ESTANCIA BASIN WATERSHED HEALTH AND MONITORING PROJECT: 2015 ANNUAL REPORT

Prepared for

ESTANCIA BASIN WATERSHED HEALTH, RESTORATION AND MONITORING STEERING COMMITTEE

Composed of: Claunch-Pinto Soil and Water Conservation District, Edgewood Soil and Water Conservation District, East Torrance Soil and Water Conservation District, Estancia Basin Water Planning Committee, Chilili Land Grant, Manzano Land Grant, New Mexico State Forestry, New Mexico Environment Department, New Mexico Forest and Watershed Restoration Institute, New Mexico Department of Agriculture
(with funding through the New Mexico Water Trust Board)

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EXECUTIVE SUMMARY

The Estancia Basin Watershed Health, Restoration and Monitoring Steering Committee (Steering Committee) oversees forest thinning projects and monitoring of forest and watershed health in the Estancia Basin in coordination with the New Mexico Forest and Watershed Restoration Institute. The primary goals of the Steering Committee are to improve forest health and create defensible space from wildfire. Funding for forest and watershed monitoring has been provided by the New Mexico Water Trust Board.

In 2007, SWCA Environmental Consultants (SWCA) was awarded a contract to conduct monitoring for forest thinning effectiveness on the eastern slopes of the Manzano Mountains. SWCA finalized a comprehensive monitoring plan in March 2008—available online at the New Mexico Forest and Watershed Restoration Institute's website (http://www.nmfwri.org/ images/ stories/pdfs/Estancia_Basin_Monitoring/EstanciaBasinMonitoring.pdf)—that provides background information, research questions, and a discussion of methods relative to forest thinning and monitoring (SWCA 2008). The monitoring study provided 3 years of pre-thinning baseline data to provide background information on all study sites prior to implementing thinning treatments and monitoring treatment effectiveness. Results from the 2008 through 2014 monitoring seasons are presented in the 2008, 2009, 2010, 2011, 2012, 2013, and 2014 annual reports, respectively, which can also be found on the New Mexico Forest and Watershed Restoration Institute's website. Results from 2015 are presented here.

The principal goals of forest and watershed monitoring are to determine the effectiveness of standard prescribed forest thinning on soils, hydrology, water yield, vegetation, and wildlife. SWCA is responsible for planning and implementing forest thinning monitoring in order to evaluate these resources. SWCA has also assumed responsibility for the South Mountain Weather Station, which was previously installed by another contractor in 2006. After monitoring began, three major wildfires (Ojo Peak, Trigo, and Big Spring) occurred in the monitoring area in late 2007 and early 2008. The Trigo fire destroyed one of the forest thinning monitoring sites, which was replaced during summer 2008. SWCA implemented a monitoring study of post-Trigo fire recovery on private forest lands from 2008 to 2012. Fire monitoring measurements were discontinued at the end of the 2011 monitoring season due to dead tree fall hazards in the monitoring areas. Monitoring in those areas may resume in the future, as conditions allow.

This 2015 Annual Report provides information on the results of forest thinning during the calendar year 2015. Initial 2008, 2009, and 2010 baseline pre-treatment monitoring data from permanent monitoring study sites provided information on rainfall, ambient and soil temperatures, soil moisture, soil surface profiles to assess erosion over time, soil surface stability, soil chemistry, bird and small mammal composition and relative abundance, and vegetation composition, structure, and cover. Monitoring data from 2011 to 2015 provides post treatment information on the above parameters along with data on medium-sized to large wildlife and livestock for the first 5 years following thinning treatments.

The monitoring sampling design employs paired monitoring plots at two piñon/juniper (*Pinus edulis/Juniperus monosperma*) woodland sites and two ponderosa pine (*Pinus ponderosa*) sites. One plot of each pair was randomly selected and treated by thinning tree stands in late 2010/ early 2011. Those thinning treatments were completed in early 2011, and SWCA will continue to monitor the above-mentioned parameters through at least June 2017 to examine the impacts and effectiveness of forest thinning treatments. Not only are paired study plots being compared to each other in a treatment/control design, but each treated plot will be monitored over time to assess changes resulting from thinning treatments.

Results from the 2015 fifth year of post-treatment monitoring data revealed similar differences in parameter values between treatment and control plots to those that occurred since 2011, but not prior to thinning treatments.

- Tree and woody vegetation structure was greatly changed from the thinning treatments, resulting in more open forest stands on the treated watersheds. Sapling conifer trees are now beginning to become common on the treated plots, indicating that follow-up maintenance thinning treatments may be needed in coming years.
- Tree growth and health have not differed much between control and treatment plots since thinning treatments were imposed. Despite a severe drought from 2011 through 2013, trees measured on thinned plots have not differed from trees on control plots.
- The severe drought that extended from 2011 through 2014 ended in 2015. All study sites received near long-term average rainfall and herbaceous vegetation growth was substantial across the area.
- During the 2015 monitoring period, several rainfall events generated surface runoff events basin-wide. Paired flows did not consistently produce more or less surface runoff from control or treated plots.
- Stream flow monitoring did not pick up any flows in 2015.
- Groundwater well monitoring in the area continued to show the same trends, however, the well at Manzano was discontinued due to being dry.
- Soil moisture was higher on treated plots than control plots, especially during dry periods following rainfall events.This pattern has held since thinning treatments were completed in 2011.
- Herbaceous vegetation canopy cover was again significantly higher on both piñon/juniper treated plots and at one ponderosa pine treated plot, when compared to the control plots. Herbaceous vegetation cover was even higher than measured on treated plots, because domestic livestock grazed the treated plots prior to vegetation measurements. Livestock grazing appeared to be more intense on the treated plots than on the control plots, perhaps in response to the increased herbaceous vegetation growth on treated plots. Herbaceous vegetation grew considerably across the region following substantialsummer rains.

- A repeat photo point monitoring protocol was developed and implemented in 2014 to serve as a rapid and low-cost method for forest thinning effectiveness monitoring. 2015 photo point photograph (five years following treatments) comparisons to 2011 photographs (less than one-year following treatments) showed a decrease in tree densities, increased herbaceous vegetation, stable soil surfaces, but an increase in regrowth of young conifer trees on one ponderosa pine site and at both piñon/juniper sites. This photo monitoring method will be applied to private lands thinning projects throughout the area in 2016 as a way to expand the thinning effectiveness monitoring.
- Bird densities and species richness in 2015 were found to be higher on treated plots than on control plots at all sites during the spring breeding season, but there was no consistent pattern in abundance during the fall migration period During the 2015 spring breeding season, bird species composition was more similar among plots based on treatment history than location at the piñon/juniper sites, but pairs of control and treatment plots at each site (i.e., by location) were more similar than by treatment history at the ponderosa pine sites. During the fall migration period, bird community compositions showed no clear associations by either treatment type or by site locations. These results indicate that the thinning treatments are affecting bird communities during the breeding season, and that more birds are attracted to the treated plots during breeding season.
- Rodent densities were moderate in 2015, and both rodent abundance and species richness were lower on most of the treatment plots than on the control plots across all sites, but especially at the piñon/juniper sites. Piñon mice (*Peromyscus truei*) dominated at the piñon/juniper sites and at one ponderosa site; deer mice (*P. maniculatus*) were most common at the other ponderosa pine site. Piñon mice were less abundant on treated plots at the piñon/juniper sites.
- Native large animals recorded from remote wildlife cameras such as mule deer (*Odocoileus hemionus*), elk (*Cervus canadensis*), and rabbits (*Leporidae*) were most abundant on control plots in 2015, while domestic livestock were far more abundant on treated plots at both piñon/juniper and ponderosa pine sites.
- Other parameters such as soil chemistry, soil surface erosion and surface stability, and have not yet shown clear and consistent differences between treatment and control plots.

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1.0 INTRODUCTION

This 2015 Annual Report provides summaries of monitoring data collected during the 2015 calendar year for the Estancia Basin Watershed Health, Restoration and Monitoring Steering Committee (Steering Committee). Details about research questions and the background and administration of this monitoring project may be found in the *Estancia Basin Watershed Health and Monitoring Project: Monitoring Plan Evaluation* (2008 Monitoring Plan) (SWCA Environmental Consultants [SWCA] 2008), which is available at the New Mexico Forest and Watershed Restoration Institute (Restoration Institute) website (http://www.nmfwri.org/images/stories/pdfs/Estancia_Basin_Monitoring/EstanciaBasinMonitoring.pdf). The 2008 Monitoring Plan provides detailed information on the background knowledge of forest thinning in the Southwest and presents the goals and methodologies for the Estancia Basin forest thinning monitoring project. The 2008 Annual Report (SWCA 2009) also provides important background information about the Trigo wildfire monitoring project that was initiated in 2008. Previous annual reports for 2008, 2009, 2010, 2011, 2012, 2013, and 2014 summarize overall monitoring findings from those 6 years, and they also may be found at the Restoration Institute website.

The Steering Committee oversees forest thinning and effectiveness monitoring of forest thinning on ponderosa pine (*Pinus ponderosa*) forests and piñon/juniper (*Pinus edulis/Juniperus monosperma*) woodlands on private and state lands on the eastern slopes of the Manzano Mountains, New Mexico. Principal members of the Steering Committee include the Claunch-Pinto, East Torrance, and Edgewood Soil and Water Conservation Districts; New Mexico State Forestry; and the Restoration Institute. The Restoration Institute is additionally providing oversight and public relations for forest thinning and monitoring activities.

The principal goals of the Steering Committee are to create defensible space around homes and other structures to protect against wildfire and improve overall forest health, following forest thinning prescriptions determined by New Mexico State Forestry. The primary goals of forest thinning monitoring are to determine the impacts of standard prescribed forest thinning on soils, hydrology, water yield and quality, vegetation, and wildlife.

The scope of work for this monitoring project was described in the Steering Committee's 2007 request for proposals as follows:

1. Plan and implement methods to determine how vegetation thinning and removal affect water yield.
2. Plan and implement methods of establishing reliable and repeatable vegetation monitoring methods to allow for both qualitative interpretation and quantitative documentation of change in vegetative structure and composition over time.
3. Plan and implement methods of monitoring small mammal and avian populations, which are indicators of ecosystem health.

SWCA is currently under contract for 5 years of monitoring, beginning in 2015, and is responsible for study site maintenance, data collection, data management, data analysis and interpretation, and information dissemination (including monthly meetings, monthly reports, and annual reports). The current Steering Committee plan calls for 3 years of baseline pre-thinning treatment monitoring (2008–2010), thinning treatments implemented during the winter of 2010 and 2011, and a least 10 years of post-treatment monitoring (2011–2021).

Several new subprojects were added to the overall monitoring project in 2008, including post-fire monitoring of soils, hydrology, vegetation, and wildlife on private forest lands following the Trigo wildfire. These tasks involve developing and implementing ephemeral stream and groundwater monitoring to assess the effects of both forest thinning and the Trigo fire on water resources, as well as assuming the operation and reporting for the South Mountain Weather Station (SMWS), initiated by EnviroLogic in 2006. A map of all study sites for these projects is presented in Figure 1.1.

This 2015 Annual Report is similar in format to the previous 2008, 2009, 2010, 2011, 2012, 2013, 2014 annual reports, and it provides complete data files (appended on DVD) and summaries of findings from field monitoring measurements conducted during the calendar year 2015 for the primary subprojects: 1) forest thinning monitoring of weather, soils, hydrology, vegetation, and wildlife; 2) overall Manzano watershed ephemeral stream and groundwater monitoring, associated with both forest thinning and post-wildfire monitoring.Data collected in 2008, 2009, and 2010 represent baseline conditions prior to forest thinning treatments, which were begun in late 2010 and were completed by May 2011. Data collected after thinning in 2011 provides measures of thinning treatment effectiveness and a comparison of post-treatment environmental conditions. Monitoring from subsequent years will provide data on thinning treatment effects over time.

This report provides analyses of parameter changes over the 8 years of monitoring and comparisons of paired treatment and control plots to evaluate treatment effects. Some statistical tests of parameter values between paired study plots are also provided to compare pre-thinning treatment baseline conditions to post-treatment conditions in order to determine if the paired plots differ in parameter values resulting from imposed thinning treatments. Numerous discrete data sets have been collected, and SWCA has been active in creating data collection, storage, and management plans for each of the subprojects. SWCA has created metadata for each of these data sets that outline the date range of each data set, the collection methods, the unit measurements, and the abbreviations and codes used within each data file. The metadata files will also state any caveats or general comments of which the viewer should be aware before analyzing the data.

SWCA is making these data available in a form that can be easily disseminated, using readily available software such as Microsoft Word and Excel. Some information, such as those data collected from the WatchDog Mini Weather Stations, is collected using proprietary software. These data are converted into Microsoft Excel files so they can be viewed by the general public. SWCA also intends to make the data available in forms that are easy to analyze. Some data, such as those related to the flumes, which are recorded in 5-minute intervals, must be partitioned into several files, as the data exceed Microsoft Excel's capacity of data rows. All of these data are being made available to the Restoration Institute for dissemination on its website. Note that measurements from various aspects of monitoring are reported in English units (e.g., feet, acres), while others are reported in metric units (meters, hectares). The protocols for monitoring measurements were obtained from different sources that use different units of measure. The U.S. Department of Agriculture (USDA) Agricultural Research Service Rangeland Monitoring Manual (Herrick et al. 2005) uses metric units, while the U.S. Forest Service (USFS) Forest Inventory and Analysis Guide (USFS 2005) uses English units. In general, scientific research worldwide has adopted the metric system as the standard for measurements, while some federal and state agencies use English units of measure. For ease of comparison, values are presented in this report with both English and metric units, except where not feasible.

This 2015 Annual Report provides summaries of findings from field monitoring measurements conducted during the calendar year 2015 and compares them with previous years for the above-mentioned projects and subprojects. This report is partitioned into different sections for each subproject:1.0) Introduction (this section), 2.0) Forest Thinning Monitoring, 3.0) ephemeral stream and groundwater monitoring, and 4.0) Planned Monitoring for 2016 (Year 9).

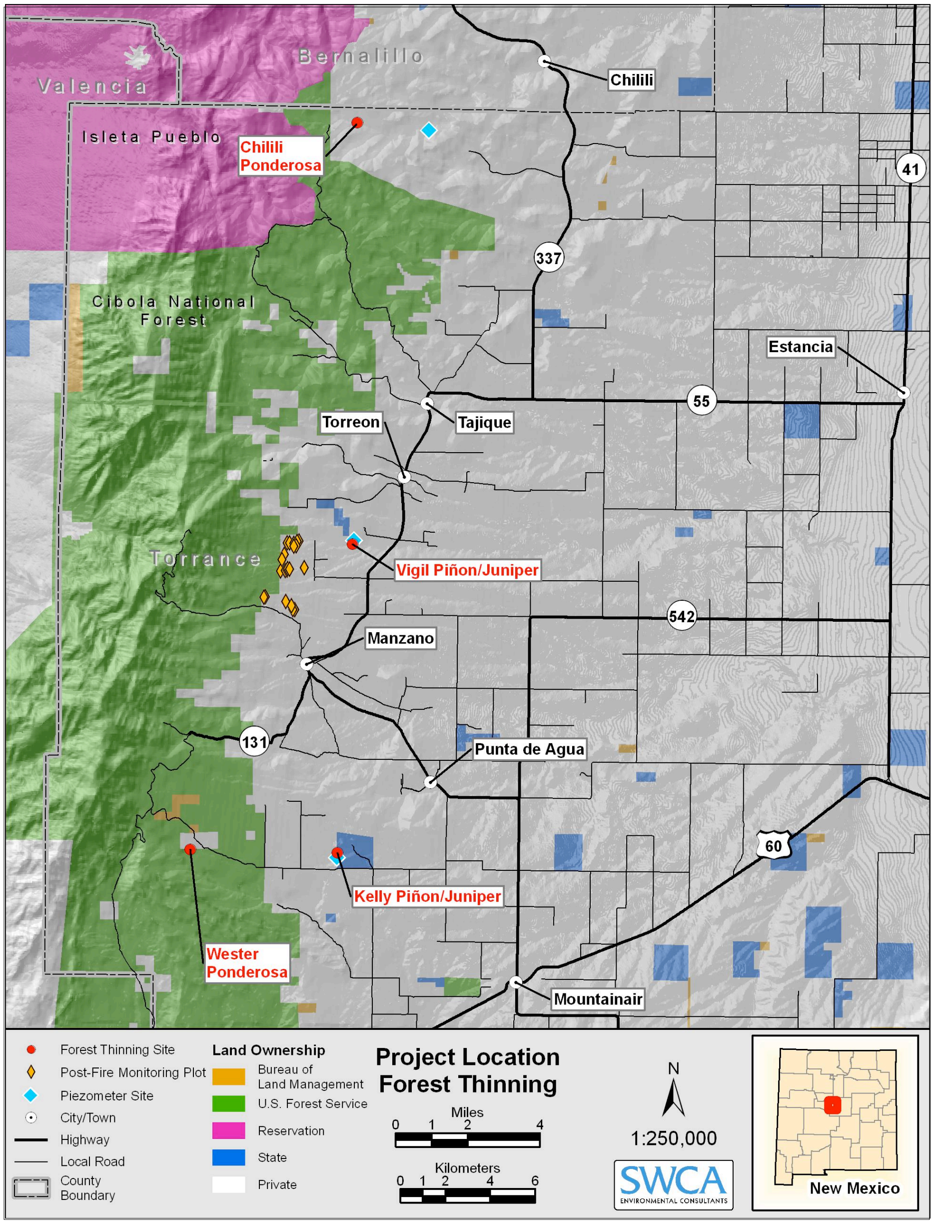


Figure 1.1. **Map of all Estancia Basin forest and watershed monitoring locations addressed in this report.**

2.0 FOREST THINNING MONITORING

Details of forest thinning monitoring protocols were provided in the 2008 Monitoring Plan (SWCA 2008). Background information on the known environmental effects of forest thinning on Southwestern forest ecosystems was also presented in the 2008 Monitoring Plan, along with detailed discussions of the experimental study design and methods used in this research to measure various environmental responses to forest thinning treatments. Since 2008 there have been some significant updates to Southwest forest ecology, climate, wildfire, and forest thinning literature. Section 2.1 provides a literature review update for all resource areas included in this monitoring study in order to keep this project up to date with the most current information.

2.01 UPDATED LITERATURE REVIEW FOR THE ENVIRONMENTAL EFFECTS OF FOREST THINNING IN THE SOUTHWEST

The sections below address recent research findings about the effects of forest thinning in the American Southwest and New Mexico on natural resources that are addressed in this monitoring study. Climate and resources are presented below in the same order as they are organized in this monitoring report.

2.01.1 CLIMATE

As mentioned in the original monitoring plan (SWCA 2008), climate change is likely to significantly affect the findings of this forest thinning monitoring study, as drought intensifies and temperatures increase across New Mexico. Recent key articles on changing Southwest and New Mexico climate by Gutzler (2013) and Llewellyn and Vaddey (2013) discuss how the climate of the Southwest has been documented as becoming warmer and less predictable, and how drought is becoming more common and more severe than in the past. The average annual ambient temperatures for the Upper and Middle Rio Grande regions of New Mexico (Colorado border to Truth or Consequences, New Mexico) has increased from 1971 to 2012 by 1.4 degrees Celsius (°C) (2.5 degrees Fahrenheit [°F]), and in mountainous areas that increase has been even greater at 1.5°C (2.7°F) (Llewellyn and Vaddey 2013). Winter temperatures (December, January, and February) have been warming by as much as 1.3°C (2.3°F) since 1970 (National Weather Service [NWS] 2015). Long-term episodic droughts have occurred in the Southwest region for centuries (Gutzler 2013), but the region is strongly affected by ongoing and projected century-scale climate change (Llewellyn and Vaddey 2013). The start of 2015 was very moist statewide, but February-March was drier than the long-term climatology for those months. However, precipitation increased from May through July and then declined in August and September. Due to the higher than average precipitation in May, June, July, and October, overall precipitation was near 150% of normal through the end of 2015. The drought monitor for December 2015 shows that almost none of the state is currently experiencing drought, and the project area specifically is categorized as not in drought (NWS 2015). Llewellyn and Vaddey (2013) attribute the climate change observed in the Southwest to human-caused increases in greenhouse gases and report on a strong regional warming trend in recent temperature data that modifies natural drought/high precipitation fluctuations by enhancing evaporative losses and decreasing snowpack in mountainous regions to the north (see Brown and Mote 2009). Current modeling predicts that peak runoff will occur earlier, leaving less water for irrigators during the hot and dry months of the pre-monsoon growing season (Elias, 2015). As the climate warms, intense storms are expected to increase in the region (Gutzler 2013), and a greater fraction of total annual precipitation is expected to come from single intense rainfall or snowfall events as compared to more frequent low-intensity storms (Allan and Soden 2008; Intergovernmental Panel on Climate Change 2007; Tebaldi et al. 2006). Petrie et al. (2014) demonstrate that fewer single storm events are determining precipitation amounts in central and southern New Mexico, especially during the monsoon season, and that the number of such storms has declined and become more variable over the last decade. These fewer but more intense events are also being documented in the region by others (Allan and Soden 2008; Groisman et al. 2008). The periodic drought and intense rainfall patterns that are projected for the region (Alexander et al. 2006; Hurd and Coonrod 2008; Gutzler 2013; Gutzler and Robbins 2011) are expected to result in significantly diminished stream flow and drier surface conditions (Llewellyn and Vaddey 2013; Seager et al. 2008), causing the Southwest's climate to become even more arid than it currently is over the coming decades.

There is strong evidence in the literature that changing climates are affecting forest resources (Allen et al. 2010; Bonan 2008; Breshears et al. 2009; Floyd et al. 2009; Negrón et al. 2009; Raffa et al. 2008; van Mantgem et al. 2009; Williams et al. 2013; Woodall et al. 2011). The 2014 National Climate Assessment (U.S. Global Change Research Program 2014) projects increased wildfire, forest disease, and drought for the Southwest region as a result of climate change. Better understanding of climate change is needed in order to aid forest resource management planning (Allen et al. 2010). Ganey and Vojta (2011) monitored tree mortality in drought-stressed mixed-conifer and ponderosa pine forests in Arizona that were impacted by drought from 1997 to 2007 (Breshears et al. 2005). They observed considerable tree mortality across almost 100% of their plots, attributed primarily to insect attack resulting from drought and tree water stress. They hypothesize that climate change will heavily impact these forest types because they are not well adapted to long-term drought conditions.

Other studies recorded mortality of piñon/juniper woodlands in the same area during this period (Breshears et al. 2005; Floyd et al. 2009; Mueller et al. 2005). Pinyon pine and ponderosa mortality can be predicted based on modelled precipitation and evapotranspiration (Huang, 2015). Findings that linked drought and insect damage to mortality in mixed-conifer and ponderosa pine–dominated forests were consistent with other similar studies in the area. In most of those studies, bark beetles of the genus *Ips* have been found in association with piñon/juniper and ponderosa pine forests, and Douglas-fir beetle (*Dendroctonus pseudotsugae*) and the fir engraver (*Scolytus ventralis*) in Douglas-fir (*Pseudotsuga menziesii*) and white fir (*Abies concolor*) forests (Breece et al. 2008; Guarín and Taylor 2005; Maloney and Rizzo 2002; Savage 1997; Stephens and Gill 2005), and insect attacks were determined to be a proximate cause of tree death, mediated by long-term drought. Extreme climate conditions were recorded throughout Ganey and Vojta's (2011) study period, particularly in 2002 when tree ring reconstruction indicated that 2002 was the third-driest year in over 1,400 years. Interestingly, their results showed that mortality was not significantly related to either elevation or stand density. This finding indicates that thinning alone may not always alleviate moisture stress–related mortality during times of drought. Ganey and Vojta (2011) also reported the relative high mortality of large trees versus smaller trees, and they expressed concern that other studies had similar findings (e.g., Floyd et al. 2009; Mueller et al. 2005; van Mantgem et al. 2009) and that large trees are not only of commercial value, but also provide important habitat and are already rare on the landscape (Ganey and Vojta 2011).

Research by Savage et al. (1996) and Savage et al. (2013) demonstrate that regeneration and establishment of ponderosa pine is sensitive to specifically timed temperature and precipitation patterns—particularly important are series of wet years (Brown and Wu 2005). Such climate patterns are now affected by persistent drought that exacerbates ponderosa pine regeneration and establishment, and increases severe wildfire (Savage et al. 2013). The long periods of drought that have been recently observed throughout the Southwest, in combination with altered forest management practices and fire exclusion policies over the last century, have resulted in frequent landscape-level high-severity fires that are beyond the range of natural variability (Allen et al. 2002; Covington and Moore 1994a, 1994b). Savage et al. (2013) suggest that for Southwest ponderosa pine forests, a specific climate window critical for regeneration is narrowed by a synchronous occurrence of high-severity fire and drought. Spring and summer drought conditions have been related to high seedling mortality (Rietveld and Heidmann 1976), and sparse vegetation cover and low moisture in the fall result in early freezes that also reduce the period available for seedling establishment. Growing periods are becoming shorter under drought conditions due to moisture stress. Furthermore, droughts can exert legacy effects that impair vegetative growth even after drought conditions end (Petrie, 2015). Under a climate change scenario of warmer and drier conditions, Savage et al. (2013) predict that ponderosa pine regeneration may become all but eliminated in the Southwest. Similarly, Williams et al. (2013), utilizing a forest drought stress index for Southwestern forests and projected climate models, suggest a transition of forests in the Southwestern Uniteds States towards distributions unfamiliar to modern civilization.

2.01.2 SOILS

Soils provide plants with structural support, nutrients, and symbiotic soil biota; therefore, soil disturbance on a site can decrease plant and forest productivity. The effect of forest management disturbance on soil sustainability is still relatively poorly known, and many questions remain on the subject (Harrison et al. 2011; Johnson et al. 2012; Morford et al. 2011; Powers et al. 2005). According to a recent nationwide survey of forest managers, soil productivity is a consideration of 80% of respondents when selecting fuel reduction techniques (Busse et al. 2014); however, few respondents ranked soil as a primary concern when selecting techniques when compared to other factors like cost, effectiveness, ease of use, and environmental factors. Most research to date has centered on the impacts of mechanized thinning operations that can result in severe soil disturbance (Elliott et al. 1996; Miller and Sirois 1986; Rice et al. 1972). Few studies have examined the impact of hand thinning operations on soil properties. Busse et al.'s (2014) study found that most practitioners were not concerned about the impact of hand thinning on soil properties due to a lack of mechanized equipment and low-intensity treatments, which could explain a lack of literature on the subject. Soil erosion, combined with other impacts from forest disturbance, such as soil compaction, can reduce forest sustainability and soil productivity. Forest soils are protected from erosion by litter and duff, and forest litter is a primary component, providing nutrients and retaining water. Thinning operations (both mechanized and hand thinning) can remove ground surface organic materials, thereby impacting nutrient levels and making soils vulnerable to erosion. While studies have observed reductions in litter cover following thinning, they have not observed consistent changes to soil pH, total carbon, nitrogen, or phosphorous of thinned forests (Overby, 2015). Maintaining soil surface litter cover is the easiest way to prevent accelerated erosion (Moghaddas 2013). Page-Dumroese et al. (2000) have used computer-based soil simulation models and found that, in many cases, the presence of at least 50% ground cover (e.g., vegetation or leaf litter) could prevent accelerated erosion rates. Robichaud et al. (2010) also suggest that levels of exposed bare soil less than 30% to 40% following forest thinning can generally keep soil erosion rates “acceptably low.”

Many tree thinning/fuel treatment approaches can be planned and implemented with minimal bare soil exposure, thereby limiting subsequent erosion (Moghaddas 2013). Researchers have found that tree cutting by itself does not cause significant erosion (Berg and Azuma 2010; Moghaddas 2013; Wayman and North 2007), and timber harvest operations usually cause less erosion per unit area than roads (Elliott et al.1996). The greatest disturbance to soils is associated with heavy machinery such as wheeled or tracked skidders used to drag logs to landing areas (Litschert and MacDonald 2009; Williamson and Nielson 2000); these often cause severe soil disturbance and ground cover removal (Moghaddas 2013). Steep slopes are also more vulnerable to erosion and machine impacts. Cram et al. (2007) have studied disturbance and erosion on intermediate (10%–25%) and steep (26%–43%) slopes in a thinned New Mexico mixed-conifer forest. They conclude that operations on steep slopes generally caused more soil disturbance, but maintaining soil cover and minimizing large areas of bare soil were sufficient to prevent increased erosion and sedimentation levels.

Timber harvesting can also lead to soil compaction and increased soil erosion, adversely impacting soil and vegetation productivity (Busse et al. 2014; Ponder et al. 2012; Yoho 1980). Physical soil changes due to compaction have been enumerated by many (see Page-Dumroese et al. 2006) and can include decreases in soil porosity; disturbance to the organic layer (Robichaud et al. 1993); rooting volume and aeration; increases in soil bulk density, strength, and water content; and reduced infiltration rates and subsequent increased surface runoff and erosion (Greacen and Sands 1980). Compaction impacts are site-specific, with varied effects on forest stand productivity (Froehlich and McNabb 1984; Gomez et al. 2002; Greacen and Sands 1980). In addition to the disturbance associated with felling operations, a decrease in the number of trees within a stand results in a decrease in evapotranspiration, which contributes to increased surface flow, stream flow, and even channel erosion (Elliott et al. 1996). Most compaction studies have again focused on operations that use heavy machinery, and therefore results may not be easily extrapolated to small-scale non-mechanized projects.

Soil erosion resulting from forest thinning operations in turn impacts forest productivity by decreasing soil water availability (Swanson et al. 1989), removing plant-available nutrients, and causing degradation of soil structure (Elliott et al. 1996). Removal of the loose, organic surface materials promotes surface sealing and crusting, which decrease infiltration capacity and may increase erosion (Childs et al. 1989). Erosion also results in loss of important soil biota, such as mycorrhizal fungi, which facilitate nutrient uptake by plants (Amaranthus et al. 1989, 1996). Forest management can directly and indirectly change nutrient stores at a site. Vegetation harvest removes nutrients in wood and/or crowns, immediately affecting local nutrient pools (Powers et al. 2005). The greatest concentration of nutrients and maximum water-holding capacity are in the uppermost soil horizons; loss of these surface layers from erosion is therefore most damaging to forest productivity (Moghaddas 2013). Shallow soils are the most at risk, and therefore the largest declines in productivity are most likely to occur in marginal dry environments (Elliott et al. 1996).

Impacts to soils from thinning operations have been described as extraordinarily complex, reflecting interactions among disturbance levels, soil water-holding capacities, nutrient cycling properties, and climate (Elliott et al. 1996). As observed by Childs et al. (1989) and Dickerson (1976) environmental degradation and erosion resulting from thinning operations could be attributed to various factors, including compaction, soil surface disturbance, depletion of surface organic horizons, and removal of vegetative cover.

2.01.3 HYDROLOGY

Numerous studies worldwide have demonstrated that changing forest density can in turn change forest water yield (Baker 1986, 2003; Bosch and Hewlett 1982; Brown et al. 1974; DeBano et al. 2004; Douglass 1983; Gary 1975; Harr 1983; Hibbert 1967; Hornbeck et al. 1997; Kattleman and Ice 2004; Keppeler and Zeimer 1990; National Research Council 2008; Reinhart et al. 1964; Robles et al. 2014; Stednick 1996; Troendle 1983; Troendle and Leaf 1980; Troendle et al. 2010). In general, reducing forest cover has been found to increase water yield, though stream flow response has also been found to be closely related to climate, particularly the amount and timing of precipitation (Troendle et al. 2010). Stednick (1996) reported that in a review of 95 watersheds, annual runoff increased by nearly 2.5 millimeters (mm) for each 1% of watershed landscape harvested. Various studies have reported that approximately 20% of the basal area of the vegetation must be removed in order to see significant changes in annual runoff (Bosch and Hewlett 1982; Hibbert 1967; Stednick 1996). Others have found that runoff increases were negligible for basal area reductions below 30%, and that runoff increases were contingent upon time since treatment and winter precipitation (October–April) thresholds (Baker 2003; Brown et al. 1974; Robles et al. 2014). A recent study by Biederman et al. (2015) did not observe consistent changes in runoff from forests experiencing extensive bark beetle-induced die-off. They compared gauged streams from catchments before and after infestation, as well as looking at matched control watersheds and conclude that their results are consistent with increased transpiration by surviving vegetation and increased snow sublimation and evaporation following die-off (Biederman et al. 2015). An application of experimental rainfall to a thinned New Mexico forest did not detect significant effects of thinning on time to peak runoff, time to runoff initiation, runoff ratio, or sediment yield, suggesting that thinning can be accomplished without increasing erosion potential (Garduno 2015). However, Baker (2003) reported increased runoff of 15% to 40% when a 30% to 100% reduction in basal area of ponderosa pine forests was implemented.

Troendle et al. (2010) discuss the importance of assessing the degree to which the management activity alters net precipitation to the soil by altering interception losses and infiltration characteristics and the soil moisture evaporation and transpiration. The timing of a change in stream flow within a year depends on when precipitation or snowmelt exceeds both evapotranspiration demand and soil moisture recharge requirements. The Fool Creek watershed study in central Colorado has been ongoing since the late 1960s and demonstrated that for the first 15 years after thinning treatments, there was increased average annual and peak runoff flows by up to 16.2 cm (6.4 inches) a year, with an average increase in water yield of 10.0 cm (3.9 inches) in the first year after thinning. Increases were attributed to reduced evapotranspiration in the thinned portion of the watershed, decreased interception, and increased amount of water contained in the snowpack. By 28 years post-harvest, regrowth in the thinned areas caused a significant decline in average water yields (Troendle et al. 2010). Similar hydrologic responses to thinning have been reported in other studies in the Rocky Mountain region, including Wagon Wheel Gap, Colorado (Bates and Henry 1928), Dead Horse Creek, Wyoming (Troendle and King 1987), Coon Creek, Wyoming (Troendle et al. 2001), and South Dakota thinning (Anderson 1980). Troendle et al. (2010) suggest that lessons learned from the higher elevation watersheds widely studied can be applied to lower elevations but reduced precipitation levels (particularly in the form of snowpack) in these areas must be considered when predicting the intensity of the hydrological response. This is borne out by Haupt (1979) who found that in drier ponderosa pine forests, a reduction in basal area did not detectably increase the snow water equivalent on south, east, and west aspects, but did substantially increase the snow water equivalent on north-facing slopes.

Thinning treatments can also reduce soil moisture depletion and evapotranspiration, especially in wet years (Dietrich and Meiman 1974; Troendle 1987, 1988; Troendle and Kaufman 1987; Troendle and Meiman 1984). In dry years, residual trees may use any additional soil moisture created as a result of thinning. This means that the relationship between stand density and soil water depletion is statistically significant in wet years when there is less competition for soil water, while in dry years, there may be no correlation between basal area and soil water depletion because evapotranspiration from the residual stand may use all of the available water, regardless of the reduction in stand density. During drought periods, summer precipitation is low and soil water reserves are often depleted on all aspects and across a wide range of stand densities and forest types. Under these conditions, tree thinning treatments may not cause an increase in annual water yields unless precipitation amounts exceed evaporative demand (Troendle et al. 2010).

Robles et al. (2014), in a modeled study of thinning impacts on runoff in Southwestern ponderosa pine, demonstrate that modeled runoff from thinned forests was approximately 20% greater than unthinned forests and that runoff gains occurred during droughts and pluvials.

Similar to historic studies, the researchers agree that runoff gains were temporary, ceasing 6 years after thinning. Baker (2003) hypothesize that thinning effects on runoff in the historic Bearver Creek watershed study were short lived due to regeneration of the understory vegetation. The Robles et al. (2014) study evaluates thinning and runoff as it relates to the work proposed under the Four-Forest Restoration Initiative (4FRI) (USFS 2013). The 4FRI is a congressionally funded program to accelerate mechanical thinning and prescribed burning across four national forest with the objective of re-establishing forest structure, pattern, and composition to improve forest resiliency and function. Robles et al. (2014) suggest that the accelerated thinning, as proposed under 4FRI, can improve surface water runoff, a key ecosystem function, and provide other indirect benefits to soil moisture and productivity.

A number of studies have found that runoff gains resulting from thinning were best predicted by winter precipitation totals (Baker 1986; Brown et al. 1974; Robles et al. 2014). Robles et al. (2014) report increased runoff totals even during drought years in thinned forests. They also suggest that the same thinning intensity in a pluvial year could generate double the additional runoff as compared to unthinned watersheds. This has implications for forest management decisions if drought and pluvial periods can be predicted. Objectives to increase runoff could more likely be met by thinning in a pluvial period, while objectives to reduce catastrophic wildfire or drought mortality could be met through thinning during drought periods (Robles et al. 2014).

In 2013 Wyatt reviewed 37 studies worldwide (31 peer-reviewed articles) to answer the questions “how do restoration thinning treatments conducted in conifer-dominated watersheds affect the water budget?” and “how do restoration thinning treatments impact the groundwater system?” Results from those studies showed that water yield can increase from 10% to 35% when 20% to 100% of a conifer-dominated watershed is treated. Groundwater results were inconclusive. All studies showed a positive response of surface water yield resulting from forest thinning treatments, but responses varied across climatic types. Wyatt (2013) suggests that additional research and reviews are needed to address the uncertainties and variances found across studies of forest treatment effects on surface water yield and groundwater recharge. Literature to date suggests that in general the greatest hydrologic response will be detected when more than 20% of the watershed is thinned and for dry forests the response may be only negligible or difficult to detect except during wet years.

2.01.4 TREES

Thinning and prescribed fire have been implemented in Southwest forests and woodlands in recent years as a means of returning stands to more historical conditions, thereby decreasing wildfire risk and improving forest health (Feeney et al. 2008; Keane et al. 2002; Sala et al. 2005). Researchers have shown how thinning of these forests is effective in increasing individual tree growth (Feeney et al. 1998; Ronco et al. 1985; Skov et al. 2005), decreasing tree water stress (Kolb et al. 1998; Skov et al. 2005; Wallin et al. 2004), increasing tree defense against bark beetles through increased resin production (Kolb et al. 1998), and increasing leaf nitrogen concentration and hence photosynthetic capacity in some cases (Feeney et al. 1998; Wallin et al. 2004; Zausen et al. 2005).

Fulé et al. (2007) studied long-term ecosystem response to ponderosa pine thinning treatments in Arizona. They reported that ponderosa pine trees grew significantly faster in treated units than in controls, enough to reach the reference level of basal area in 6 years. However they also reported increased post-treatment mortality of large trees in treated units (10.9 large trees per hectare [ha]), compared to control units (6.2 trees per ha). They suggested that although mortality of large trees is a concern, the treated units had vigorous growth and low density, indicating that they would be relatively resistant to future drought and fire events. Sala et al. (2005) found that in the short term (1–3 years), thinning alone or thinning followed by burning had resulted in increased soil water availability and improved physiological performance of second-growth (Skov et al. 2005) and old-growth (Feeney et al. 1998) ponderosa pine. They reported that, despite minimal differences in soil resource availability, trees in managed units where basal area was reduced had improved gas exchange and growth compared with trees in unmanaged units.

Bark Beetles, Tree Parsites and Disease

Much attention has been placed in recent years on the effects of thinning on bark beetle infestations throughout the western United States. For over a decade, Southwest forests and woodlands have been subjected to increased drought, insect infestation, and disease, which have resulted in a decline in forest health (Clifford et al. 2008; Shaw 2008). Mortality from drought and bark beetle infestation of ponderosa pine, piñon/juniper, and other forest and woodland species throughout the Southwest region increased dramatically between 2000 and 2003 (Zausen et al. 2005). Piñon pine was especially affected, with 774,771 ha of piñon across New Mexico and Arizona showing evidence of bark beetle attack by 2003. Some areas experienced greater than 90% piñon mortality (Gaylord et al. 2013), while juniper mortality was significantly lower. Piñon mortality was largely a result of the piñon ips bark beetle (*Ips confuses*) (Figure 2.1), which generally attacks water-stressed or recently dead trees (Raffa et al. 2008; Rogers 1995). A plethora of recent research has focused on the effects that restoration treatments have on the species resistance/susceptibility to bark beetles in ponderosa pine forests (Gaylord 2014).

Figure 2.1. Piñon beetle (*Ips confuses*). This individual has been caught in pine sap or pitch, the usual defense against the beetle.

The piñon ips bark beetle is always present at low numbers in piñon woodlands, attacking unhealthy trees. When persistent drought occurs and piñon pines become water stressed and the trees produce a "stress scent" (i.e., beta-pinene), this attracts the beetles. Trees that were already stressed preferentially succumbed to severe drought in 2002: results suggest that sapwood cavitation, low carbon assimilation and low resin defense predispose piñon pine trees to bark beetle attacks and mortality during severe drought (Gaylord, 2015). Because of water stress, the trees are not able to fight the beetles with their usual defense of resin (sap or pitch), so the beetles are able to bore into the inner bark or cambium. Therefore, piñon ips outbreaks are a symptom of drought, and the beetles are able to attach to otherwise healthy trees because of drought and water stress. During drought, the beetle populations grow rapidly into outbreak situations because so many trees are vulnerable. The *Ips* beetles slowly kill the trees by boring through the bark and laying eggs from which larvae hatch; the larvae begin feeding on the inner living bark or cambium, cutting through the cambium where tree sap flows, cutting off sap flow, and essentially girdling the trees. The beetles also carry blue stain fungus into the attacked host tree, which grows in the vascular tissue and clogs the tree's ability to draw water and transport carbon and nutrients. The combination of larval feeding on cambium and the fungus infection generally proves fatal to the tree (Christopherson 2013), especially if the tree is already water stressed.

There is wide acceptance globally of a positive relationship between drought stress and beetle attack in trees (Huberty and Denno 2004; Jactel et al. 2012; Raffa et al. 2008). Biotic and abiotic stresses such as high inter-tree competition, defoliation, lightning strikes, and fire damage are also thought to influence tree susceptibility to bark beetle attack (Berryman 1976; Bradley and Tueller 2001; Christiansen et al. 1987; Fettig et al. 2007; Ruel et al. 1998; Wallin et al. 2003). Another agent responsible for piñon damage in the region is the piñon needle scale (*Matsucoccus acalyptus Herbert*), a native sap-sucking insect that can defoliate branches, killing small trees or weakening trees to the extent they fall victim to the ips beetle.

Drought stress can affect trees physiologically, impacting their hydraulic function (McDowell et al. 2008; Ryan 2011; Sala et al. 2010), reducing carbohydrate production, and reducing carbon available for resin production (McDowell et al. 2008; Sala et al. 2010). Thinned stands of several pine species have been reported to be less susceptible to tree-killing bark beetles (e.g., Amman et al. 1988; Brown et al. 1987; Fettig et al. 2007; Gaylord 2014; Mitchell et al. 1983; Sartwell and Stevens 1975; Schowalter and Turchin 1993). Research in northern Arizona, for example, has suggested greater ponderosa pine resistance to bark beetles, based on higher resin flow and overall improvements in tree vigor, in thinned or thinned and then burned stands compared to unthinned stands (Feeney et al. 1998; Kolb et al. 1998; Wallin et al. 2004). Other research has attributed improved resilience to bark beetle in thinned stands to changes in microclimate (temperature, wind movement, and stand structure) (Gaylord 2014). Some studies have proposed thresholds below which bark beetle attacks on ponderosa pine are less probable (Gaylord et al. 2010; Negrón et al. 2000; Negrón and Popp 2004); however, an established threshold basal area still remains inconclusive (Gaylord 2014).

To examine the impacts of the recent drought in New Mexico, Gaylord et al. (2013) tested the hypothesis that drought predisposes trees to insect attack in New Mexico. They quantified the effects of water availability on insect attacks and mortality of piñon pine and oneseed juniper. They found that piñon began dying 1 year after drought initiation, with higher mortality in plots that had removal of 45% of the ambient annual precipitation, relative to other treatments (i.e., irrigation to produce 125% of ambient annual precipitation, a control, and ambient precipitation). They found that beetles (both bark and twig) were present in 92% of dead piñon trees. For juniper, treatments had no effects on insect attack or resistance but needle browning was highest in the plots that underwent water removal. They concluded that their results provided strong evidence that more than 1 year of severe drought and water stress predisposes piñon to insect attacks and increases mortality, whereas 3 years of the same drought only causes partial canopy loss in juniper.

Zausen et al. (2005) reported that thinning stands to lower tree densities (with and without ensuing prescribed fire treatments) decreased ponderosa pine water stress during the peak of the dry season (late June) 8 to 16 years after thinning and 3 to 10 years after the most recent prescribed burn, compared with unmanaged stands in northern Arizona. They attributed these results to increased water availability to trees resulting from decreased tree competition in thinned stands. Under predicted climate change scenarios of increasing drought in the southwest, Gaylord (2014) states that thinning treatments may mitigate water stress and thereby lessen the likelihood of bark beetle–driven tree mortality.

Piñon and juniper in the region are also infected with dwarf mistletoe (*Arceuthobium divaricatum*) (Figure 2.2) on pines and true mistletoes (*Phoradendron* spp.) on juniper. They are small parasitic flowering plants that draw water and nutrients from the host tree, impacting the tree's normal growth and reproductive processes. Increased mortality is associated with severe mistletoe infestations (Mathiasen et al. 2002), but even in minor cases, dwarf mistletoe and true mistletoe are thought to increase the host tree's susceptibility to other damaging agents such as insect and disease.

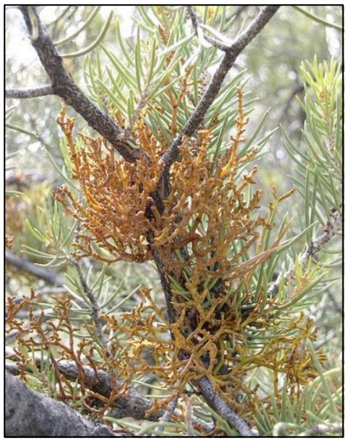


Figure 2.2. Dwarf mistletoe (*Arceuthobium divaricatum*) growing on piñon.

2.01.5 UNDERSTORY VEGETATION

A major objective of ponderosa pine restoration is to increase understory and shrub production (Korb 2001), making herbaceous production a critical response variable for monitoring the effects of treatments during ponderosa pine forest restoration. Most research to date has explored the effect of thinning and fire on understory production, and generally indicates that thinning and burning increases understory vegetation biomass (Abella 2009; Fulé et al. 2001; Laughlin and Fulé 2008; Moore et al. 2006; Stoddard et al. 2011; Stoddard and McGlone 2008; Wienk et al. 2004, Thomas, 2015, Jacobs, 2015). Studies of the effects of thinning on understory species have been reviewed by Korb and Springer (2003), with a general conclusion that understory productivity in ponderosa pine is inversely related to the density of the overstory trees (Laughlin et al. 2005; Smith 2011).

Thinning treatments to reduce overstory density have repeatedly been shown to increase understory productivity, particularly when pre-treatment stands are dense (Bedunah et al. 1988; McConnell and Smith 1970; Metlen and Fiedler 2006; Moore and Deiter 1992; Thysell and Carey 2001). For example, Jacobs (2015) found total understory cover increased several-fold at 3 to 5 years post-treatment. Understory responses in dry forests are thought to be driven by changes in availability of limiting resources, primarily nitrogen and water (Coomes and Grubb 2000; Kolb and Robberecht 1996; Riegel et al. 1992, 1995). Stoddard et al. (2011) found that plant species richness was positively related to both the percent change in canopy cover and basal area as a result of tree removal. Total plant cover was always greater in all treated units than in control units. Plant cover was positively correlated to both the percent change in canopy cover and the tree basal area.

Smith (2011) found that precipitation is a strong determinant of understory response following thinning, concluding that long-term drought can compromise the ability of vegetation to respond to management. Climate influences on understory response have been discussed in a number of studies (Bataineh et al. 2006; Fulé et al. 2002; Moore et al. 2006; Sabo et al. 2008), with a general finding of a strong positive correlation between annual precipitation and understory productivity and diversity (Abella and Covington 2004; Smith 2011).

Matchett et al. (2010) evaluated short-term effects of thinning methods on Southwest piñon/juniper woodlands. They found that thinning treatments increased the abundance of herbaceous vegetation, with pre-treatment tree dominance dictating the strength of the increase. Increases in perennial grass cover and density in response to thinning was usually greatest at lower levels of pre-treatment piñon/juniper dominance. Native annual forb cover and density responded fairly equally along the tree dominance gradient. Shrub abundance declined in response to pre-treatment tree dominance, and the response to thinning treatments appeared more subtle than for herbaceous vegetation. Species richness within the two thinning treatments steadily increased relative to the control over the course of the 3 years following treatment. Species richness was also consistent across the piñon/juniper dominance gradient. Stimulation of herbaceous cover may have implications for fire spread by enhancing continuity of surface fuels especially during dry years. Matchett et al. (2010) concluded that thinning-induced increases in perennial grass cover in areas of high tree dominance were mainly due to an increase in growth of individuals present prior to the treatment, as opposed to an increase due to the recruitment of new individuals. Ramirez et al. (2008) report similar finding in piñon/juniper forest in New Mexico, where thinning treatments increased herbaceous vegetation cover and biomass.

Since 1990 researchers and land managers have teamed up to experiment with restoration techniques in piñon/juniper woodlands (Jacobs et al. 2002). The primary restoration treatment (thinning and application of slash mulch) in this study was demonstrated to be an effective remediation technique for increasing herbaceous cover, stabilizing soils, and supporting surface fire. Monitoring showed that the restoration treatment also increased the resilience of vegetation to drought effects.

According to a review of research by Abella (2009), previous studies did not consistently show an increase in ground flora diversity in ponderosa pine forests. Studies that have shown significant increases in species richness with treatment (Laughlin and Fulé 2008; Metlen and Fiedler 2006; Moore et al. 2006) have reported a minimum threshold basal area required (down to 10 square meters [m^2]/ha) before a significant increase in herbaceous production can occur. Previous research has also shown similar relationships between ground flora production and basal area. Clary and Ffolliott (1966) found that ground flora biomass was higher in thinned stands (compared with unthinned stands) with residual basal area of 5 to 18 m^2/ha, but there was no significant difference among treatments when post-thinning basal area exceeded 18 m^2/ha.

Abella and Covington (2004) found that total mean species richness per square meter did not differ significantly among control, low-, and medium-intensity thinning treatments, but high-intensity treatment areas did yield a richness twice as high as other treatments. This indicates that a lower limit stand density threshold needs to be passed before species richness increases. Few studies have assessed community composition following thinning in Southwest forests. Only Abella and Covington (2004) have statistically evaluated overall community compositional differences among treatments. They reported subtle but positive native species compositional differences between control plots and thinned and burned plots 3 years after treatment. Many authors have noted the need for future research to distinguish the effects of mechanical thinning on understory dynamics from the effects of prescribed burning (Abella 2009; Fulé et al. 2001). Abella revisited this question in 2015 and concluded that plant community responses of species richness, cover, and composition were diverse and depend on soil partent material, the specifics of thinning implementation, and the presence or exclusion of grazing (Abella et al, 2015).

In a review of the National Fire and Fire Surrogates study by McIver et al. (2013), the authors state that most literature on the effects of treatment on ecosystem processes finds that standard fuel treatments generally cause modest effects on most components of dry-forest ecosystems, the magnitude of effects correlates well with the intensity of the treatment and most variables quickly recover to pre-treatment levels.

Exotic Invasive Weeds

Non-native or exotic invasive weed species' response to thinning has been studied by various researchers (Allen et al. 2002; Crawford et al. 2001; Griffis et al. 2001; Hunter et al. 2006; Keeley 2006; Korb and Springer 2003; Stoddard and McGlone 2008). Thinning has generally been found to promote the establishment of exotic weed species (Hunter et al. 2006). Nelson et al. (2008) found that exotic species showed small but highly significant increases in cover and richness in response to both thinning and burning. Cover and richness of exotic herbs showed small increases with intensity and disturbance and time since treatment. In contrast they found no significant effect of thinning or burning on understory plant composition, nor significant differences among treatments in canopy cover and species richness of native plants. Griffis et al. (2001) report a stronger response by exotic species than native species to thinning in terms of species richness and abundance.

Stoddard and McGlone (2008), studying a Southwest ponderosa pine forest, found that disturbances associated with restoration treatments facilitated the establishment of exotic weed species. Exotic weed species abundance and richness increased significantly in response to treatment intensity. Within 2 years of treatment, exotic species made up 50% and 45% of the indicator species in the medium- and high-intensity plots, respectively. After 2 years, exotic species had declined in proportion to native species cover and richness, and after 6 years exotic species were only a minor component of the plant community.

The exotic invasive weed cheatgrass (*Bromus tectorum*) is a concern in piñon/juniper-dominated woodlands following treatment (Matchett et al. 2010). Application of wood chips following mastication has been shown to reduce the dominance of cheatgrass (Wolk and Rocca 2009), which may be a result of increased soil carbon from the mulch leading to reduced soil nitrogen levels that inhibit growth of invasive plants. Alternatively, the mulch may significantly shade the soil surface and inhibit seed germination and seedling growth. Reduced dominance of exotic invasive weeds such as cheatgrass can lead to increased density, cover, and diversity of native species, which could provide an additional benefit of tree thinning treatments.

2.01.6 BIRDS

The effects of thinning on bird populations have most often been based on observational studies after forest treatments or post-fire salvage logging or wildfire (Bock and Block 2005; Kotliar et al. 2002; Saab and Powell 2005; Saab et al. 2004). According to Hutto et al. (2014), birds are a highly effective and useful ecological indicator group since large numbers of species can be detected using a single method (Hutto 1998; Hutto et al. 2014). This is especially important in evaluating forest restoration because each species is associated with a distinct vegetation condition, and their community structure is sensitive to forest structure (Hutto et al. 2014). Many authors have found that the removal of small-diameter trees typical of fuel reduction treatments has a neutral to positive effect on avian species (Gaines et al. 2010; Hurteau et al. 2008; Kalies et al. 2009; Verschuyl et al. 2011; White et al. 2013), though studies have revealed that responses are generally species specific and/or vary over time, attributable to the pace of vegetation response of the understory and overstory strata (Yegorova 2013). Yegorova (2013) found that bird-vegetation relationships are highly dynamic, which the author attributes to intrinsic population processes rather than plasticity in avian habitat selection.

Kalies et al. (2009) in a study of wildlife responses to thinning on Southwestern conifer forests found that at the guild level, aerial foraging birds benefited from small-diameter tree removal, but they had negative responses to large-diameter tree overstory removal. Ground shrub–foraging birds responded positively to overstory removal, suggesting that the treatment was effective in maintaining or enhancing understory and shrub cover (Ffolliott and Gottfried 1989; Yorks et al. 2000). Woodpeckers, however, declined following overstory removals. The occurrence probability of bark foragers and seed eaters was more closely associated with abiotic variables—that is, annual variability of food resources like bark beetles (Gaylord 2014), seed mast, and composition of tree species. Foliage insectivores, which glean invertebrates from foliage of trees and shrubs, were associated with higher tree cover and fuel reduction that reduced cover of these species impacted the foliage insectivores. Bark gleaners responded differently, with their response being related to intermediate canopy cover and reduced shrub cover. Gaylord (2014) relates bark beetle infestations, like those observed across the Southwest, with providing insectivorous avian species with increased food, as well as creating habitat for cavity-nesting birds.

Hurteau et al. (2008) reported an increase in western bluebird (*Sialia mexicana*) densities as a result of increasing foraging opportunities post thinning (Hurteau et al. 2008). Wightman and Germaine (2006) had similar findings—treatments to reduce tree densities and increase herbaceous vegetation provide a more abundant food source and improved habitat quality for bluebirds. Mountain chickadees (*Poecile gambeli*) have been found to be negatively affected by timber management practices (Hurteau et al. 2008); reductions in density of the species are consistent with other studies (Franzreb 1978; McCallum et al. 1999).

A study by Hurteau et al. (2008) found that treatments to reduce forest fuels had little effect on avian diversity over 4 years, but did affect some aspects of species composition and abundance. Their results suggest that although the small-scale forest treatments they studied may have influenced the avian species present, natural annual variation in density is a stronger source of variation. Similarly Szaro and Balda (1986) found that various intensities of forest thinning treatments influenced bird density and species richness, but treatments had a greater influence on community composition.

White et al. (2013) used computer simulations to evaluate avian response to fuel reduction treatments in coniferous forests. They suggested that although fuel reduction treatments may provide or improve suitable habitat for some species, treatments may cause an overall but minimal cumulative reduction in species richness. Simulations also showed that treatments that created a more complex stand structure increased the occurrence probability by greater than 30% for a larger number of avian species than typical fuel reduction methods and led to smaller predicted reductions in species richness. Their models suggest that a greater number of avian species would be retained by using a treatment that adds or retains forest structural heterogeneity. Kalies et al. (2009) similarly found that a mosaic of forest conditions may be the most appropriate technique for providing suitable habitat for a wide range of forest passerines. They suggest that landscape-level forest treatments applied by land managers throughout the country will have only modest effects on avian species.

2.01.7 RODENTS

The effectiveness of thinning to either promote or maintain habitat has been demonstrated for a number of forest-floor specialist and generalist rodent species (Carey and Wilson 2001; Gitzen et al. 2007; Hayward et al. 1999; Klenner and Sullivan 2003; Suzuki and Hayes 2003; Wilson and Carey 2000). Thinning treatments have increased forest spatial and temporal heterogeneity, diversified habitat available for wildlife, and in turn restored a native, diverse assemblage of animal species (Allen et al. 2002; Noss et al. 2006), with any significant management action likely to favor some species over others (McIver et al. 2013). McIver et al. (2013) in a review of fire surrogate studies across the United States summarize that species that favor drier microhabitat conditions have been found to respond more positively to management actions that increase heat and light at the forest floor (Huang et al. 2007), expose bare mineral soil (Boerner et al. 2009), increase grass cover due to decreased shrub cover (Collins et al. 2007), and increase within stand heterogeneity (Gundale et al. 2006).

Several studies have found early and positive responses of small forest-floor mammals to thinning (Converse et al. 2006a, 2006b; Muzika et al. 2004; Sullivan et al. 2005; Suzuki and Hayes 2003; Wilson and Carey 2000; Wilson and Forsman 2013) with numerous authors reporting that reductions in canopy cover may increase herbaceous plant and shrub cover (Bagne and Finch 2010; Block et al. 2005; Carey and Johnson 1995; Converse et al. 2006b; Lee et al. 2008). This understory response leads to increased structure and plant diversity on the forest floor, providing food, shelter, and protective cover for small mammals. Responses were generally strongest in forests that originally lacked understory cover and shrub components (Wilson and Forsman 2013).

In a 6-year study of thinning effects on small mammal populations in ponderosa pine forests, Bagne and Finch (2010) found positive or neutral effects of thinning on the small mammals examined. Out of 4 years of post-thinning data, positive effects lasted for up to 3 years post-thinning. As found by other authors, positive effects were attributed to increases in downed woody debris (Converse et al. 2006a, 2006b; Manning and Edge 2004), herbaceous understory plants (Converse et al. 2006a; Manning and Edge 2004; Suzuki and Hayes 2003), and habitat heterogeneity (Carey and Wilson 2001; Muzika et al. 2004). Bagne and Finch (2010) reported a lack of negative effects on small mammals, which they suggest indicates that ecosystem function remains intact following large-scale thinning with minimal soil disturbance in this watershed. Some species were positively affected for a short period as well. Bage and Finch (2010) also found that precipitation, flooding potential, and capture probabilities are important when examining changes in small mammal populations and likely influenced the timing of small mammal responses to thinning; thus, abiotic influences need to be considered when evaluating treatment effects. McIver et al. (2013), in a comprehensive review of the literature, suggest that across a broad spectrum of ecosystems, treatment responses tended to be subtle or non-existent, suggesting a single entry of mechanical treatment is unlikely to cause major or persistent changes in most ecosystem properties. Any changes that did occur where subtle and transient, lasting only 1 to 3 years (Boerner et al. 2009; Coates et al. 2008).

The Northern Arizona University Ecological Restoration Institute (2010) evaluated post-treatment time periods of ponderosa pine forest thinning treatments and found that different periods of time since treatment was an important factor affecting the densities of four key species over time. They concluded that species associated with denser cover were the only ones to increase in occupancy with increased density. The presence of slash piles and duration of the slash piles' presence produced positive occupancy responses from all but one small mammal, the pocket gopher (Geomyidae), a burrowing species. They acknowledged the importance of downed wood as an important habitat feature for some members of the small mammal community, but concluded that the presence of downed wood is less important than overstory and understory vegetation composition and structure. The presence of downed wood or slash is important for some species (Chambers 2002; Converse et al. 2006a), particularly deer mice (*Peromyscus maniculatus*), because of its use for cover, nesting, and food. A number of researchers have found that most ground-dwelling rodents responded positively to small-diameter tree removal and that deer mouse densities increased following treatment (Kalies 2010; Kalies et al. 2009; Zwolak 2009). Converse et al. (2006a) also found that although woody debris created during thinning operations may provide greater protective cover for small mammals, the eventual removal of these materials may result in reductions of small mammal populations. Further, thinning operations may open forests, increasing the success of predators hunting small mammals (Gese et al. 1995).

2.01.8 WILDFIRE

Historically, wildfires have played a key role in maintaining the proper functioning of ponderosa pine forests in the American Southwest. Research has shown that pre-settlement fire return intervals ranged from 2 to 15 years in southwestern ponderosa pine stands (Swetnam and Baisan

1996; Swetnam and Betancourt 1998), including the Manzano Mountains in central New Mexico. However, forest management practices (wildfire suppression) coupled with intensive livestock grazing (loss of herbaceous understory fuels) over the past century has greatly limited natural fires and their ecological effects on these ecosystems. Ponderosa pine forests were once composed of "park-like" stands with considerable understory vegetation and with heterogeneous spatial stand structure patterns (Allen et al. 2002), largely the result of the repeat occurrence of surface fires. In many watersheds throughout the Southwest, over 90 percent of ponderosa pine forests are considered at high risk of crown fires because of dense structure and unnaturally high levels of accumulated fuels (Allen et al. 2002; Covington and Moore 1994a, 1994b). Given the state of these forests, high-severity wildfires are now commonplace and are associated with a number of significant and undesirable ecological impacts (Covington and Moore 1994a; Fulé et al. 1997). High-severity wildfires have the ability to disrupt, damage, and destroy ecosystem functioning through the consumption of the under- and overstory vegetative cover and the protective litter and duff layers.

The damaging effect of present-day wildfires on ecosystem function, in particular vegetative response, has been well documented throughout the literature (Abella et al. 2012a; Brown et al. 2000; Campbell et al. 1977**;** Ffolliott et al. 2008; Neary et al. 2008), with many studies relating plant response to burn severity (Lentile et al. 2007; Lyon and Stickney1976; Ryan and Noste 1985). Lentile et al. (2007) in a study of eight wildfires found that post-fire vegetation species richness varied highly among patches burned with low, moderate, and high severity. They state that variation could be attributed to fine-scale variability in post-fire effects to soil, the pre-fire vegetation cover, and the degree of resilience of the pre-fire vegetation to fire. Others have found post-fire plant composition to be correlated with post-fire climate (Whelan 1995), litter and duff consumption and forest floor condition (Lentile et al. 2007), and seed production (Lyon and Stickney 1976). The relationships between native and exotic species response following fire report various and inconsistent relationships between native and exotic species richness and cover (Abella et al. 2012b, Fornwalt et al. 2010, Hunter et al. 2006, Keeley et al. 2003; Stohlgren et al. 1999). A literature review by Abella et al. (2012) concludes that native and exotic plant species richness and cover have been primarily found to be positively correlated.

In order to mitigate fire effects on ecosystem resources, high-severity burn areas typically undergo post-fire rehabilitation treatments, including the application of mulch, contour felling of trees, and aerial seeding of exotic grasses (Peppin et al. 2014; Robichaud 2000). The later of these techniques, aerial seeding, is the most often used method by land managers due to the ease of application, relative low cost compared to the other techniques, ability to limit the establishment of local exotic invasive and weedy species, and is mandated by U.S. federal polices when economical (Beyers 2004; Peppin et al. 2010; Stella et al. 2010). Pyke et al. (2002) suggested that in the absence of intensive post-fire rehabilitation of native species, exotic invasive exotic species will out-compete many native plants, increasing fire risk and changing the ecology of wildland areas. Species typically used in the seeding process are exotic annuals or short-lived perennials; these species typically have low productive potential in forested systems and/or are sterile hybrids (Beyers 2004; Everett et al. 1990; Peppin et al. 2010;

Robichaud 2000; Stella et al. 2010). Grasses tend to be used most often because of their ability to establish and colonize sites in a short period (Barclay et al. 2004; Everett et al. 1990).

The effectiveness of aerial seeding is still heavily debated and critics suggest that post-fire seeding can suppress the native post-fire herbaceous flora and out-compete shrub and tree seedlings (Beyers 2004). Several recent studies have addressed the effectiveness of post-wildfire seeding (Barclay et al. 2004; Hunter et al. 2006; Peppin et al. 2010; Stella et al. 2010), but long-term studies (>2 years) are still lacking (Peppin et al. 2014). Consensus does exist, however, that the success of post-fire seeding can be largely attributed to weather conditions within the treatment area (Peppin et al. 2014), especially the amount and timing of precipitation events (Peppin et al. 2010). A literature review on post-wildfire seeding of exotic grasses in the western United States was conducted by Peppin et al. (2010) in order to answer a number of questions on the effectiveness of aerial seeding. The questions posed in this review were: does aerial seeding 1) reduce erosion, 2) does it reduce exotic plant cover, and 3) how does it affect native plant communities? They reported that seeding used for rehabilitation to reduce erosion and exotic plant establishment had no conclusive results. This review and other reviews (Beyers 2004; Beyers et al. 1994; Beyers et al. 1998) also concluded that aerial seeding following a wildfire generally decreases native cover within the first few growing seasons, but there is not enough long-term data to determine how this situation changes through time.

2.02 PROJECT DESCRIPTION

Forest thinning projects on private lands on the eastern slopes of the Manzano Mountains are overseen by the Steering Committee and include projects in both ponderosa pine forests and piñon/juniper woodlands. Forest thinning monitoring has been designed to address forest thinning in both of these forest types, so four monitoring study sites have been established: two in ponderosa pine forests and two in piñon/juniper woodlands. Each ponderosa pine site has been paired with a piñon/juniper site in the same watershed, so that each of the two watersheds has a ponderosa pine and a piñon/juniper monitoring site. One pair of sites is situated at the northern end of the study area (eastern slopes of the Manzano Mountains), and the other is at the southern end (see Figure 1.1). Two paired study plots have been installed at each of the four study sites. Descriptions of physical site characteristics such as slope, aspect, parent materials, plant associations, and habitat types are provided in the 2008 Monitoring Plan (SWCA 2008). Surface elevations of the flumes on the thinning plots can be seen in Table 2.1 below. All study sites chosen are representative of the surrounding area; for example, all sites, excluding the Wester property, undergo a livestock grazing regime, which is typical of the private land use in the Manzano Mountains. One plot from each pair was randomly selected for forest thinning treatments, and the other plot of the pair serves as an untreated control. Parameters being measured for monitoring at each of the eight study plots include rainfall, ambient temperature, soil moisture and temperature, soil chemistry, soil movement, soil surface stability, soil surface hydrology runoff, vegetation canopy cover and species composition, vegetation vertical structure, tree stand structure, density, composition and health, and bird and small mammal species composition and abundance.

Table 2.1. Surface Elevations of the Flumes on the Forest Thinning Plots

Site	Elevation (meters)	Elevation (feet)
Chilili (treatment)	2,288	7,507
Chilili (control)	2,292	7,520
Wester (treatment)	2,267	7,436
Wester (control)	2,275	7,466
Kelly (treatment)	2,114	6,937
Kelly (control)	2,111	6,925
Vigil (treatment)	2,068	6,783
Vigil (control)	2,073	6,802

Actual forest thinning treatments were implemented in November 2010 and were completed by May 2011. This 2015 report presents the fifth year of post-thinning treatment data and comparisons of paired study plots. From 2011 onward, the various environmental parameters being measured have been compared between the treatment and control study plots, and each study plot will be compared to itself over time.

2.03 FOREST THINNING TREATMENTS

One study plot of each forest thinning monitoring pair (plots 1 and 2) was randomly selected to be treated with the standard New Mexico State Forestry prescribed thinning treatment (piñon/ juniper or ponderosa pine prescriptions) in late 2010 and early 2011, with the other plot being left as a control (plots T and C, respectively). The minimum area and boundaries for thinning treatments were determined for each of those four plots and mapped with a sub-meter accuracy global positioning system (GPS) unit in October and November 2009. Those GPS coordinates were used to produce geographic information system (GIS) maps of the treatment areas and boundaries for each of the four treatment study plots (maps of the thinning areas are presented in the 2009 Annual Report [SWCA 2010]). The thinning treatment areas for each of those plots included the entire subwatershed that was previously defined and mapped in 2007, the vegetation/soils measurement plot, and the mammal and bird sampling plot, all within the area of each treatment plot to be thinned. A minimum treatment buffer area of 10 meters (m) (33 feet) was extended from the boundaries of each subwatershed and study plot to ensure that all areas from which soil, hydrology, vegetation, and animal measurements are being collected were thinned on those treatment plots. Table 2.2 shows which plots were treated by tree thinning and which ones remained undisturbed as controls.

Table 2.2. Treated and Control Plots across the Four Monitoring Study Sites

Site	Treated Plot	Control Plot
Chilili	Plot 1	Plot 2
Kelly	Plot 2	Plot 1
Vigil	Plot 1	Plot 2
Wester	Plot 1	Plot 2

Note that results presented above refer to plot number, and all treated plots were plot number 1 except at the Kelly site where the treated plot was number 2.

Tree thinning treatments were conducted as planned and were inspected by New Mexico State Forestry to ensure that all protocols were followed and that the thinning was conducted to the standards developed by the agency for the region for both ponderosa pine and piñon/juniper woodland. In addition to reducing the density of trees on treatment monitoring plots, the thinning process also required that small branches from cut trees be chipped on-site and spread on the ground surface. Large-diameter wood was removed from the sites for firewood. Figure 2.3 through Figure 2.6 show views of both the non-treated control plots and adjacent treatment plots where trees were thinned from each of the four monitoring sites. Plots were photographed in late fall 2010 and early spring 2011, following tree thinning treatments. Note the open structure of the tree stands and wood chips spread over the ground surfaces of the thinned plots.

a. Non-thinned control plot (plot C).

b. Thinned treatment plot (plot T).

Figure 2.3. Kelly piñon/juniper site thinning treatment plot after excess trees were removed in late 2010.

a. Non-thinned control plot (plot C).

b. Thinned treatment plot (plot T).

Figure 2.4. The Vigil piñon/juniper site following tree thinning treatments in late 2010. Note the open stand and wood chips. Stacked wood was removed shortly after the photograph was taken.

a. Non-thinned control plot (plot C).

b. Thinned treatment plot (plot T).

Figure 2.5. The Chilili ponderosa pine site following tree thinning.

a. Non-thinned control plot (plot C).

b. Thinned treatment plot (plot T).

Figure 2.6. The Wester ponderosa pine site in early spring 2011 following tree thinning. The stacked wood was removed in early summer 2011.

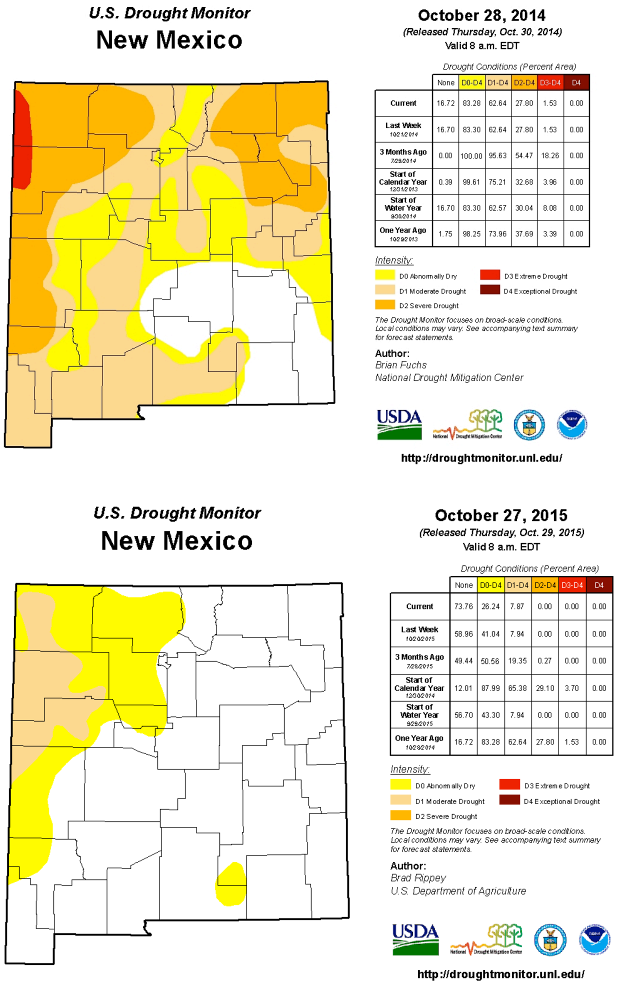
2.04 RAINFALL AND TEMPERATURES

Spectrum WatchDog automated data-logging rain gauges installed at each of the paired vegetation and soils monitoring plots at all of the study sites (see Figure 1.1) have run continuously since they were installed in November 2007 (Figure 2.7). The WatchDog stations are located in openings in the tree canopy in order to reduce effects of interception. Additional details regarding the setup of the weather stations are provided in the 2008 Monitoring Plan (SWCA 2008). The tipping bucket rain gauges on the WatchDog stations are set to record rainfall and snowmelt sums at 1-hour intervals continuously. In fall 2008, a graduated cylinder rain gauge was added to each of the automated rain gauge locations to serve as backups in case of power failure or other malfunction of the data logger (Figure 2.8). These graduated rain gauges and their recorded values are checked monthly when Time Domain Reflectometer (TDR) soil moisture and temperature readings are taken; mineral oil is also added to these gauges at this time to prevent evaporation of water collected. The WatchDog stations are set to record ambient temperature, soil moisture 10 cm (4 inches) below the soil surface (−10 cm), and soil temperature at −10 cm, all at 1-hour intervals. Soil moisture and temperature data from each WatchDog station provide baseline comparisons for the Field Scout TDR 200 soil water content and soil temperature data that are sampled monthly at each study plot. All data from the stations are off-loaded approximately every three months and entered into a database. Summaries for precipitation, ambient temperature, soil moisture, and soil temperature from 2015 on all thinning plots are presented as examples below. Also presented below are long-term graphs of each of these variables (2009–2015) showing any trends that may be occurring climatically within the region.

During the 2015 monitoring period, the drought that occurred in 2011, 2012, 2013 and 2014 ended in the state of New Mexico(Figure 2.9). The project area fell within the category of exceptional drought in 2011, which means there were exceptional and widespread crop/pasture losses and shortages of water in reservoirs, streams, and wells, creating water emergencies. In 2012 and 2013, the drought was categorized as severe. A severe drought can cause water shortages resulting in a loss in crops and pasture lands. In 2014 the moisture that the area received was able to reduce the drought rating to moderate, which can result in some damages to crops, pastures, streams, and wells. Moderate droughts can cause localized water shortages and result in water-use restrictions. In 2015 the moisture the area received effectively ended the long-term drought.

Figure 2.7. WatchDog mini weather station at the Wester ponderosa pine site.

Figure 2.8. Graduated rain gauges are used for backup in the case of failure of one of the WatchDog weather stations.



October 28, 2014
(Released Thursday, Oct. 30, 2014)
Valid 8 a.m. EDT

Drought Conditions (Percent Area)

	None	D0-D4	D1-D4	D2-D4	D3-D4	D4
Current	16.72	83.28	62.64	27.80	1.53	0.00
Last Week 10/21/2014	16.70	83.30	62.64	27.80	1.53	0.00
3 Months Ago 7/29/2014	0.00	100.00	95.63	54.47	18.26	0.00
Start of Calendar Year 12/31/2013	0.39	99.61	75.21	32.68	3.96	0.00
Start of Water Year 9/30/2014	16.70	83.30	62.57	30.04	8.08	0.00
One Year Ago 10/29/2013	1.75	98.25	73.96	37.69	3.39	0.00

Intensity:

🟨 D0 Abnormally Dry
🟥 D3 Extreme Drought
🟧 D1 Moderate Drought
🟫 D4 Exceptional Drought
🟧 D2 Severe Drought

The Drought Monitor focuses on broad-scale conditions. Local conditions may vary. See accompanying text summary for forecast statements.

Author:
Brian Fuchs
National Drought Mitigation Center

http://droughtmonitor.unl.edu/

October 27, 2015
(Released Thursday, Oct. 29, 2015)
Valid 8 a.m. EDT

Drought Conditions (Percent Area)

	None	D0-D4	D1-D4	D2-D4	D3-D4	D4
Current	73.76	26.24	7.87	0.00	0.00	0.00
Last Week 10/20/2015	58.96	41.04	7.94	0.00	0.00	0.00
3 Months Ago 7/28/2015	49.44	50.56	19.35	0.27	0.00	0.00
Start of Calendar Year 12/30/2014	12.01	87.99	65.38	29.10	3.70	0.00
Start of Water Year 9/29/2015	56.70	43.30	7.94	0.00	0.00	0.00
One Year Ago 10/28/2014	16.72	83.28	62.64	27.80	1.53	0.00

Intensity:

🟨 D0 Abnormally Dry
🟥 D3 Extreme Drought
🟧 D1 Moderate Drought
🟫 D4 Exceptional Drought
🟧 D2 Severe Drought

The Drought Monitor focuses on broad-scale conditions. Local conditions may vary. See accompanying text summary for forecast statements.

Author:
Brad Rippey
U.S. Department of Agriculture

http://droughtmonitor.unl.edu/

Figure 2.9. Drought monitor map of New Mexico from the weeks of October 28, 2014, and October 27, 2015, showing the project area located within Torrance County decreasing to no drought as compared to the moderate drought the county faced in 2014 (U.S. Drought Monitor 2015).

2.04.1 PRECIPITATION

Hourly precipitation totals have been summed to monthly totals, and there are similar monthly precipitation totals for the paired study plots at the Vigil PJ study sites (Figure 2.10). The graph in Figure 2.10 shows similar monthly precipitation values for the paired study plots, as was typical at all of the study sites. Annual precipitation values for 2009–2015 averaged for the ponderosa pine and piñon/juniper sites are shown below in Figure 2.11. This figure clearly shows the variability in precipitation values throughout the study period, with 2011 and 2012 below the long-term average of 36.6 cm (14.4 inches) (Western Regional Climate Center 2014) and 2013 and 2014 closer to the average. In 2015 there was an increase in precipitation values at both the piñon/juniper sites and the ponderosa pine sites. The total precipitation received in 2015 at the piñon/juniper sites was 43.31 cm (17.05 inches) as compared to 37.46 cm (14.75 inches) in 2014. The total precipitation at the ponderosa sites was 48.13 cm (18.95) in 2015 as compared to 33.7 cm (13.27 inches) in 2014. The long-term average is from a weather station in Mountainair that has a period of record beginning May 1, 1902 (Western Regional Climate Center 2014).

All tipping bucket rain gauges were functioning properly during the 2015 monitoring season, except for Kelly T, which malfunctioned in 2015.

Figure 2.10. Monthly cumulative precipitation (rainfall and snow) from the two paired Vigil PJ study plots in 2015.

Figure 2.11. Annual precipitation values from 20011–2015 on the piñon/juniper and ponderosa pine sites, which highlight the drought in 2011–2012 and increased precipitation 2013-2015.

2.04.2 AMBIENT TEMPERATURE

An example of monthly averages of hourly ambient temperatures is presented for the Kelly PJ study sites (Figure 2.12). This graph shows similar monthly average ambient temperatures for the paired study plots, as was typical at all of the study sites. The average ambient temperatures are also presented for 2009–2015, which had a steady rise in temperature from 2009 to 2012, then a decrease in average temperature from 2012 to 2013 and then a stabilization in 2014 and 2015 near the long-term average (Figure 2.13). The average temperature at the piñon/juniper (10.6°C [51.12°F]) sites in 2015 was near the long-term average for the area, while the ponderosa pine sites (9.5°C [49.18°F]), which are higher in elevation, had average temperatures that were less than the long-term average for the region (10.8°C [51.5°F]). This average was taken from the long-term weather station located in Mountainair (Western Regional Climate Center 2015).

Figure 2.12. Monthly average ambient temperatures from the two paired Kelly PJ study plots in 2015.

Figure 2.13. Annual average ambient temperature values at the piñon/juniper and ponderosa pine sites, 2009–2015.

2.04.3 SOIL MOISTURE

An example of monthly averages of hourly −10 cm soil moisture readings are presented for the paired study plots at the Vigil PJ site (Figure 2.14). Soil moisture was measured with Watermark soil moisture probes that measure soil water tension in kilopascal (kPa) values that are directly equivalent to California Bearing Ratio (cbr) values for soil water saturation. Results for paired plots were generally similar. More detailed information on the trends in soil moisture can be found in Section 2.5.1 below on soil TDR measurements.

Figure 2.14. Monthly average soil moisture tensions (−10 cm) from the two paired Vigil PJ study plots in 2015.

2.04.4 SOIL TEMPERATURE

An example of monthly averages of hourly −10 cm soil temperature readings are presented for the paired study plots at the Vigil PJ sites (Figure 2.15). The graphs show similar monthly average soil temperatures between the paired study plots (T and C) at both study sites, which was generally the pattern across all sites.

Figure 2.15. Monthly average soil temperature (-10 cm) from the two paired Vigil PJ study plots in 2015.

2.05 SOILS

2.05.1 ENTIRE STUDY PLOT SOIL WATER CONTENT AND TEMPERATURE (TDR)

Continuous hourly soil moisture and temperature measurements recorded by the WatchDog station at each plot only provide a single reference point measurement for each plot, measured and recorded hourly. In order to sample soil moisture and temperature from locations throughout each vegetation and soil monitoring plots, a portable Field Scout TDR 200 soil moisture meter was used. Further information on the detailed methods can be found in the 2008 Annual Report (SWCA 2009).

Average percent soil volumetric water content on the piñon/juniper and ponderosa plots from 2008 through 2015 is displayed below in Figure 2.16 and Figure 2.17. These results show that the piñon/juniper and ponderosa sites are acting in similar fashion prior to the thinning treatments completed in 2011, but after 2012 the piñon/juniper sites showed a decrease in average soil moisture on both control and treatment plots while the ponderosa sites show an increase in average soil moisture on control and treatment plots. Average annual soil moisture between the paired plots is presented below for 2008–2015 from all forest thinning plots (Figure 2.18–Figure 2.21). These figures indicate that the treated sites retain on average more soil moisture throughout the year, especially after storm events and during times of drought. Both PJ and ponderosa pines sites showed the same trends as previous years with the treatment retaining slightly more soil moisture than the controls. Whether these findings continue to persist into the future remains to be seen.

Figure 2.16. Annual average soil moisture percentage for the piñon/juniper sites, 2008–2015; moisture readings were averaged annually from the monthly readings.

Figure 2.17. Annual average soil moisture percentage for the ponderosa sites, 2008–2015; moisture readings were averaged annually from the monthly readings.

Figure 2.18. Average annual soil moisture readings taken at the Chilili site, 2008–2015.

Figure 2.19. Average annual soil moisture readings taken at the Kelly site, 2008–2015.

Figure 2.20. Average annual soil moisture readings taken at the Vigil site, 2008–2015.

Figure 2.21. Average annual soil moisture readings taken at the Wester site, 2008–2015.

2.05.2 SOIL SURFACE STABILITY

Soil surface stability was measured and scored in August 2015 using the Soil Stability Test Kits developed by the U.S. Department of Agriculture Agricultural Resource Service (Herrick et al. 2005) (Figure 2.22). Further details of the measurement methods and a review of the literature can be found in the 2008 Monitoring Plan (SWCA 2008). Figure 2.23 through Figure 2.26 provide average soil surface stability scores for each of the four sample sites for 2008–2015, except for Chilili, which is 2009–2015. Figure 2.27 through Figure 2.30 provide average subsurface (1 cm below the soil surface, or -1 cm) soil stability scores for each of the four sampling sites for 2008–2015, except for Chilili, which is 2009–2015.

In general, the data show there was not much of a change in soil surface or subsurface stability from 2008 to 2015, meaning the thinning practices did not initially affect stability. The data do show, however, that the stability scores are higher on the ponderosa pine sites (Chilili and Wester) than on the piñon/juniper sites (Kelly and Vigil) (Figure 2.31 and Figure 2.32). This difference can largely be attributed to the large accumulation of organic matter that occurs underneath tree canopies in the ponderosa pine vegetation type, which can add as much as 2,000 pounds/acre/year of fine fuels (Ffolliott et al. 1968). Most soils at the sites measured were underneath litter layers and contained organic material and fungi.

Figure 2.22. Soil stability test in use on the study sites.

Figure 2.23. Soil surface stability average scores for Chilili, 2009–2015 (18 subsamples/subplot).

Figure 2.24. Soil surface stability average scores for Kelly, 2008–2015 (18 subsamples/subplot).

Figure 2.25. Soil surface stability average scores for Vigil, 2008–2015 (18 subsamples/subplot).

Figure 2.26. Soil surface stability average scores for Wester, 2008–2015 (18 subsamples/subplot).

Figure 2.27. Soil subsurface (-1 cm) stability average scores for Chilili, 2009–2015 (18 subsamples/subplot).

Figure 2.28. Soil subsurface (-1 cm) stability average scores for Kelly, 2008–2015 (18 subsamples/ subplot).

Figure 2.29. Soil subsurface (-1 cm) stability average scores for Vigil, 2008–2015 (18 subsamples/ subplot).

Figure 2.30. Soil subsurface (-1 cm) stability average scores for Wester, 2008–2015 (18 subsamples/subplot).

Figure 2.31. Soil surface stability average scores for the piñon/juniper and ponderosa sites, 2008–2015.

Figure 2.32. Soil subsurface (-1 cm) stability average scores for the piñon/juniper and ponderosa sites, 2008–2015.

2.05.3 SOIL MOVEMENT

Soil movement was monitored using soil movement bridges (called soil erosion bridges in the 2008 report) (Figure 2.33) modeled after White and Loftin (2000). Permanent bridge support posts were installed at consistent, systematically determined, and unbiased locations at one of each of the vegetation and soil subplots for a total of three bridges at each paired plot at all four sites. Please refer to the 2008 Annual Report for detailed monitoring protocols and literature associated with soil movement (SWCA 2009). Figure 2.34 shows the micro-soil topography profile from one of the three sampling points at the Kelly piñon/juniper site for 2008–2015. The graph clearly shows the yearly variability associated with soil movement on a plot and a slight trend for overall soil loss over the 7-year period. Figure 2.35 through Figure 2.38 show average soil profile values averaged over all points per bridge, and over three bridges per paired plot, for 2008, 2009, 2010, 2011, 2012, 2013, 2014, and 2015. These figures show little overall change in average soil surface levels over that 6-year period and between the control and treatment plots. The processes of soil erosion and soil deposition can clearly be seen when plotting data from all 8 years. Over a series of years, this study will document losses and/or gains to the soil surface profiles at each bridge site and will provide average values for each of the eight plots in this study.

Through 5 years of post-treatment monitoring, no differences have been observed between the control and treatment. Overall, it does not appear that the treatments have caused damage to the soil resources. Whether these results persist into the future is still up for debate, with only future monitoring providing a conclusive answer.

Figure 2.33. Measurement of soil surface topography using a soil movement bridge helps understand the yearly variability associated with soil topography.

Figure 2.34. Soil surface profile from the East soil movement bridge located at the Kelly piñon/juniper control plot over 2008–2015, showing variation in the soil surface profile over a 8-year period. Each point 1–21 on the x axis represents one measurement point from the soil surface to the level bridge above the surface. Point 11 is the set point (head of a spike) for calibration.

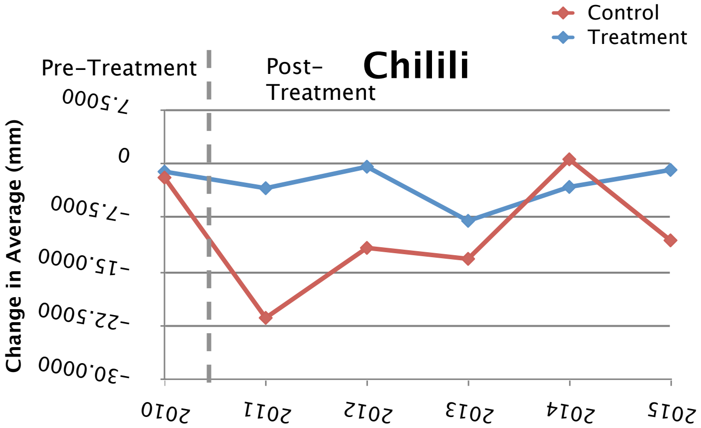


Figure 2.35. **Average soil surface profiles for the Chilili sites, averaged from three soil movement bridges located on each of the paired study plots over the 5-year period, 2010–2015.**

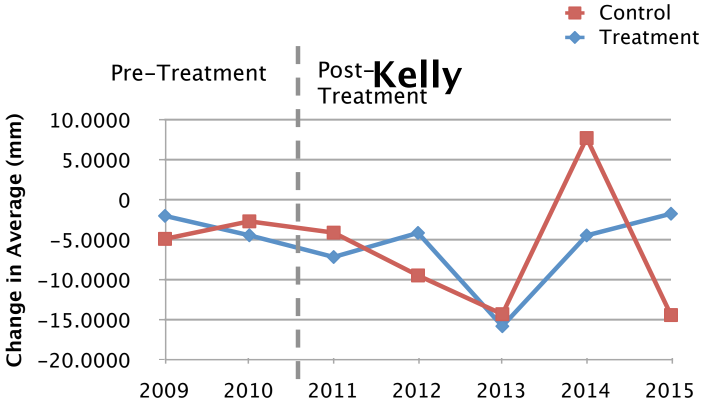


Figure 2.36. **Average change to soil surface profiles for the Kelly sites, averaged from three soil movement bridges located on each of the paired study plots over the 6-year period, 2009–2015.**

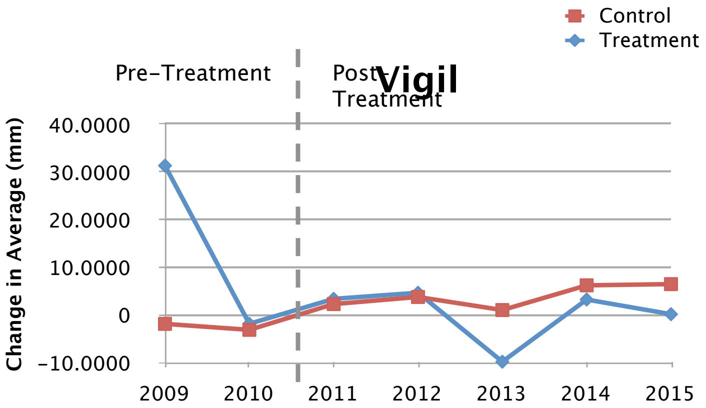


Figure 2.37. **Average change to soil surface profiles for the Vigil sites, averaged from three soil movement bridges located on each of the paired study plots over the 6-year period, 2009–2015.**

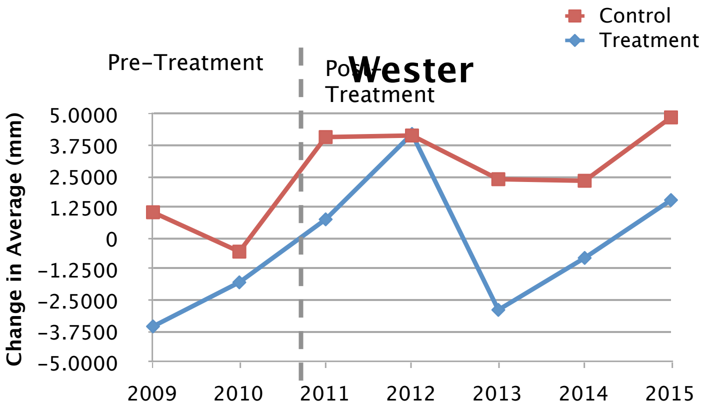


Figure 2.38. **Average change to soil surface profiles for the Wester sites, averaged from three soil movement bridges located on each of the paired study plots over the 6-year period, 2009–2015.**

2.05.4 SOIL CHEMISTRY

The chemistry of the soil is an important parameter in the overall health and functioning of a watershed. In particular, the top layer of soil, the A-horizon, is important because it is the zone where most biological activity occurs and is therefore the most fertile layer. The A-horizon is also the layer of soil most susceptible to disturbance because it is exposed at the surface to the elements of nature and humans. Soil chemistry plays a key role in sustaining the productivity of plants and soil biota, which directly affect the ability of soil to infiltrate water. Understanding the chemical makeup of a soil before treatment or disturbance can shed light on how restoration techniques affect the chemical composition of the soil.

Baseline measurements of soil chemistry were obtained in 2008, 2009, and 2010 before thinning treatments at the Kelly, Vigil, and Wester sites; Chilili was not included until the 2009 sampling because this plot had yet to be established. The purpose of taking these measurements was to quantify changes to soil chemistry potentially caused by thinning activities. The methods used in 2008, however, were slightly different than those used in 2009 and 2010 and can be a reason for any large differences seen between years. The soil samples were obtained using a 4-cm-diameter (1.6-inch-diameter), 20-cm-deep (8-inch-deep) impact soil corer at the four corners of the three established vegetation plots (Figure 2.39). In 2008 the 12 subsamples were placed in labeled separate bags in order to attempt in-house analysis with Cardy soil kits. The variability associated with these kits, however, proved to be too great for reliable results, so the subsamples were combined into one bag for each site and sent to the New Mexico State University Soils and Water Testing (SWAT) laboratory for further analysis. In 2009 through 2015, the 12 subsamples were combined into the same bag at the time of sampling. These pooled samples were considered to be representative of the study areas. The 2009, 2010, and 2011 samples were sent to the SWAT laboratory for analysis. The New Mexico State University SWAT laboratory closed in early 2012, so the 2012, 2013, 2014, and 2015 samples were sent to the Soil, Water, and Plant Testing Laboratory at Colorado State University (CSU). These methods followed the USFS Forest Inventory and Analysis Guide procedures (USFS 2005).

Figure 2.39. Soil cores were taken using an impact corer, shown above, for chemical analysis.

The variables measured by the SWAT and CSU laboratories included saturated paste pH, electronic conductivity, total soluble salts (sodium, calcium, and magnesium), sodium adsorption ratio, organic matter, nitrogen (nitrate) (NO_3), bicarbonate phosphorous, potassium, and a texture estimate. The results of the soil organic matter content and the macro nutrient nitrogen from samples taken from 2008–2015 are presented in Figure 2.40 through Figure 2.47.

The various soil chemistry compounds varied quite a bit at a given plot, between paired plots, between sites, and between years. This amount of background variation will be important to consider in determining if thinning treatments affect soil chemistry. Such treatment differences will need to be above this background variation.

Figure 2.40. Organic matter concentrations measured at the Chilili sites, 2009–2015.

Figure 2.41. Organic matter concentrations measured at the Kelly sites, 2008–2015.

Figure 2.42. Organic matter concentrations measured at the Vigil sites, 2008–2015.

Figure 2.43. Organic matter concentrations measured at the Wester sites, 2008–2015.

Figure 2.44. Nitrate concentrations measured at the Chilili sites, 2009–2015.

Figure 2.45. Nitrate concentrations measured at the Kelly sites, 2008–2015.

Figure 2.46. Nitrate concentrations measured at the Vigil sites, 2008–2015.

Figure 2.47. Nitrate concentrations measured at the Wester sites, 2008–2015.

2.06 FOREST THINNING HYDROLOGIC MONITORING

Monitoring flumes (Parshall flumes) complete with pressure transducers were installed at all four study sites to study the impacts of tree thinning to surface flow (Figure 2.48). For more detailed information on the methodology, site location, and relevant background information, please refer to the 2008 Monitoring Plan (SWCA 2008).

Figure 2.48. Parshall flume located at the thinned Chilili site.

During the 2015 monitoring period, rainfall occurred in the project area on 15% of the days monitored compared with 14% of the days in 2014, 12% in 2013 and 17% of the days monitored in 2012. However, like most years, a majority of these rainfall events (~75%) was relatively small and totaled less than 2.5 mm (0.1 inch). During the 2015 monitoring period, 28 flow events were recorded across all watersheds. This was an increase from 2013, but still flows generally did not occur without at least 7.6 mm (0.3 inch) of rainfall, which has been the case since the beginning of the project. The plots located in the ponderosa pine sites generated runoff with slightly less rain (7.6 mm [0.3 inch]), whereas the piñon/juniper sites required about 12.7 mm (0.5 inch) of rain to generate runoff events.

During the 2015 monitoring period, there were no basin-wide storm events that generated flow across all study sites simultaneously. The Kelly control flume did not record any flow events during the 2015 monitoring season, whereas the Wester control flume recorded 10 events. The Chilili flumes were not analyzed because of a lack of weather station data to corroborate precipitation-triggered runoff events. There were a total of 28 recorded flow events, but there were no clear trends in 2015 between control and treatment; some plots had more flow recorded at the control, whereas others had more at the treatment. This may be due to the intensity of the precipitation, which can be significantly different from one watershed to the other during the monsoon season. The rain gages at the site only measure hourly precipitation; therefore, being able to tease this out would be difficult. Another factor that could be contributing to the reduction of flows on the treated watersheds could be a result of the recovery of the herbaceous cover which has the ability to limit the amount of overland flow and increase the soil infiltration.

Six of the flumes recording events in 2015 had paired events measured, which means both the control and treatment watersheds had a flow event. Below in Figure 2.49 and Figure 2.50 paired flow events from the Vigil and Wester watersheds are analyzed. The summary results of these flows can be found in Table 2.3 through Table 2.5.

All Parshall flumes were functioning properly during the 2015 season.

Figure 2.49. Hydrograph showing the Vigil treatment and control treatment during a storm flow event on July 7, 2015.

Table 2.3. Summary of Runoff Event for the Vigil treatment and control, July 7, 2015.

Runoff Parameters	Study Sites	
	Vigil Treated	Vigil Control
Flow start	2:35	3:00
Flow stop	4:00	4:40
Peak stage (feet)	0.35	0.43
Peak flow (cubic feet/second)	0.198	0.27
Flow duration (minutes)	85	100
Total volume of flow (cubic feet)	400	158
Watershed area (acres)	0.68	0.1
Volume of flow per acre (cubic feet/acre)	588	1586
Total rainfall (inches)	1.05	2.27
Total volumetric rainfall (cubic feet)	2591	824
Rainfall/Runoff ratio	0.22	1.92

Figure 2.50. Hydrograph showing the storm flow at the Wester treatment and control site that occurred on August 24, 2015.

Table 2.4. Summary of Runoff Event for the Wester Sites, August 24, 2015.

Runoff Parameters	Study Sites	
	Wester Treated	Wester Control

Flow start	4:38	4:33
Flow stop	5:03	5:08
Peak stage (feet)	0.06	0.34
Peak flow (cubic feet/second)	0.014	0.19
Flow duration (minutes)	25	35
Total volume of flow (cubic feet)	10.2	164
Watershed area (acres)	1.03	6.76
Volume of flow per acre (cubic feet/acre)	9.9	24.2
Total rainfall (inches)	1.07	0.79
Total volumetric rainfall (cubic feet)	4000	19385
Rainfall/Runoff ratio	0.0025	0.0012

With respect to site hydrology, there are four conditions that could change because of forest thinning or from the effects of wildfire: 1) increased frequency of flow, 2) greater duration and volume of flow, 3) increased peak flow, and 4) a greater ratio of runoff to rainfall.

2.06.1 FLOW FREQUENCY, DURATION, AND VOLUME

Frequency of flow will be analyzed over time as data are collected; however, based on the period of record so far, a baseline has been established for the remaining parameters. The parameters of flow duration and volume will likely be the least useful in assessing effects from forest thinning, as these parameters are highly dependent on rainfall duration and intensity. In general, the ponderosa pine sites generated flows of longer duration and greater volume than did the piñon/juniper sites, which can likely be attributed the elevation differences (see Table 2.1). A summary of the number of flow events (frequency), flow duration, and flow volume for the observed runoff events is shown in Table 2.5.

Table 2.5. Summary of Flow Frequency, Duration, and Volume, 2008–2015

Location	Number of Flow Events	Range of Duration (minutes)	Mean Duration (minutes)	Range of Volume (cubic feet)	Mean Volume (cubic feet)
Chilili treatment	12	55-175	131	3-225	89
Chilili control	5	75-160	117	35-124	80
Kelly control	10	15-150	89	14-1840	303
Kelly treatment	4	125-200	155	22-503	293
Vigil treatment	29	10-230	106	2-400	55
Vigil control	17	30-215	132	3-343	56
Wester treatment	8	10–205	86	1-334	78
Wester control	19	25-330	91	3–1832	278
Control Sites	**51**	**10-330**	**107**	**3-1840**	**179**
Treatment Sites	**53**	**10-230**	**119**	**1-503**	**128**

2.06.2 PEAK FLOW/STAGE

Peak flow can be affected by the intensity of rainfall, but it is also a measure of the flashiness of flow; particularly in post-fire monitoring, runoff can occur rapidly with large peaks appearing very quickly. The highest peak stage in 2015 was recorded at the Kelly Control plot (0.134 m [0.439 feet]), compared to the greatest recorded peak flow of 1.29 feet recorded at the Wester control plot on July 2, 2010 (coinciding with the greatest observed daily rainfall). A summary of peak stage runoff events for all years is shown in Table 2.6.

Table 2.6. Peak Stage of Runoff Events, 2008–2014

Location	Number of Flow Events	Range of Peak Stage (feet)	Median Peak Stage (feet)
Chilili treatment	12	0.19–0.76	0.475
Chilili control	5	0.11–0.57	0.375
Kelly control	10	0.14–1.038	0.175
Kelly treatment	4	0.02–0.69	0.23
Vigil treatment	29	0.06–0.46	0.19
Vigil control	17	0.22–0.4	0.27
Wester treatment	8	0.15–0.85	0.19
Wester control	19	0.12–1.29	0.38
Control Sites	**51**	**0.11-1.29**	**0.30**
Treatment Sites	**53**	**0.02-0.85**	**0.27**

2.06.3 RAINFALL/RUNOFF RATIO

The rainfall/runoff ratio is perhaps the most useful parameter to observe. All other parameters can vary due solely to the magnitude or intensity of rainfall; the rainfall/runoff ratio normalizes the flow events, although intensity and antecedent soil moisture conditions will still affect the amount of runoff. The rainfall/runoff ratio looks at the percentage of rainfall falling on the watershed and leaving as surface runoff. A value of zero indicates no water left the watershed, and a value of 1 indicates all water falling on the watershed was observed leaving as surface runoff (this is highly unlikely). In natural settings, the rainfall/runoff ratio typically falls in the 0.1 to 0.3 range. The rainfall/runoff ratios observed during flow events from the watersheds are summarized in Table 2.7. In general, rainfall/runoff ratios were highly variable, including some extremely high values; however, almost 70% of the flow events had rainfall/runoff ratios of less than 0.10. Ponderosa pine sites exhibited a slightly lower rainfall/runoff ratio than piñon/juniper sites, which can likely be attributed to the large amounts of litter and duff that serve as a sponge and retain the water.

Table 2.7. Rainfall/Runoff Ratio for Observed Flow Events, 2008–2015

Location	Number of Flow Events	Range of Rainfall/Runoff Ratio	Mean Rainfall/Runoff Ratio
Chilili treatment	12	0.0001–0.003	0.002
Chilili control	5	0.005–0.008	0.007
Kelly control	10	0.01–0.88	0.18

Kelly treatment	4	0.12-1.07	0.61
Vigil treatment	29	0.001–0.227	0.044
Vigil control	17	0.14–6.09	1.42
Wester treatment	8	0.0004–0.01	0.004
Wester control	19	0.0001–1.05	0.10
Control sites	**51**	**0.0001-6.09**	**0.42**
Treatment sites	**53**	**0.0001-1.07**	**0.16**

2.07 TREES AND OTHER VEGETATION

For details regarding the research questions, monitoring protocols, and plot design for vegetation monitoring, as well as a full literature review, please refer to the 2008 Monitoring Plan (SWCA 2008).

2.07.1 TREES AND WILDFIRE FUELS

Tree monitoring measurements in the fall of 2015 included observations of canopy dieback, disease or damage, live and dead status.

Basal Area Measurements

Basal area measurements were taken in fall 2015 (Table 2.8).

Table 2.8. Treatment Designation for All Plots (with basal area totals) and 2015

Site	Total Basal Area (square feet/acre) 2008	Total Basal Area (square feet/acre) 2015
Chilili treatment	101	65
Chilili control	103	113
Kelly control	89	103
Kelly treatment	86	48
Vigil treatment	68	41
Vigil control	38	47
Wester treatment	133	108
Wester control	126	101

Stand Structure

Diameter measurements of trees were taken in 2015. These are used below to demonstrate the stand structure and various size classes at each site. Figure 2.51 and Figure 2.52 show the size class structure of ponderosa pine trees (diameter at breast height [DBH]) at the ponderosa pine sites, Chilili and Wester. Figure 2.53 and Figure 2.54 show the size class structure of piñon and juniper trees (diameter at root crown [DRC]) at the piñon/juniper sites, Kelly and Vigil.

Figure 2.51. **Size classes of ponderosa pine trees measured at DBH on the Chilili control and treatment plots.**

Figure 2.52. **Size classes of ponderosa pine trees measured at DBH on the Wester control and treatment plots.**

Figure 2.53. **Size classes of piñon/juniper trees measured at DRC on the Kelly control and treatment plots.**

Figure 2.54. **Size classes of piñon/juniper trees measured at DRC on the Vigil control and treatment plots.**

Figure 2.51 through Figure 2.54 illustrate the difference in size class distribution between control and treatment plots for each site. At both ponderosa pine sites (Chilili and Wester), the control plots have a greater number of trees distributed in the lower size classes (1–7 inches, and to a lesser extent 8–11 inches), compared to the treatment plots where trees are more uniformly distributed across size classes and the number of smaller-diameter trees (1–4 inches) is reduced. For the piñon/juniper control plots on both sites (Kelly and Vigil), the greatest numbers of trees fall in size classes 5–8 and 9–12 inches DRC, respectively, and there are relatively fewer large-diameter trees. Both piñon/juniper treatment plots had no trees less than 4 inches DRC, and the remaining trees were more evenly distributed across size classes than the control plots.

Crown Dieback

Percent crown dieback is the percentage of the leafy canopy of each tree that showed signs of physiological stress (i.e., brown needles and leaves). Crown dieback could result from a number of environmental factors, e.g., drought, insect attack, competition, and disease. Measurement of crown dieback is highly dependent on the time of year; as a result, efforts are made to take measurements consistently during late September to early October each year. Figure 2.55 illustrates crown dieback across all sites.

Crown dieback levels from 2008 to 2015 are presented below by site and year (see Figure 2.55). This graph clearly shows the inherent variability associated with measuring crown dieback. Crown dieback of individual trees can be highly variable across a plot based on tree size and position and the environmental factors it is exposed to. Dieback levels for 2015 were low compared to previous years averaging 3% to 7% across all plots. We believe that dieback levels are within the normal range of variability for all 8 years.

Figure 2.55. **Average percent crown dieback of tree canopies for each thinning plot, 2008–2015.**

Tree Mortality

In total, 613 trees were tagged across all watersheds in this study, with species composition from ponderosa pine, piñon pine, oneseed juniper, and alligator juniper (*Juniperus deppeana*). In 2008 there were no dead trees tagged on any plots. Natural tree mortality has been low across all plots in all 8 years (Figure 2.56).

The 2015 season saw limited mortality (not attributed to tree cutting) on only the Kelly control and Wester control plots. The Kelly control plot saw 1% mortality on top of the 8% mortality experienced in 2014, all of which was located in the same cluster and likely a result of the preceeding years' drought and bark beetle infestation. As demonstrated in Figure 2.55, a number of plots experienced crown dieback in 2015 that may result in increased mortality in subsequent years. Tree mortality will continue to be monitored every fall.

Figure 2.56. Percent tree mortality recorded across all thinning plots from 2008–2015. Percent mortality is recorded in relation to tree status in 2008.

Wildfire Fuels

Fuel measurements have been taken using Brown's transect protocols (Brown 1974) during the fall monitoring season from 2009 to 2015. Measurements are taken within the four circular tree plots on each paired watershed. Refer to the 2008 Monitoring Plan for detailed monitoring protocols and an explanation of fuel class sizes (SWCA 2008). Figure 2.57 and Figure 2.58 illustrate the percent cover by the various fuel classes on each thinning plot measured in 2012 and 2015, respectively.

Figure 2.57. Percentage of fuel in each fuel particle size class for 2012 (1-hour, 10-hour, 100-hour, 1,000-hour) on all thinning plots.

Figure 2.58. Percentage of fuel in each fuel particle size class for 2015 (1-hour, 10-hour, 100-hour, 1,000-hour) on all thinning plots.

With reference to Figure 2.57 and Figure 2.58, the piñon/juniper plots tended to have a higher accumulation of 1-hour fuels (fine fuels 0.0–0.6 cm [0.00–0.25 inch] in diameter) compared to the ponderosa plots, with the exception of the Chilili treatment plot. Conversely, 100-hour and 1,000-hour fuels (woody debris > 2.5 cm [1 inch] in diameter and > 8 cm [3 inches] in diameter, respectively) were more common at the ponderosa sites. 100-hour fuels have increased at all of the PJ sites over the last 3 years. Each paired plot was relatively consistent in terms of fuel loading by size class, except in the case of Chilili where the treatment plot had significantly higher 1-hour fuels; 1-hour fuels at the treatment plot have decreased on 2012 levels.

Figure 2.59 shows that both Chilili plots had considerably more duff and litter than the other plots. The volume of litter and duff found on the forest floor is related to both productivity and decomposition. There is very little difference in duff and litter depths between treatment and control sites because residual wood chips left over from treatment were spread thinly following the required prescription, so as not to significantly alter the fuel loading at each site.

Figure 2.59. Average combined duff and litter depths on all thinning plots, measured in inches for 2015.

The variation in litter and duff between the Wester and Chilili sites could be related to differing decomposition rates as a result of differences in elevation and moisture regimes. Decomposition has been found to be positively correlated with moisture gradient with greater decomposition on more productive sites (Keane 2008); this would explain the greater depths of duff at Chilili (a higher elevation and more productive ponderosa pine forest) versus Wester (a lower elevation, drier and more open stand ponderosa pine forest). Overall duff and litter depths were higher on the ponderosa sites than the piñon/juniper sites, which is to be expected since litter and duff cover in ponderosa pine is almost continuous across the landscape, while litter and duff is isolated in patches immediately below the canopies of trees in piñon/juniper woodlands.

Figure 2.60 shows the tons/acre of woody dead and downed fuels at each site. The piñon/juniper sites had relatively low fuel loading compared to the ponderosa sites, because the piñon/juniper sites tended to have fewer large-diameter woody fuels. The piñon/juniper sites exhibited greater fine fuel loading, however, likely due to lower canopy cover that permits the growth of graminoids and forbs (Figure 2.61 and Figure 2.62). Shrub cover was limited at both piñon/juniper sites. The Wester plots also had low loading compared to the Chilili plots (see Figure 2.61); this site was relatively open, with fewer 1,000-hour fuels consequently lowering the tons/acre totals. Chilili treatment and control plots have noticeably higher fuel loadings than all other sites; these are dense plots with many more 1,000-hour fuels (many downed trees and stumps) (see Figure 2.62), which raised their total tons/acre. The fuel loading at the Chilili control site increased considerably since 2012, while the treatment site had lower fuel loading than 2012 levels; this can be attributed to more 1,000-hour fuels on the control plot. The high levels for the Chilili control in 2015 are likely due to heavy concentrations of large-diameter fuels that have accumulated at that site.

Figure 2.60. Fuel loading (in tons/acre) of dead and downed woody debris for all thinning plots, 2009–2015.

Figure 2.61. **Wester control, showing the low fuel loading on the plot and lack of large-diameter dead and downed fuels.**

Figure 2.62. **Chilili control, showing high fuel loading with evidence of large-diameter dead and downed fuels.**

2.08 TREE CANOPY VISUAL STRUCTURE MONITORING

Tree canopy horizontal (crown cover) and vertical structure (i.e., lower tree branches) were measured on the wildlife plots every fall from 2010 through 2012. However, since tree structure changes very slowly in contrast with herbaceous vegetation, tree canopy and vertical structure on the wildlife plots are now being measured every 3 years since 2012, including 2015.

Tree canopy horizontal structure on the wildlife plots was measured by using a standard spherical densiometer for measuring tree upper canopy closure. A vertical structure method that is presented below was used to measure lower tree vertical canopy closure. Tree canopy structure was measured in the fall of 2010 through 2013, and again in 2015 when other vegetation measurements were made. Vegetation vertical canopy structure was measured on each of the four vegetation and soils subplots, and on all of the wildlife monitoring plots. The method was adapted from Herrick et al. (2005) and consisted of a 2-m-long (6.6-foot-long), 5-cm-diameter (2-inch-diameter) white polyvinyl chloride (PVC) pipe pole partitioned into three different 2-m (6.6-foot) height layers, each with continuous 10-cm (4-inch) black/white increment markings. The 2-m (6.6-foot) PVC measurement pipe was partitioned into four different vertical 0.5-m (1.6-foot) segments or heights above the ground surface: segment one = 2.0–1.5 m (6.6–4.9 feet), segment two = 1.5–1.0 m (4.9–3.3 feet), segment three = 1.0–0.5 m (3.3–1.6 feet), and segment four = 0.5–0.0 m (1.6–0.0 feet) above the ground surface. An observer recorded vegetation canopy obstruction of the black and white marked areas on the pole, while another person held the pole vertical at three locations across the center line of each 30-m (98-foot) vegetation and soils monitoring subplots, one reading at 10 m (33 feet), one at 20 m (66 feet), and one at 30 m (98 feet). On the vegetation/soils plots, the observer was located 10 m (33 feet) toward the center of the plot from the pole for each canopy measurement. An overall visual obstruction average score was then calculated for each segment of the pole over each of the three lines per subplot, and an overall average score for each segment was then calculated for each plot.

On the wildlife monitoring plots, both vertical structure and densiometer measurements were taken at 11 locations on each wildlife plot at 12 existing vegetation quadrat points, along the middle lines of six quadrats running north-south, and east-west through the middle of each plot, at 10-m (33-foot) intervals. Vertical vegetation structure profiles are not only important for assessing wildlife habitat, but also for fire fuels structure.

Changes in tree horizontal canopy cover as measured by a spherical densitometer showed a reduction in tree upper canopy cover on all of the treatment plots in 2011 and 2012 compared to the control plots following tree thinning in late 2010 (Figure 2.57). However, apparently due to large variation values from measurement points, those differences were not statistically different except for the Wester ponderosa pine site where tree canopy cover was significantly less on the plot that was thinned. In 2015, tree canopy cover increased slightly but not significantly on 5 of the 8 plots, and declined slightly but not significantly on 3 of the 8 plots (Figure 2.57). In 2015, control plots had significantly more tree canopy foliage than treated plots at the Chilili ($P = 0.01$) and Wester ($P < 0.0001$) ponderosa pine sites, but not at either of the piñon/juniper sites.

Changes in tree lower vertical canopy structure from ground level to a height of 2 m (6.6 feet) also showed a reduction in lower tree canopy density on the plots that were thinned when comparing the treatment to control plots after thinning in 2011, and in 2012 (Figure 2.58). In 2010 prior to tree thinning, paired plots and the two ponderosa pine sites, Chilili and Wester, had significantly different lower canopy structure (P = 0.006, P = 0.02, respectively), but treatment and control plots at the Kelly and Vigil piñon/juniper sites were not different. In 2011 following tree thinning, lower tree canopy densities were significantly different at the Chilili and Wester ponderosa pine sites, even more than in 2010 (P < 0.0001, P = 0.001, respectively), and significantly different at the Vigil piñon/juniper site (P < 0.0001), but not at the Kelly piñon/juniper site. These differences were still present in 2012. In 2015 vertical canopy structure was significantly greater at the Chillili (P < 0.0001), Kelly (P = 0.026) and Vigil (P < 0.0001) sites, but not at the Wester site (Figure 2.58). These findings indicate that forest thinning had a greater effect on lower canopy structure of trees than the upper canopy and that forest thinning did open the tree canopy on thinned plots compared to adjacent non-thinned control plots, and thinned plots still have more open horizontal and vertical canopy structure five years following thinning treatments.

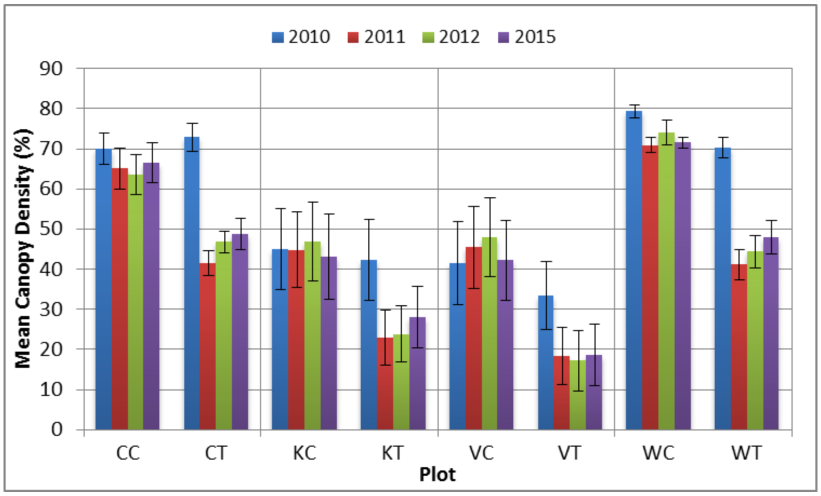


Figure 2.57. **Upper tree canopy cover scores as measured from a spherical densiometer on each of the monitoring plots. Densiometer scores are means (averages) from 11 locations per plot and range from 0 to 96, similar to percent cover.**

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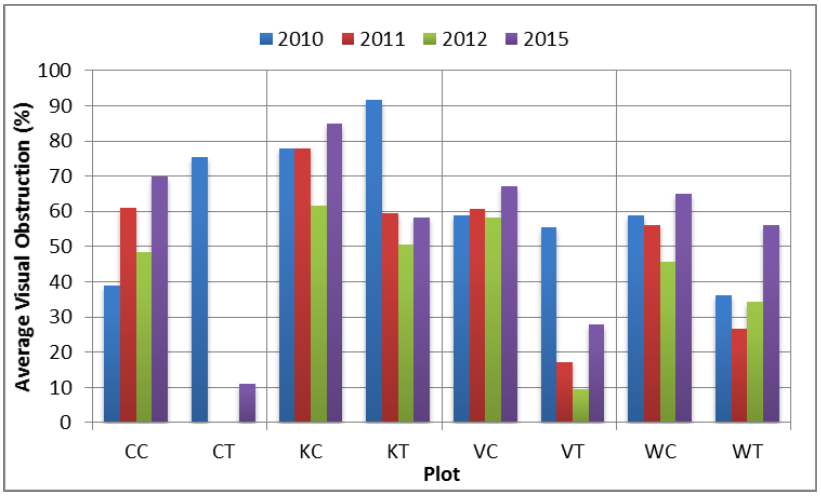


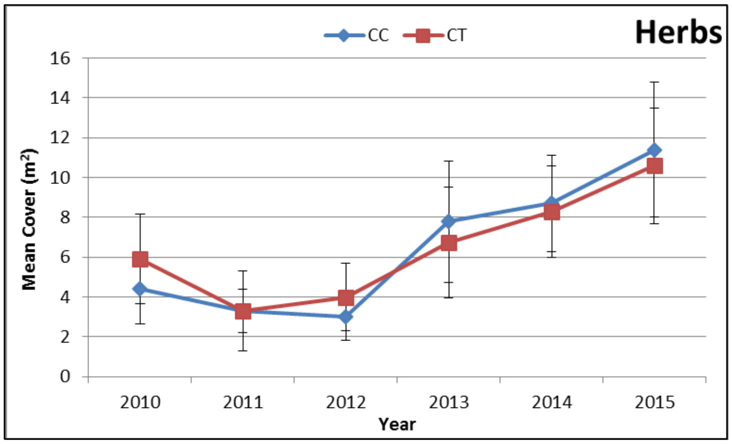
Figure 2.58. **Vegetation vertical structure cover from ground level to a height of 2 m (6.6 feet), measured with a vertical structure pole. The higher the score, the denser the vertical canopy cover (note that Chilili treatment plot had a value of zero in 2011 and 2012).**

2.08.1 VEGETATION AND GROUND SURFACE COVER MONITORING

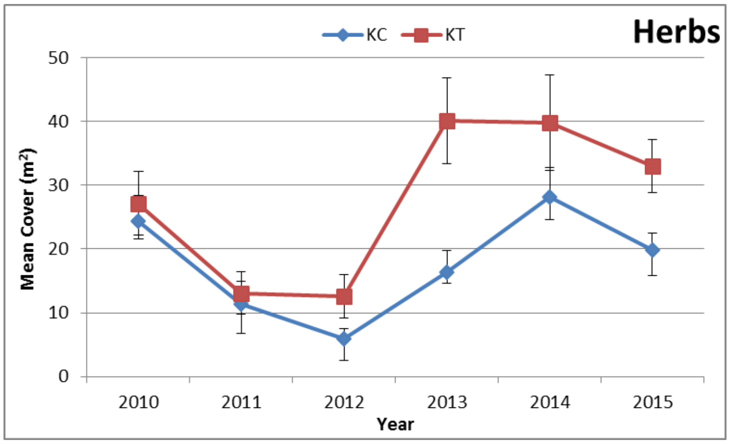
Herbaceous vegetation was again measured along line intercepts and quadrats from the vegetation and soils plots at each site as presented in the 2009 Annual Report (SWCA 2010). Additionally, in 2010, SWCA initiated more extensive vegetation measurements on the wildlife plots in order to characterize vegetation composition and structure as habitat for wildlife on those plots and to provide quantitative data to determine how vegetation or habitat changed on the wildlife plots relative to forest thinning treatments. Those vegetation measurements were taken again in 2011, 2012, 2013, 2014 and 2015 and are being used to characterize vegetation changes on study plots relative to forest thinning treatments. Vegetation was measured from 1-m^2 (10.8-square-foot) quadrats located at each of the 36 permanently marked rodent trapping stations on each wildlife plot in a six by six grid, with stations at 10-m (33-foot) intervals (50 × 50–m [164 × 164–foot] plot). All herbaceous plant species, cacti, and woody shrubs were measured on each of those 1-m^2 (10.8-square-foot) quadrats. The total canopy cover and maximum height in centimeters of each species was measured per quadrat. Vegetation quadrat data were also categorized by growth form (e.g., shrub, cacti, grass, forb, total herbaceous vegetation (grasses and forbs) (herbs), and life history (annual or perennial). In addition to vegetation, soil surface cover categories also were measured on the quadrats, including bare soil, leaf litter (and dead and downed woody material), rock, and cryptobiotic (cryptogam) soil surface crusts. Measures of wood chip coverage on the ground resulting from forest thinning practices were added in 2011 and have been continued each year.

The vegetation and ground cover data measured from the replicated quadrats on wildlife plots provide the most appropriate data for statistical testing for differences in those cover values resulting from thinning treatments, because there is sufficient sample replication (n = 36 on each paired plot) to perform parametric statistical tests. Also, those 36 sampling quadrats were evenly distributed over relative large areas (plots 50 m [164 feet] on a side), providing a good sampling representation of each of the paired study plots. Data from each vegetation and ground cover type were used to test for differences between paired plots using parametric paired t-tests. Ideally, there should have been no significant differences between paired plots prior to thinning treatments. If thinning has an effect on any of those cover types, then a significant difference would be expected following thinning treatments.

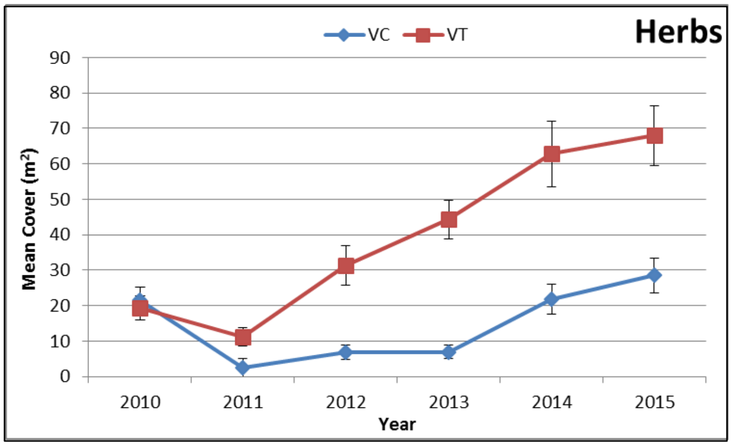
Results for herbaceous understory vegetation and soil surface cover types measured from the thirty-six 1-m^2 (10.8-square-foot) quadrats in the fall of 2015 are presented in Figure 2.59, a–bb, providing separate graphs for forbs, grasses, all herbs, and each soil surface cover type. Results from 2010 prior to tree thinning treatments and in 2011 through 2015 following thinning treatments also are presented in Figure 2.59, a–bb, to show annual change in those variables over time. Results of statistical paired t-tests of differences between mean cover values for each of the different vegetation and ground surface cover types measured in 2015 and shown in Figure 2.59 are presented in Table 2.9.



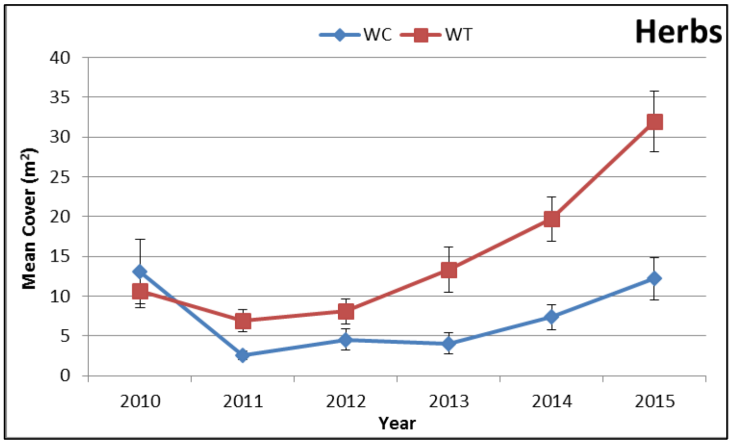
a. Herbs, Chilili.



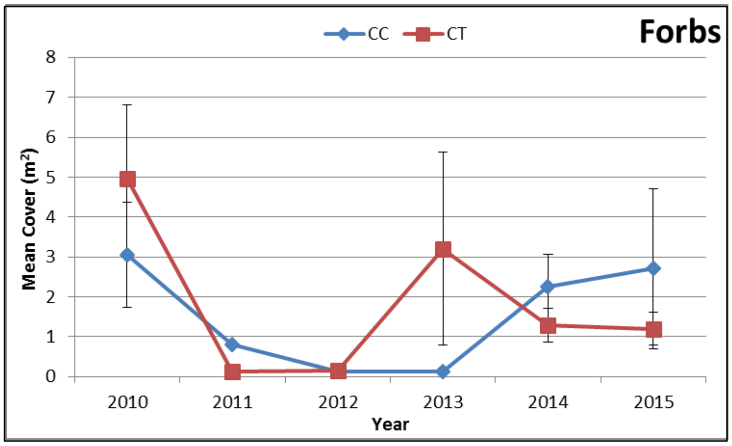
b. Herbs, Kelly.



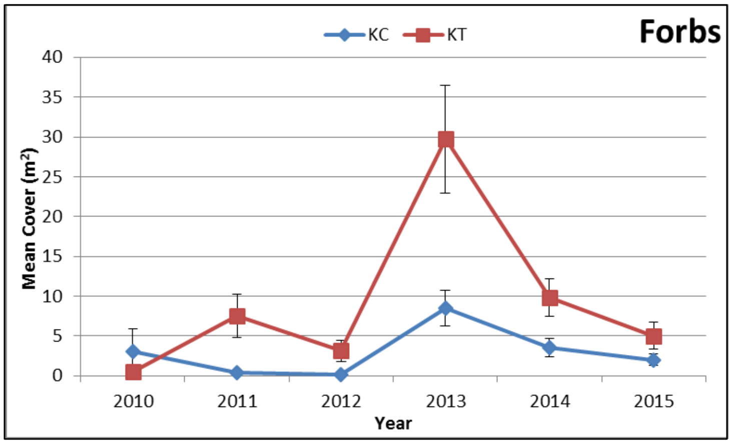
c. Herbs, Vigil.



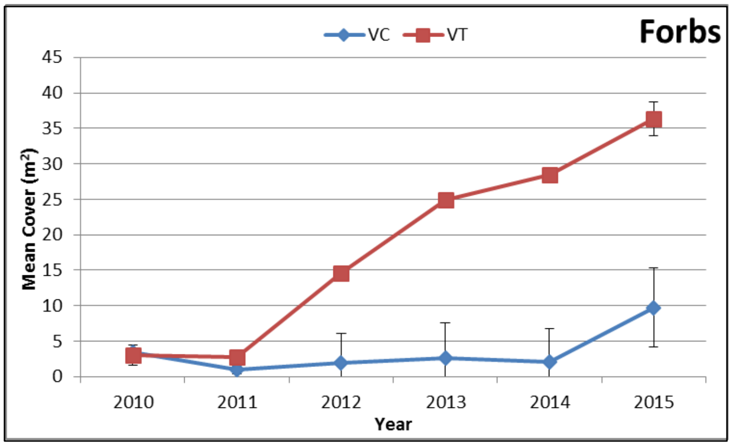
d. Herbs, Wester.



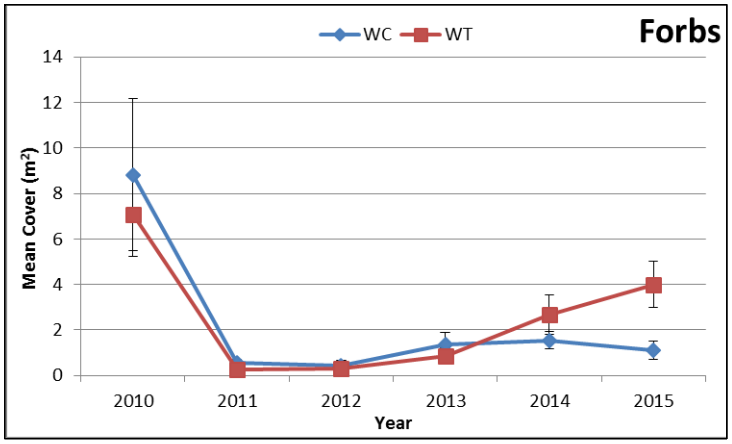
e. Forbs, Chilili.



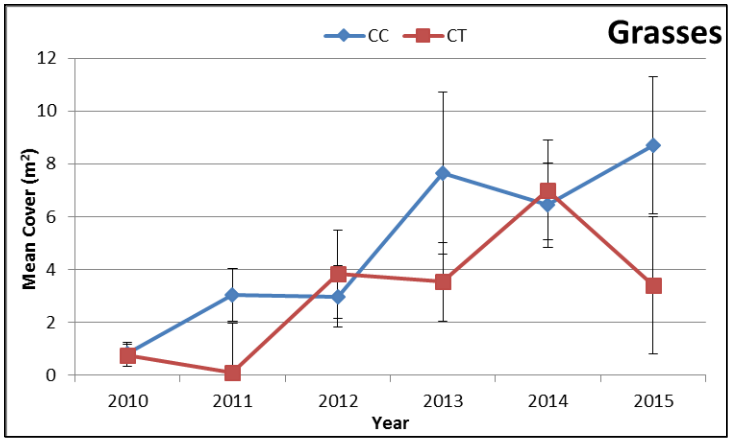
f. Forbs, Kelly.



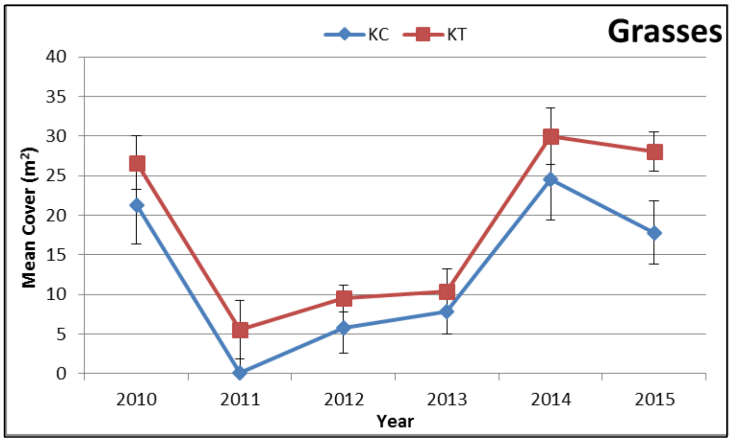
g. Forbs, Vigil.



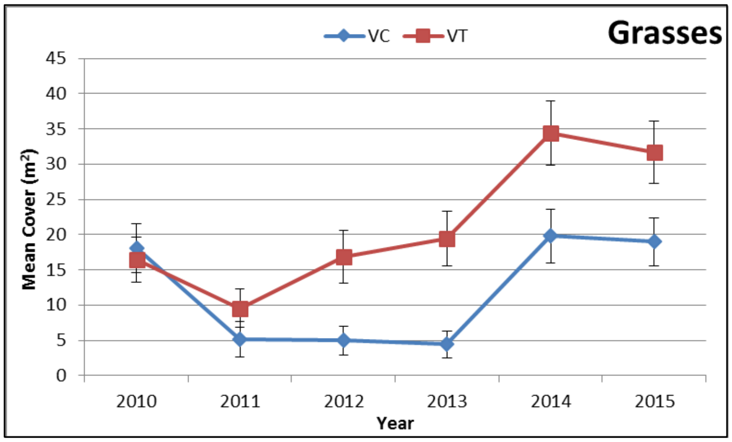
h. Forbs, Wester.



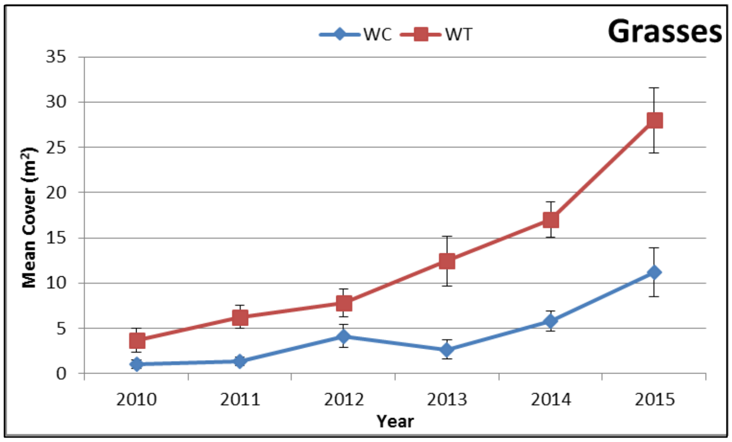
i. Grass, Chilili



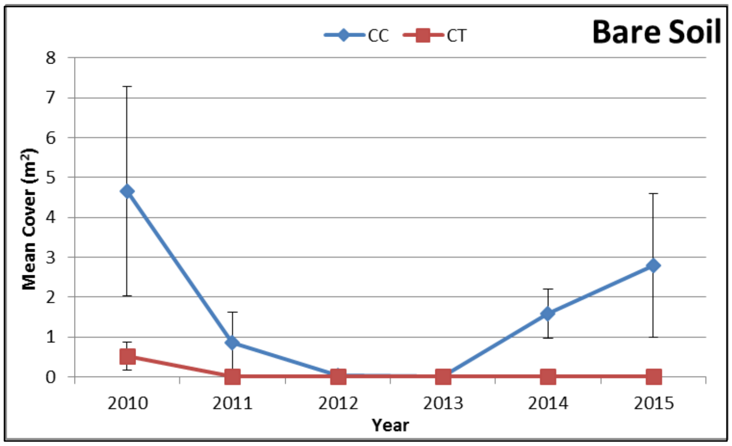
j. Grass, Kelly.



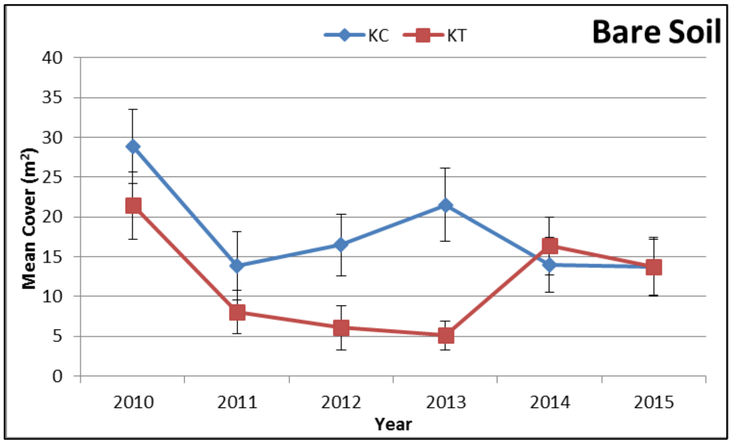
k. Grass, Vigil.



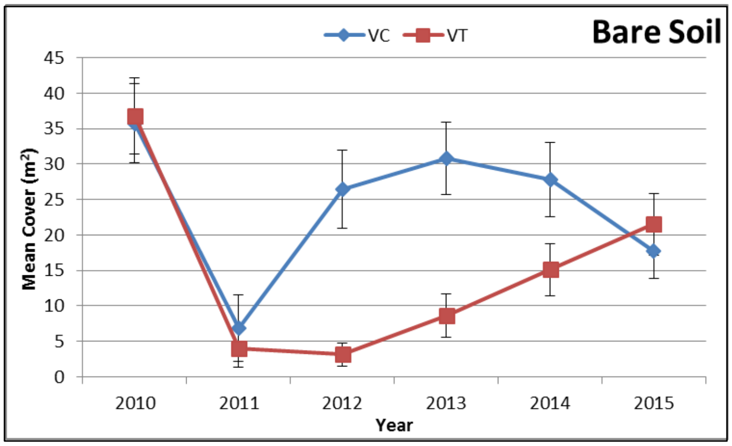
l. Grass, Wester.



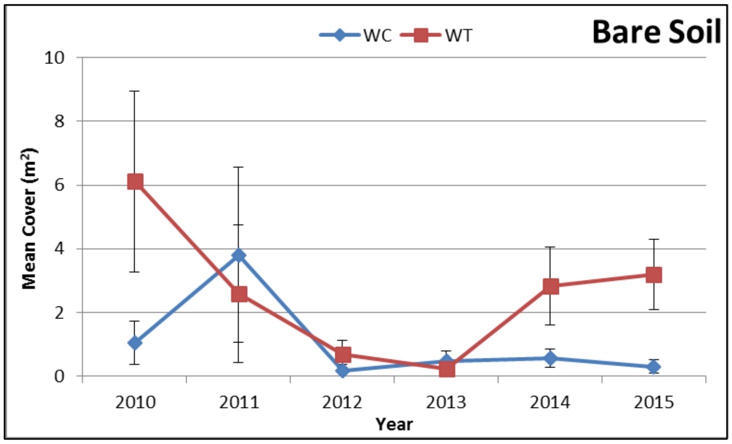
m. Bare soil, Chilili.



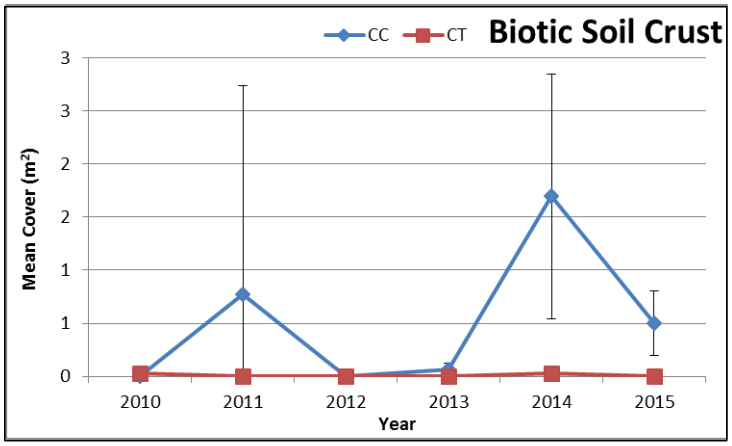
n. Bare soil, Kelly.



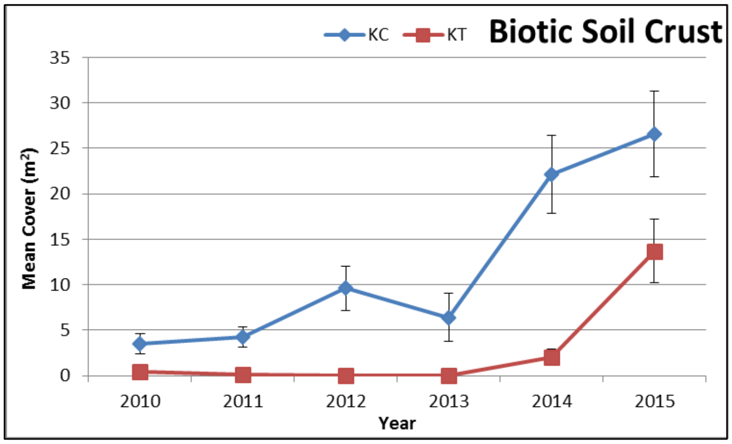
o. Bare soil, Vigil.



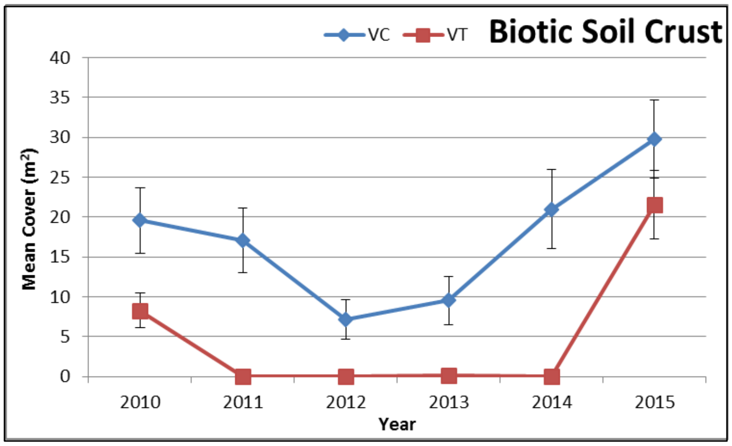
p. Bare soil, Wester.



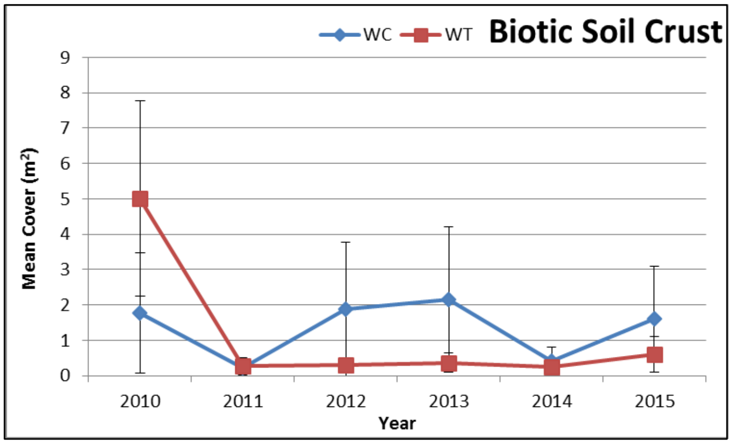
q. Biotic soil crust, Chilili.



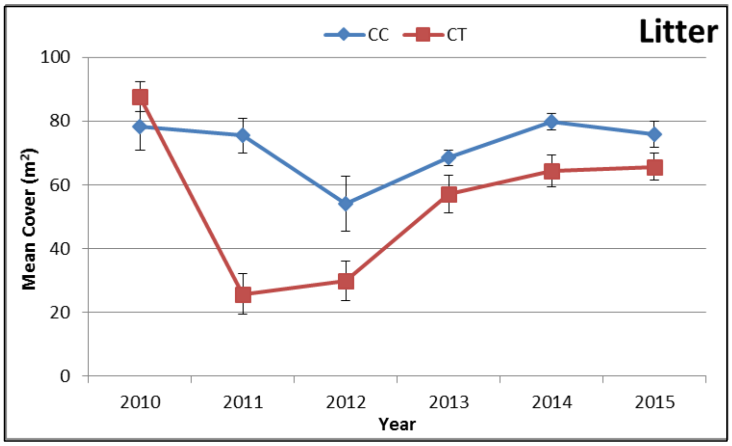
r. Biotic soil crust, Kelly.



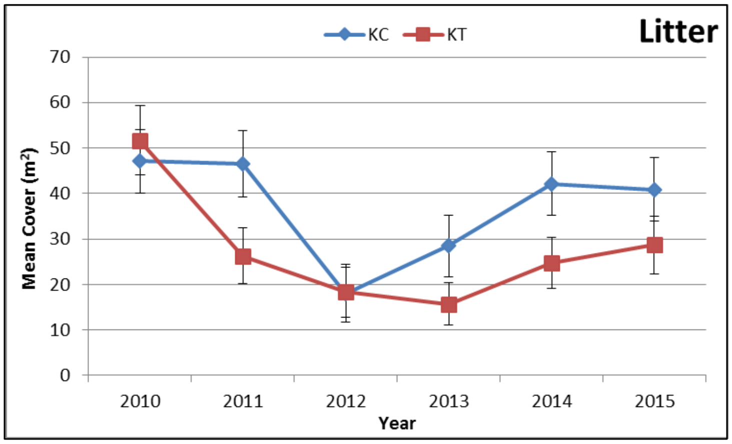
s. Biotic soil crust, Vigil.



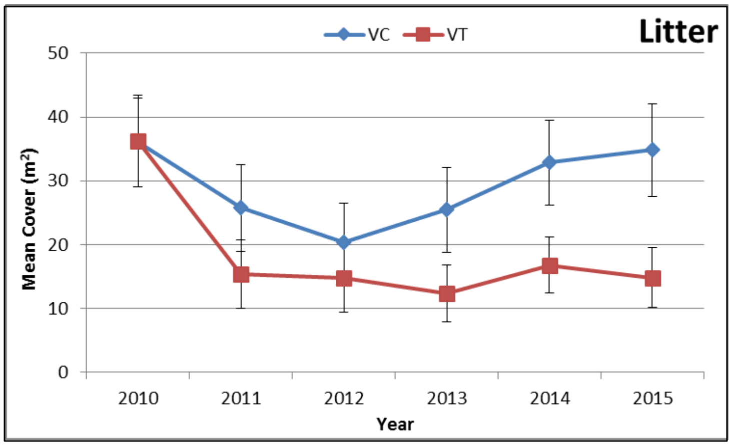
t. Biotic soil crust, Wester.



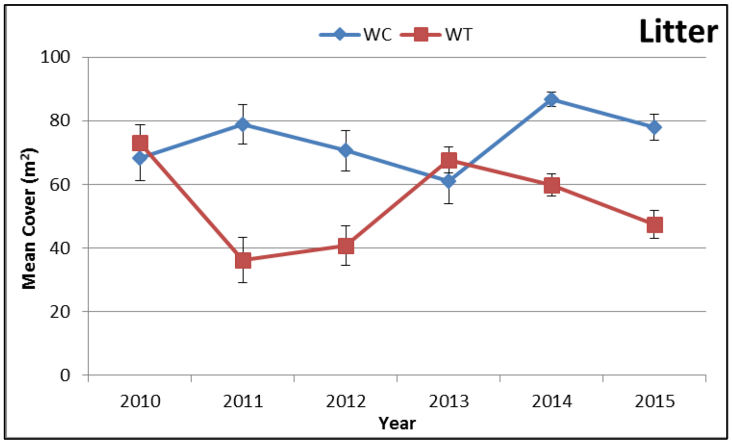
u. Leaf litter, Chilili.



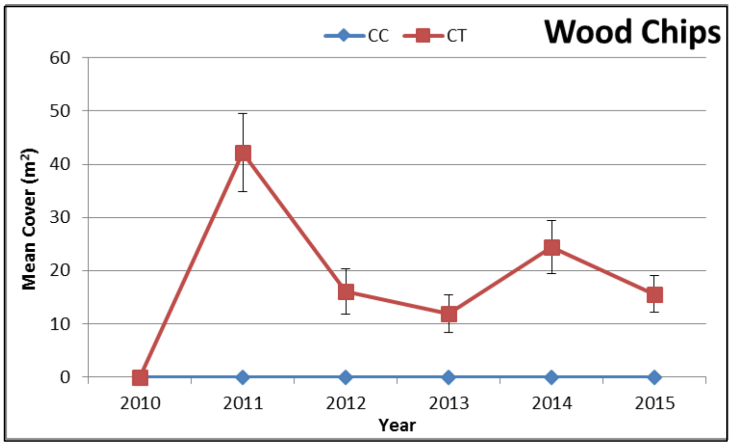
v. Leaf litter, Kelly.



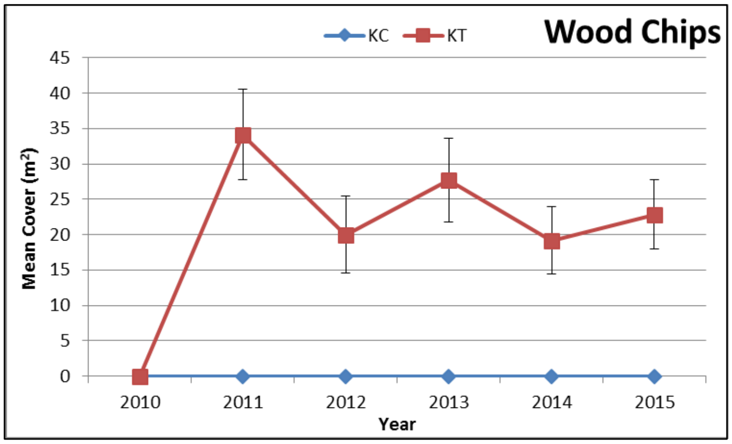
w. Leaf litter, Vigil.



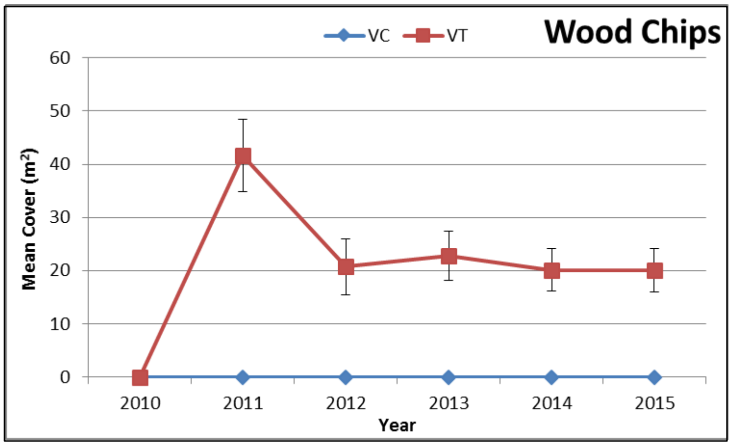
x. Leaf litter, Wester.



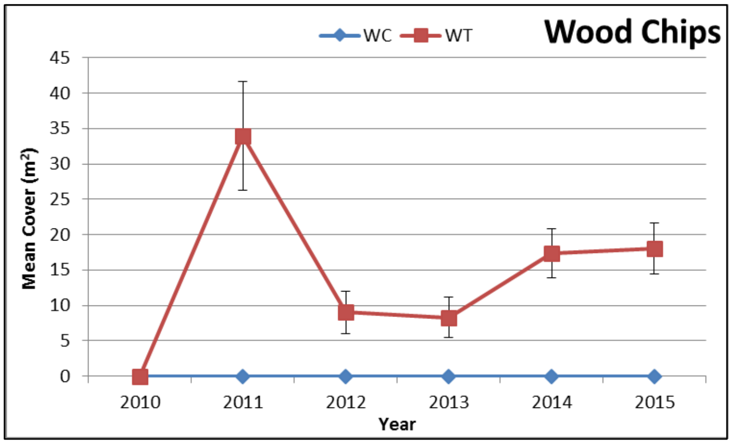
y. Wood chips, Chilili.



z. Wood chips, Kelly.



aa. Wood chips, Vigil.



bb. Wood chips, Wester.

Figure 2.59. These graphs illustrate the mean values of cover type found across all vegetation quadrats among all of the study sites and paired study plots from fall 2010 through 2015. Thinning treatments occurred on the treatment plots between 2010 and 2011. Note that the vertical axis scales vary among these graphs in order to best present each cover type. Error bars represent ± one standard error of the mean.

Table 2.9. Test Results for Paired T-tests of No Difference Between Mean Values of Vegetation and Ground Cover Types Measured from Vegetation Quadrats on Each Study Plot Pair at the Four Study Sites in 2015. Refer to Figure 2.59 for graphical illustrations of differences in mean values over years.

Site	Parameter	Control Mean	Treatment Mean	p-value (significance)
Chilili	All herbs (forbs and grasses)	3.4	2.9	0.8953
	Forbs	2.7	1.2	0.4696
	Grasses	8.7	3.4	0.8581
	Bare soil	2.8	0.0	0.1275
	Cryptobiotic crust	0.5	0.0	0.0995
	Leaf litter	75.8	65.7	0.0895
	Wood chips	**0.0**	**15.6**	**<0.0001**
Kelly	**All herbs (forbs and grasses)**	**19.8**	**33.0**	**0.0103**
	Forbs	2.0	5.0	0.1210
	Grasses	**17.8**	**28.0**	**0.0350**
	Bare soil	13.7	13.7	0.9914
	Cryptobiotic crust	**26.6**	**13.7**	**<0.0001**
	Leaf litter	40.9	28.7	0.2014
	Wood chips	**0.0**	**22.8**	**<0.0001**
Vigil	**All herbs (forbs and grasses)**	**28.6**	**68.0**	**0.0002**
	Forbs	**9.7**	**36.3**	**<0.0001**
	Grasses	**19.0**	**31.7**	**0.0248**
	Bare soil	17.7	21.5	0.5162
	Cryptobiotic crust	**29.8**	**21.5**	**<0.0001**
	Leaf litter	**34.8**	**14.8**	**0.0226**
	Wood chips	**0.0**	**20.1**	**<0.0001**
Wester	**All herbs (forbs and grasses)**	**12.2**	**32.0**	**<0.0001**
	Forbs	**1.1**	**4.0**	**0.0096**
	Grasses	**11.2**	**28.0**	**0.0004**
	Bare soil	**0.3**	**3.2**	**0.0149**
	Cryptobiotic crust	1.6	0.6	0.5219
	Leaf litter	**77.9**	**47.4**	**<0.0001**
	Wood chips	**0.0**	**18.0**	**<0.0001**

Note: Results in rows that are presented in **bold** were significantly different ($p < 0.05$) between control and treatment plots. All tests were with sample sizes of 36; p-values of less than 0.05 represent significant differences.

Total herbaceous vegetation canopy cover was significantly greater on all treated plots at all sites except for Chilili in 2015 (see Figure 2.59, a–d; Table 2.9). Total herbaceous canopy cover was not significantly different between control and treatment plots at any site in 2010 prior to tree thinning treatments, but total herbaceous cover was significantly greater on treated plots at all sites except Chilili from 2011 through 2015 following thinning treatments. The amount of herbaceous vegetation cover on treatment plots has tended to increase even more relative to control plots at the three sites other than Chilili through 2015, except for a leveling off and decline in total canopy cover at the Kelly site between 2013 and 2015 (Figure 2.59, a–d). The Vigil site has shown the greatest increase of herbaceous vegetation on the treatment plot compared to the control plot over time. Total herbaceous vegetation cover has been consistently about twice as high at the two piñon/juniper sites compared to the two ponderosa pine sites over time.

Forb canopy cover was significantly higher on the treated plots at both of the piñon/juniper sites from 2011 through 2014, but not at either of the ponderosa pine sites (see Figure 2.59, e–h). Forb cover at the Vigil site continued to increase even greater on the treatment plot than the control plot through 2014, but declined on the treatment plot relative to the control plot at the Kelly site in 2014. The majority of forb species were summer annual plants that grew on disturbed soils and wood chips. All of the dominant species were native, and no exotic invasive forb species were found through 2013. However, in 2014, three species of exotic invasive weeds were found represented by one plant only, and only on piñon/juniper treatment plots. Those weeds were prickly Russian thistle (*Salsola tragus*) (one plant on the Vigil treatment plot), redstem storks bill (*Erodium cicutarium*) (one plant on the Kelly treatment plot), and puncture vine (*Tribulus terrestris*) (one plant on the Vigil treatment plot). Common mullen (*Verbascum thapsus*), an additional species of exotic weed, was found at the Vigil site in small numbers in 2015. All four of these weed species are common in the study area and have been observed along roadsides and near the study sites since 2008, but have not become common on any of the study plots. An example of the forb growth at the Vigil treatment plot in 2015 is provided in Figure 2.60.

Figure 2.60. Forbs, grasses and sapling piñon and juniper trees growing on the Vigil site treatment plot, September 2015.

Grass canopy cover has tended to increase overall from 2011 through 2014, and then decreased slightly in 2015, except at the Wester site where it continued to increase (Figure 2.48, i-l). Grass cover was significantly higher on the treated plots at the Vigil piñon/juniper site and at the Wester ponderosa pine site in 2015, but there were not significant differences in grass cover at the Kelly and Chilili sites (see Figure 2.59, i–l; Table 2.9). Grass cover increased steadily on the treated plot at the Vigil site from 2010 to 2015 with significantly greater cover on the treatment plot since 2012. Grass cover was significantly higher on the treated plot at the Wester ponderosa pine site even before thinning treatments in 2010. However, through 2015, grass cover on the treatment plot has continued to increase more rapidly and proportionately more so than on the control plot, indicating a positive response to the thinning treatment. Domestic cattle grazing occurred at all sites except the Wester site from 2008 through 2015. Dominant grasses at the two piñon/juniper sites that responded positively to tree thinning were perennial species such as blue grama (*Bouteloua gracilis*) and James' galleta (*Pleuraphis jamesii*). Those grasses grew through the wood chips from existing individual plants that were in place prior to thinning treatments, unlike annual forbs that colonized the disturbed soils and wood chips. Blue grama growing through wood chips at the Kelly site in 2015 is shown in Figure 2.61.

Figure 2.61. Perennial blue grama growing through wood chips (foreground) at the Kelly piñon/ juniper site treatment plot in 2015. Note that the blue grama had been grazed down by livestock away from the cactus that protected grass from cattle grazing.

The extent of bare soil surfaces has been variable among control and treatment plots since 2010 and there have been no consistent patterns since thinning treatments in 2010 (see Figure 2.59, m–p; Table 2.9). Bare soil was significantly higher on control plots than on treated plots at the two piñon/juniper sites, but not at the two ponderosa pine sites from 2010 through 2013. In 2014 the amount of bare soil was no longer greater on the Kelly treatment plot compared to the control plot, but bare soil did remain significantly greater on the Vigil control plot in 2014 (see Figure 2.59, m–p; Table 2.9). In 2015 there was significantly more bare soil on the treatment plot only at the Wester site. Bare soil is an inverse of other ground cover features such as vegetation, leaf litter, and wood chips. The addition of wood chips in late 2010 resulted in less bare soil on treatment plots following thinning treatments at those two piñon/juniper sites. Considerable amounts of leaf litter were already present on the ground at the ponderosa pine sites, so additional wood chips did not change the amount of bare soil present as at the piñon/juniper sites where more bare soil was present to begin with. Increases in herbaceous vegetation since 2011 have likely affected variation in the amounts of exposed bare soil, but not in a clear way, probably due to changes in leaf litter, wood chips, and biotic crusts.

Cryptobiotic or biotic soil crust cover has tended to increase on all control plots from 2011 through 2014, with a decrease at the Wester control plot between 2013 and 2014. Biotic soil crust was significantly higher on the control plots than on the treatment plots at both of the piñon/ juniper sites, but not at either ponderosa pine site in 2014 and again in 2015 (see Figure 2.59, q–t; Table 2.9). However, biotic soil crust cover was significantly higher on those control plots prior to thinning treatments in 2010, so the difference is apparently not due to thinning treatment effects, even though wood chips probably covered biotic crusts on soil surfaces. Recent increases in biotic crust cover may be associated with increased late summer rainfall in 2013 through 2015.

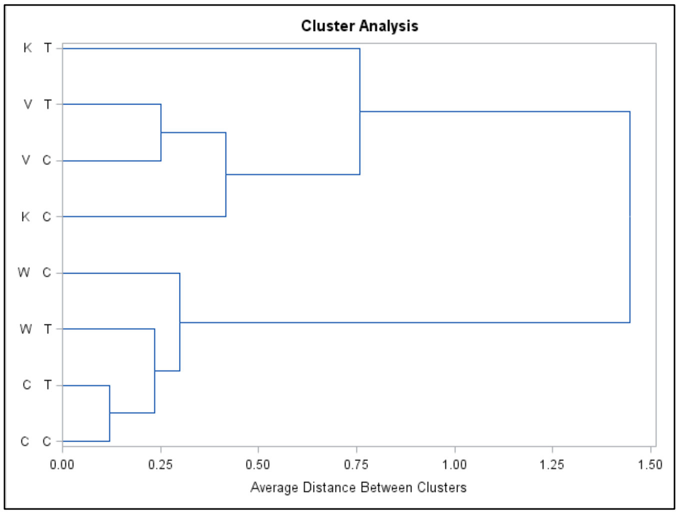
Leaf litter cover was higher on the control plots at both ponderosa pine sites in 2011 and 2012 following thinning treatments, but not in 2010 prior to treatments (see Figure 2.59, u–x; Table 2.9). Leaf litter was not significantly different between any of the paired control and treatment plots in 2013 and 2014, but was again significantly greater on control plots at both the Vigil and Wester sites and 2015 (Figure 2.59, u–x; Table 2.9). This finding may be due to the addition of wood chips in 2010–2011 that covered the extensive leaf litter layers at those two ponderosa pine sites, while the two piñon/juniper sites had less leaf litter than bare soil, as stated above. Since 2012, leaf litter is probably now starting to accumulate on the ground surface, becoming more equal between control and treatment plots. However, 2015 findings of significantly more leaf litter on the ground at the Vigil and Wester control plots indicates that high tree densities on control plots may again be contributing more leaf litter to the ground surfaces at those sites.

Wood chips have remained significantly higher in 2015 on all treated 2011 following thinning treatments (see Figure 2.59, y–bb; Table 2.9). No wood chips existed prior to thinning treatments in 2010, and they were applied only to the treated plots in late 2010. Wood chip cover on the treated plots declined by about 20% on all plots between 2011 and 2012, indicating some decomposition or redistribution and/or increased herbaceous plant canopy cover over wood chips. Since 2012, wood chip cover has remained fairly constant at the two piñon/juniper sites. In 2015, wood chip cover increased back up to about 20% cover at the two ponderosa pine sites. No new wood chips have been added at those sites. Such fluctuations are likely due to redistribution of chips following heavy rain and runoff events, and shifts in relative ground surface cover with other cover features. Figure 2.62 shows an example of wood chips that have been redistributed on the ground surface from heavy rain and runoff on the Kelly treatment plot in 2014. Similar redistribution occurred again in 2015.

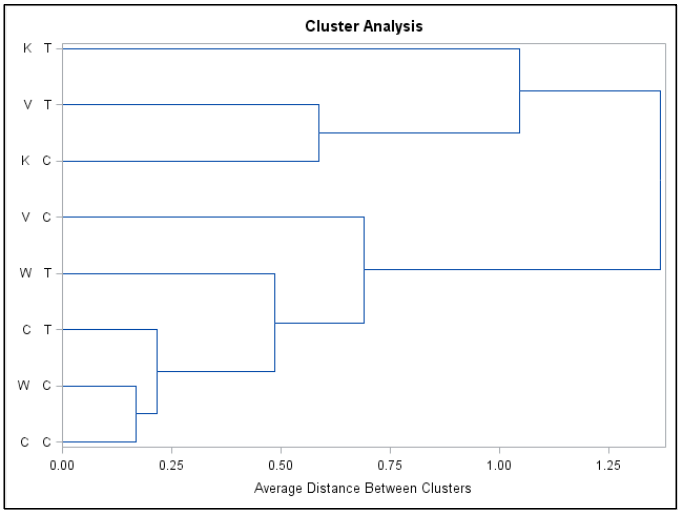
Figure 2.62. Wood chips from tree thinning treatment redistributed over the ground surface by heavy summer rain runoff at the Kelly treatment plot in 2014. The wood chips also

formed small debris dams to retain soil surface water runoff, likely increasing infiltration and reducing soil erosion.

Measurements of herbaceous vegetation on the thirty-six 1-m² (10.8-square-foot) quadrats also provided information on the canopy cover of each plant species per quadrat. The similarity of plant species composition among all of the study plots since 2010 was evaluated with the analytical method called cluster analysis (McCune and Grace 2002). Cluster analysis is useful for evaluating sets of species abundance when many species are involved. Cluster analysis compares sets of species/abundance data and determines how similar those sets are and then graphically represents their similarities as dendrograms or tree diagrams. The closer terminal branches are in those diagrams, the more similar those sets of species are in terms of composition and relative abundance. Cluster analysis dendrograms for all sites and plots for the spring and fall sampling periods for the years 2010 through 2014 are presented in Figure 2.63, a–e. Cluster analysis shows that in 2010 (see Figure 2.63 a), prior to tree thinning treatments, the ponderosa pine sites (Chilili and Wester) grouped together, the piñon/juniper sites (Kelly and Vigil) grouped together, and the paired plots at each ponderosa site were more similar to each other than to the other site. The Vigil paired plots also grouped together, but the Kelly plots were not as similar to each other as the Kelly control plot was to the Vigil plots, based on plant species compositions. There were no groupings of treatment versus control plots in 2010. In 2011, and again in 2012, those location-based groupings were less pronounced (see Figure 2.63, b–c), but still more important than similarities based on treatment versus control plots. In 2013, the Kelly and Vigil treatment plots grouped together, distinct from all other plots, probably as a result of the high densities of herbaceous plants on the treated plots. Both Chilili plots still grouped together, showing that location at Chilili was more important than treatment effects. In 2014 and in 2015 the sites again separated out primarily as all piñon/juniper and all ponderosa pine sites. The piñon/juniper plots were more similar by treatment type than location, but the ponderosa pine sites were not. These results indicate that the tree thinning treatments altered the location-based patterns found in 2010, and that the piñon/juniper sites are showing strong treatment effects on herbaceous vegetation, but the ponderosa pine sites less so.


Cluster Analysis
K T
V T
V C
K C
W C
W T
C T
C C
0.00
0.25
0.50
0.75
1.00
1.25
1.50
Average Distance Between Clusters

a. 2010.


Cluster Analysis
K T
V T
K C
V C
W T
C T
W C
C C
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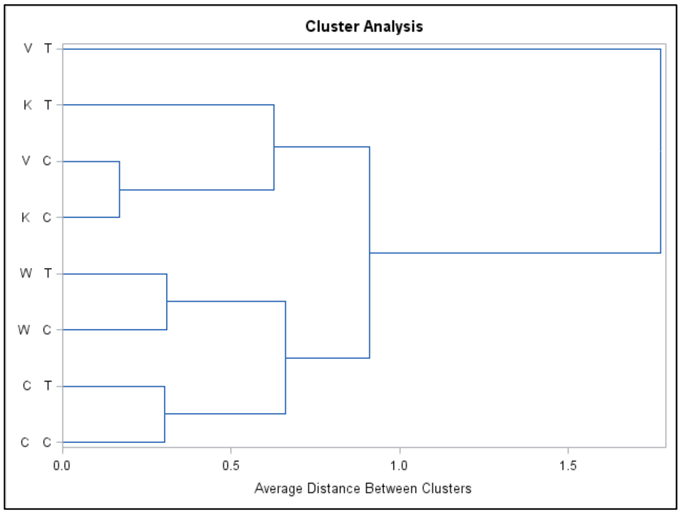
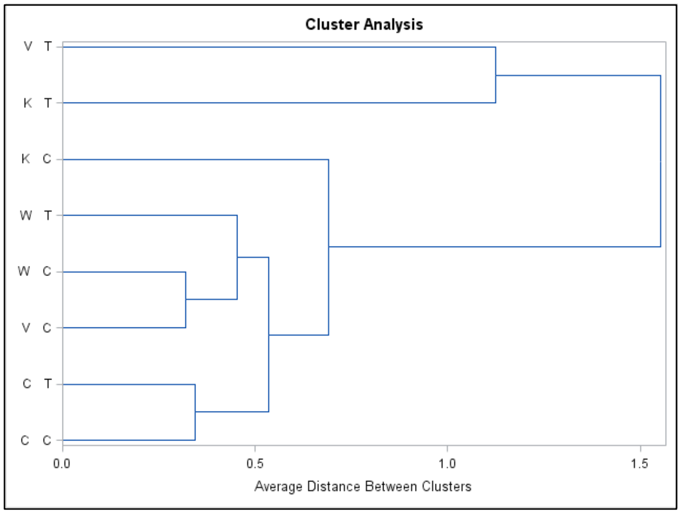
0.25
0.50
0.75
1.00
1.25
Average Distance Between Clusters

b. 2011.


Cluster Analysis
V T
K T
V C
K C
W T
W C
C T
C C
0.0
0.5
1.0
1.5
Average Distance Between Clusters

c. 2012.


Cluster Analysis
V T
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V C
C T
C C
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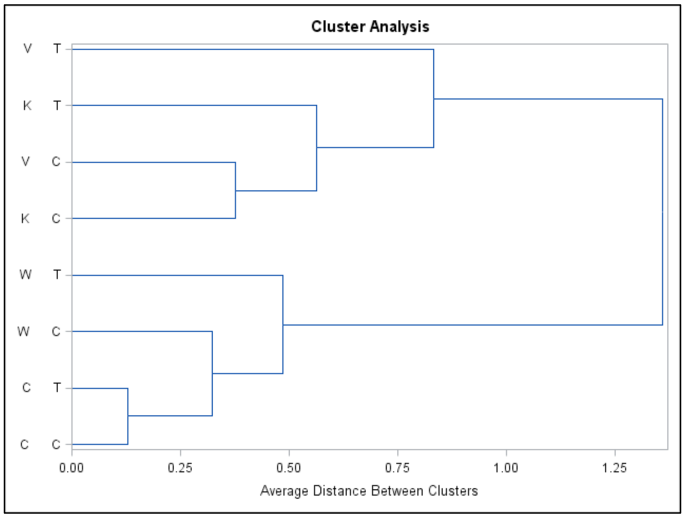
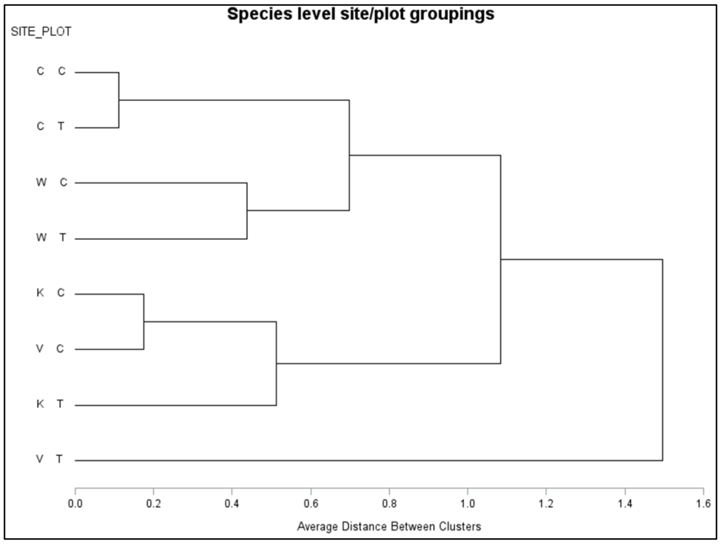
1.5
Average Distance Between Clusters

d. 2013.



e. 2014.



f. 2015.

Figure 2.63. Cluster analysis results showing the similarity of monitoring sites and paired plots based on similarity of the herbaceous plant community species compositions: a. 2010, b. 2011, c. 2012, d. 2013, e. 2014, f. 2015.

Results of vegetation and ground cover monitoring showed that forest thinning did affect the physical structure of the woodlands by reducing tree canopy and greatly affected the ground surface following the application of wood chips. Also, these findings show that at both piñon/juniper sites, herbaceous vegetation increased significantly on plots where trees were thinned, and that the patterns of location-based plant community similarities were altered by forest thinning treatments. Also, when measuring vegetation, we noticed that the vigorous growth of herbaceous plants, especially grasses, on the treated plots was being grazed heavily by domestic livestock. The removal of grass and forb canopies by livestock undoubtedly reduced canopy cover of those plants that were measured in late summer from all of the treatment plots. Therefore, actual herb growth or production on plots were likely higher on treatment plots than were measured, and the positive effects of tree thinning on those plots to herbaceous plants, especially grasses, were probably even greater than the data show. These findings are now five years following thinning treatments and graphs in Figure 2.63 show that trends between treatment and control plots are starting to show consistent patterns over time; generally greater herbaceous vegetation cover on treatment plots at the piñon/juniper sites, and at one of the ponderosa pine sites.

2.08.2 REPEAT PHOTO POINTS

Repeat photographic monitoring has been used for a variety of rapid assessment restoration monitoring purposes. Photo monitoring may be used for quantitative measurements of vegetation change by actually measuring vegetation in the photographs (Garrard et al. 2012; Hall 2001, 2002a, 2002b; Hamilton 2014; Powell 2006; Shaff et al. 2007; Tamarisk Coalition 2014). Photo monitoring also has been used for stream and wetland restoration to evaluate changes in riparian geomorphology, as well as vegetation (Kocher and Harris 2005; Shaff et al. 2007). The value of photo monitoring is that it is easy and inexpensive to take the photographs, and it takes little time or expertise to analyze the photographs. The primary drawback to qualitative photo monitoring is that the analysis of the photographs is somewhat subjective, and interpretation may vary among observers. Any photo monitoring protocol, especially interpretation and analysis, must be standardized and consistent among users in order to be accurate and effective.

The photo monitoring protocol used here was developed to be consistent with the procedures and protocols for Estancia Basin Watershed Health, Restoration and Monitoring forest thinning projects on private landowner properties, and to use the SWCA experimental monitoring data as a way to verify the scoring of repeat photographs relative to trends in the condition of soils and vegetation, including trees, based on field-measured quantitative data.

The purpose of repeat photo monitoring of forest thinning projects is to evaluate post-treatment changes in environmental parameters relative to the objectives of thinning for those parameters. Unlike high-intensity measurement monitoring, photo monitoring is a rapid assessment, qualitative evaluation of change in parameters as observed in repeat photographs over time. Rather than measuring parameter values, visual changes in parameter conditions are scored on a linear scale from low to high. Low to high rank scales are a common way of evaluating and scoring items such as Likert scales used in opinion surveys, and rank scales have been developed for photo monitoring (Garrard et al. 2012; also see Wikipedia.com 2014 for a detailed description of Likert scales). Rank scales cover a range of response values, from negative to neutral to positive, and the scores can be used to evaluate whether an attribute, parameter, or item is trending in a positive, negative, or static direction. Statistics can even be applied to rank scale scores from different people to test for significance differences in score trends among items from a series of photographs representing different photo points (Garrard et al. 2012). A rank scale is used to evaluate environmental change as positive, negative, or static for each of the forest thinning objective parameters.

The objectives for trends in change of environmental parameters of forest thinning sites (e.g., reduced tree stand density, increased herbaceous vegetation, etc.) serve as items on a response scoring scale. Those objectives become the parameters that are then evaluated for change. Forest environment parameters that are used as items for the evaluations of repeat photographs must be parameters that can be observed and evaluated in the photographs. The parameters used also should be those that are being quantitatively measured at the four experimental forest thinning monitoring sites by SWCA. Quantitative data collected from the experimental monitoring sites are used to verify the scoring of environmental items in the repeat photo monitoring.

Repeat Photo Items for Evaluation

The listing of forest thinning monitoring parameters presented above (e.g., soils, hydrology, vegetation, etc.) provides the list of parameters for repeat photo monitoring as well. However, only attributes of the parameters that can be viewed and evaluated in photographs can be used for photo monitoring. From the above list, the following parameters will be evaluated for change in repeat photographs from the forest thinning project sites:

- *Soils:* 1) Erosion and 2) surface stability.
    - Thinning objectives are to: 1) reduce soil erosion.
    - Soil erosion will appear as bare soil with surface rills, litter dams among bare soil, and rock and twig pedestals. Surface stability can be evaluated by differentiation of loose friable soil surfaces from crusted soil surfaces, and bare soil versus litter or wood chip cover.
- *Hydrology:* Runoff amounts cannot be evaluated from photographs, so hydrology will not be included in photo monitoring. However, indications of high levels of soil erosion involve high levels of runoff.
- *Trees and Woody Vegetation:* 1) Prescribed thinning treatment values for basal areas and age classes by species, 2) a change in growth and health of remaining trees, and 3) a reduction in vertical (standing) and dead/down (on the ground) wildfire fuels.
    - Thinning objectives are to: 1) reduce basal areas to prescribed levels, 2) increase growth and health of trees, and 3) reduce standing fire fuels.

  - Changes in tree density, vertical structure, and tree health are relatively easy to observe in repeat photographs.

- *Herbaceous Vegetation:* 1) A change in the canopy cover of herbaceous vegetation, 2) a change in the species composition and diversity of herbaceous vegetation, and 3) a change in the abundance and cover of invasive exotic weed species.
  - Thinning objectives are to: 1) increase herbaceous canopy cover, 2) favor establishment of perennial native herbs (grasses and forbs), and 3) reduce the cover of invasive exotic weed species.
  - Changes in herbaceous vegetation canopy cover and species diversity are relatively easy to observe in repeat photographs. Photographs were near the end of the summer growing season to view the maximum growth of herbaceous vegetation. Some but not all exotic invasive weeds may be observed in photographs.
- *Wildlife:* Cannot be evaluated from photographs.

Evaluating and Scoring Repeat Photos

Environmental change is evaluated by comparing photographs from the same photo point of the same view, taken at different times. In most cases, the photograph taken at the latest date is compared to the original pre-treatment or baseline photograph. However, any pair of photographs may be compared, depending on the need to evaluate change over any particular time period. Repeat photographs are evaluated for environmental change using photo monitoring evaluation forms where each environmental parameter is scored and other information recorded as follows:

1. Soil Erosion and Surface Stability
   - +2: Considerable decrease in soil erosion and increased surface stability
   - +1: Some decrease in soil erosion and increased surface stability
   - 0: No change in soil erosion or surface stability
   - -1: Some increased in soil erosion and reduced surface stability
   - -2: Considerable increase in soil erosion and reduced surface stability

Comments (*note other changes that are not scored and any uncertainty or questions about scoring*): Status of wood chip and leaf litter, amount of bare soil surfaces and their appearances, down woody material status, rivulet formation, etc. Leaf litter and wood chips increase surface stability and reduce erosion potential. Bare soil surfaces generally have lower surface stability and are prone to erosion, especially if surface crusts are lacking.

2. Tree Density and Vertical Wildfire Fuels

- +2: Considerably lower tree density and vertical fire fuels
- +1: Lower tree density and vertical fire fuels
- 0: No change in tree density and vertical fire fuels
- -1: Greater tree density and vertical fire fuels
- -2: Considerably greater tree density and vertical fire fuels

Comments (*note other changes that are not scored and any uncertainty or questions about scoring*): Change in tree species composition, size classes, etc.

3. Tree and Other Woody Vegetation Growth and Health

- +2: Considerable growth and more healthy trees
- +1: Some increased growth and more healthy trees
- 0: No change in tree growth or health
- -1: Some decreased tree growth and tree health
- -2: Considerable decreased tree growth and health

Comments (*note other changes that are not scored and any uncertainty or questions about scoring*): Condition by species, descriptive signs of health and growth, identification of insect/disease or other damage.

4. Herbaceous Vegetation

- +2: Considerably greater herbaceous vegetation cover and diversity
- +1: Greater herbaceous vegetation cover
- 0: No change in herbaceous vegetation cover and diversity
- -1: Lower herbaceous vegetation cover and diversity
- -2: Considerably lower herbaceous vegetation cover and diversity

Comments (*note other changes that are not scored and any uncertainty or questions about scoring*): Change in species composition, canopy height, dominant native grasses, any exotic invasive weeds, etc.

5. Other Observable Changes

Comments: Note any other changes not addressed above that may reflect site conditions relative to soils, hydrology, vegetation, wildlife, and wildfire fuels. Note how livestock grazing may affect visible vegetation cover.

Repeat Photo Analysis

The above scoring is conducted on a Photo Monitoring Restoration Effectiveness Analysis: 2-Photo Comparison photo monitoring data form that is a Microsoft Excel spreadsheet and calculates an overall score for each repeat photo comparison analysis for each tree thinning project, in this case each control and treatment plot at each site. Multiple photo point photographs and scores for a particular project and time period are then averaged to provide an overall average score by using a Photo Monitoring Restoration Effectiveness Analysis: Multi-Photo Averages form. Finally, a Photo Monitoring Restoration Effectiveness Analysis: Multi-Photo Score Trend form is used to determine score trends over time (for both single photo points and from averaged multiple photo points) and to actually evaluate score trends over time. The same scoring is applied to all photographs taken from any particular plot and across all sites and plots.

Analysis and Interpretation of Photo Monitoring Data

Scores from photo monitoring data forms may be combined across various dates, locations, etc., to evaluate changes in individual parameter scores and overall project scores over time. Any combination of repeat photographs may be used, depending on the various needs to evaluate change over time relative to tree thinning projects. The Photo Monitoring Restoration Effectiveness Analysis: Multi-Photo Score Trend form allows one to evaluate scores of interest that are recorded in a table on the spreadsheet, and then those values are graphed over time to evaluate score trends. A trend summary table also may be constructed to summarize trends across the various parameters for any particular tree thinning project or for series of projects representing certain geographic areas, forest types, or other categories.

The growth and health of vegetation each year depends considerably on weather conditions prior to the dates that photo-point photographs are taken. Analysis of repeat photographs must include considerations for previous weather conditions, especially rainfall, prior to each photograph or series of photographs analyzed. The interpretation of repeat photographs and score trends must include a discussion of weather/climate conditions over the range of time that the photographs represent. The growth and health of vegetation observable in the photographs may be more the result of past weather/climate than the thinning treatment itself.

As with weather, livestock grazing can have significant effects on the cover and height of herbaceous vegetation, the amount of bare soil, and the surface stability and the erosion of soil surfaces. If a site has experienced heavy livestock grazing, this is noted in the comments. In such cases, livestock grazing, like weather, may have a greater impact and observable effect than the thinning treatment alone.

Any other environmental factors or land management/use practices, such as follow-up thinning treatments, brush control, erosion control, etc., that may affect the appearance of soils, trees, and herbaceous vegetation are documented and considered when evaluating repeat photographs for tree thinning affects. All of the above are considered for commenting at the bottom of each

Photo-Monitoring Restoration Effectiveness Analysis: 2-Photo Comparisons form for each repeat photograph.

The tree thinning treatments were started and completed at all sites during the winter of 2010–2011.Using the above protocols, repeat photo point photographs from the treatment plots at all four sites have been evaluated for changes relative to tree thinning treatments between 2011, less than one year following thinning treatments, and 2015, five years following treatments. Repeat photo point photographs taken in All 2011 and 2015 photographs and scoring analysis worksheets are presented in Appendix B. The 2014 Annual Report provided a different analysis of repeat photo point photographs, evaluating trends prior to the thinning treatments in 2010, up to 2014. The analysis presented here focuses on trends over the past 5 years following the thinning treatments in 2010.

Summary trend scores and a graph of those scores for the Chilili ponderosa pine treatment plot are presented in Table 2.10 and Figure 2.64. The score for herbaceous vegetation and the overall score changed in a slight positive way from 2011 to 2015, but no other parameters changed.. These findings tend to correspond to the tree and herbaceous vegetation findings based on the measurements above. However, soil erosion measurements have shown an increase in soil loss at the Chilili treatment plot that is not reflected in the photographs.

Table 2.10. Repeat Photo Point Interpretation Scores Averaged over Three Photo Points for the Chilili Ponderosa Pine Treatment Plot, 2011 to 2015 (1 year post treatment to 5 years post treatment.

Year 1	Year 5	Overall	Soil	Tree Density	Tree Health	Herbs
2011	2015	0.3	0	0	0	0.3

Note: Zero means no change over the time between photographs, positive numbers mean favorable change, and negative numbers mean unfavorable change.

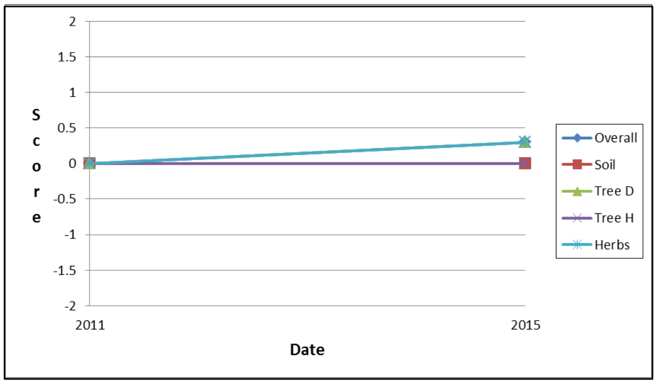


Figure 2.64. Parameter trend scores based on repeat photo point interpretation for the Chilili ponderosa pine site from 2011 through 2015.

Summary trend scores and a graph of those scores for the Kelly piñon/juniper treatment plot from 2011 to 2015are presented in Table 2.11 and Figure 2.65. Results show that the herbaceous vegetation, tree health and overall score increased from 2011 to 2015, but the tree density score declined due to an increase in small juniper and piñon establishement and growth.

Table 2.11. Repeat Photo Point Interpretation Scores Averaged over Three Photo Points for the Kelly Piñon/juniper Treatment plot, 2011–2015.

Year 1	Year 5	Overall	Soil	Tree Density	Tree Health	Herbs
2011	2015	1	0	-1	1	1

Note: Zero means no change over the time between photographs, positive numbers mean favorable change, negative numbers mean unfavorable change.

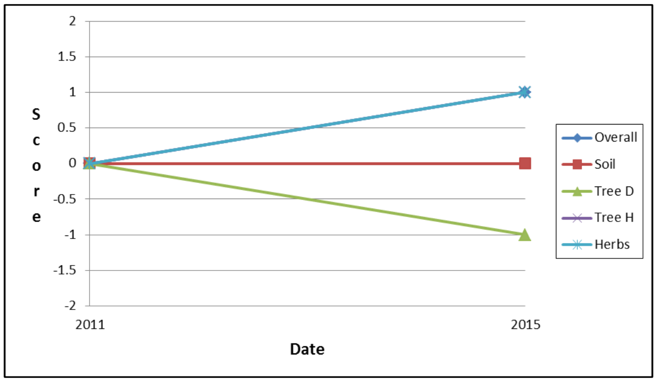


Figure 2.65. Parameter trend scores based on repeat photo point interpretation for the Kelly piñon/juniper treatment plot from 2011 through 2015.

Summary trend scores and a graph of those scores for the Vigil piñon/juniper treatment plot are presented in Table 2.12 and Figure 2.66. Results show that the herbaceous vegetation, tree health and the overall score increased from 2011 to 2015, but the tree density score declined over the same time due to the appearance of small juniper and piñon trees.

Table 2.12. Repeat Photo Point Interpretation Scores Averaged over Three Photo Points for the Vigil Piñon/juniper Treatment plot, 2011–2015.

Year 1	Year 5	Overall	Soil	Tree Density	Tree Health	Herbs
2011	2015	1.3	0	-1	1.3	1

Note: Zero means no change over the time between photographs, positive numbers mean favorable change, negative numbers mean unfavorable change.

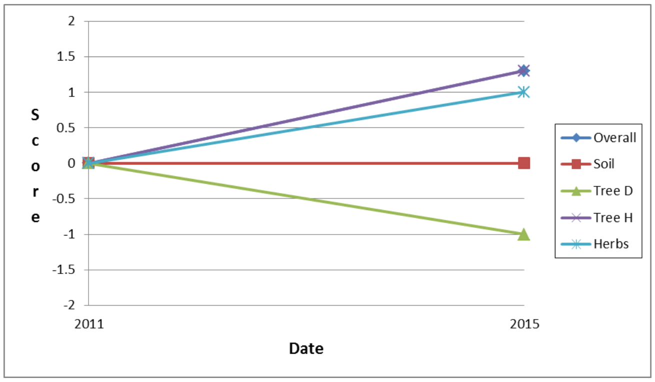


Figure 2.66. Parameter trend scores based on repeat photo point interpretation for the Vigil piñon/juniper treatment plot from 2011 through 2015.

Summary trend scores and a graph of those scores for the Wester ponderosa pine treatment plot are presented in Table 2.13 and Figure 2.67. Herbaceous vegetation, tree health and the overall score increased between 2011 and 2015, while soil and tree density remained the same.

Table 2.13. Repeat Photo Point Interpretation Scores Averaged over Three Photo Points for the Wester Ponderosa Pine Treatment plot, 2011–2015. No photos were taken in 2010.

Year 1	Year 5	Overall	Soil	Tree Density	Tree Health	Herbs
2011	2015	0	0	0	0.7	1

Note: Zero means no change over the time between photographs, positive numbers mean favorable change, negative numbers mean unfavorable change.

Figure 2.67. Parameter trend scores based on repeat photo point interpretation for the Wester ponderosa pine site treatment plot from 2011 through 2015.

Overall, repeat photo point analysis results were similar to the trends for the same parameters that we have quantitatively measured on the study plots, showing improvements in the status of herbaceous vegetation and tree health, including the growth of young trees, which in turn is causing a negative trend due to increased numbers of small trees providing ladder fuels for wildfires.

2.08.3 PHOTOGRPAHIC DOCUMENTATION OF POST-THINNING CONIFER TREE REESTABLISHMENT

Additional photographs were taken on the monitoring plots in 2015 to document the growth of conifer trees following the 2010/2011 thinning treatments. Ponderosa pine establishment has been minimal at both the Chilili and Wester ponderosa pine sites. No obvious ponderosa pine seedlings have been observed at the Wester site, on either control or treatment plots. However, numerous ponderosa pine seedlings were first observed at Chilili treatment plot in the late summer of 2011 (Figure 2.68), but none on the control plot. Figure 2.69 shows an example of 3 ponderosa pine saplings on the Chilili treatment plot in 2015 that are almost 30 cm (1-foot) tall. The ponderosa pine seedlings and saplings have been observed growing in the wood chip mulch, but not in any locations lacking wood chips. The wood chip mulch apparently created a microenvironment condusive to ponderosa pine seed germination and establishment. Many similar sized ponderosa pine saplings were observed on the Chilili treatment plot, but none were observed on the control plot where leaf litter was present but wood chips were not.

Figure 2.68. Ponderosa pine seedlings first becoming established and photographed on the Chilili treatment plot in late summer 2011.

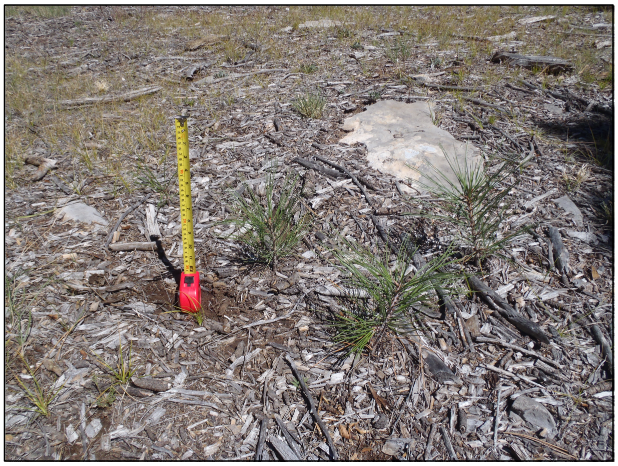
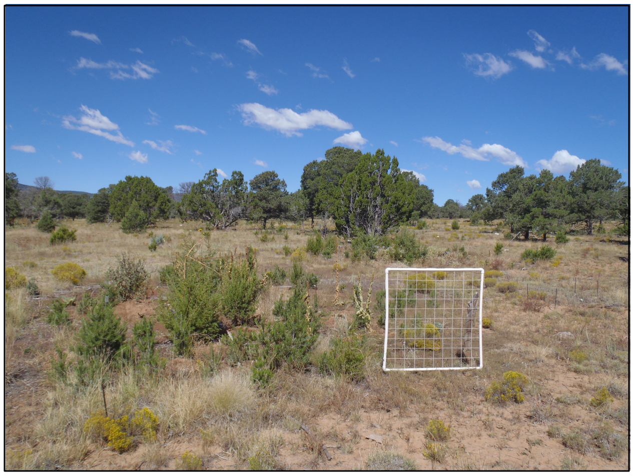


Figure 2.69. Ponderosa pine saplings photographed on the Chilili treatment plot in 2015, that had germinated in the summer of 2011 (Figure XX above).

Conifer establishment also has been occurring at the two piñon/juniper sites, on the treatment plots but not on the control plots. One-seed juniper and piñon saplings have both been growing on patches of leaf litter that had accumulated under conifer trees that were removed in 2010/2011 (Figure 2.70 and Figure 2.71). Some of those saplings were probably present at the time of tree thinning and others apparently have germinated since. The saplings at both sites are approaching 1 m (3-feet) in height, and are clearly associated with historic tree litter/duff patches. Few have been observed in other locations such as wood chips spread over the ground, as observed above at the Chilili ponderosa site.

a.



b.

Figure 2.70. **One-seed juniper and piñon sapling growth at the Vigil piñon/juniper treatment site in 2015. a. Numerous saplings growing in a leaf litter patch that had accumulated under trees that were removed in 2010/2011. b. Another patch of saplings at the Vigil treatment site, the measurement frame is 1 m in height for scale. Also note how tree recruitement is largely limited to leaf litter patches were trees were removed.**

Figure 2.71. Piñon pine regeneration on the Kelly treatment plot photographed in 2015. The frame is 1 m tall.

2.09 WILDLIFE

Birds and small mammals are being monitored to determine if forest thinning affects native wildlife species. Both birds and small mammals were recorded from separate 50 × 50–m (164 × 164–foot) wildlife study plots that are immediately adjacent to each of the two vegetation and soils monitoring study plots at the four study sites. Birds and mammals were measured in late spring (May/June) and early fall (September/October) each year from 2008 through 2015, for 3 consecutive days on each study plot.

2.09.1 BIRDS

The species composition and relative abundance of birds on all study plots were recorded by observing birds by point counts from one location at the center of each wildlife study plot. Each point count was conducted for 20 minutes at dawn for three consecutive mornings on each study plot in both spring and fall. Spring counts are intended to assess breeding bird use of the forest and woodland habitats, and fall counts are intended to assess migratory bird use of the same habitats. Many of the bird observations were based on hearing songs and calls and identifying those to species. Additionally, visual observations were often recorded.

Counts of individual birds in 2015 revealed that more birds were found on all treated plots than on control plots in the spring, and while more birds were found at all control plots in the fall, except for Vigil(Figure 2.72). Numbers of bird species were also consistently higher on treatment plots at all sites in the spring, while more species were found at all control plots in the fall, except Wester (Figure 2.73).

Figure 2.72. Numbers of individual birds recorded from thinning treatment and control plots across the four study sites in both spring and fall 2015.

Figure 2.73. Numbers of bird species recorded from thinning treatment and control plots across the four study sites in both spring and fall 2015.

Figure 2.74 presents the species and numbers of individuals of those species summarized in Figure 2.72 and Figure 2.73. Examination of Figure 2.74 shows that in most cases, different species of birds dominate the spring and fall bird communities. Such findings should be expected, given that some of the species that breed at those sites in the spring migrate south in the fall and are replaced by species and individuals that migrated to the sites from locations farther north and/or higher in elevation.

a. Chilili control and treatment plots, spring 2015.

b. Chilili control and treatment plots, fall 2015.

c. Kelly control and treatment plots, spring 2015.

d. Kelly control and treatment plots, fall 2015.

e. Vigil control and treatment plots, spring 2015.

f. Vigil control and treatment plots, fall 2015.

g. Wester control and treatment plots, spring 2015.

h. Wester control and treatment plots, fall 2015.

Figure 2.74. Numbers of individuals of each bird species recorded from all control and treatment study plots in 2015, both spring and fall.

Figure 2.75 presents total numbers of birds from control and treatment plots, both in spring and fall, from all four study sites from fall 2008 through fall 2014. In general, overall bird counts increased over the years up to 2011 and 2012, but then declined in 2013 and 2014. Comparisons of treated versus control plots since thinning treatments in late 2010 have shown increases on treated plots at the two piñon/juniper sites, but not at the ponderosa pine sites.

a. Chilili, spring.

b. Chilili, fall.

c. Kelly, spring.

d. Kelly, fall.

e. Vigil, spring.

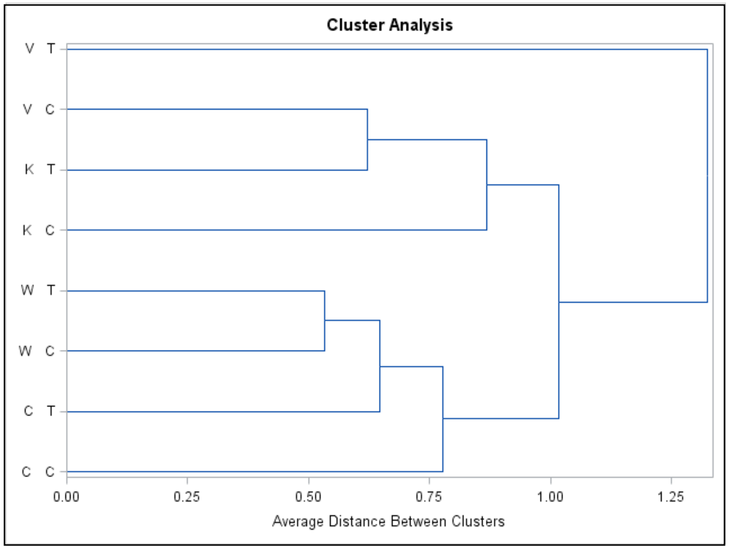
f. Vigil, fall.

g. Wester, spring.

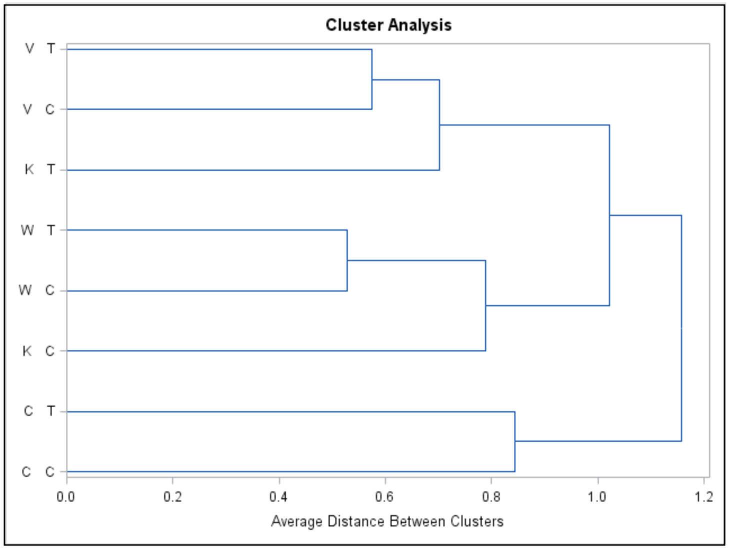
h. Wester, fall.

Figure 2.75. Total numbers of birds from both control and treatment plots at all four study sites, fall 2008 to fall 2015.

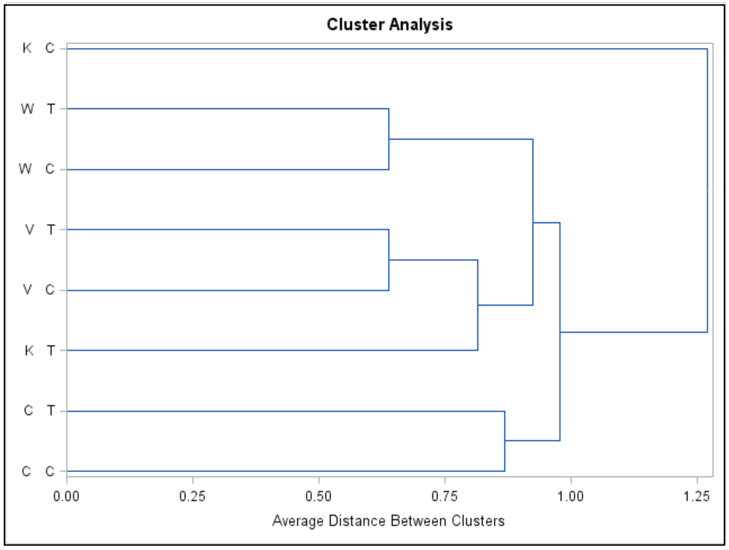
Cluster analysis dendrograms for all sites and plots for the spring and fall sampling periods for 2008 through 2015 are presented in Figure 2.76, a–p. Cluster analysis shows that over the 8-year period from 2008 through 2015, bird communities were most similar to each other based on location. The ponderosa pine sites and plots within sites tended to group together, and the piñon/juniper sites and plots within sites tended to group together. This pattern was especially pronounced during the spring breeding period when birds locate nests and establish feeding territories in preferred habitats. Through 2014, the bird communities have not shown a strong response to forest thinning treatments; if that had been the case, plots would have been grouped together based on treatment status rather than location. In 2015, bird species compostions and counts did show treatment effects at the ponderosa pine sites, with the Chilili and Wester control plots and treatment plots grouping together based on treatment rather than location. However, the Kelly and Vigil pinion-juniper sites still grouped based on location, not treatment. The same pattern was similar in the fall of 2015, except that the Chilili control site differed from all other sites, apparently due to very low bird species richness and numbers of individuals. These findings indicate that thinning treatment effects may be starting to affect bird communities in the ponderosa pine sites, five years following thinning treatments.



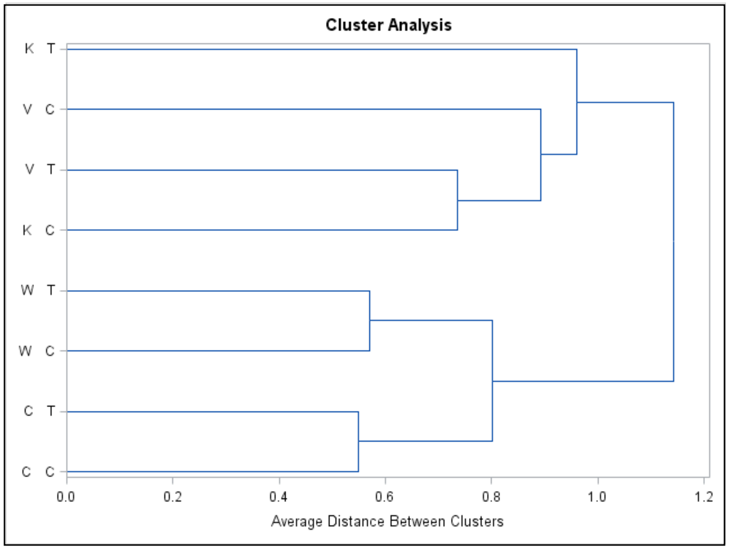
a. 2008, fall.



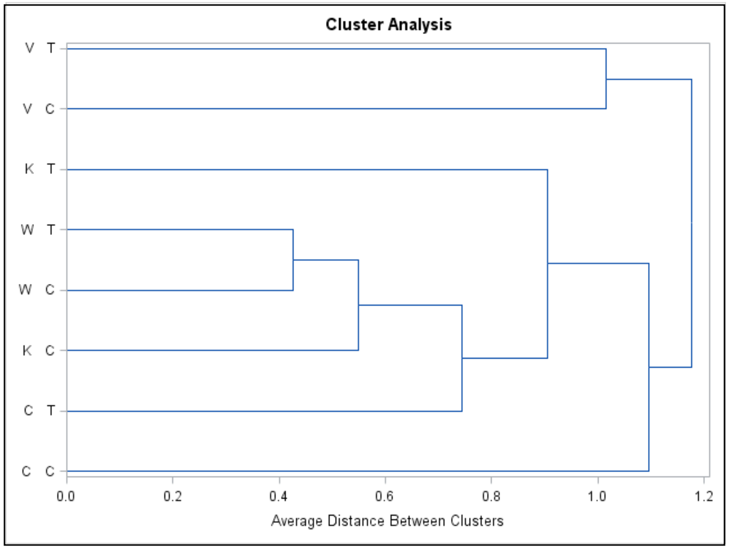
b. 2009, spring.



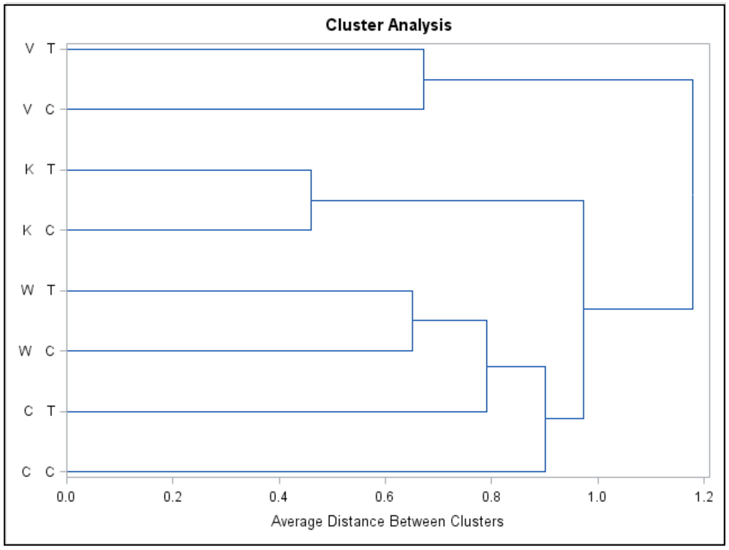
c. 2009, fall.



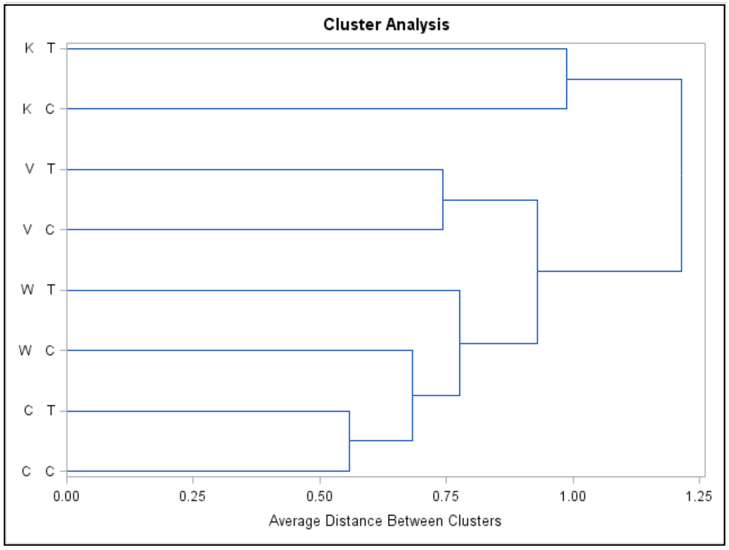
d. 2010, spring.



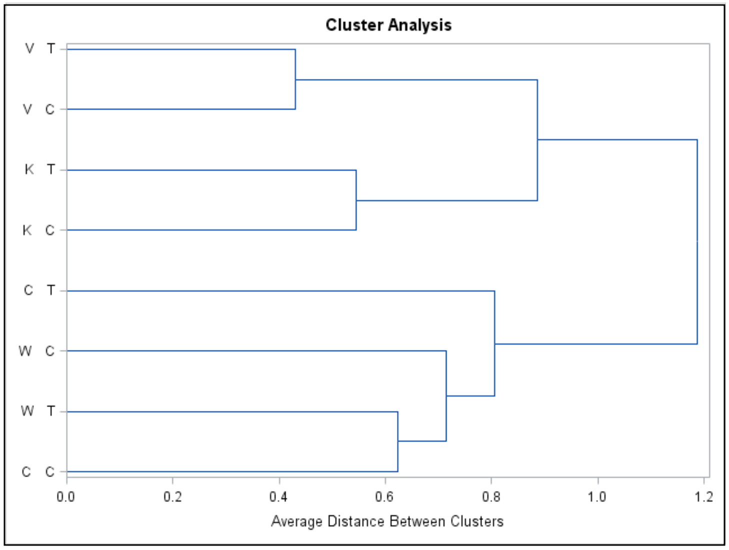
e. 2010, fall.



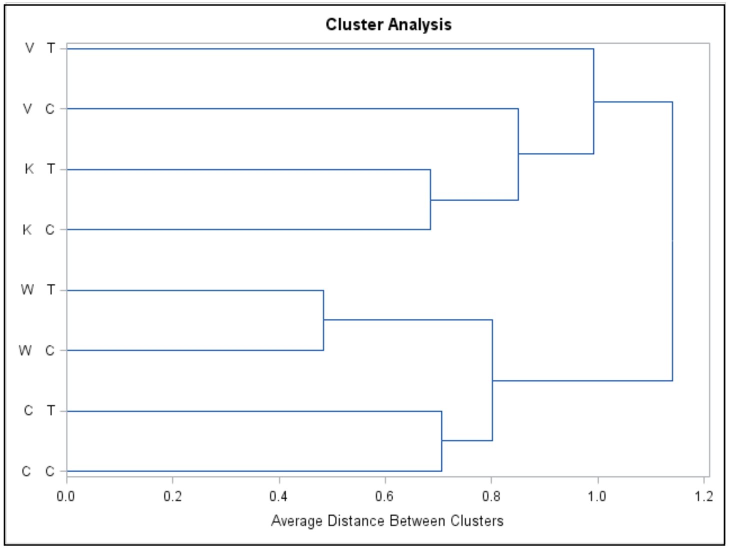
f. 2011, spring.



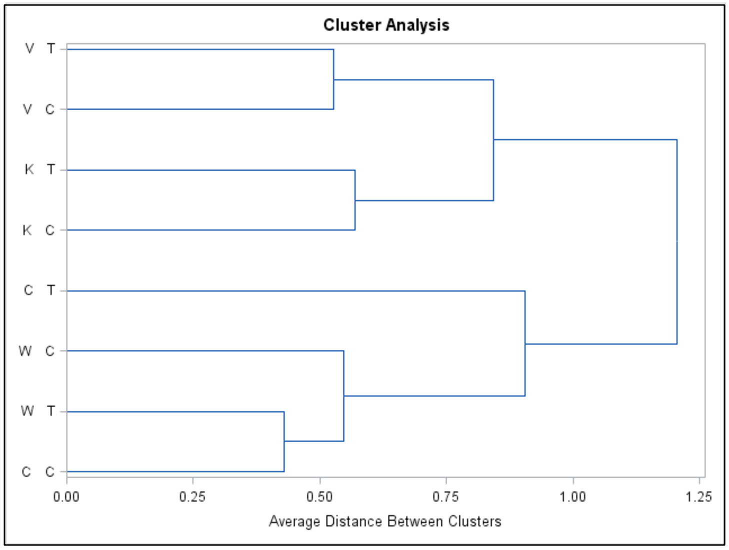
g. 2011, fall.



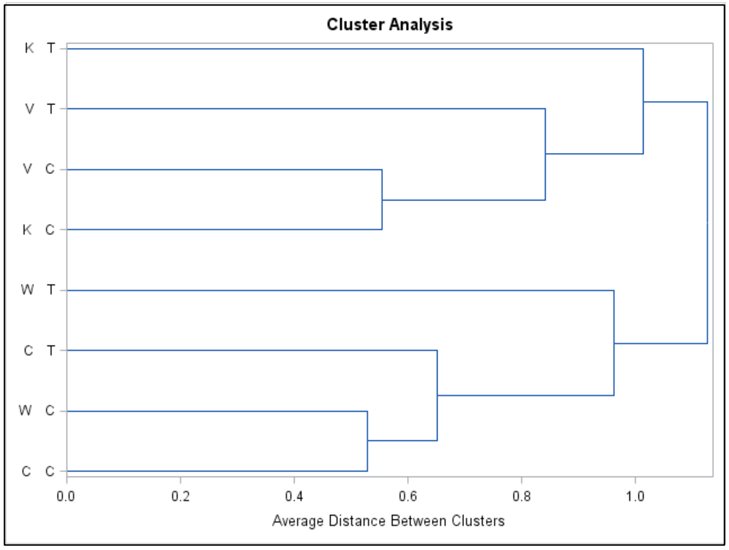
i. 2012, spring.



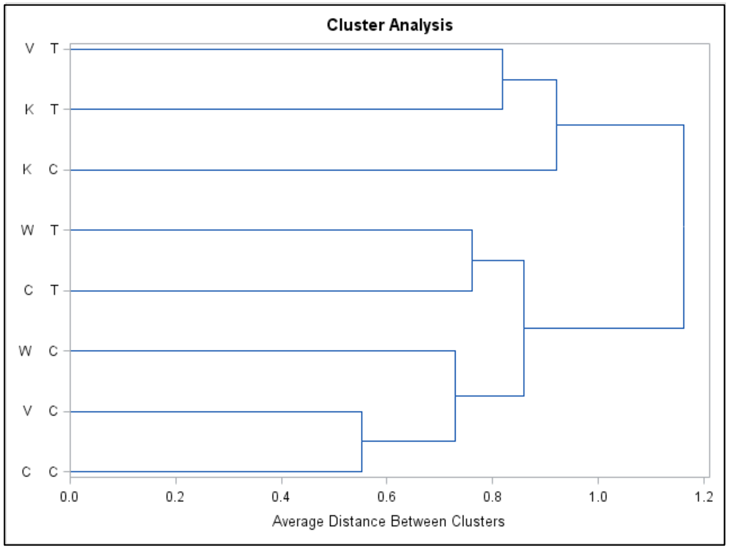
j. 2012, fall.



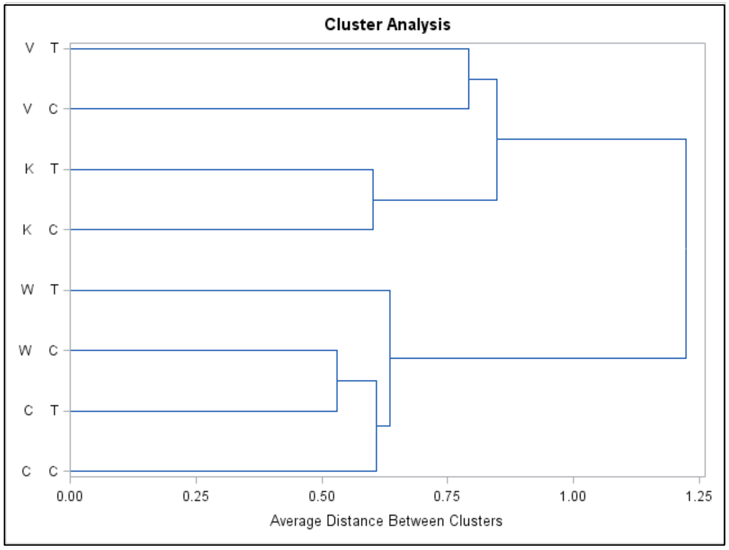
k. 2013, spring



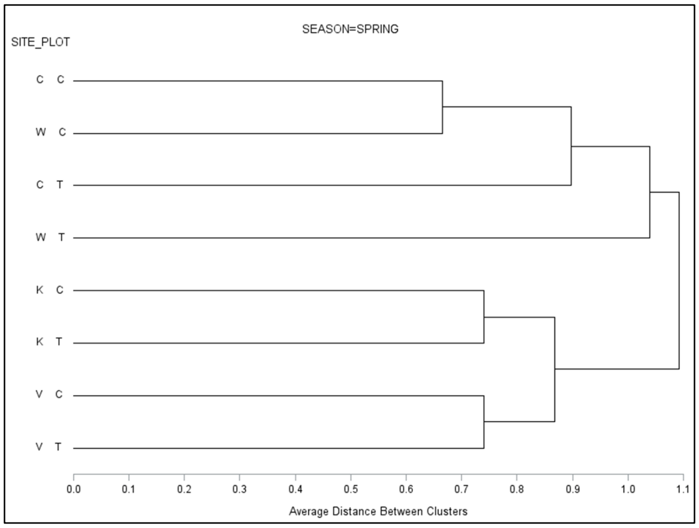
l. 2013, fall.



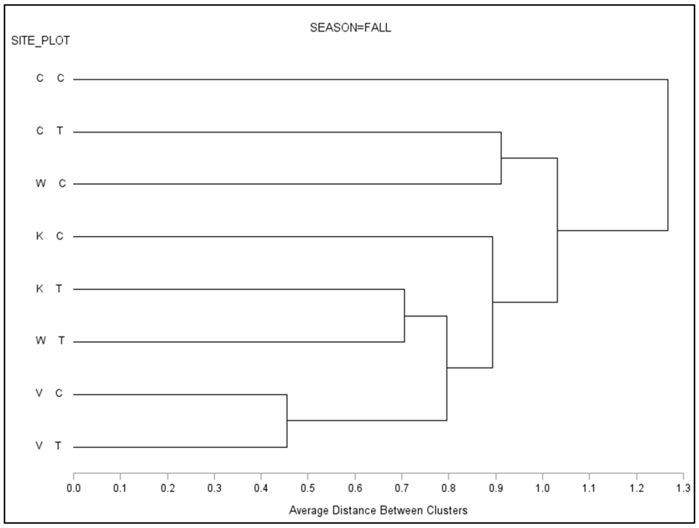
m. 2014, spring.



n. 2014, fall.



o. 2015, spring



p. 2015, fall

Figure 2.76. Cluster analysis dendrograms showing similarities of monitoring sites/plots based on bird species composition, spring and fall 2008–2010 prior to tree thinning treatments (a–e), spring and fall 2011 (f–g), and spring and fall 2012–2015 following thinning treatments (i–p).

2.09.2 SMALL MAMMALS

Small mammals (rodents) were sampled from a single six by six–trap grid (36 traps total) of live-capture rodent traps set at 10-m (33-foot) intervals on each of the wildlife monitoring plots for three consecutive nights in spring and fall, the same dates that birds were sampled in 2008 through 2015. Counts of individual rodents in 2015 revealed that rodent densities were considerably greater on the treated plots than on the control plots, and that rodent densities declined through the summer and were greater in the spring than in the fall than in the spring (Figure 2.77). Numbers of rodent species tended to be greater on the treatment plots than on the control plots in the spring, but greater on control plots than treatment plots in the fall (Figure 2.78). These findings show that in 2015, both numbers of individual rodents were consistently greater on control plots than treatment plots, numbers decreased from spring to fall, and numbers of rodent species were greater on treatment plots in the spring, but greater on control plots in the fall. These findings indicate that rodents preferred the control plots to the treatment plots, indicating better habitat and food resources on the control plots. Both piñon and ponderosa pine produced large numbers of seeds in 2015, perhaps providing more food for rodents on plots were trees were most dense.

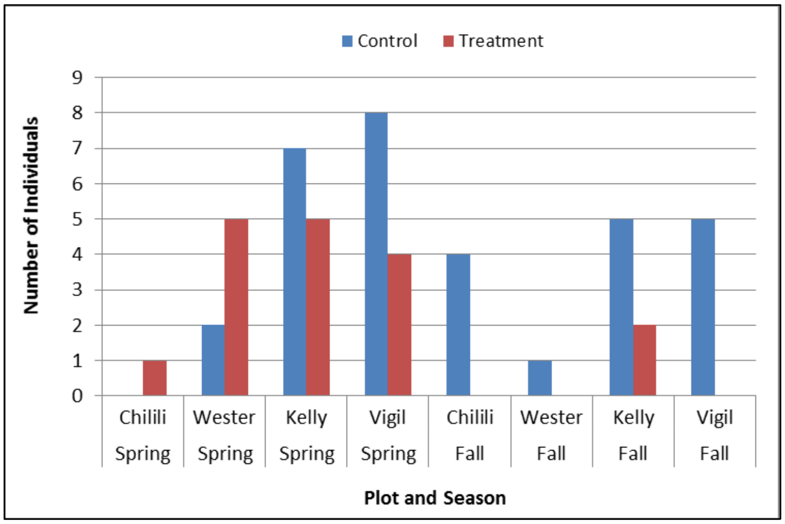


Figure 2.77. Numbers of individual rodents recorded from thinning treatment and control plots across the four study sites in both spring and fall, 2015.

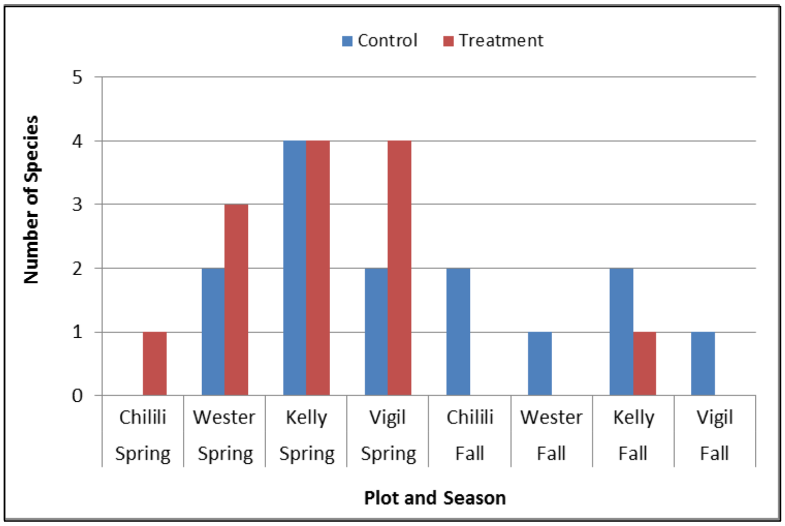
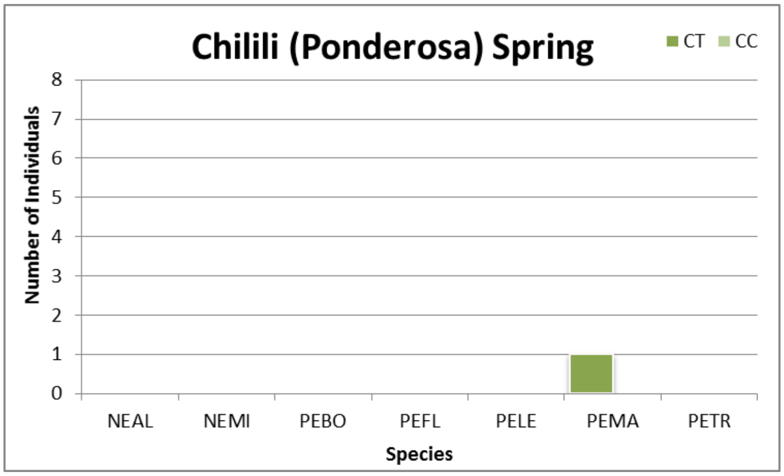


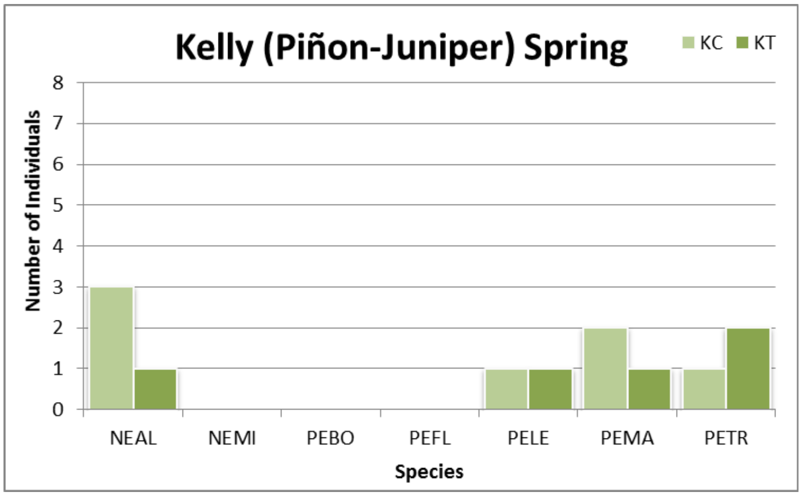
Figure 2.78. Numbers of rodent species recorded from thinning treatment and control plots across the four study sites in both spring and fall, 2015.

Figure 2.79 presents the species and numbers of individuals of those species summarized in Figure 2.77 and Figure 2.78. Rodent abundance and species composition varied considerably among the different sites in 2015. The piñon mouse (*Peromyscus truei*) was the dominant species only at the Vigil piñon/juniper site, where individuals were common on the treatment plot but absent from the control plot, both in the spring and fall. The piñon mouse was uncommon at the Wester ponderosa pine site in both the spring and fall, where numbers were too low to indicate a difference between control and treatment plots. For unknown reasons the piñon mouse was rare at the Kelly pinion/juniper site in the spring and absent in the fall; piñon mice were common at the Kelly site in previous years. The deer mouse (*Peromyscus maniculatus*) was uncommon at the Chilili site, primarily on the control plot in the fall, and also uncommon at Kelly and Wester sites in the spring, but absent from the Vigil site. Except at Chillili where deer mice were most abundant on the control plot, numbers were too low elsewhere to demonstrate a difference between treatment and control plots. The white-throated woodrat (*Neotomoa albigula*) was common at the Kelly site in both the spring and fall, and consistently more abundant on the control plot than on the treatment plot. The Chilili control plot was the only other location where the white-throated woodrat was found. The white-footed mouse (*Peromyscus leucopus*) was uncommon at both the Kelly and Vigil piñon/juniper sites in the spring only, but uncommon enough not to show differences in control or treatment plots. The silky pocket mouse (*Perognathus flavus*) and the plains pocket mouse (*Perognathus flavescens*) were both rare at the Vigil treatment plot in the spring.

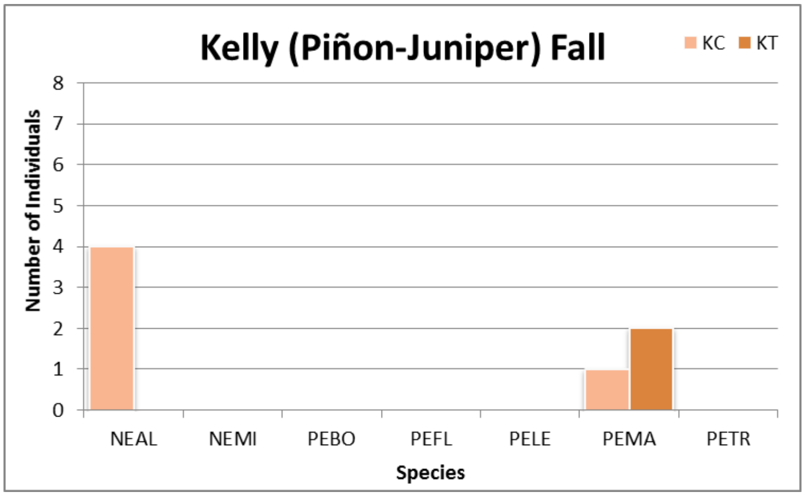


a. Chilili control and treatment plots, spring 2015.

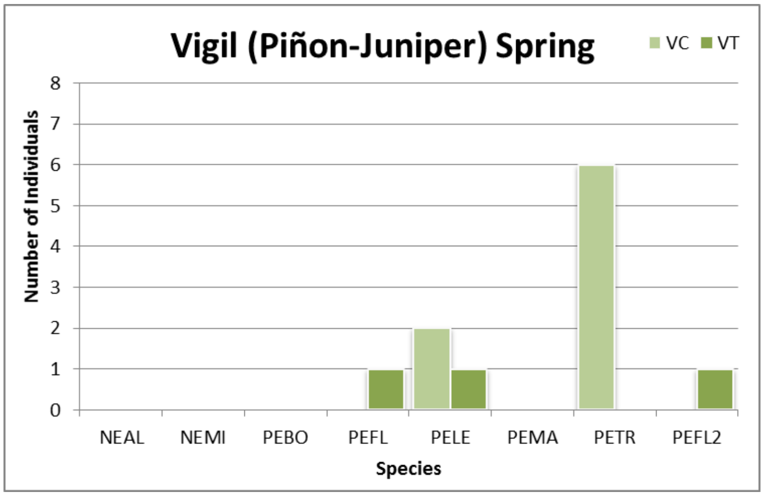
b. Chilili control and treatment plots, fall 2015.



c. Kelly control and treatment plots, spring 2015.

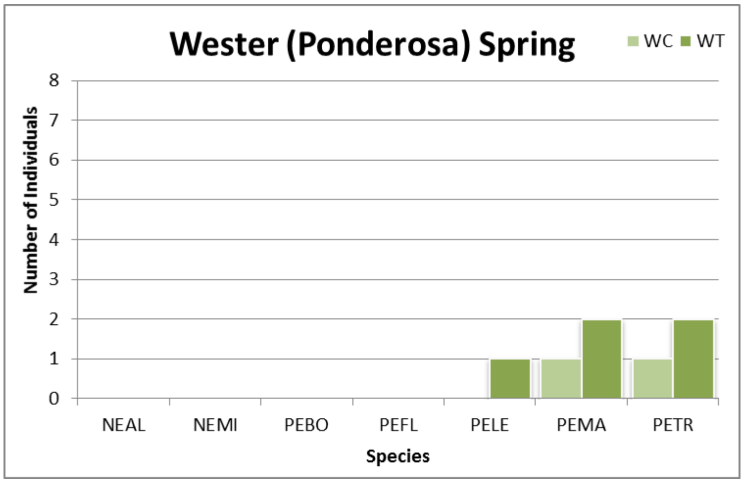


d. Kelly control and treatment plots, fall 2015.



e. Vigil control and treatment plots, spring 2015.

f. Vigil control and treatment plots, fall 2015.



g. Wester control and treatment plots, spring 2015.

h. Wester control and treatment plots, fall 2015.

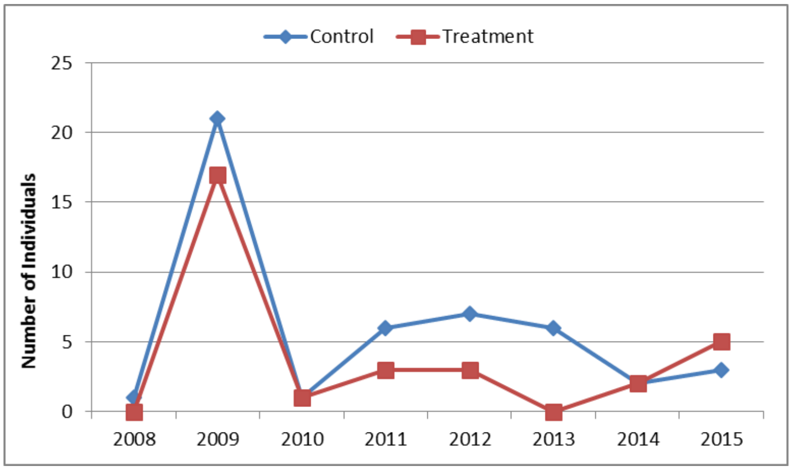
Figure 2.79. Numbers of individual rodents of each species recorded from all control and treatment study plots in 2015, both spring and fall. All rodent species found in 2015 are presented on each graph for comparative purposes, even if none were found at particular sites. Rodent species codes correspond to the following names: NEAL = *Neotoma albigula* (white-throated wood rat), NEMI = *Neotoma micropus* (Southern plains wood rat), PEBO = *Peromyscus boylii* (brush mouse), PEFL = *Perognathus flavus* (silky pocket mouse), PEFL2 = *Perognathus flavescens* (plains pocket mouse), PELE = *Peromyscus leucopus* (white-footed mouse), , PEMA = *Peromyscus maniculatus* (deer mouse), PETR = *Peromyscus truei* (piñon mouse).

Figure 2.80, a–d, presents total numbers of rodents from control and treatment plots, combined over both spring and fall, from all four study sites from fall 2008 through fall 2015. In general, overall rodent counts peaked in 2009, declined in 2010, increased slightly in 2011 and 2012, remained constant in 2013, and declined through 2014. Rodent numbers have increased on control versus treated plots at both ponderosa pine sites since tree thinning treatments in late 2010, indicating that deer mice densities have declined on treated plots at those two sites. However, in 2015, rodent numbers at the Wester site were slightly greater on the treatment plot. At the two piñon/juniper sites, the numbers of piñon mice have generally declined on the treatment plots through 2015, indicating that reductions in piñon and juniper tree densities results in declines in piñon mice as well. In 2015, overall rodent numbers were simiar or lower at the Chilili and Kelly sites, and number increased slightly at the Wester sited and on the control plots at the Vigil site. Numbers of rodent species were not high enough to perform a meaningful cluster analysis of rodent communities across the sites as was done for birds above.

a. Chilili.

b. Kelly.

c. Vigil.



d. Wester.

Figure 2.80. Total numbers of rodents from both control and treatment plots at all four study sites, 2008–2015.

2.09.3 WILDLIFE CAMERAS

Wildlife cameras (Figure 2.81) were established on the forest thinning paired control and treatment monitoring plots in February 2012 to evaluate how medium and large wildlife species are using the control versus treated study plots. The original cameras were all Leaf River IR5 infrared cameras that had a detection sensor up to 21 m (70 feet). In 2013 we replaced the Leaf River cameras (due to operational issues) with Truth Cam model 63010 cameras, which have performed very well through 2015. One camera was placed on each of the eight control and treatment study plots among the four sites, Chilili, Kelly, Vigil, and Wester. The wildlife cameras were erected near the center of each wildlife monitoring plot, approximately 1.2 m (4 feet) from the ground and oriented toward open areas free of trees up to 20 m (66 feet) away from each camera. The cameras operate during day and night using a movement sensor infrared flash. Camera photograph cards were offloaded each month.

Figure 2.81. Automatic wildlife camera (Truth Cam, model 63010).

Figure 2.82 and Figure 2.83 and present summaries of numbers of different types of animals recorded from wildlife cameras during 2015. Figure 2.82 presents all types of animals summed over all treatment and control plots from all sites. These findings show that in general, native wildlife species such as mule deer (*Odocoileus hemionus*), elk (*Cervus canadensis*), and wild Merriam's turkey (*Meleagris gallopavo*) tended to be equally frequent on control and treatment plots, while domestic livestock, both cattle and horses, were more frequent on treatment plots than control plots. Figure 2.83 shows that domestic livestock were especially abundant at the Kelly and Vigil piñon/juniper sites, but native wildlife were more abundant at most other sites. These findings indicate that native wildlife species prefer the more dense stands of trees remaining on the control plots than the more open habitats created on the treatment plots. In contrast, domestic livestock appear to prefer the more open treated plots where trees were removed and where herbaceous vegetation cover is higher. The physical structure of tree stands may be important, but also the increased growth of herbaceous vegetation on treated piñon/juniper plots appears to be more attractive to domestic livestock.

Figure 2.82. Summary of total photographs of different types of animals recorded from wildlife cameras during 2015.

Figure 2.83. Summary of types of animals recorded from control and treatment plots at each study site during 2015.

3.0 EPHEMERAL WATERSHED STREAM MONITORING

Background information on the stream piezometers can be found in the 2009 Annual Report (SWCA 2010). In addition to the paired watershed flumes, piezometers were installed on three nearby streams in order to gauge surface flows on a larger scale (Figure 3.1). The 2015 monitoring season, like previous seasons, saw very few flows; in fact in 2015 there were no recorded flows. However, large flows did occur within the Vigil and Chilili sites, however, the size of the storms again destroyed the stream piezometers at both sites. The piezometer at the Kelly site was not damaged and properly functioned throughout the entire 2015 monitoring season. The piezometers at both the Chilili and Vigil site will be relocated to a more stable stretch of channel in order to prevent washouts. Due to the damage caused at the Vigil and Chilili stream piezometers the flood, the stages could not be recorded.

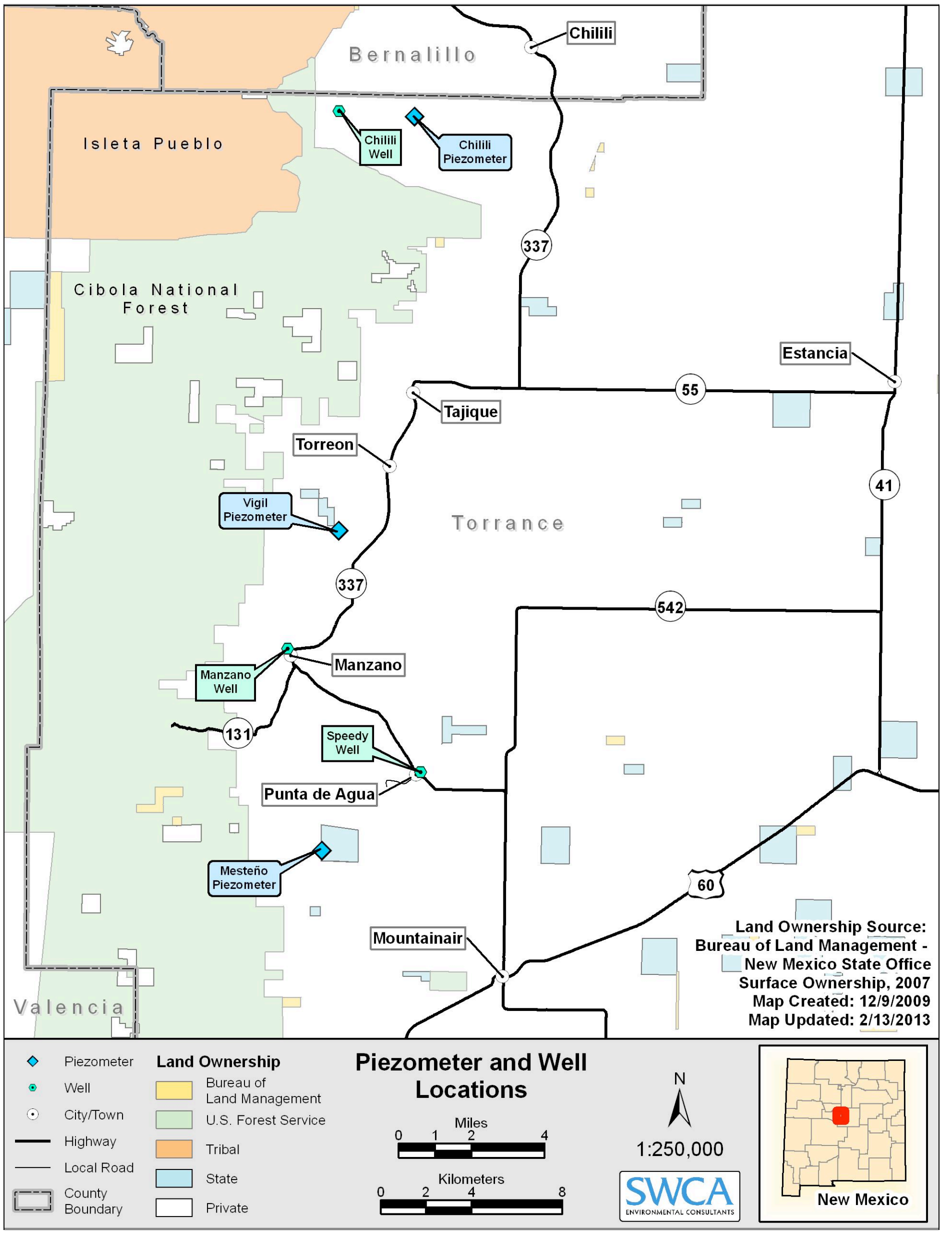


Figure 3.1. **Location of the piezometers and wells within the Estancia Basin.**

3.01 GROUNDWATER WELL MONITORING

The monitoring study is evaluating infiltration rates in the Estancia Basin by using deep pressure sensors to monitor the level of groundwater in relation to stream flow events. By monitoring the groundwater levels in private wells located close to stream monitoring locations, changes in recharge can be observed and the impact of thinning and burned areas can be compared to these groundwater levels to asses any changes.

Ideally, this project will evaluate infiltration rates in the control areas versus burned areas and relate this information to nearby groundwater levels. This could be accomplished by monitoring private wells located close to stream monitoring locations. Sandia National Laboratory and the U.S. Geological Survey are currently initiating well monitoring programs. Both entities have been receptive to sharing data when they become available, though neither knows if data would be available near the project's piezometer locations in the immediate future. The monitoring will use deep pressure sensors to monitor the level of groundwater in relation to stream flow events. If these data are available, they will be compared to the collected data from this project.

SWCA installed three well monitoring devices during early to mid-June 2009. These well monitoring locations are at Chilili, Manzano, and Punta de Agua (see Figure 3.1). Each monitoring well is equipped with Solinst Levelogger Junior pressure transducers that were programmed to record values hourly. The Chilili site is approximately 30 m (98 feet) from the western flume. The well is approximately 15 m (50 feet) deep, and depth to groundwater when installed is approximately 8 feet (25 feet). The Manzano well is shallow, approximately 8 m (25 feet) deep is dry expect for brief periods after precipitation events. The Punta de Agua well is in "downtown" Punta. The well is approximately 37 m (120 feet) deep, and depth to groundwater is approximately 28 m (91 feet) when installed. SWCA offloads data quarterly at each well location. During 2015, like the 2014 season, all wells showed a general decline throughout the year (Figure 3.2). The well at Manzano remained dry for much of the 2014 season and therefore there is no data to present.

Figure 3.2. Well data from the Chilili site, 2014-2015, showing a declining water level until the snowmelt, which generates a small rise in groundwater depth followed by another decline.

4.0 PLANNED MONITORING FOR 2016 (YEAR 9)

SWCA will continue the current monitoring efforts for year 9 of this project by monitoring the post-thinning treatment conditions in the late spring. If additional funding is awarded, monitoring will continue in fall 2016 and spring 2017. SWCA will also continue to manage the SMWS and the associated weather data if funding is awarded.

Post-wildfire monitoring has been suspended because of safety issues regarding falling dead trees. Post-fire monitoring may commence if sufficiently few dead trees remain at the monitoring sites. At this time, SWCA does not anticipate changes in the current monitoring designs or methods for forest thinning monitoring. Reporting will include regular monthly progress reports and a 2016 Annual Report.

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Appendix A.
LIST OF PLANT SPECIES ENCOUNTERED ON FOREST MONITORING STUDY PLOTS

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Appendix A. List of Plant Species Encountered on Forest Monitoring Study Plots

Group/Family	Genus	Species	Code	Common Name	Form	Life History
Gymnosperms						
Cypressaceae	*Juniperus*	*deppeana*	JUDE2	Alligator juniper	Tree	Perennial
Cypressaceae	*Juniperus*	*monosperma*	JUMO	Oneseed juniper	Tree	Perennial
Cypressaceae	*Juniperus*	*scopulorum*	JUSC2	Rocky Mountain juniper	Tree	Perennial
Pinaceae	*Pinus*	*edulis*	PIED	Piñon pine	Tree	Perennial
Pinaceae	*Pinus*	*ponderosa*	PIPO	Ponderosa pine	Tree	Perennial
Angiosperms: Dicotyledons						
Amaranthaceae	*Amaranthus*	*albus*	AMAL	Prostrate pigweed	Forb	Annual
Amaranthaceae	*Amaranthus*	*cruentus*	AMCR	Red amaranth	Forb	Annual
Amaranthaceae	*Amaranthus*	*palmeri*	AMPA	Carelessweed	Forb	Annual
Anacardiaceae	*Rhus*	*trilobata*	RHTR	Skunkbush sumac	Shrub	Perennial
Apiaceae	*Lomatium*	*dissectum*	LODI	Fernleaf biscuitroot	Forb	Perennial
Asteraceae	*Achillea*	*millefolium*	ACMI2	Common yarrow	Forb	Perennial
Asteraceae	*Ageratina*	*herbacea*	AGHE5	Fragrant snakeroot	Forb	Perennial
Asteraceae	*Anaphalis*	*margaritacea*	ANMA	Western pearly everlasting	Forb	Perennial
Asteraceae	*Antennaria*	*microphylla*	ANMI3	Littleleaf pussytoes	Forb	Perennial
Asteraceae	*Artemisia*	*carruthii*	ARCA14	Carruth’s sagewort	Forb	Perennial
Asteraceae	*Artemisia*	*dracunculus*	ARDR4	Taragon	Forb	perennial
Asteraceae	*Artemisia*	*frigida*	ARFR4	prairie sagewort	Forb	Perennial
Asteraceae	*Artemisia*	*ludoviciana*	ARLU	White sagebrush	Forb	Perennial
Asteraceae	*Aster*	*falcatus*	ASFA3	Russian milkvetch	Forb	Annual
Asteraceae	*Bahia*	*dissecta*	BADI	Ragleaf bahia	Forb	Annual
Asteraceae	*Brickellia*	*eupatorioides*	BREU	False boneset	Forb	Perennial
Asteraceae	*Brickellia*	*grandiflora*	BRGR	Tasselflower brickel	Forb	Perennial
Asteraceae	*Chaetopappa*	*ericoides*	CHER2	Rose heath	Forb	Perennial
Asteraceae	*Circium*	*undulatum*	CIUN	Wavyleaf thistle	Forb	Annual
Asteraceae	*Conyza*	*canadensis*	COCA5	Canadian horseweed	Forb	Annual
Asteraceae	*Erigeron*	*divergens*	ERDI4	Spreading fleabane	Forb	Biennial
Asteraceae	*Erigeron*	*flagellaris*	ERFL	Trailing fleabane	Forb	Biennial
Asteraceae	*Erigeron*	*formosissimus*	ERFO3	Beautiful fleabane	Forb	Perennial
Asteraceae	*Erigeron*	*speciosus*	ERSP4	Aspen fleabane	Forb	Perennial
Asteraceae	*Erigeron*	*divergens*	ERDI4	Spreading fleabane	Forb	Biennial
Brassicaceae	*Lepidium*	*alyssoides*	LEAL4	Mesa pepperwort	Forb	Perennial
Brassicaceae	*Schoenocrambe*	*linearifolia*	SCLI12	Slimleaf plainsmustard	Forb	Perennial
Brassicaceae	*Sisymbrium*	*altissimum*	SIAL2	Tall tumblemustard	Forb	Annual/Biennial
Cactaceae	*Cylindropuntia*	*imbricata*	CYIM2	Tree cholla	Succulent	Perennial
Cactaceae	*Echinocereus*	*viridiflorus*	ECVI2	Nylon hedgehog cactus	Succulent	Perennial
Cactaceae	*Escobaria*	*vivipera*	ESVI2	Spinystar cactus	Succulent	Perennial
Cactaceae	*Grusonia*	*clavata*	GRCL	Club cholla	Succulent	Perennial
Cactaceae	*Opuntia*	*engelmannii*	OPEN3	Cactus apple	Succulent	Perennial
Cactaceae	*Opuntia*	*phaeacantha*	OPPH	Tulip pricklypear	Succulent	Perennial
Cactaceae	*Opuntia*	*macrorhiza*	OPMA2	Twistspine pricklypear	Succulent	Perennial
Cactaceae	*Opuntia*	*polyacantha*	OPPO	Plains pricklypear	Succulent	Perennial

Group/Family	Genus	Species	Code	Common Name	Form	Life History
Caryophyllaceae	*Cerastium*	*brachypodum*	CEBR3	Shortstalk chickweed	Forb	Perennial
Caryophyllaceae	*Cerastium*	*nutans*	CENU2	Nodding chickweed	Forb	Annual/Perennial
Caryophyllaceae	*Pseudostellaria*	*jamesiana*	PSJA2	Tuber starwort	Forb	Perennial
Caryophyllaceae	*Silene*	*scouleri*	SISC7	Simple campion	Forb	Perennial
Chenopodiaceae	*Chenopodium*	*capitatum*	CHCA4	Blight goosefoot	Forb	Perennial
Chenopodiaceae	*Chenopodium*	*fremontii*	CHFR3	Fremont's goosefoot	Forb	Perennial
Chenopodiaceae	*Chenopodium*	*graveolens*	CHGR2	Fetid goosefoot	Forb	Annual
Chenopodiaceae	*Chenopodium*	*incanum*	CHIN2	Mealy goosefoot	Forb	Annual
Chenopodiaceae	*Chenopodium*	*leptophyllum*	CHLE4	Narrowleaf goosefoot	Forb	Annual
Chenopodiaceae	*Salsola*	*kali*	SAKA	Russian thistle	Forb	Annual
Euphorbiaceae	*Chamaesyce*	*albomarginata*	CHAL11	Whitemargin sandmat	Forb	Perennial
Euphorbiaceae	*Chamaesyce*	*chaetocalyx*	CHCHC3	Bristlecup sandmat	Forb	Perennial
Euphorbiaceae	*Chamaesyce*	*fendleri*	CHFE3	Threadstem sandmat	Forb	Perennial
Euphorbiaceae	*Chamaesyce*	*serpyllifolia*	CHSE6	Thymeleaf sandmat	Forb	Annual
Fabaceae	*Astragalus*	*mollisimus*	ASMO7	Wooly locoweed	Forb	Perennial
Fabaceae	*Astragalus*	*nuttallianus*	ASNU4	Smallflowered milkvetch	Forb	Perennial
Fabaceae	*Dalea*	*purpurea*	DAPU5	Purple prairie clove	Forb	Perennial
Fabaceae	*Hoffmannseggia*	*drepanocarpa*	HODR	Sicklepod holdback	Forb	Perennial
Fabaceae	*Lotus*	*wrightii*	LOWR	Wright's deervetch	Forb	Perennial
Fabaceae	*Lupinus*	*kingii*	LUKI	King's lupine	Forb	Perennial
Fabaceae	*Psoralidium*	*tenuiflorum*	PSTE5	Slimflower scurfpea	Forb	Perennial
Fabaceae	*Robinia*	*neomexicana*	RONE	New Mexico locust	Tree	Perennial
Fabaceae	*Vicea*	*americana*	VIAM	American vetch	Forb	Perennial
Fagaceae	*Quercus*	*gambelii*	QUGA	Gambel oak	Tree	Perennial
Fagaceae	*Quercus*	*grisea*	QUGR3	Gray oak	Tree	Perennial
Fagaceae	*Quercus*	*turbinella*	QUTU2	Sonoran scrub oak	Tree	Perennial
Geraniaceae	*Geranium*	*caespitosum*	GECAF	Fremont's geranium	Forb	Perennial
Hydrophyllaceae	*Nama*	*dichotomum*	NADI	Wishbone fiddleleaf	Forb	Annual
Lamiacea	*Agastache*	*pallidiflora*	AGPA	Bill Williams Mountain giant hyssop	Forb	Perennial
Lamiacea	*Hedeoma*	*drummondii*	HEDR	Drummond's false pen	Forb	Annual
Lamiacea	*Salvia*	*subincisa*	SASU7	Sawtooth sage	Forb	Annual
Linaceae	*Linum*	*aristatum*	LIAR3	Bristle flax	Forb	Annual
Linaceae	*Linum*	*vernale*	LIVE2	Chihuahuan flax	Forb	Annual
Malvaceae	*Spheralcea*	*angustifolia*	SPAN3	Copper globemallow	Forb	Perennial
Malvaceae	*Spheralcea*	*coccinea*	SPCO	Scarlet globemallow	Forb	Perennial
Malvaceae	*Spheralcea*	*fendleri*	SPFE	Fendler's globemallow	Forb	Perennial
Malvaceae	*Spheralcea*	*grossulariifolia*	SPGR2	Gooseberryleaf globe	Forb	Perennial
Malvaceae	*Spheralcea*	*hastulata*	SPHA	Spear globemallow	Forb	Perennial
Monotropaeae	*Monotropa*	*hypopithys*	MOHY3	Pinesap	Forb	Perennial
Nyctaginaceae	*Mirabilis*	*linearis*	MILI3	Narrowleaf four o'clock	Forb	Perennial
Nyctaginaceae	*Mirabilis*	*oxybaphoides*	MIOX	Smooth spreading four o’clock	Forb	Perennial
Oleaceae	*Menodora*	*scabra*	MESC	Rough menodora	Forb	Perennial
Onagraceae	*Oenothera*	*caespitosa*	OECA10	Tufted evening primrose	Forb	Annual
Oxalidaceae	*Oxalis*	*violacea*	OXVI	Violet woodsorrel	Forb	Perennial

Group/Family	Genus	Species	Code	Common Name	Form	Life History
Papaveraceae	*Argemone*	*squarrosa*	ARSQ	Hedgehog pricklypoppy	Forb	Perennial
Onagraceae	*Oenothera*	*caespitosa*	OECA10	Tufted evening primrose	Forb	Annual
Polemoniaceae	*Ipomopsis*	*aggregata*	IPAG	Scarlet gilia	Forb	Annual
Polygonaceae	*Eriogonum*	*alatum*	ERAL4	Winged buckwheat	Forb	Annual
Polygonaceae	*Eriogonum*	*annuum*	ERAN4	Annual buckwheat	Forb	Annual
Polygonaceae	*Eriogonum*	*microthecum*	ERMI4	Slender buckwheat	Shrub	Perennial
Polygonaceae	*Eriogonum*	*racemosum*	ERRA3	Redroot buckwheat	Forb	Perennial
Polygonaceae	*Eriogonum*	*wrightii*	ERWR	Bastardsage	Forb	Perennial
Polygonaceae	*Polygonum*	*douglasii*	PODO4	Douglas' knotweed	Forb	Annual
Portulacaceae	*Phemeranthus*	*brevicaulis*	PHBR15	Dwarf fameflower	Forb	Perennial
Portulacaceae	*Portulaca*	*oleracea*	POOL	Little hogweed	Forb	Annual
Portulacaceae	*Portulaca*	*pilosa*	POPI3	Kiss me quick	Forb	Annual
Primulaceae	*Androsace*	*septentrionalis*	ANSE4	Pygmyflower rockjasmine	Forb	Annual
Ranunculaceae	*Thalictrum*	*fendleri*	THFE	Fendler's meadow-rue	Forb	Perennial
Santalaceae	*Comandra*	*umbellata*	COUM	Bastard toadflax	Forb	Perennial
Primulaceae	*Androsace*	*septentrionalis*	ANSE4	Pygmyflower rockjasmine	Forb	Annual
Scrophulariaceae	*Castilleja*	*integra*	CAIN14	Wholeleaf Indian paintbrush	Forb	Perennial
Scrophulariaceae	*Cordylanthus*	*tenuis*	COTE3	Slender birdbeak	Forb	Annual
Scrophulariaceae	*Cordylanthus*	*wrightii*	COWR2	Wrights bird's beak	Forb	Annual
Scrophulariaceae	*Penstemon*	*barbatus*	PEBA2	Beardlip penstemon	Forb	Perennial
Scrophulariaceae	*Penstemon*	*jamesii*	PEJA	James' beardtongue	Forb	Perennial
Scrophulariaceae	*Penstemon*	*oliganthus*	PEOL	Apache beardtongue	Forb	Perennial
Scrophulariaceae	*Penstemon*	*virgatus*	PEVI4	Upright blue beardtongue	Forb	Perennial
Scrophulariaceae	*verbascum*	*thapsus*	VETH	Common mullein	Forb	Biennial
Solanaceae	*Physalis*	*hederifolia*	PHHE4	Ivyleaf groundcherry	Forb	Perennial
Solanaceae	*Solanum*	*elaeagnifolium*	SOEL	Silverleaf nightshade	Forb	Perennial
Solanaceae	*Solanum*	*triflorum*	SOTR	Cutleaf nightshade	Forb	Perennial
Verbanaceae	*Glandularia*	*bipinnatifida*	GLBIC	Davis Mountain mock vervain	Forb	Perennial
Verbanaceae	*Verbena*	*macdougalii*	VEMA	MacDougal verbena	Forb	Annual
Viscaceae	*Phoradendron*	*juniperinum*	PHJU	Juniper mistletoe	Herb	Perennial/Juniper parasite
Viscaceae	*Phoradendron*	*macrophyllum*	PHMA18	Colorado desert mist	Herb	Perennial
Angiosperms: Monocotyledons						
Agavaceae	*Yucca*	*baccada*	YUBA	Banana yucca	Succulent	Perennial
Agavaceae	*Yucca*	*glauca*	YUGL	Soapweed yucca	Succulent	Perennial
Commelinaceae	*Commelina*	*dianthifolia*	CODI4	Birdbill dayflower	Forb	Perennial
Cyperaceae	*Carex*	*geophila*	CAGE	White Mountain sedge	Sedge	Perennial
Cyperaceae	*Cyperus*	*esculentus*	CYES	Yellow nutsedge	Sedge	Perennial
Cyperaceae	*Cyperus*	*fendlerianus*	CYFE2	Fendler's flatsedge	Sedge	Perennial
Liliaceae	*Allium*	*cernuum*	ALCE2	Nodding onion	Forb	Perennial
Poaceae	*Achnatherum*	*robustum*	ACRO7	Sleepygrass	Grass	Perennial
Poaceae	*Alopecurus*	*aequalis*	ALAE	Shortawn foxtail	Grass	Perennial
Poaceae	*Andropogon*	*gerardii*	ANGE	Big bluestem	Grass	Perennial
Poaceae	*Aristida*	*adscensionis*	ARAD	Sixweeks threeawn	Grass	Annual
Poaceae	*Aristida*	*arizonica*	ARAR6	Arizona threeawn	Grass	Perennial

Group/Family	Genus	Species	Code	Common Name	Form	Life History
Poaceae	*Aristida*	*divaricata*	ARDI5	Poverty threeawn	Grass	Perennial
Poaceae	*Aristida*	*purpurea*	ARPU9	Purple threeawn	Grass	Perennial
Poaceae	*Blepharoneuron*	*tricholepsis*	BLTR	Pine dropseed	Grass	Perennial
Poaceae	*Bouteloua*	*aristidoides*	BOAR	Needle grama	Grass	Annual
Poaceae	*Bouteloua*	*curtipendula*	BOCU	Sideoats grama	Grass	Perennial
Poaceae	*Bouteloua*	*gracilis*	BOGR2	Blue grama	Grass	Perennial
Poaceae	*Bromus*	*arvensis*	BRAR5	Field brome	Grass	Annual
Poaceae	*Elymus*	*canadensis*	ELCA4	Canada wildrye	Grass	Perennial
Poaceae	*Elymus*	*elymoides*	ELEL5	Squirreltail	Grass	Perennial
Poaceae	*Elymus*	*hystrix L.*	ELHY	Eastern bottlebrush	Grass	Perennial
Poaceae	*Eragrostis*	*cilianensis*	ERCI	Stinkgrass	Grass	Annual
Poaceae	*Eragrostis*	*curvula*	ERCU2	Weeping lovegrass	Grass	Annual
Poaceae	*Eragrostis*	*mexicanus*	ERME	Mexican lovegrass	Grass	Annual
Poaceae	*Koeleria*	*macrantha*	KOMA	Prairie junegrass	Grass	Perennial
Poaceae	*Lolium*	*perenne*	LOPE	Perennial ryegrass	Grass	Annual
Poaceae	*Lycurus*	*phleoides*	LYPH	Common wolfstail	Grass	Perennial
Poaceae	*Lycurus*	*setosus*	LYSE3	Bristly wolfstail	Grass	Perennial
Poaceae	*Monroa*	*squarrosa*	MOSQ	False buffalograss	Grass	Annual
Poaceae	*Muhlenbergia*	*minutissima*	MUMI2	Annual muhly	Grass	Annual
Poaceae	*Muhlenbergia*	*montana*	MUMO	Mountain muhly	Grass	Perennial
Poaceae	*Muhlenbergia*	*thurberi*	MUTH	Thurber's muhly	Grass	Perennial
Poaceae	*Muhlenbergia*	*torreyi*	MUTO2	Ring muhly	Grass	Perennial
Poaceae	*Muhlenbergia*	*richardsonii*	MURI	Mat muhly	Grass	Perennial
Poaceae	*Panicum*	*capillare*	PACA6	Witchgrass	Grass	Annual
Poaceae	*Pascopyrum*	*smithii*	PASM	Western wheatgrass	Grass	Perennial
Poaceae	*Piptatherum*	*micranthum*	PIMI7	Littleseed ricegrass	Grass	Perennial
Poaceae	*Pleuraphis*	*jamesii*	PLJA	James' galleta	Grass	Perennial
Poaceae	*Poa*	*fendleriana*	POFE	Muttongrass	Grass	Perennial
Poaceae	*Setaria*	*viridis*	SEVI4	Green bristlegrass	Grass	Annual
Poaceae	*Sporobolus*	*cryptandrus*	SPCR	Sand dropseed	Grass	Perennial
Poaceae	*Thinopyrum*	*ponticum*	THPO7	Tall wheatgrass	Grass	Perennial
Non-Vascular Plants						
–	multiple	multiple	MOSS	Moss	Crypt	Perennial
–	multiple	multiple	CRUST	Cryptobiotic crust	Crypt	Perennial

Taxonomy and names follow the U.S. Department of Agriculture (2014) PLANTS Database.

Appendix B.

REPEAT PHOTO POINT PHOTOGRAPHS

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Chilili Control 2008

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Chilili Control 2010

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Chilil Control 2011

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Chilili Treatment 2008

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Chilili Treatment 2010

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Kelly Control 2008

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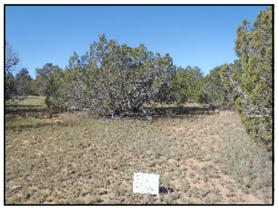

Kelly Treatment 2008

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Kelly Treatment 2011

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Vigil Control 2008

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Wester Control 2008

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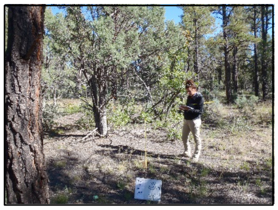

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Attachments

CD with an electronic .pdf version of the report