ESTANCIA BASIN WATERSHED HEALTH AND MONITORING PROJECT: 2013 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)

CLAUNCH-PINTO SOIL AND WATER CONSERVATION DISTRICT
Fiscal Agent
P.O. Box 129
Mountainair, New Mexico 87036
Telephone: 505.847.2243

Prepared by

David Lightfoot, Ph.D.
Cody Stropki, Ph.D.
Victoria Amato, M.S.
Chris Garrett, M.S.
Anne Russell, B.S.

SWCA ENVIRONMENTAL CONSULTANTS
5647 Jefferson Street NE
Albuquerque, New Mexico 87109
Telephone: 505.254.1115; Facsimile: 505.254.1116
www.swca.com

March 2014

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 plan calls for 2 years of pre-thinning data to provide background information on all study sites prior to implementing thinning treatments and monitoring treatment effectiveness. Results from the 2008, 2009, 2010,2011, and 2012 monitoring seasons are presented in the 2008, 2009, 2010, 2011, and 2012 annual reports, which can also be found on the New Mexico Forest and Watershed Restoration Institute's website.

The principal goals of forest and watershed monitoring are to determine the effectiveness of standard prescribed forest thinning on soils, hydrology, water yield and quality, 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 2013 Annual Report provides information on the results of forest thinning during the calendar year 2013. Summaries of weather data from the South Mountain Weather Station, which serve as a baseline for monitoring area climate data, are also provided. Initial 2008, 2009, and 2010 baseline pre-treatment monitoring data from permanent monitoring study sites provide 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 2013 represent information on the above parameters along with data on medium- to large-sized wildlife and livestock for the first 3 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 2015 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 third year of post-treatment monitoring data revealed similar differences in parameter values between treatment and control plots to those that occurred last year, 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.
- Tree basal areas were reduced on the treatment plots according to New Mexico State Forestry guidelines: Chilili pre-treatment basal area was 210 square feet/acre and was reduced to 80 square feet/acre, Wester basal area was 220 square feet/acre pre-treatment and 99 square feet/acre post-treatment, Kelly basal area was 155 square feet/acre pre-treatment and 47 square feet/acre post-treatment, and Vigil basal area was 124 square feet/acre pre-treatment and 39 square feet/acre post-treatment.
- Tree diameter size class measurements of all treatment and control plots showed that control plots had on average a larger number of trees in the smaller diameter classes when compared to treatment plots where trees were more evenly distributed between diameter size classes.
- During the 2012 monitoring period, relatively few rainfall events generated surface runoff events basin-wide. In fact flows only occurred at the Vigil watersheds where the treated watersheds had higher peak flows and runoff ratios when compared to the controls. Future monitoring of flow events will reveal if this increased runoff on thinned sites persists and for how long.
- Soil moisture was higher on treated plots than control plots, especially during dry periods following rainfall events.
- Herbaceous vegetation canopy cover was significantly higher on both piñon/juniper treated plots and at one ponderosa pine treated plot compared to the control plots, and cover was significantly higher on the treatment plot at one ponderosa pine site, but there was no significant difference at the other ponderosa pine site. Herbaceous vegetation cover was probably even higher than measured on treated plots, because domestic livestock grazed the treated plots prior to vegetation measurements. Despite the drought, herbaceous vegetation grew considerably in the region following heavy July and early September rains.

- Bird densities and species richness were found to be higher on treated plots than control plots, especially at piñon/juniper sites. Bird species composition was more similar among pairs of control and treatment plots at each site and forest type (i.e., by location) during the spring breeding season, but more similar among treatment plots and control plots (i.e., by treatment type) during the fall migration time. These results indicate that the thinning treatments are not affecting bird communities during the breeding season, but that more birds are attracted to the treated plots during fall migration.
- Rodent densities remained relatively low in 2013, and were lower on most of the treatment plots than on control plots. Piñon mice dominated at the piñon/juniper sites and at one ponderosa site; deer mice were most common at the other ponderosa pine site. Piñon mice were especially less abundant on treated plots at the piñon/juniper sites, while deer mice were present only on treated plots at the piñon/juniper sites.
- Native large animals recorded from remote wildlife cameras such as mule deer, elk, and rabbits were most abundant on control plots, 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 bird communities have not yet shown differences between treatment and control plots.

Given that 2013, like 2012 and 2011, was an extreme drought year in the region, some parameter responses may have been dampened by a lack of rainfall.

This page left intentionally blank

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

This page left intentionally blank

1.0 INTRODUCTION

This 2013 Annual Report provides summaries of monitoring data collected during the 2013 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, and 2012 summarize overall monitoring findings from those five 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 to 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 seven years of monitoring, beginning in 2008, 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 three years of baseline pre-thinning treatment monitoring (2008–2010), thinning treatments implemented during the winter of 2010 and 2011, and four years of post-treatment monitoring (2011–2015).

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 (note that the SMWS is located north of Edgewood, New Mexico, and is not on the map presented in Figure 1.1, but is on the map presented as Figure 4.1 in Chapter 4).

This 2013 Annual Report is similar in format to the previous 2008, 2009, 2010, 2011, and 2012 annual reports, and it provides complete data files (appended on DVD) and summaries of findings from field monitoring measurements conducted during the calendar year 2013 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; and 3) SMWS weather and soil moisture data collection, including addenda representing the four quarterly 2013 reports. 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 will then provide measures of thinning treatment effectiveness and a comparison of post-treatment environmental conditions. Monitoring data from subsequent years will provide data on thinning treatment effects over time.

This report provides analyses of parameter changes over the six 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 five-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 2013 Annual Report provides summaries of findings from field monitoring measurements conducted during the calendar year 2013 and compares them with previous years for the above-mentioned projects and subprojects. This report is partitioned into different sections for each subproject: 1) introduction (this section), 2) forest thinning monitoring, 3) ephemeral stream and groundwater monitoring, 4) SMWS data, and 5) planned monitoring for 2014 (year seven).

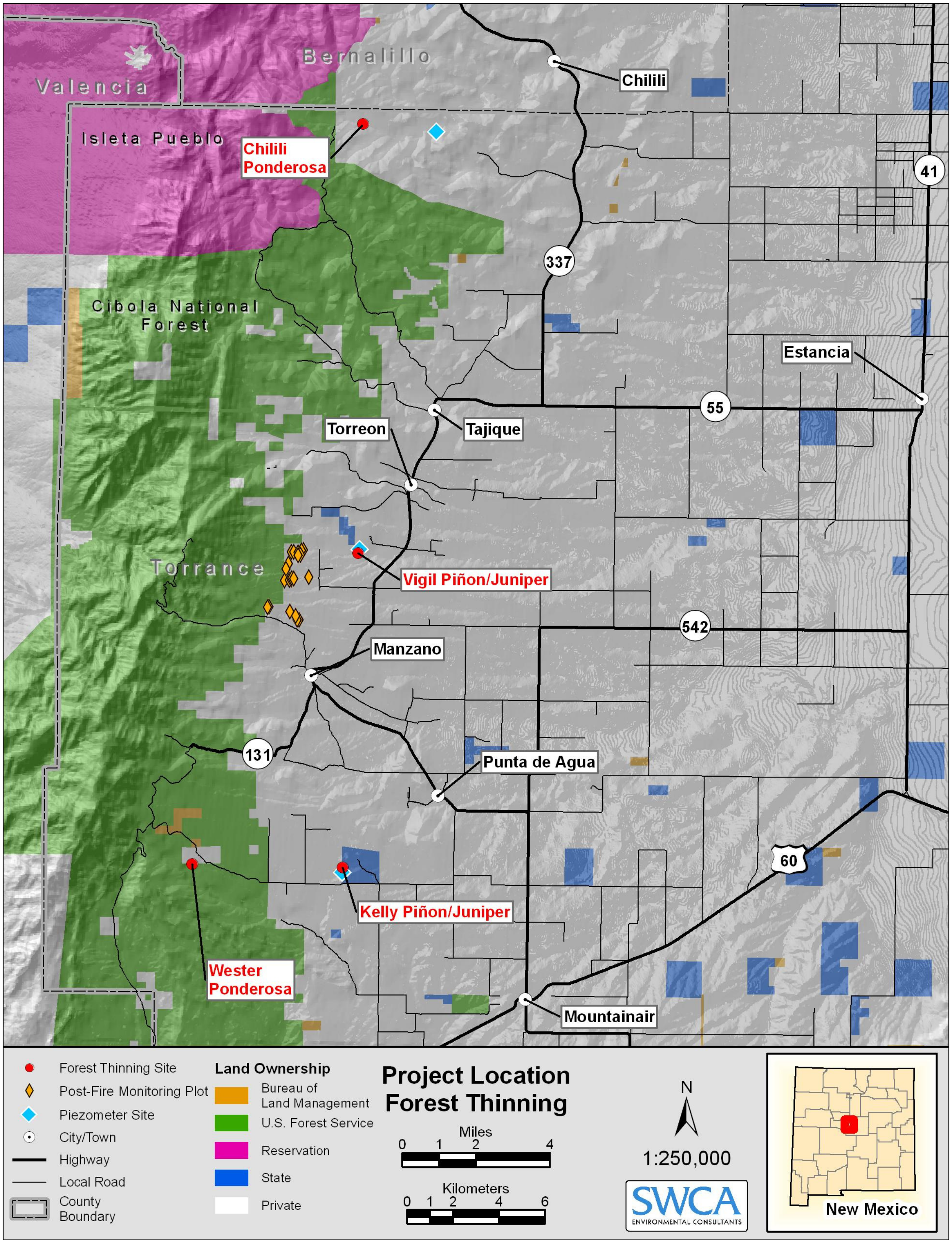


Figure 1.1. **Map of all Estancia Basin forest and watershed monitoring locations addressed in this report.**

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.

2.1 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.

2.1.1 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 Rio Grande Region of New Mexico (Colorado border to Truth or Consequences, NM) has increased from 1971-2012 by 2.5º F, and in mountainous areas that increase has been even greater at 2.7º F (Llewellyn and Vaddey 2013). 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). Gutzler (2013) and Llewellyn and Vaddey (2013) attribute this climate change 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). 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 [IPCC] 2007; Tebaldi et al. 2006). Petrie et al. (2014) demonstrated 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 and Robbins 2011; Gutzler 2013, Llewellyn and Vaddey 2013) are expected to result in significantly diminished stream flow and drier surface conditions (Seager et al. 2008, Llewellyn and Vaddey 2013), causing the Southwest 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; Negron et al. 2009; Raffa et al. 2008; van Mantgem et al. 2009) and that better understanding of climate change is needed 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 hypothesize that climate change will heavily impact these forest types since they are not well adapted to long-term drought conditions. They observed considerable tree mortality across almost 100% of their plots, attributed primarily to insect attack resulting from drought and tree water stress.

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). 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 and white fir 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 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. (2013) and Savage et al. (1996) demonstrated 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 1994). 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. 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.

2.1.2 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 there are many questions that remain on the subject (Harrison et al. 2011; Johnson et al. 2012; Morford et al. 2011; Powers et al. 2005). 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.

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. Maintaining soil surface litter cover is the easiest way to prevent accelerated erosion (Moghaddas 2013). Page-Dumroese et al. (2000) 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 suggested 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) studied disturbance and erosion on intermediate (10% to 25%) and steep (26% to 43%) slopes in a thinned New Mexico mixed-conifer forest. They concluded 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 (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 2006). 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.

2.1.3 HYDROLOGY

Numerous studies worldwide have demonstrated that changing forest density can in turn change forest water yield (Bosch and Hewlett 1982; Brown et al. 1974; DeBano et al. 2004; Douglas 1983; Gary 1975; Harr 1983; Hibbert 1967; Hornbeck et al. 1997; Kattleman and Ice 2004; Keppeler and Zeimer 1990; National Research Council [NRC] 2008; Reinhart et al. 1964; 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). 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 centimeters (cm) a year, with an average increase in water yield of 10.0 cm 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).

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 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) suggested 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.

2.1.4 TIMBER

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).

Fule 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 ha–1), compared to control units (6.2 trees ha–1). 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

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 hectares (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).

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. 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; 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 impact 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; 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, in thinned or thinned and then burned stands compared to unthinned stands (Feeney et al. 1998; Kolb et al. 1998; Wallin et al. 2004).

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 one-seed 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 (ppt), relative to other treatments (i.e., irrigation to produce 125% of ambient annual ppt; a control; and ambient ppt). 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 as indicated by higher Cp 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.

Piñon and juniper in the region are also infected with dwarf mistletoe (*Arceuthobium divaricatum*) (Figure 2.2) on pines and true mistletoes 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.

2.1.5 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; Fule et al. 2001; Laughlin and Fulé 2008; Moore et al. 2006; Stoddard et al. 2011; Stoddard and McGlone 2008; Wienk et al. 2004).

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). 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.

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 (in press) 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 (in press) 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; Fule et al. 2001).

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; 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.

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.

2.1.6 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 et al. 2004; Saab and Powell 2005). 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.

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 benefit from small-diameter tree removal, but they have negative responses to overstory removal. Tree foliage and bole foraging birds responded neutrally or positively to the small-diameter tree removal but negatively to overstory removal. Woodpeckers, however, declined following overstory removals. 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). 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, 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.

Hurteau et al. (2008) reported an increase in western bluebird densities as a result of increasing foraging opportunities (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 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.

2.1.7 RODENTS

The effectiveness of thinning to either promote or maintain habitat has been demonstrated for a number of forest-floor specialists and generalists (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. 2006a).

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. 2006c; Lee et al.

2008). This understory response results in 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 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.

Northern Arizona University Ecological Restoration Institute (NAU ERI) looked at time since treatment effects in ponderosa pine on mammal density and reported time since treatment was an important variable for four key species (2010). 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, a burrowing species. The presence of downed wood or slash is important for some species (Chambers 2002; Converse et al. 2006a), particularly deer mice, 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). The NAU study 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. 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).

2.1.8 WILDFIRE

In many watersheds throughout the Southwest, over 90% 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 1994; Fulé et al. 1997). Wildfires, which burn at a high severity, have the ability to disrupt, damage, and destroy ecosystem functioning through the consumption of the understory and overstory vegetative cover and the protective litter and duff layers. The damaging effect of wildfires on ecosystem functioning, in particular vegetative response, has been well documented throughout the literature (Abella et al. 2012; Brown et al. 2000; Cambell et al. 1977; Ffolliott et al. 2008; Neary et al. 2008). Many studies have assessed the relationship between native and exotic species response following fire (Abella et al. 2012; Fornwalt et al. 2010; Hunter et al. 2006; Keeley et al. 2003; Stohlgren et al. 1999), reporting various relationships between native and exotic species richness and cover. Abella et al. (2012) write that based on a synthesis of literature a general postulate is that native plant species richness and cover are positively correlated with exotic richness and cover. In their Arizona ponderosa pine study, they also found that environmental variables (specifically soil total nitrogen and water-holding capacity) were more strongly related than disturbance to exotic species richness and cover (Abella et al. 2012).

Research has found the loss of surface cover following a high-intensity wildfire greatly increases erosion rates as well as surface runoff and subsequent sediment transportation (Benavides-Solorio and MacDonald 2001; DeBano et al. 1998; Moody and Martin 2001; Neary et al. 2008; Rich 1962). The high incidence of exposed bare soil and low tree canopy cover following high-intensity wildfire also increases the potential for non-native species invasions (Hunter et al. 006; Wolfson et al. 2005).

To mitigate fire effects on ecosystem resources, high-severity burn areas typically undergo post-wildfire rehabilitation treatments. Pyke et al. (2002) suggested that in the absence of intensive post-fire rehabilitation of native species, non-native species will outcompete many native plants, increasing fire risk and changing the age structure of the wildland area. The methods most commonly used in rehabilitation efforts include the application of mulch, contour felling of trees, and aerial seeding (Robichaud et al. 2000). The latter of these techniques, aerial seeding, is the most commonly used method by land managers due to the ease of application, relative low cost compared to the other techniques, and ability to limit non-native species invasion, and is mandated by U.S. federal polices when economical (Beyers 2004; Peppin et al. 2010; Stella et al. 2010). Species typically used in the seeding process are non-native 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 et al. 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. Several recent studies have addressed the effectiveness of post-wildfire seeding (Barclay et al. 2004; Peppin et al. 2010; Stella et al. 2010).

Peppin et al. (2010) conducted a systematic review of the literature on post-wildfire seeding in the western United States in order to answer a number of questions on the effectiveness of aerial seeding. The questions posed in this review were: does aerial seeding reduce erosion or reduce non-native plant cover, and how does seeding affect native plant communities? From this comprehensive review, which looked at over 30 publications, the researchers concluded that the ability for seeding to reduce erosion and non-native establishment is still inconclusive. The review also yielded that aerial seeding following a wildfire generally decreases native cover within the first few growing seasons, but there are not much long-term data to assess how this structure changes through time.

2.2 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 2013 report presents the third 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.

2.3 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 submeter 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.

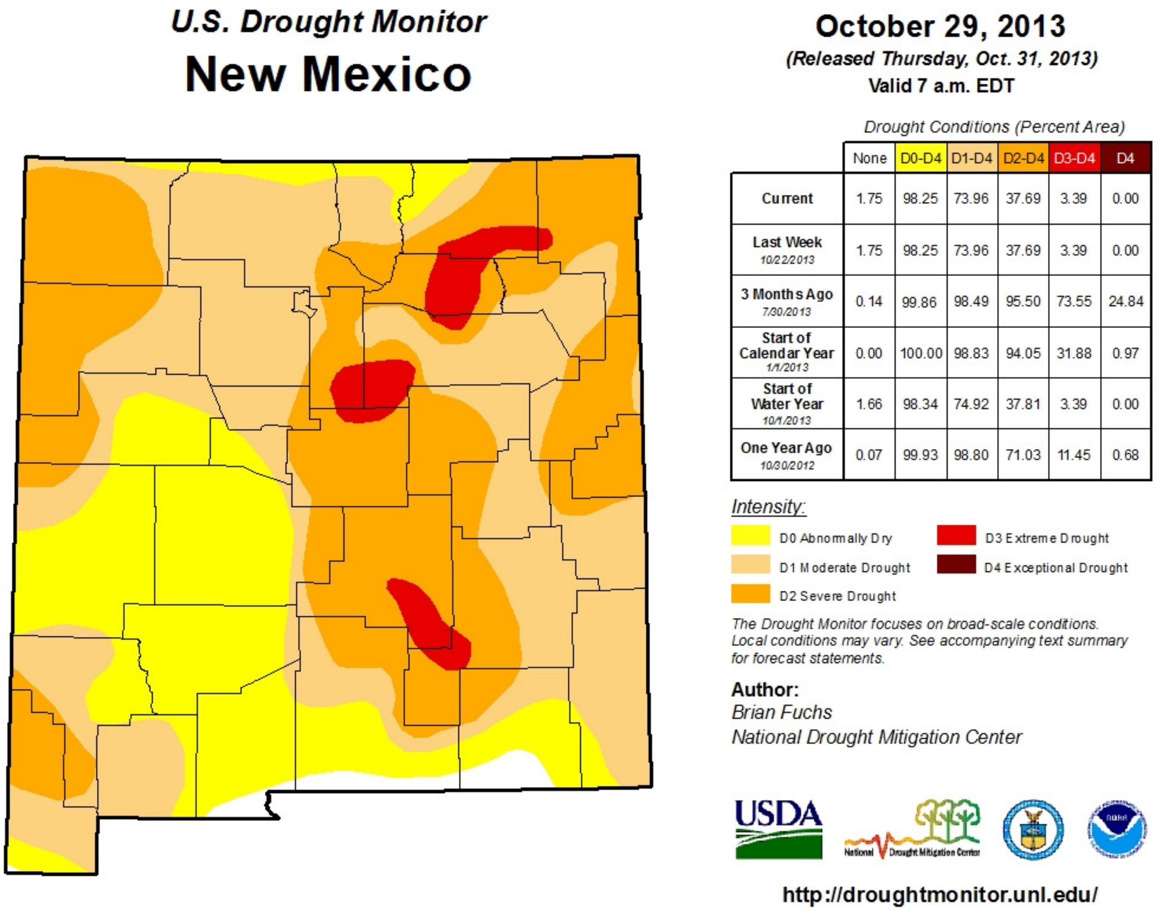
2.4 RAINFALL AND TEMPERATURES

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 2013 on all thinning plots are presented as examples below. Also presented below are long-term graphs of each of these variables (2009–2013) showing any trends that may be occurring climatically within the region.

During the 2013 monitoring period the drought that occurred in 2011 and 2012 still persisted throughout the state of New Mexico, particularly over the project area (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.

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.



U.S. Drought Monitor

New Mexico

October 29, 2013

(Released Thursday, Oct. 31, 2013)

Valid 7 a.m. EDT

Drought Conditions (Percent Area)

	None	D0-D4	D1-D4	D2-D4	D3-D4	D4
Current	1.75	98.25	73.96	37.69	3.39	0.00
Last Week 10/22/2013	1.75	98.25	73.96	37.69	3.39	0.00
3 Months Ago 7/30/2013	0.14	99.86	98.49	95.50	73.55	24.84
Start of Calendar Year 1/1/2013	0.00	100.00	98.83	94.05	31.88	0.97
Start of Water Year 10/1/2013	1.66	98.34	74.92	37.81	3.39	0.00
One Year Ago 10/30/2012	0.07	99.93	98.80	71.03	11.45	0.68

Intensity:

- D0 Abnormally Dry
- D1 Moderate Drought
- D2 Severe Drought
- D3 Extreme Drought
- D4 Exceptional 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/

Figure 2.9. Drought monitor map of New Mexico from the week of October 29, 2013, showing the project area located within Torrance County experiencing a severe drought (U.S. Drought Monitor 2013).

2.4.1 PRECIPITATION

Hourly precipitation totals have been summed to monthly totals, and there are similar monthly precipitation totals for the paired study plots at the Kelly piñon/juniper 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 2009–2013 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 14.4 inches (Western Regional Climate Center 2013) and a slight increase in precipitation values in 2013 at both ponderosa pine and piñon/juniper sites. The total precipitation received in 2013 was 18.15 inches at the piñon/juniper sites and 12.34 inches at the ponderosa pine sites, showing the ponderosa pine sites remain below average. 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 2013).

All tipping bucket rain gauges were functioning properly during the 2013 monitoring season.

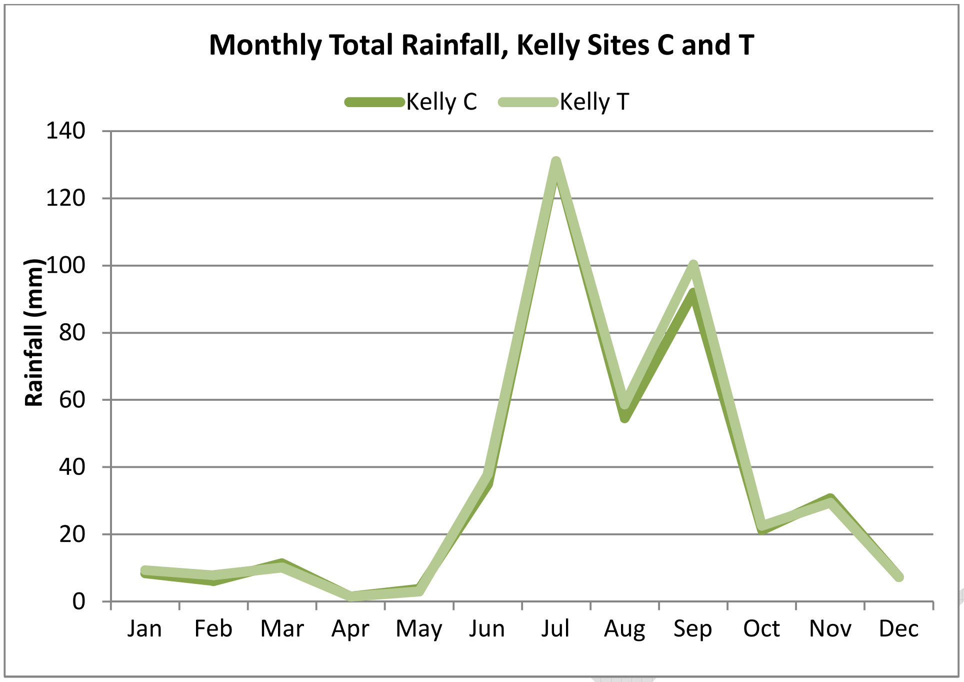


Figure 2.10. **Monthly cumulative precipitation (rainfall and snow) from the two paired Kelly piñon/juniper study plots in 2013.**

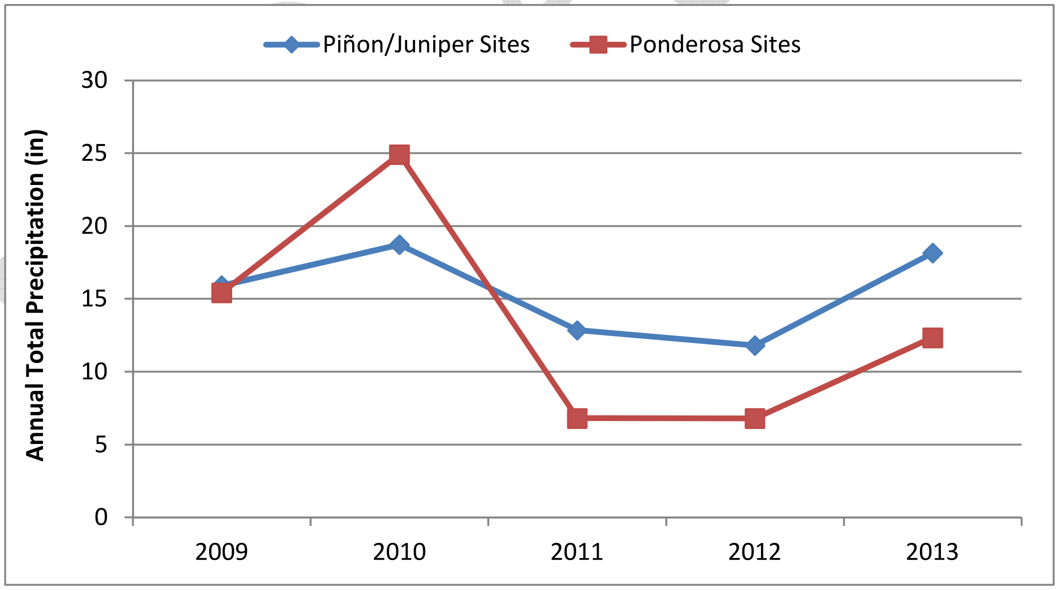


Figure 2.11. **Annual precipitation values from 2009–2013 on the piñon/juniper and ponderosa pine sites, which highlight the drought in 2011–2012 and increased precipitation in 2013.**

2.4.2 AMBIENT TEMPERATURE

An example of monthly averages of hourly ambient temperatures is presented for the Kelly piñon/juniper 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 2008–2013, which had a steady rise in temperature from 2008 to 2012 and then a decrease in average temperature from 2012 to 2013 for the piñon/juniper sites and a leveling off of temperature for the ponderosa pine sites (Figure 2.13). The average temperature at the piñon/juniper (10.2°C [50.36°F]) site was well above the long-term average for the area, while the ponderosa pine sites (9.7°C [49.46°F]), which are higher in elevation, had average temperatures that were equal to 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 2014). The temperature increase seen in this study is also what has been occurring statewide and even at a national scale.

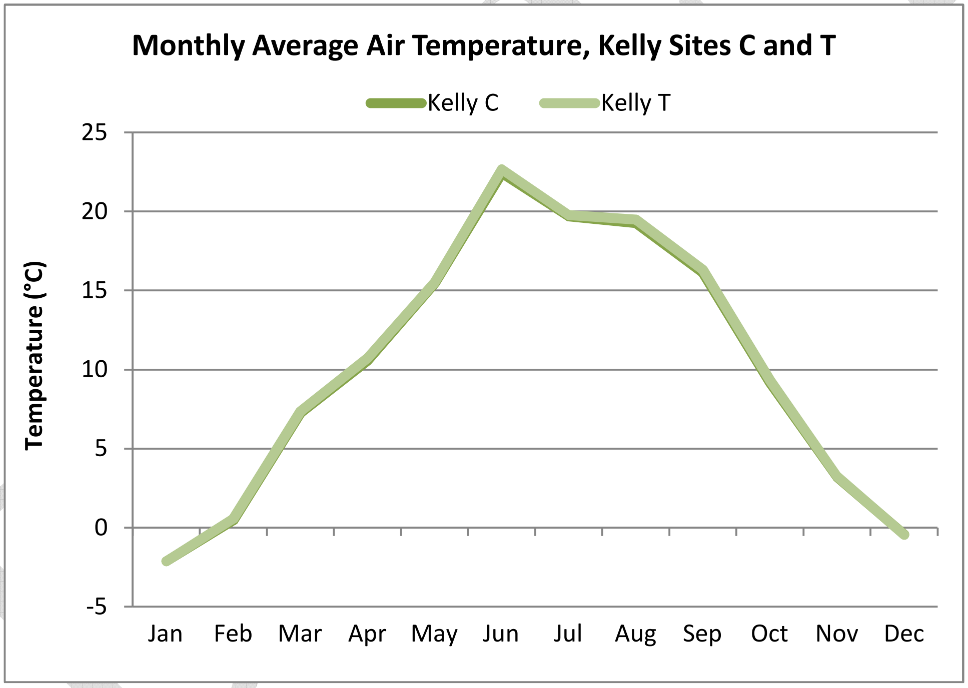


Figure 2.12. Monthly average ambient temperatures from the two paired Kelly piñon/juniper study plots in 2013.

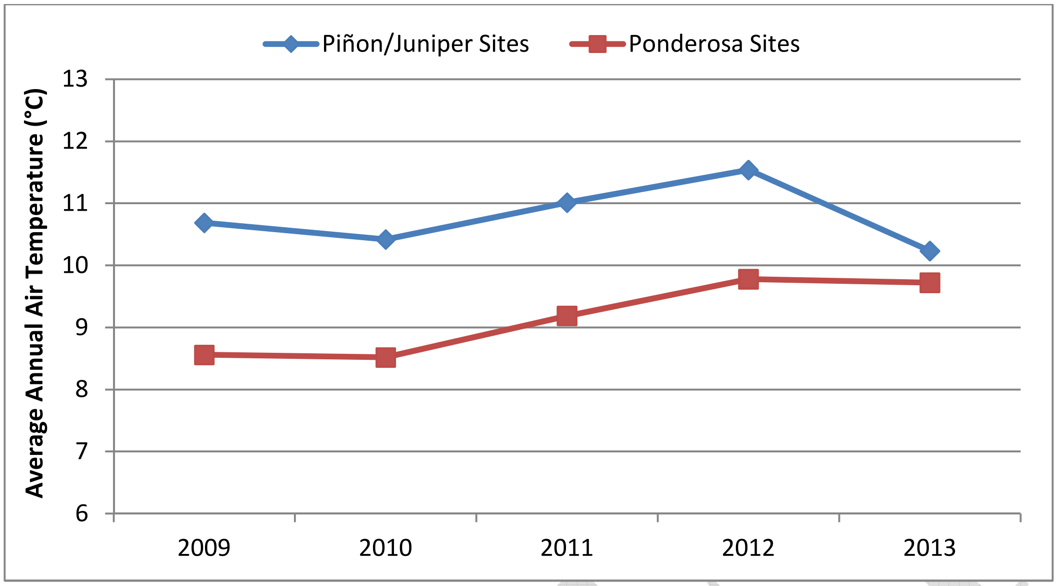


Figure 2.13. Annual average ambient temperature values at the piñon/juniper and ponderosa pine sites, 2009–2013.

2.4.3 SOIL MOISTURE

An example of monthly averages of hourly –10 cm soil moisture readings are presented for the paired study plots at the Kelly piñon/juniper 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.

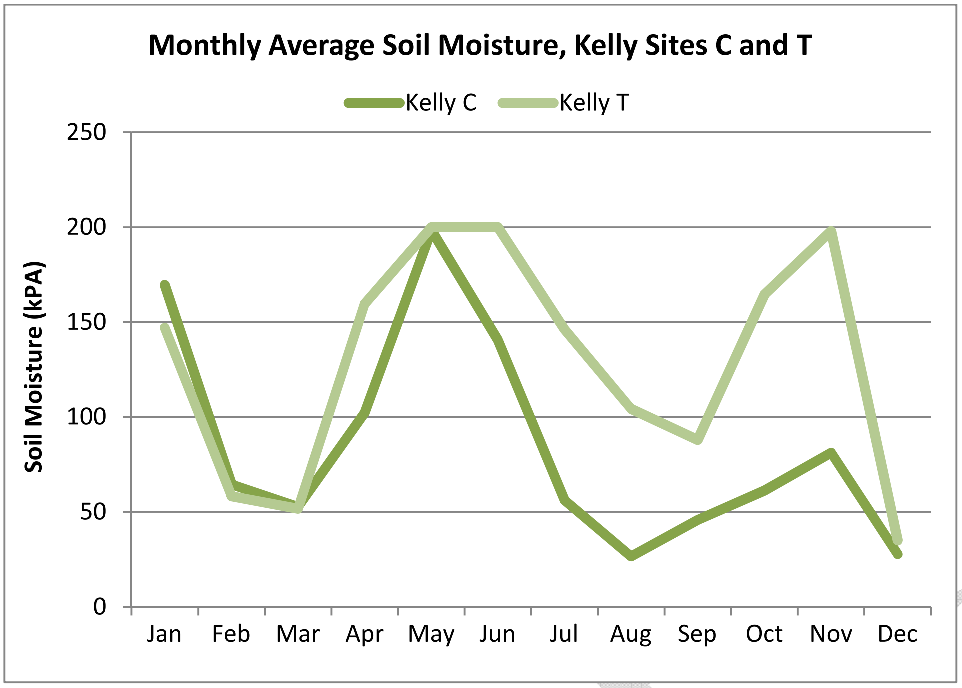


Figure 2.14. Monthly average soil moisture tensions (–10 cm) from the two paired Kelly piñon/juniper study plots in 2013.

2.4.4 SOIL TEMPERATURE

An example of monthly averages of hourly –10 cm soil temperature readings are presented for the paired study plots at the Kelly piñon/juniper 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.

A figure showing the annual soil temperatures for both the ponderosa and piñon/juniper sites is also presented below (Figure 2.16).

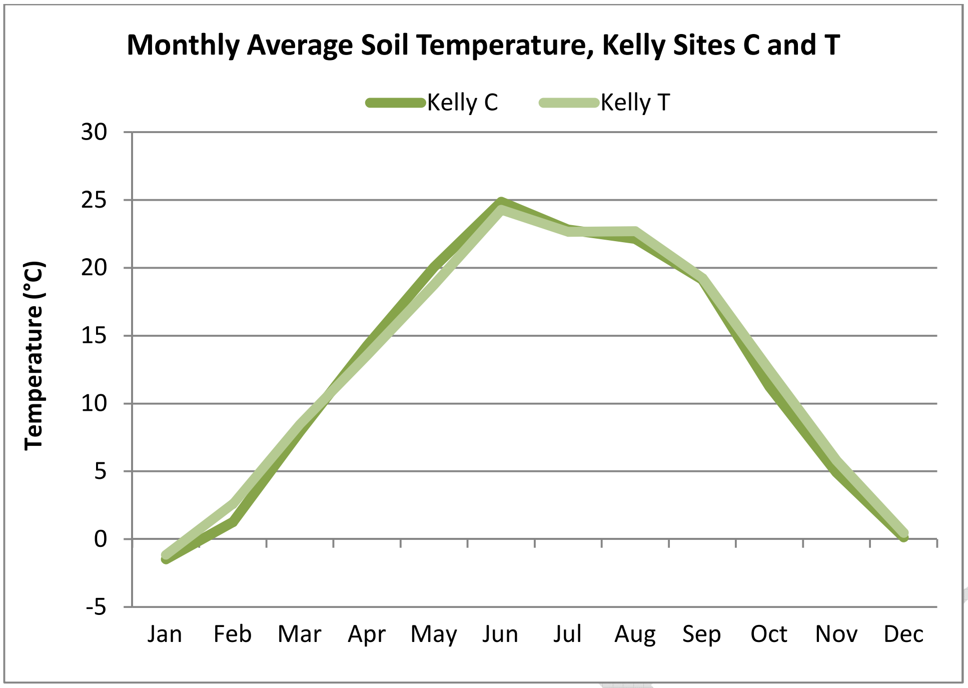


Figure 2.15. **Monthly average soil temperature (-10 cm) from the two paired Kelly piñon/juniper study plots in 2013.**

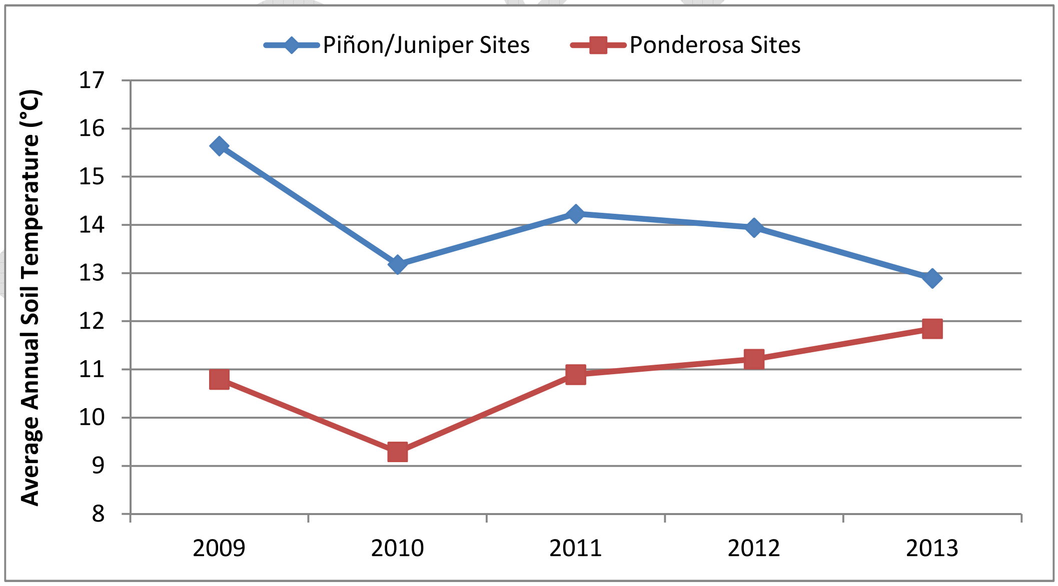


Figure 2.16. **Annual average soil temperature values at the piñon/juniper and ponderosa pine sites, 2009–2013.**

2.5 SOILS

2.5.1 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 2013 is displayed below in Figure 2.17 and Figure 2.18. 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 2008–2013 from all forest thinning plots (Figure 2.19–Figure 2.22). 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. These figures also show the difference in soil moisture between the treatment and control is more pronounced on the piñon/juniper plots than the ponderosa pine plots. This finding can likely be contributed to the decrease in canopy cover and the increase in ground cover in the form of wood chips. This pattern has been evident for both 2012 and 2013 monitoring periods, whether these findings continue to persist into the future remains to be seen.

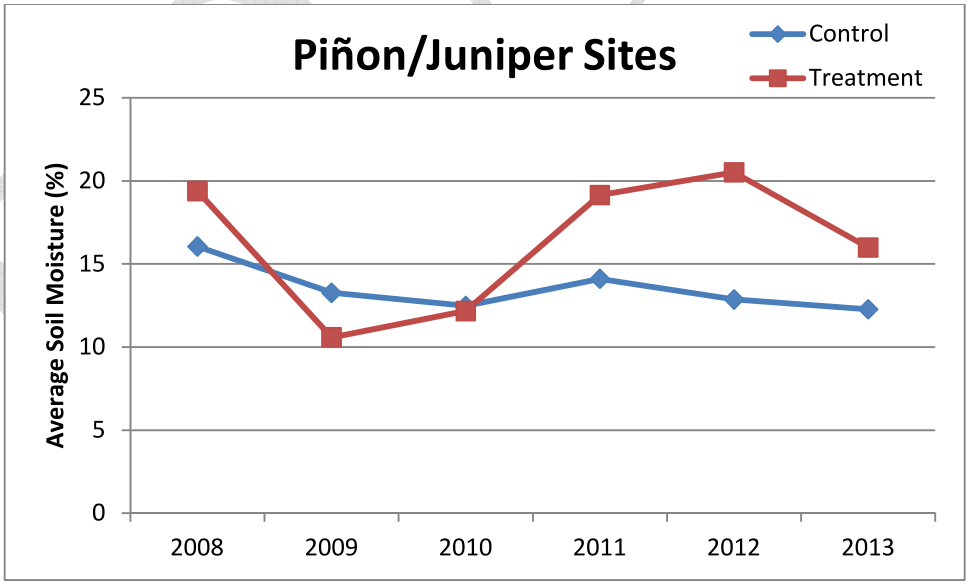


Figure 2.17. Annual average soil moisture percentage for the piñon/juniper sites, 2008–2013; moisture readings were averaged annually from the monthly readings.

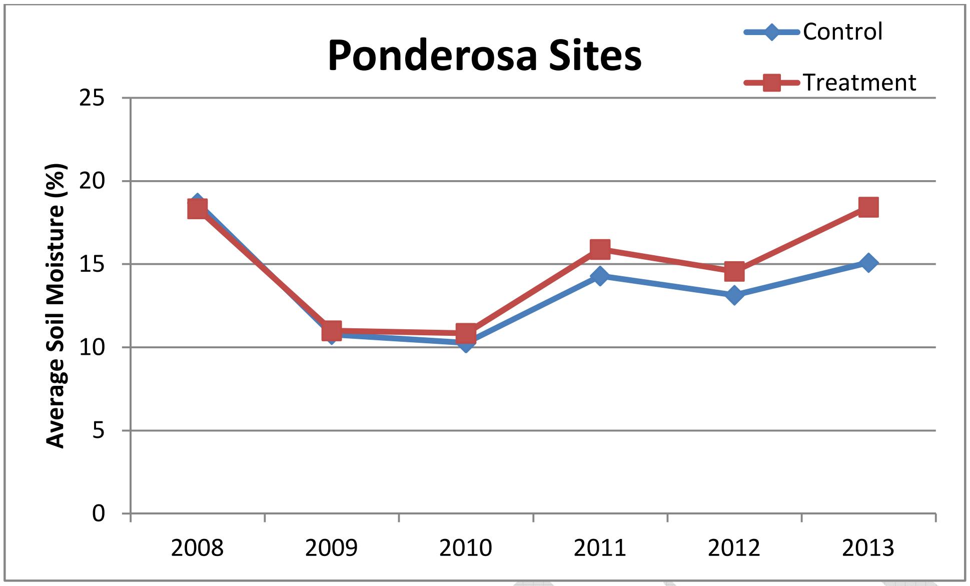


Figure 2.18. Annual average soil moisture percentage for the ponderosa sites, 2008-2013; moisture readings were averaged annually from the monthly readings.

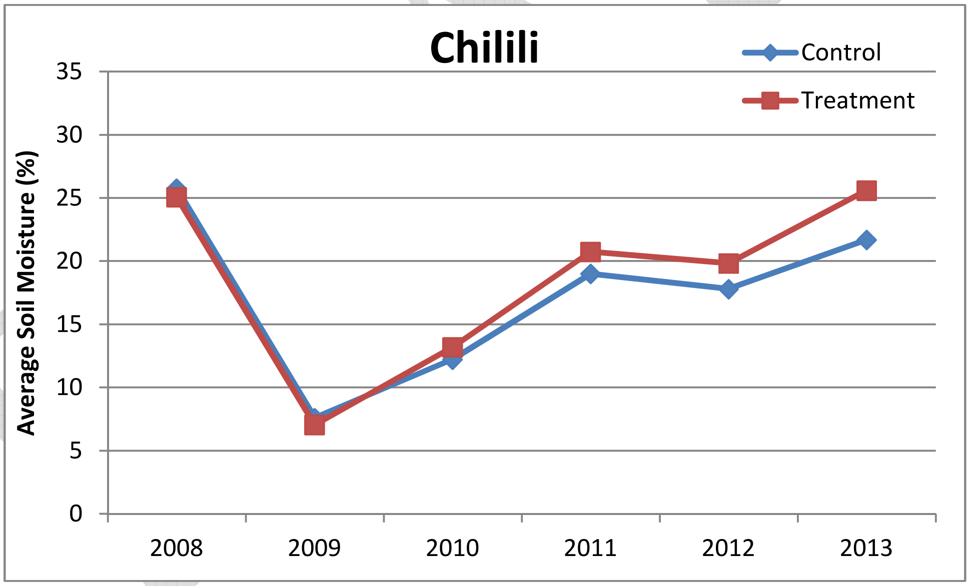


Figure 2.19. Average annual soil moisture readings taken at the Chilili site, 2008–2013.

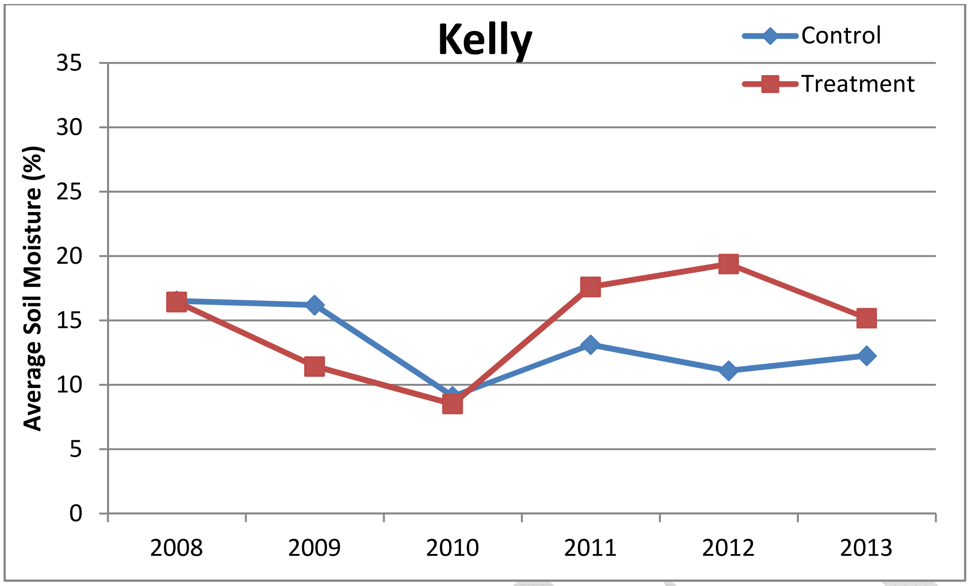


Figure 2.20. **Average annual soil moisture readings taken at the Kelly site, 2008–2013.**

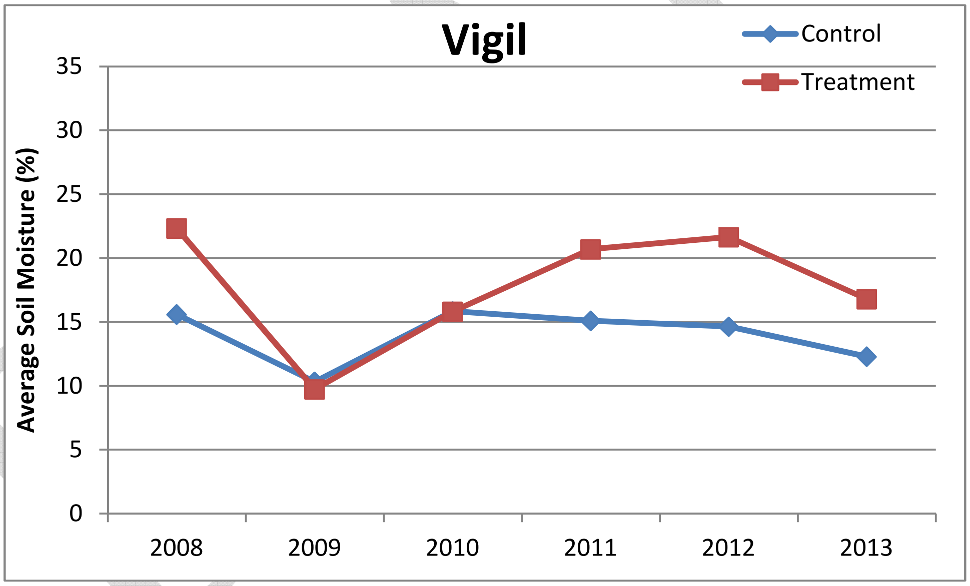


Figure 2.21. **Average annual soil moisture readings taken at the Vigil site, 2008–2013.**

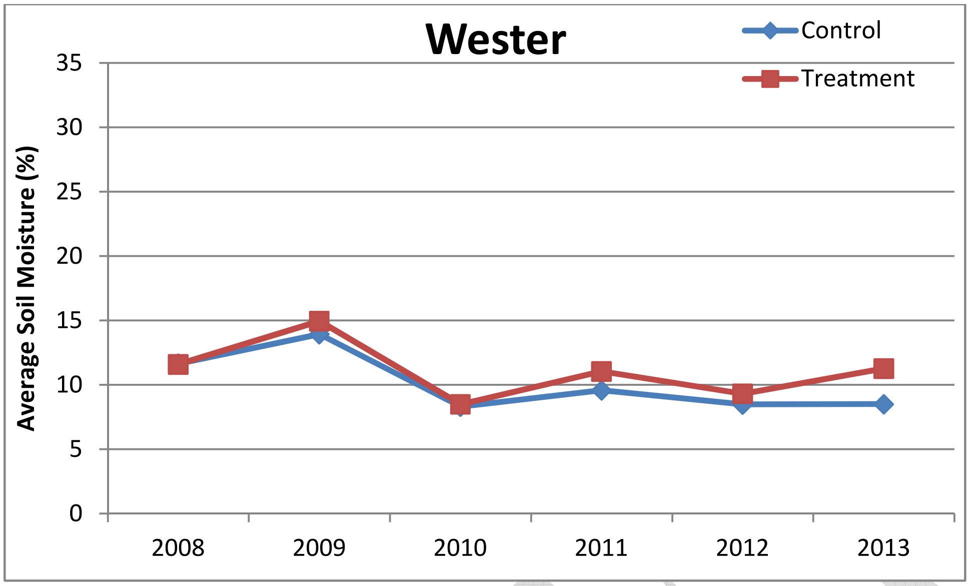


Figure 2.22. Average annual soil moisture readings taken at the Wester site, 2008–2013.

2.5.2 SOIL SURFACE STABILITY

Soil surface stability was not measured in 2013. Since only small changes in surface stability are detected on an annual basis, it was decided that this measurement would be transitioned to a 3-year monitoring cycle and so will be measured again in the spring of 2015.

2.5.3 SOIL MOVEMENT

Soil movement was monitored using soil movement bridges (called soil erosion bridges in the 2008 report) (Figure 2.23) 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 2.24 shows the micro-soil topography profile from one of the three sampling points at the Kelly piñon/juniper site for 2008–2013. The graph clearly shows the yearly variability associated with soil movement on a plot and a slight trend for overall soil loss over the Five-year period. Figure 2.5 through Figure 2.28 show average soil profile values averaged over all points per bridge, and over three bridges per paired plot, for 2008, 2009, 2010, 2011, 2012, and 2013. These figures show little overall change in average soil surface levels over that five-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 five 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 two years of post-treatment monitoring, no differences have been observed between the control and treatment. However, one factor to take into consideration is the lack of large precipitation events. For the past two years, the project area has been in a severe drought that has resulted in very few overland flow events that typically move large amounts of sediment occurring. 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 2.23. Measurement of soil surface topography using a soil movement bridge helps understand the yearly variability associated with soil topography.

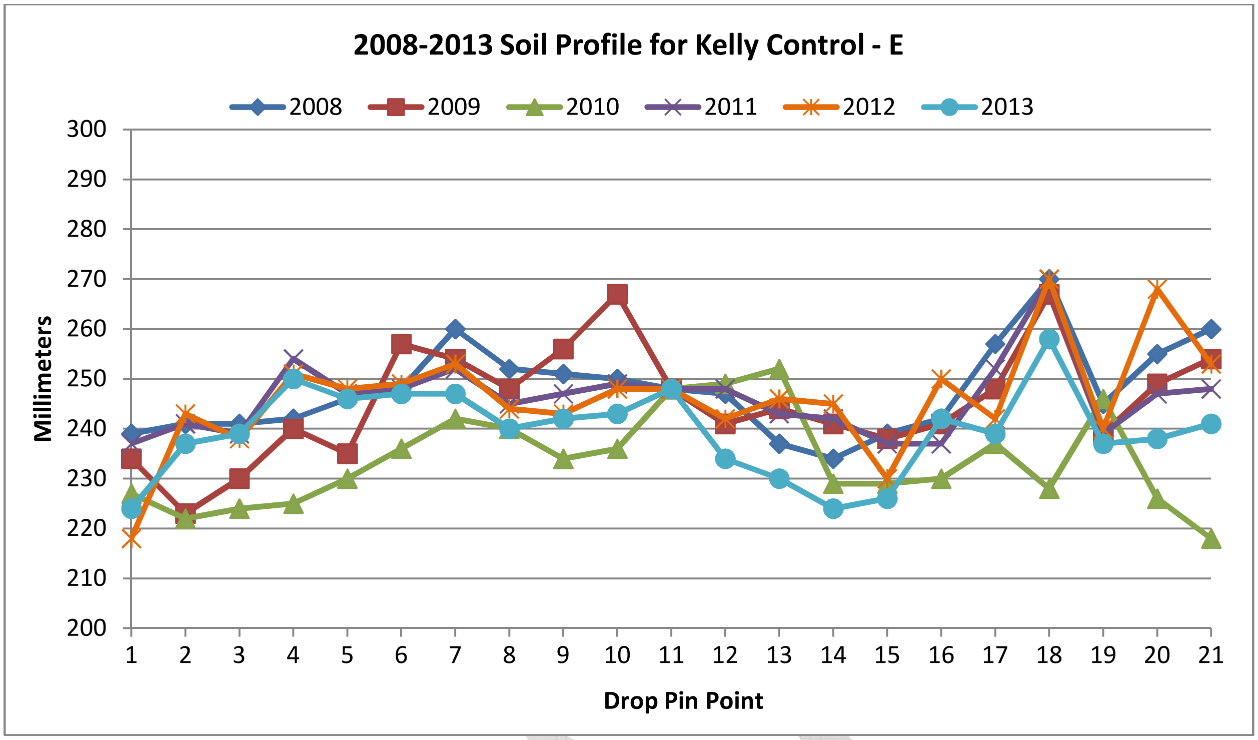


Figure 2.24. Soil surface profile from the soil movement bridge located at the Kelly piñon/juniper control site over 2008–2013, showing variation in the soil surface profile over a five-year period. Each point 1–21 on the x axis represents one measurement point from the soil surface to the level bridge above the surface. Point 11 is the set point (head of a spike) for calibration.

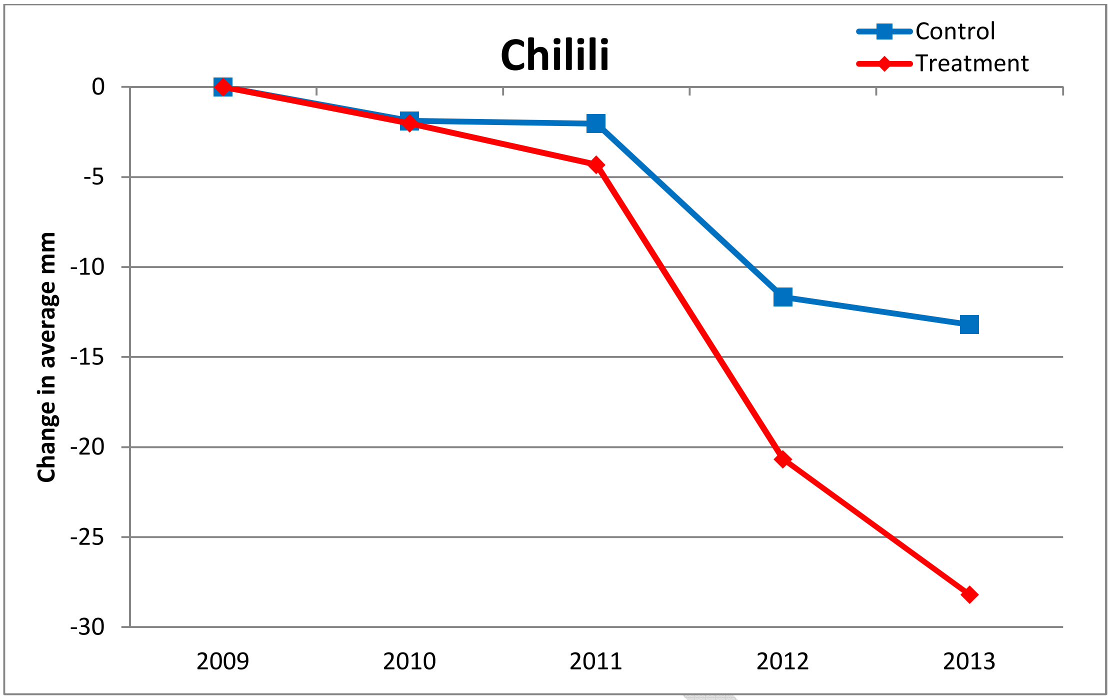


Figure 2.25. **Average soil surface profiles for the Chilili sites, averaged from three soil movement bridges located on each of the paired study plots over the three-year period, 2009–2013.**

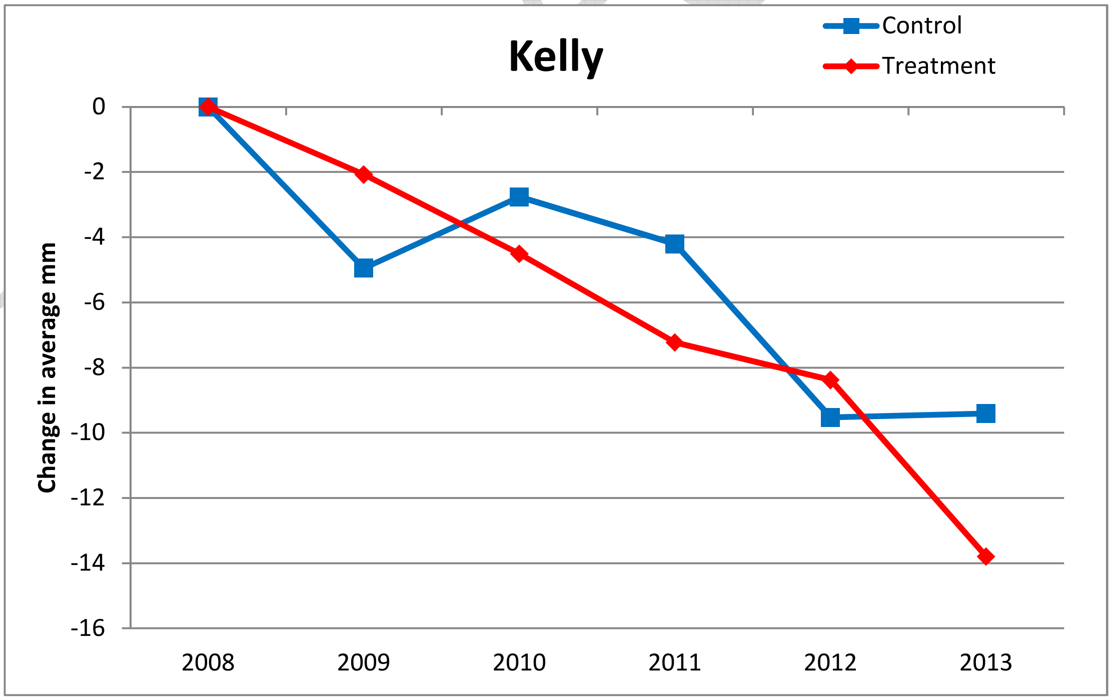


Figure 2.26. **Average soil surface profiles for the Kelly sites, averaged from three soil movement bridges located on each of the paired study plots over the five-year period, 2008–2013.**

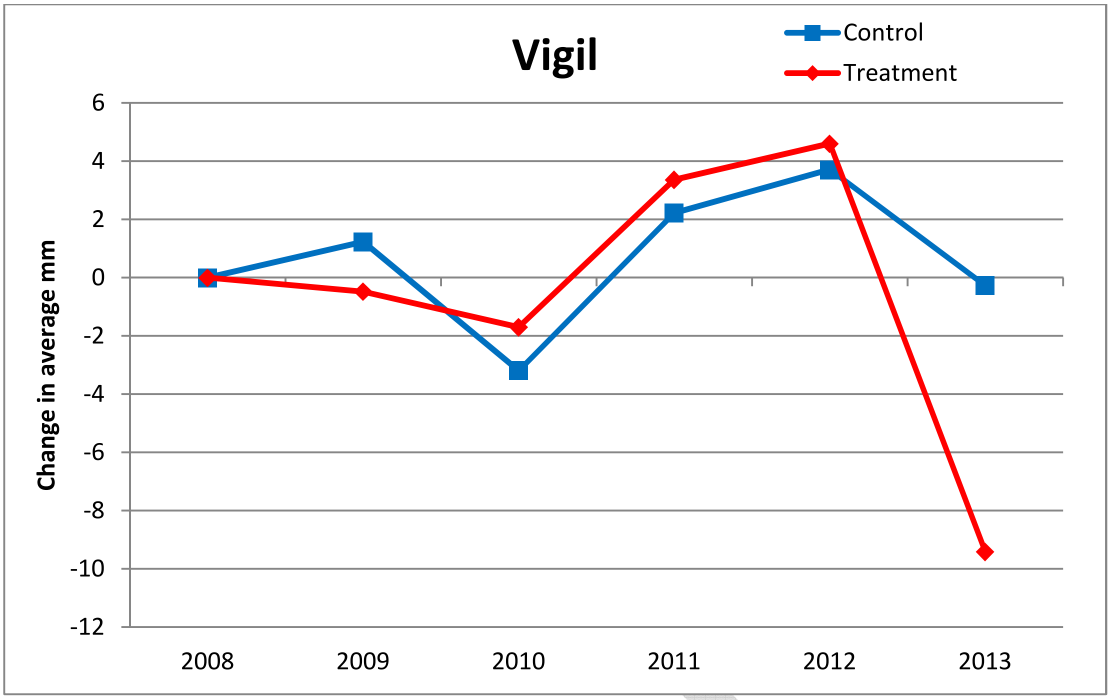


Figure 2.27. **Average soil surface profiles for the Vigil sites, averaged from three soil movement bridges located on each of the paired study plots over the five-year period, 2008–2013.**

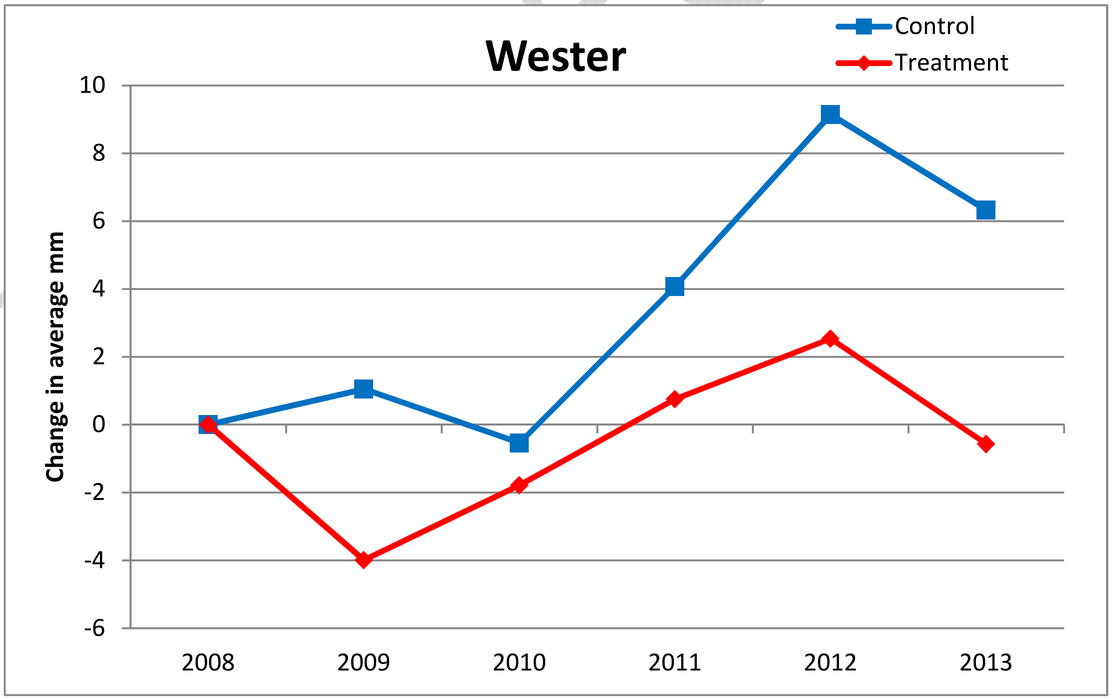


Figure 2.28. **Average soil surface profiles for the Wester sites, averaged from three soil movement bridges located on each of the paired study plots over the five-year period, 2008–2013.**

2.5.4 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 2.29). 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, 2010, 2011, 2012, and 2013, 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 and 2013 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 2.29. 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 2008–2013 are presented in Figure 2.30 through Figure 2.37.

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.

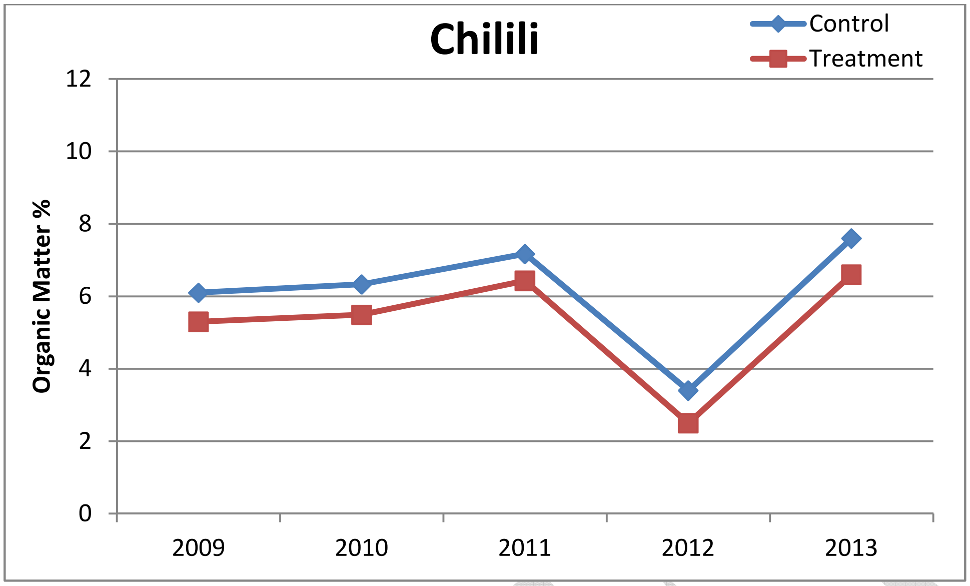


Figure 2.30. Organic matter concentrations measured at the Chilili sites, 2009–2013.

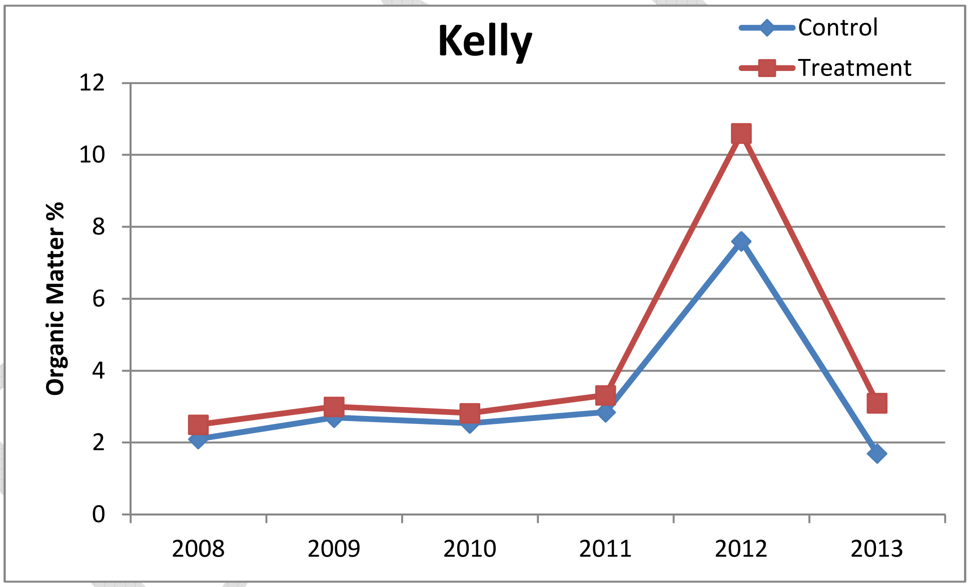


Figure 2.31. Organic matter concentrations measured at the Kelly sites, 2008–2013.

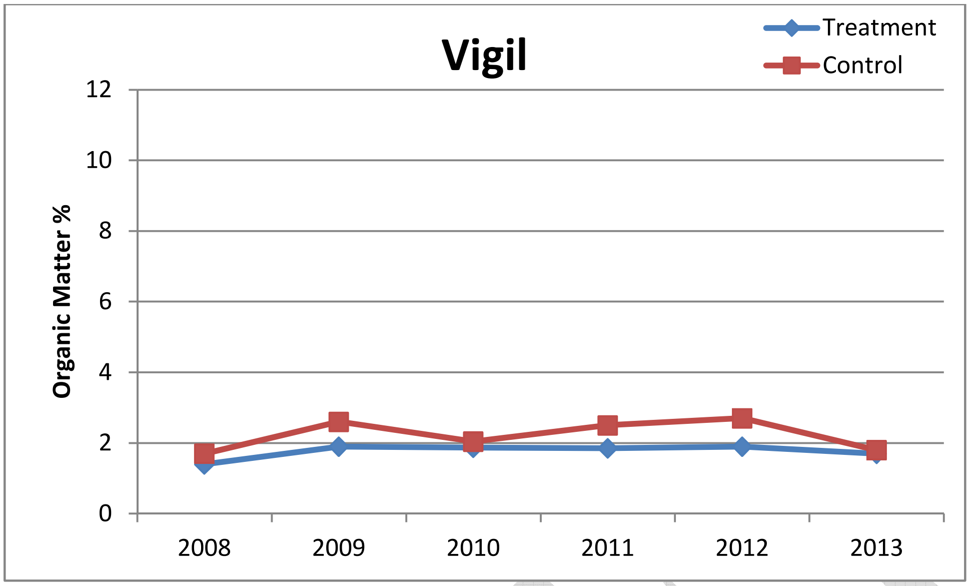


Figure 2.32. **Organic matter concentrations measured at the Vigil sites, 2008–2013.**

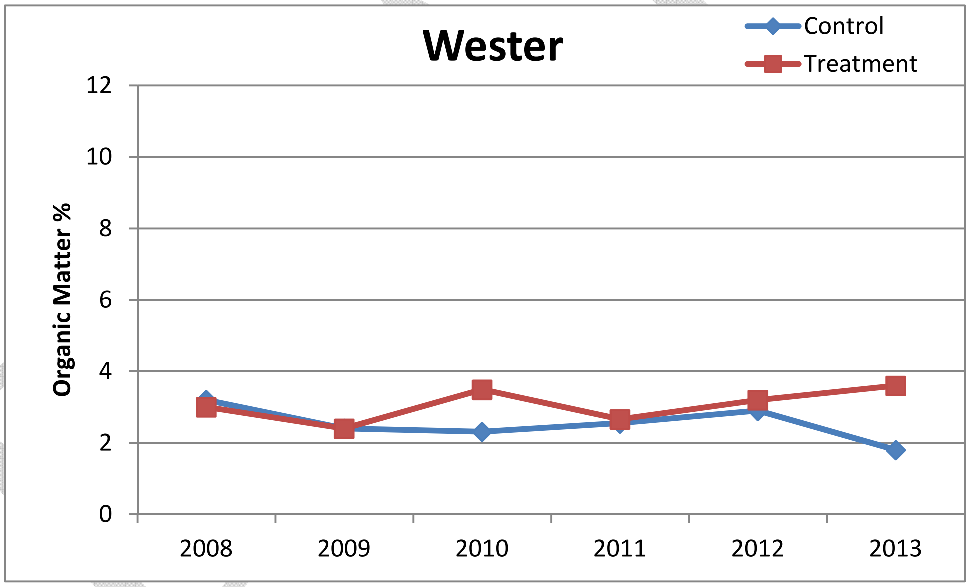


Figure 2.33. **Organic matter concentrations measured at the Wester sites, 2008–2013.**

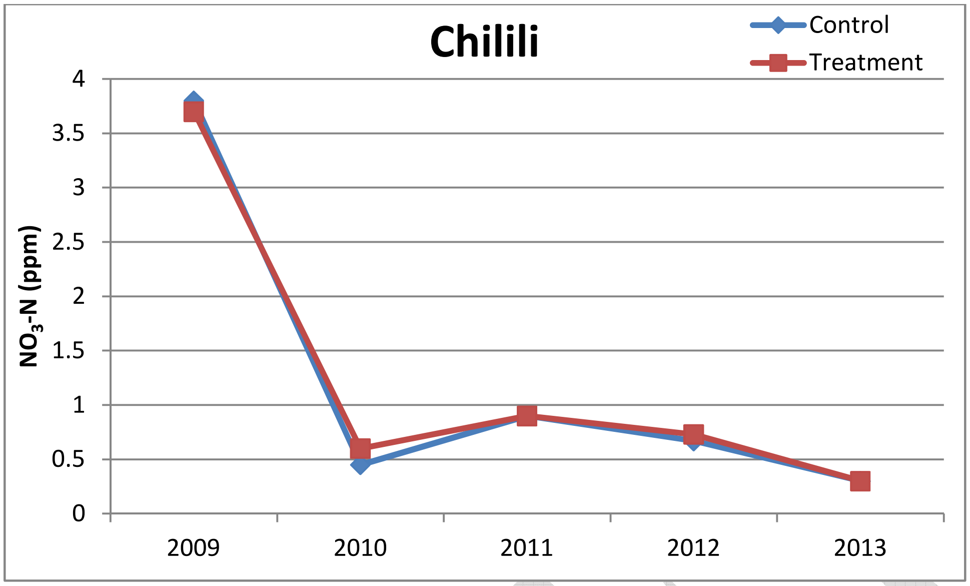


Figure 2.34. Nitrate concentrations measured at the Chilili sites, 2009–2013.

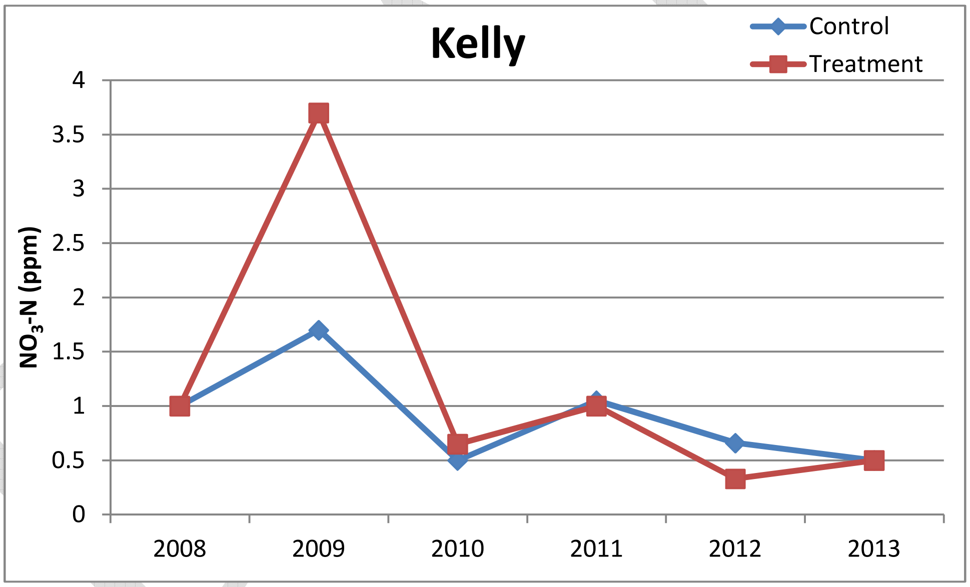


Figure 2.35. Nitrate concentrations measured at the Kelly sites, 2008–2013.

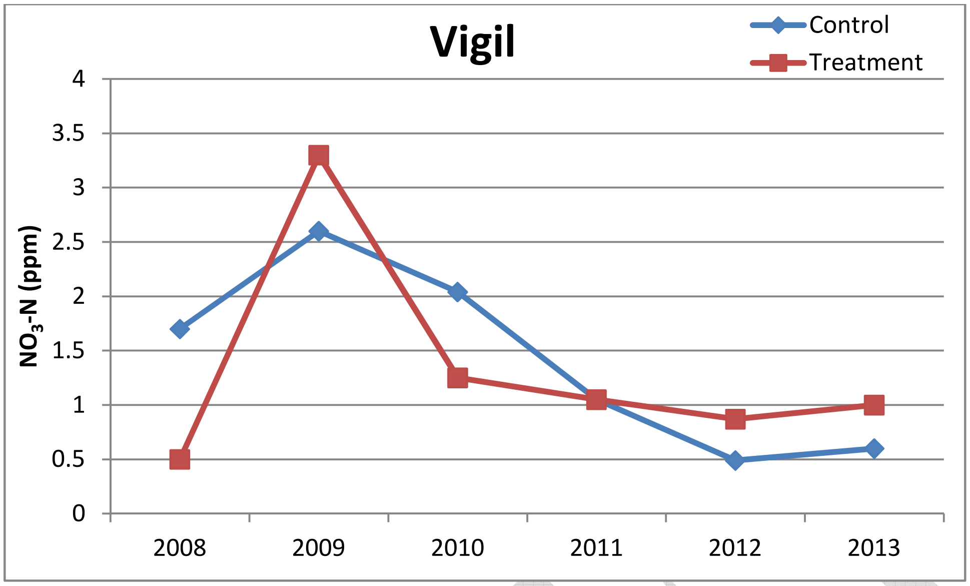


Figure 2.36. Nitrate concentrations measured at the Vigil sites, 2008–2013.

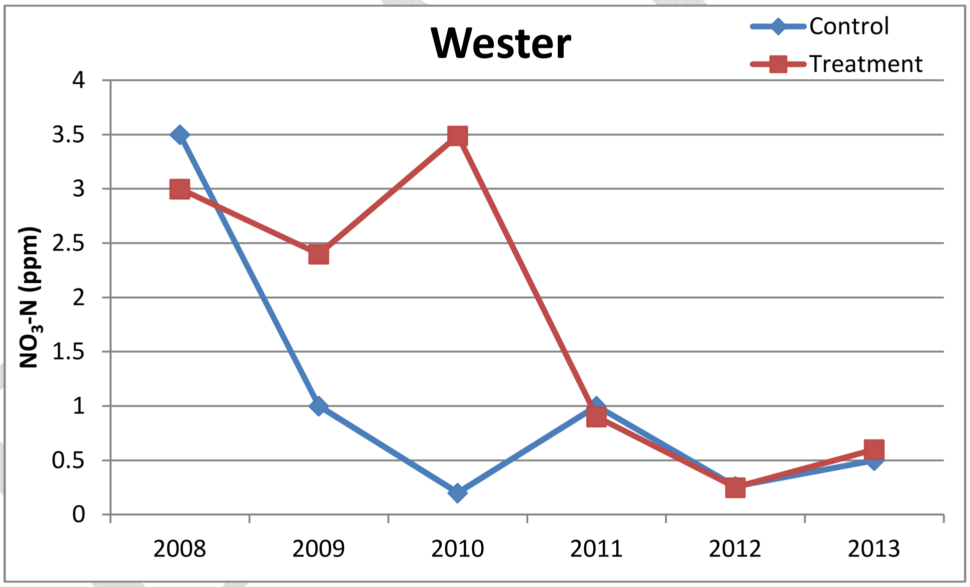


Figure 2.37. Nitrate concentrations measured at the Wester sites, 2008–2013.

2.6 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 2.38). For more detailed information on the methodology, site location, and relevant background information, please refer to the 2008 Monitoring Plan (SWCA 2008).

Figure 2.38. Parshall flume located at the thinned Chilili site.

During the 2013 monitoring period, rainfall occurred in the project area on 12% of the days monitored compared with 17% of the days monitored in 2012. However, about 85% of these rainfall events were relatively small and totaled less than 2.5 mm (0.1 inch). During the same monitoring period, only three flow events were recorded across all watersheds, which was the lowest number of flows recorded thus far in the study. 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 2013 monitoring period, there were no basin-wide storm events that generated flow across all study sites simultaneously. Many of the flumes did not even record flow events during the 2013 monitoring season, which is a product of the persistent drought over the region. The flumes that did not record surface flow events during the 2013 season included the Vigil control, both Kelly sites and both Chilili sites. Even though there were very few recordable storm events, trends that were beginning to show in 2011 and 2012 are still persisting. The flumes that did record events were the Vigil treatment, which recorded three flow events, and the Wester treatment and control, which recorded one paired event (Figure 2.39 and Figure 2.40). The results of these flows can be found in Table 2.3 through Table 2.5. Whether the differences in peak flows on the treated watershed versus the control watershed persist remains to be seen, and with more future flows this picture will become clearer. All Parshall flumes were functioning properly during the 2013 season.

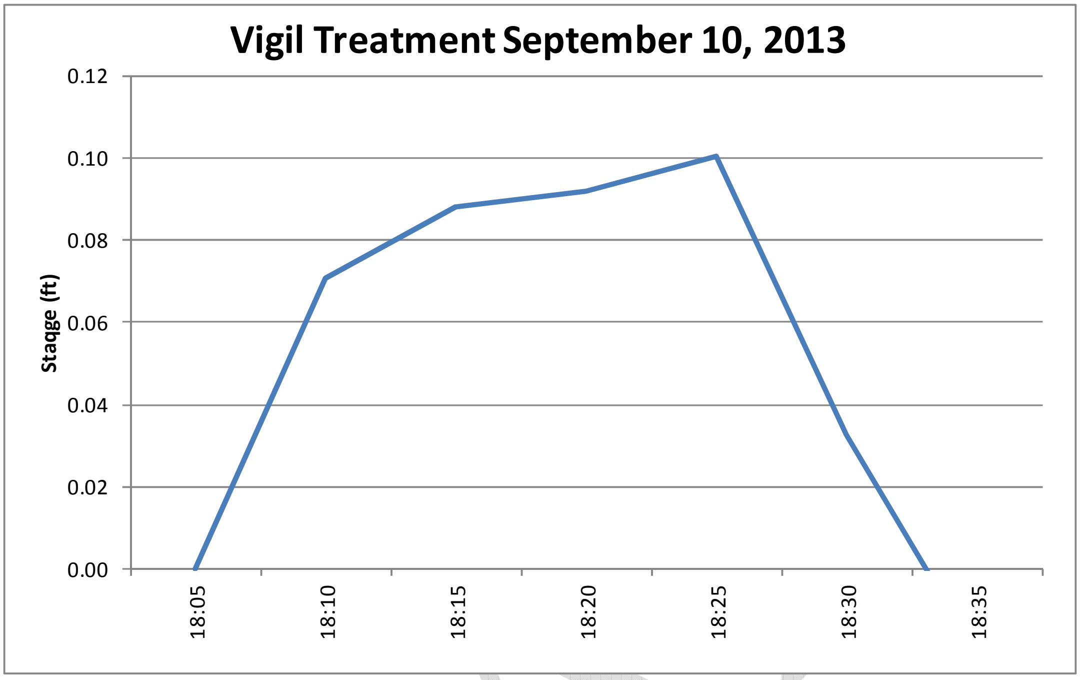


Figure 2.39. Hydrograph showing plot from the Vigil treatment during a storm flow event on September 10, 2013.

Table 2.3. Summary of Runoff Event for the Wester Sites, September 12, 2013

Runoff Parameters	Study Sites	
	Wester Treated	Wester Control
Flow start	16:27	16:27
Flow stop	17:02	17:02
Peak stage (feet)	0.507	0.421
Peak flow (cubic feet/second)	0.349	0.259
Flow duration (minutes)	35	335
Total volume of flow (cubic feet)	487.8	237
Watershed area (acres)	1.03	1.47
Volume of flow per acre (cubic feet/acre)	473.59	35.06
Total rainfall (inches)	1.47	1.47
Total volumetric rainfall (cubic feet)	5,496.18	5,496.18
Rainfall/Runoff ratio	0.089	0.043

Table 2.4. Summary of Runoff Event for the Vigil Treatment Site, September 10, 2013

Runoff Parameters	Study Site
	Vigil Treated
Flow start	18:05
Flow stop	18:30
Peak stage (feet)	0.10
Peak flow (cubic feet/second)	0.03
Flow duration (minutes)	25
Total volume of flow (cubic feet)	28.8
Watershed area (acres)	0.68
Volume of flow per acre (cubic feet/acre)	42.35
Total rainfall (inches)	0.9
Total volumetric rainfall (cubic feet)	3,135
Rainfall/Runoff ratio	0.01

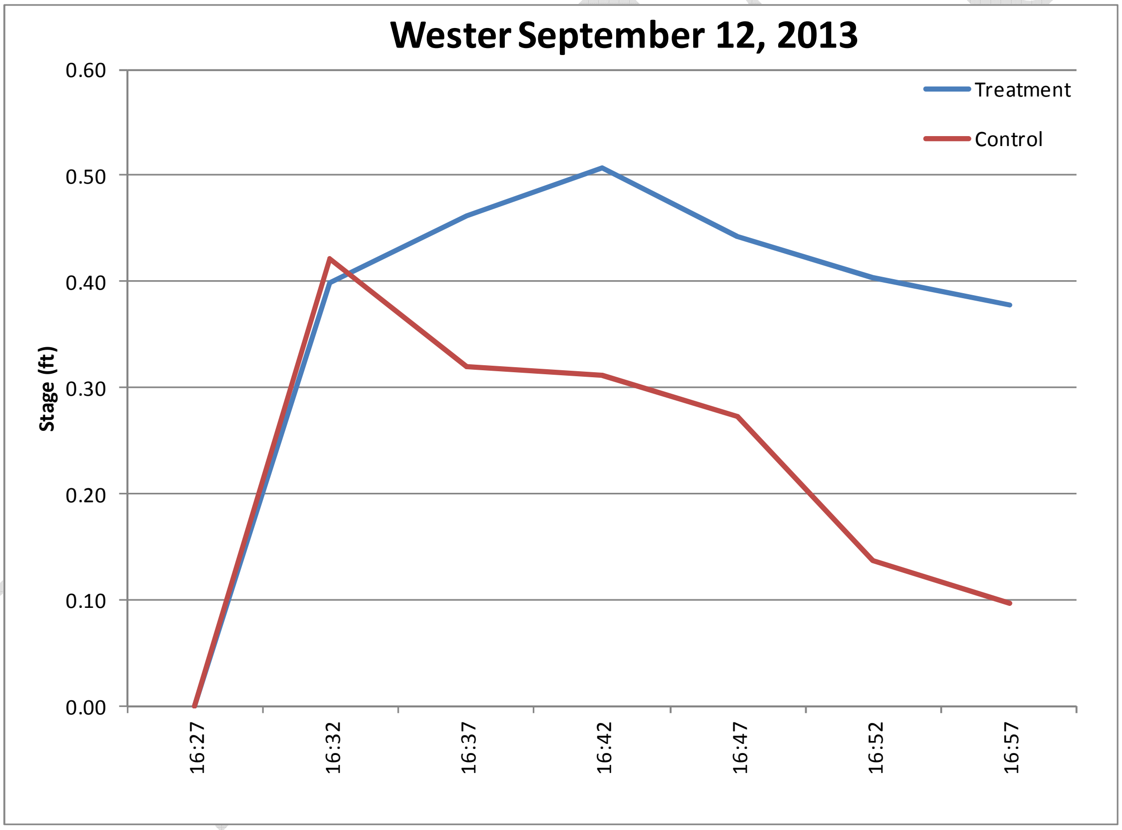


Figure 2.40. Hydrograph showing the storm flow at the treatment and control Wester site that occurred on September 12, 2013

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.

2.6.1 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, 2008–2013

Location	Number of Flow Events	Range of Duration (minutes)	Median Duration (minutes)	Range of Volume (cubic feet)	Median Volume (cubic feet)
Chilili treatment	3	55–840	512.5	245–17,751	9,197
Chilili control	8	25–715	167.5	36–2,564	920.5
Kelly control	4	25–35	30	38–392	54.5
Kelly treatment	1	15	15	69	69
Vigil treatment	14	15–115	40	46–197	117
Vigil control	7	20–80	50	123–290	218
Wester treatment	6	10–235	102.5	39–4,765	210
Wester control	8	10–760	90	42–9,458	444
All ponderosa pine	**25**	**10–840**	**95**	**35–9,458**	**468.5**
All piñon/juniper	**23**	**15–115**	**32.5**	**38–392**	**93**

2.6.2 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 was recorded at the Chilili treatment site during 2011 (0.11 m [0.37 feet]), while the greatest recorded peak flow of 1.29 feet was recorded at the Wester control site 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, 2008–2013

Location	Number of Flow Events	Range of Peak Stage (feet)	Median Peak Stage (feet)
Chilili treatment	3	0.19–0.76	0.475
Chilili control	8	0.11–0.57	0.375
Kelly control	4	0.14–0.39	0.175
Kelly treatment	1	0.02–0.23	0.23
Vigil treatment	14	0.06–0.46	0.19
Vigil control	7	0.22–0.28	0.27
Wester treatment	6	0.15–0.85	0.19
Wester control	8	0.12–1.29	0.38
All ponderosa pine	**25**	**0.11–1.29**	**0.35**
All piñon/juniper	**23**	**0.02–0.46**	**0.175**

2.6.3 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. Note that some rainfall/runoff values were not calculated due to missing rainfall data. 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, 2008–2013

Location	Number of Flow Events	Range of Rainfall/Runoff Ratio	Median Rainfall/Runoff Ratio
Chilili treatment	3	0.01–0.561	0.056
Chilili control	8	0.003–0.550	0.022
Kelly control	4	0.045–0.460	0.088
Kelly treatment	1	–	–
Vigil treatment	14	0.022–0.160	0.056
Vigil control	7	0.063–0.654	0.439
Wester treatment	6	0.029–0.058	0.044
Wester control	8	0.015–0.848	0.407
All ponderosa pine	**25**	**0.003–0.848**	**0.058**
All piñon/juniper	**23**	**0.022–0.479**	**0.075**

2.7 TREES

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).

2.7.1 TREES AND WILDFIRE FUELS

Tree monitoring measurements in the fall of 2013 included observations of canopy dieback, disease or damage, and live and dead status.

Basal Area Measurements

Basal area measurements were taken in spring 2011. Since basal areas are unlikely to have changed between 2011 and 2013, no basal area measurements were taken during this field season. The 2011 basal area measurements are shown in Table 2.8.

Table 2.8. Treatment Designation for All Plots (with basal area totals), 2011

Site	Average Basal Area (square feet/acre) 2008	Average Basal Area (square feet/acre) 2011
Chilili treatment	210	79
Chilili control	194	194
Kelly control	106	106
Kelly treatment	155	47
Vigil treatment	124	39
Vigil control	129	129
Wester treatment	220	99
Wester control	213	213

Stand Structure

Diameter measurements of trees were taken in 2012. Since diameter measurements are unlikely to have changed significantly since 2012, diameter measurements were not taken in 2013. The measurements from 2012 are used below to demonstrate the stand structure and various size classes at each site. Figure 2.41 and Figure 2.42 show the size class structure of ponderosa pine trees (diameter at breast height [DBH]) at the ponderosa pine sites, Chilili and Wester. Figure 2.43 and Figure 2.44 show the size class structure of piñon and juniper trees (diameter at root crown [DRC]) at the piñon/juniper sites, Kelly and Vigil.

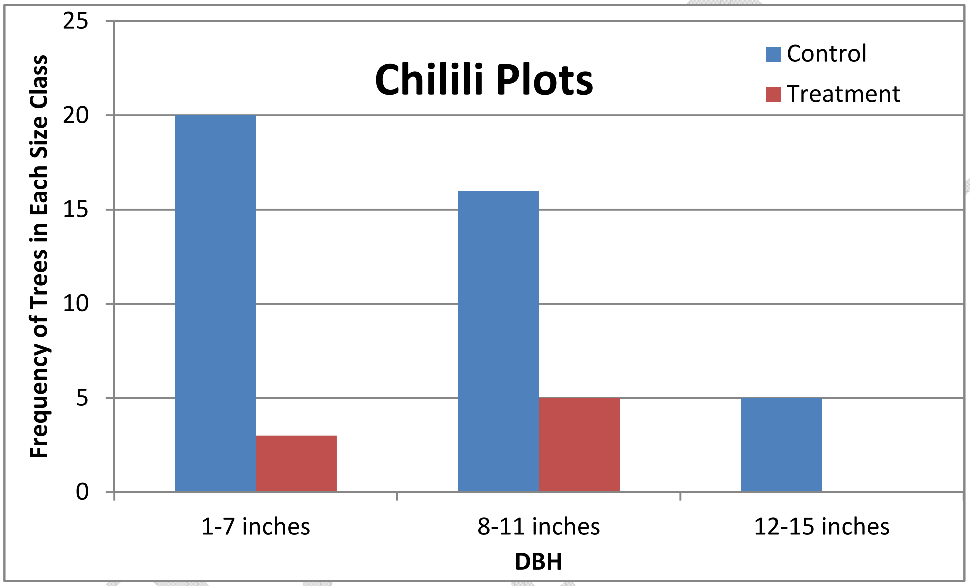


Figure 2.41. Size classes of ponderosa pine trees measured at DBH on the Chilili control and treatment plots.

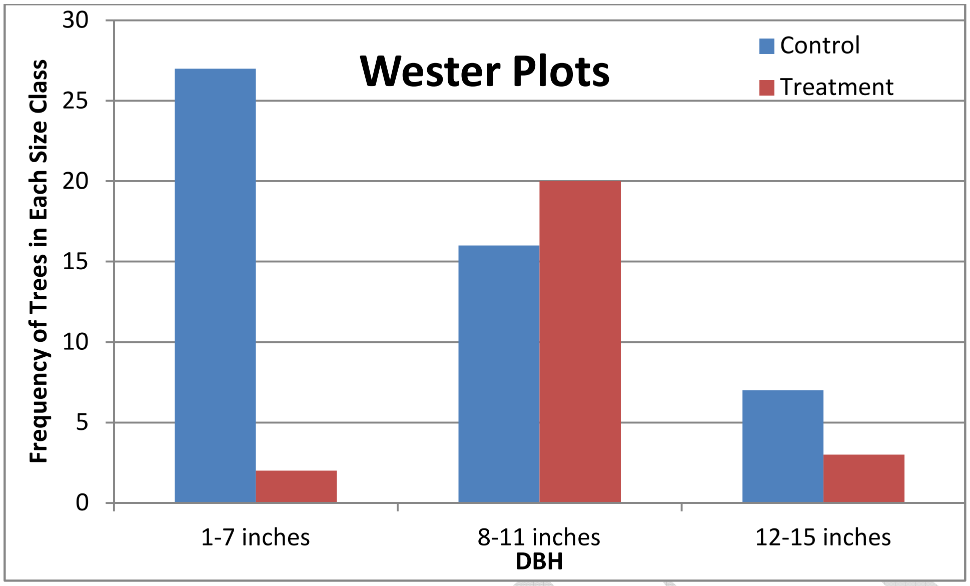


Figure 2.42. Size classes of ponderosa pine trees measured at DBH on the Wester control and treatment plots.

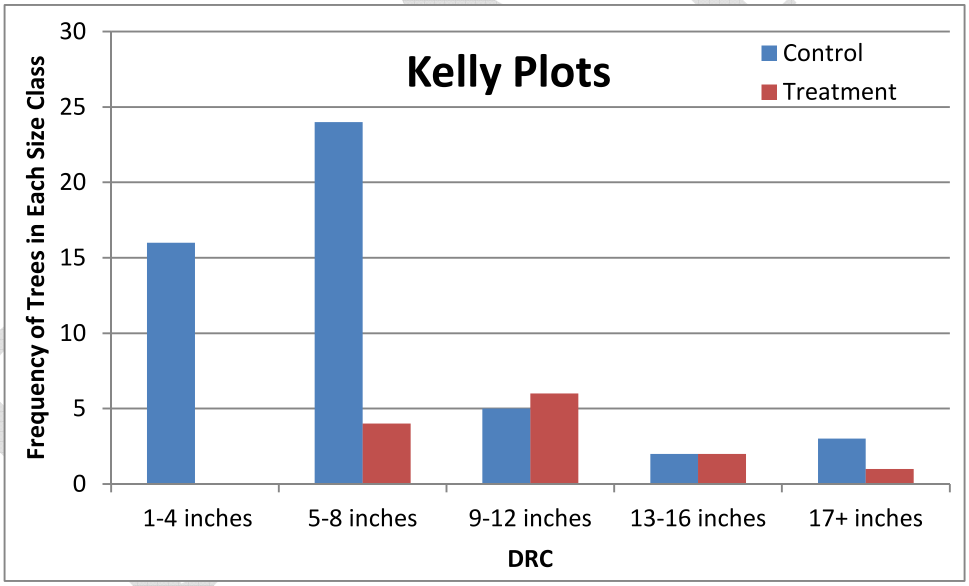


Figure 2.43. Size classes of piñon/juniper trees measured at DRC on the Kelly control and treatment plots.

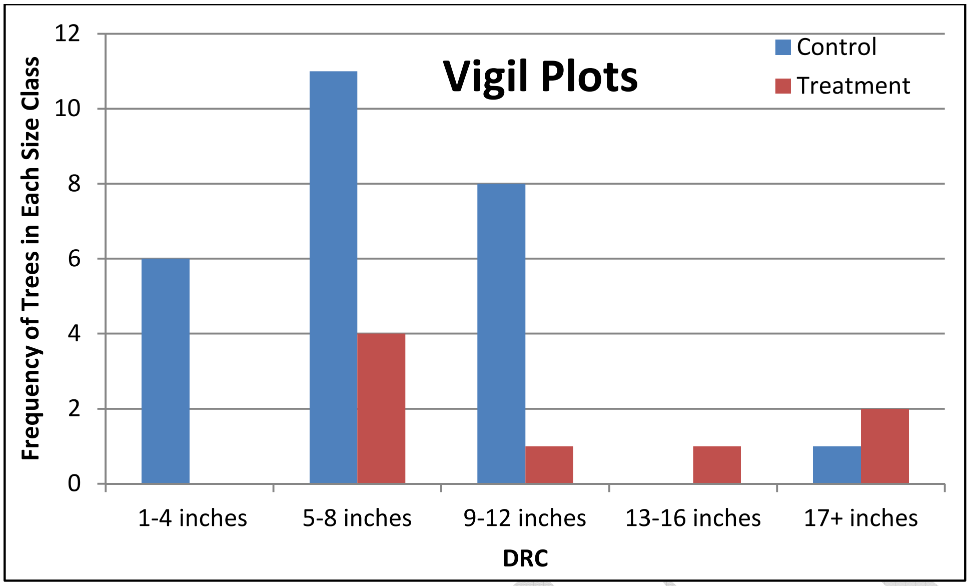


Figure 2.44. **Size classes of piñon/juniper trees measured at DRC on the Vigil control and treatment plots.**

Figure 2.41 through Figure 2.44 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, for example, 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 2.45 illustrates crown dieback across all sites.

Crown dieback levels from 2008 to 2013 are presented below by site and year (see Figure 2.45). 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 2012 were low compared to previous years across all plots, except on the Chilili plots where dieback had increased by 8% to 10% on both plots. Dieback in 2013 decreased at the Chilili site and stayed relatively constant compared with 2012 rates at the Wester site. Dieback however increased significantly on both piñon/juniper sites, most noticeably at the Kelly control site with an increase of over 25%. Much of this dieback is likely related to piñon die-off that was observed to be particularly prevalent at the Kelly control site. We believe that dieback levels are within the normal range of variability for all six years.

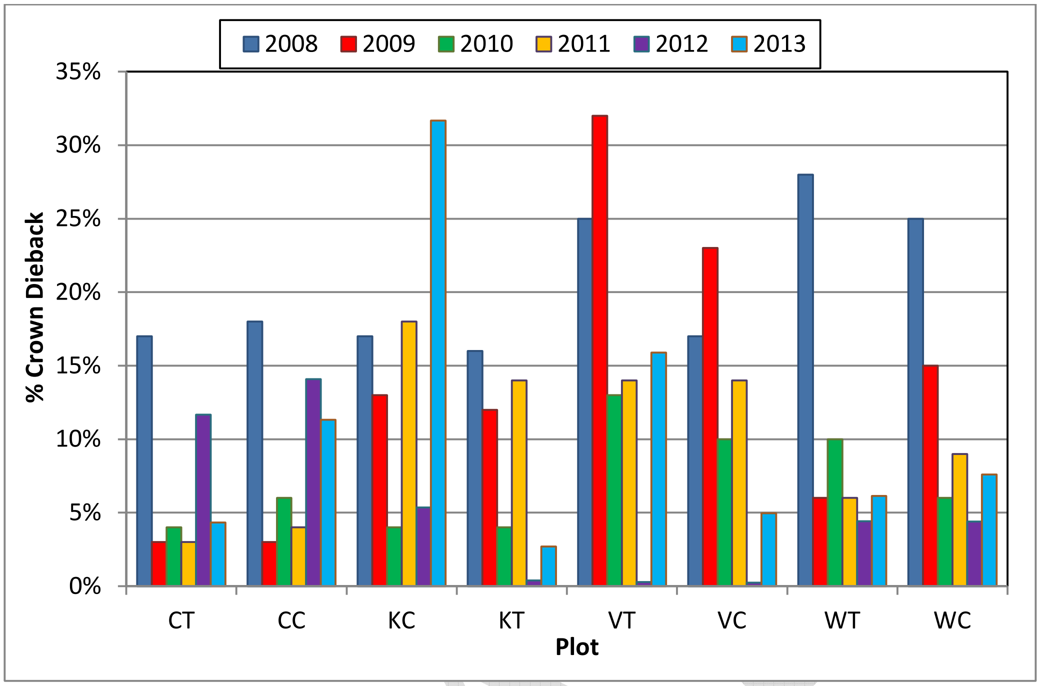


Figure 2.45. Average percent crown dieback of tree canopies for each thinning plot, 2008–2013.

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 six years (Figure 2.46).

The 2012 season saw the highest mortality (not attributed to tree cutting) of all six years, with most mortality recorded on previously thinned plots (with the exception of Wester). The Vigil treatment was the only plot to experience mortality in 2013; 100% of this mortality was of piñon pine and was attributed to bark beetle infestation. Conversely, the Vigil control plot showed no tree mortality in any of the six years. The Vigil treatment plot has experienced the greatest percent mortality of all plots; similarly, this plot also exhibited higher crown dieback than other plots in previous years, suggesting that the trees on that plot were experiencing environmental stress possibly leading to the high mortality in 2012 and 2013. The Chilili and Kelly treatment plots also suffered tree mortality of a little less than 10% in 2012. The Wester control plot was the only control plot that exhibited mortality in 2012. This plot had suffered tree mortality in 2009, 2011, and 2012, which is likely a result of competitive stress and/or beetle damage. A number of trees at the Wester site suffered broken/dead tops and showed signs of drought stress. As demonstrated in Figure 2.45, a number of plots experienced crown dieback in 2013 that may result in increased mortality in subsequent years. Tree mortality will continue to be monitored every fall.

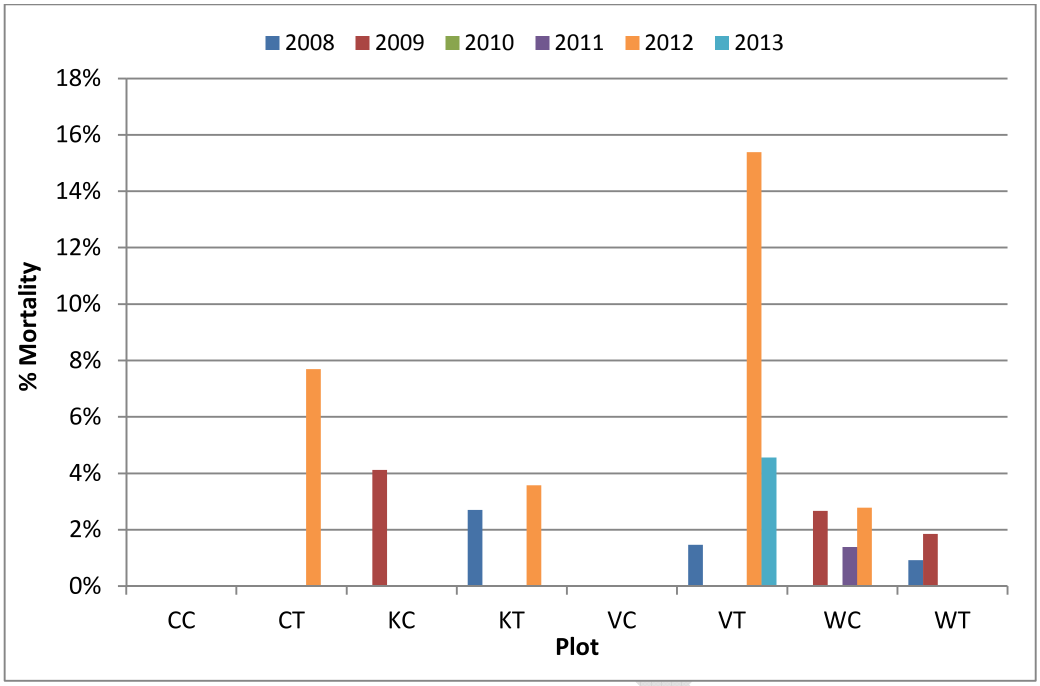


Figure 2.46. Percent tree mortality recorded across all thinning plots from 2008–2013. Percent mortality is recorded in relation to tree status in 2008.

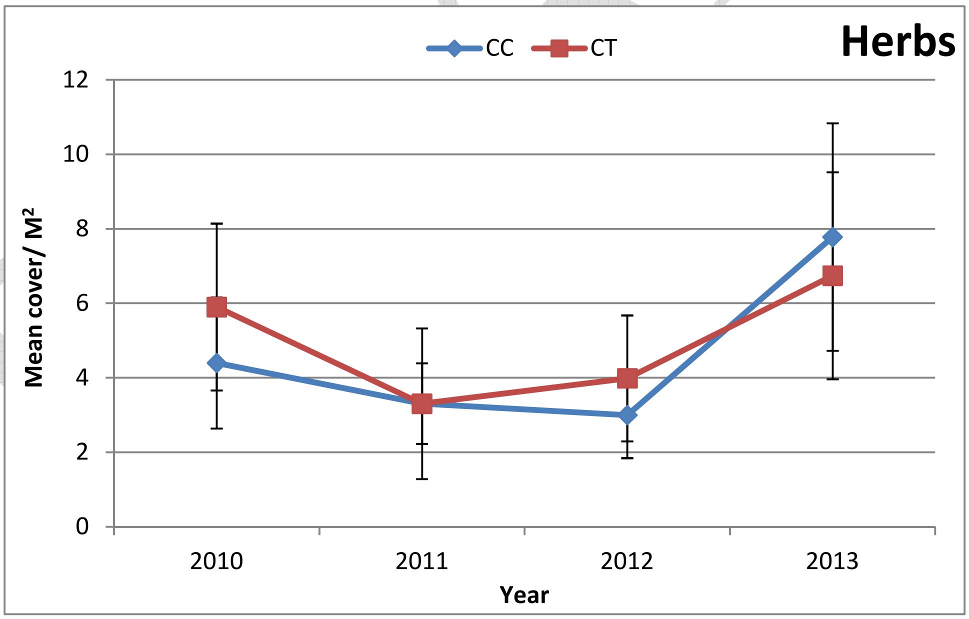
2.7.2 VEGETATION AND GROUND SURFACE COVER MONITORING

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. 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, and 2013 and will be 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 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) 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 are continued each year, including in 2013.

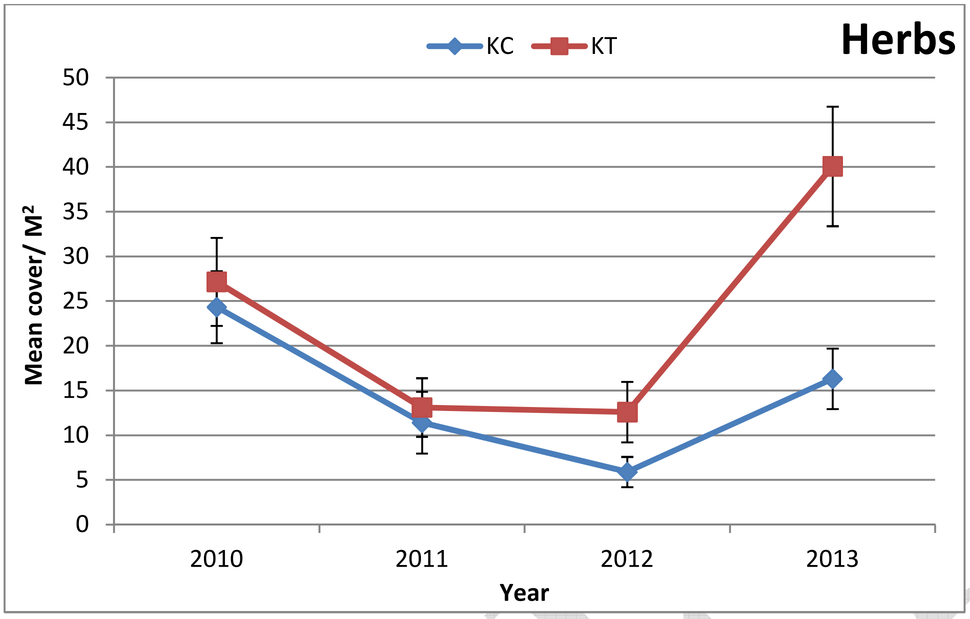
The vegetation and ground cover data measured from the replicated quadrats on wildlife plots provide the most appropriate data for statistical testing for differences in those cover values resulting from thinning treatments, because there is sufficient sample replication to perform parametric statistical 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 a standard parametric t-test. Ideally, there should be 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.

Tree canopy cover and vegetation 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 three years since 2012. The next tree canopy and structure measurements will be taken in 2015.

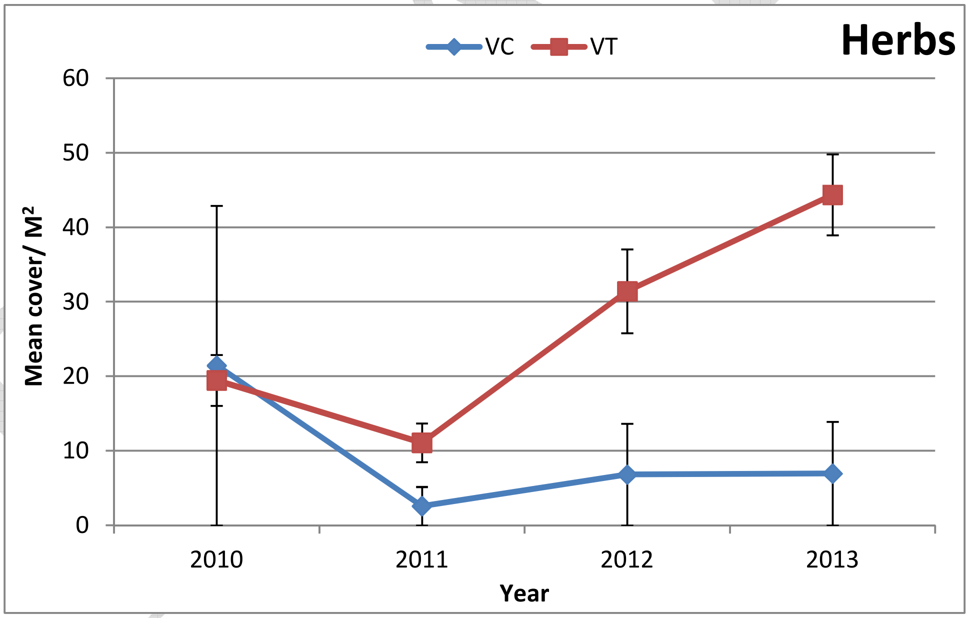
Results for herbaceous understory vegetation and soil surface cover types measured from the thirty-six 1-m^2 (10.8-square-foot) quadrats in the fall 2013 are presented in Figure 2.47, a–bb, providing separate graphs for forbs, grasses, and each soil surface cover type. Results from 2010 prior to tree thinning treatments and in 2011 and 2012 following thinning treatments also are presented in Figure 2.47, 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 2013 and shown in Figure 2.47 are presented in Table 2.9.



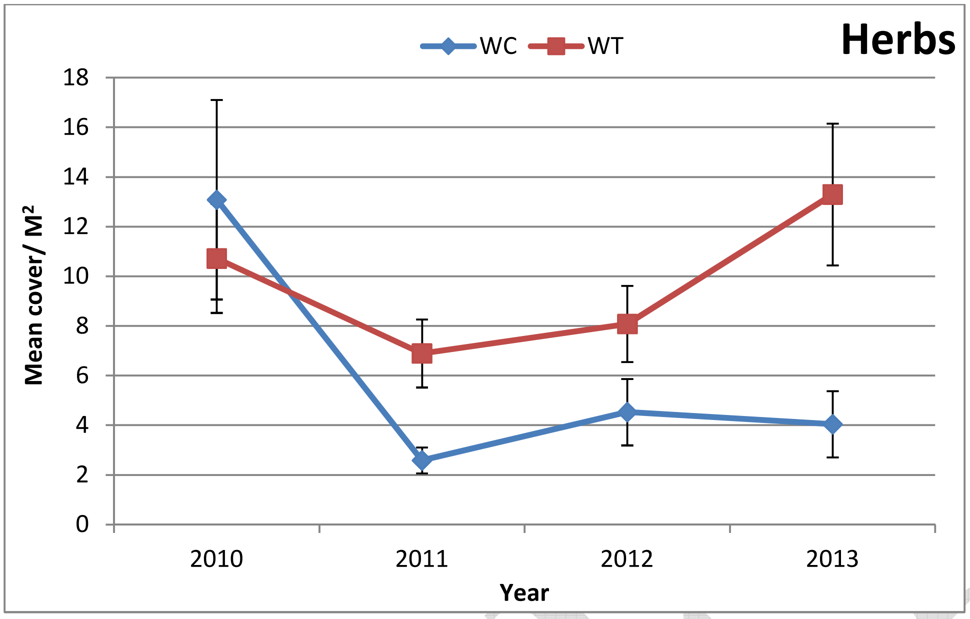
a. Herbs, Chilili.



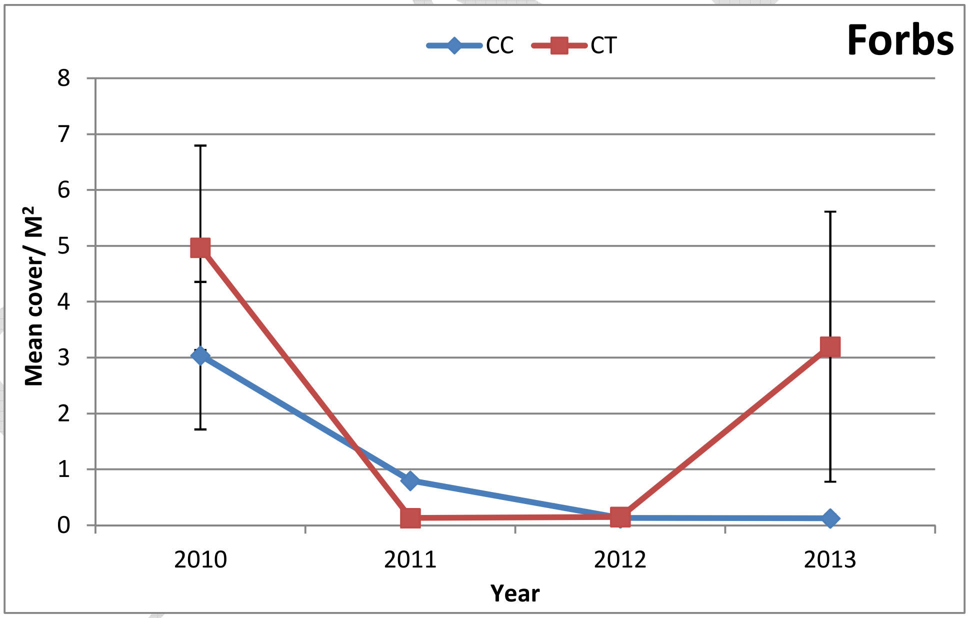
b. Herbs, Kelly.



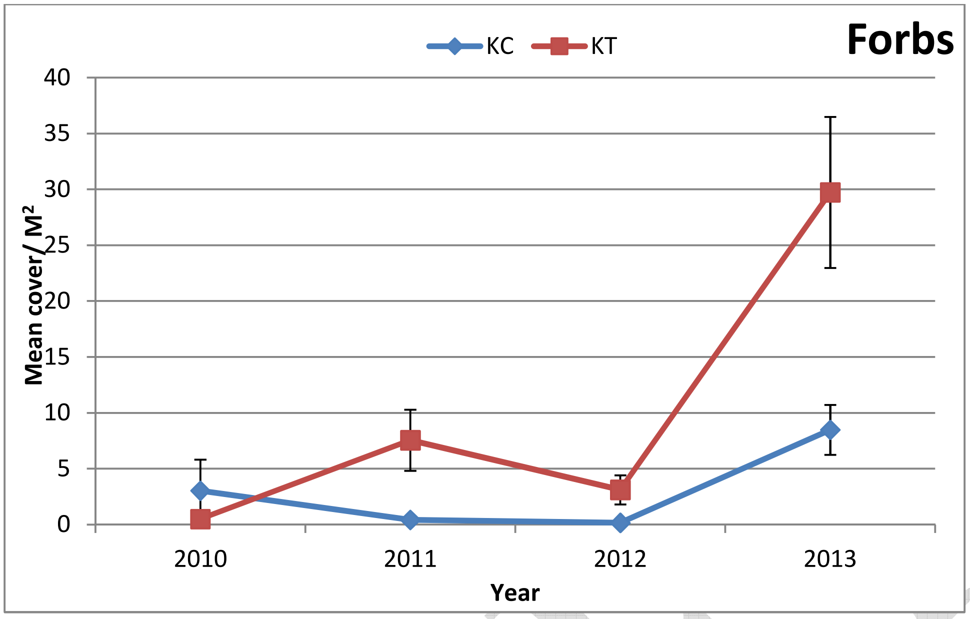
c. Herbs, Vigil.



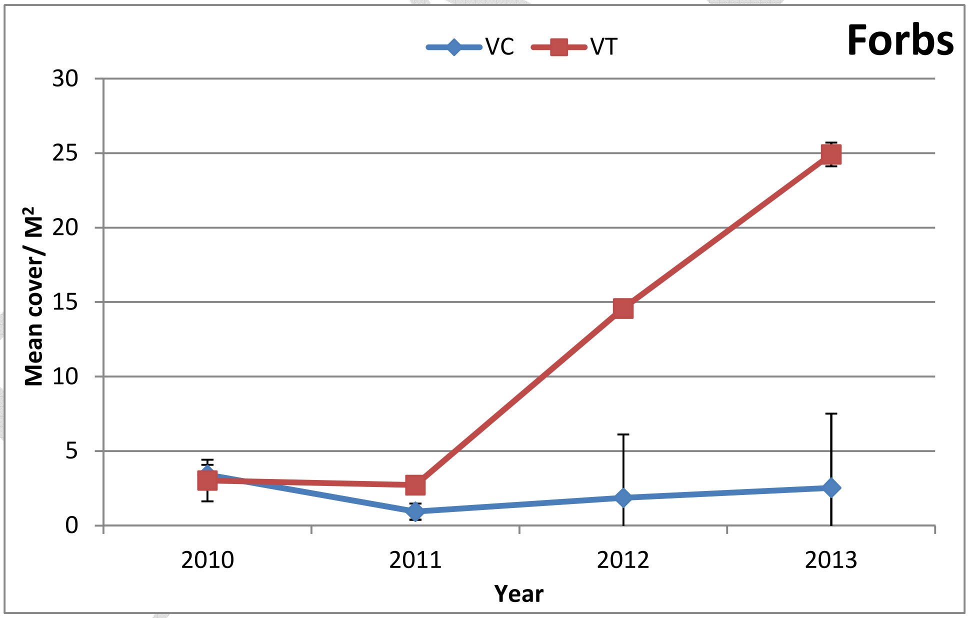
d. Herbs, Wester.



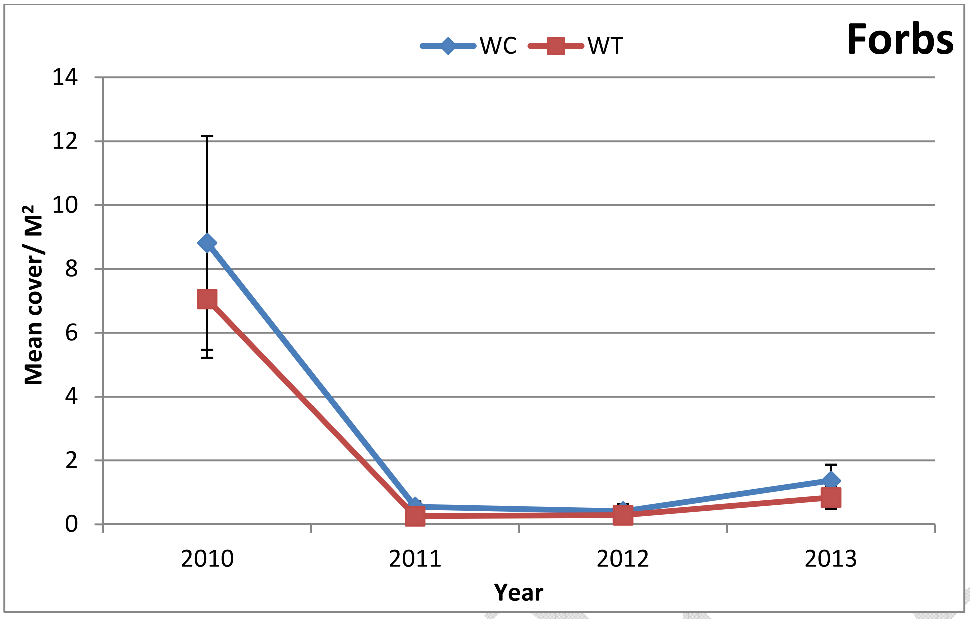
e. Forbs, Chilili.



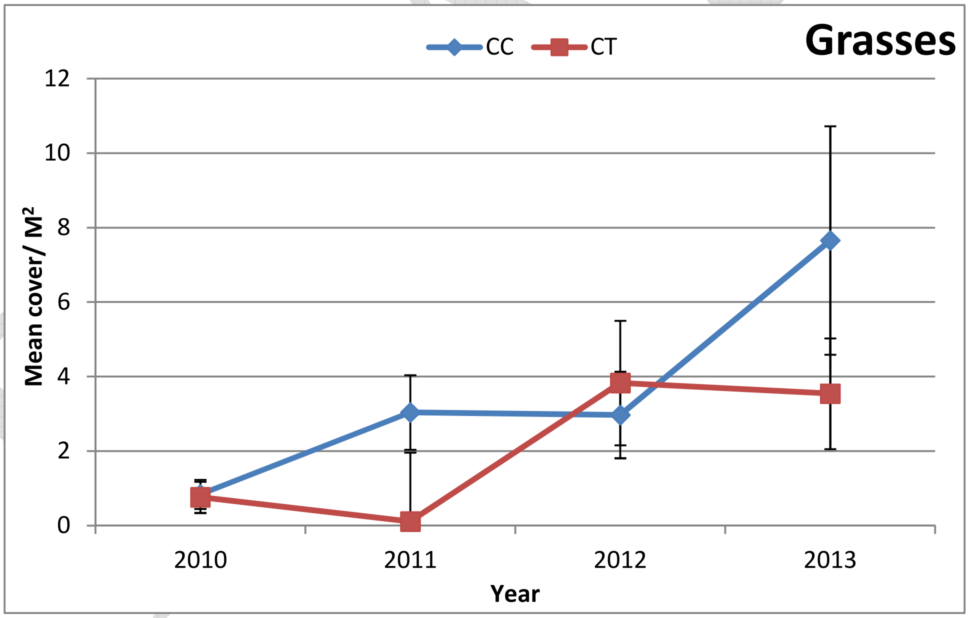
f. Forbs, Kelly.



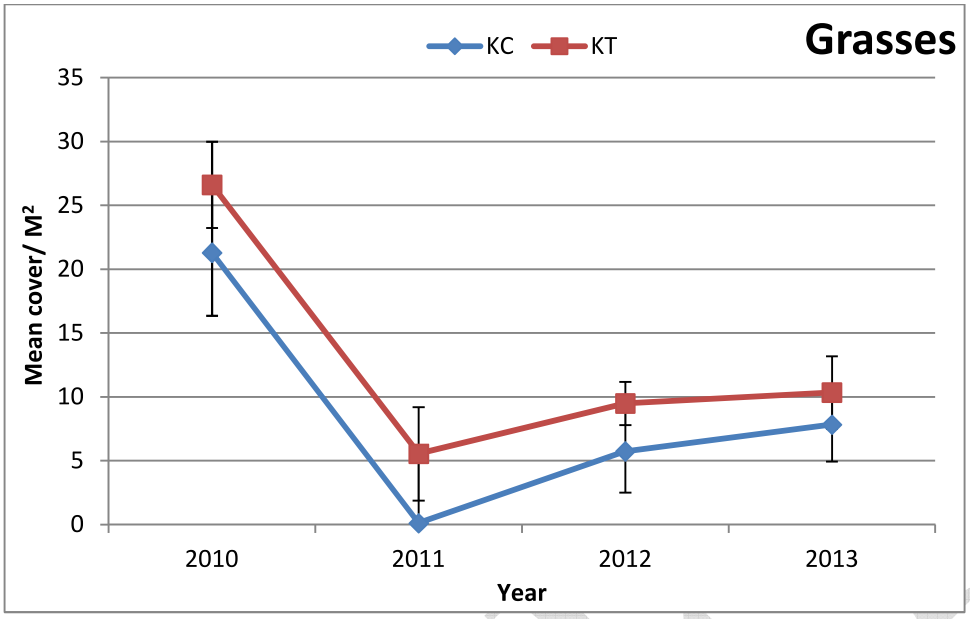
g. Forbs, Vigil.



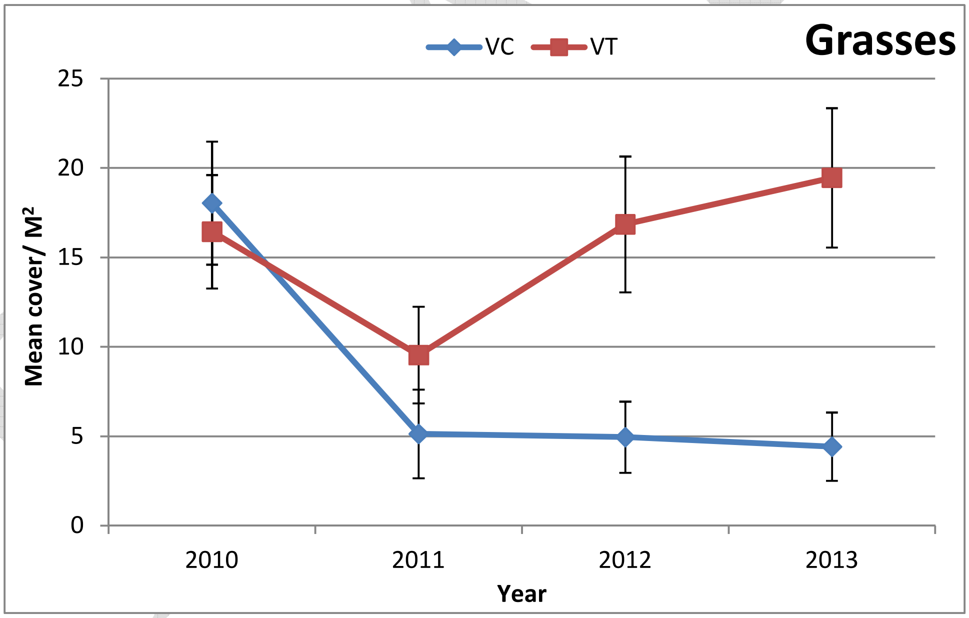
h. Forbs, Wester.



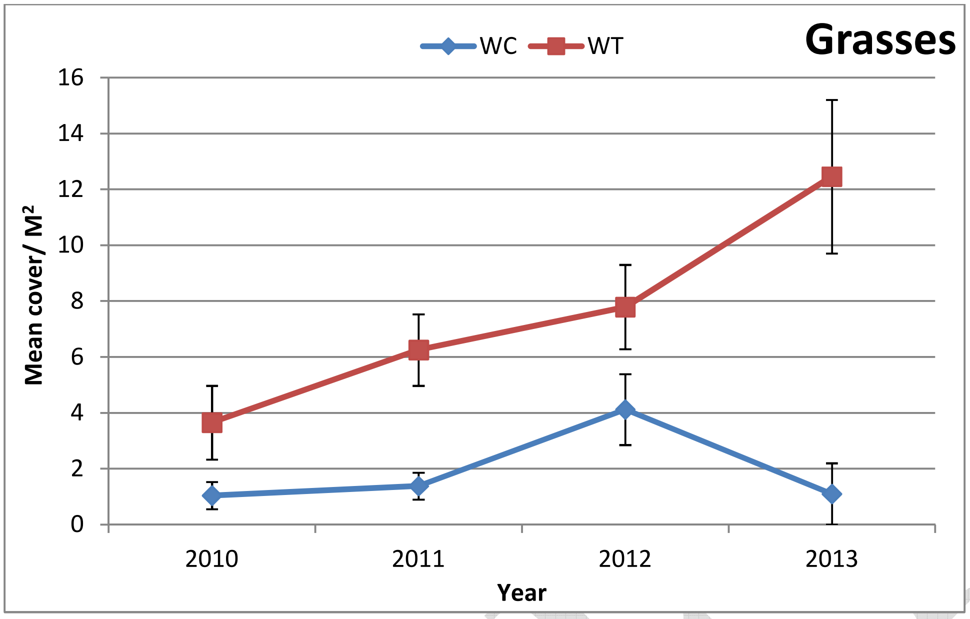
i. Grass, Chilili



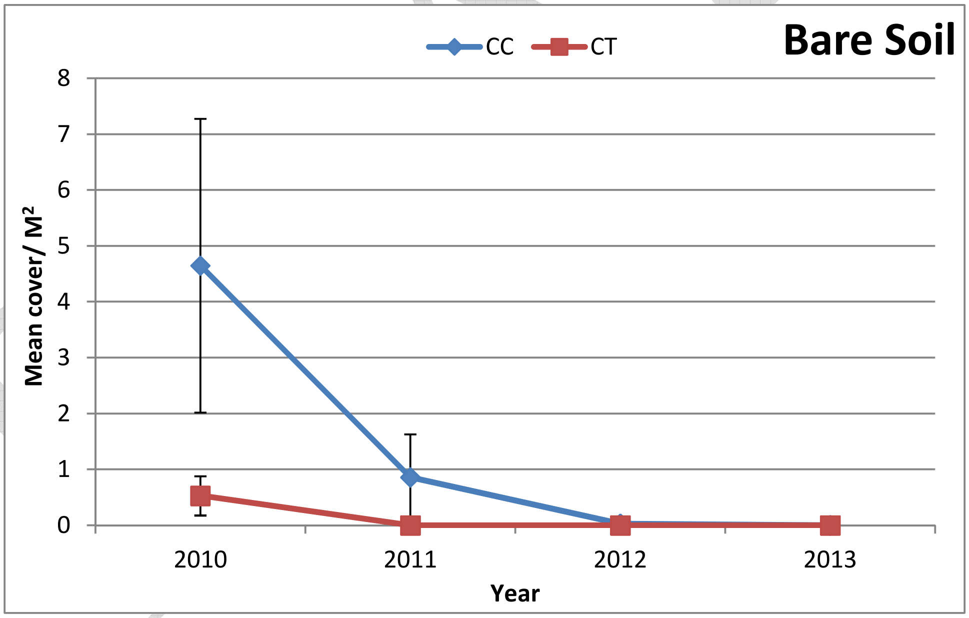
j. Grass, Kelly.



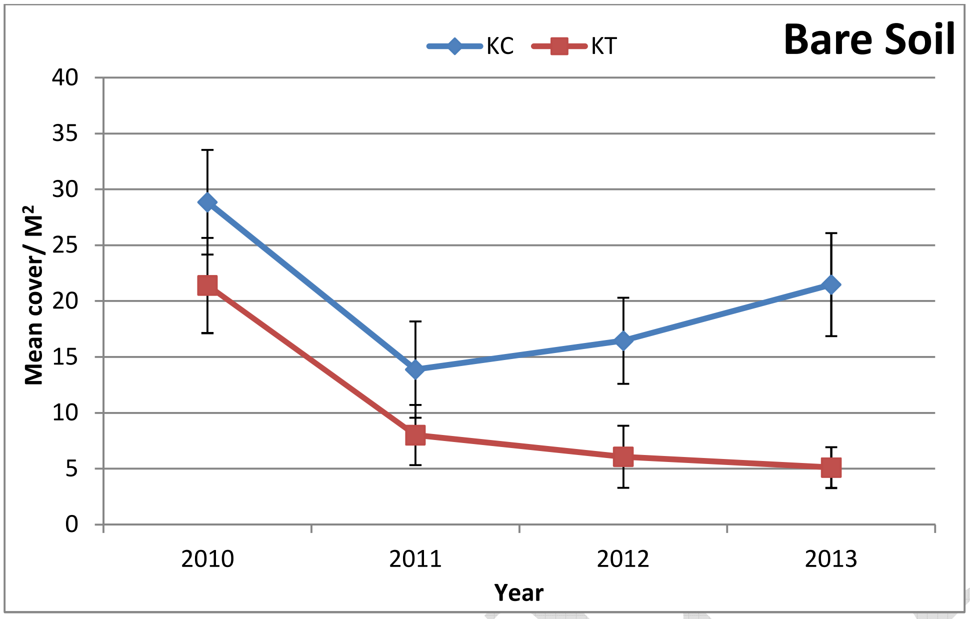
k. Grass, Vigil.



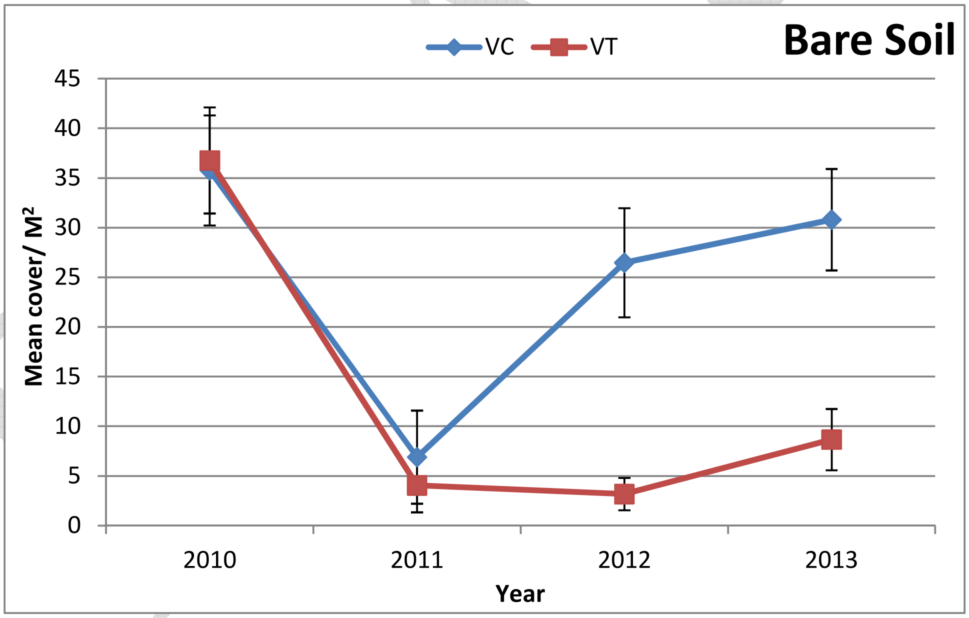
l. Grass, Wester.



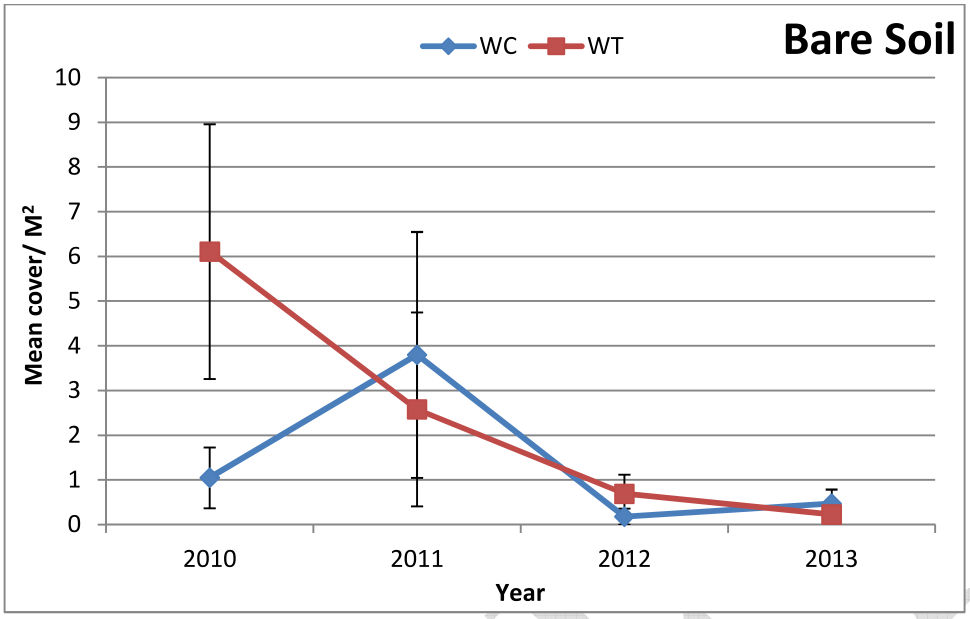
m. Bare soil, Chilili.



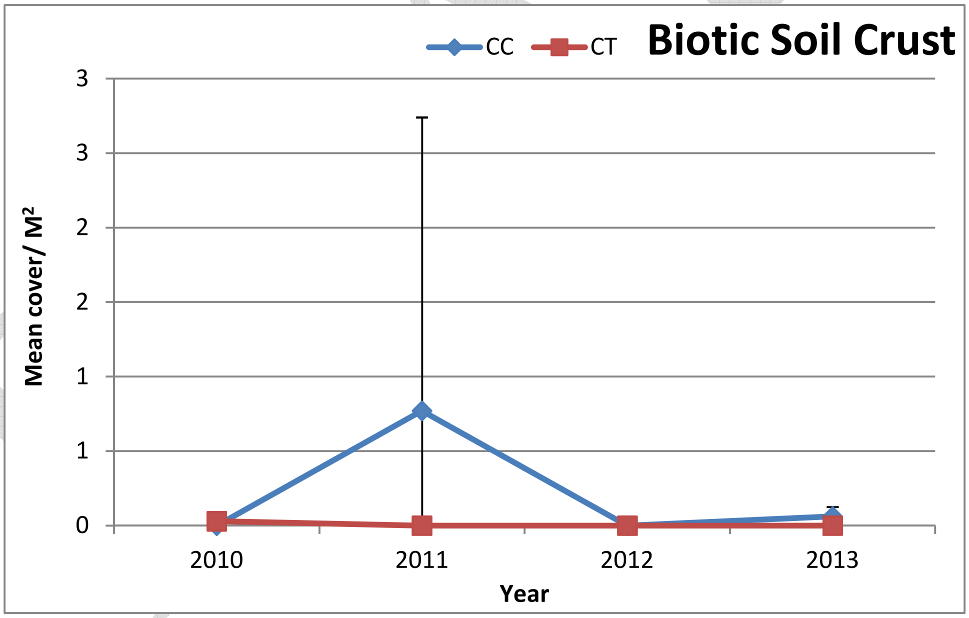
n. Bare soil, Kelly.



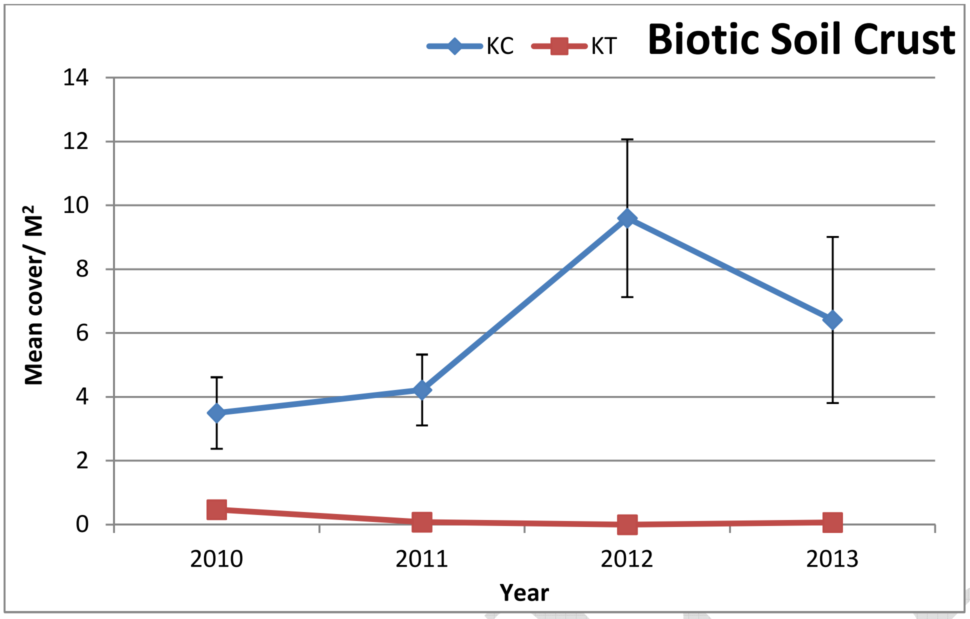
o. Bare soil, Vigil.



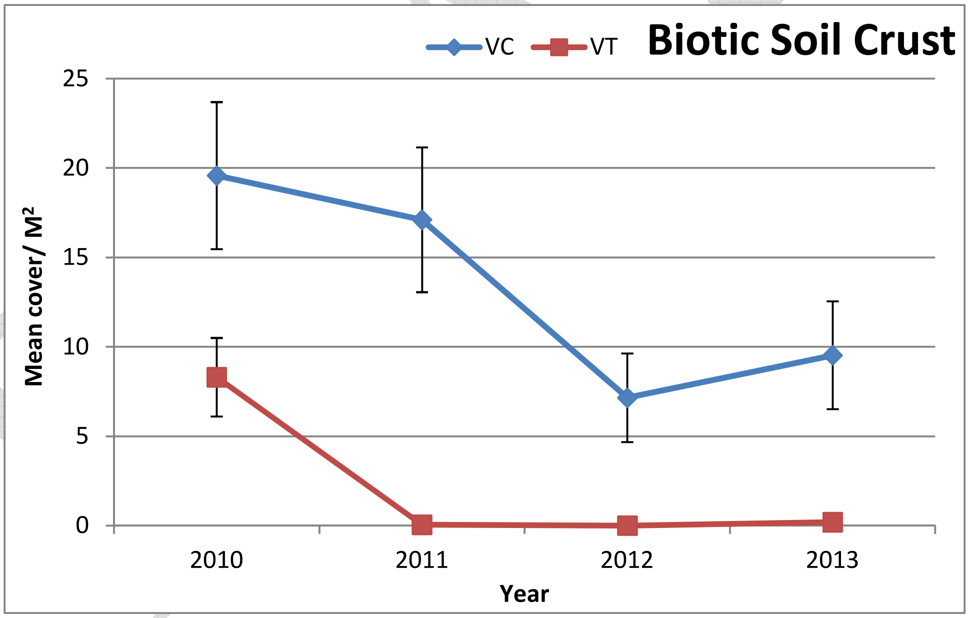
p. Bare soil, Wester.



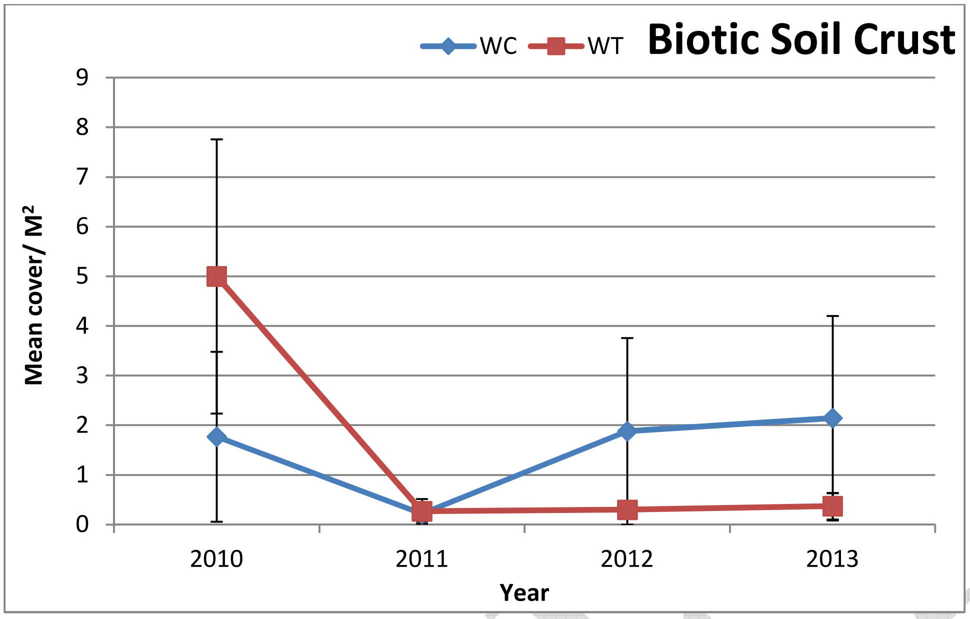
q. Biotic soil crust, Chilili.



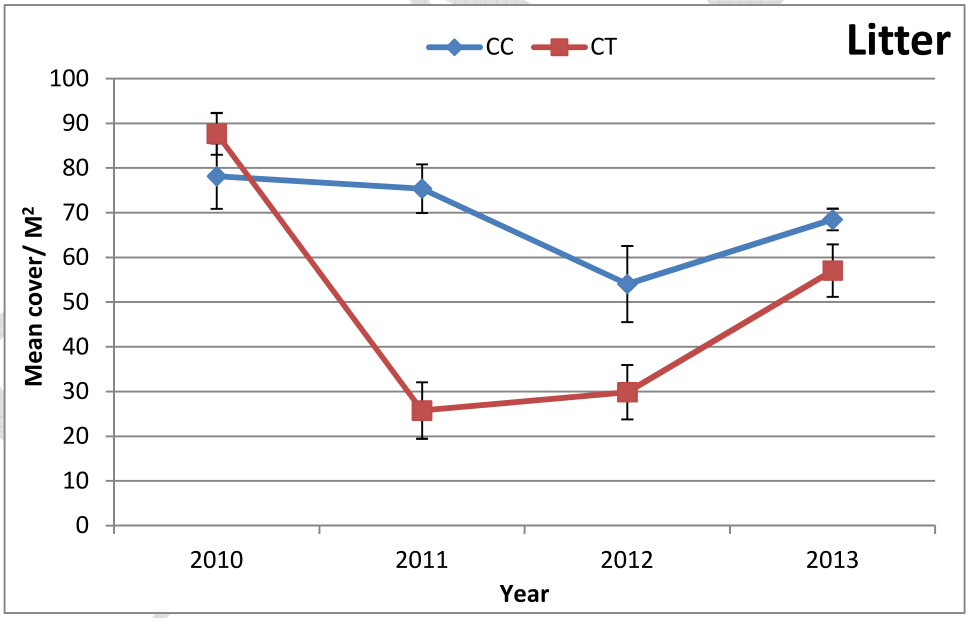
r. Biotic soil crust, Kelly.



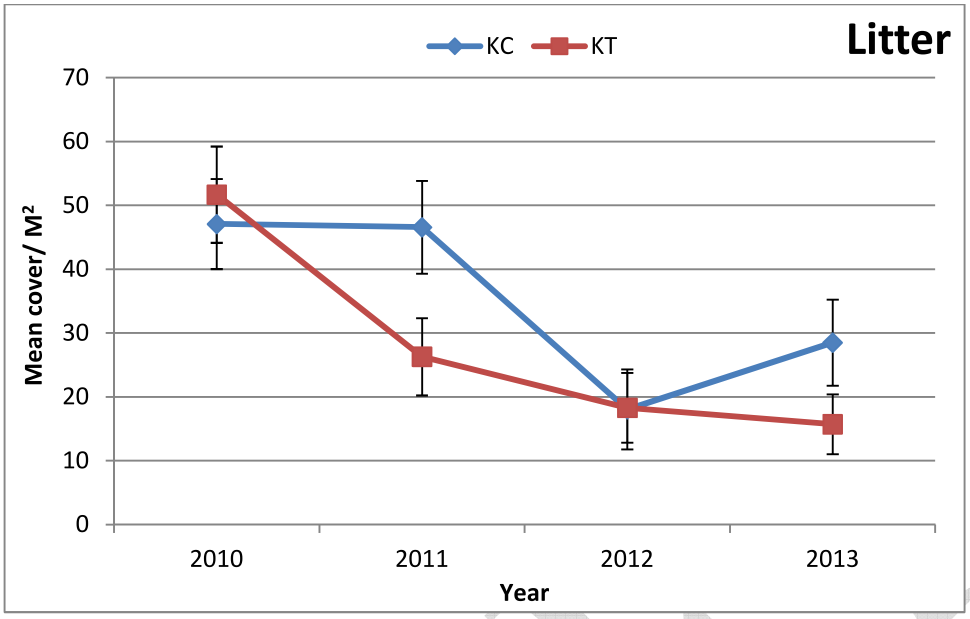
s. Biotic soil crust, Vigil.



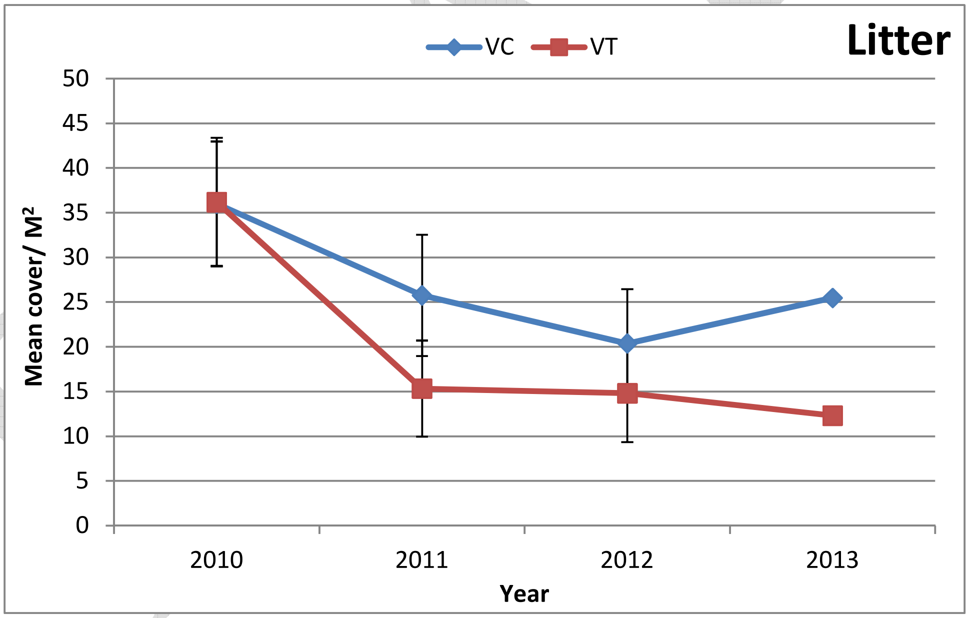
t. Biotic soil crust, Wester.



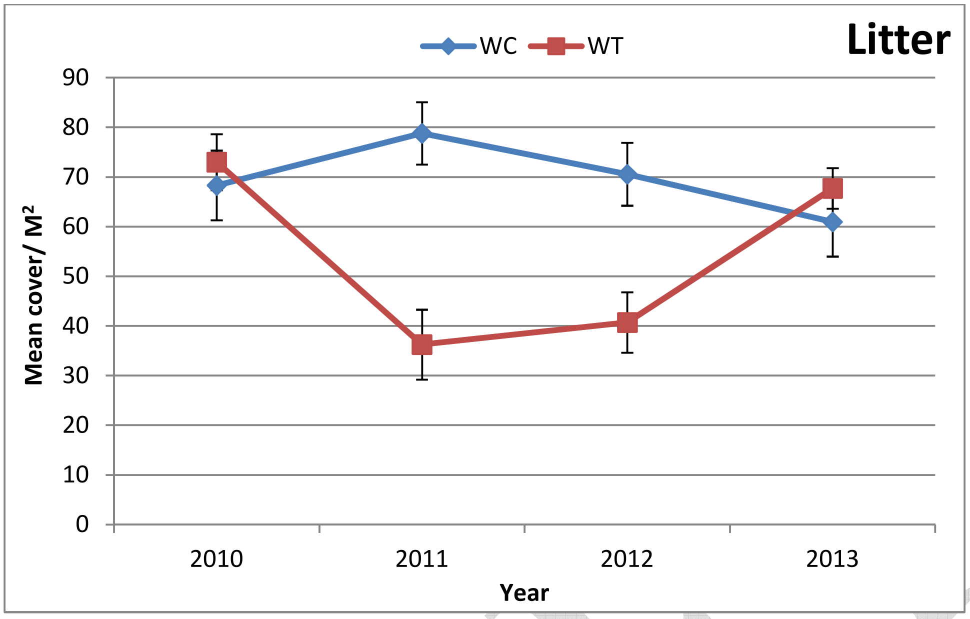
u. Leaf litter, Chilili.



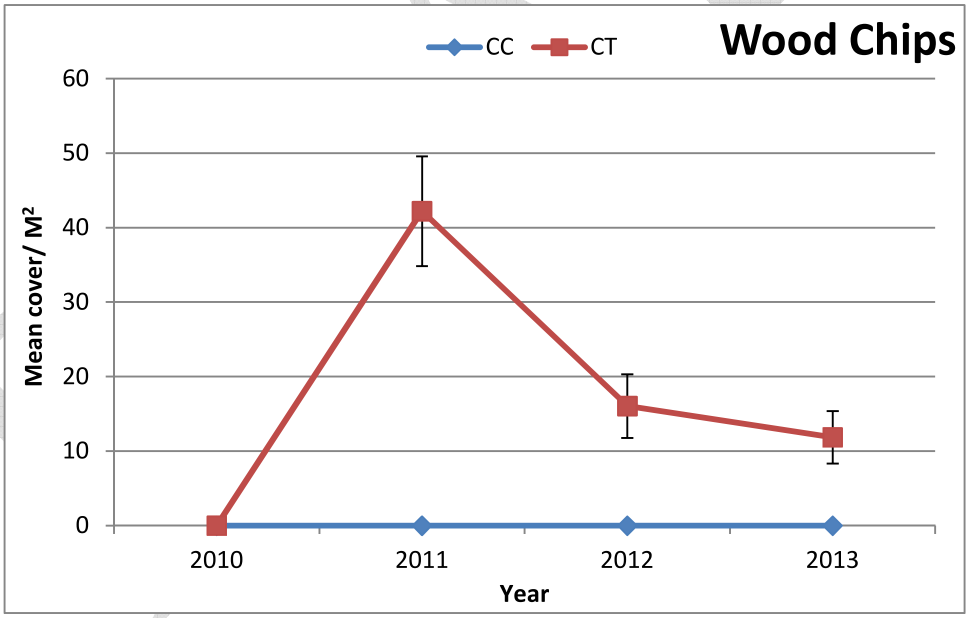
v. Leaf litter, Kelly.



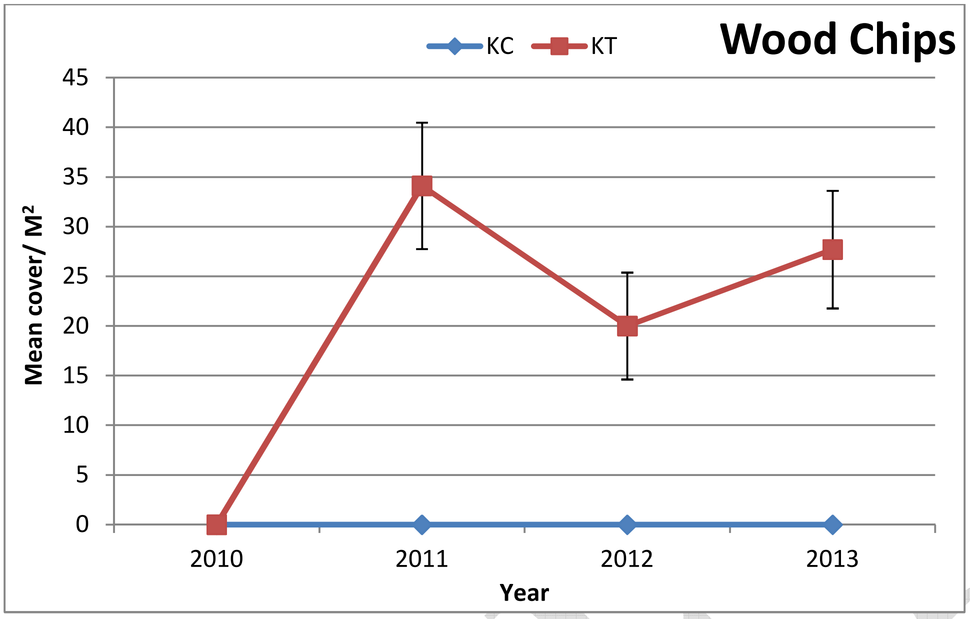
w. Leaf litter, Vigil.



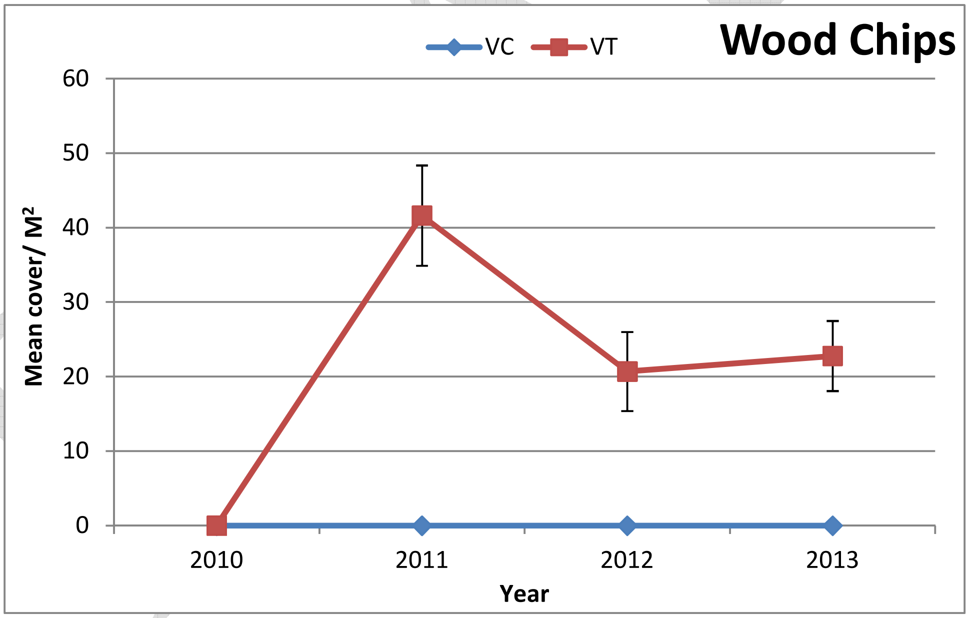
x. Leaf litter, Wester.



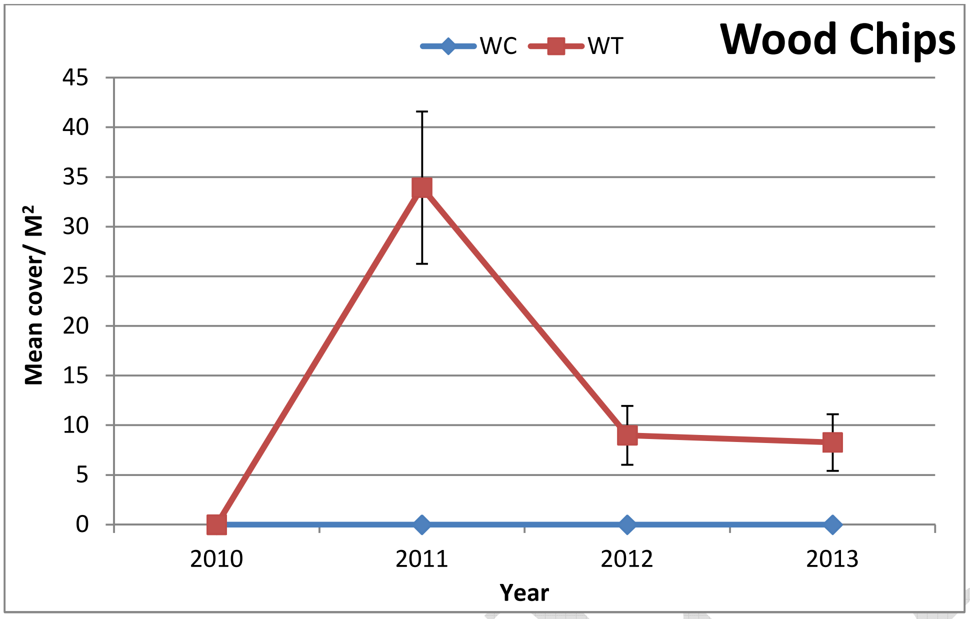
y. Wood chips, Chilili.



z. Wood chips, Kelly.



aa. Wood chips, Vigil.



bb. Wood chips, Wester.

Figure 2.47. These graphs illustrate the mean values cover type found across all vegetation quadrats among all of the study sites and paired study plots from fall 2010 through 2013. 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 2013

Site	Parameter	Control Mean	Treatment Mean	P-value (significance)
Chilili	All herbs (forbs and grasses)	7.8	6.7	0.80
	Forbs	0.1	3.2	0.23
	Grasses	7.7	3.5	0.22
	Bare soil	0	0	0
	Cryptobiotic crust	0.1	0	0.30
	Leaf litter	68.5	57.1	0.18
	Wood chips	**0**	**11.9**	**0.002**
Kelly	**All herbs (forbs and grasses)**	**16.3**	**40.1**	**0.002**
	Forbs	**8.5**	**29.7**	**0.004**
	Grasses	7.8	10.3	0.53
	Bare soil	**21.5**	**5.1**	**0.001**
	Cryptobiotic crust	**6.4**	**0.1**	**0.02**
	Leaf litter	28.5	15.7	0.12
	Wood chips	**0**	**27.7**	**<0.0001**

Site	Parameter	Control Mean	Treatment Mean	P-value (significance)
Vigil	**All herbs (forbs and grasses)**	**7.0**	**44.4**	**<0.0001**
	Forbs	**2.5**	**24.9**	**<0.0001**
	Grasses	**4.4**	**19.5**	**0.0009**
	Bare soil	**30.8**	**8.7**	**0.0004**
	Cryptobiotic crust	**9.5**	**0.2**	**0.003**
	Leaf litter	25.5	12.3	0.11
	Wood chips	**0**	**22.8**	**<0.0001**
Wester	**All herbs (forbs and grasses)**	**4.0**	**13.3**	**0.005**
	Forbs	1.4	0.8	0.40
	Grasses	**2.7**	**12.5**	**0.002**
	Bare soil	0.5	0.2	0.50
	Cryptobiotic crust	2.1	0.4	0.40
	Leaf litter	61.0	67.7	0.40
	Wood chips	**0**	**8.3**	**0.006**

Note: All tests were with sample sizes of 36; p-values of less than 0.05 represent significant differences. Parameters in bold represent those with significant differences between paired plots. Refer to Figure 2.47for graphical illustrations of differences in mean values.

Total herbaceous vegetation canopy cover was significantly greater on all treated plots at all sites except for Chilili in 2013 (Figure 2.47 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 2013 following thinning treatments. The amount of herbaceous vegetation cover on treatment plots has tended to increase even more relative to control plots through 2013 (Figure 2.47 a–d). Forb canopy cover was significantly higher on the treated plots at both of the piñon/juniper sites in 2013, but not at either of the ponderosa pine sites (Figure 2.47, f–h). Forb cover at both the Kelly and Vigil sites was considerably higher in 2013 than it was before thinning treatments in 2010 and after treatments in 2011 and 2012. Forb cover on control plots remained relatively constant over the three-year period at the Vigil site, but also increased on the control plot at the Kelly site in 2013. The majority of forb species were summer annual plants that grew on disturbed soils and wood chips. All of the dominant species were native; no exotic invasive forb species were found. An example of the forb growth at the Vigil treatment plot in 2013 is provided in Figure 2.48.

Figure 2.48. Forbs growing on disturbed soils and wood chips at the Kelly site treatment plot, October 2013. The visible dominant species is native annual forb *Chenopodium gravolens*.

Grass cover was significantly higher only on the treated plot at the Vigil piñon/juniper site and at the Wester ponderosa pine site in 2013 (Table 2.9; Figure 2.47, i–l). Grass cover increased steadily on the treated plot at the Vigil site from 2010 to 2013, but remained similar on the control plot. Grass cover was significantly higher on the treated plot at the Wester ponderosa pine site even before thinning treatments in 2010. However, in 2013 grass cover on the treatment plot continued to increase over 2012, while it declined on the control plot from 2012, indicating a positive response to the thinning treatment. 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 Vigil site is shown in Figure 2.49.

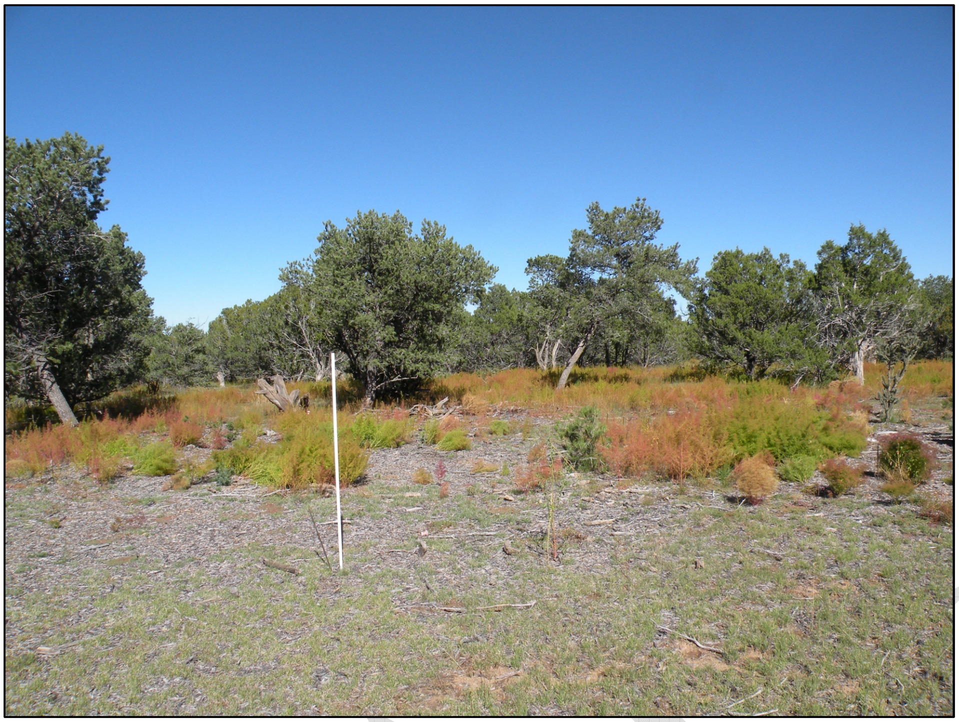
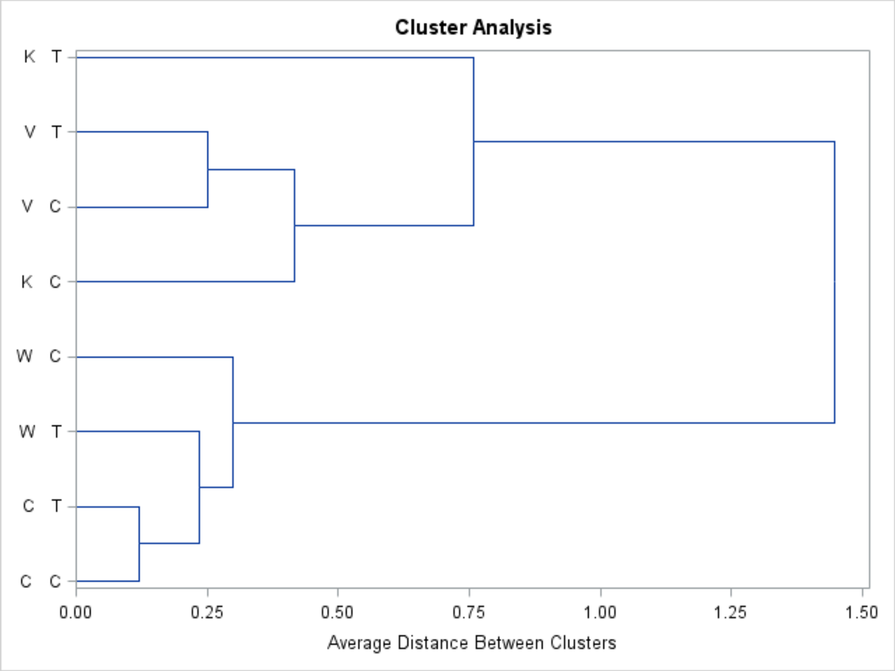


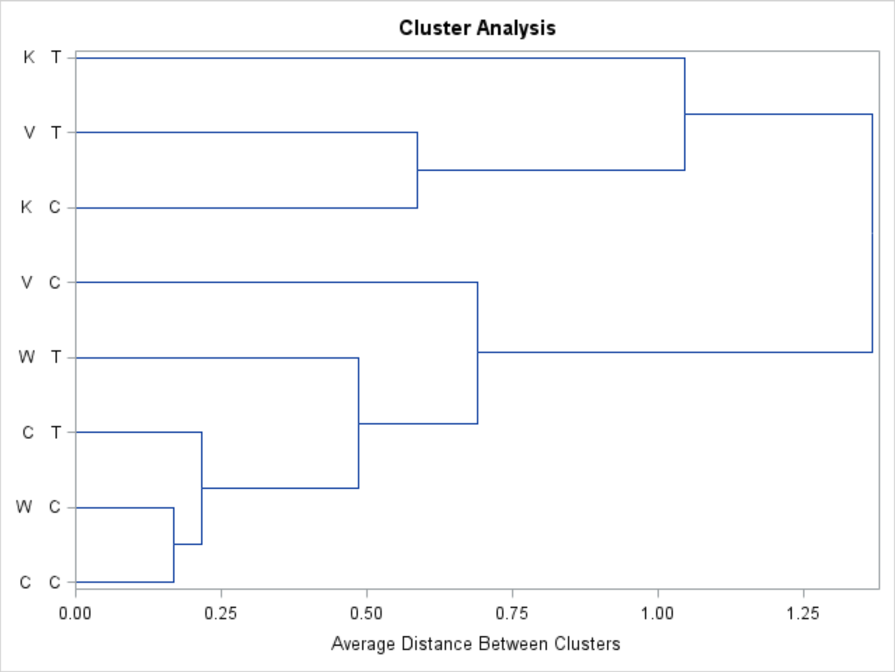
Figure 2.49. Perennial blue grama growing through wood chips (foreground) at the Kelly piñon/juniper site treatment plot in 2013. Note that the blue grama had been grazed down by livestock.

Bare soil was significantly higher on control plots than treated plots at the two piñon/juniper sites, but not at the two ponderosa pine sites in 2013, or in 2010, 2011, or 2012 (Figure 2.47, m–p; see Table 2.9). Bare soil is an inverse ground cover to leaf litter and wood chips, and 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. Cryptobiotic or biotic soil crust cover 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 2013 (see Table 2.9; Figure 2.47, q–t). 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. Leaf litter cover was not significantly different between any of the paired control and treatment plots in 2013, but it 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 Table 2.9; Figure 2.47, u–x). 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. Wood chips remained significantly higher on all treated plots in 2012, as in 2011 following thinning treatments (see Table 2.9; Figure 2.47, y–bb). 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. Wood chip cover remained the same and even increased slightly but not significantly at the Kelly and Vigil sites, indicating a shift in relative ground surface cover with other cover categories or redistribution of wood chips by grazing livestock, since no new wood chips have been added since 2010.

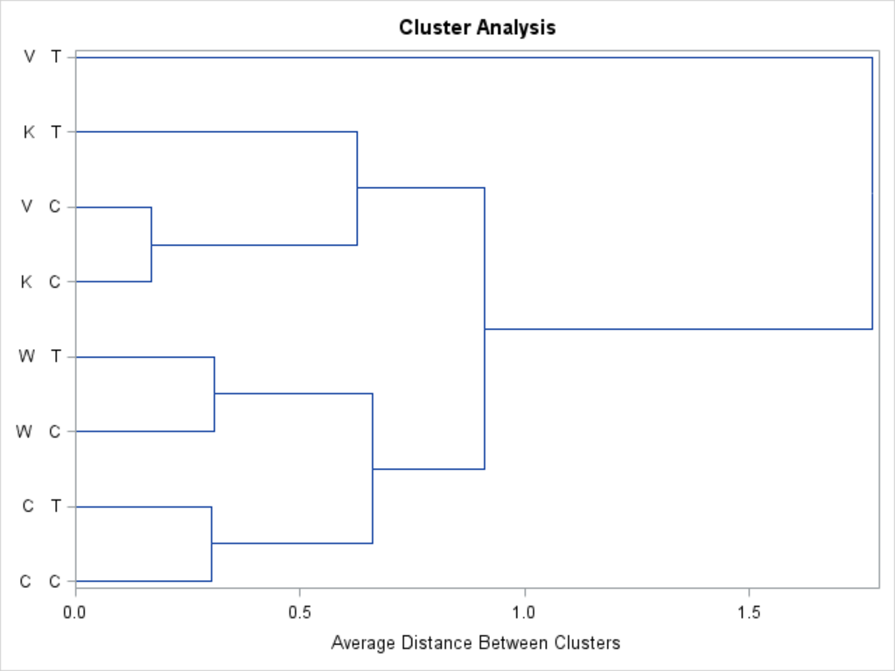
Measurements of herbaceous vegetation on the thirty-six 1-m² (10.8-square-foot) quadrats also provided information on the canopy cover of each plant species per quadrat. The similarity of plant species composition among all of the study plots since 2010 was evaluated with the analytical method called 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 2013 are presented in Figure 2.50, a–d. Cluster analysis shows that in 2010 (see Figure 2.50, 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 2.50, 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. 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

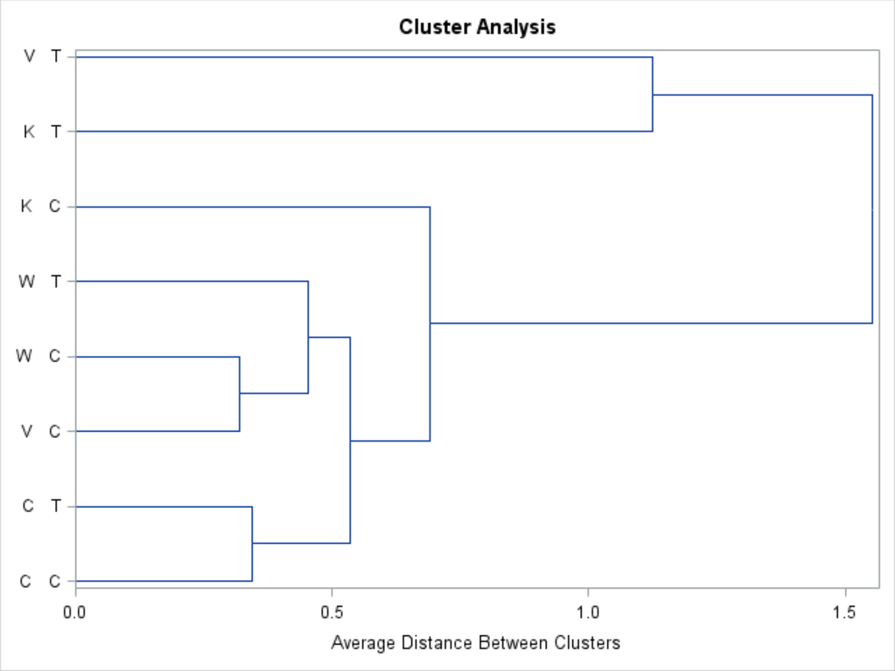
a. 2010.


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

b. 2011.



c. 2012.



d. 2013.

Figure 2.50. 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.

Results of vegetation and ground cover monitoring showed that forest thinning did affect the physical structure of the woodland 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 (see wildlife camera results below). The removal of grass and forb canopies by livestock undoubtedly reduced canopy cover of those plants that were measured in October 2012 and 2013 from all of the treatment plots. Therefore, actual herb canopy covers 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 three years following thinning treatments. As vegetation adjusts to the removal of some trees and the effects of wood chips decomposing on the ground surface, more changes in vegetation and ground cover features are likely in years to come as a response to tree thinning.

2.8 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) in 2008, 2009, 2010, 2011, 2012, and 2013 for three consecutive days on each study plot.

2.8.1 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 2013 revealed that more birds were found on all treated plots than on control plots in both the spring and fall, except for at the Wester site during the fall (Figure 2.51). Numbers of bird species were also always higher (or equal at Wester in the fall) on treated plots than control plots in both spring and fall, especially at the two piñon/juniper sites (Figure 2.52).

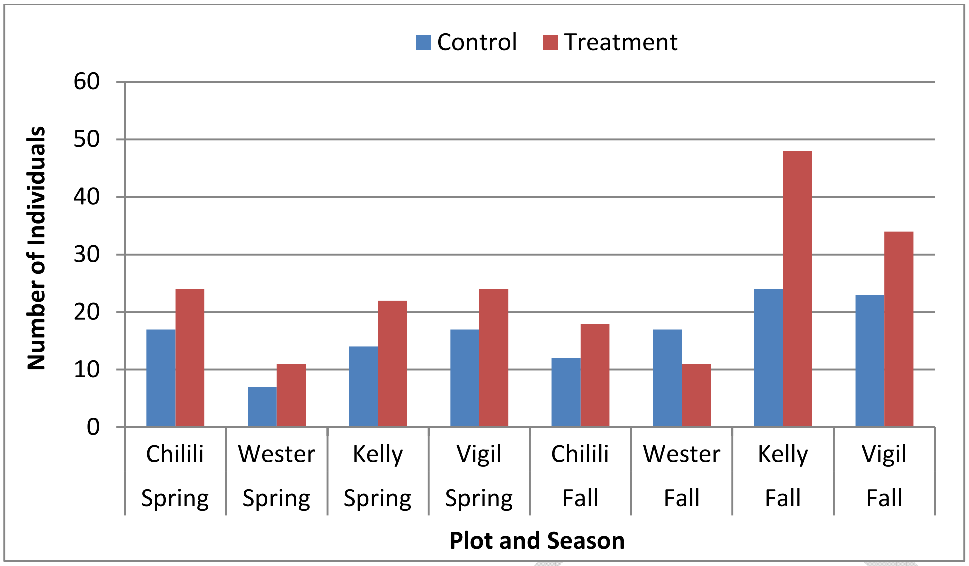


Figure 2.51. Numbers of individual birds recorded from thinning treatment and control plots across the four study sites in both spring and fall 2013.

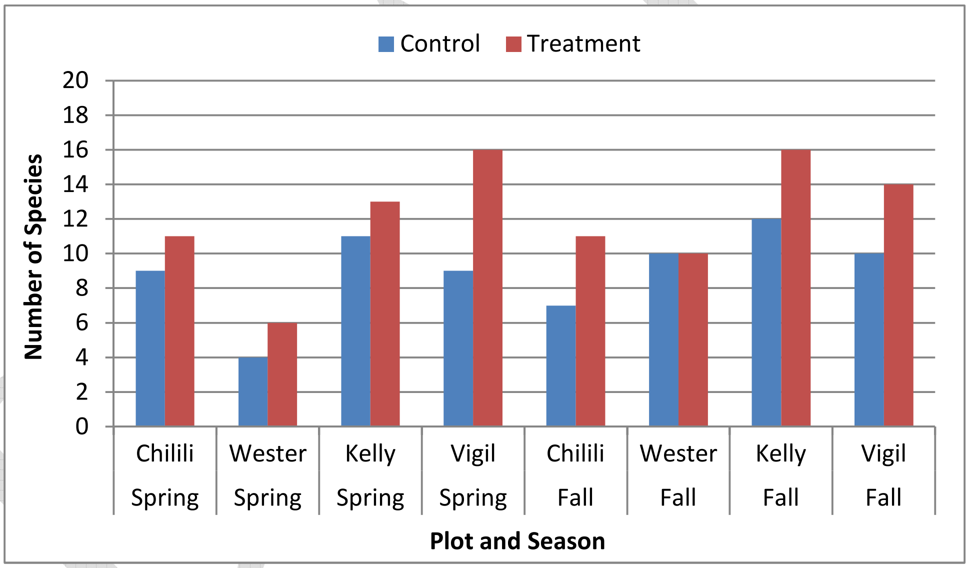
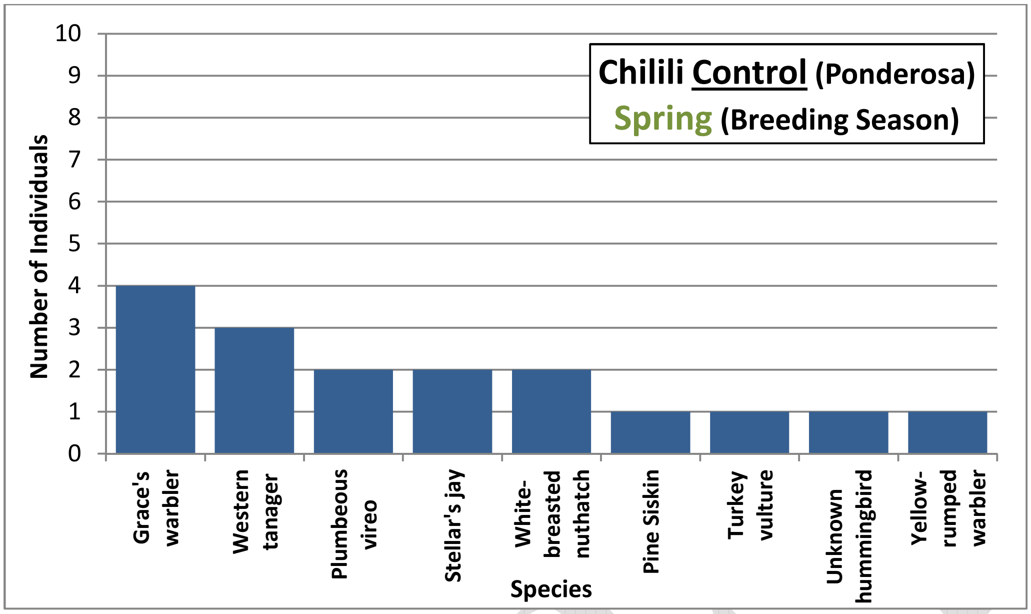
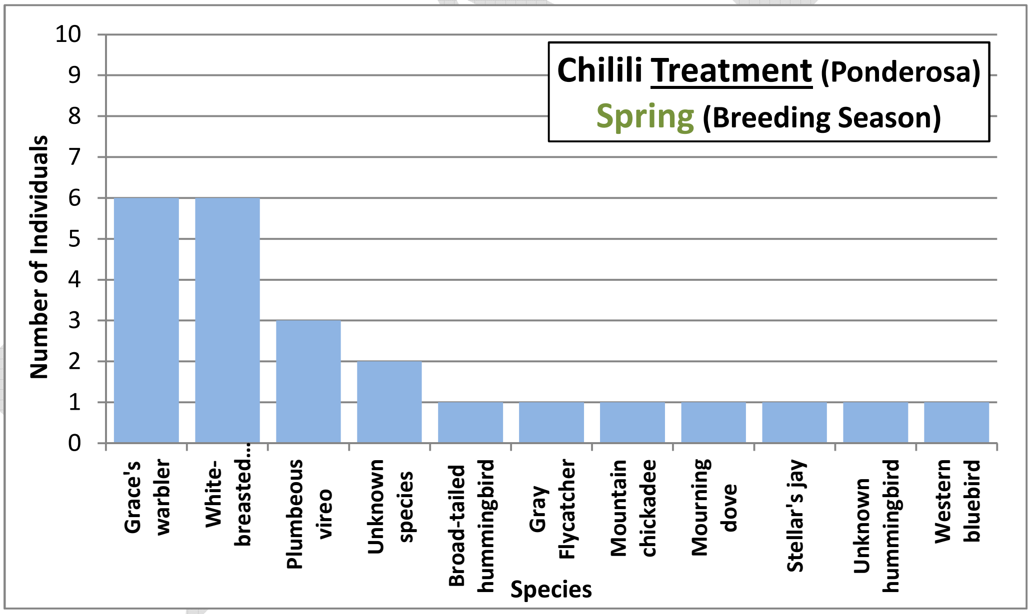


Figure 2.52. Numbers of bird species recorded from thinning treatment and control plots across the four study sites in both spring and fall 2013.

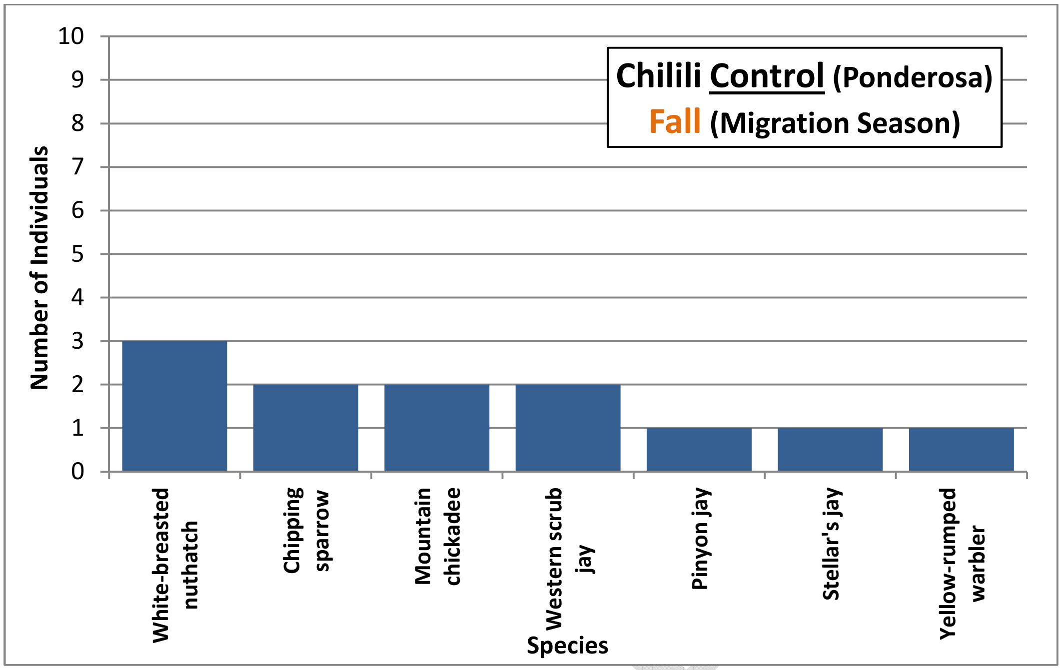
Figure 2.53 presents the species and numbers of individuals of those species summarized in Figure 2.51 and Figure 2.52. Examination of Figure 2.53 shows that in most cases, different species of birds dominate the spring and fall bird communities. Such findings should be expected, given that some of the 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.



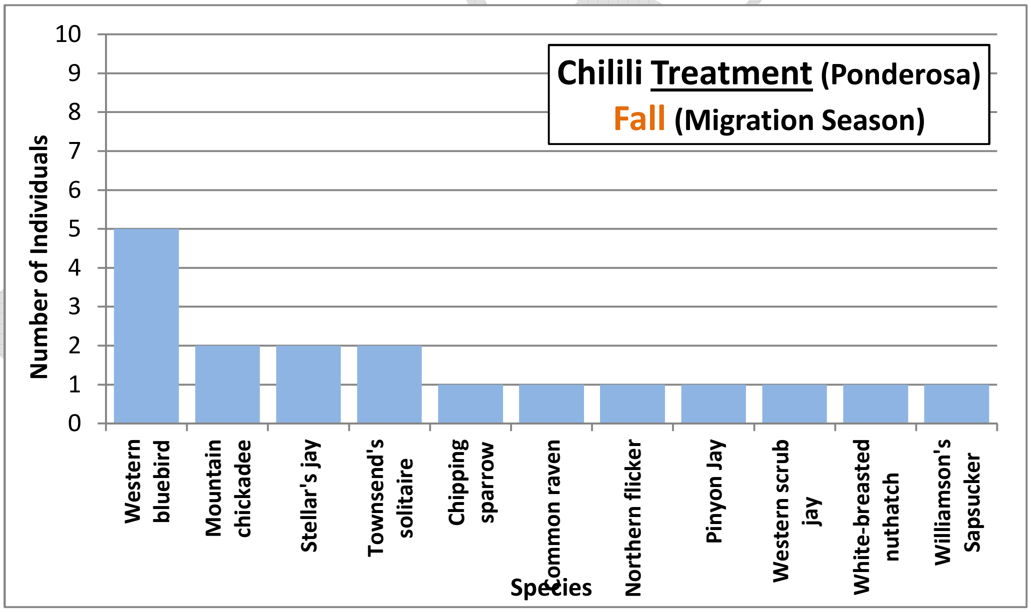
a. Chilili control plot, spring 2013.



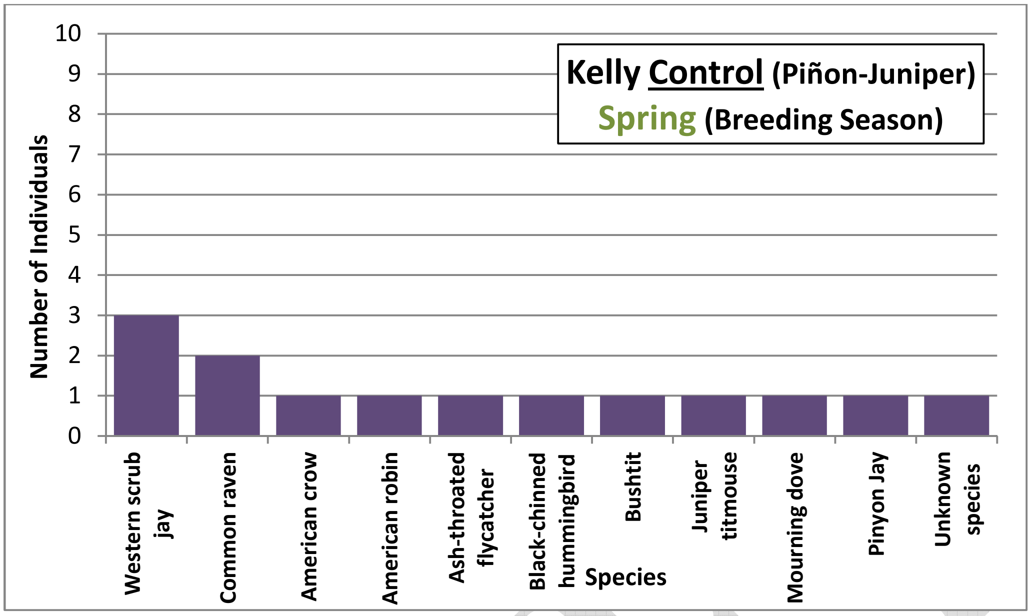
b. Chilili treatment plot, spring 2013.



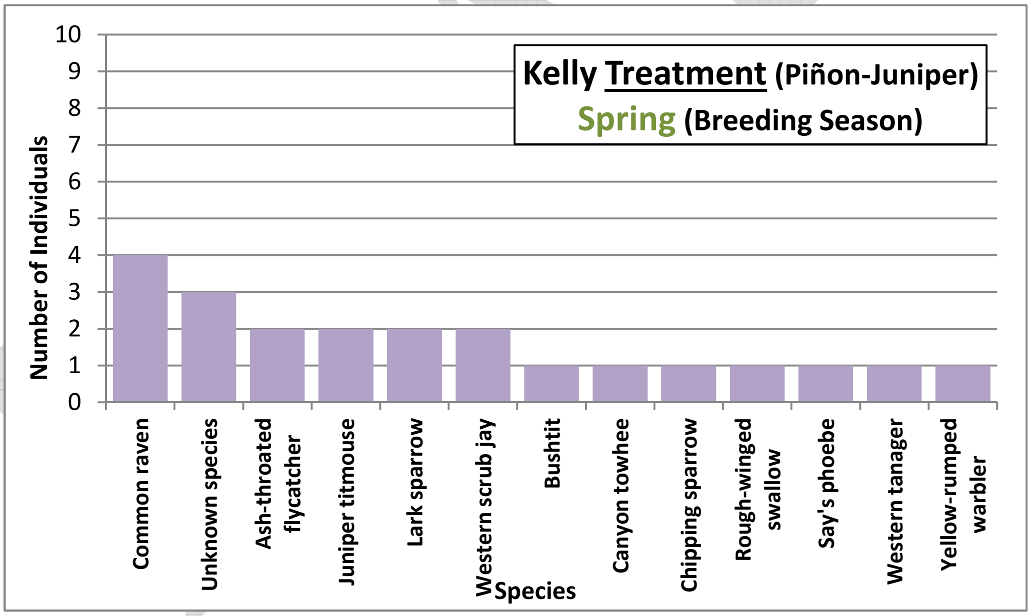
c. Chilili control plot, fall 2013.



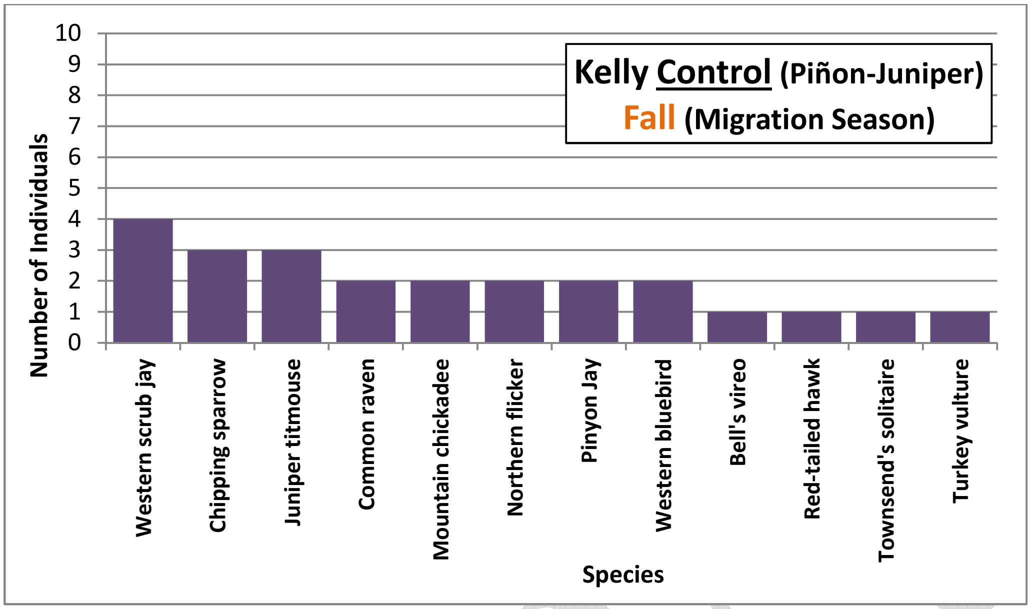
d. Chilili treatment plot, fall 2013.



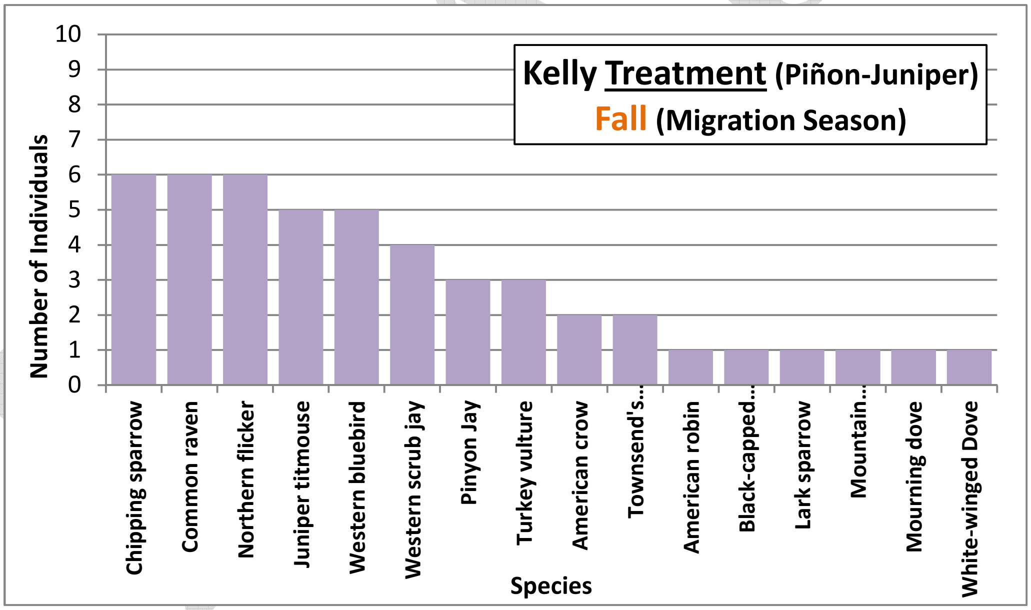
e. Kelly control plot, spring 2013.



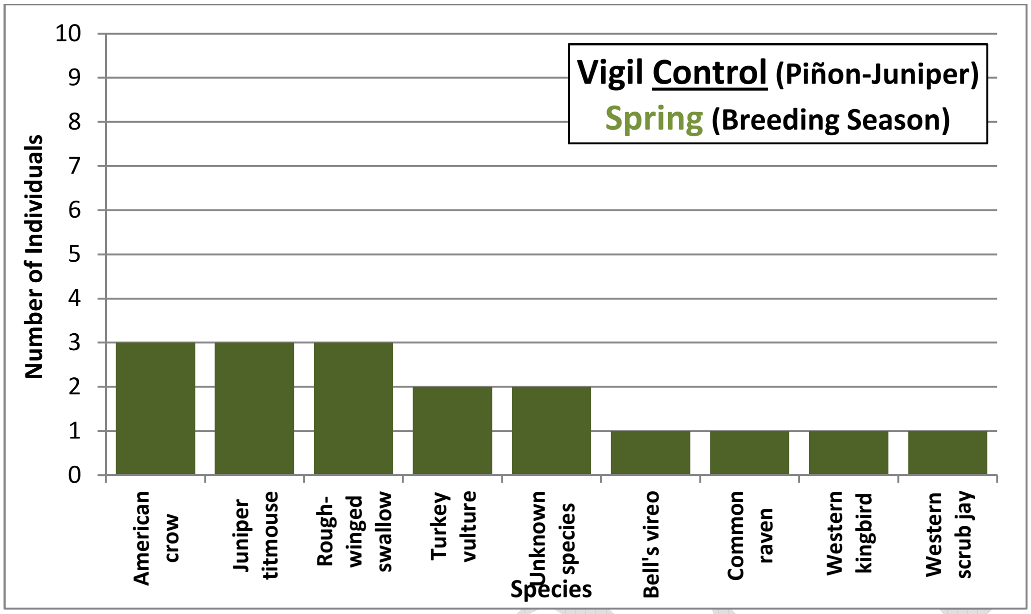
f. Kelly treatment plot, spring 2013.



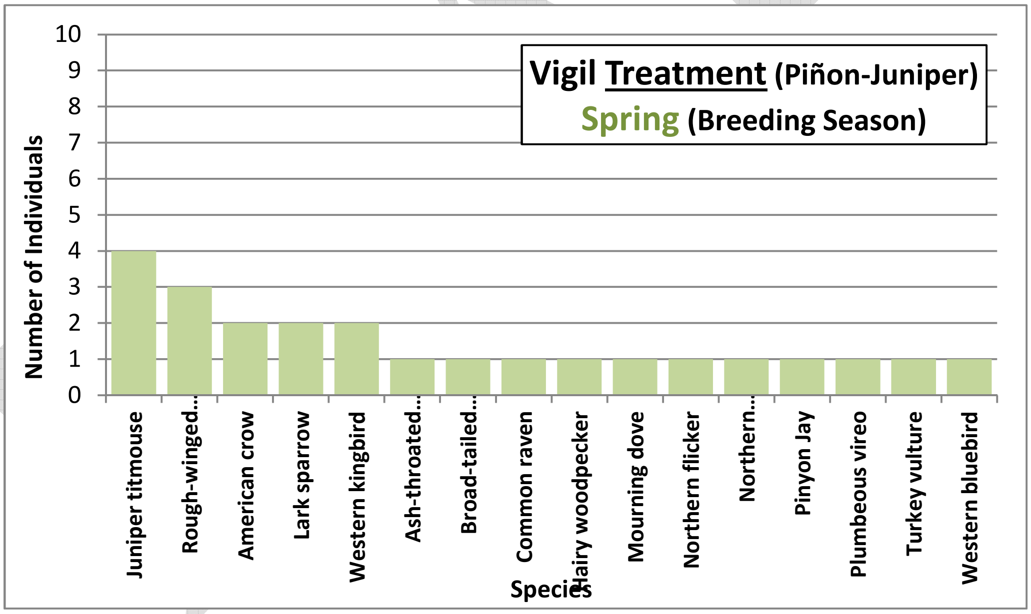
g. Kelly control plot, fall 2013.



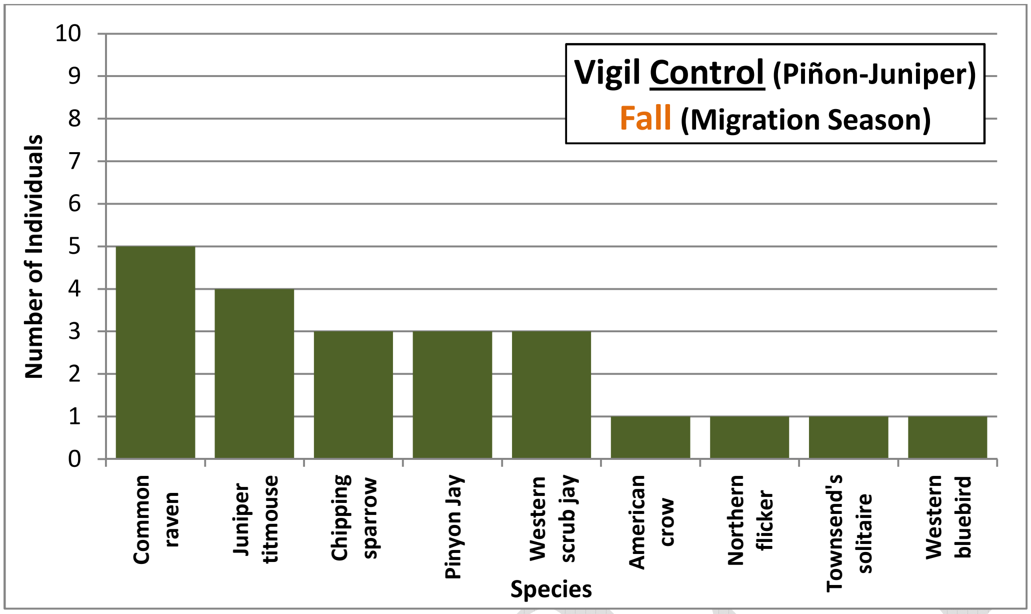
h. Kelly treatment plot, fall 2013.



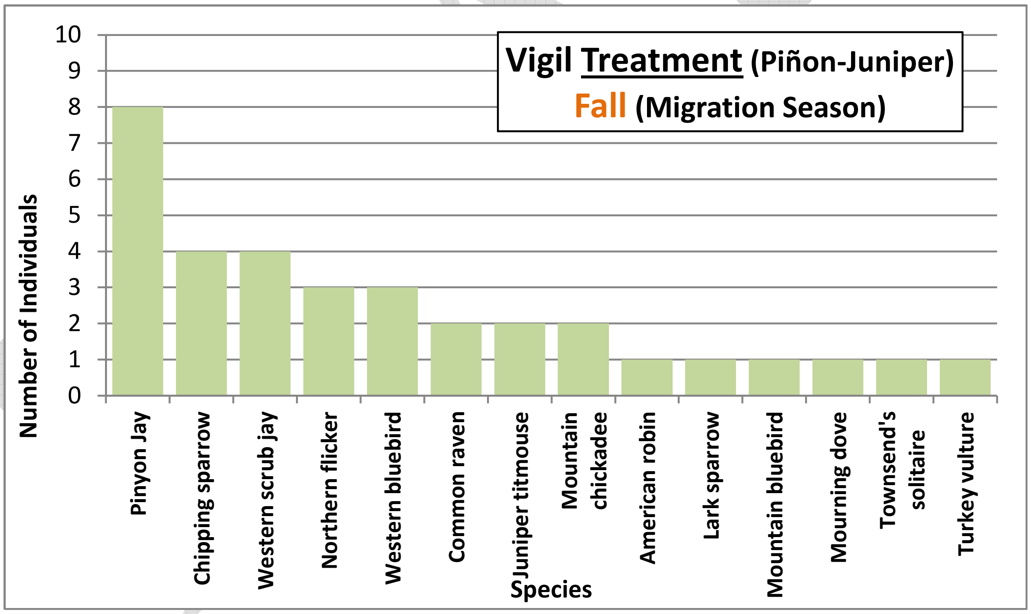
i. Vigil control plot, spring 2013.



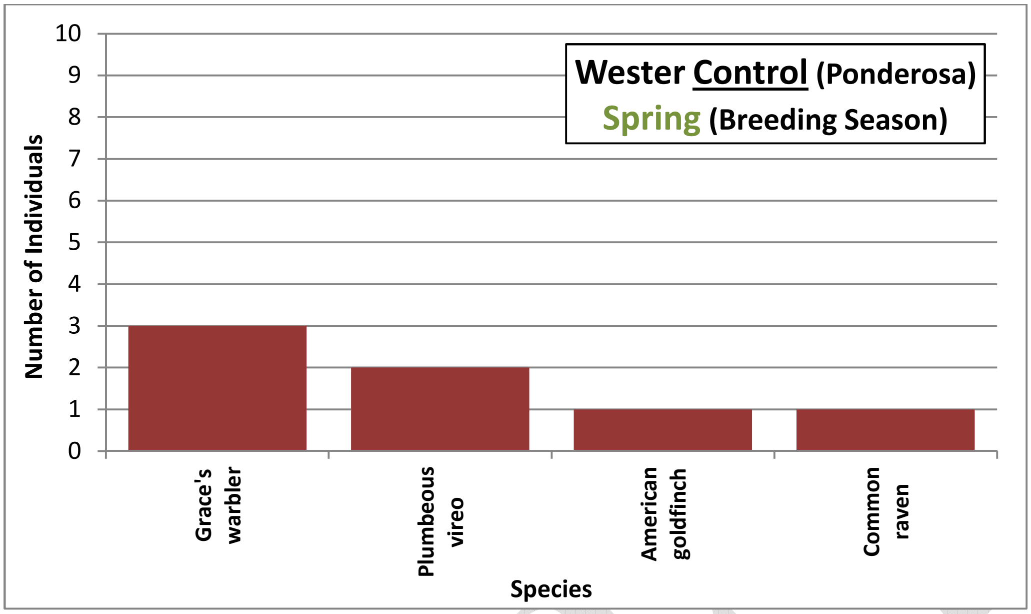
j. Vigil treatment plot, spring 2013.



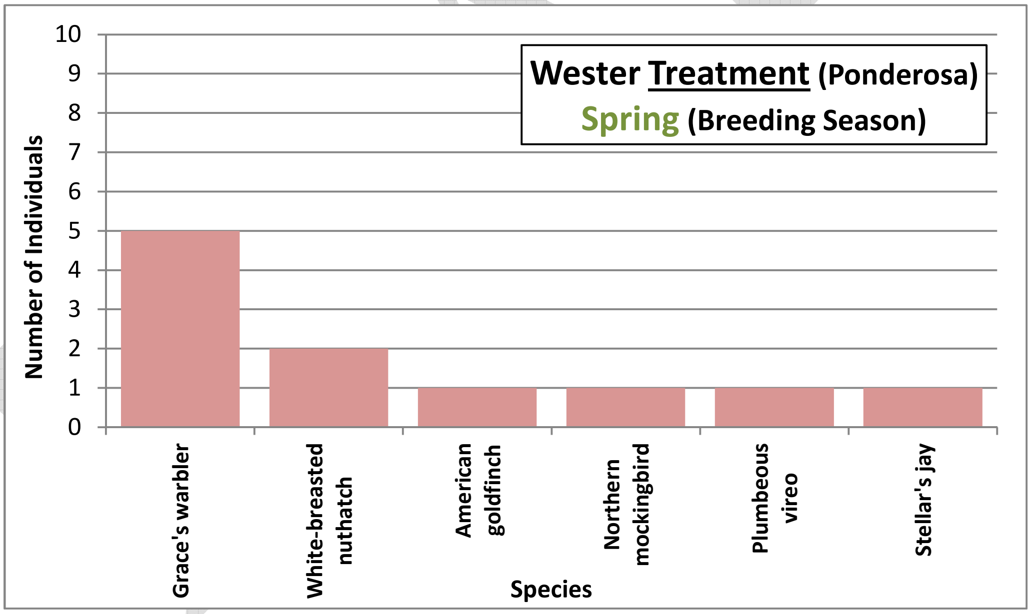
k. Vigil control plot, fall 2013.



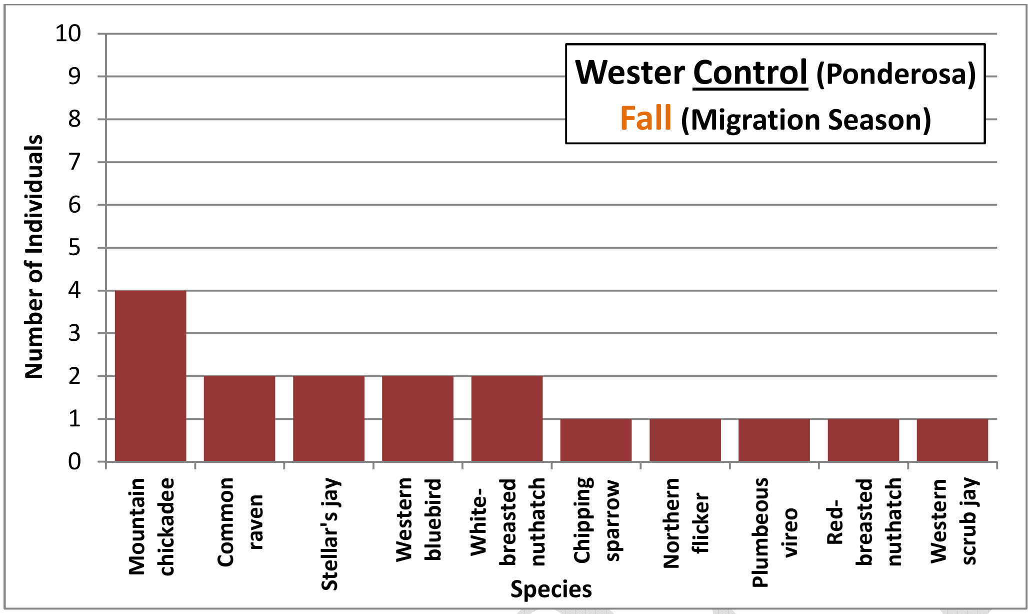
l. Vigil treatment plot, fall 2013.



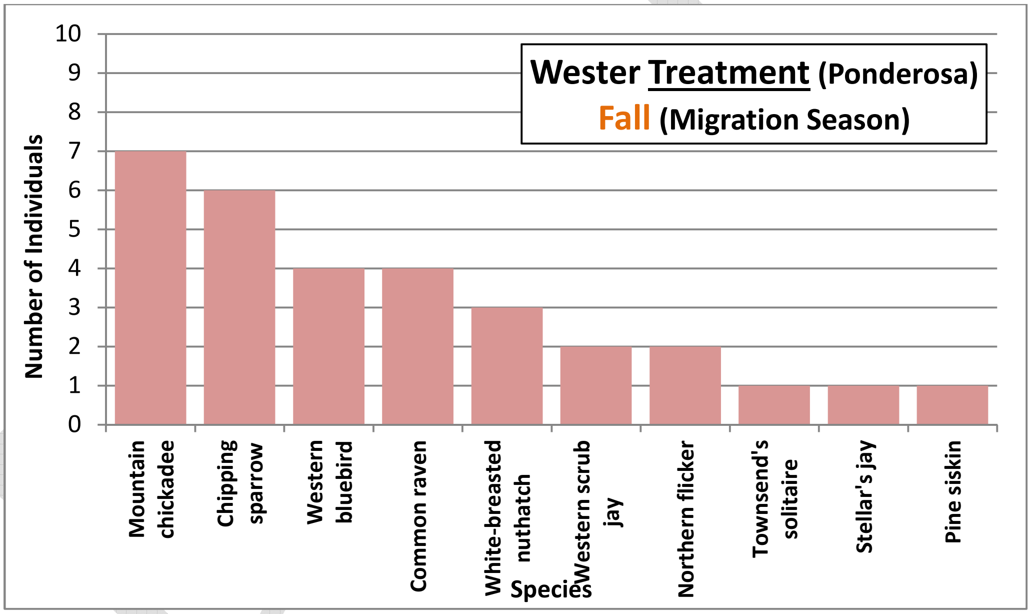
m. Wester control plot, spring 2013.



n. Wester treatment plot, spring 2013.



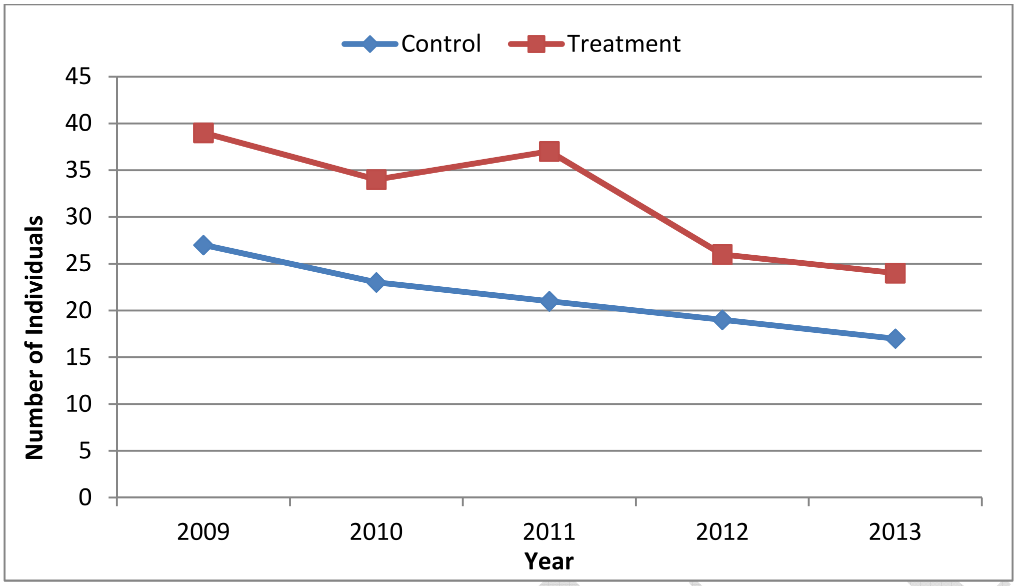
o. Wester control plot, fall 2013.



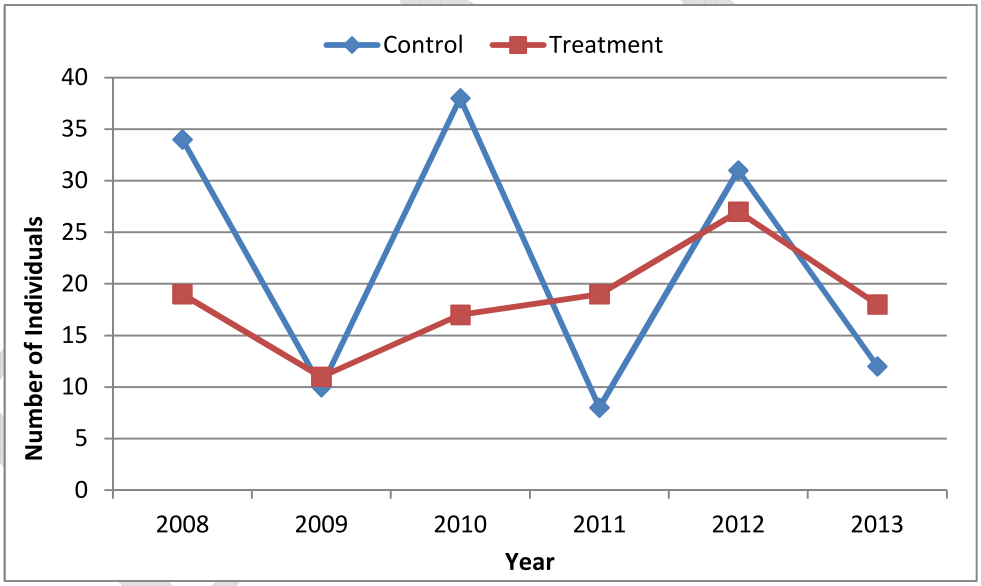
p. Wester treatment plot, fall 2013.

Figure 2.53. Numbers of individuals of each bird species recorded from all control and treatment study plots in 2013, both spring and fall.

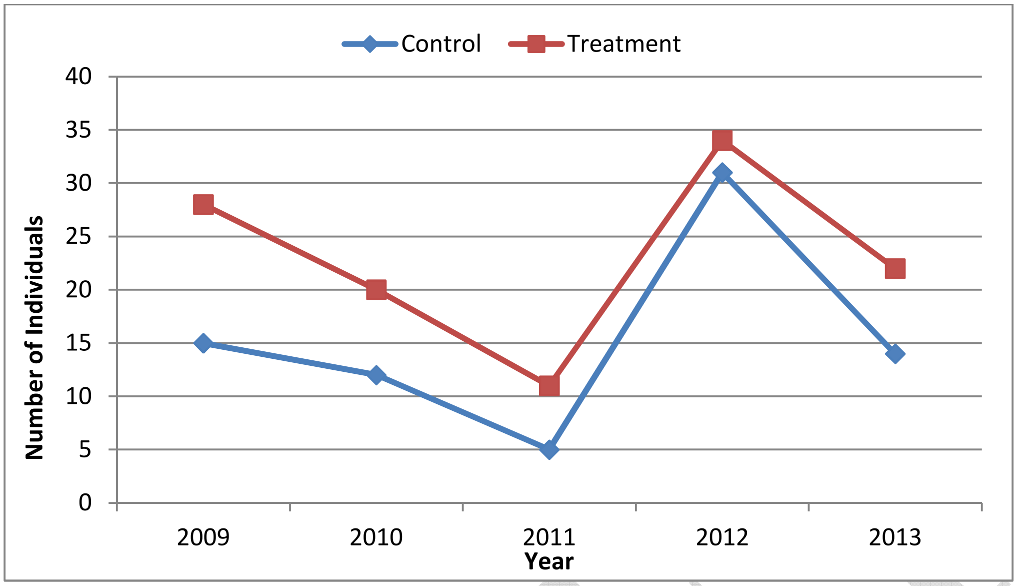
Figure 2.54 presents total numbers of birds from control and treatment plots, both in spring and fall, from all four study sites from fall 2008 through fall 2013. In general, overall bird counts increased over the years up to 2012, but then declined in 2013. Comparisons of treated versus control plots 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.



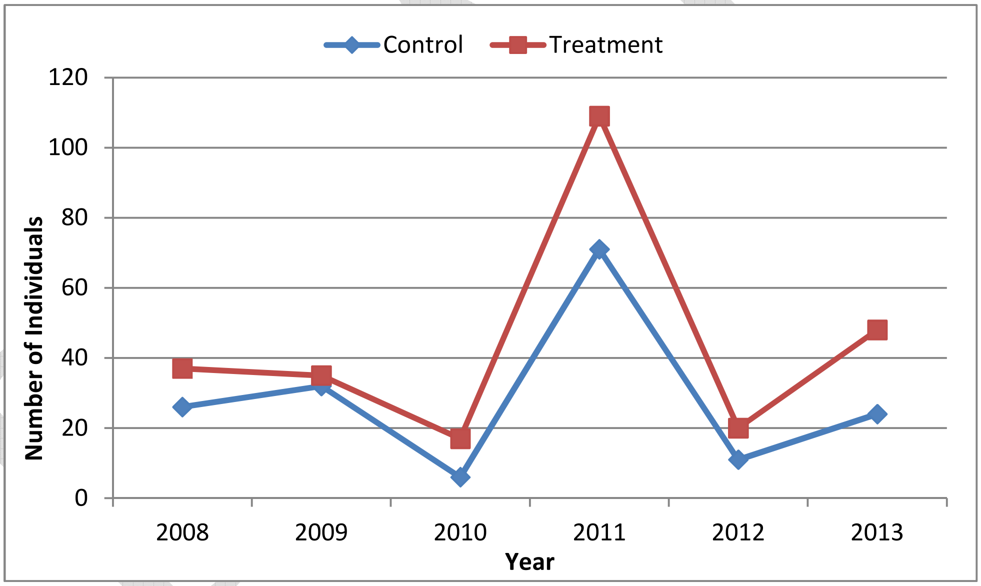
a. Chilili, spring.



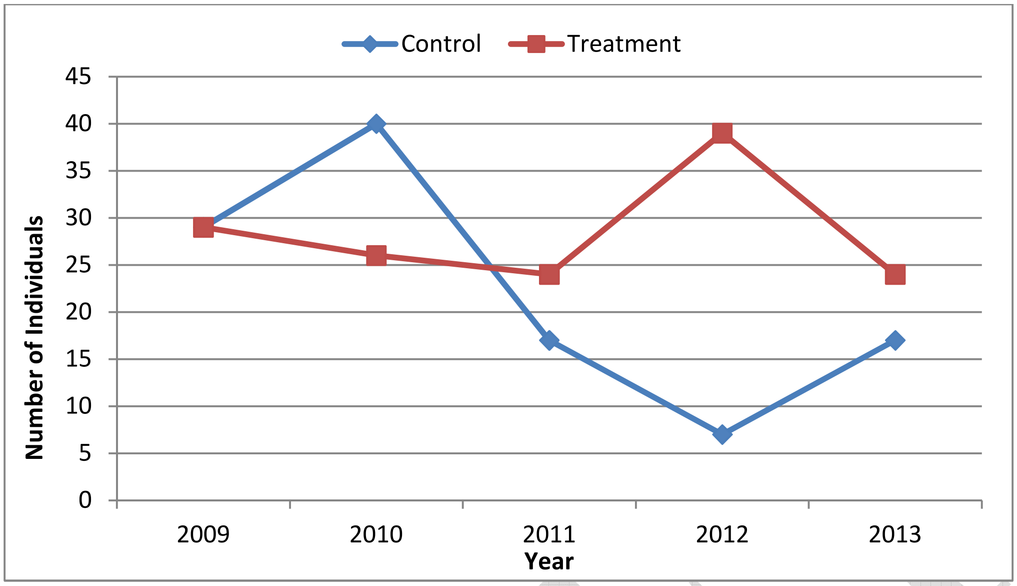
b. Chilili, fall.



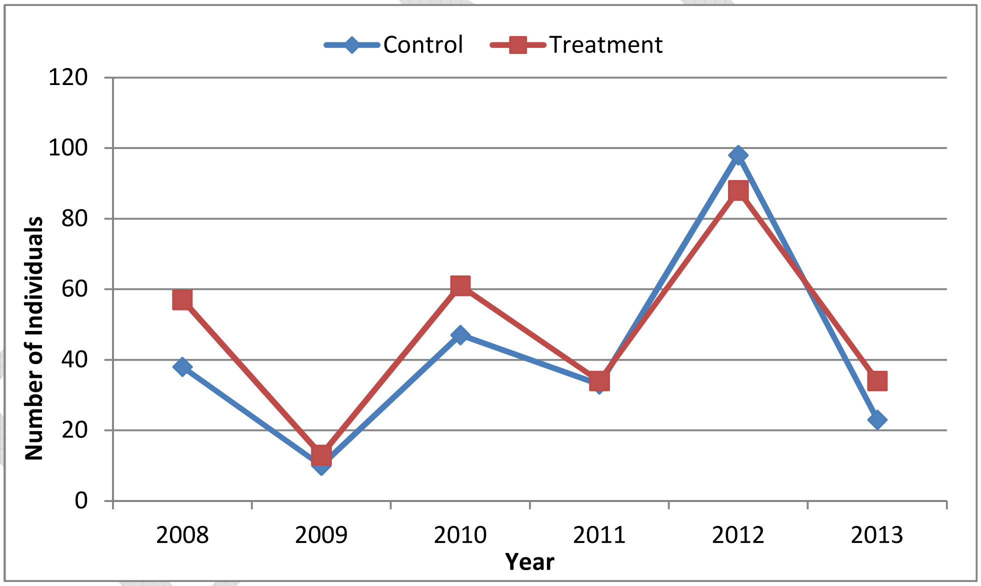
c. Kelly, spring.



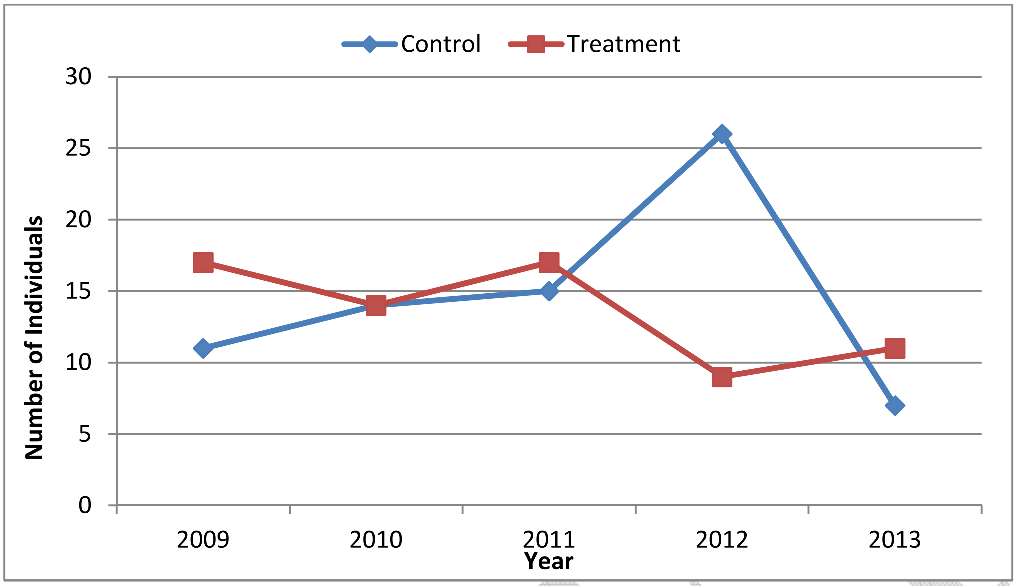
d. Kelly, fall.



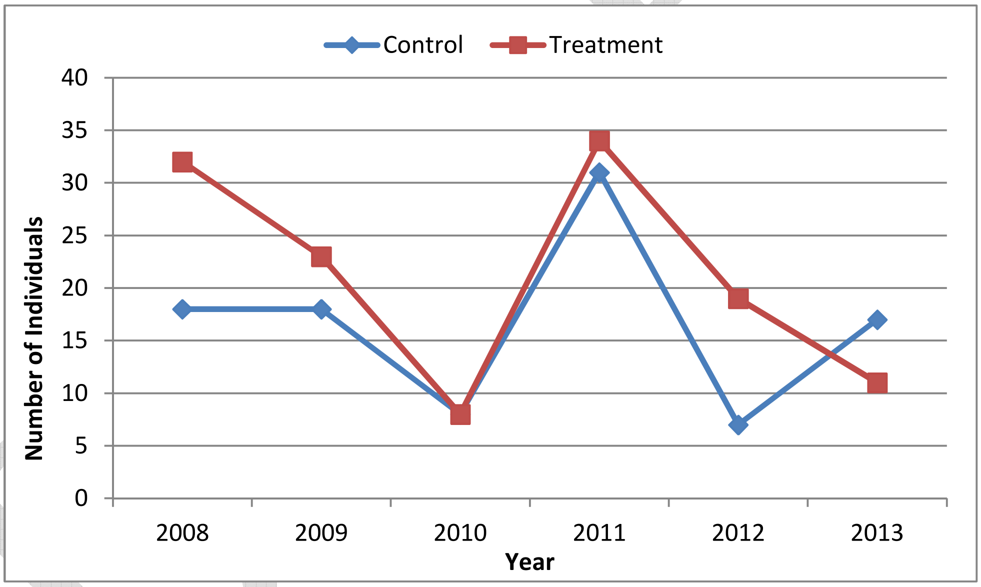
e. Vigil, spring.



f. Vigil, fall.



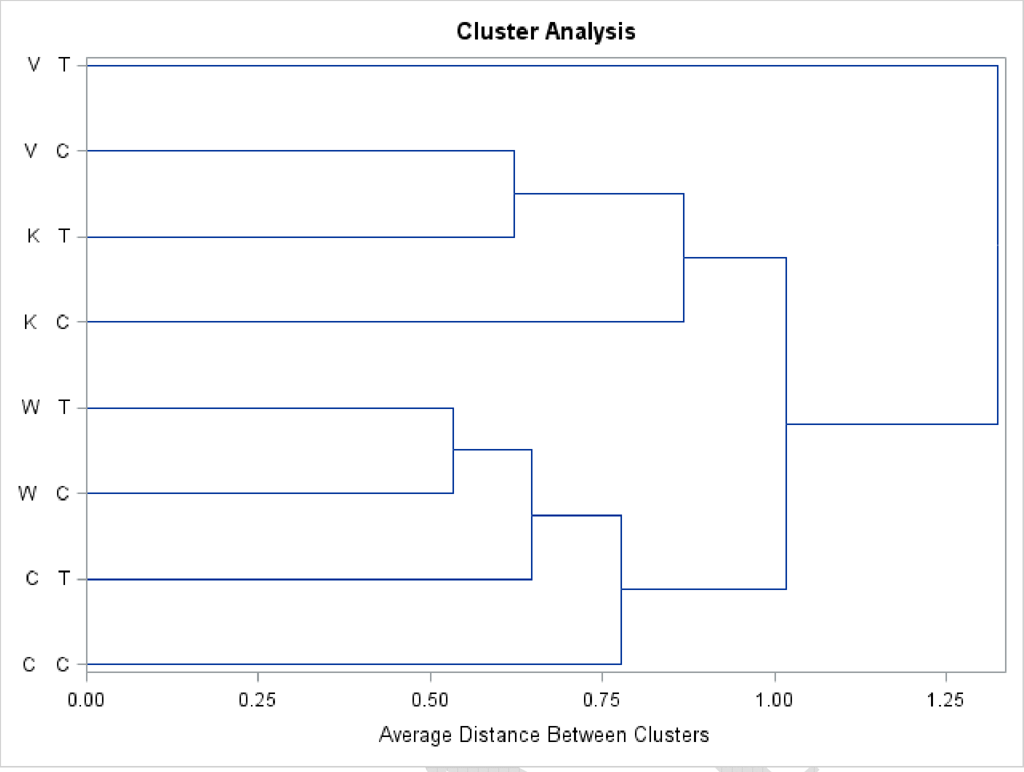
g. Wester, spring.



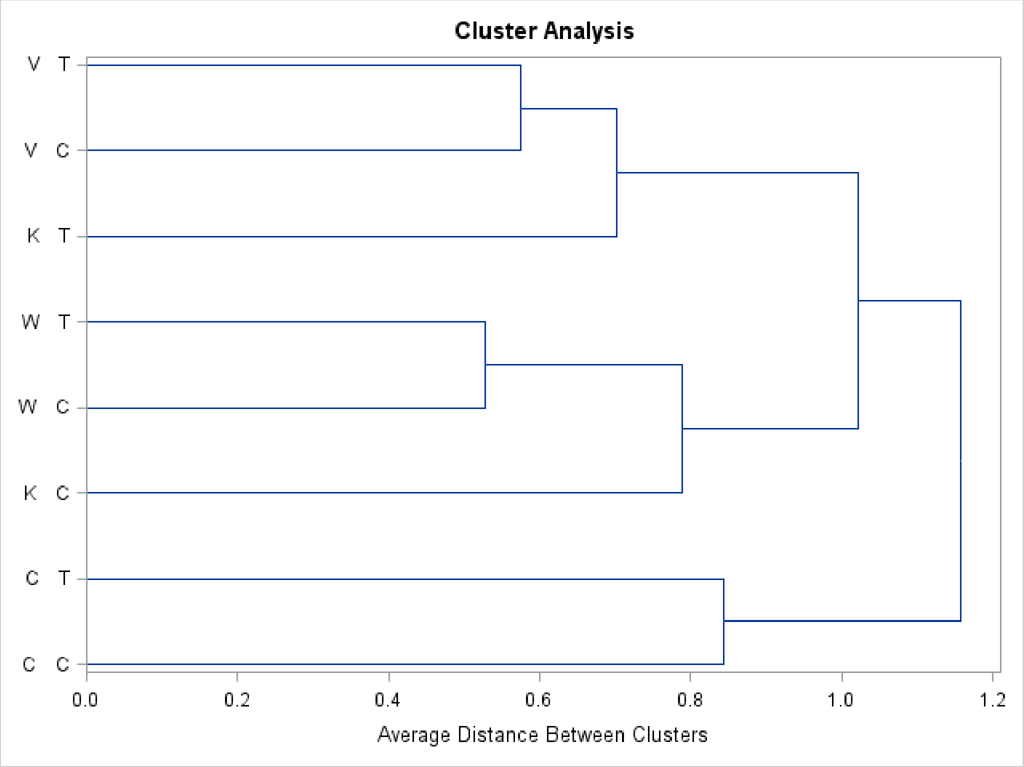
h. Wester, fall.

Figure 2.54. Total numbers of birds from both control and treatment plots at all four study sites, fall 2008 to fall 2013.

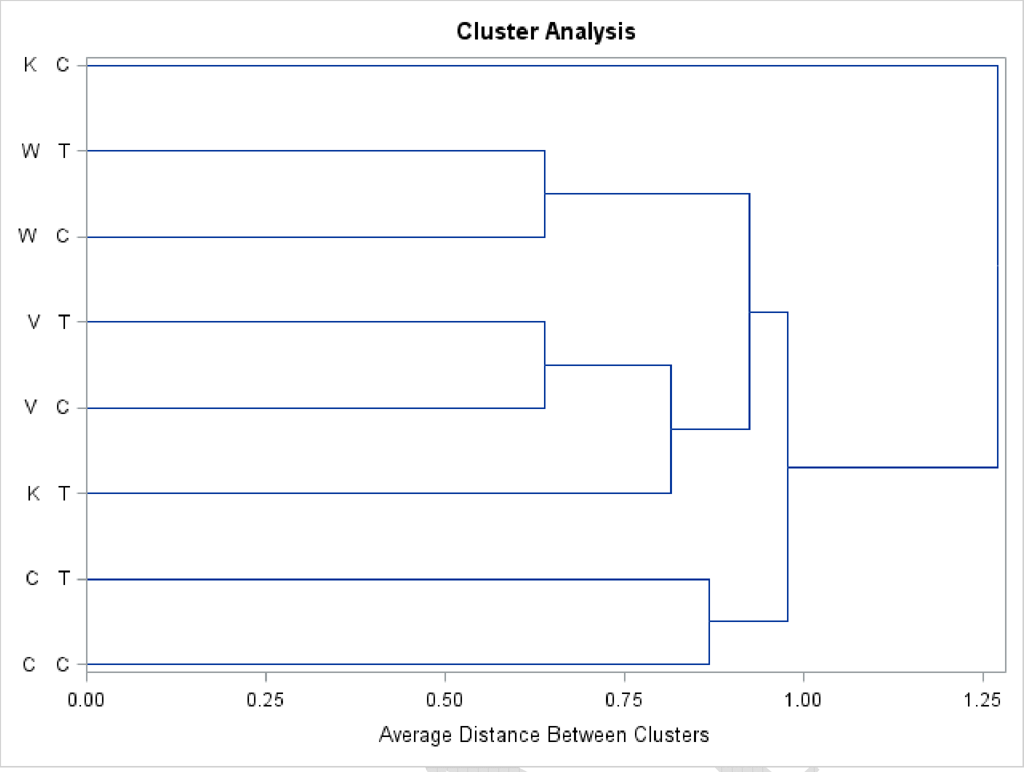
Cluster analysis dendrograms for all sites and plots for the spring and fall sampling periods for 2010 through 2013 are presented in Figure 2.55, a–l. Cluster analysis shows that over the six-year period from 2008 through 2013, bird communities were most similar to each other based on location. The ponderosa 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. As of fall 2013, the bird communities have not shown a strong response to forest thinning treatments; if that had been the case, plots would have grouped together based on treatment status rather than location.



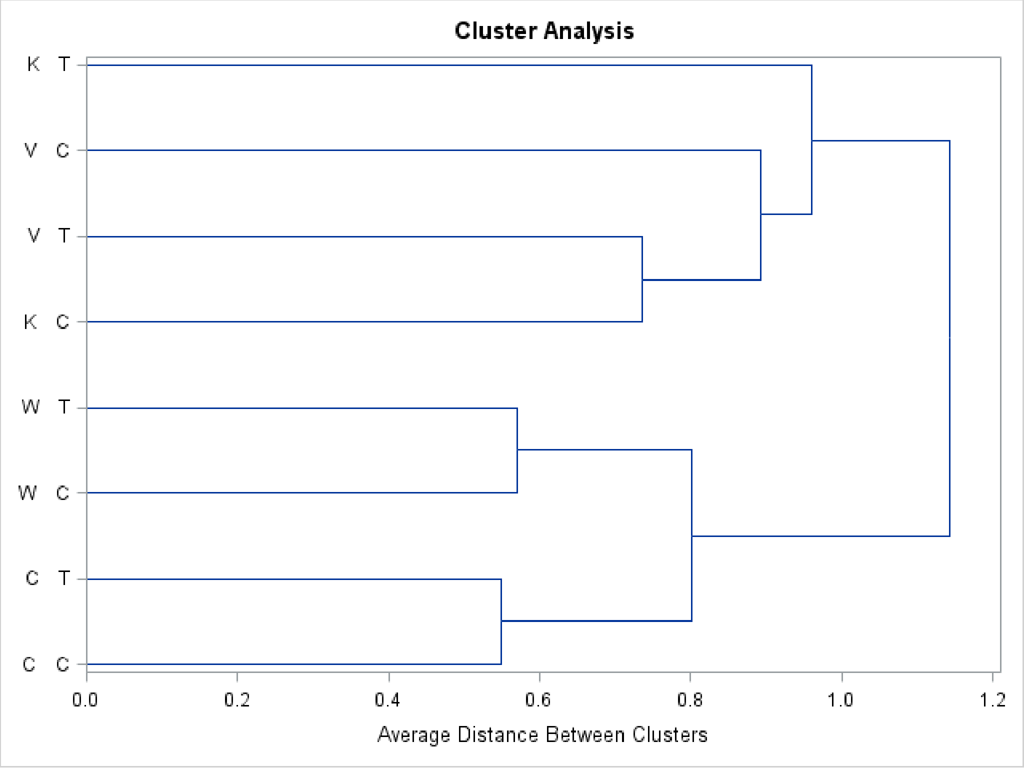
a. 2008, fall.



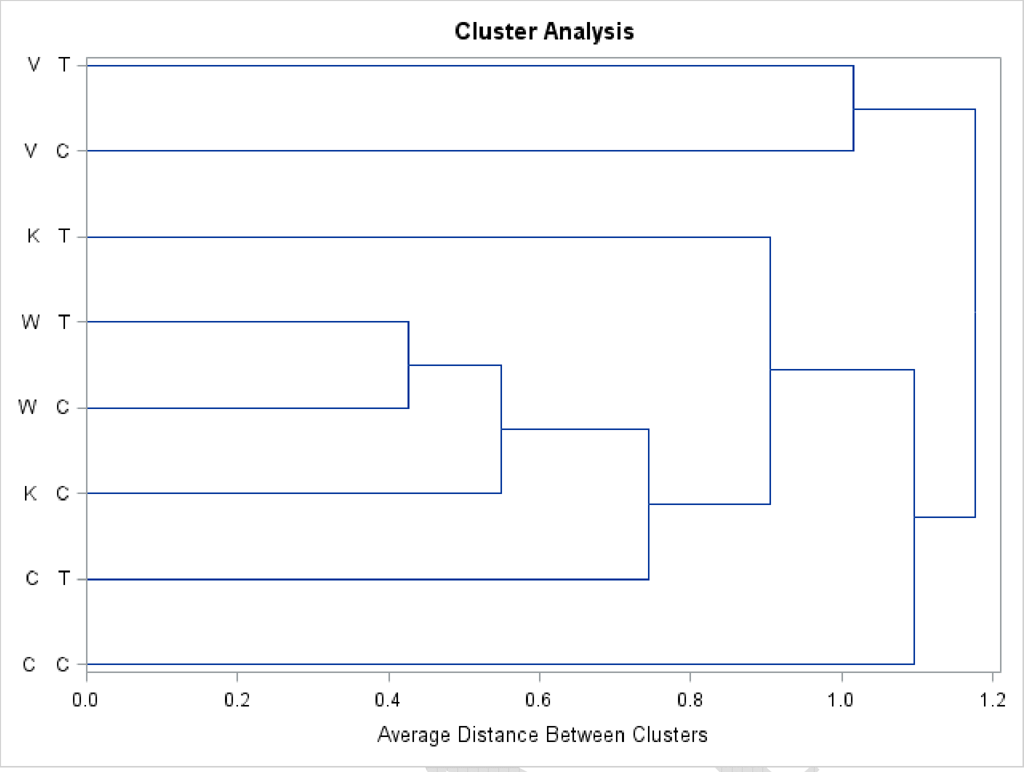
b. 2009, spring.



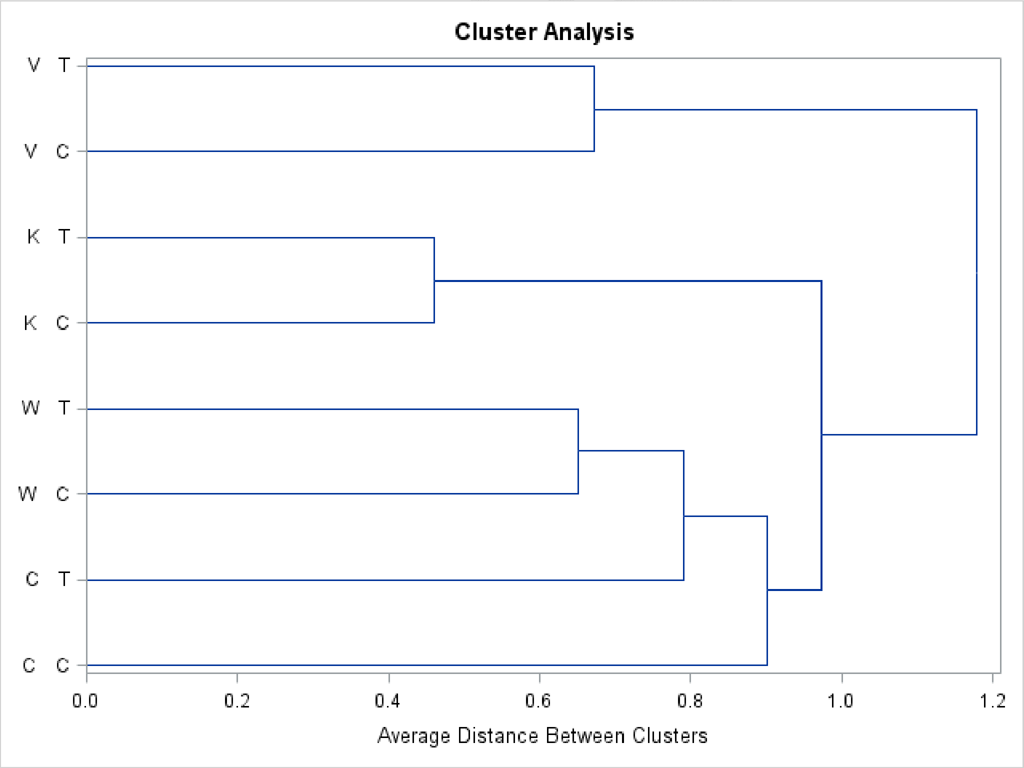
c. 2009, fall.



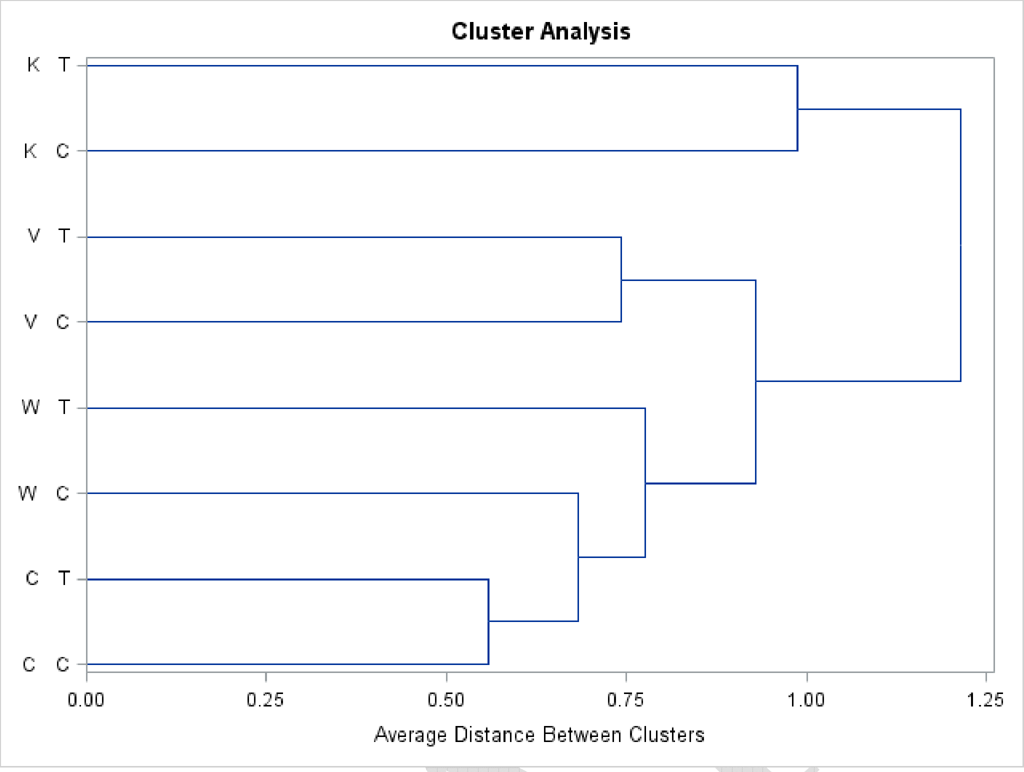
d. 2010, spring.



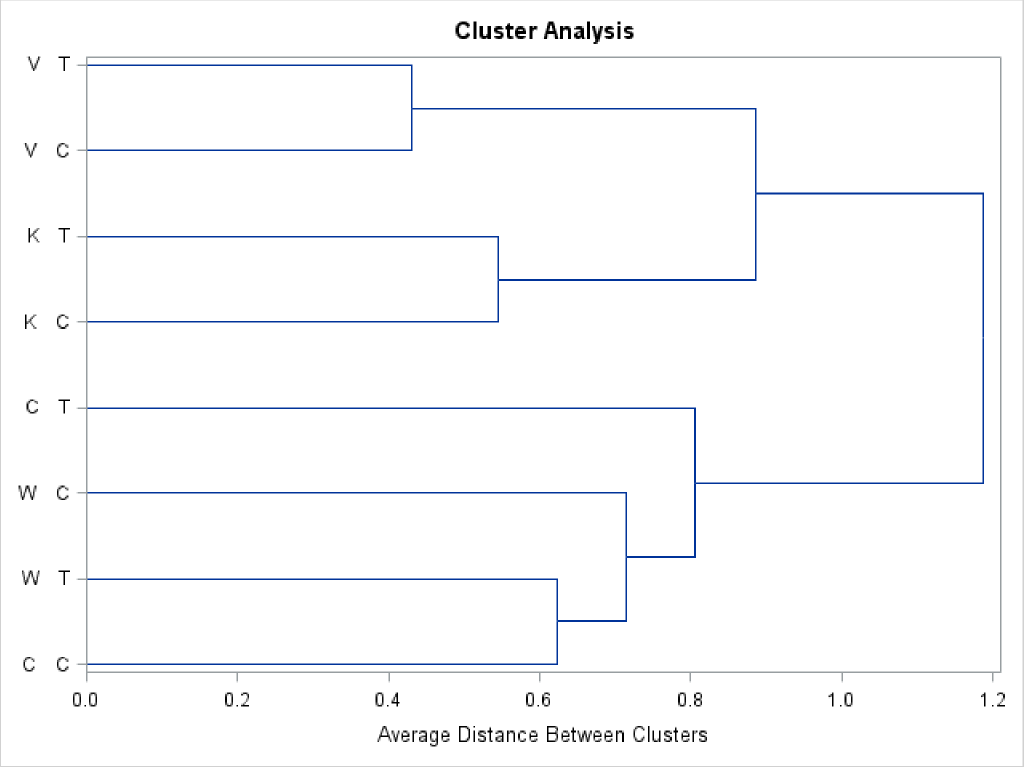
e. 2010, fall.



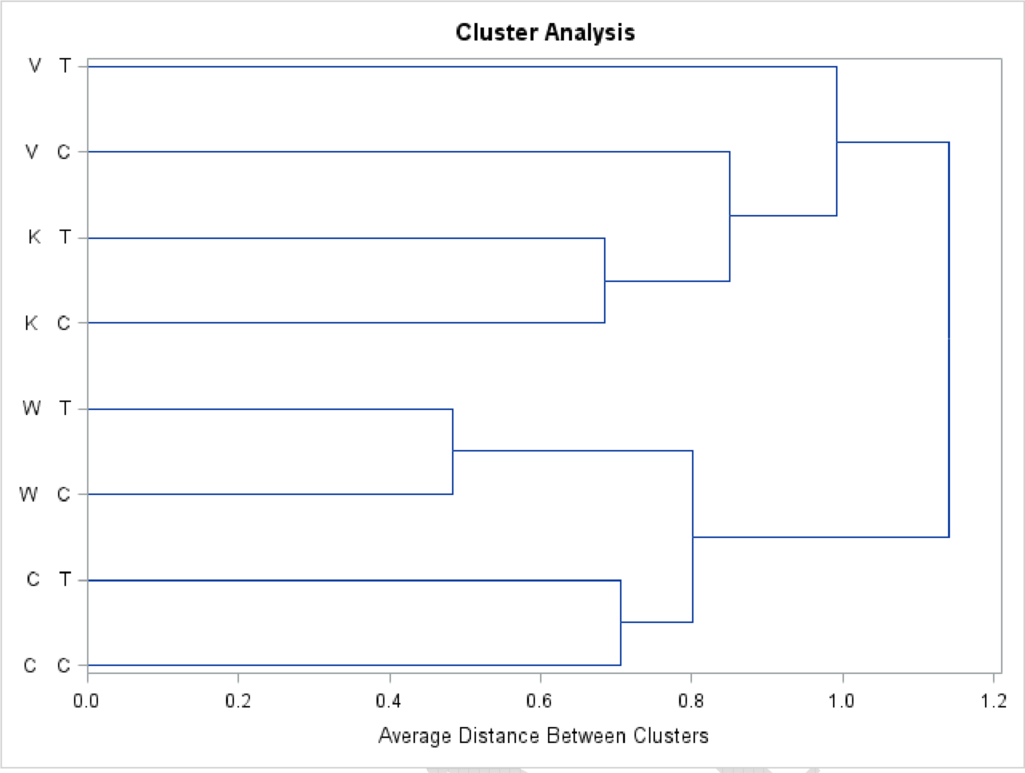
f. 2011, spring.



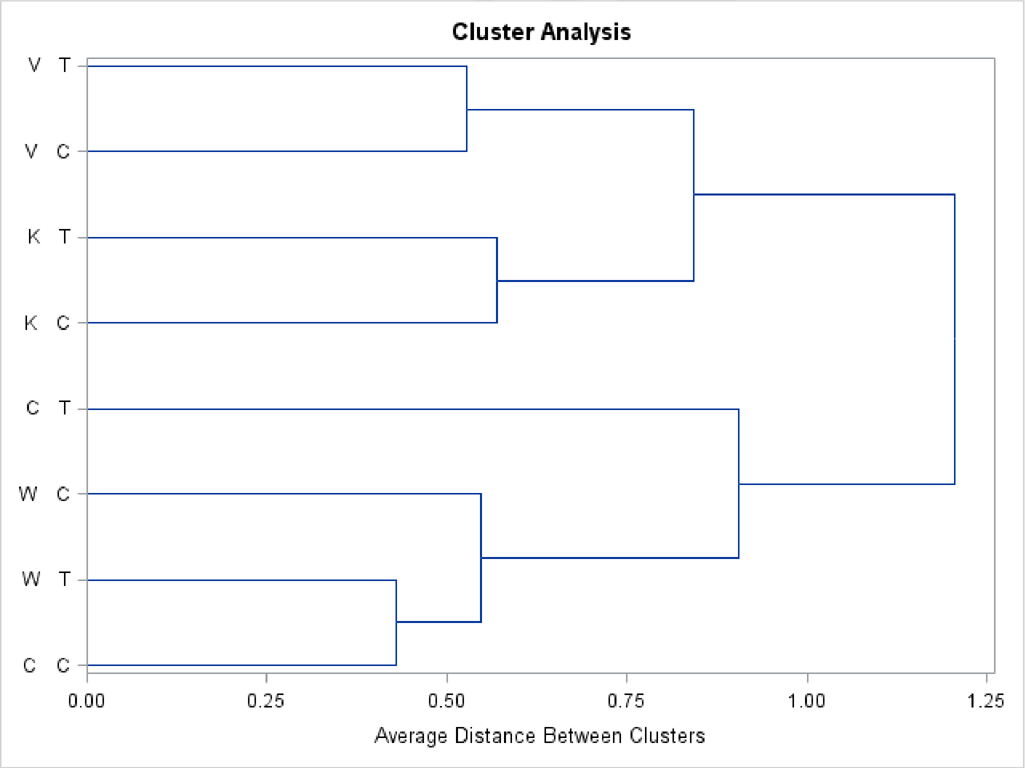
g. 2011, fall.



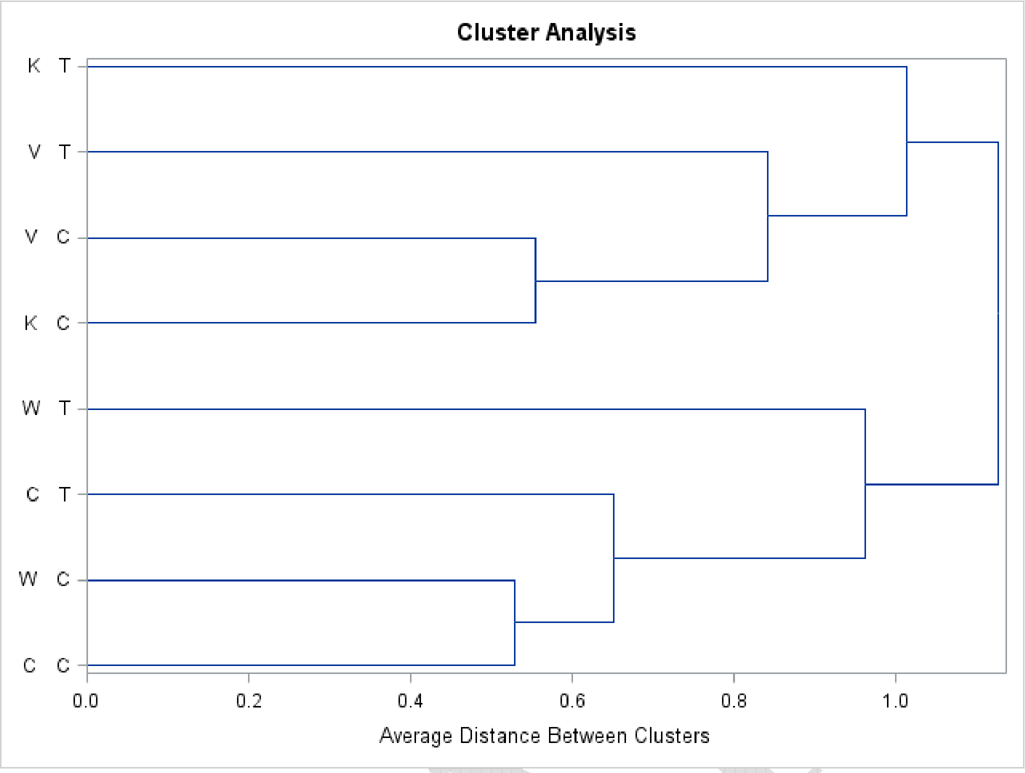
i. 2012, spring.



j. 2012, fall.



k. 2013, spring



l. 2013, fall.

Figure 2.55. 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 2012–2013 following thinning treatments (i–l).

2.8.2 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, 2009, 2010, 2011, 2012, and 2013. Samples from spring and fall are useful to follow trends in adults and juveniles in order to assess breeding status and production over each year, but season species composition generally does not change as with birds.

Counts of individual rodents in 2013 revealed that rodent densities tended to greater than or equal to those on the treated plots, and that rodent densities rose through the summer and were greater in the fall than in the spring, especially at the piñon/juniper sites (Figure 2.56). Numbers of rodent species tended to be higher on treated plots than control plots in both the fall and spring, especially at the two piñon/juniper sites (Figure 2.57). These findings are consistent with indications that that rodents are responding to the increased herbaceous vegetation and likely food resources on the treated plots in the fall.

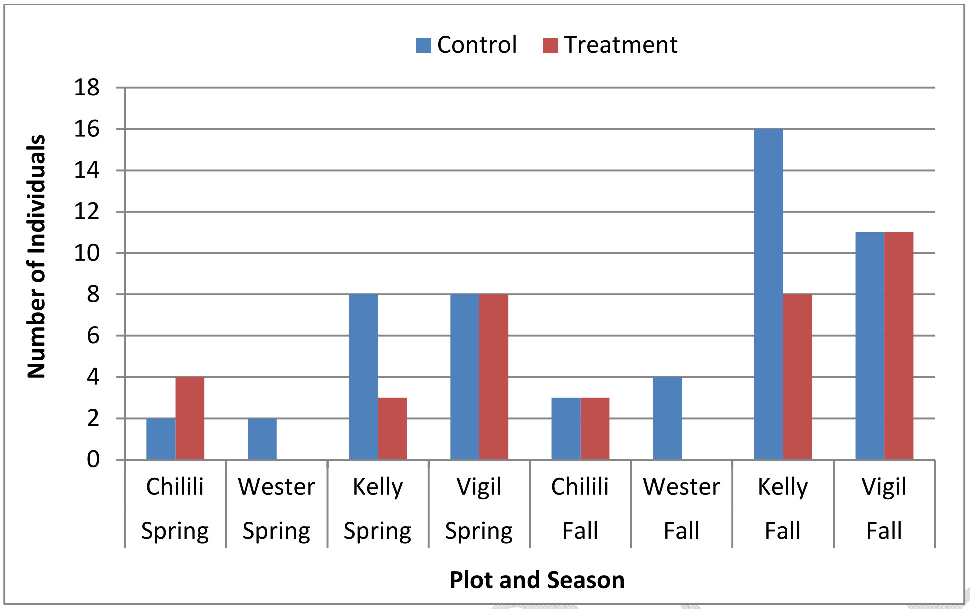


Figure 2.56. Numbers of individual rodents recorded from thinning treatment and control plots across the four study sites in both spring and fall, 2013.

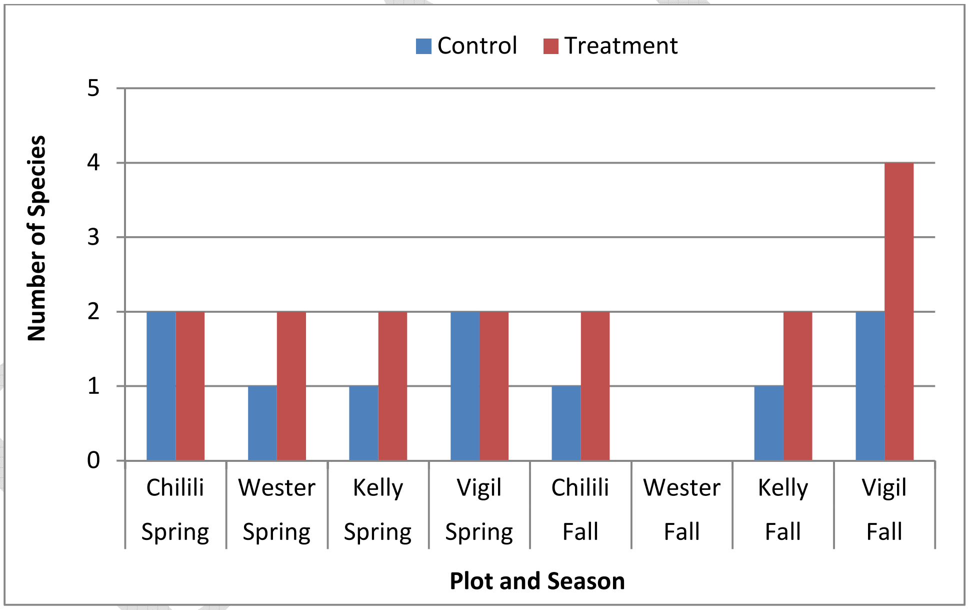
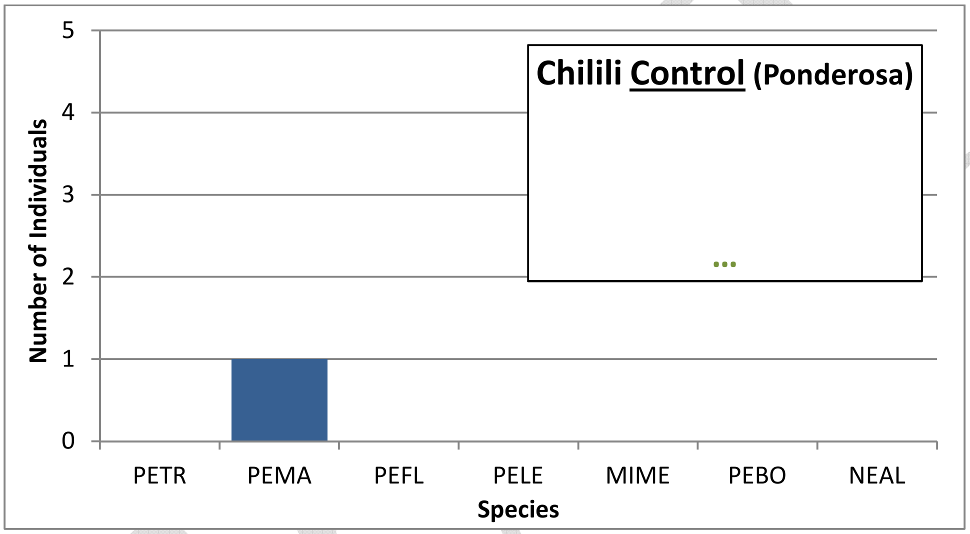


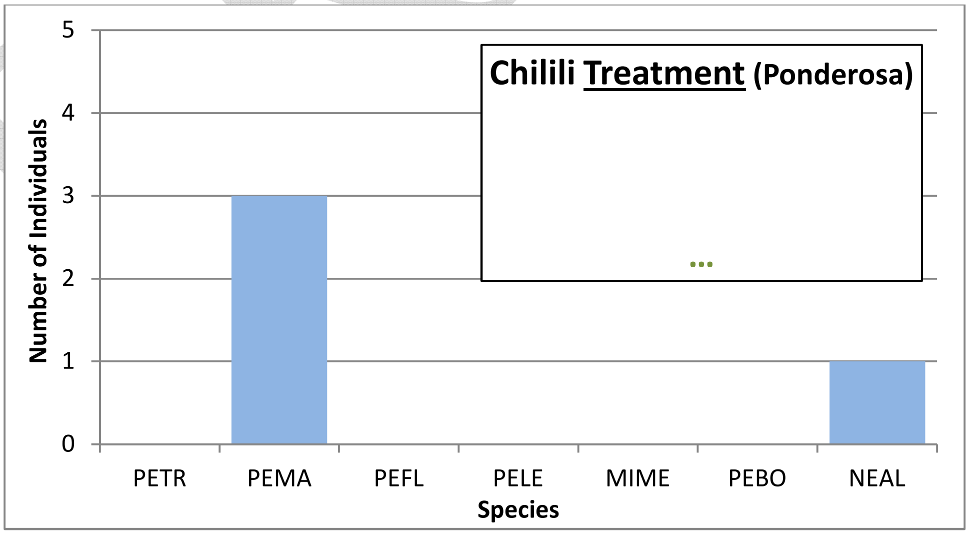
Figure 2.57. Numbers of rodent species recorded from thinning treatment and control plots across the four study sites in both spring and fall, 2013.

Figure 2.58 presents the species and numbers of individuals of those species summarized in Figure 2.56 and Figure 2.57. Examination of Figure 2.58 shows that the deer mouse (*Peromyscus maniculatus*) was the dominant species at the two ponderosa pine sites and the piñon mouse (*P. truei*) was the dominant rodent species at the two piñon/juniper sites, in both spring and fall 2013. However, the deer mouse was the most abundant rodent on the treated plot at the Vigil piñon/juniper site in the spring, while the piñon mouse was the most abundant rodent on the

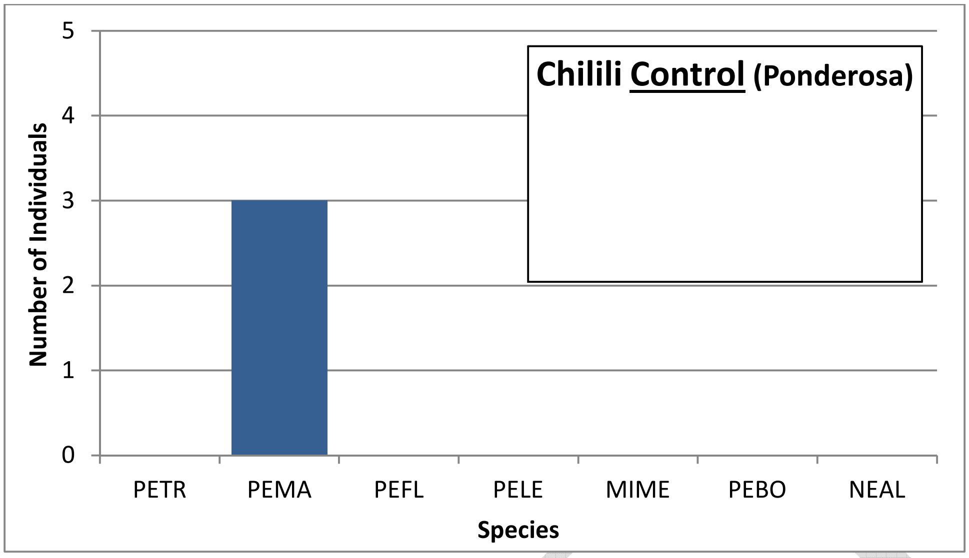
Vigil control plot in the spring and fall, and on the treated plot in the fall. The deer mouse was absent from the Kelly piñon/juniper control plot in the spring and fall, but was present on the Kelly treated plot in the spring and fall, and was more abundant than the piñon mouse on the treated plot in the spring, but less abundant than the piñon mouse in the fall. These findings indicate that tree thinning in the piñon/juniper woodlands causes a decline in piñon mice and an increase in deer mice. The deer mouse was more common on the Chilili treatment plot in the spring, but was more common on the control plot in the fall. The brush mouse (*P. boylii*) was found on the Chili treatment plot in fall. The Wester ponderosa pine site had very low rodent numbers, and no rodents were trapped on the treatment plot in the spring or fall of 2013. Only piñon mice were found at the Wester control plot in the spring, and both deer mice and piñon mice were found at the Wester control plot in the fall of 2013.



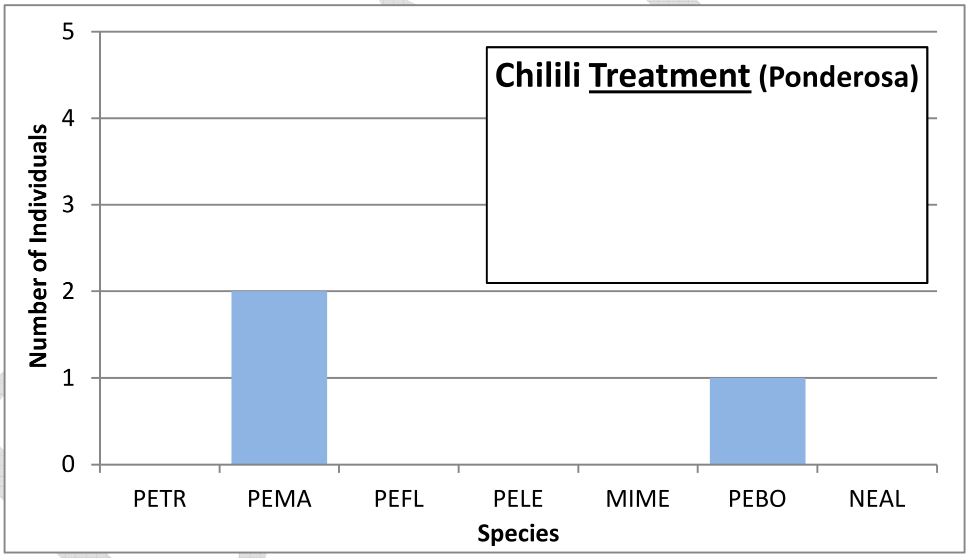
a. Chilili control plot, spring 2013.



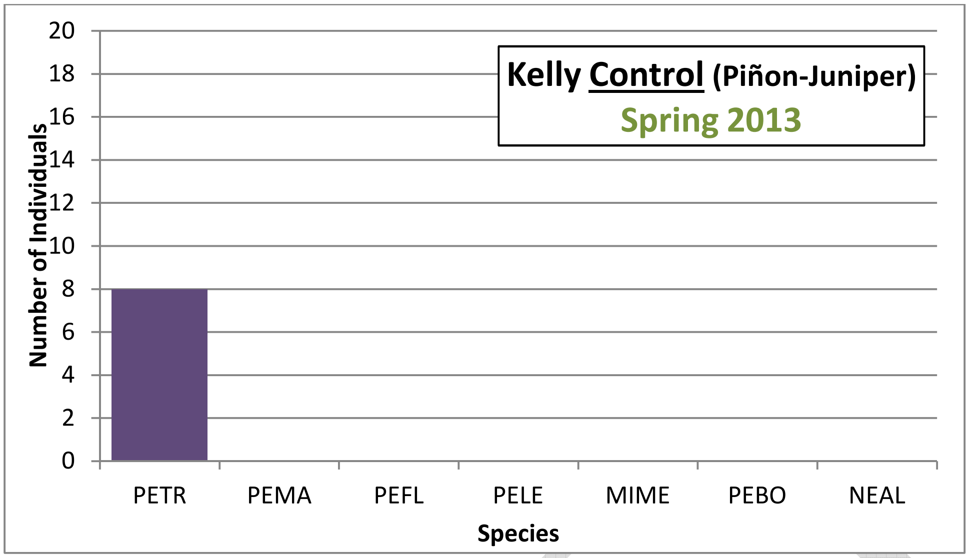
b. Chilili treatment plot, spring 2013.



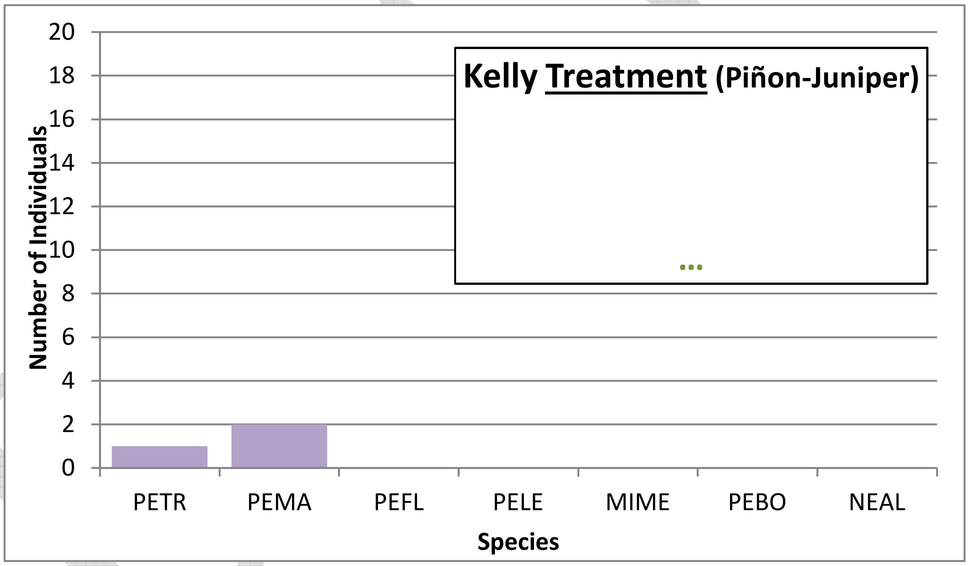
c. Chilili control plot, fall 2013.



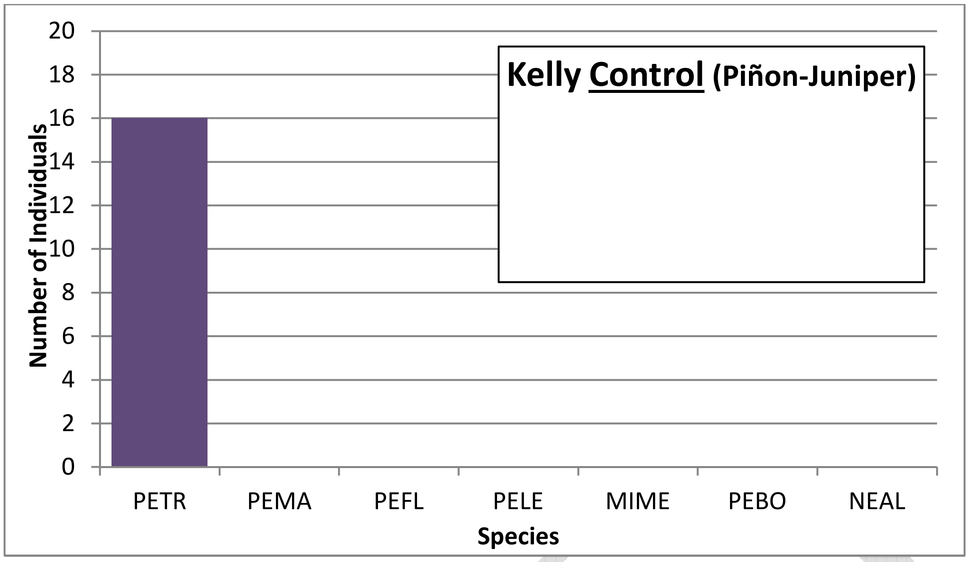
d. Chilili treatment plot, fall 2013.



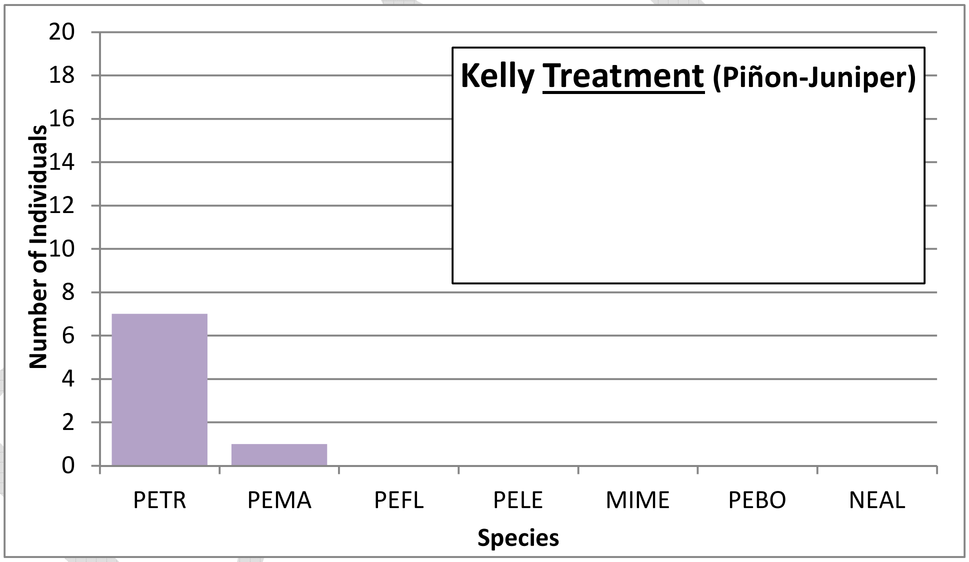
e. Kelly control plot, spring 2013.



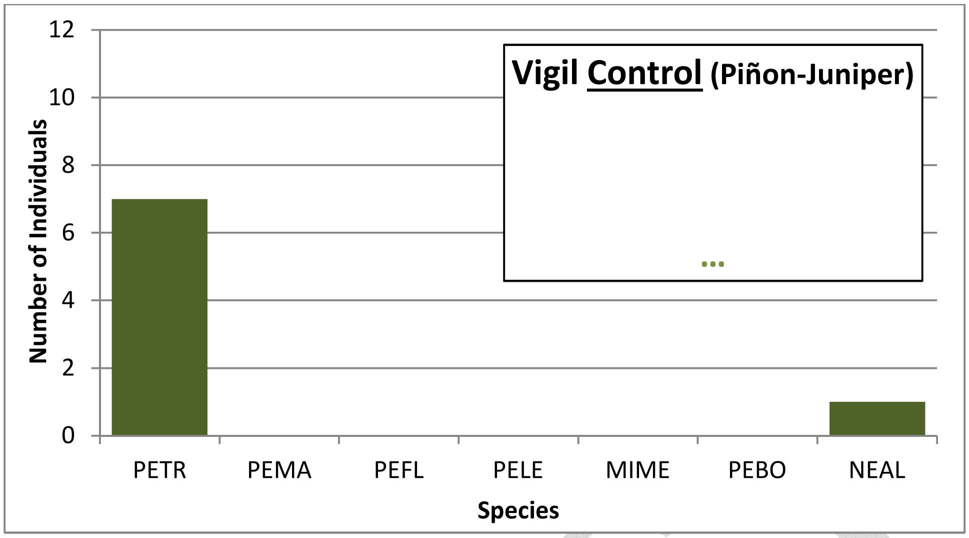
f. Kelly treatment plot, spring 2013.



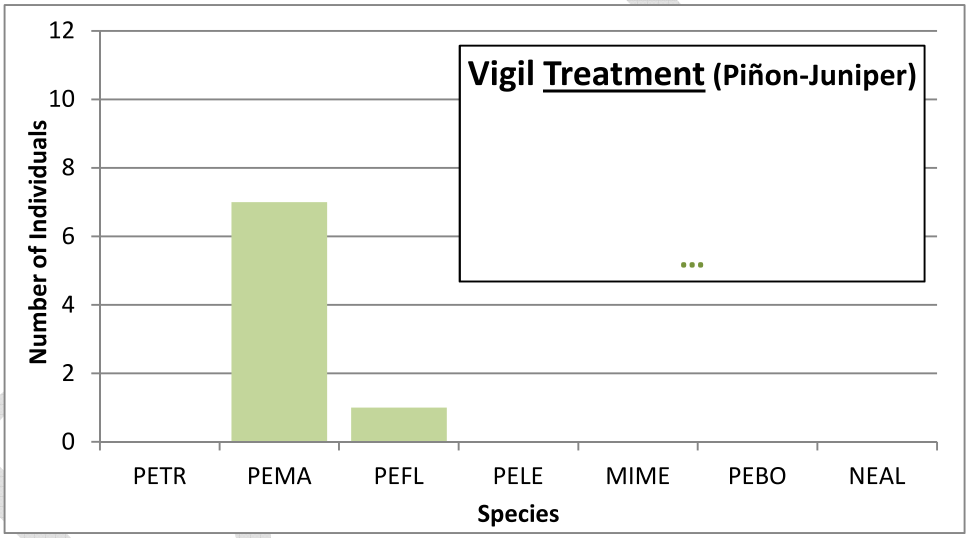
g. Kelly control plot, fall 2013.



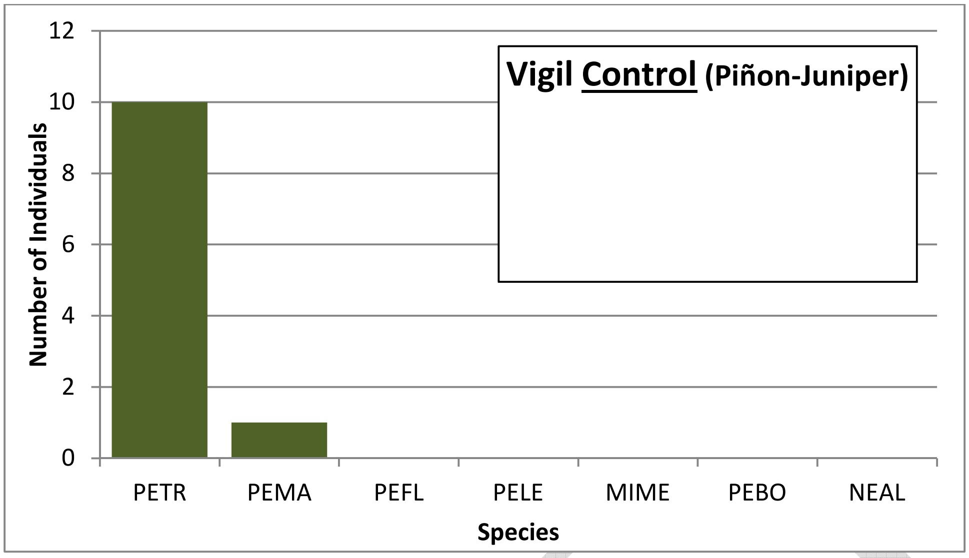
h. Kelly treatment plot, fall 2013.



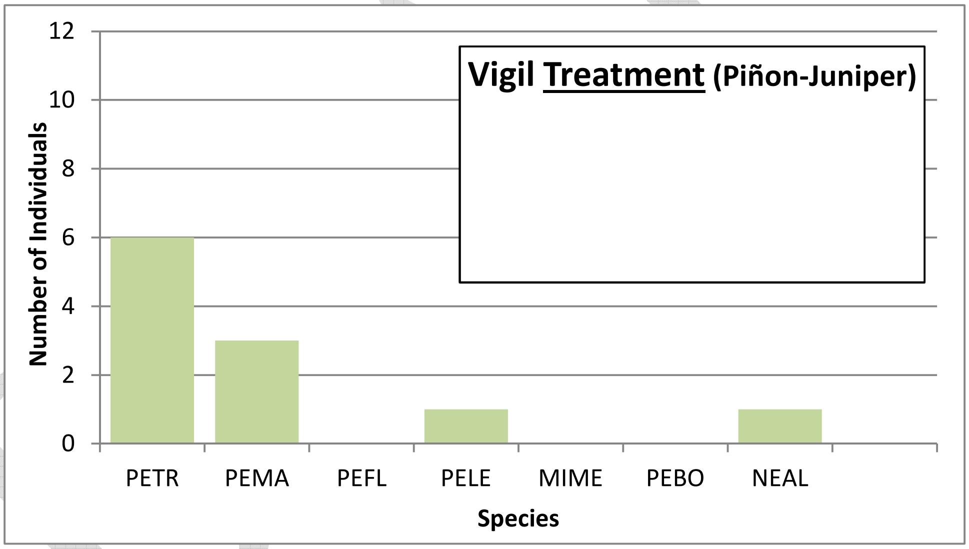
i. Vigil control plot, spring 2013.



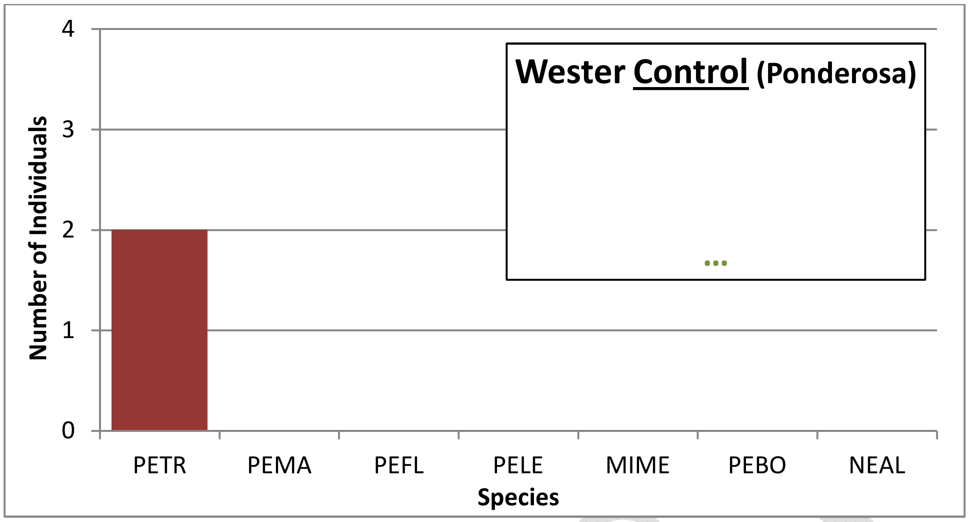
j. Vigil treatment plot, spring 2013.



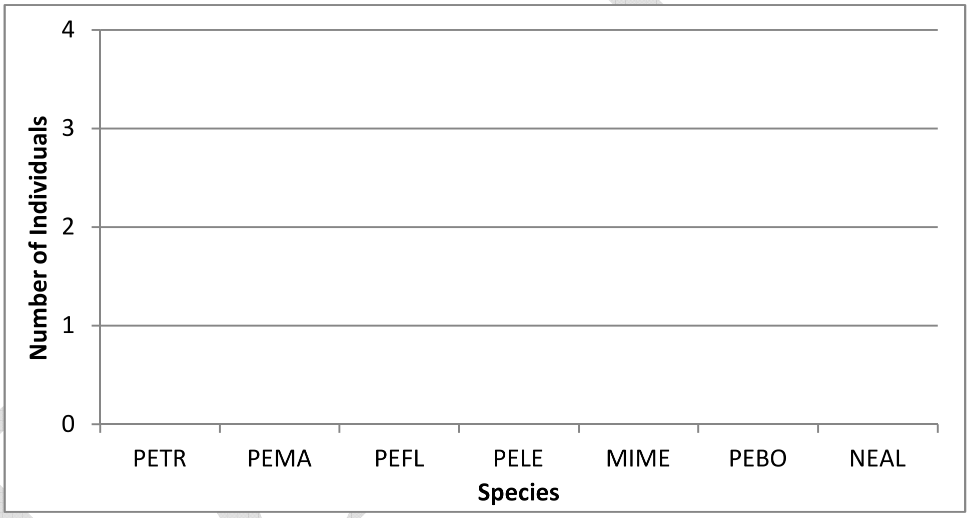
k. Vigil control plot, fall 2013.



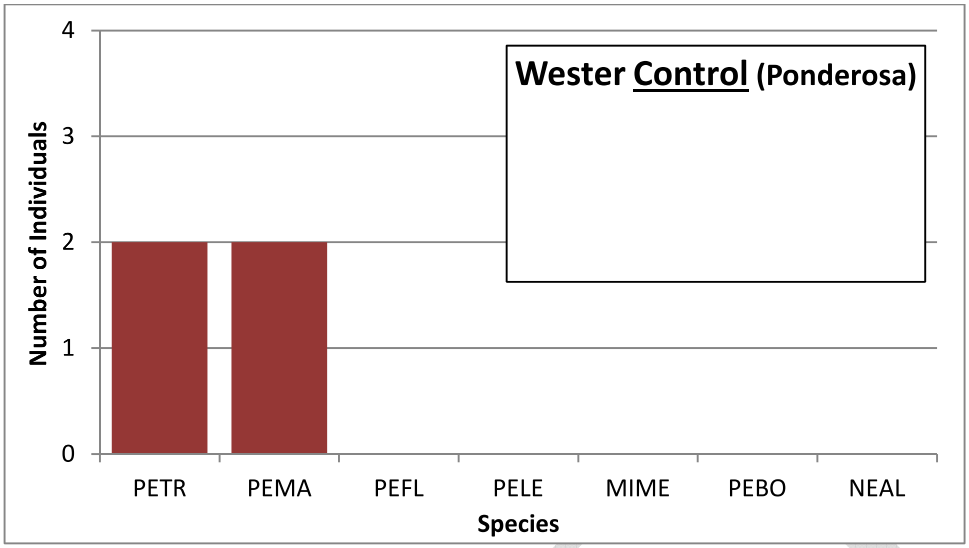
l. Vigil treatment plot, fall 2013.



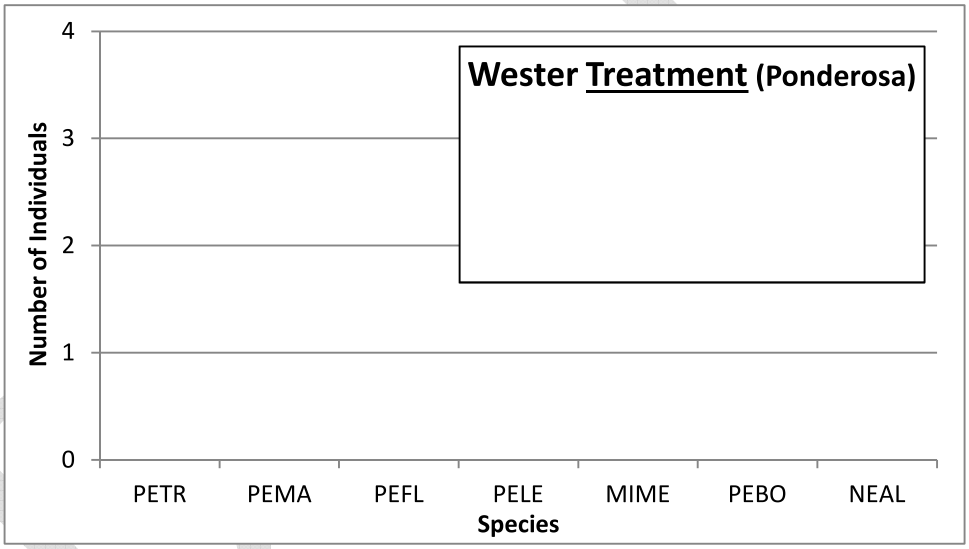
m. Wester control plot, spring 2013.



n. Wester treatment plot, spring 2013. No rodents were found.



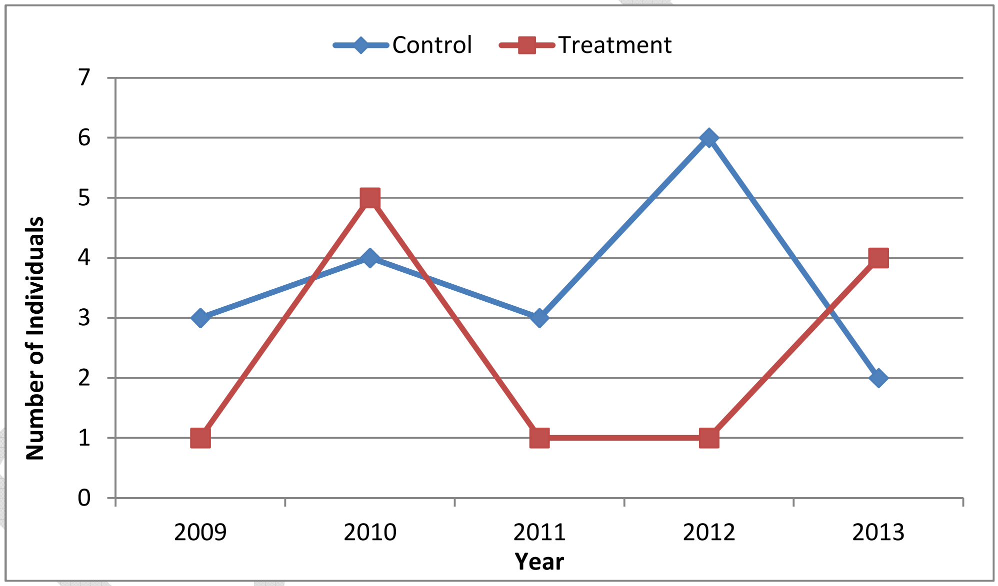
o. Wester control plot, fall 2013.



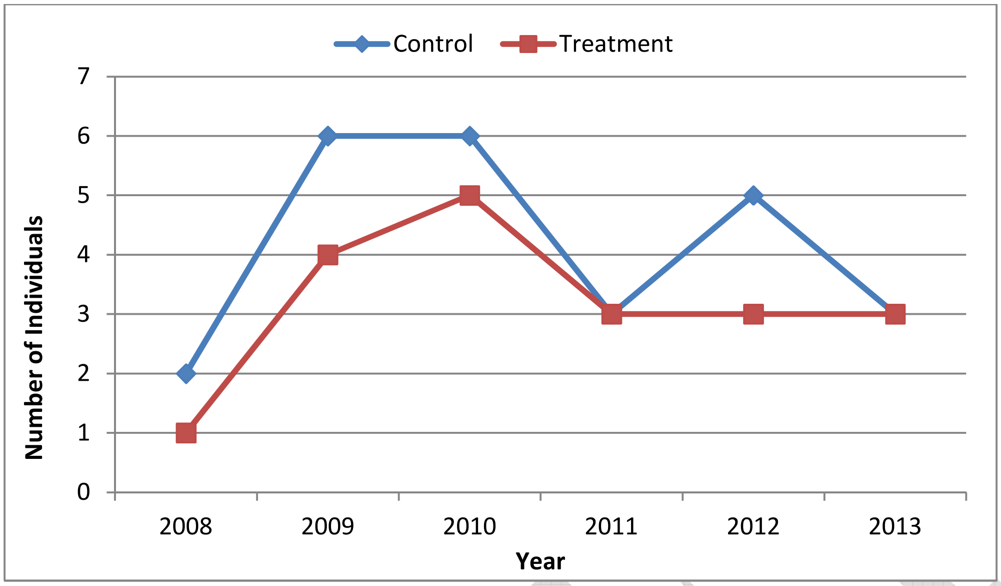
p. Wester treatment plot, fall 2013. No animals were found.

Figure 2.58. Numbers of individual rodents of each species recorded from all control and treatment study plots in 2012, both spring and fall. Rodent species codes correspond to the following names: PETR = *Peromyscus truei* (piñon mouse), PEMA = *Peromyscus maniculatus* (deer mouse), PEFL = *Perognathus flavus* (silky pocket mouse), PELE = *Peromyscus leucopus* (white-footed mouse), MIME = *Microtus mexicanus* (Mexican vole), PEBO = *Peromyscus boylii* (brush mouse), NEAL = *Neotoma albigula* (white-throated wood rat).

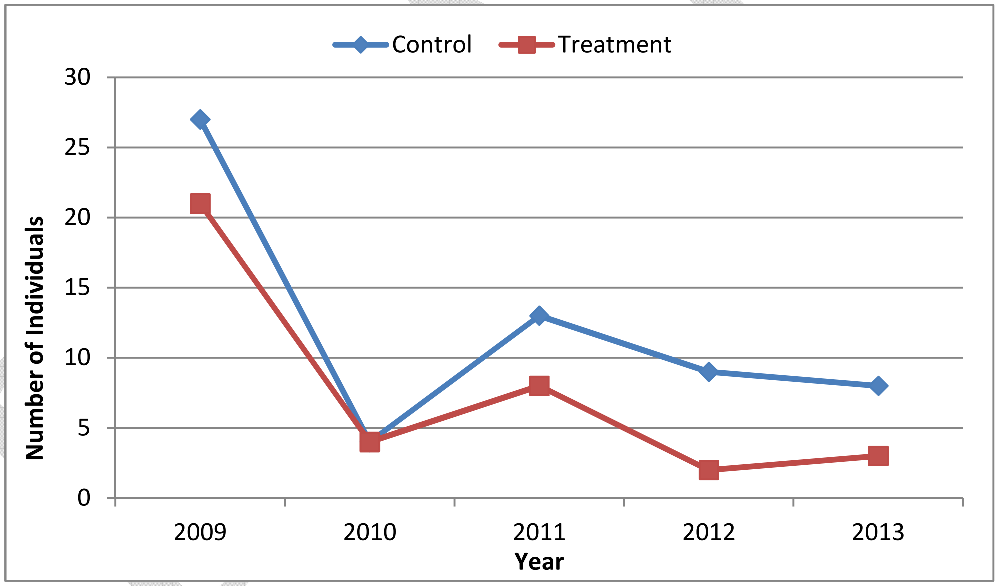
Figure 2.59, a–h, presents total numbers of rodents from control and treatment plots, both spring and fall, from all four study sites from fall 2008 through fall 2013. In general, overall rodent counts peaked in 2009, declined in 2010, increased slightly in 2011 and 2012, and remained constant in 2013. 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, especially the Wester site. In 2013, spring rodent numbers were higher on the Chilili treatment plot than on the control plot, while at the Wester site rodent numbers have remained higher on the control plot than the treatment plot since 2011. At the two piñon/juniper sites, rodent numbers have been similar between the treatment and control plots at the Vigil site since 2008, but at the Kelly site, rodent numbers have been consistently higher on the control plot in the spring, and were higher on the control plot in the fall in 2013. In summary, rodent numbers have been higher on control plots at one ponderosa pine site and at one piñon/juniper site, but there have been no clear consistent differences between treatment and control plots at the other two sites. Numbers of rodent species were not high enough to perform cluster analysis of rodent communities across the sites as was done for birds above.



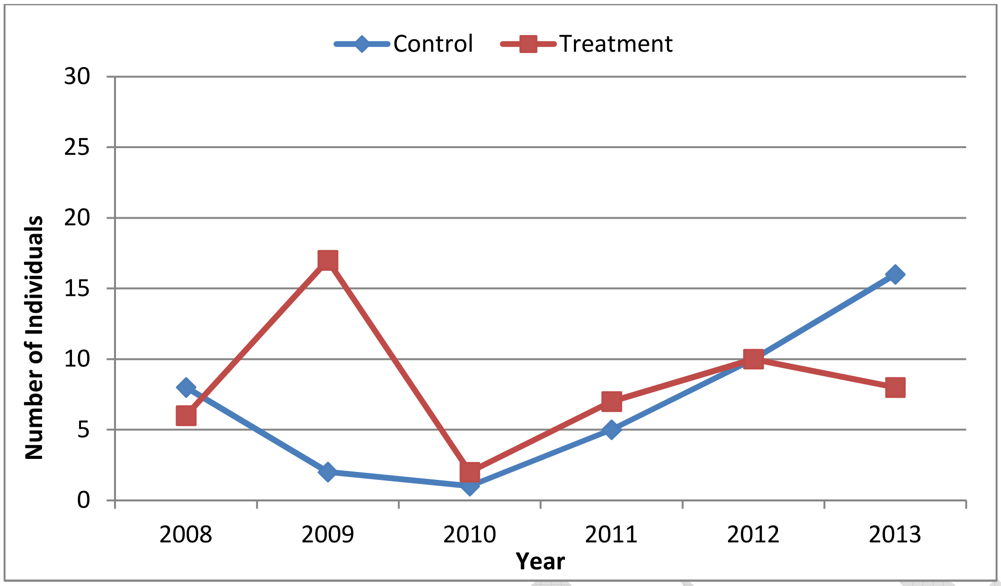
a. Chilili spring.



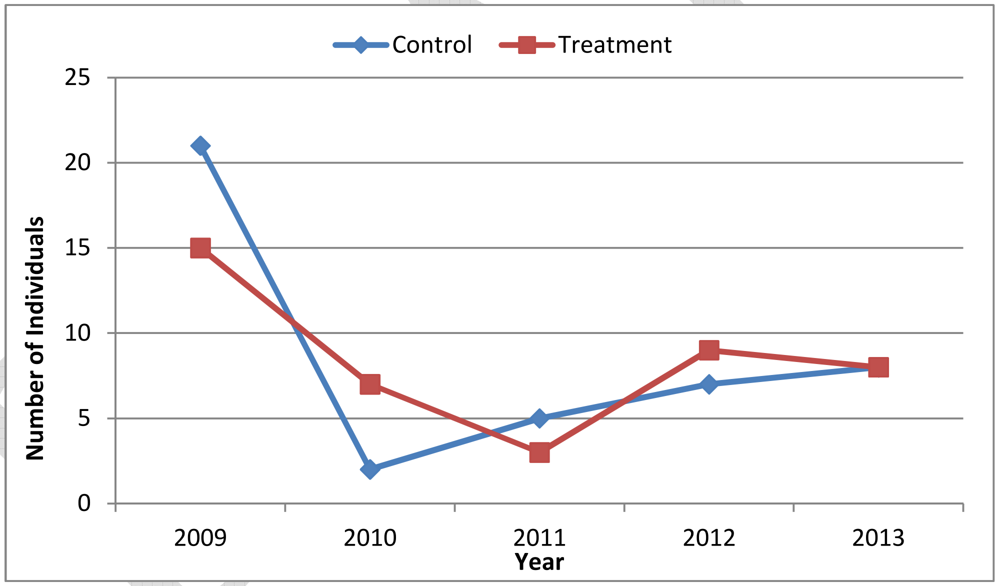
b. Chilili fall.



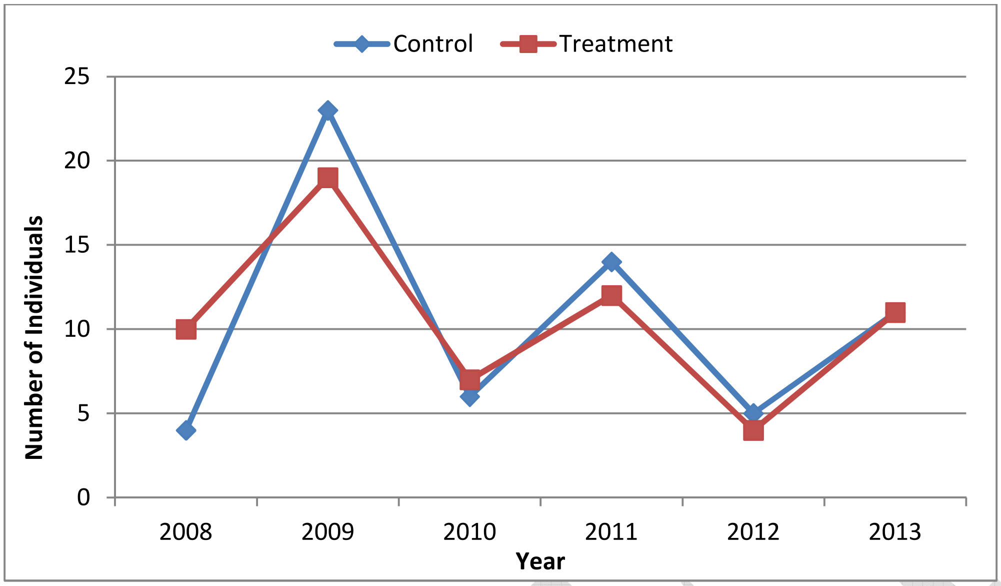
c. Kelly spring.



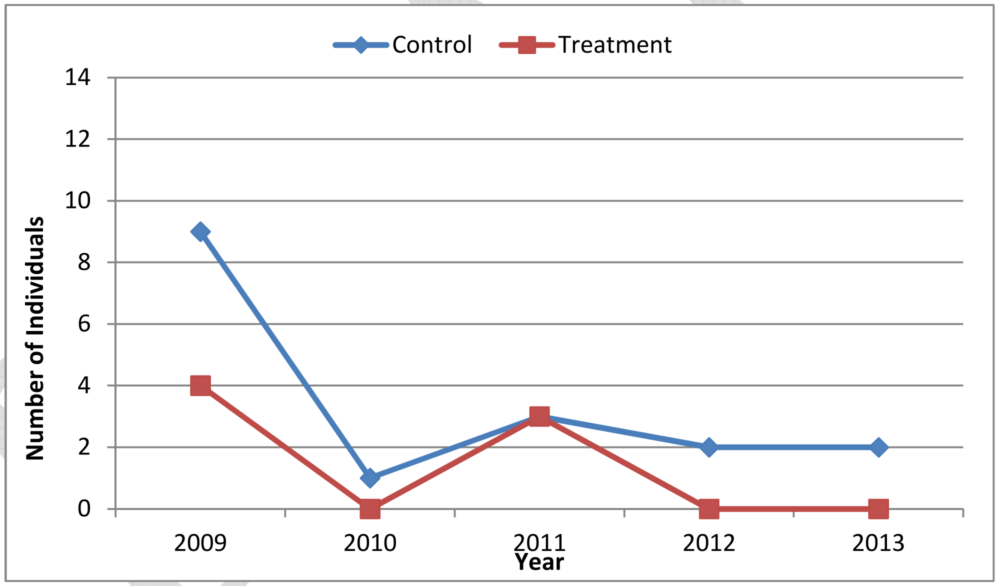
d. Kelly fall.



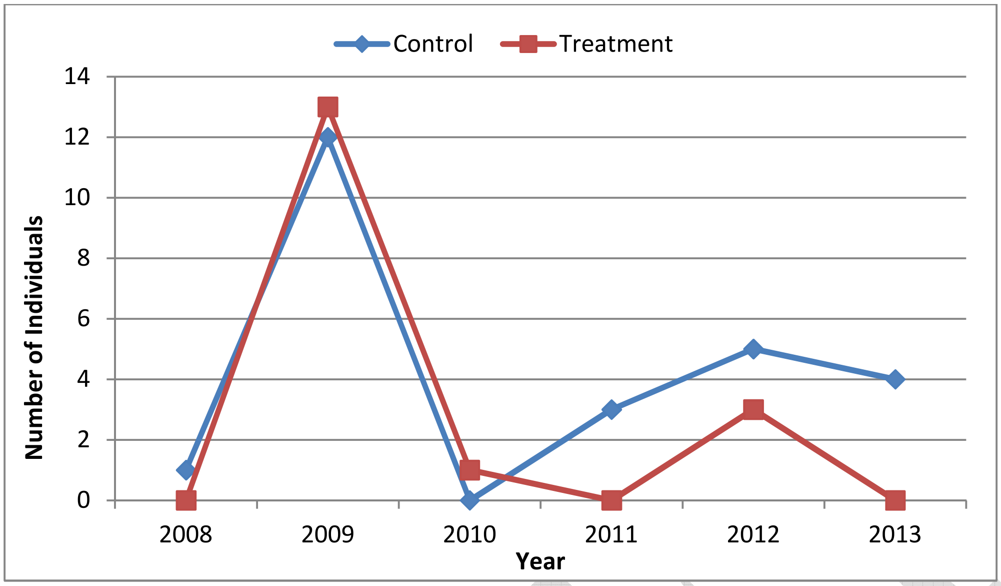
e. Vigil spring.



f. Vigil fall.



g. Wester spring.



h. Wester fall.

Figure 2.59. Total numbers of rodents from both control and treatment plots at all four study sites, fall 2008–fall 2013.

2.8.3 WILDLIFE CAMERAS

Wildlife cameras (Figure 2.60) 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 cameras were all Leaf River IR5 infrared cameras that had a detection sensor up to 21 m (70 feet). 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 2.60. Automatic wildlife camera.

Figure 2.61 and Figure 2.62 present summaries of numbers of different types of animals recorded from wildlife cameras during 2013. Figure 2.61 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 turkey (*Meleagris gallopavo*) are more frequent on control than treatment plots, while domestic livestock, both cattle and horses, are more frequent on treatment plots than control plots. Figure 2.62 shows that domestic livestock were especially abundant at the Kelly piñon/juniper site and were about equally abundant on both the control and treatment plots at the Wester ponderosa pine site. 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.

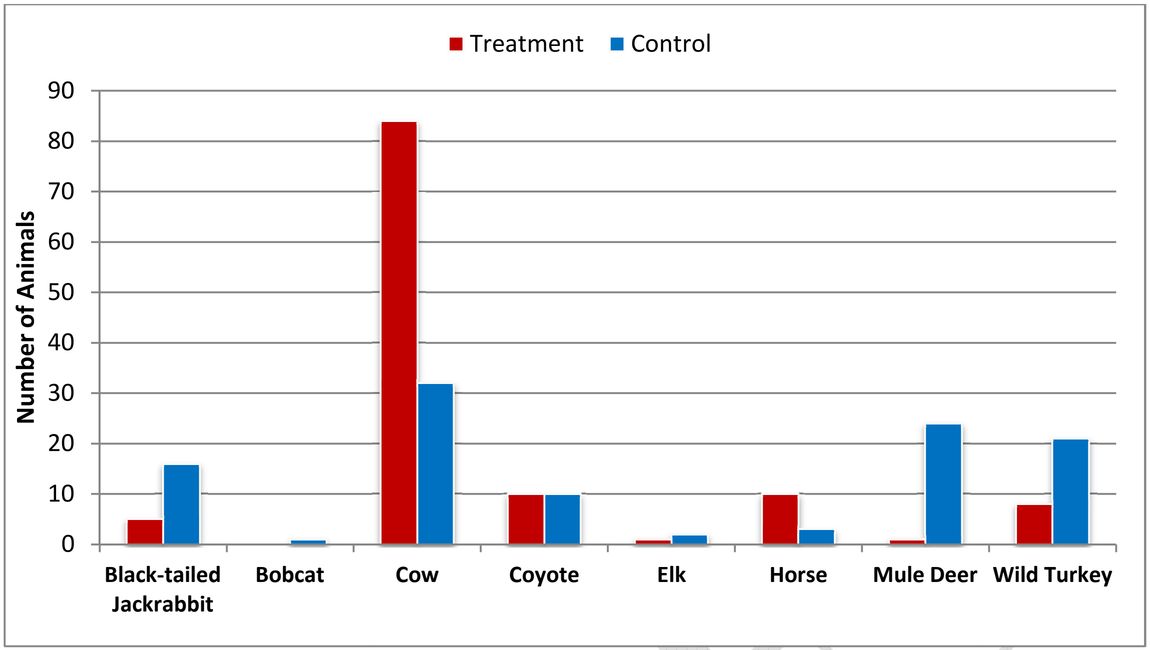


Figure 2.61. Summary of total photographs of different animals recorded from wildlife cameras during 2013.

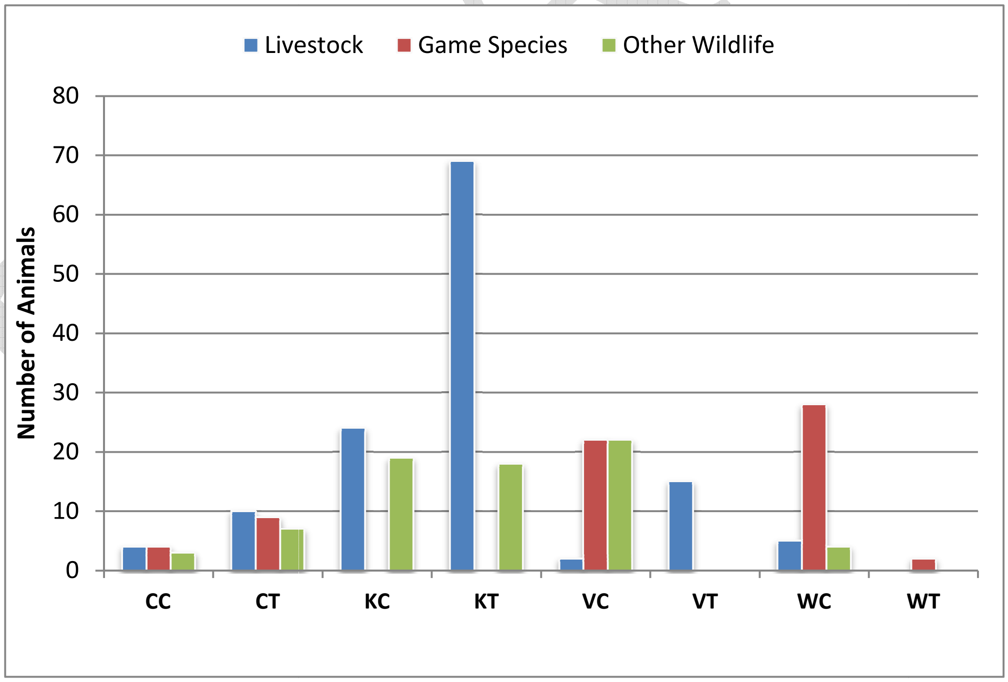


Figure 2.62. Summary of types of animals recorded from control and treatment plots at each study site during 2013.

3.0 EPHEMERAL WATERSHED STREAM MONITORING

Background information on the stream piezometers can be found in the 2009 Annual Report. 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 2013 monitoring season saw very few flows; however, a large flow did occur at the Vigil site and destroyed the stream piezometer (Figure 3.2). Due to the damage caused to the stream piezometer from the flood, the stage could not be recorded; however, it was estimated that the peak flow reached nearly 1.5 m (5 feet) (Figure 3.3). A new gauge made of galvanized steel was installed to replace the damaged gauge on a subsequent site visit. The other gauges at the Chilili and Kelly site did not record any flows during the 2013 monitoring season, but seemed to be in good working order.

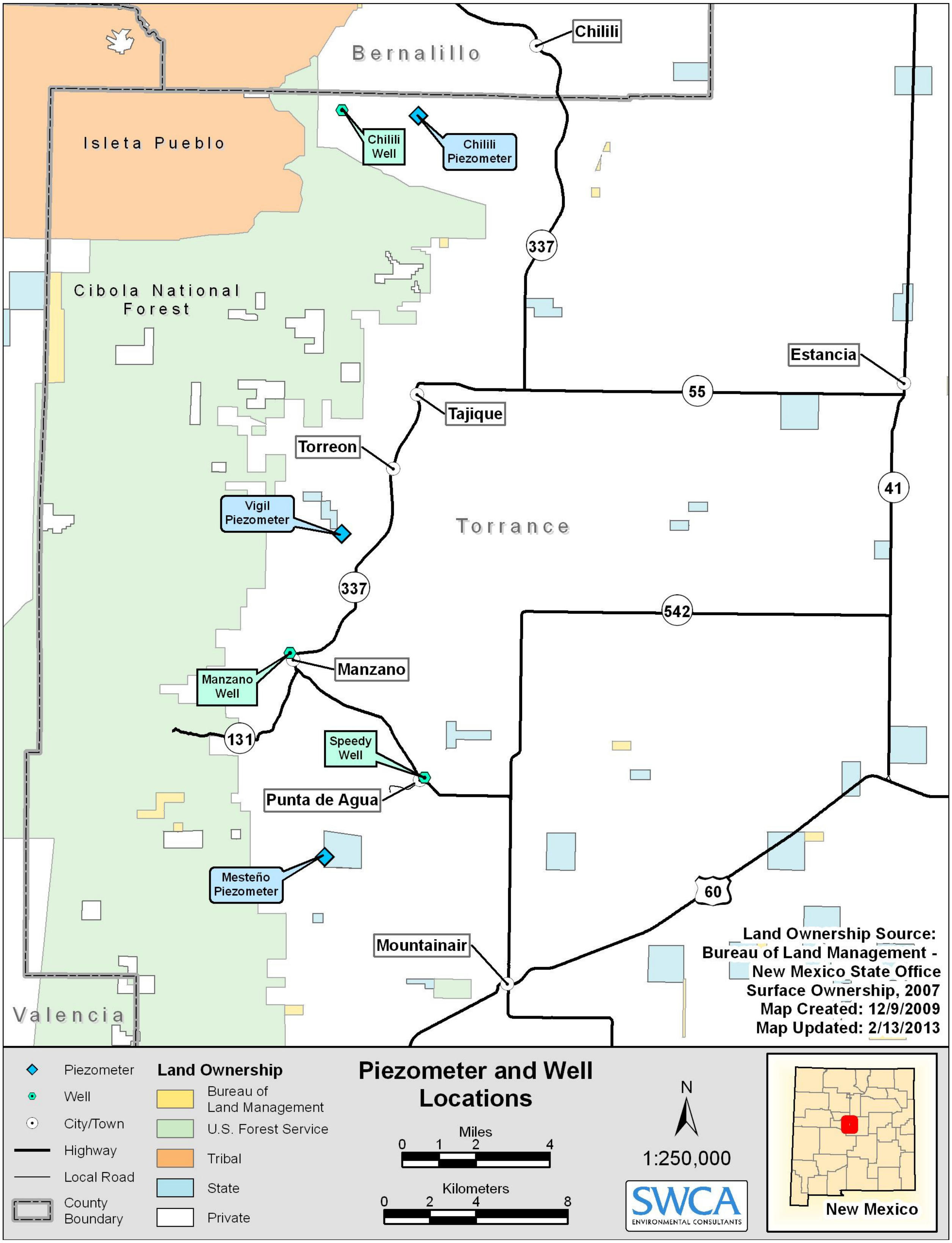


Figure 3.1. Location of the piezometers and wells within the Estancia Basin.

Figure 3.2. The Vigil piezometer in the fall of 2013 after a storm event compromised the gauge.

Figure 3.3. View of the Vigil piezometer facing downstream showing the high water mark (1.5 meters) as a red line above the piezometer.

3.1 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 will offload data quarterly at each well location.

Figure 3.4 through Figure 3.6 display the well data from each of the three locations monitored in the Estancia Basin. During 2013 all wells showed a general decline throughout the year. The well at Punta de Agua showed a steady decline through the course of the year, while the well at Chilili showed a response to the snowmelt as can be seen by the peak in late April 2013. The well at Manzano remained dry for much of the 2013 season.

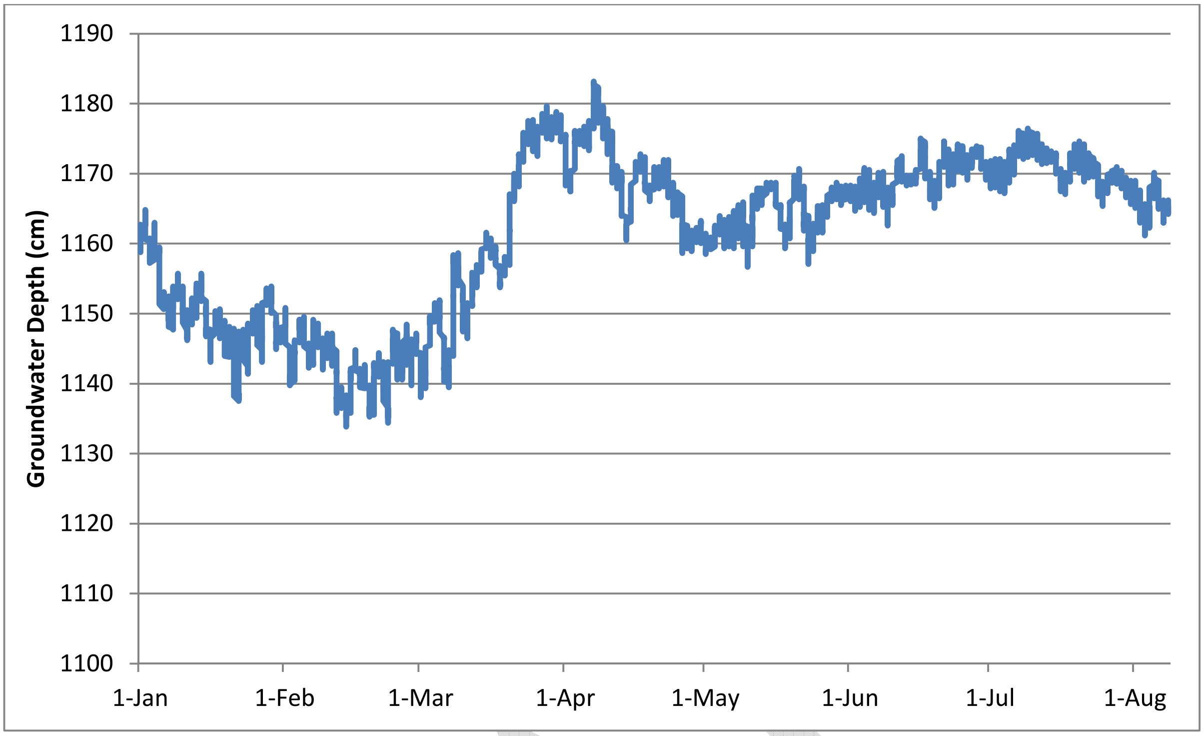


Figure 3.4. **Well data from the Chilili site showing a declining water level until the snowmelt which generates a small rise in groundwater depth.**

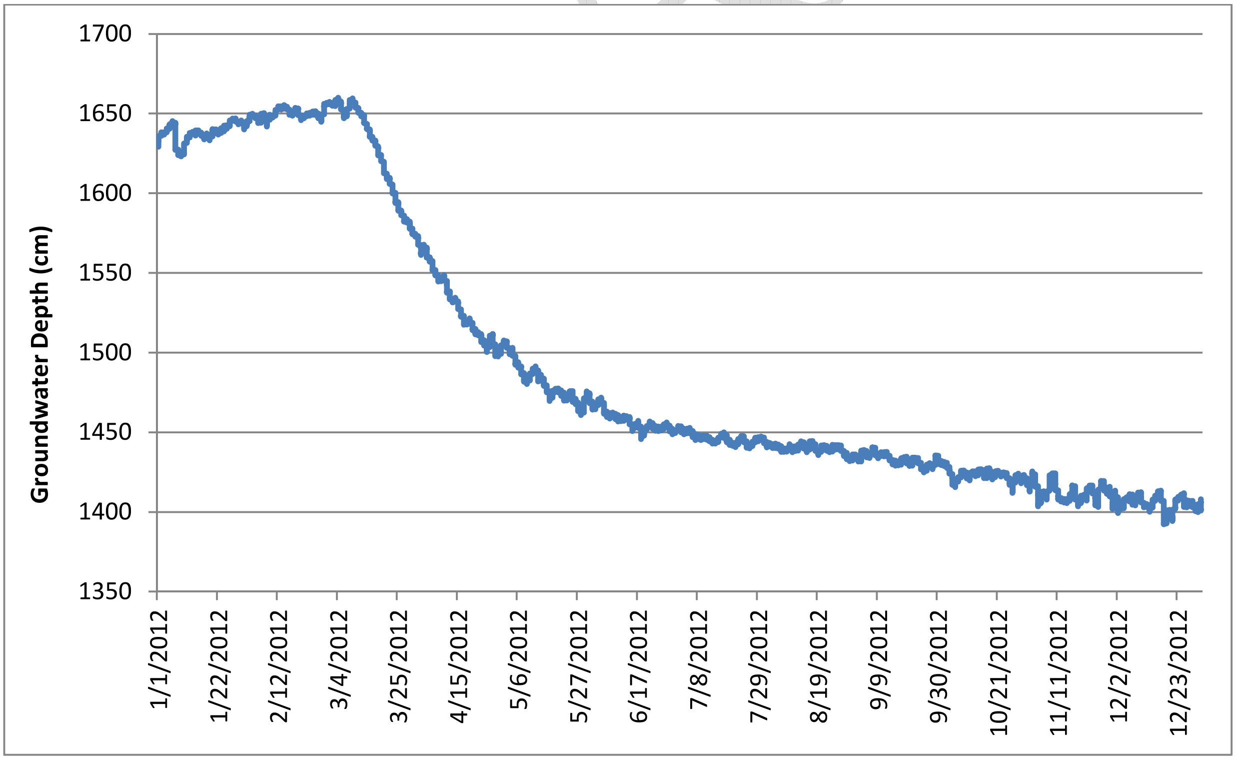


Figure 3.5. **Well data from the Punta de Agua site showing steady rise of the groundwater over the summer months.**

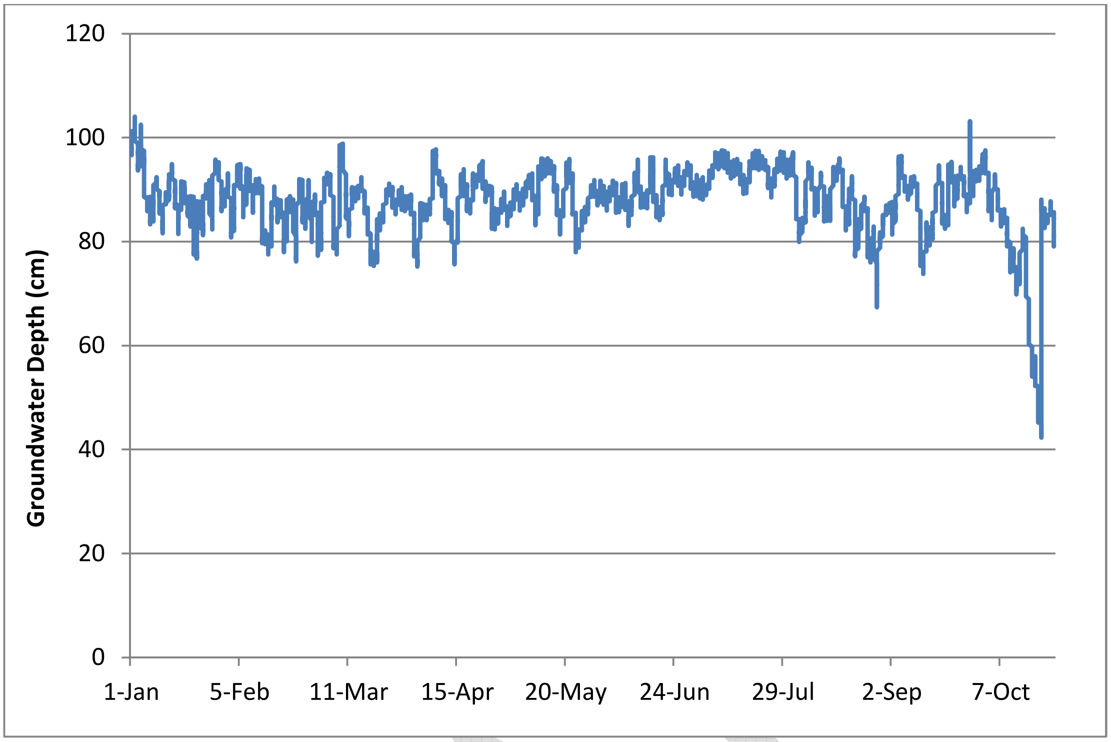


Figure 3.6. Well data from the Manzano site showing the well being dry for most of the year.

4.0 SOUTH MOUNTAIN WEATHER STATION

The SMWS was installed by EnviroLogic to provide meteorological, soil moisture, and temperature data as part of the Estancia Basin Watershed Health and Restoration Program overseen by the Steering Committee. EnviroLogic installed the SMWS in September 2006 to initiate site-specific monitoring of rainfall and soil water content at various soil depths. For details on site selection and monitoring protocols, please refer to the 2008 Monitoring Plan (SWCA 2008). The SMWS is within the Edgewood Soil and Water Conservation District on private property near South Mountain, Santa Fe County, New Mexico, approximately 19 kilometers (12 miles) north of the town of Edgewood (Figure 4.1). The intent of EnviroLogic was to assess water infiltration through soil depths, relate that to meteorological variables, and then compare two measured locations to determine the effects of forest thinning projects on groundwater recharge.

The SMWS measures precipitation, wind speed and direction, air temperature, humidity, and solar radiation. Soil moisture and temperature probes are situated at various depths at two locations with distinct vegetation structure types: one site within a piñon/juniper stand and one site in an adjacent open area consisting of short grasses. EnviroLogic referred to these locations as “Tree” and “Meadow,” respectively. The Tree site is situated approximately 30 m (98 feet) northeast of the SMWS within a grouping of oneseed juniper and piñon pine trees. The Meadow site is situated approximately 11 m (36 feet) northwest of the SMWS, in vegetation dominated by blue grama and broom snakeweed (*Gutierrezia sarothrae*).

SWCA is now responsible for the management of the SMWS and the maintenance, summation, and distribution of the data collected at this station. The following sections summarize the data collected since SWCA assumed responsibility for SMWS in April 2008. SWCA prepared a report, “South Mountain Weather Station: History, Data Summaries, and Continued Operation,” summarizing the data collected from 2006 and 2007 by EnviroLogic, and submitted that report to the Steering Committee. This report is available at the Restoration Institute’s website (http://www.nmfwri.org/).

During the 2013 monitoring season, New Mexico, particularly Torrance and Bernalillo Counties, experienced a severe drought (see Figure 2.7). The effects of the drought can clearly be seen in the result summaries below. The soil moisture measurement at both the Meadow and Tree sites showed long periods of drying, with only the near-surface sensors showing variation (Figure 4.2 to Figure 4.8). There were also no storms in 2013 that were able to produce deep seepage, which would register with the sensors in the deep bore-holes. In fact, over the course of the monitoring period, the deep soil moisture sensors at both the Meadow and Tree sites have remained constant with no variation. The effects of the drought can also be seen when looking at Figure 4.8, which displays the monthly averages of relative temperature and relative humidity. This graph shows high temperatures in June and July with low average relative humidity.

The data displayed below in Figure 4.2 through Figure 4.8 are summarized as monthly averages of relevant meteorological data. Figure 4.9 through Figure 4.11 summarize the relevant meteorological data on an annual basis from 2009 through 2013. These graphs display any trends that may be occurring within the project area. Figure 4.9 shows the annual precipitation and average ambient air temperature on the same graph, and it can clearly be seen that the pattern displayed is the same trend seen at the Watchdog weather stations. Figure 4.10 and Figure 4.11 show the soil moisture changes over the past 4 years at both the Meadow and Tree sites.

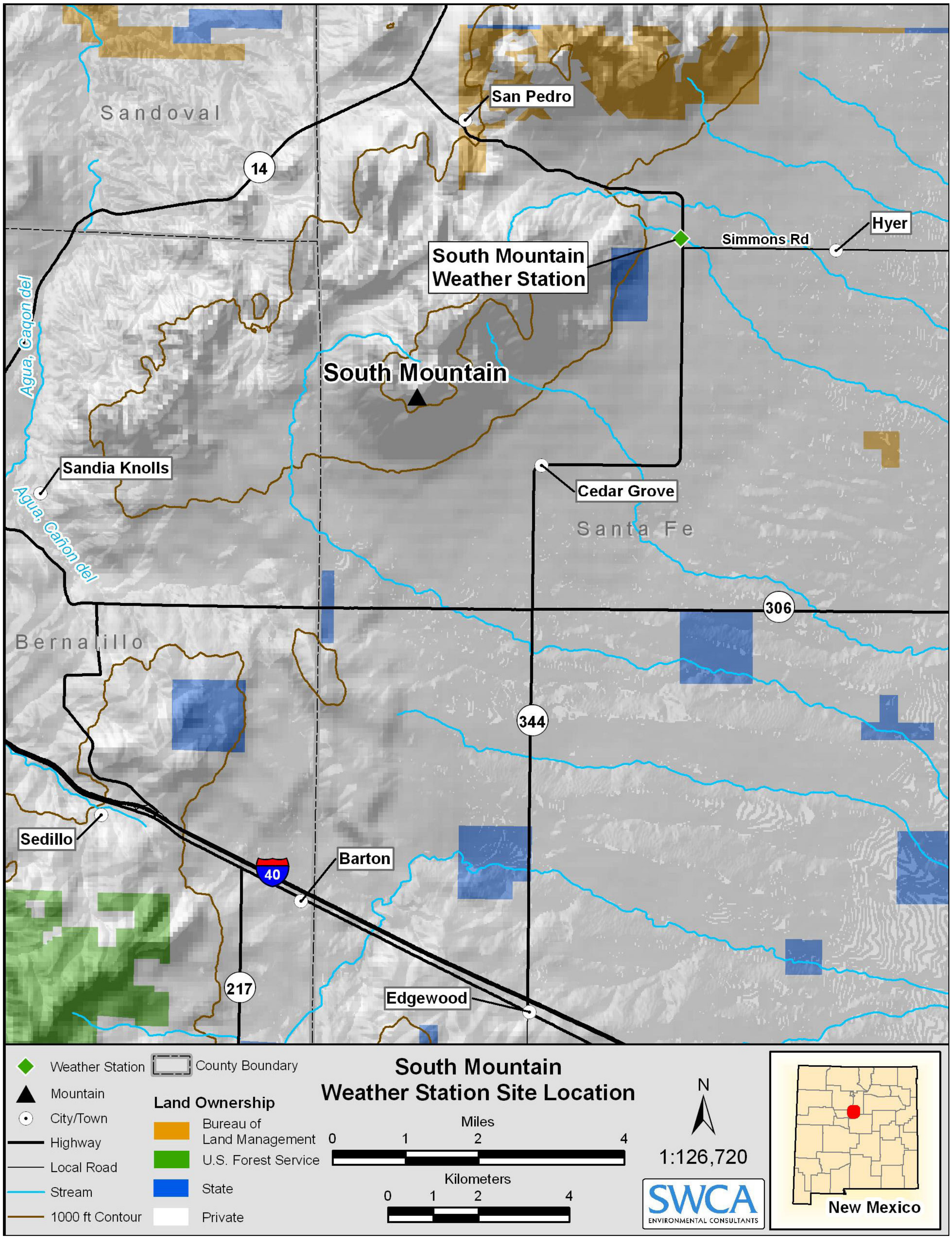


Figure 4.1. **Location of the South Mountain Weather Station.**

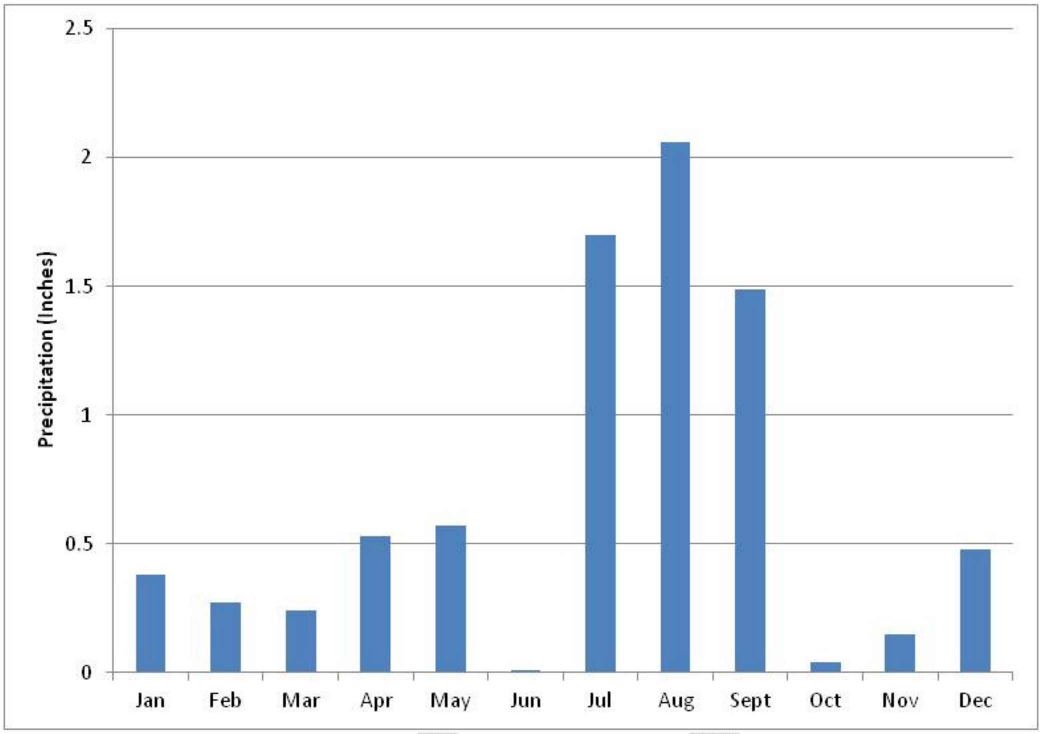


Figure 4.2. Graph showing monthly total rainfall over the course of 2013.

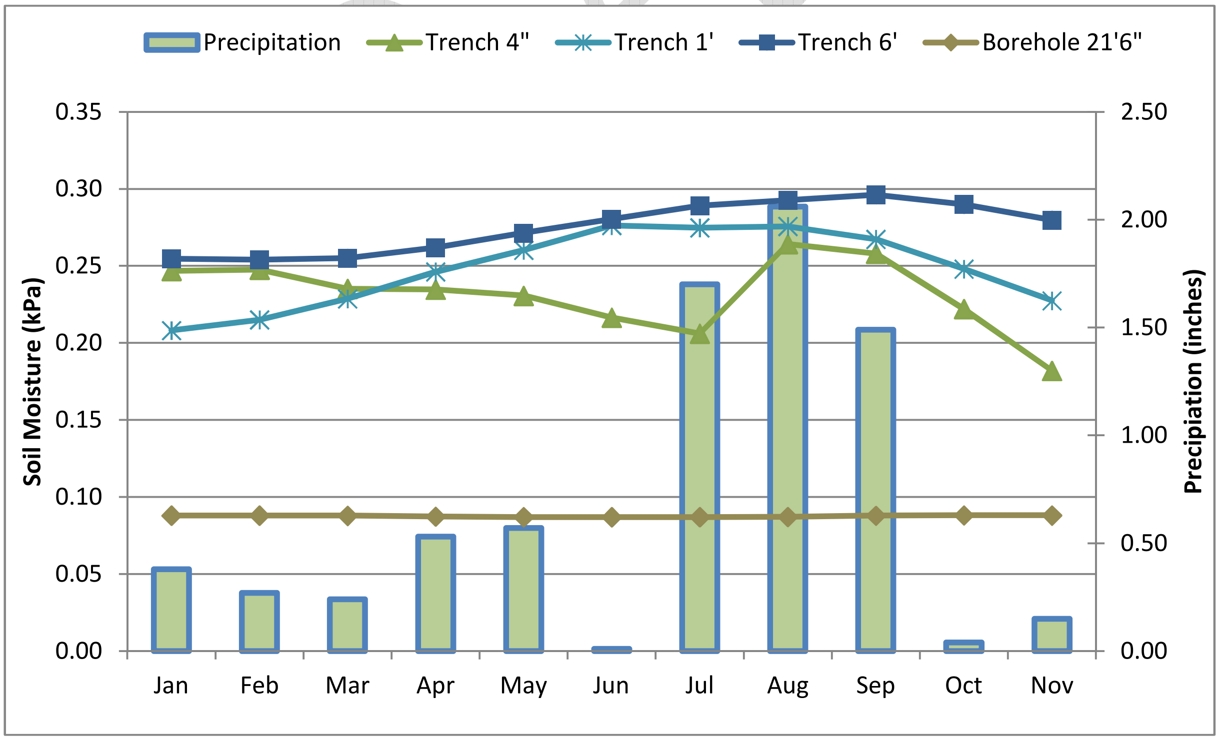


Figure 4.3. Tree site monthly average soil moisture and total precipitation for 2013.

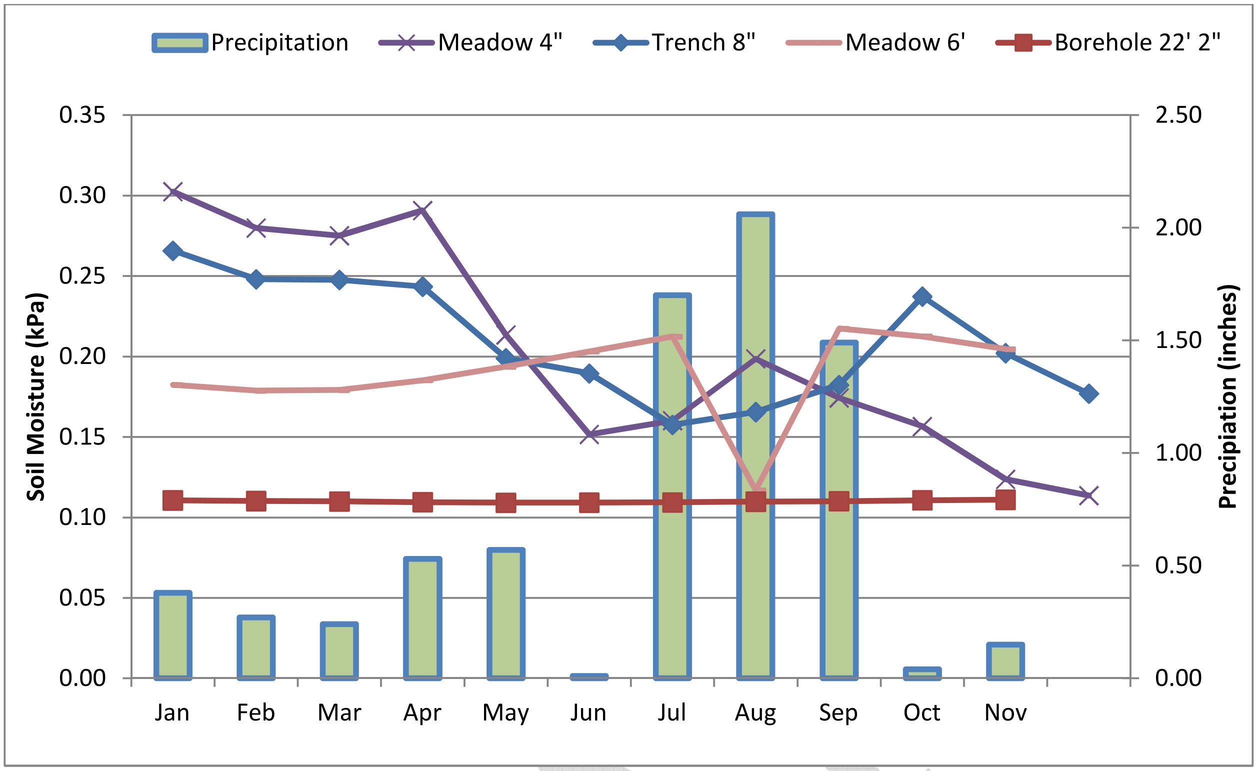


Figure 4.4. Meadow site average monthly soil moisture and total precipitation for 2013.

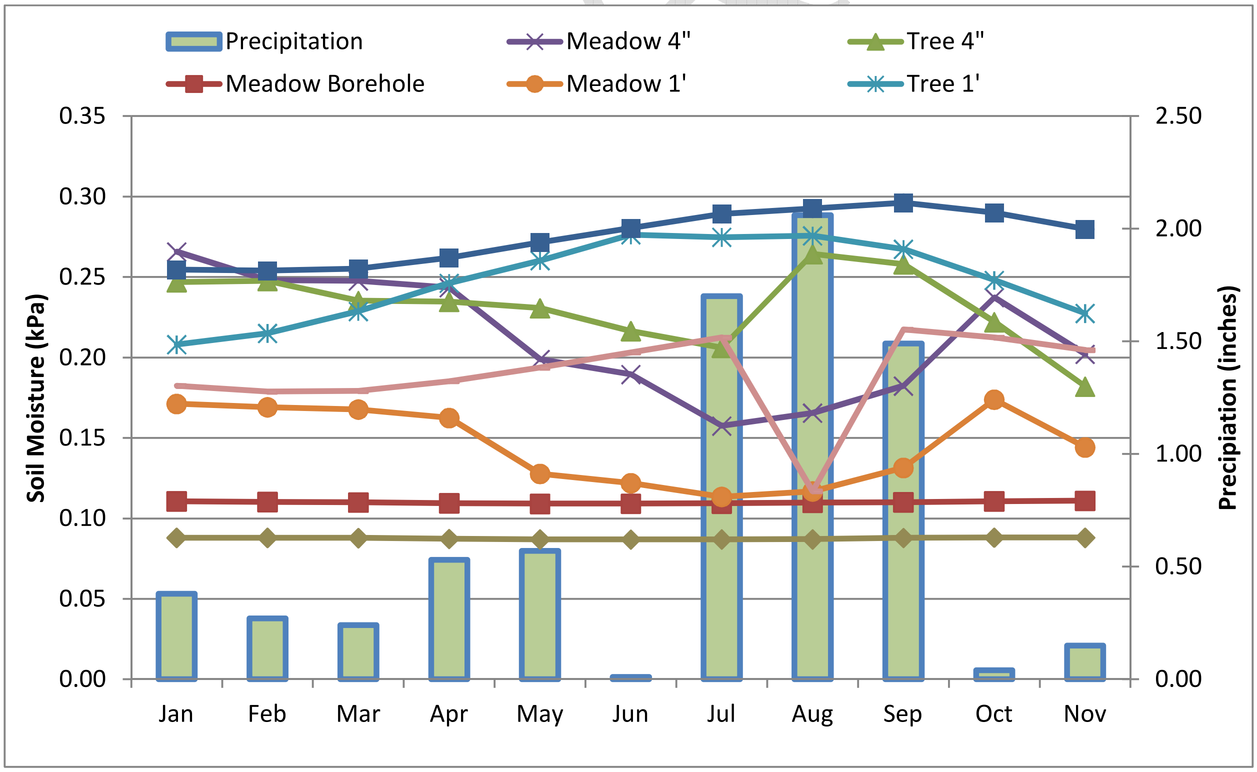


Figure 4.5. Tree and Meadow site average monthly soil moisture and total precipitation for 2013.

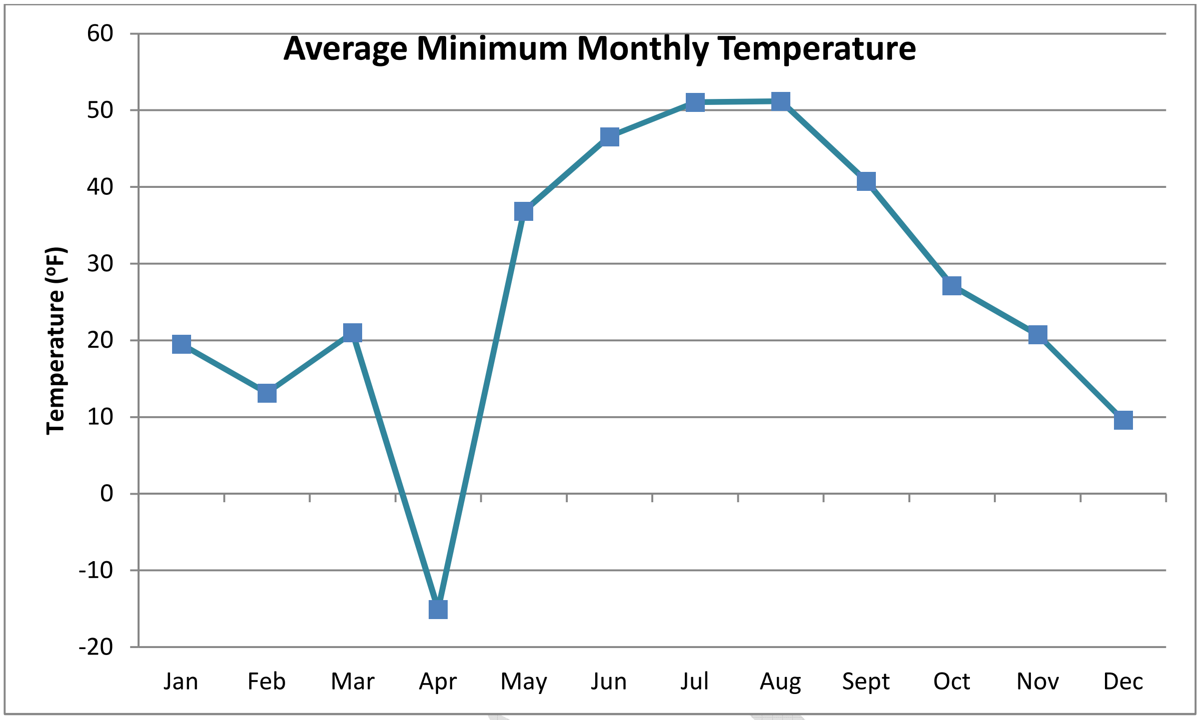


Figure 4.6. **Minimum monthly temperature experienced at the SMWS during 2013.**

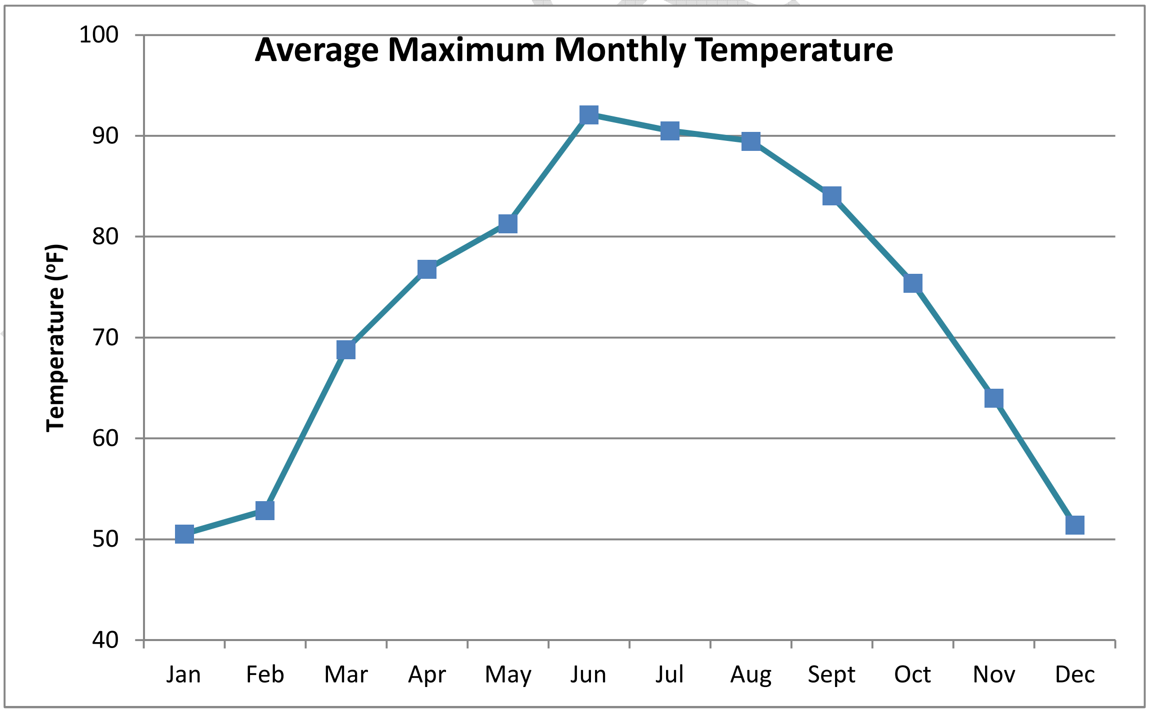


Figure 4.7. **Maximum monthly temperature experienced at the SMWS during 2013.**

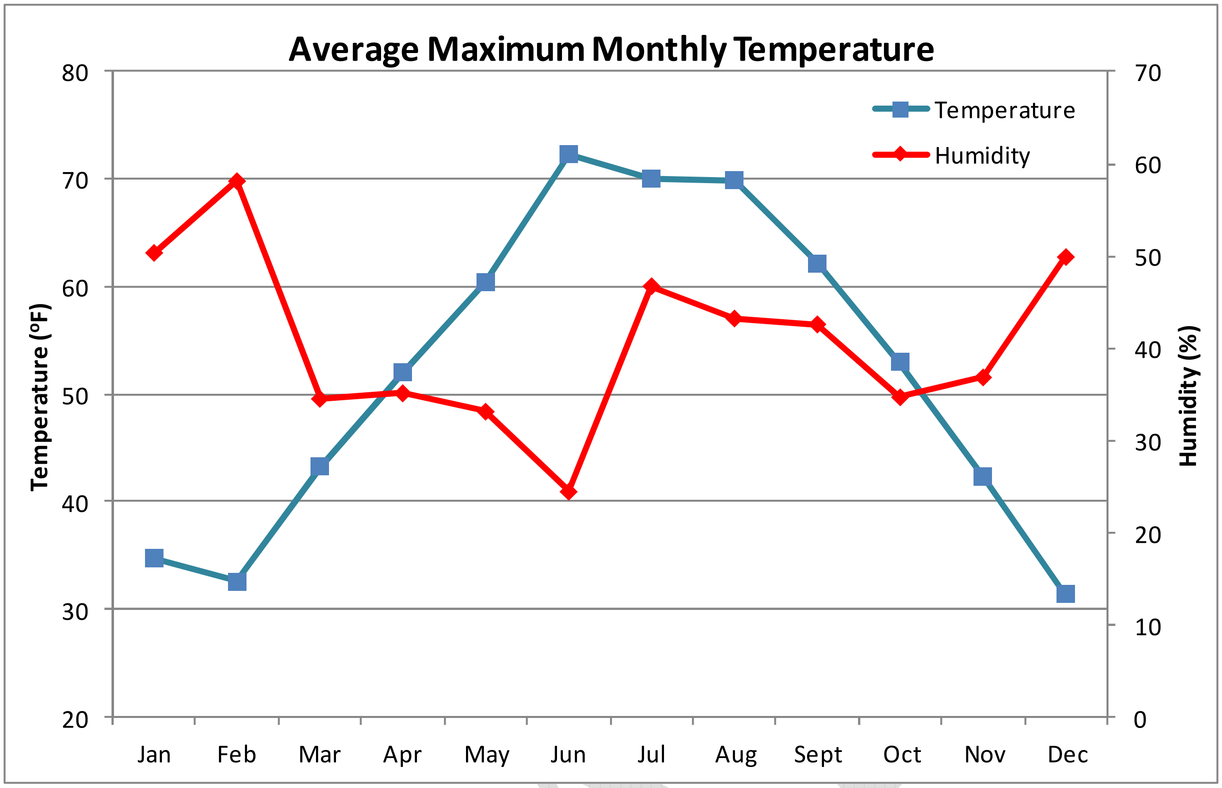


Figure 4.8. **Daily average temperature and relative humidity over the course of 2013.**

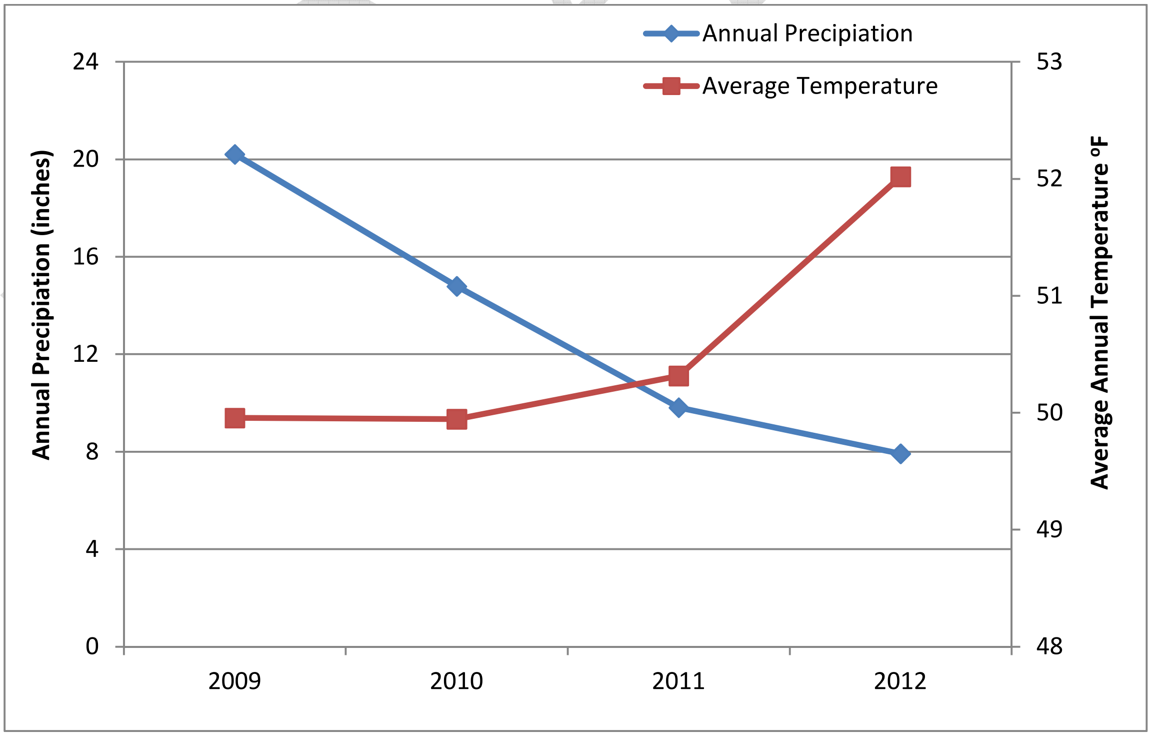


Figure 4.9. **Annual precipitation and average annual ambient temperature at the SMWS 2009–2013.**

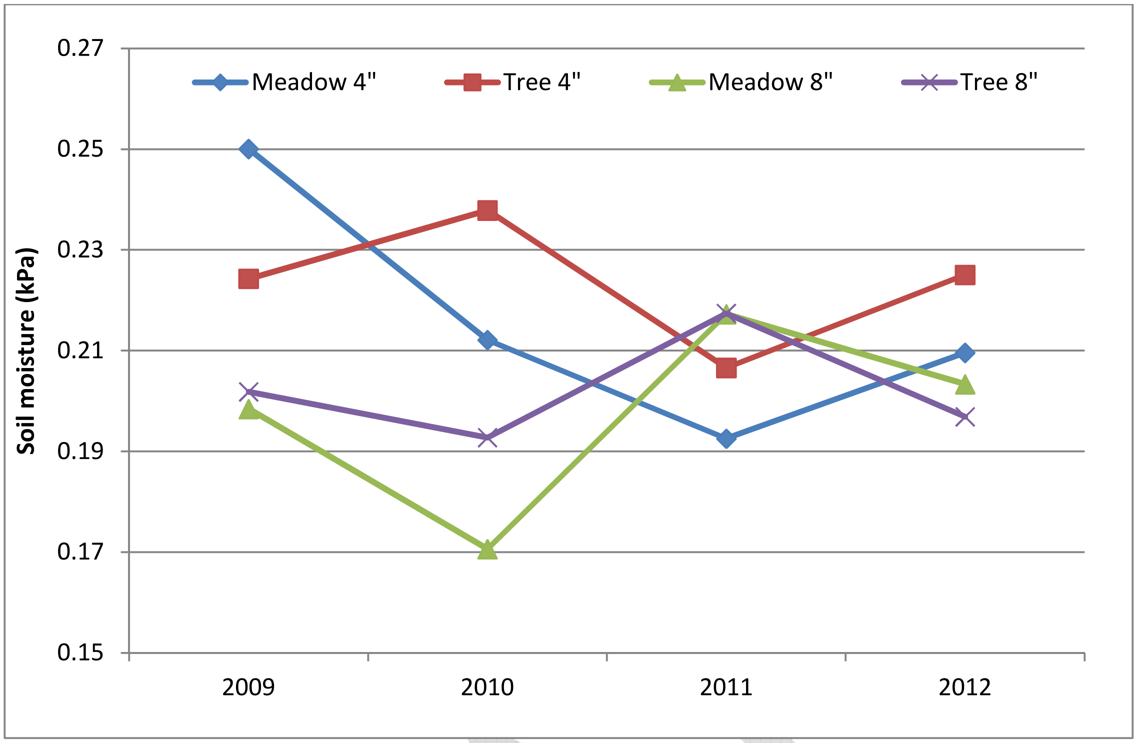


Figure 4.10. Average annual soil moisture from the two shallow depths at the tree and meadow sites with no difference seen between the 4- and 8-inch depths.

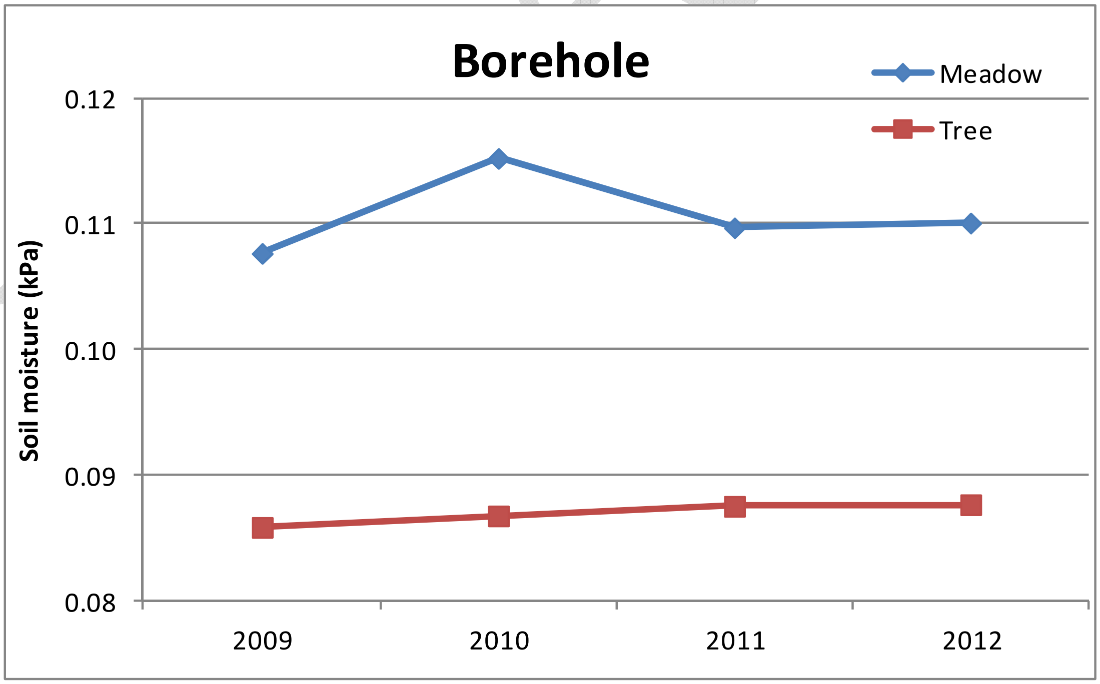


Figure 4.11. Average annual soil moisture from the two deep bore-hole depths (21 feet 6 inches) at the Tree and Meadow sites show no significant time differences throughout the 4 years of monitoring.

5.0 PLANNED MONITORING FOR 2014 (YEAR SEVEN)

SWCA will continue the current monitoring efforts for year seven of this project by monitoring the post-thinning treatment conditions in the late spring. If additional funding is awarded monitoring will continue in fall 2014 and spring 2015. SWCA will also continue to manage the SMWS and the associated weather data if funding is awarded.

Post-wildfire monitoring has been suspended for 2014 because of safety issues regarding falling dead trees. Post-fire monitoring may commence in a year or two, 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 2014 Annual Report.

6.0 ACKNOWLEDGEMENTS AND CONTRIBUTORS

The New Mexico Water Trust Board provided funding for this project. The Estancia Basin Watershed Health, Restoration and Monitoring Steering Committee provided oversight and coordination of this project, in cooperation with the New Mexico Forest and Watershed Restoration Institute and New Mexico State Forestry. Dierdre Tarr of the Claunch-Pinto Soil and Water Conservation District and Joe Zebrowski from the New Mexico Forest and Watershed Restoration Institute provided valuable oversight and support. The Bouton, Candelaria, Kelly, Mitchell, Neff, Salazar, Sanchez, Vigil, and Wester families kindly offered access to their land to conduct forest thinning and monitoring research, along with the Chilili Land Grant, Manzano Land Grant, and the Manzano Mountain Retreat. New Mexico State Forestry, the U.S. Forest Service, the U.S. Geological Survey, and the Claunch-Pinto, East Torrance, and Edgewood Soil and Water Conservation Districts have all provided advice and support. Vernon Kohler and Kelly Archuleta from the Claunch-Pinto and Edgewood Soil and Water Conservation Districts have been assisting with field data collections. Mike Matush from the New Mexico Environment Department, Surface Water Quality Bureau, has been helpful in designing and installing stream monitoring stations. The Estancia Basin Water Planning Committee also contributed funding to install the new Chilili ponderosa pine monitoring study site. Joseph Fluder, project manager from 2007–2012, now provides oversight, and quality assurance and control. In addition to the authors, SWCA staff Ryan Trollinger, Justin Elza, and Alayne Szymanski contributed to the preparation of this report.

This page left intentionally blank

7.0 LITERATURE CITED

Abella, S. R. 2004. Tree thinning and prescribed burning effects on ground flora in Arizona ponderosa pine forests: A Review. Journal of Arizona-Nevada Academy of Sciences. 36(2):68-76.

Abella, S.R. 2009. Tree thinning and prescribed burning effects on ground flora in Arizona ponderosa pine forests: A review.

Abella, S.R., and W.W. Covington. 2007. Forest-floor treatments in Arizona ponderosa pine restoration ecosystems: No short-term effects on plant communities. *Western North American Naturalist 67.*

Abella, S.R., and W.W. Covington. 2004. Monitoring an Arizona ponderosa pine restoration: Sampling efficiency and multivariate analysis of understory vegetation. *Restoration Ecology* 12.

Allan, R.P., and B.J. Soden. 2008. Atmospheric warming and the amplification of precipitation extremes. *Science* 321:1481–1484.

Allen, C.D., M. Savage, D.A. Falk, K.F. Suckling, T.W. Swetnam, T. Schulke, P.B. Stacey, P. Morgan, M. Hoffman, and J.T. Klingel. 2002. Ecological restoration of southwestern ponderosa pine ecosystems: a broad perspective. *Ecological Applications* 12(5):1418–1433.

Allen, C. 2008. Ecohydrology of Piñon-Juniper Woodlands in the Jemez Mountains, New Mexico: Runoff, Erosion, and Restoration. P 65. In: In: Gottfried, Gerald J.; Shaw, John D.; Ford, Paulette L., compilers. 2008. Ecology, management, and restoration of piñon-juniper and ponderosa pine ecosystems: combined proceedings of the 2005 St. George, Utah and 2006 Albuquerque, New Mexico workshops. Proceedings RMRS-P-51. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Allen, C.D., A.K. Macalady, H. Chenchouni, D. Bachelet, N. McDowell, M. Vennetier, T. Kitzberger, A. Rigling, D.D. Breshears, E.H. Hogg, P. Gonzalez, R. Fensham, Z. Zhang, J. Castro, N. Demidova, J.-H. Lim, G. Allard, S.W. Running, A. Semerci, and N. Cobb. 2010. A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. Forest Ecology and Management 259:660–684.

Alexander, L.V., et al. 2006. Global observed changes in daily climate extremes of temperature and precipitation. *Journal of Geophysical Research* 111:D05109.

Amaranthus, M.P., J.M. Trappe, and R.J. Molina. 1989. Long-term forest productivity and the living soil. In *Sustained Productivity of Forest Soils*, edited by S.P. Gessel, D.S. Lacate, G.F. Weetman, and R.F. Powers, pp. 36–52. Proceedings of the 7th North American Forest Soils Conference, Faculty of Forestry Publication, University of British Columbia, Vancouver.

Amaranthus, M.P., D.S. Page-Dumroese, A. Harvey, E. Cazares, and L.F. Bednar. 1996. Soil compaction and organic matter affect conifer seedling nonmycorrhizal and ectomycorrhizal root tip abundance and diversity. Research Paper PNW-RP-494. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

Amman, G.D., M.D. McGregor, R.F. Schmitz, and R.D. Oakes. 1988. Susceptibility of lodgepole pine to infestation by mountain pine beetles following partial cutting of stands. Canadian Journal of Forest Research 18:688–695.

Anderson, M.T. 1980. Water quantity and quality of three adjacent Black Hills watersheds. Unpublished M.S. thesis. Rapid City: South Dakota School of Mines and Technology.

Bagne, K.E., and D.M. Finch. 2010. Response of small mammal populations to fuel treatment and precipitation in a ponderosa pine forest, New Mexico. *Restoration Ecology* 18(S2):409–417.

Bates, C.G., and A.J. Henry. 1928. Forest and stream flow experiment at Wagon Wheel Gap, Colorado. *U.S. Monthly Weather Review* (Supplement 30). Washington, D.C.

Bedunah, D., W. Pfingsten, G. Kennett, and E. Willard. 1988. Influence of stand canopy density to forage production. In: *Proceedings of the Future Forests of the Mountain West: A Stand Culture Symposium*, edited by W.C. Schmidt, pp. 99–107. USDA Forest Service General Technical Report INT-243. Ogden, Utah: Intermountain Research Station.

Bock, C.E., and W.M. Block. 2005. Fire and birds in the southwestern United States. *Studies in Avian Biology* 30:14–32.

Bonan, G.B. 2008. Forests and climate change: forcings, feedbacks, and the climate benefits of forests. *Science* 320:1444–1449.

Bosch, J.M., and J.D. Hewlett. 1982. A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. *Journal of Hydrology* 55:2–23.

Breece, C.R., T.E. Kolb, B.G. Dickson, J.D. McMillin, and K.M. Clancy. 2008. Prescribed fire effects on bark beetle activity and tree mortality in southwestern ponderosa pine forests. *Forest Ecology and Management* 255:119–128.

Breshears, D.D., N.S. Cobb, P.M. Rich, K.P. Price, C.D. Allen, R.G. Balice, W.H. Romme, J.H. Kastens, M.L. Floyd, J. Belknap, J.J. Anderson, O.B. Myers, and C.W. Meyer. 2005. Regional vegetation die-off in response to global-change-type drought. Proceedings of the National Academy of Science 102:15144–15148.

Breshears, D.D., O.B. Myers, C.W. Meyer, F.J. Barnes, C.B. Zou, C.D. Allen, N.G. McDowell, and W.T. Pockman. 2009. Tree die-off in response to global-change-type drought: mortality insights from a decade of plant water potential measurements. Frontiers in Ecology and Environment 7:185–189.

Brown, J.K. 1974. *Handbook for Inventorying Downed Woody Material*. U.S. Department of Agriculture Forest Service General Technical Report INT-16.

Brown, H.E., M.B. Baker, J.J. Rogers, Jr., W.P. Cleary, J.L. Kovner, F.R. Larsen, C.C. Avery, and R.E. Campbell. 1974. Opportunities for Increasing Water Yields and Other Multiple Use Values on Ponderosa Pine Forest Lands. Research Paper RM-129. Rocky Mountain Forest and Range Experiment Station. Fort Collins, Colorado: USDA Forest Service.

Brown, M.W., T.E. Nebeker, and C.R. Honea. 1987. Thinning increases loblolly pine vigor and resistance to bark beetles. *Southern Journal of Applied Forestry* 11:28–31.

Brown, P., and R. Wu. 2005. Climate and disturbance forcing episodic tree recruitment in a southwestern ponderosa pine landscape. *Ecology* 86 (11):3030–3038.

Carey, A.B., and S.M. Wilson. 2001. Induced spatial heterogeneity in forest canopies: response of small mammals. *Journal of Wildlife Management* 65:1014–1027.

Clary, W.P., and P.F. Ffolliott. 1966. *Differences in Herbage-Timber Relationships between Thinned and Unthinned Ponderosa Pine Stands*. Research Note RM-74. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station.

Childs, S.W., S.P. Shade, D.W.R. Miles, E. Shepard, and H.A. Froehlich. 1989. Soil physical properties: importance to long-term productivity. In *Sustained Productivity of Forest Soils*, edited by S.P. Gessel, D.S. Lacate, G.F. Weetman, and R.F. Powers, pp. 53–67. Proceedings of the 7th North American Forest Soils Conference, Faculty of Forestry Publication. University of British Columbia, Vancouver.

Converse, S.J., W.M. Block, and G.C. White. 2006a. Small mammal population and habitat responses to forest thinning and prescribed fire. *Forest Ecology and Management* 228:263–273.

Converse, S.J., G.C. White, and W.M. Block. 2006b. Small mammal responses to thinning and wildfire in ponderosa pine-dominated forest of the southwestern United States. *Journal of Wildlife Management* 70:1711–1722.

Coomes, D.A., and P.J. Grubb. 2000. Impacts of root competition in forests and woodlands: a theoretical framework and review of experiments. Ecological Monographs 70:171–207.

Covington, W.W., and M.M. Moore. 1994. Postsettlement changes in natural fire regimes and forest structure. *Journal of Sustainable Forestry* 21(2)153–181.

Cram, D.S., T.T. Baker, A.G. Fernald, A. Madrid, and B. Rummer. 2007. Mechanical thinning impacts on runoff, infiltration, and sediment yield following fuel reduction treatments in a southwestern dry mixed conifer forest. *Journal of Soil and Water Conservation* 62:359–366.

Crawford, J.A., C.H.A. Wahren, S. Kyle, and W.H. Moir. 2001. Responses of exotic plant species to fires in *Pinus ponderosa* forests in northern Arizona. *Journal of Vegetation Science* 12:261–268.

DeBano, L.F., P.F. Ffolliott, and M.B. Baker, Jr. 2004. Fifty years of watershed management in the Southwest: some lessons learned. In *A Century of Forest and Wildland Watershed Lessons*, edited by G.G. Ice, and J.D. Stednick. Washington, D.C.: Society of American Foresters.

Dietrich, T.L., and J.R. Meiman. 1974. *Hydrologic Effects of Patch Cutting of Lodgepole Pine*. Hydrology Paper 26. Fort Collins, Colorado: Colorado State University.

Douglass, J.E. 1983. The potential for water yield augmentation from forest management in the eastern United States. *Water Resource Bulletin* 19(3):351–358.

Elliott, W.J, D. Page-Dumroese, and P.R. Robichaud. 1996. *The Effects of Forest Management on Erosion and Soil Productivity.* Presented at the Symposium on Soil Quality and Erosion Interaction sponsored by The Soil and Water Conservation Society of America. Keystone, Colorado.

Feeney, S.R., T.E. Kolb, W.W. Covington, and M.R. Wagner. 1998. Influence of thinning and burning restoration treatments on presettlement ponderosa pines at the Gus Pearson Natural Area. *Canadian Journal of Forest Research* 28:1295–1306.

Ffolliott, P.F., and G.J. Gottfried. 1989. *Production and Utilization of Herbaceous Plants in Small Clearcuts in an Arizona Mixed Conifer Forest*. RM-494. USDA Forest Service Rocky Mountain Forest and Range Experiment Station.

Floyd, M.L., M. Clifford, N.S. Cobb, D. Hanna, R. Delph, P. Ford, and D. Turner. 2009. Relationship of stand characteristics to drought-induced mortality in three southwestern piñon–juniper woodlands. *Ecological Applications* 19:1223–1230.

Franzreb, K.E. 1978. Tree species used by birds in logged and unlogged mixed-coniferous forests. *Wilson Bulletin* 90:221–238.

Froehlich, H.A., and D.H. McNabb. 1984. Minimizing soil compaction in Pacific Northwest forests. In *Forest Soils and Treatment Impacts*, edited by E.L. Stone, pp. 159–192. Knoxville, Tennesse: Proceedings of the 6th American Forest Soils Conference.

Fule, P.Z., C. McHugh, T.A. Heinlein, and W.W. Covington. 2001. Potential fire behavior is reduced following forest restoration treatments. In *Ponderosa Pine Ecosystems Restoration and Conservation: Steps Toward Stewardship*, edited by R.K. Vance, C B. Edminster, W.W. Covington, and J.A. Blake, editors, pp. 28–35. Proceedings RMRS-P-22. Ogden, Utah: U.S. Forest Service, Rocky Mountain Research Station.

Fule, P.Z., J.P. Roccaforte, and W.W. Covington. 2007. Posttreatment tree mortality after forest ecological restoration, Arizona, United States. *Environmental Management* 40:623–634.

Gaines, W.L., M. Haggard, J. Begley, J. Lehmkuhl, and A. Lyons. 2010. Short-term effects of thinning and burning restoration treatments on avian community composition, density, and nest survival in the Eastern Cascades dry forests, Washington. *Forest Science* 56:88–99.

Ganey, J.L., and S.C. Vojta. 2011. Tree mortality in drought-stressed mixed-conifer and ponderosa pine forests, Arizona, USA. *Forest Ecology and Management* 261:162–168.

Gary, Howard L. 1975. *Watershed Management Problems and Opportunities for the Colorado Front Range Ponderosa Pine Type*. Research Paper RM-139. Rocky Mountain Forest and Range Experiment Station. Fort Collins, Colorado: USDA Forest Service.

Gaylord, M.L, T.E. Kolb, W.T. Pockman, J.A. Plaut, E.A. Yepez, A.K. Macalady, R.E. Pangle, and N.G. McDowell. 2013. Drought predisposes pinon-juniper woodlands to insect attacks and mortality. *New Phytologist* 1–12.

Gitzen, R.A., S.D. West, C.C. Maquire, T. Manning, and C.B. Halpern. 2007. Response of terrestrial small mammals to varying amounts and patterns of green-tree retention in Pacific Northwest forests. *Forest Ecology and Management* 251:142–155.

Gomez, A., R.F. Powers, M.J. Singer, W.R. Horwath. 2002. Soil compaction effects on growth of young ponderosa pine following litter removal in California's Sierra Nevada. *Soil Science Society of America Journal* 66:1334–1343.

Greacen, E.L., and R. Sands. 1980. Compaction of forest soils — a review. *Australian Journal of Soil Research* 18:163–189.

Griffis, K.L., J.A. Crawford, M.R. Wagner, and W.H. Moir. 2001. Understory response to management treatments in northern Arizona ponderosa pine forests. *Forest Ecology and Management* 146:239–245.

Guarín, A., and A.H. Taylor. 2005. Drought triggered tree mortality in mixed conifer forests in Yosemite National Park, California, USA. *Forest Ecology and Management* 218:229–244.

Gutzler, D.S. 2013. Special feature: sustainability on the U.S./Mexico border, regional climatic considerations for borderlands sustainability. *Ecosphere* 4 (1) 7:1–12.

Gutzler, D.S., and T.O. Robbins. 2011. Climate variability and projected change in the western United States: regional downscaling and drought statistics. *Climate Dynamics* 37:835–849

Harr, R.D. 1983. Potential for augmenting water yield through forest practices in western Washington and western Oregon. *Water Resources Bulletin* 19(3):383–393.

Hayward, G.D., S.H. Henry, and L.F. Ruggiero. 1999. Response of red-backed voles to recent patch cutting in subalpine forest. *Conservation Biology* 13:168–176.

Herrick, J.E., J.W. Van Zee, K.M. Havstad, L.M. Burkett, and W.G. Whitford. 2005. *Monitoring Manual for Grassland, Shrubland and Savanna Ecosystems. Quick Start. Design, Supplementary Methods and Interpretation, 1 and 2*. U.S. Department of Agriculture Agricultural Research Service Jornada Experimental Range, Las Cruces, New Mexico. Tucson: University of Arizona Press. Available at: http://usda-ars.nmsu.edu/JER/Monit_Assess/monitoring_main.php. Accessed December 2008.

Hewlett, J. D., & Hibbert, A. R. 1967. Factors affecting the response of small watersheds to precipitation in humid areas. *Forest hydrology*, 275-290.

Hornbeck, J.W., C.W. Martin, and C. Eagar. 1997. Summary of water yield experiments at Hubbard Brook Experimental Forest. New Hampshire. *Canadian Journal of Forest Research* 27:2043–2052.

Huberty, A.F., and R.F. Denno. 2004. Plant water stress and its consequences for herbivorous insects: a new synthesis. *Ecology* 85:1383–1398.

Hunter, M.E., P.N. Omi, E.J. Martinson, and G.W. Chong. 2006. Establishment of nonnative plant species after wildfires: effects of fuel treatments, abiotic and biotic factors, and post-fire grass seeding treatments. *International Journal of Wildland Fire* 15:271–281.

Hurd, B.H., and J. Coonrod. 2008. *Climate Change and its Implications for New Mexico's Water Resources and Economic Opportunities*. Technical Report 45. Las Cruces, New Mexico: New Mexico State University Agricultural Experiment Station.

Hurteau, S.R., T.D. Sisk, W.M. Block, and B.G. Dickson. 2008. Fuel-reduction treatment effects on avian community structure and diversity. *Journal of Wildlife Management* 75(5):1168–1174.

Intergovernmental Panel on Climate Change (IPCC). 2007. *Climate Change 2007: I. The Physical Science Basis*. New York: Cambridge University Press.

Jacobs, B.F., R.G. Gatewood, and C.D. Allen. 2002. Watershed restoration in degraded piñon-juniper woodlands: a paired watershed study, 1996–1998(9). Unpublished report on file at U.S. Geological Survey (USGS) Jemez Mts. Field Station.

Jactel H., J. Petit, M. Desprez-Loustau, S. Delzon, D. Piou, A. Battisti, and J. Koricheva. 2012. Drought effects on damage by forest insects and pathogens: a metaanalysis. *Global Change Biology* 18:267–276.

Kalies, E.L. 2010. The effects of forest management on small mammal community dynamics in ponderosa pine ecosystems. Doctoral dissertation. Flagstaff, Arizona: School of Forestry, Northern Arizona University.

Kalies, E.L., C.L. Chambers, and W.W. Covington. 2009. Wildlife responses to thinning and burning treatments in southwestern conifer forests: A meta-analysis. *Forest Ecology and Management* 259:333–342.

Kattleman, R., and G.G. Ice. 2004. Dry/cold, wet/warm, and transient snow: Regional differences in forest snow hydrology. In *A Century of Forest and Wildland Watershed Lessons*, edited by George G. Ice and J.D. Stednick. Washington, D.C.: Society of American Foresters.

Keane, R.E. 2008. Surface Fuel Litterfall and Decomposition in the Northern Rocky Mountains. Rocky Mountain Research Station Research Paper, RMRS-RP-70.

Keane, R.E., K.C. Ryan, T.T. Veblen, C.D. Allen, J. Logan, and B. Hawkes. 2002. *Cascading Effects of Fire Exclusion in Rocky Mountain Ecosystems: A Literature Review*. General Technical Report GTR-91. USDA Forest Service, Rocky Mountain Research Station.

Keeley, Jon. E. 2006. Fire management impacts on invasive plants in the Western United States. *Conservation Biology* 20(2):375–384.

Keppeler, E.T., and R.R. Zeimer. 1990. Water yield and summer low flows at Casper Creek in northwestern California. *Water Resources Research* 26(3):1669–1679.

Klenner, W., and T.P. Sullivan. 2003. Partial and clearcut harvesting of high elevation spruce-fir forests: implications for small mammal communities. *Canadian Journal of Forest Research* 33:2283–2296.

Kolb, P.F., and R. Robberecht. 1996. *Pinus ponderosa* seedling establishment and the influence of competition with the bunchgrass *Agropyron spicatum*. International Journal of Plant Sciences 157:509–515.

Koepke, D.F., T.E. Kolb, and H.D. Adams. 2010. Variation in woody plant mortality and dieback from severe drought among soils, plant groups, and species within a northern Arizona ecotone. *Oecologia* 163:1079–1090.

Kolb, T.E., K.M. Holmberg, M.R. Wagner, and J.E. Stone. 1998. Regulation of ponderosa pine foliar physiology and insect resistance mechanisms by basal area treatments. *Tree Physiol*ogy 18:375–381.

Korb, J. 2001. Understory plant community dynamics in southwest ponderosa pine forest restoration. Doctorate dissertation.

Kotliar, N.B., S.J. Hejl, R.L. Hutto, V.A. Saab, C.P. Melcher, and M.E. McFadzen. 2002. Effects of fire and post-fire salvage logging on avian communities in conifer-dominated forests of the western United States. *Studies in Avian Biology* 25:49–64.

Kuenzi, A.M., P.Z. Fule, and C.H. Sieg. 2008. Effects of fire severity and pre-fire stand treatment on plant community recovery after a large wildfire. Forest Ecology and Management 255:855–865.

Laughlin, D.C., and P.Z. Fule. 2008. Wildland fire effects on understory plant communities in two fireprone forests. *Canadian Journal of Forest Research* 38:133–142.

Litschert, S.E., and L.H. MacDonald. 2009. Frequency and characteristics of sediment delivery pathways from forest harvest units to streams. *Forest Ecology and Management* 259:143–150.

Llewellyn, D. and S. Vaddey. 2013. West-Wide climate risk assessment: Upper Rio Grande Impact Assessment. U.S. Department of Interior, Bureau of Reclamation, Upper Colorado Region, Albuquerque Office Area. Albuquerque, NM. 138 pages, plus appendices.

Maloney, P.E., and D.M. Rizzo. 2002. Pathogens and insects in a pristine forest ecosystem: the Sierra San Pedro Martir, Baja, Mexico. *Canadian Journal of Forest Research* 32:448–457.

Manning, J.A., and W.D. Edge. 2004. Small mammal survival and downed wood at multiple scales in managed forests. *Journal of Mammalogy* 85:87–96.

Matchett, J., M. Brooks, A. Halford, D. Johnson, and H. Smith, 2010, Evaluating the effects of pinyon thinning treatments at a wildland urban interface: El Portal, CA.

McCallum, D.A., R. Grundel, and D.L. Dahlsten. 1999. Mountain chickadee (*Poecile gambeli*). Account 453 in *The Birds of North America*, edited by A. Poole and F. Gill. Philadelphia, Pennsylvania: Academy of Natural Sciences, and Washington, D.C.: The American Ornithologists' Union.

McConnell, B.R., and J.G. Smith. 1970. Response of understory vegetation to ponderosa pine thinning in eastern Washington. Journal of Range Management 23:208–212.

McDowell N., W.T. Pockman, C.D. Allen, D.D. Breshears, N. Cobb, T. Kolb, J. Plaut, J. Sperry, A. West, D.G. Williams, et al. 2008. Mechanisms of plant survival and mortality during drought: why do some plants survive while others succumb to drought? Tansley Review. *New Phytologist* 178:719–739.

McCune, B., and J.B.Grace.2002. Analysis of Ecological Communities. MjM Software Design, Gleneden Beach, Oregon.

Metlen, K.M., and C.E. Fiedler. 2006. Restoration treatment effects on the understory of ponderosa pine/Douglas fir forests in western Montana, USA. *Forest Ecology and Management* 222:355–369.

Miller, J.H., and D.L. Sirois. 1986. Soil disturbance by skyline yarding vs. skidding in a loamy hill forest. Soil Science Society of America Journal 50:462–464.

Mitchell, R.G., R.H. Waring, and G.B. Pitman. 1983. Thinning lodgepole pine increases tree vigor and resistance to mountain pine beetle. *Forest Science* 29:204–211.

Moghaddas, E. 2013. Soils. In *Science Synthesis to Promote Resilience of Social-Ecological Systems in the Sierra Nevada and Southern Cascades,* edited by J. Long, C. Skinner, M. North, P. Winter, B. Zielinski, C. Hunsaker, B. Collins, J. Keane, F. Lake, J. Wright, E. Moghaddas, A. Jardine, K. Hubbert, K. Pope, A. Bytnerowicz, M. Fenn, M. Busse, S. Charnley, T. Patterson, L. Quinn-Davidson, and H. Safford. USDA Forest Service Pacific Southwest Research Station. Albany, California: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. Available at: http://www.fs.fed.us/psw/publications/reports/psw_sciencesynthesis2013/psw_sciencesynthesis2013_5_0.pdf. Accessed February 2014.

Moore, M.M., and D.A. Deiter. 1992. Stand density index as a predictor of forage production in northern Arizona ponderosa pine forests. *Journal of Range Management* 45:267–271.

Mueller, R.C., C.M. Scudder, M.E. Porter, R.T. Trotter, III, C.A. Gehring, and T.G. Whitham. 2005. Differential tree mortality in response to severe drought: evidence for long-term vegetation shifts. *Journal of Ecology* 93:1085–1093.

Muzika, R.M., S.T. Grushecky, A.M. Liebhold, and R.L. Smith. 2004. Using thinning as a management tool for gypsy moth: the influence on small mammal abundance. *Forest Ecology and Management* 192:349–359.

Negron, J.F., J.D. McMillin, J.A. Anhold, and D. Coulson. 2009. Bark beetle-caused mortality in a drought-affected ponderosa pine landscape in Arizona, USA. *Forest Ecology and Management* 257:1353–1362.

Nelson, C.R., C.B. Halpern, and J.K. Agee. 2008. Thinning and burning result in low level invasion by nonnative plants but neutral effects on natives. *Ecological Applications* 18(3):762–770.

National Research Council (NRC). 2008. *Hydrologic Effects of a Changing Forest Landscape*. Committee on hydrologic impact of forest management. National Research Council. Washington, D.C.: National Academies Press.

Northern Arizona University Ecological Restoration Institute. 2010. *Guidelines for Managing Small Mammals in Restored Ponderosa Pine Forests of Northern Arizona.* Ecological Restoration Institute Working Paper, No. 23.

Page-Dumroese, D., M. Jurgensen, W. Elliot, T. Rice, J. Nesser, T. Collins, and R. Meurisse. 2000. Soil quality standards and guidelines for forest sustainability in northwestern North America. *Forest Ecology and Management* 138:445–462.

Petrie, M.D., S.L. Collins, D.S. Gutzler and D.M. Moore. 2014. Regional trends and local variability in monsoon precipitation in the northern Chihuahuan Desert, USA. Journal of Arid Environments 103:63-70.

Powers, R.F. 2006. Long-term soil productivity: genesis of the concept and principles behind the program. *Canadian Journal of Forest Research* 36:519–528.

Powers, R.F., D.A. Scott, F.G. Sanchez, R.A. Voldseth, D. Page-Dumroese, J.D. Elioff, and D.M. Stone. 2005. The North American long-term soil productivity experiment: Findings from the first decade of research. *Forest Ecology and Management* 220:31–50.

Raffa, K.F., B.H. Aukema, B.J. Bentz, A.L. Carroll, J.A. Hicke, M.G. Turner, and W.H. Romme. 2008. Cross-scale drivers of natural disturbances prone to anthropogenic amplification: the dynamics of bark beetle eruptions. *BioScience* 58:501–517.

Ramirez, H., A. Fernald, A. Cibils, M. Morris, S. Cox, and M. Rubio. 2008. Variation in herbaceous vegetation and soil moisture under treated and untreated oneseed juniper trees. In *Ecology, Management, and Restoration of Piñon-Juniper and Ponderosa Pine Ecosystems: Combined Proceedings of the 2005 St. George, Utah and 2006 Albuquerque, New Mexico Workshops*, compiled by G.J. Gottfried, J.D. Shaw, and P.L. Ford. Proceedings RMRS-P-51. Fort Collins, Colorado: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Reinhart et al. 1964

Rice, R.M., J.S. Rothacher, and W.F. Megahan. 1972. Erosional consequences of timber harvesting: an appraisal. In Watersheds in Transition, American Water Resources Association Proceedings Series 14, pp. 321–329. Urbana, Illinois.

Riegel, G.M., R.F. Miller, and W.C. Krueger. 1992. Competition for resources between understory vegetation and overstory ponderosa pine. *Ecological Application*s 2:71–85.

Rietveld, W.J., and L.J. Heidmann. 1976. Direct seeding ponderosa pine on recent burns in Arizona. Research Note RM-312. USDA Forest Service Rocky Mountain Forest Range Experimental Station.

Robichaud, P.R., C.H. Luce, and R.E. Brown. 1993. Variation among different surface conditions in timber harvest sites in the Southern Appalachians. In *International Workshop on Soil Erosion: Proceedings, Moscow, September 1993,* pp. 231–241. West Lafayette, Indiana: Center of Technology Transfer and Pollution Prevention, Purdue University.

Robichaud, P.R., L.H. MacDonald, and R.B. Foltz. 2010. Fuel management and erosion. In: *Cumulative Watershed Effects of Fuel Management in the Western United States*, edited by W.J. Elliot, I.S. Miller, and L. Audin, pp. 79–100. General Technical Report RMRS-GTR-231. Fort Collins, Colorado: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Ronco Jr., F., C.B. Edminster, and D.P. Trujillo. 1985. Growth of ponderosa pine thinned to different stocking levels in northern Arizona. Research Paper RM-262. Fort Collins, Colorado: USDA Forest Service Rocky Mountain Forest Range and Experiment Station.

Rogers, T.J. 1995. Insect and disease associates of the pinon-juniper woodlands. In *Desired Future Conditions for Pinonjuniper Ecosystems*, edited by D.W. Shaw, E.F. Aldon, and C. LoSapio, pp. 107–108. General Technical Report RM-GTR-258. Fort Collins, Colorado: USDA Forest Service, Rocky Mountain Forest and Range Experiment Station.

Ryan, M.G. 2011. Tree responses to drought. *Tree Physiology* 31:237–239.

Saab, V.A., J. Dudley, and W.L. Thompson. 2004. Factors influencing occupancy of nest cavities in recently burned forests. *Condor* 106:20–36.

Saab, V.A., and H.D.W. Powell. 2005. Fire and avian ecology in North America: process influencing pattern. *Studies in Avian Biology* 30:1–13.

Sala, A., G.D. Peters, L.R. McIntyre, and M.G. Harrington. 2005. Physiological responses of ponderosa pine in western Montana to thinning, prescribed fire and burning season. *Tree Physiology* 25:339–348.

Sala A., F. Piper, and G. Hoch. 2010. Physiological mechanisms of drought-induced tree mortality are far from being resolved. *New Phytologist* 186:274–281.

Sartwell, C., and R.E. Stevens. 1975. Mountain pine beetle in ponderosa pine: prospects for silvicultural control in second growth stands. *Journal of Forestry* 73:136–140.

Savage, M., P.M. Brown, and J.J. Feddema. 1996. The role of climate in a pine forest regeneration pulse in the southwestern United States. Ecoscience 3 (3):310–318.

Savage, M. 1997. The role of anthropogenic influences in a mixed-conifer forest mortality episode. *Journal of Vegetation Science* 8:95–104.

Savage, M., J. Nystrom Mast, and J.J. Feddema. 2013. Double whammy: high severity fire and drought in ponderosa pine forests of the southwest. *Canadian Journal of Forest Research* 43:570–583.

Schowalter, T.D., and P. Turchin. 1993. Southern pine beetle infestation development: interaction between pine and hardwood basal areas. *Forest Science* 39:201–210.

Seager, R., et al. 2008. Model projections of an imminent transition to a more arid climate in southwestern North America. *Science* 316:1181–1184.

Seidel, D. J., Q. Fu, W. J. Randel, and T. J. Reichler. 2008. Widening of the tropical belt in a changing climate. Nature Geoscience 1:21–24.

Skov, K.R., T.E. Kolb, and K.F. Wallin. 2005. Difference in radial growth response to restoration thinning and burning treatments between young and old ponderosa pine in Arizona. *Western Journal of Applied Forestry* 20:36–43.

Stednick, John D. 1996. Monitoring the effects of timber harvest on annual water yield. *Journal of Hydrology* 176:79–95.

Stephens, S.L., and S.J. Gill. 2005. Forest structure and mortality in an old-growth Jeffrey pine-mixed conifer forest in northwestern Mexico. *Forest Ecology and Management* 205:15–28.

Stoddard, M.T., and C.M. McGlone. 2008. *Effects of Ecological Restoration Alternative Treatments on Nonnative Plant Species Establishment.* In *Fort Valley Experimental Forest—A Century of Research 1908–2008*, coordinated by S.D. Olberding, and M.M. Moore. USDA Forest Service RMRS-P-55. Fort Collins, Colorado: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Stoddard, M.T., C.M. McGlone, P.Z. Fule, D.C. Laughlin, and M.L. Daniels. 2011. Native plants dominate understory vegetation following ponderosa pine forest restoration treatments. *Western North American Naturalist* 71 (2):206–214.

Sullivan, T.P., D.S. Sullivan, P.M.F. Lindgren, and D.B. Ransome. 2005. Long-term responses of ecosystem components to stand thinning in young lodgepole pine forest II: diversity and population dynamics of forest floor small mammals. *Forest Ecology and Management* 205:1–14.

Suzuki, N., and J.P. Hayes. 2003. Effects of thinning on small mammals in Oregon coastal forests. *Journal of Wildlife Management* 67:352–371.

Swanson, F.J., J.L. Clayton, W.F. Megahan, and G. Bush. 1989. Erosional processes and long-term site productivity. In *Maintaining the Long-Term Productivity of Pacific Northwest Forest Ecosystems*, edited by D.A. Perry, R. Meurisse, B. Thomas, R. Miller, J. Boyle, J. Means, C.R. Perry, and R.F. Powers, pp. 67–82. Portland, Oregon: Timber Press.

SWCA Environmental Consultants (SWCA). 2008. Estancia Basin Watershed Health and Monitoring Project: Monitoring Plan Evaluation. Albuquerque: SWCA Environmental Consultants.

———. 2009. Estancia Basin Watershed Health and Monitoring Project: 2008 Annual Report. Albuquerque: SWCA Environmental Consultants.

———. 2010. Estancia Basin Watershed Health and Monitoring Project: 2009 Annual Report. Albuquerque: SWCA Environmental Consultants.

———. 2011. Estancia Basin Watershed Health and Monitoring Project: 2010 Annual Report. Albuquerque: SWCA Environmental Consultants.

———. 2012. Estancia Basin Watershed Health and Monitoring Project: 2011 Annual Report. Albuquerque: SWCA Environmental Consultants.

———. 2013. Estancia Basin Watershed Health and Monitoring Project: 2012 Annual Report. Albuquerque: SWCA Environmental Consultants.

Szaro, R.C., and R.P. Balda. 1986. Relationships among weather, habitat structure, and ponderosa pine forest birds. *Journal of Wildlife Management* 50:253–260.

Thysell, D.R., and A.B. Carey. 2001. Manipulation of density of *Pseudotsuga menziesii* canopies: preliminary effects on understory vegetation. *Canadian Journal of Forest Research* 31:1513–1525.

Troendle, C.A. 1983. The potential for water yield augmentation from forest management in the Rocky Mountain Region. *American Water Resources Bulletin* 19(3):359–373.

Troendle, C.A. 1987. The potential effect of partial cutting and thinning on streamflow from the subalpine forest. Research Paper RM-274. Fort Collins, Colorado: USDA Forest Service.

Troendle, C.A. 1988. Effect of partial cutting and thinning on the water balance of the subalpine forest. In *Proceedings of Future Forests of the Mountain West: A Stand Culture Symposium,* Sept. 29 to Oct. 3, 1986, compiled by W.C. Schmidt. Int.-GTR-243. Missoula, Montana, and Ogden, Utah: USDA Forest Service Intermountain Research Station.

Troendle, C.A., and M.R. Kaufmann. 1987. Influences of forests on the hydrology of subalpine forests.. In: *Management of Subalpine Forests: Building on 50 Years of Research. July 6–9, 1987,* coordinated by C.A. Troendle, M.R. Kaufmann, R.H. Hamre, and R.P. Winokur, pp. 68–78. RM-GTR-149. Silver Creek and Fort Collins, Colorado: USDA Forest Service Rocky Mountain Forest and Range Experiment Station.

Troendle, C.A., and R.M. King. 1985. The Fool Creek watershed, 30 years later. *Water Resources Research* 21(12):1915–1922.

Troendle, C.A., and C.F. Leaf. 1980. Hydrology. Chapter III. In *An Approach to Water Resources Evaluation of Non-Point Silvicultural Sources*. EPA-600/8-80-012. Athens, Georgia: U.S. Environmental Protection Agency.

Troendle, C.A., and J.R. Meiman. 1984. Options for harvesting timber to control snowpack accumulation. In Proceedings of the 52nd Annual Western Snow Conference, pp. 86–97. Sun Valley, Idaho.

Troendle, C.A., M.S. Wilcox, G.S. Bevenger, and L.S. Porth. 2001. The Coon Creek water yield augmentation project: Implementation of timber harvesting technology to increase streamflow. *Journal of Forest Ecology and Management* 143:179.

Troendle, C.A., L.H. MacDonald, C.H. Luce, and I.J. Larsen. 2010. Fuel management and water yield. In *Cumulative Watershed Effects of Fuel Management in the Western United States.* USDA Forest Service RMRS-GTR-231. Available at: http://www.fs.fed.us/rm/pubs/rmrs_gtr231/rmrs_gtr231_126_148.pdf. Accessed February 2014.

U.S. Drought Monitor. 2011. U.S. Drought Monitor, New Mexico, October 4, 2011. Available at: http://droughtmonitor.unl.edu/DM_state.htm?NM,W. Accessed October 7, 2011.

U.S. Forest Service (USFS). 2005. Forest Inventory and Analysis Guide. Version 3.0, with Phase 3 modifications (2006). Available at: http://www.fia.fs.fed.us/library/field-guides-methods-proc/. Accessed 2007.

Van Mantgem, P.J., N.L. Stephenson, J.C. Byrne, L.D. Daniels, J.F. Franklin, P.Z. Fulé, M.E. Harmon, A.J. Larson, J.M. Smith, A.H. Taylor, and T.T. Veblen. 2009. Widespread increase of tree mortality rates in the western United States. *Science* 323:521–524.

Verschuyl, J., S. Riffell, D. Miller, and T.B. Wigley. 2011. Biodiversity response to intensive biomass production from forest thinning in North American forests: a meta-analysis. *Forest Ecology and Management* 261:221–232.

Wallin, K.F., T.E. Kolb, K.R. Skov, and M. R. Wagner. 2004. 7-year results of thinning and burning restoration treatments on old ponderosa pines at the Gus Pearson Natural Area. *Restoration Ecology* 12:239–247.

Wayman, R.B., and M. North. 2007. Initial response of a mixed-conifer understory plant community to burning and thinning restoration treatments. *Forest Ecology and Management* 239:32–44.

Western Regional Climate Center. 2013. Mountainair, New Mexico Weather (295965). Available at: http://www.wrcc.dri.edu/cgi-bin/cliMAIN.pl?nm5965.

White, A.W., E.Z. Zipkin, P.N. Manley, and M.D. Schlesinger. 2013. Simulating avian species and foraging group responses to fuel reduction treatments in coniferous forests. *Forest Ecology and Management* 304:261–274.

White, C.S., and S.R. Loftin. 2000. Response of two semiarid grasslands to cool-season prescribed fire. Journal of Range Management 53:52–61.

Wienk, C.L., C.H. Sieg, and G.R. McPherson. 2004. Evaluating the role of cutting treatments, fire and soil seed banks in an experimental framework in ponderosa pine forests of the Black Hills, South Dakota. *Forest Ecology and Management* 192:375–393.

Wightman, C.S., and S.S. Germaine. 2006. Forest stand characteristics altered by restoration affect western bluebird habitat quality. *Restoration Ecology* 14:653–661.

Williamson, J.R., and W.A. Neilsen. 2000. The influence of forest site on rate and extent of soil compaction and profile disturbance of skid trails during ground-based harvesting. *Canadian Journal of Forest Research* 30:1196–1205.

Wilson, S.M., and A.B. Carey. 2000. Legacy retention versus thinning: influences on small mammals. *Northwest Science* 74:131–145.

Wilson, T.M., and E.D. Forsman. 2013. Thinning effects on spotted owl prey and other forest dwelling small mammals. In *Density Management in the 21st Century: West Side Story*, pp 79–90. PNW-GTR-880.

Wolk, B., and M. Rocca. 2009. Thinning and chipping ponderosa pine changes understory plant communities on the Colorado Front Range. *Forest Ecology and Management* 257(1):85–95.

Wyatt, C. 2013. *How Restoration Thinning Treatments on Conifer Dominated Watersheds Affect the Water Budget. Evidence Based Conservation Systematic Review. Fact Sheet: Restoration Treatments and Their Impacts on the Water Budget.* Flagstaff, Arizona: Northern Arizona University Ecological Restoration Institute.

Yorks, T.E., S. Dabydeen, and P.J. Smallidge. 2000. Understory vegetation –environment relationships in clearcut and mature secondary forest of western Maryland. *Northeastern Naturalist* 7:205–220.

Zausen, G.L., T.E. Kolb, J.D. Baley, and M.R. Wagner. 2005. Long-term impacts of stand management on ponderosa pine physiology and bark beetle abundance in northern Arizona: A replicated landscape study. *Forest Ecology and Management* 218:291–305.

Yoho, N.S. 1980. Forest management and sediment production in the South - a review. *Southern Journal of Applied Forestry* 4:27–36.

Zwolak, R. 2009. A meta-analysis of the effects of wildfire, clearcutting and partial harvest on the abundance of North American small mammals. *Forest Ecology and Management* 258:539–545.

U.S. Department of Agriculture (USDA). 2010. PLANTS database. Available at: http://plants.usda.gov/java. Accessed January 2011.

This page left intentionally blank

APPENDIX A
LIST OF PLANT SPECIES ENCOUNTERED ON FOREST MONITORING STUDY PLOTS

This page left intentionally blank

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

Group/Family	Genus	Species	Code	Common Name	Form	Life History
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
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

Group/Family	Genus	Species	Code	Common Name	Form	Life History
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
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

Group/Family	Genus	Species	Code	Common Name	Form	Life History
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
Poaceae	*Aristida*	*divaricata*	ARDI5	Poverty threeawn	Grass	Perennial
Poaceae	*Aristida*	*purpurea*	ARPU9	Purple threeawn	Grass	Perennial

Group/Family	Genus	Species	Code	Common Name	Form	Life History
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 (2010) PLANTS Database.

This page left intentionally blank

Attachments

DVD with all raw data files along with an electronic .pdf version of the report

Addenda

(SMWS quarterly reports)