Riparian System Responses to Fire and Flood Disturbance in Capulin Canyon, Bandelier National Monument, NM

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TABLE OF CONTENTS

List of Tables and Figures

Abstract

Disturbance is a driver of riparian ecosystem dynamics, and riparian areas are typically resilient to disturbance events. However, responses to extreme disturbances are not well-documented. We examined riparian responses to extreme disturbance in a montane canyon in northern New Mexico. Multiple severe fires burned extensive areas of live and dead organic material in the eastern Jemez Mountains between 1996 and 2011. Runoff, stormflow discharges, and sediment transport increased greatly after the fires. Capulin Creek flows through a canyon which drains part of the east Jemez Mountains and was severely burned in 2011 and severely flooded in 2013. Here we report results of repeat sampling of canyon bottom geomorphology and riparian vegetation along six transects in Capulin canyon, sampled before (in 2006) and after (2019) the 2011-2013 fire and flood disturbances. Sampling included repeat topographic and landform surveys along six monumented cross-sections to determine geomorphic change and woody vegetation cover, basal area, and stem density by species. Post-disturbance cross-sections revealed significant erosion and a shift from a trapezoidal channel/valley geometry to a wider, braided geometry. We found a dramatic decrease in riparian vegetation between 2006 and 2019. For example, the mean total basal area (per transect), in 2006 was 17.35 ± 12.92 m^2/ha (mean ± standard deviation); whereas in 2019 it was only 0.32 ± 0.49 m^2/ha. *Pinus ponderosa*, *Alnus oblongifolia*, *Acer negundo,* and *Juniperus scopulorum* accounted for the majority of the basal area pre disturbance, with mean basal areas of 10.15 m^2/ha, 4.57 m^2/ha, 1.66 m^2/ha, and 1.55 m^2/ha, respectively. Post-fire and -flood, mean basal areas for *P. ponderosa, A. oblongifolia,* and *J. scopulorum* were all 0 and for *A. negundo* was 0.23 ± 0.51 m^2/ha. Other relatively abundant species in 2019, were *Quercus gambelii* and *Salix lutea* with mean basal areas of 0.075 ± 0.17 m^2/ha and 0.01 ± 0.03 m^2/ha, respectively. Our results will help to inform future management decisions regarding potential restoration actions in highly disturbed canyon ecosystems at Bandelier National Monument and elsewhere around the West.

Introduction

Riparian ecosystems are ecologically important because they are areas of concentrated biodiversity (Naiman et al. 2005) and can support rare or regionally restricted species (Sabo et al. 2005 in Jacobs et al. 2015). This is especially apparent in Arizona and New Mexico, where 80% of all vertebrate species spend half their lives in riparian corridors and many of these are classified as threatened or endangered on state or federal listings (Baker et al. 2004). Also, riparian areas provide ecosystem services including flood control, sediment retention, water purification, and their contribution to productive wildlife populations and fisheries (Naiman et al. 2005).

Disturbance can influence the structure and function of riparian systems. In the Southwest, riparian systems are often influenced by disturbances such as flooding and wildfire, which can alter stream geomorphology and riparian vegetation (Baker et al. 2004). Bendix and Hupp (2000), emphasize flood regimes as an important attribute of fluvial landforms. Additionally, periodic fire promotes stream alteration (Legleiter et al. 2003). As a result, altered stream geomorphology can influence vegetative species diversity, structure, and function during postfire and post-flood periods (Gresswell 1999; Smith et al. 2009; Kominoski et al. 2013). However, the magnitude of post-disturbance change varies with position in a watershed, disturbance intensity, and the spatial and temporal extent of the disturbance(s) (Legleiter et al. 2003; Parsons et al. 2005; Romme et al. 2011).

Flooding is a hydrogeomorphic process whose influence on riparian areas can depend on flow regime (Scott et al. 1997; Steiger et al. 2005). Flow-induced sediment deposition, scouring, and vegetation removal can create new space for riparian species to grow, especially pioneer species such as willows and cottonwoods (González et al. 2019). Subsequently, vegetation may create roughness, slowing down the flow of water and leading to more sediment deposition (Bendix and Hupp, 2000). In addition, lack of flow can lead to low water availability and subsequent dieback or mortality within stands of woody riparian vegetation (Scott et al. 1999; Shafroth et al. 2000; Cooper et al. 2003).

Fire and floods differ in their influence on riparian vegetation. Bendix and Cowell (2013) explained how their findings demonstrated the greater importance of fire over flooding in terms of its impact on the process of overstory regeneration. They contrasted the higher magnitude of the pulses in regeneration caused by fires versus floods, in that fires can be stand-clearing, whereas floods have a patchier spatial distribution in terms of their disturbance influence.

The combination of fire and floods influences riparian vegetation as well. Typically, this is exemplified in fire's relationship to erosion and then erosion and runoff's contribution to high flows in riparian areas. Mechanisms by which fire can influence erosion and runoff is via the creation of a hydrophobic layer of soil and reduced roughness. High severity fire can create a hydrophobic top layer of soil and kill rooted vegetation in burn areas (Cannon and Reneau 2000). Thus, when areas that were severely burned experience heavy rains, there is the potential for the hydrophobic top layer of soil to become mobile, moving downhill as erosion (Veenhuis, 2002). Dwire and Kaufman (2003) reported that erosional events contribute sediment, which is transported by water and distributed along banks or on top of pre-existing vegetation. This creates new space for vegetation to establish, restarting successional dynamics.

Fire affects riparian vegetation through a complex process with multiple biotic and abiotic factors at play. Fires cause mortality of vegetation and stimulate regeneration (Stromberg and Rychener 2010). Halofsky and Hibbs (2009) found that in the understory, an increase in percent exposed mineral soil correlated with an increase in seeder species such as conifers, whereas lower percent exposed mineral soil correlated with an increase in sprouting species. Smith et al. (2009) found that during the first year postfire, which also happened to be during the peak of a drought, cottonwood resprouts on the Middle Rio Grande, NM had a low survival rate. Additionally, they found that in postfire areas, depth to groundwater was an influential factor. During drought, depth to groundwater influenced native cottonwoods and willows, which require high water tables and have less of a chance of surviving where water tables are relatively deep.

Resprouting is a common trait exhibited by riparian trees in response to fire. In their reviews of the ecological consequences of fire in riparian areas, Pettit and Naiman (2007), Dwire and Kaufman (2003), and Bock and Bock (2014) suggested that resprouting could be an adaptation to fire. Pettit and Naiman (2007) and Bock and Bock (2014) explained that resprouting by riparian species was also an adaptation to other disturbances like flooding. Halofsky and Hibbs' (2009) discussion of regeneration via resprouting emphasized the ecological benefit of the resprouting process in stream channel health, in that newly sprouted vegetation fortifies banks, prevents erosion, and provides shade. Smith et al. (2009) confirmed that resprouting was a postfire regeneration strategy of both native and exotic riparian species. Contrasting native Fremont cottonwood (*Populus fremontii)* and non-native tamarisk (*Tamarix* sp.), survival strategies in their research on the San Pedro River in Southeastern Arizona, Stromberg and Rychener (2010) discussed how vigorous resprouting was the primary strategy of tamarisk, postfire. Meanwhile, older, taller Fremont cottonwood exhibited traits like thick bark to survive fire over resprouting. Discussing the relationship between sprout rate and age class, they also observed that Goodding's willow (*Salix gooddingii*), did not exhibit the commonly observed trend amongst riparian species of a reduced sprouting rate with an increase in age. In their research on tree establishment following severe fire in the Sespe Wilderness in Southern California, Bendix and Cowell (2013) discussed how white alder (*Alnus rhombifolia*) went years without re-establishing via resprouting or a robust seedbed unlike native cottonwood, willow, and oak species that re-stablished sooner during the post-fire period. Ellis (2001) found that Rio Grande cottonwood regenerated after severe fire via resprouting. Smith and Finch (2018) found that fire-caused topkill was a common cause of death for mature cottonwoods along the Middle Rio Grande River. Reproductive strategies including root sprouts, epicormic sprouts, and root suckers could be utilized by cottonwoods, but only under favorable conditions because tamarisk and Russian olive were superior in postfire regeneration compared to cottonwoods.

To improve our understanding of riparian responses to extreme disturbance, we studied the response of a canyon bottom system in north-central New Mexico to fire and flood disturbance. In 2011 the Las Conchas fire burned more than 150,000 acres (194 km^2) in the Eastern Jemez Mountains of

North-Central New Mexico, including within the boundaries of the Bandelier National Monument, an area of significance because of its rich natural history. Capulin Creek is an intermittent stream that flows through the Santa Fe National Forest, into Bandelier National Monument, and after leaving the monument, joins the Rio Grande River. During the Las Conchas Fire, sixty percent of Capulin Creek's riparian zone burned at high severity (Jacobs et al. 2015; Figures 3 and 4). Even though specific details on the magnitude and nature of erosion impacting Capulin Creek are not known, channel erosion was reported during the four years following the Las Conchas fire (Stephen Monroe, NPS, Southern Colorado Plateau Network, hydrologist, written communication 28 May 2015 in Kellerlynn 2015). Additionally, erosion harmed macroinvertebrate and fish populations in Capulin Creek (Cannon and Gartner, 2005 in Tillery et al. 2011, Dyer et al. 2016, Stumpf, 2016). All fish were eradicated from Capulin Creek by these disturbances (Jacbos et al. 2015). Understanding how Capulin Creek has transitioned would provide land managers with a more accurate understanding of its overall ecological health. This understanding could better inform future management decisions for purposes like aquatic species habitat improvement, protection of archaeological sites, and recreation. Additionally, understanding the recovery and dynamics of riparian vegetation along Capulin Creek could help with planning and prioritize passive or active restoration efforts that aim for a revegetated riparian area along Capulin Creek to help mitigate threats to human interests downstream.

Previous research on Capulin Creek existed for us to use as a basis of comparison to gauge Capulin Creek's response to the extreme disturbance of a mixed severity fire and subsequent flooding. In 2006, eleven stream cross-sections from this reach were sampled by Scott and Reynolds (2007) as part of larger ongoing research by the National Park Service's Southern Colorado Plateau Inventory and Monitoring Network. Therefore, pre-Las Conchas fire basal area, species lists, percentage cover by species, and stream geomorphic data along these cross-sections were available (Scott and Reynolds 2007). Our first research question was: Has extreme disturbance changed the topography of Capulin Creek within the sample reach? Our second question was: In the aftermath of mixed-severity fire and flooding, is vegetation recovering and via which life history strategies? We hypothesized that the topography of Capulin Creek would be altered post-disturbance. Concerning vegetation, we hypothesized that prior to the Las Conchas fire, there would be higher total basal area and stem density of seeder species and in relatively larger size classes compared to resprouting species. Conversely, after disturbance, there would be lower total stem density and basal area of seeder species compared to resprouting species.

Study Area

Our sample reach was located on Capulin Creek, within the boundaries of Bandelier National Monument. This reach was previously sampled in 2006 by Scott and Reynolds and starts approximately 7.50 kilometers Northwest of Capulin Creek's confluence with the Rio Grande (Figure 1). Bandelier is located in Los Alamos County, NM, in the north-central part of the state, on the Pajarito Plateau, an area formed by ancient eruptions of the Valles Caldera (Figure 2). Several southeast-trending, steep-walled canyons were carved out of the Pajarito Plateau via erosion, including Frijoles, Alamo, and Capulin canyons, which are within the boundaries of the monument (Muldavin et al. 2011 in Jacobs et al. 2015). Within Capulin Canyon runs Capulin Creek, whose headwaters are on the Santa Fe National Forest and Valles Caldera National Preserve (Jacobs et al. 2015). Its watershed area is approximately 51 square kilometers and it ranges from 1,628 to 2,652 meters in elevation (Cannon and Reneau 2000). The climate of the area is marked by cold, dry winters and warm wet summers (Jacobs et al. 2015). Intra-annual temperatures range from -7.60°C to 30.20°C. Average annual precipitation for Los Alamos, the nearest weather station, for the period 1902-2016, was 464.31 millimeters (Western Regional Climate Center, 2019). Though precipitation falls in the form of snow during the winter, the majority of precipitation arrives as rain during monsoonal events, occurring July through September (PRISM Climate Group, 2019). The Capulin Canyon study site's vegetation community was classified as canyon riparian and ponderosa pine savanna or forest prior to the Las Conchas fire (Muldavin et al. 2011 in Jacobs et al. 2015). Bandelier is historically significant because of its rich collection of prehistoric archaeological sites (Jacobs et al. 2015).

Methods

Topographic surveying

We relocated the endpoints of the transects (rebar) originally sampled by Scott and Reynolds (2007). Where rebar had been removed by the 2013 flood, we established a new endpoint. Topography along each transect (between the endpoints) was surveyed using a total station in 2006 (Scott and Reynolds 2007). In 2019, we re-surveyed the topography using a Real Time Kinematic, Global Positioning System.

Establishing the Transect Line

First, we located the endpoints of the transects originally surveyed and monumented by Scott and Reynolds (2007). Scott and Reynolds originally sampled eleven transects. We re-sampled the odd numbered ones (six in total). The surveyed transect often extended far outside of the riparian areas and into the uplands on each side of the valley. If we deemed it unnecessary to collect data for the entire surveyed transect length, then we abbreviated the length for gathering data. Zero always started on the left headpin side of each transect. Scott and Reynolds (2007) sampled eleven transects. In 2019, we re-sampled the odd numbered transects (6 transects). Transect lengths were: 38.20 meters, 31 meters, 23.20 meters, 30 meters, 35.30 meters, and 27.70 meters.

Line Intercept

To delineate the geomorphic surfaces along the transect, we used the line intercept method (Bonham 1989; Elzinga et al. 1998). We noted a start and stop distance for each surface along the line. Geomorphic surfaces included: channel (water), depositional bar, bank, cut bank, lower flood plain, upper flood plain, boulder levee, overflow channel, secondary channel, lower terrace, terrace, upper terrace, upland, and slopes to or from any of these surfaces to others. In addition, we applied the line intercept method to estimate the cover of woody species along each line. Specifically, we noted the start and stop distances of live leaves or needles, by species and size class, that overlapped the transect line, including above it (Table 1 lists the observed woody species). Tree size classes were: seedling (< 1m tall), sapling (≥ 1 meter tall and ≤ 2.5 centimeters DBH), pole (2.5-15 centimeters DBH), overstory (≥ 15 - 50 centimeters DBH), and ≥ 50 centimeters DBH). Shrub height size classes were < 2 meters and ≥ 2 meters.

Belt Transect

We measured all trees that were at least 1.37 meters tall (breast height) within five meters of the line on both sides of the line – i.e., within a 10-meter wide belt. We measured the diameter of every stem at breast height (DBH), noted the species of the stem, and noted the quadrant in which it occurred (upstream of the line on right bank side; upstream of the line on left bank side; downstream of the line on right bank side; downstream of the line on left bank side). We included stems that extended partially beyond 5 meters from the line if 50% or more of the stem was within the 5-meter distance.

Data Analysis

Cross-sectional Area

We entered the 2006 and 2019 topographic survey data into WinXSPRO Cross-Section Analyzer Version 3.0 software (USDA Forest Service, Rocky Mountain Research Station, Stream Systems Technology Center, Fort Collins, CO), to compare 2006 cross-sectionional areas with 2019 ones, using the Compare Areas tool.

Percent Cover by Species

With the line intercept data from Scott and Reynolds' 2006 data and our 2019 data, we calculated percent cover by woody species and size class. First, we totaled the start and stop distances of each species and size class on each transect. Then we divided these distance totals for each species and size class by the total length of the transect. Also, we calculated the mean and standard deviation for each species' percent cover. Finally, we performed a paired t-test in MS Excel on the total percent cover values per transect from 2006 and 2019.

Basal Area

With belt transect data from 2006 and 2019, we calculated basal area for each tree species in meters^2/hectare (Avery and Burkhart 2002). Also, we calculated the mean and standard deviation of each species' basal area from the six transects. Then we performed a paired t-test in MS Excel on the basal area values per transect from 2006 and 2019.

Stem Density

With belt transect data from 2006 and 2019, we calculated stem density (stems/hectare) in each year, dividing the number of stems by the area sampled in hectares. Then we performed a paired t-test in MS Excel on the stem density values per transect from 2006 and 2019.

Results

Cross-section Changes

Transect 1 had the largest difference in area between 2006 and 2019 of all the transects, with erosion of 33 meters^2. The left bank floodplain surface eroded between 2006 and 2019. The 2019 thalweg migrated approximately 6 meters towards the left bank side and was 8 centimeters higher in elevation compared to 2006 (Figure 5).

Transect 3 is the only transect that decreased in area between 2006 and 2019. 0.50 meters^2 of material was deposited since 2006. Erosion of terrace and floodplain surfaces occurred on the left bank side of the transect and the thalweg migrated approximately 6 meters towards the left bank side. In 2019, the thalweg elevation was 45 centimeters higher than in 2006 (Figure 6).

Transect 5 eroded 6.90 meters^2 between 2006 and 2019. The thalweg migrated approximately 7 meters towards the right bank side. In 2019, the thalweg was 38 centimeters lower in elevation than in 2006 (Figure 7).

Transect 7, formerly trapezoidal-shaped, eroded 16.20 meters^2 between 2006 and 2019. The thalweg migrated approximately 8 meters towards the right bank side and in 2019, and was 9 centimeters higher in elevation than in 2006 (Figure 8).

Transect 9 was also formerly trapezoidal in shape. It eroded 20.60 meters^2 between 2006 and 2019. In 2019, the thalweg migrated approximately 13 meters to the right bank side and was 10 centimeters lower in elevation than in 2006 (Figure 9).

Transect 11 eroded 15.70 meters^2 between 2006 and 2019. The thalweg migrated approximately 5 meters to the left bank side and was 21 centimeters lower in elevation than in 2006 (Figure 10).

Percent Cover by Species

Percent cover values per transect from 2006 and 2019 were significantly different (p = 0.0002). Mean percent cover by species between 2006 and 2019 differed (Table 2). Also, relative percent cover in 2006 and 2019 varied by transect (Figures 11-16).

Basal Area

Total basal area per transect was significantly different between 2006 and 2019 ($p = 0.012$). Total mean basal area per transect was 17.04 m^2/hectare ± 12.63 (basal area ± standard deviation) in 2006 and in 2019 it was 0.32 m^2/hectare ± 0.49. Mean basal area by species differed between 2006 and 2019 (Table 3). Also, relative basal area by size class per transect varied between 2006 and 2019 (Figures 17-22).

Density

Stem density per transect was significantly different between 2006 and 2019 ($p = 0.00064$). Mean stem density by species and size class differed between 2006 and 2019 (Table 4).

Discussion

Topographic survey results of the six stream cross-sections reflected significant changes in the topography of each transect, which provide insight about vegetation changes along the sample reach. Every cross-section except transect 3 increased in area due to erosion, which confirmed our first hypothesis that the topography of Capulin Creek would be altered post-disturbance. Sediment and biomass from floodplain, channel shelf, terrace, and terrace riser surfaces were excavated by disturbance. This excavation of sediment and biomass created open, more easily moistened surfaces suitable for recruitment of pioneer species like *S. gooddingii* and *P. deltoides* (Shafroth et al. 1998; Lytle and Merrit, 2004). Meanwhile *A. negundo* and *Q. gambelii* recruited along terrace surfaces and results sugeest that they were utilizing a resprouting life history strategy in the aftermath of the Las Conchas Fire. Species that were not detected in 2006 along the transect, but were detected along the 2019 transect were *P. deltoides*, *S. goodddingii*, *S. exigua*, *S. lutea*, and *Parthenocissus vitacea*, demonstrating that these pioneer species were able to establish in the aftermath of disturbance (Table 2).

During the Las Conchas fire, the majority of the Capulin Canyon riparian area burned at mixed severity, which likely drove the dramatically lower observed total basal area, percent cover, and stem density on all transects in 2019 relative to 2006. Bendix and Cowell (2010) did similar post-disturbance research in a riparian area of the Sespe Wilderness in Southern California and had similar results. They found significantly less basal area of *P. fremontii*, *Q. dumosa*, *Salix* spp., and *A. rhombifolia* post-fire, which all had relatively high basal areas pre-fire. Meanwhile, in our research, *P. ponderosa*, *A. oblongifolia*, and to a lesser extent *J. scopulorum*, had relatively high basal areas pre-fire, but were not detected on transects in 2019 (Table 3). As it pertains to *P. ponderosa* and *J. scopulorum*, these results support our hypothesis that prior to the Las Conchas fire, seeder species would account for a higher total stem density and basal area compared to resprouters, but after disturbance there would be lower total basal area and stem density of seeders compared to resprouters. It should be noted that life history strategies for *Alnus* species include sprouting (Fryer 2014). Therefore, labeling *A. oblongifolia* as exclusively a seeder (in alignment with our hypothesis), is inaccurate. Bendix and Cowell (2010) found that *P. fremontii*, *Q. dumosa*, and *Salix* spp., eventually re-established, but *A. rhombifolia* did not. They explained that *A. rhombifolia*, lost seed sources in the fire and the topography of the research area did not encourage seed dispersal from farther seed sources. The similarly rugged terrain of Capulin Canyon also makes it difficult for species whose local seed sources burned in the Las Conchas Fire. This could explain why we did not detect any *A. oblongifolia* and *J. scopulorum* in 2019 despite the fact that these species accounted for a significant portion of the basal area and stem density in 2006. We did observe a potential seed source for *P. ponderosa* in the uplands near transect 5. Future repeat sampling would help in determining if these *P. ponderosa* trees are viable seed sources.

The lack of detection of *P. ponderosa*, *J. scopulorum*, and *A. oblongifolia* may be indicative of the effects of mixed severity fire. During the Las Conchas Fire the majority of the riparian zone burned at high severity (Stumpf and Monroe, 2012 in Jacobs et al. 2015). Seed sources and the seed bank may have burned to the extent that they were rendered useless in contributing to the post-fire regeneration strategy of these two species. In their 2006 sampling of the same transects, Scott and Reynolds (2007) noted the relatively high cover of *A. oblongifolia* poles and that this likely represented recruitment after the 1996 Dome Fire, which burned a large portion of the Capulin Creek Watershed (Jacobs et al. 2015). Differences in the recruitment of *A. oblongifolia* during both postfire eras could be due to the fact that the Dome Fire burned at a lower intensity than the Las Conchas fire did in the portion of Capulin Creek where the sample reach is located (Cannon and Reneau 2000). There is a lack of literature on the fire effects on *A. oblongifolia* and the life history strategies of *A. oblongifolia*, making comparison of our results difficult.

Regarding mean basal area and stem density, we observed an increase in *P. deltoides*, *S. gooddingii*, and *Q. gambelii* in 2019 relative to 2006 (Tables 3 and 4). In their research on Upper San Pedro River riparian tree regeneration, Stromberg and Rychener (2009) found significant decreases in the stem density and basal area of *P. fremontii* and *S. gooddingii* after fire. However, the composition of pre-fire transects in their experiment were different from ours, in that theirs were dominated by *P. fremontii* and *S. gooddingii* and not all stems were killed by the fire. Thus, larger, more mature trees were contributing to postfire basal area and stem density values in their study. Along our transects, mixed severity fire eliminated most of the biomass and created a clean slate that was beneficial for *P. deltoides* and *S. gooddingii* recruitment. This could explain why 2019 mean stem density increases were 813% for *P. deltoides* and 1,523% for *S. gooddingii* and mean basal area increases were 25% for *P. deltoides* and 490% for *S. gooddingii* relative to 2006.

Concerning *A. negundo*, results from our sampling provide us with a better understanding of its life history strategy. *A. negundo* accounted for the highest mean percent cover, basal area, and stem density in 2019, despite its considerable negative percent change in all measurements in 2019 relative to 2006. *A. negundo* accounted for a significant portion of the stems measured for belt transects and multiple stems were often emanating from a common individual, consistent with resprouting. On Oak Creek in Northern Arizona, Kolb et al. (1997) observed that most of the *A. negundo* at the site were resprouts from flood-damaged mature stems and suggested the importance of asexual reproduction for *A. negundo* in frequently flooded areas. Galuszka and Kolb's (2002) work on West Clear Creek, Arizona found almost no regeneration of *A. negundo* seedlings after severe flooding. Unfortunately, there is a lack of literature on fire effects on *A. negundo* to which we can compare our results to (Smith and Finch, 2018). Resprouting from mature individuals may be the primary life history strategy for *A. negundo* in response to disturbance. In order for us to have a better understanding of this, more research should be done on fire effects on *A. negundo*.

The fact that *Q. gambelii* accounted for a large amount of percent cover, basal area, and stem density in 2019, suggests that it also resprouted. When high severity fire burns, *Q. gambelii* vigorously resprouts and often dominates post-fire vegetation (Kaufmann et al. 2016). This was paralleled in our results, as in 2019, *Q. gambelii* saplings and poles increased in stem density by 157% and 605%, respectively, relative to 2006 (Table 4). Having evolved in the Southwest in fire-adapted *P. ponderosa* ecosystems, *Q. gambelii* persists in burned areas and likely will remain in the region for the foreseeable future (Abella and Fulé, 2008).

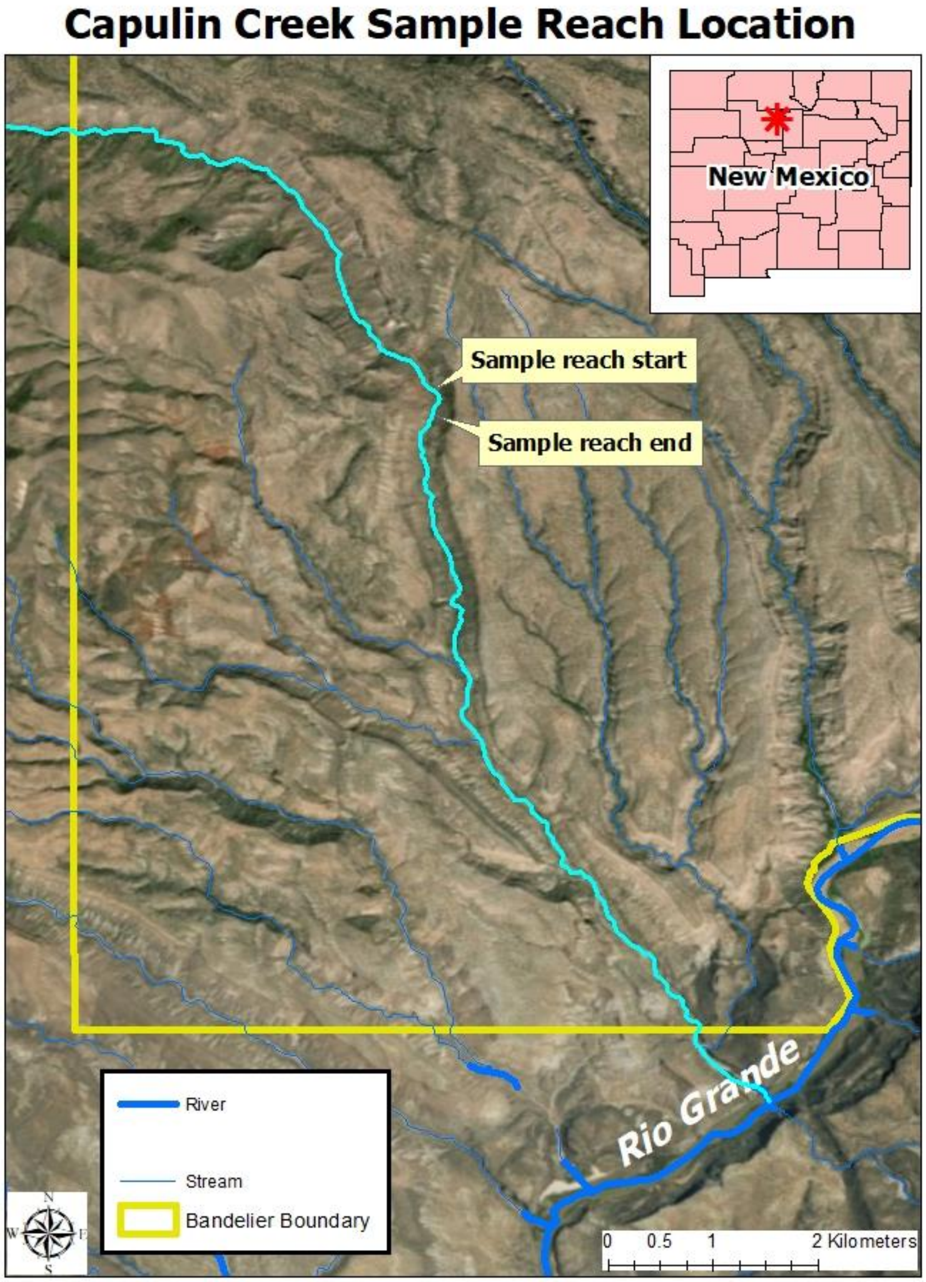
Our results suggest that mixed severity fire burning the same area more than once over a relatively short period of time may influence changes in species composition. In Capulin Canyon's case, "relatively short," means fifteen years. Changes in fire interval influence the amount of time it takes for vegetation to recover (Leonard et al. 2017). Repeat fires change vegetation communities, soil conditions, frequency, occurrence, and dispersal of floods, and erosion events, manipulating channel and floodplain geomorphology (Poff and Ward, 1990 in Gresswell, 1999). In their research on the fire history of the Jemez Mountains, Swetnam et al. (2016) found that for ≥25% fires in the region during the pre-fire suppression era (1681-1860), the mean return interval was 27.83 years and the fires were often low severity ones. Both the 1996 Dome Fire and the 2011 Las Conchas Fire burned significant portions of the Capulin Canyon watershed (Jacobs et al. 2015). Our results suggest that composition shifted from an *A. oblongifolia*, *P. ponderosa* and *J. scopulorum* dominated community to one featuring *A. negundo* and *Q. gambelii* as well as shrub species like *Forestiera pubescens* var. *pubescens*, and *Rhus trilobata*. Additionally, *Salix exigua* and *Salix lutea*, which were not detected in 2006, had notable mean percent cover values in 2019, suggesting a composition shift featuring these pioneer species.

Although we observed recruitment of pioneer species, namely *P. deltoides* and *S. goodddingii*, we did not see as dense of post-disturbance recruitment as previously reported in the literature. Populus seedlings and saplings between 1 and 4 years old have been reported to have densities ranging from ca. 10,000-3,500,000 individuals/hectare (Lytle and Merritt 2004). In our research, the mean density of *P. deltoides* was 51.75 individuals/hectare. One reason for this notable difference in seedling density could be that both species lack nearby seed sources. Based on sampling farther up in the watershed, we know that viable seed sources do not exist upstream of this reach of Capulin Creek (P. Shafroth, personal communication). The nearest location is most likely the Rio Grande River, which is approximately 7.50 km downstream. Thus, we suggest that the seedlings we observed likely originated from wind-dispersed seeds. The rugged topography of Capulin Canyon could contribute to relatively low density of wind-dispersed seed, but we did not measure this as part of this study.

In summary, our re-sampling of transects on Capulin Creek highlighted the influence of disturbance, namely mixed-severity fire and ensuing floods on the topography and vegetation along the creek. The absence of *P. ponderosa*, *J. scopulorum*, and *A. oblongifolia* may be indicative of mixed severity fire, in that seed sources could have been eliminated during the Las Conchas fire (the one exception being *P. ponderosa*, which has a nearby seed source, but that has not proven viable yet). *P. deltoides* and *S. gooddingii* appeared to be responding to the significant changes in the topography of transects, recruiting on floodplain and channel shelf surfaces. Meanwhile, *Q. gambelii* and *A. negundo* resprouted on terrace and slope to terrace surfaces, demonstrating a resprouting strategy that may be a response to mixed-severity fire, though literature pertaining to fire effects on *A. negundo* is lacking. Additionally, the relatively brief amount of time between two mixed severity fires, the 1996 Dome fire and 2011 Las Conchas fire, may have caused shifts in the composition of Capulin Creek. Lastly, less dense recruitment of *P. deltoides* and *S. gooddingii* than reported in the literature and a lack of upstream seed sources suggests that observed seedlings in 2019 likely originated from wind-dispersed seeds.

Management Implications

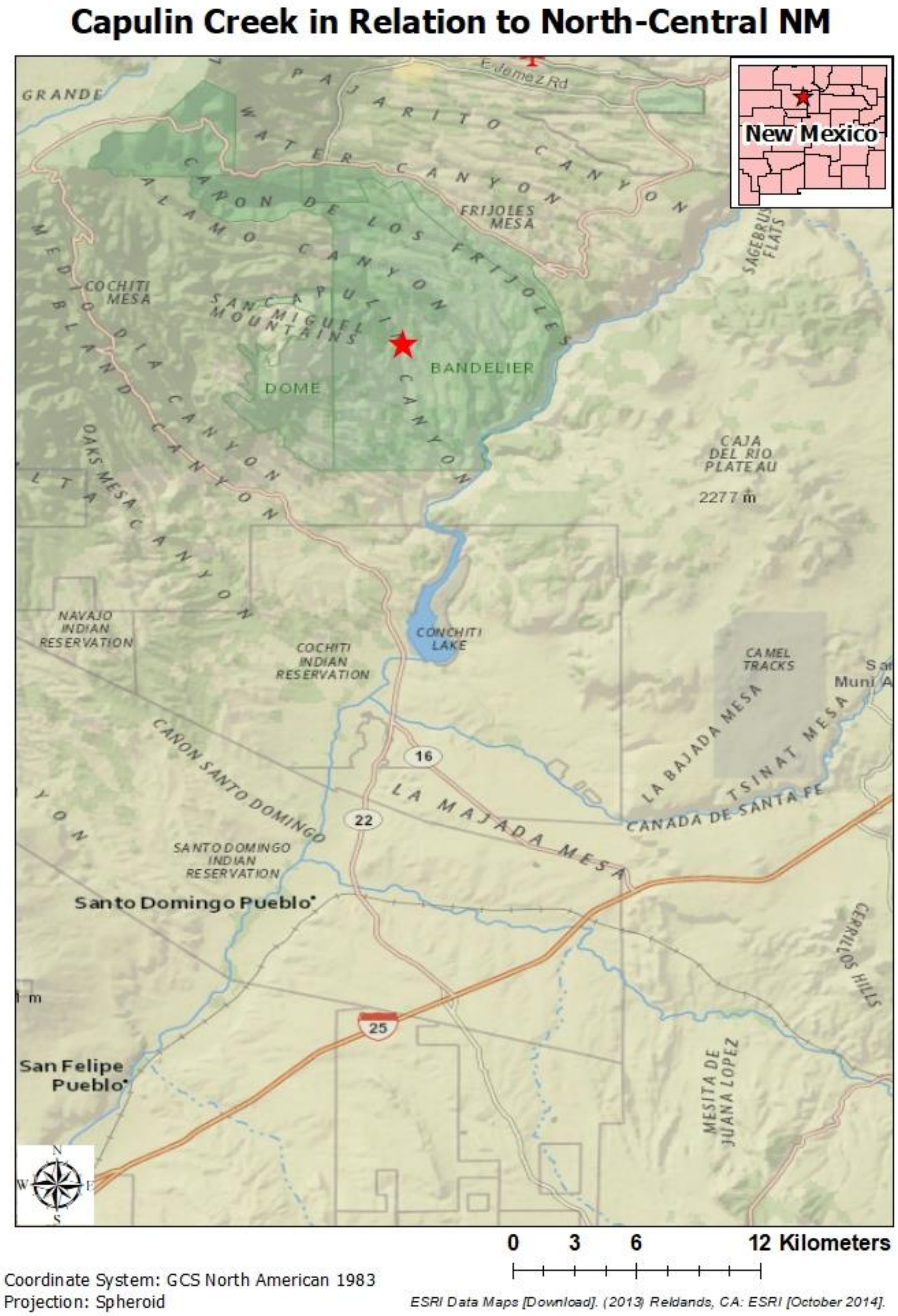
Bandelier National Monument resource managers should take the results of this re-sampling of the Southern Colorado Plateau Network Capulin Creek reach into consideration when planning restoration efforts. First, the topography of the sample reach of Capulin Creek has changed significantly and the relationship of stream morphology and riparian vegetation should be taken into consideration in future restoration efforts. Second, resource managers should consider the fact that the structure and composition of riparian vegetation shifted post-disturbance. In 2006, this area was dominated by *A. oblongifolia*, *P. ponderosa*, *J. scopulorum*, and *Q. grisea* in larger size classes. Conversely, in 2019 this area was dominated by *P. deltoides*, *S. gooddingii*, *A. negundo*, and *Q. gambelii* in sapling and pole size classes. Restoration goals pertaining to specific species should incorporate consideration of these shifts in determining whether or not to take passive or active approaches. The apparent lack of viable seed sources for some species (e.g., *A. oblongifolia*) may suggest a need for active restoration measures. Meanwhile, the presence of pioneer species like *P. deltoides* and *S. gooddingii*, which regenerated naturally, may suggest that passive approaches would better fit this reach of Capulin Creek.



Coordinate System: GCS North American 1983
Projection: Spheroid

ESRI Data Maps [Download]. (2013) Reldands, CA: ESRI [October 2014].

Figure 1: Map of study area within Bandelier National Monument.


Capulin Creek in Relation to North-Central NM

Coordinate System: GCS North American 1983
Projection: Spheroid

ESRI Data Maps [Download]. (2013) Reldands, CA: ESRI [October 2014].

Figure 2: Map of Bandelier National Monument in relation to the rest of North-Central New Mexico. Red star on larger map signifies the sample reach of Capulin Creek.

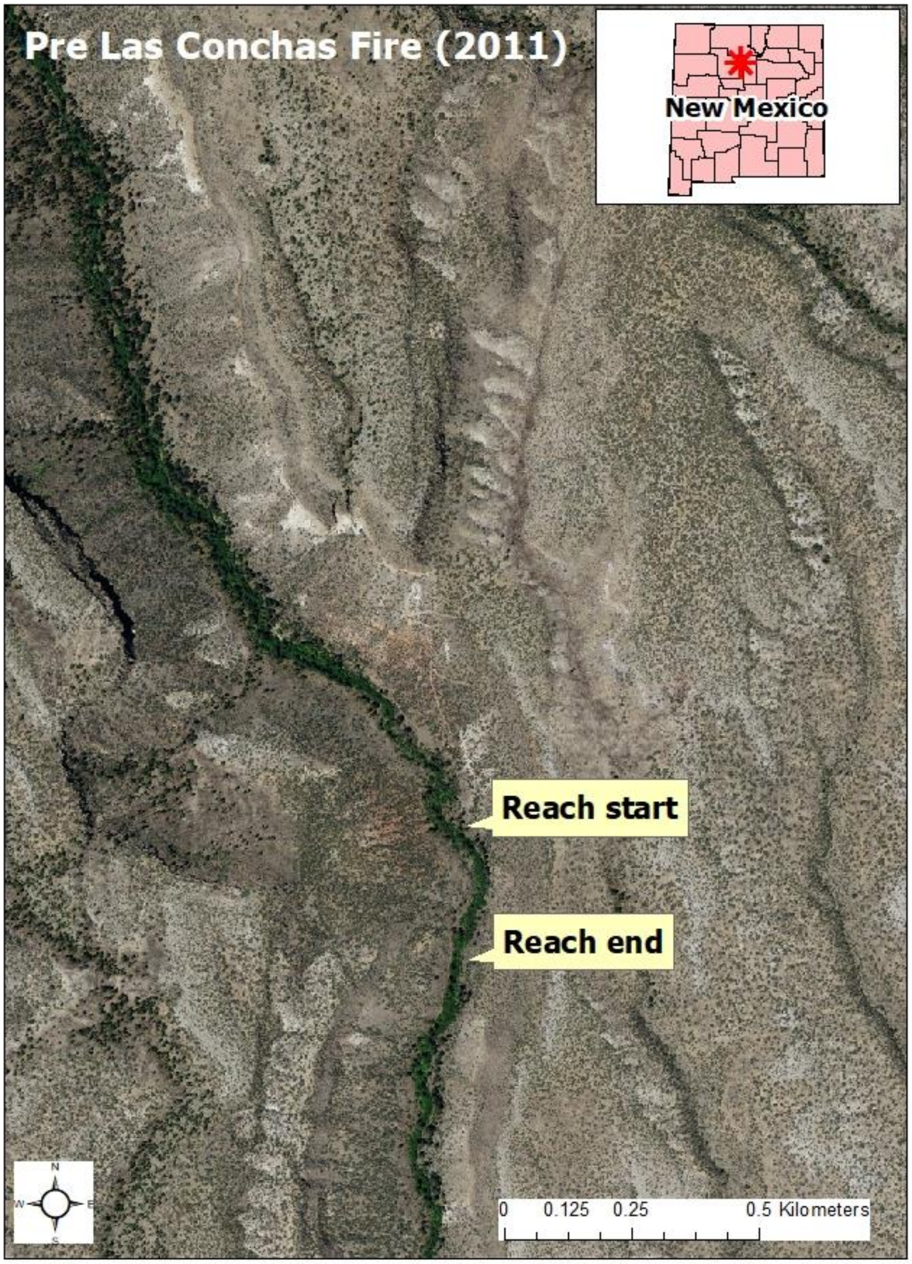


Figure 3: Aerial imagery of study area prior to the Las Conchas fire. Note the dense vegetation growing

along the stream.

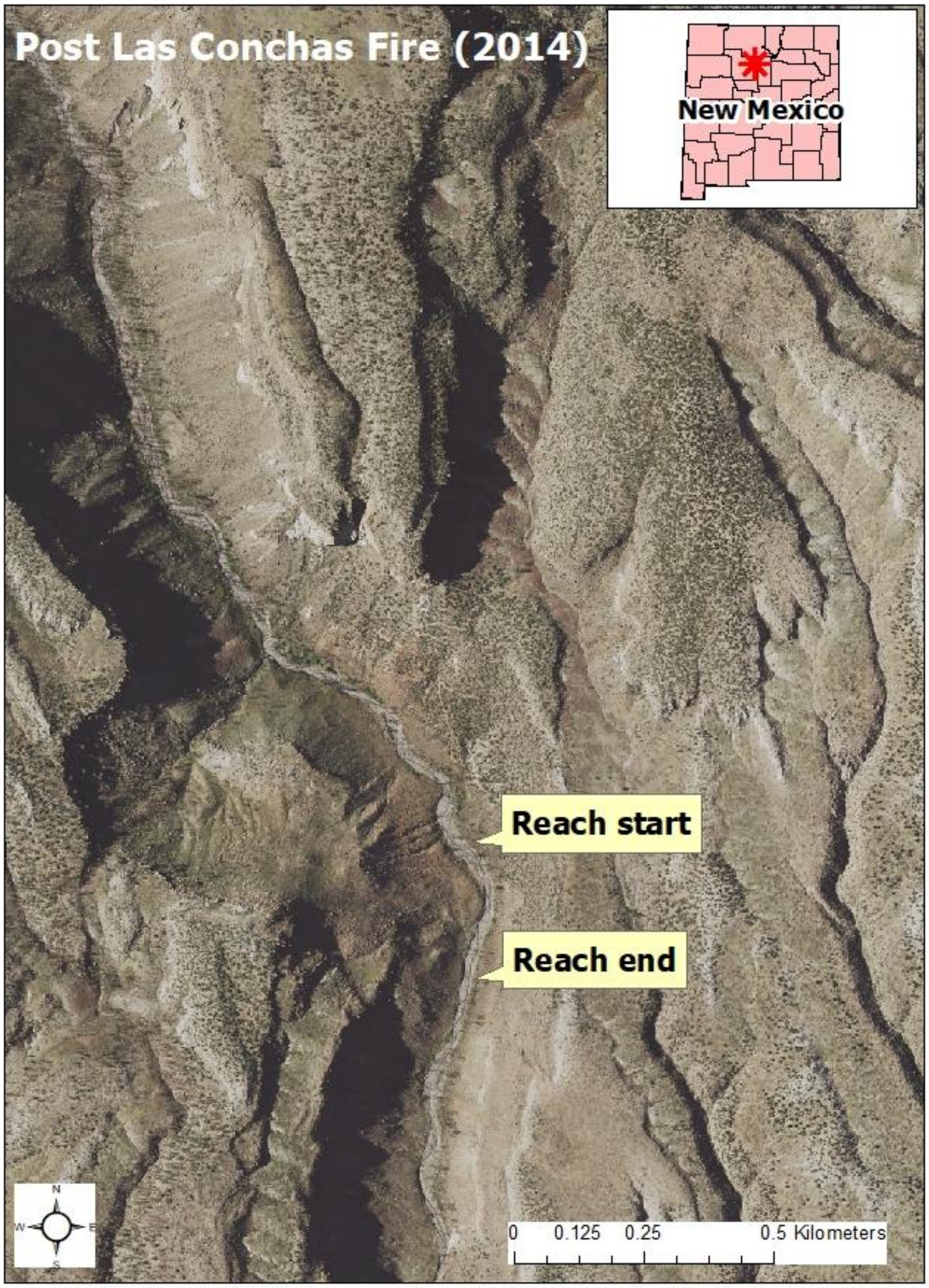


Figure 4: Aerial imagery of study area after the Las Conchas fire. Note the lack of vegetation growing along the stream.

Species	**Family**	**Life Form**
Acer negundo	Aceraceae	Tree
Alnus oblongifolia	Betulaceae	Tree
Foresteria pubescens var. *pubescens*	Oleaceae	Shrub
Juniperus scopulorum	Cupresaceae	Tree
Parthenocissus vitacea	Vitaceae	Vine
Pinus ponderosa	Pinaceae	Tree
Populus deltoides	Salicaceae	Tree
Ptelea trifoliata	Rutaceae	Shrub
Quercus gambelii	Fagaceae	Tree
Rhus trilobata	Anacardiaceae	Shrub
Rosa woodsii	Rosaceae	Shrub
Salix exigua	Salicaceae	Shrub
Salix gooddingii	Salicaceae	Tree
Salix irrorata	Salicaceae	Shrub
Salix lutea	Salicaceae	Shrub

Table 1: List of all woody species from field sampling, including life form type.

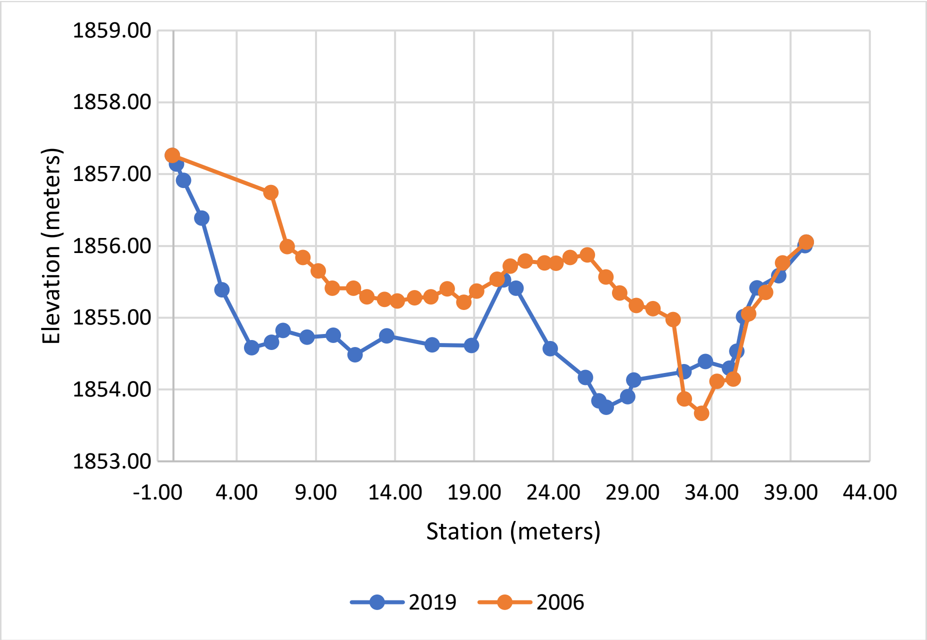


Figure 5: Transect 1 cross-sectional area comparison between 2006 and 2019. 33 meters^2 eroded between 2006 and 2019.

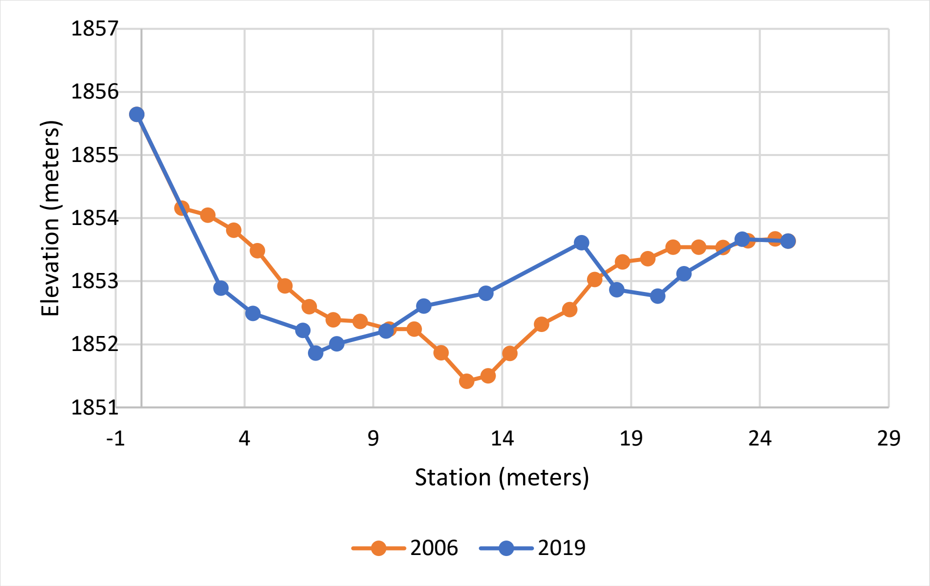


Figure 6: Transect 3 cross-sectional area comparison between 2006 and 2019. The cross-section decreased in area by 0.50 meters^2 between 2006 and 2019.

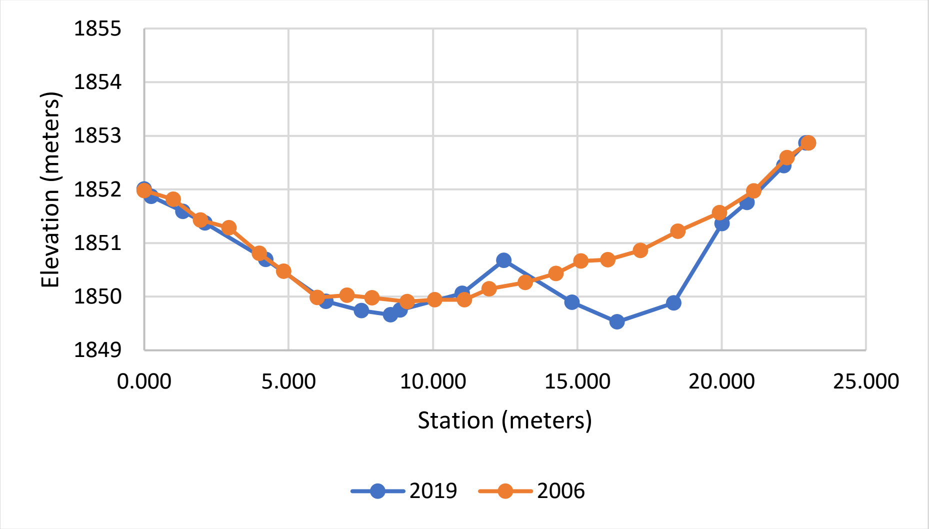


Figure 7: Transect 5 cross-sectional area comparison between 2006 and 2019. 6.90 meters^2 eroded between 2006 and 2019.

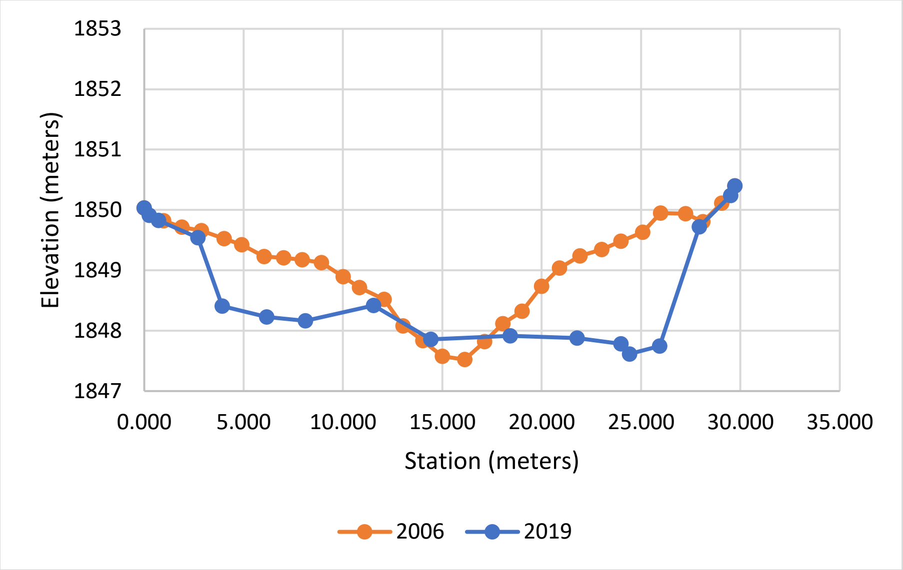


Figure 8: Transect 7 cross-sectional area comparison between 2006 and 2019. 16.20 meters^2 eroded between 2006 and 2019.

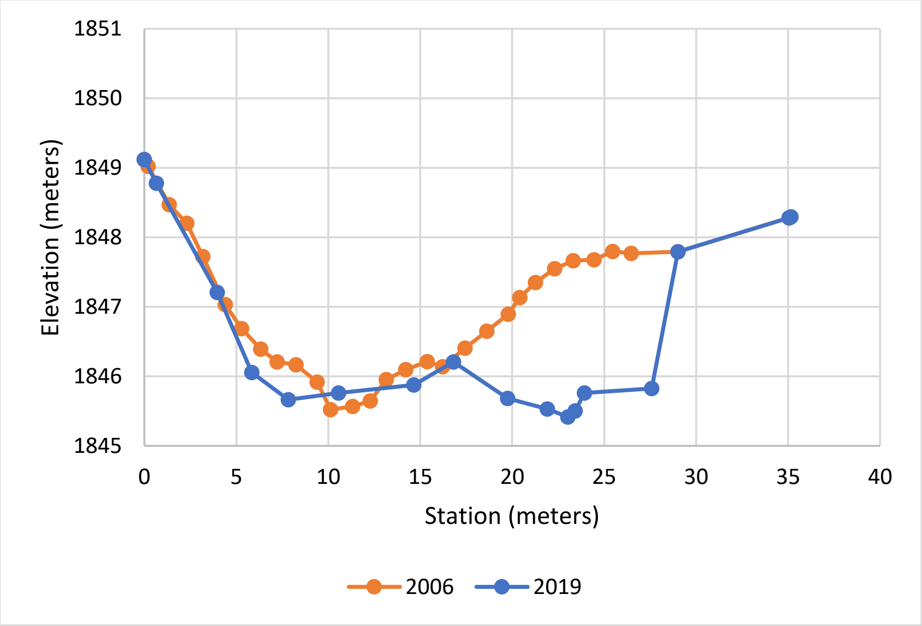


Figure 9: Transect 9 cross-sectional area comparison between 2006 and 2019. 20.60 meters^2 eroded between 2006 and 2019.

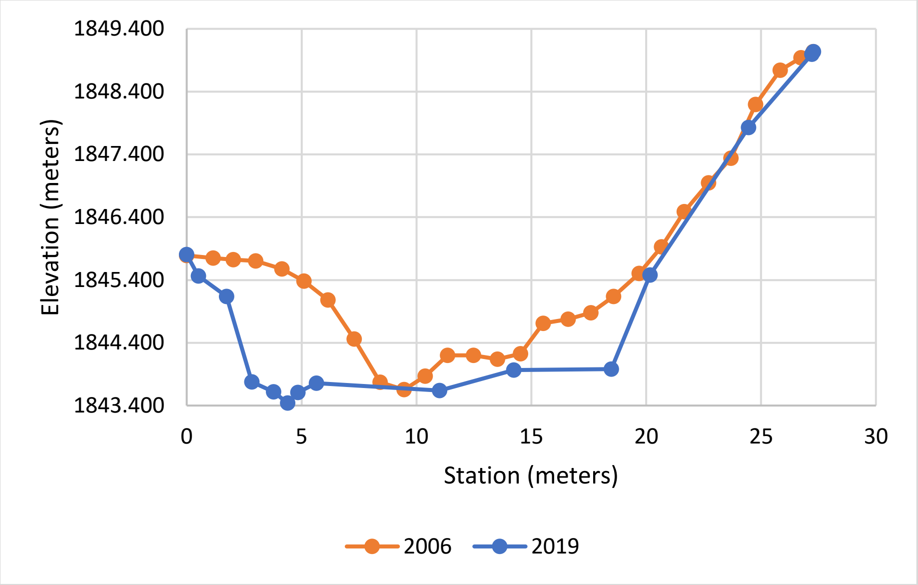


Figure 10: Transect 11 cross-sectional area comparison between 2006 and 2019. 15.70 meters^2 eroded between 2006 and 2019.

Species	**2006 Mean Cover ± Standard Deviation**	**2019 Mean Cover ± Standard Deviation**	**Percent Change**
Acer negundo	16.74 ± 18.90	2.17 ± 5.32	-87%
Alnus oblongifolia	34.55 ± 21.08	0	not detected in 2019
Foresteria pubescens var. pubescens	16.47 ± 9.11	5.09 ± 6.82	-69%
Juniperus scopulorum	1.93 ± 4.72	0	not detected in 2019
Parthenocissus vitacea	0	2.30 ± 5.63	not detected in 2006
Pinus ponderosa	4.31 ± 10.56	0	not detected in 2019
Populus deltoides	0	0.10 ± 0.16	not detected in 2006
Ptelea trifoliata	4.62 ± 11.13	0	not detected in 2019
Quercus gambelii	0.31 ± 0.76	1.99 ± 4.87	538%
Rhus trilobata	0.16 ± 0.26	0.9162 ± 2.2443	461%
Rosa woodsii	2.13 ± 5.21	0.81 ± 1.26	-62%
Salix exigua	0	4.10 ± 5.57	not detected in 2006
Salix gooddingii	0	0.09 ± 0.23	not detected in 2006
Salix irrorata	0	0.25 ± 0.44	not detected in 2006
Salix lutea	0	0.66 ± 1.36	not detected in 2006

Table 2: Mean percent cover by species in 2006 and 2019 and percent change between years, along 6 study transects in Capulin Creek.

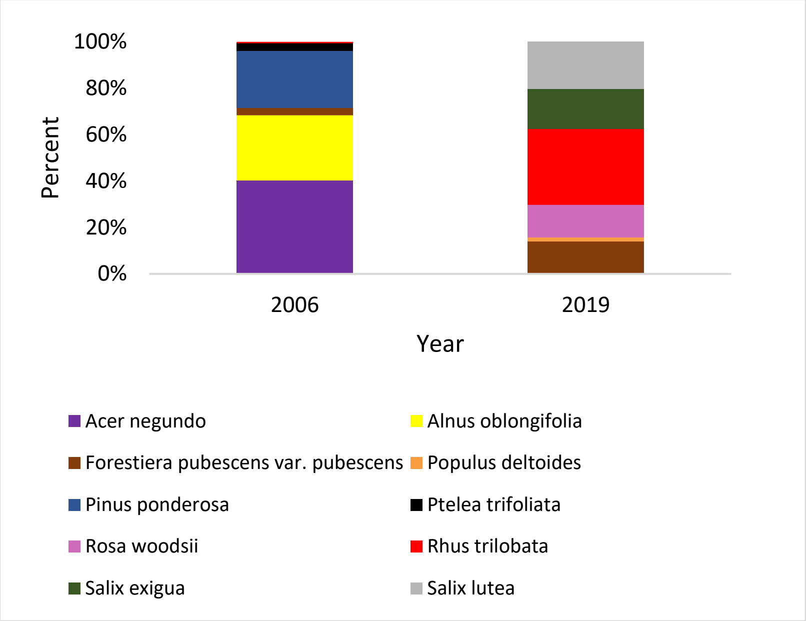


Figure 11: Transect 1 relative percent cover from 2006 and 2019. Note the shift in composition, namely the absence of *P. ponderosa* and *A. oblongifolia* from the transect in 2019.

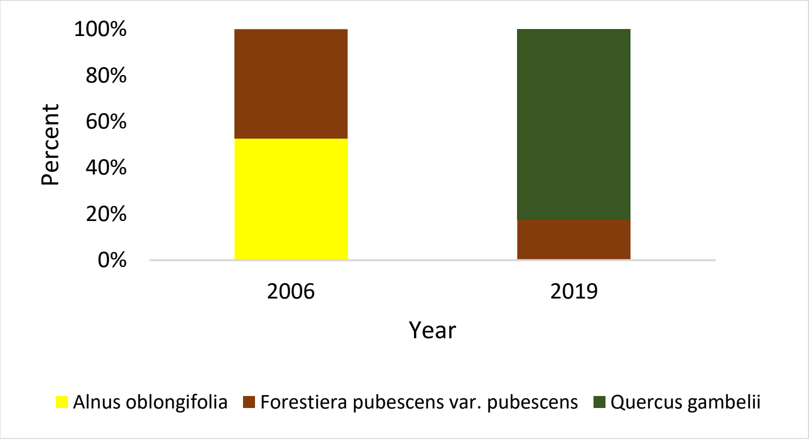


Figure 12: Transect 3 relative percent cover from 2006 and 2019. Note the shift in composition, namely the relatively large percent of *A. oblongifolia* in 2006 and its lack of detection on the transect in 2019.

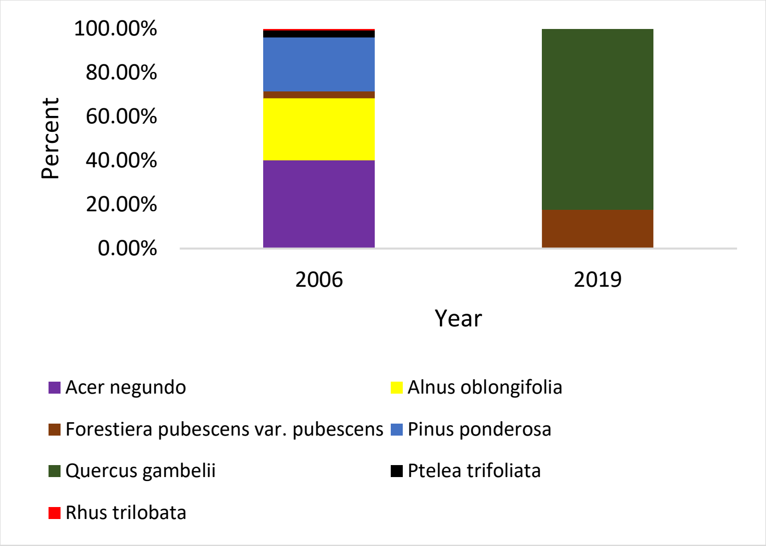


Figure 13: Transect 5 relative percent cover from 2006 and 2019. Note the shift in composition, namely the relatively large percent of *A. oblongifolia* in 2006 and its lack of detection on the transect in 2019.

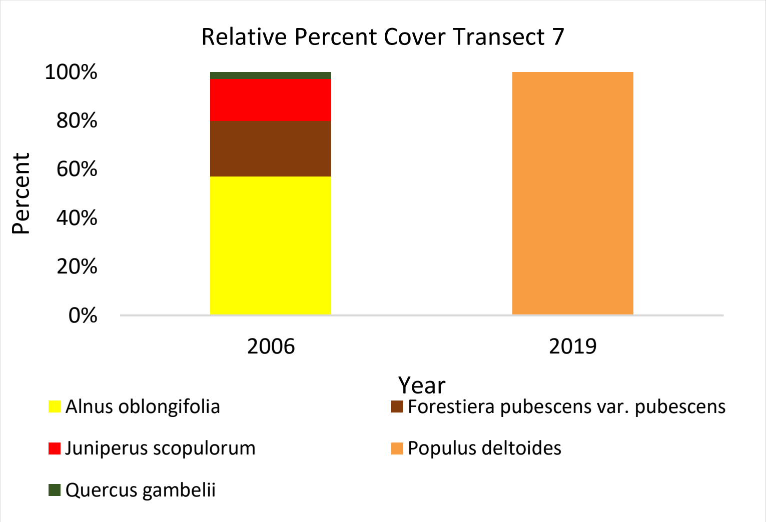


Figure 14: Transect 7 relative percent cover from 2006 and 2019. Note the shift in composition, namely the relatively large percent of *A. oblongifolia* in 2006 and its lack of detection on the transect in 2019.

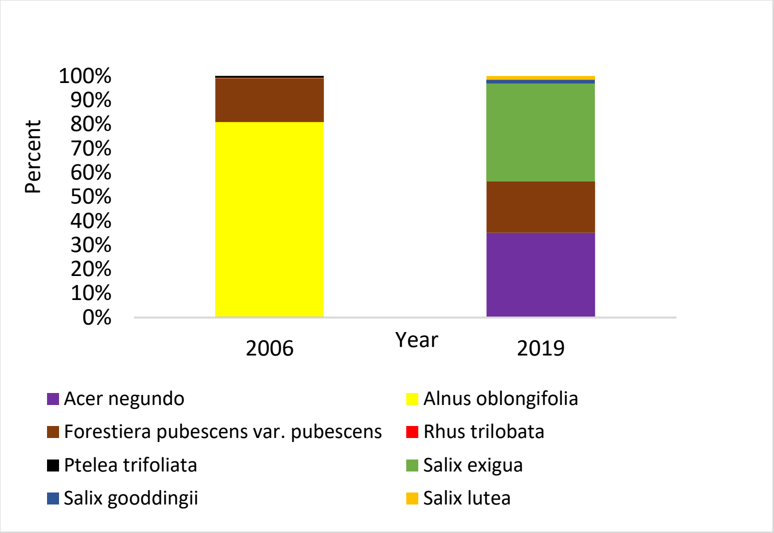


Figure 15: Transect 9 relative percent cover from 2006 and 2019. Note the shift in composition, namely the relatively large percent of *A. oblongifolia* in 2006 and its lack of detection on the line intercept in 2019.

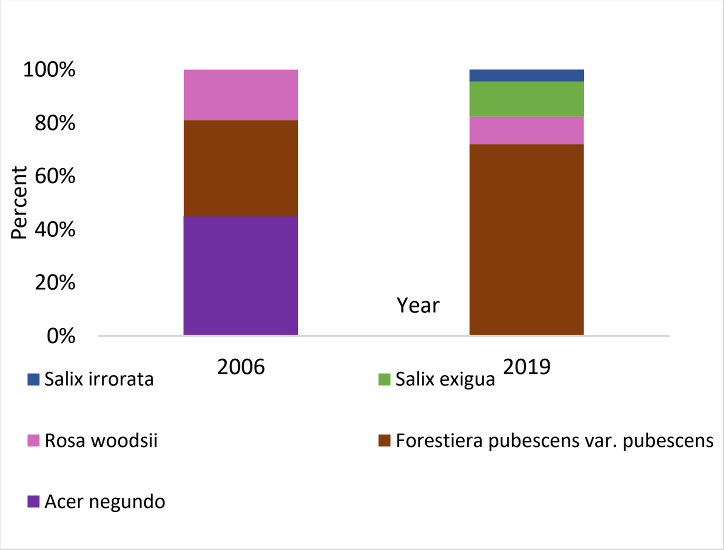


Figure 16: Transect 11 relative percent cover from 2006 and 2019. Note the shift in composition, namely the detection of *S. exigua* and *S. irrorata* on the line intercept in 2019 when they were not detected in 2006.

Species	**2006 Mean BA ± Standard Deviation**	**2019 Mean BA ± Standard Deviation**	**Percent Change**
Acer negundo	1.66 ± 0.7126	0.23 ± 0.51	-86%
Alnus oblongifolia	4.57 ± 2.65	0	not detected in 2019
Juniperus scopulorum	0.63 ± 1.55	0	not detected in 2019
Pinus ponderosa	10.15 ± 13.19	0	not detected in 2019
Populus fremontii	0.0004 ± 0.001	0.0005 ± 0.0009	25%
Quercus gambelii	0.006 ± 0.008	0.07 ± 0.17	0.04%
Quercus grisea	0.01 ± 0.03	0	not detected in 2019
Salix gooddingii	0.001 ± 0.003	0.007 ± 0.005	490%
Salix lutea	0	0.01 ± 0.03	not detected in 2006

Table 3: Mean basal area (meters^2/hectare) in 2006 and 2019 and percent change between years, along 6 study transects in Capulin Creek.

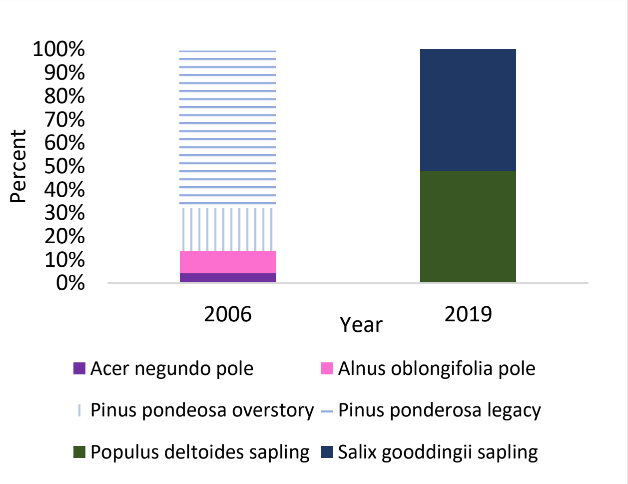


Figure 17: Transect 1 relative basal area by species and size class in 2006 and 2019. Note the shift in composition and structure between 2006 and 2019.

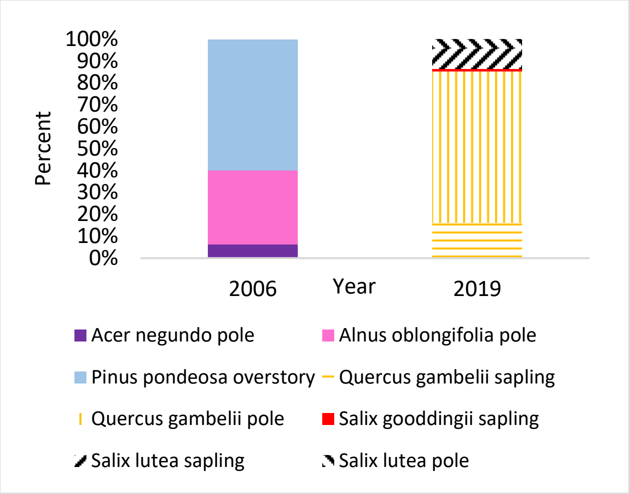


Figure 18: Transect 3 relative basal area by species and size class in 2006 and 2019. Note the shift in composition and structure between 2006 and 2019.

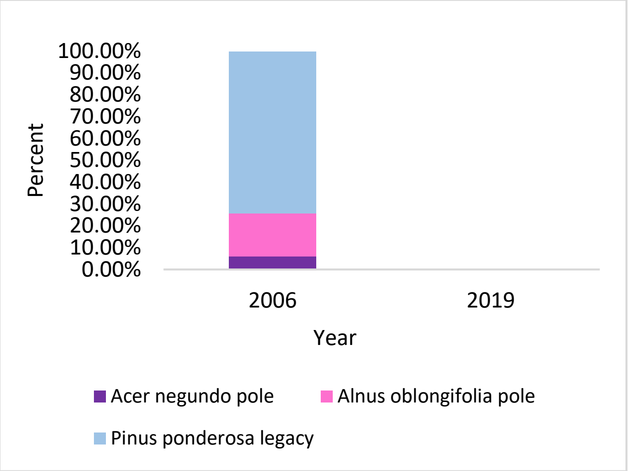


Figure 19: Transect 5 relative basal area by species and size class in 2006 and 2019. Note the shift in composition and structure between 2006 and 2019, namely that no trees were detected on the transect in 2019.

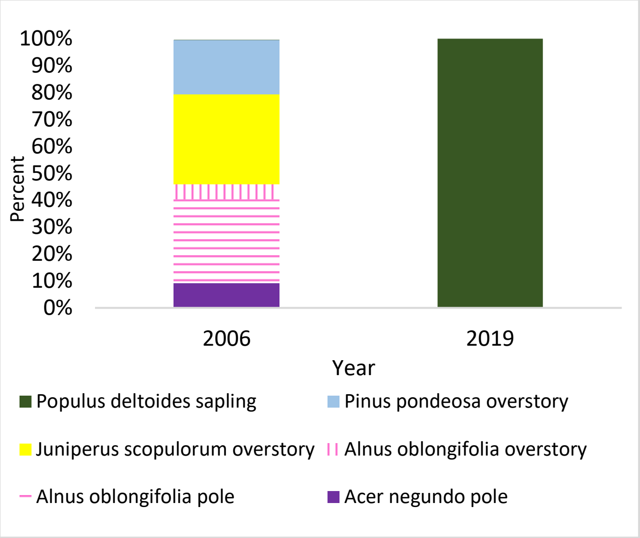


Figure 20: Transect 7 relative basal area by species and size class in 2006 and 2019. Note the shift in composition and structure between 2006 and 2019, namely that *P. deltoides* was the only tree species detected in 2019.

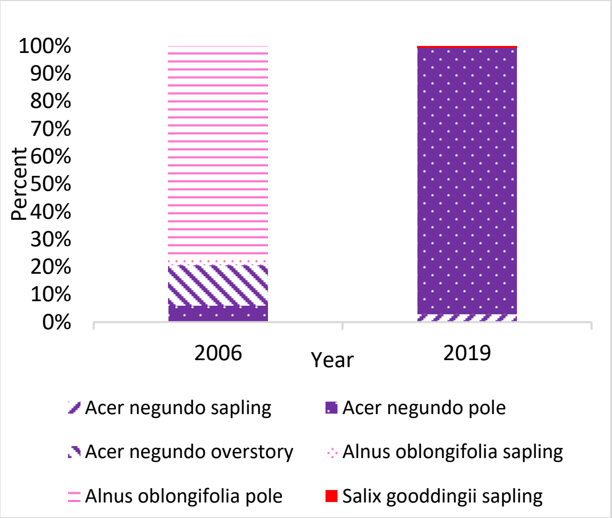


Figure 21: Transect 9 relative basal area by species and size class in 2006 and 2019. Note the shift in composition and structure between 2006 and 2019.

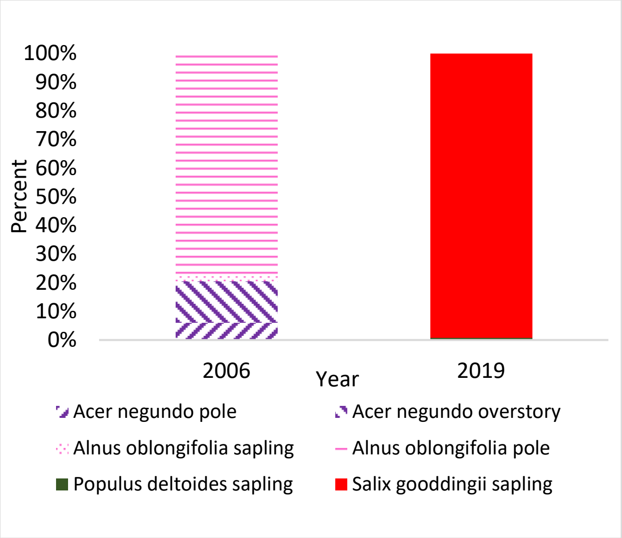


Figure 22: Transect 11 relative basal area by species and size class in 2006 and 2019. Note the shift in composition and structure between 2006 and 2019.

Species	**2006 Mean Density ± Standard Deviation**	**2019 Mean Density ± Standard Deviation**	**Percent Change**
Acer negundo sapling	156.03 ± 220.16	27.78 ± 62.11	-82%
Acer negundo pole	601.45 ± 353.61	90.28 ± 201.87	-85%
Acer negundo overstory	6.41 ± 15.70	0	not detected in 2019
Alnus oblongifolia sapling	177.51 ± 209.09	0	not detected in 2019
Alnus oblongifolia pole	1143.01 ± 727.16	0	not detected in 2019
Alnus oblongifolia overstory	5.67 ± 13.89	0	not detected in 2019
Juniperus scopulorum overstory	11.34 ± 27.77	0	not detected in 2019
Pinus ponderosa legacy	9.91 ± 24.26	0	not detected in 2019
Pinus ponderosa overstory	35.38 ± 33.36	0	not detected in 2019
Populus deltoides sapling	5.67 ± 13.89	51.75 ± 36.93	+813%
Quercus gambelii sapling	27.65 ± 31.78	70.97 ± 158.69	+157%
Quercus gambelii pole	6.41 ± 15.70	45.16 ± 100.98	+605%
Quercus grisea sapling	36.43 ± 89.23	0	not detected in 2019
Quercus grisea pole	6.07 ± 14.87	0	not detected in 2019
Salix gooddingii sapling	5.67 ± 13.89	92.05 ± 53.71	+1523%
Salix lutea sapling	0	45.16 ± 100.98	not detected in 2006
Salix lutea pole	0	6.45 ± 14.43	not detected in 2006

Table 4: Mean stem density (stems/hectare) by species and size class in 2006 and 2019 and percent change between years, along 6 study transects in Capulin Creek.

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