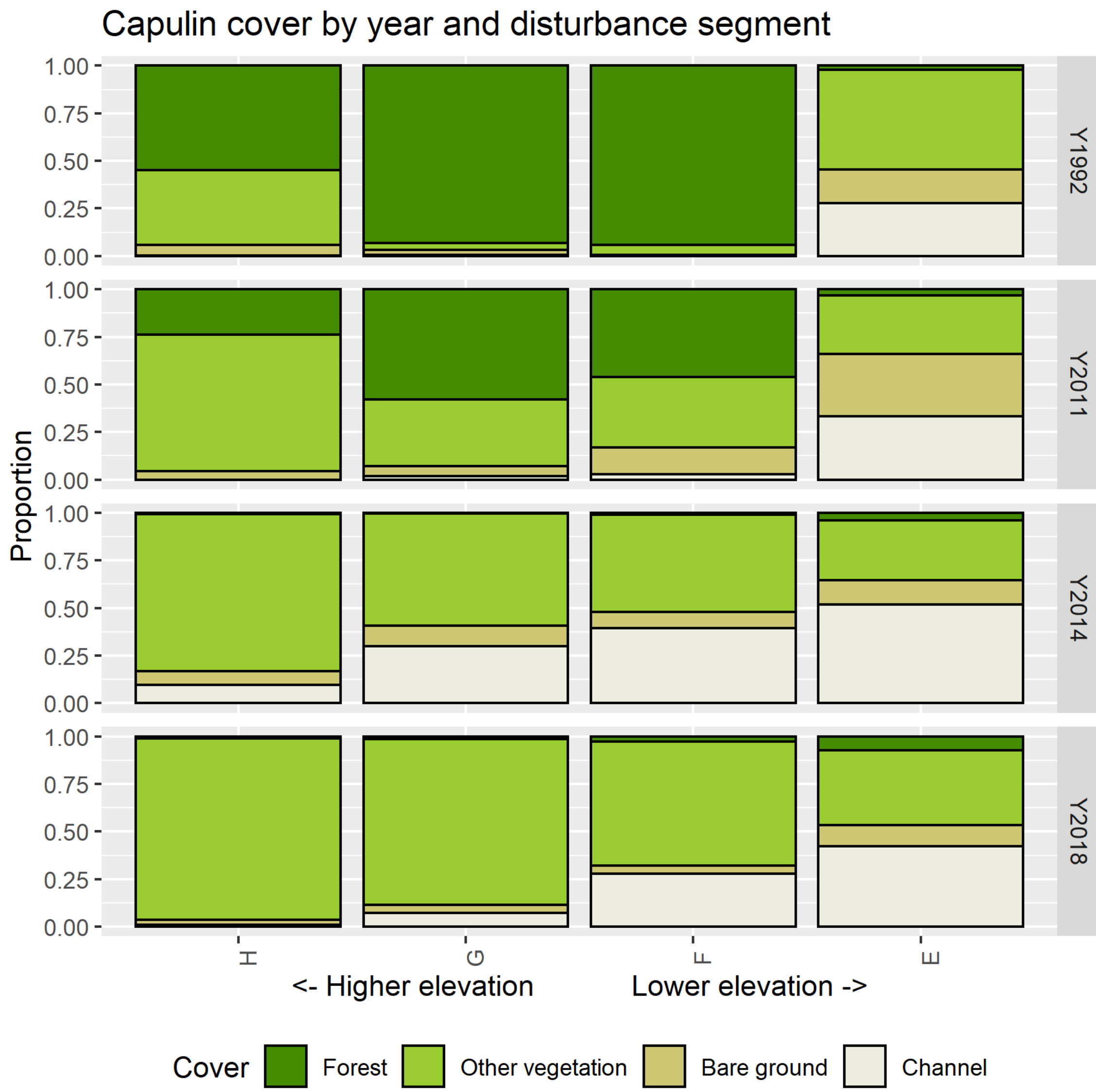

Figure 9. Proportions of vegetative cover observed by disturbance segment and year in Capulin Canyon.

Discussion

Canyon bottoms in the Pajarito Plateau are of particular interest to land cover researchers because they represent a portion of the semiarid landscape with the most availability of water and propensity for propagules to disperse by wind, seasonal stream flows and floods. In the aftermath of extreme wildfires, these areas are likely to experience the most extreme flood disturbances, while also possessing some of the most opportune conditions for vegetation to re-establish. In this study, large differences were observed between vegetative composition of canyon bottoms before and after intense fires and subsequent flood disturbances. Cover counts within designated stream segments that had different degrees and types of disturbance occur were shown to be independent of disturbance type.

In the context of occurrence of stand-replacing fires of unprecedented intensity and extent, examining the way that vegetation develops in the immediate aftermath and extended time period following disturbances will give insight to the development of novel ecosystems in the study area, and more broadly as climate intensifies on a global scale. Anecdotal evidence has suggested that certain types of vegetation cover have been slow or unable to reestablish. Specifically, riparian alder (*Alnus* sp.) and birch (*Betula* sp.) may be locally extirpated from Capulin; and yellow ladyslipper (*Cypripedium* sp.) and grape fern (*Botrychium* sp.) from Frijoles Canyon. In the seven years following the Las Conchas Fire, which was the main disturbance event in the temporal and geographic areas of the present study, forest cover has not recovered to pre-fire levels in either Frijoles or Capulin canyons. In Capulin Canyon, the 1996 Dome Fire had a large impact on forest cover proportions that is still clear in 2011, when the next set of aerial imagery is available. Comparing the trajectories of the recovery in the aftermath of the Las Conchas Fire between the two canyons may give insight into how single vs. multiple extreme disturbance events variously impact these two parallel systems, and also be compared to studies of riparian disturbance on broader regional and global scales.

The data analysis does not reflect more fine-scale observations that could be made were more samples analyzed for the years between 1991-2011, and 2012-2013. In the time period between 2014-2018, no large fires burned, and the most significant changes to cover type are a transition from bare ground to other vegetation that may or may not progress to forest in subsequent years. Continued monitoring of the area will help understanding and management of forest re-establishment, and to determine and project if the “Other vegetation” that is establishing on former bare ground will stabilize as forest cover or herbaceous vegetation.

Large stand-replacing fires and downstream removal of vegetation due to subsequent stormflow produce directly observable impacts on the landscape. While fire is a natural part of disturbance regimes, consistent processes of small-scale disturbance and revegetation would result in relatively even proportions of cover types over time (Naiman, Decamps, et al., 2005). Drastic shifts in cover type proportions thus likely indicate that the degree and extent of disturbance is unusual. The large differences in vegetative cover following fire and flood disturbances indicate that the system has been disturbed to a larger degree than usual, however it is possible that over a longer period, the systems will recover to their pre-fire dynamics. To draw consequential long-term conclusions about vegetation trajectories in the study area, repeated measurements should be taken to track the development of vegetation on site, and comparisons made to analogous situations regionally and worldwide.

The manual methodology employed, wherein buffered points were manually inspected, in some cases multiple times and by multiple sets of eyes, was time-intensive and in future research, more efficient methods may be used to quickly identify significant changes to landscape cover in satellite imagery. Use of supervised or unsupervised raster classification, or training artificial intelligence to interpret the landscape at a high level, as has been done in the Sierra Nevada Mountains of California, could make interpretation of both large and small study areas faster, more informative, and easier to implement on a larger scale (Parisa & Nova, 2020).

The manual methodology was selected for its simplicity, and the ability to specifically track crown size of forest vegetation. The human eye can make decisions about textural characteristics that a computer would not be able to if using a built-in classifier in ArcGIS Pro. In lieu of more advanced AI algorithms, making manual decisions about each point allowed the data to be reported with a high level of confidence and detail. However, further research could compare the results of computer-classified results with those generated by a human, to determine if the increased level of detail conferred by human decision-making is worth the additional investment of time and resources. In the short-term, employing human eyes to perform assessments such as the one in this study may be less resource-intensive because it does not require the development of an advanced computer program and algorithm, and the associated limitations. However, if the technology is developed and able to be deployed on a large scale across regions, this could allow for easier replicability of the methodology, and provide deeper insights and a wider basis for comparisons of large-scale disturbances.

Conclusion

The interpretation of remotely-sensed imagery of Frijoles and Capulin canyons within the Bandelier National Monument quantifies the clear disturbances caused by intense large-scale fires and floods throughout the area. While the fire regime appears to be unprecedented in the last several millennia according to preserved bog records, continuing to study the ongoing development and trajectory of vegetative changes will be essential to understanding how these disturbances affect ecosystems and their functions in the short and long term (Anderson, Jass, et al., 2008). In the absence of robust reference systems to which the disturbances in Bandelier can be compared, directly studying the area to understand its development and changes may be the most effective method of understanding local ecosystem dynamics.

Integrating analysis of vegetation dynamics with changing geomorphic conditions, and adapting management decisions to ongoing research findings and novel developments, will help to inform when and where intervention may be necessary, and track the effectiveness of actions taken to mitigate or reverse damages inflicted by fire disturbance. Quantifying the effects that disturbances have had on ecosystem services typically provided by the riparian systems, and comparing those results to the changes to landscape cover over time, may provide further insight for determining which types of vegetation are most important and effective for preserving the critical functions that the site has historically provided. Comparing vegetative cover directly with data sources such as digital elevation models and stormflow measurements may reveal additional patterns and correlations between riparian variables.

Riparian forests in the arid and semiarid Southwestern United States serve as critical habitat for a variety of plant and animal species, provide essential ecosystem services, and are of immense cultural and recreational value to human populations. Changes to climate conditions and disturbance regimes are triggering drastic shifts in vegetation and ecosystem dynamics, for which recovery trajectories are unclear. Continuing to study and adapt management to preserve functional ecosystems and habitats will be critical to ensure the continued existence of habitable landscapes that preserve native ranges of local flora and fauna and manage for novel ecosystems in the context of fundamentally or permanently altered dynamics.

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Appendix

This appendix includes data that was excluded from the general analysis. Wetted ground, shadow, and no data cells were excluded from analysis, as well as the lowermost reaches of Frijoles Canyon, which were anomalous due to the large proportions of bare ground in the delta, and deep shadows up to km 1.5 of the river. Tables 6-9 exhibit the raw cover proportions of Frijoles and Capulin canyons. Figures 10-11 present the proportions of all cover types, in Frijoles and Capulin canyons, by year.

Table A1. Frijoles Canyon raw cover proportions without channel designated.

Cover type	Year1992	Year2011	Year2014	Year2018
Forest	60%	56%	13%	14%
Other	14%	17%	34%	58%
Bare	8%	9%	29%	10%
Wetted	2%	0%	0%	0%
Shadow	17%	18%	25%	18%
Unidentifiable	1%	0%	0%	0%

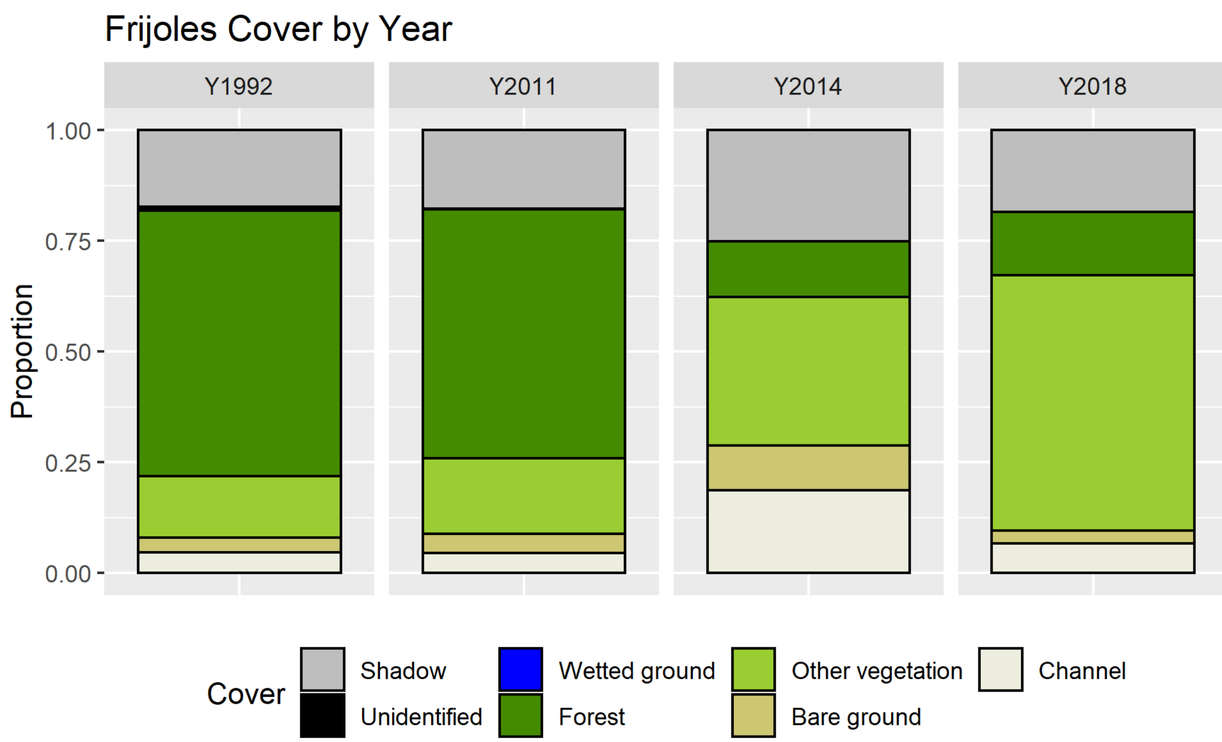

Figure A1. Proportions of vegetative cover observed in Frijoles Canyon overall by year

Table A2. Counts of sample points by cover type, disturbance segment, and year for Frijoles Canyon. Data represented graphically as proportions in Figure 5.

Year	Disturbance segment	Bare	Bare channel	Forest	Other vegetation
1992	A	10	1	209	18
	B	25	3	254	19
	C	1	0	419	17
	D	18	0	346	191
2011	A	26	1	205	39
	B	19	2	313	65
	C	4	0	364	8
	D	13	0	269	218
2014	A	21	14	144	68
	B	65	89	50	145
	C	46	200	0	226
	D	71	2	66	246
2018	A	8	2	154	94
	B	23	15	35	269
	C	8	40	13	430
	D	18	1	96	392

Table A3. Capulin Canyon raw proportions without channel designated.

Cover Type	Year1992	Year2011	Year2014	Year2018
Forest	47%	26%	6%	3%
Other	26%	47%	56%	71%
Bare	9%	17%	30%	21%
Wetted	0%	0%	0%	0%
Shadow	17%	11%	8%	5%
Unidentifiable	1%	0%	0%	0%

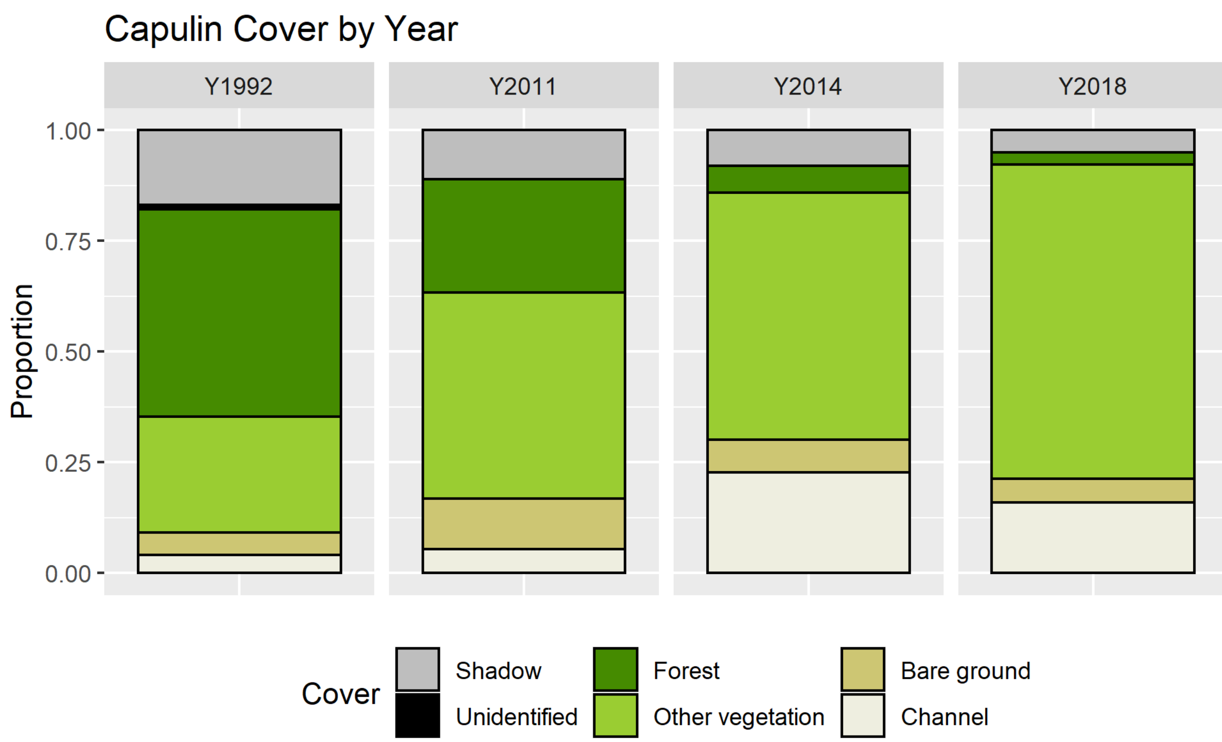

Figure A2. Proportions of vegetative cover observed in Capulin Canyon overall by year.

Table A4. Counts of sample points by cover type, disturbance segment, and year for Capulin Canyon. Data represented graphically as proportions in Figure 9.

Year	Disturbance segment	Bare	Bare channel	Forest	Other vegetation
1992	E	66	104	8	196
	F	1	2	377	20
	G	5	1	176	7
	H	33	2	332	239
2011	E	126	128	12	119
	F	67	14	221	177
	G	9	3	99	60
	H	30	0	158	477
2014	E	47	194	15	117
	F	42	189	5	245
	G	22	62	1	122
	H	51	66	5	575
2018	E	40	154	26	143
	F	21	137	14	320
	G	9	16	3	190
	H	19	8	9	706