SWCA
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ESTANCIA BASIN WATERSHED HEALTH AND MONITORING PROJECT: MONITORING PLAN EVALUATION

Prepared for

ESTANCIA BASIN WATERSHED HEALTH, RESTORATION AND MONITORING STEERING COMMITTEE

CLAUNCH-PINTO SOIL AND WATER CONSERVATION DISTRICT

Prepared by

SWCA ENVIRONMENTAL CONSULTANTS

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Executive Summary

SWCA Environmental Consultants (SWCA) developed this experimental environmental monitoring plan in cooperation with the Claunch-Pinto Soil and Water Conservation District and the Estancia Basin Watershed Health, Restoration and Monitoring Project Steering Committee to assess the environmental effects of forest thinning practices on Estancia Basin forest and watershed health. The principal goals of this monitoring project are to determine what effects forest thinning practices in both ponderosa pine woodlands and piñon/juniper woodlands have on soils, soil surface hydrology, vegetation, and animals. Healthy forests are considered productive, sustainable, and resilient to environmental disturbances at the landscape level, while providing sustainable habitat for wildlife and resources for people. Healthy watersheds maintain sustainable natural (unaltered by human impacts) hydrological processes, surfacewater flow, soil infiltration, natural recharge of subsurface water tables, and sustainable water quality. SWCA designed and begun to implement an experimental monitoring study aimed at determining how standard forest thinning practices in the Estancia Basin affect forest and watershed health. We experimentally impose standard thinning treatments on study plots paired with non-treated control plots in both ponderosa forest and piñon/juniper woodlands in two sub-watersheds of the Estancia Basin. We then use established U.S. Department of Agriculture (USDA) forest and rangeland measurement and monitoring methods to comparatively follow the response of surface soils, hydrology, vegetation including trees, and native animals on those treated and control study plots over time. The duration of this particular study will be five years, including one to two years of pretreatment measurements to establish the pre-existing environmental background status of all study plots. SWCA will conduct data management, and the New Mexico Forest and Watershed Restoration Institute will work with us to provide annual summaries of research findings from this study on their website for public access. The results of this experimental monitoring study will provide information on how current standard forest thinning activities affect forest and watershed health in the Estancia Basin. Given the significance of recent climate change on global and Southwestern ecosystems, this monitoring study will also provide empirical data on the effects of climate change to these particular ecosystems, and for adaptive management to account for such changes relative to forest management in the Estancia Basin watershed.

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1.0 INTRODUCTION

1.1 PURPOSE

This forest and watershed health environmental monitoring plan was prepared in response to the Claunch-Pinto Soil and Water Conservation District's (CPSWCD) request for proposals (RFP) (CPSWCD 2007) soliciting the development and implementation of a comprehensive environmental monitoring program to evaluate the effects of forest thinning activities on watershed function and health as part of the Estancia Basin Watershed Health, Restoration and Monitoring Project. This monitoring plan fulfills the first phase of the monitoring portion of the overall project by providing a guide for subsequent monitoring activities. The New Mexico Water Trust Board granted funding for this project to the CPSWCD.

1.2 SCOPE OF WORK

The scope of work described in the CPSWCD RFP has three parts:

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.

1.3 PROJECT LOCATION

This forest thinning monitoring project is located on the eastern slopes of the Manzano Mountains in Torrance County, New Mexico. This project is located in the western portion of the Estancia Basin watershed on lands administered by the Claunch-Pinto, East Torrance, and Edgewood Soil and Water Conservation Districts (Figure 1.1).

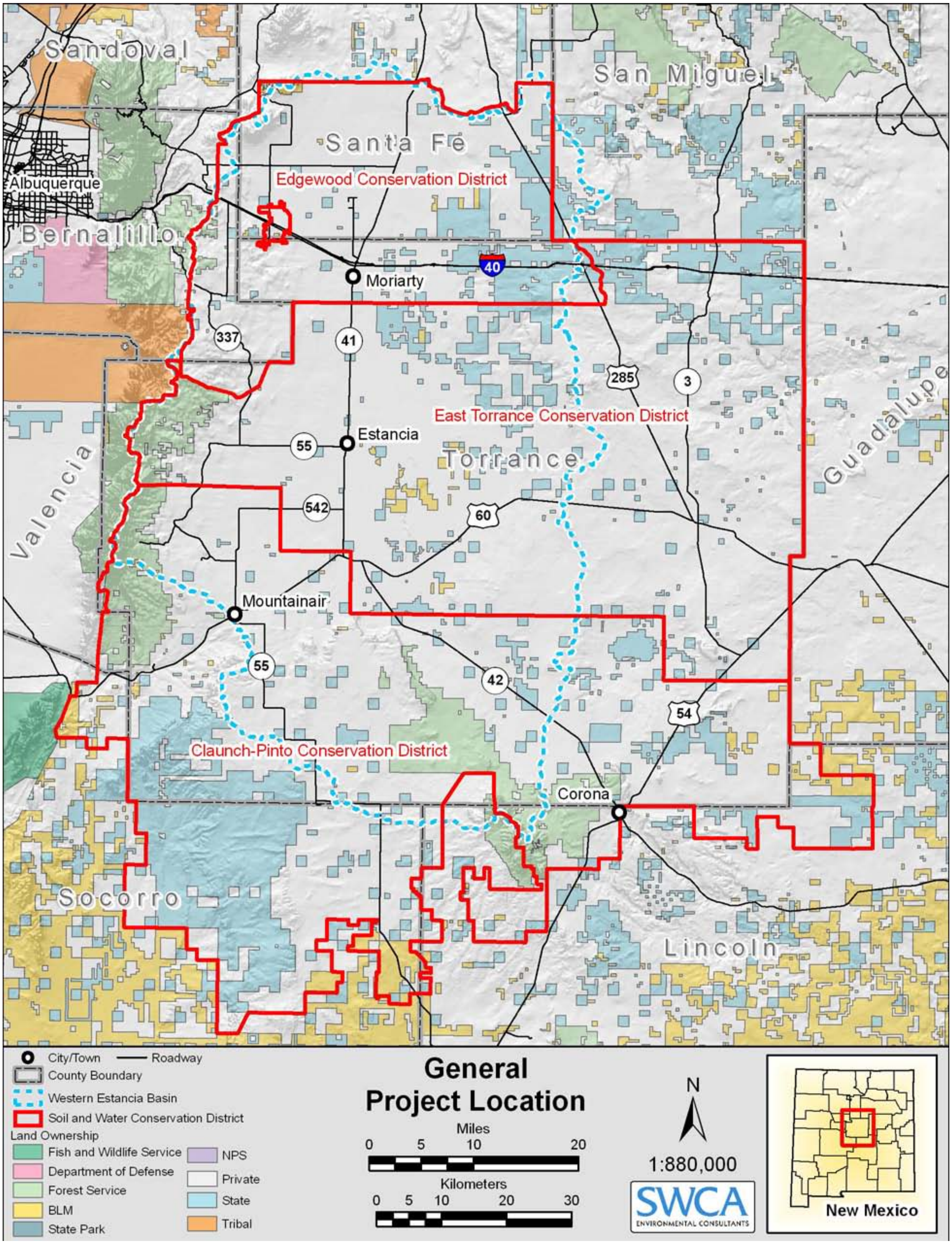

Figure 1.1. Project location.

1.4 GOALS OF ENVIRONMENTAL MONITORING

Environmental or ecological monitoring is becoming an important aspect of natural resources management and environmental restoration (Elzinga et al. 2001; Spellerberg 2005; Herrick et al. 2005, 2006). The primary objective of environmental monitoring is to provide data on parameters or elements (e.g., plant production, soil erosion, species composition, etc.) that are potentially affected [either "with the potential to be affected" or "that may be affected"] by some type of environmental treatment or impact so that the effects of that environmental impact on those parameters or elements can be evaluated objectively (Spellerberg 2005). Monitoring has become especially important for environmental restoration as a means of evaluating the effectiveness of restoration treatments (Herrick et al. 2006). Monitoring-study design for natural resources management should include several important steps or considerations (Herrick et al. 2005). The first two steps should occur during the first year of a monitoring study. The first step is to define management and monitoring purpose and objectives. The second step is to select monitoring sites and parameters or elements to measure. The third step is the actual short-term monitoring or collection of measurement data for the next 2–5 years. This step includes the collection, analysis, and interpretation of monitoring data and adjustment of monitoring methodology as needed based on short-term findings (e.g., adaptive management if monitoring methods are achieving desired goals). Finally, long-term monitoring continues after five years. Long-term monitoring likely will lead to changes in overall resource management strategies based on findings from the monitoring data. Such changes in management strategies may also lead to changes in monitoring objectives and design. Both management and monitoring objectives are likely to evolve over the long term, as both the system of interest and resource management needs change.

1.5 FOREST AND WATERSHED HEALTH MANAGEMENT IN THE SOUTHWEST

Ponderosa pine and mixed-conifer communities in the Southwest have experienced major changes in ecological structure, composition, and process over the last century (Cooper 1960; Covington et al. 1997; Fulé et al. 1997; Allen et al. 2002). Before Euroamerican settlement, Southwestern ponderosa pine woodlands were composed of low-density, park-like stands (Covington and Moore (1994a) with dense grass understory and highly flammable leaf litter (Stone et al. 1999). Historically these forests experienced frequent low intensity wildfire (Kaufmann et al. 1998), creating heterogeneous forest spatial patterns at local and landscape scales (Allen et al. 2002). Disruption to the natural fire regime, harvesting, and intensive grazing practices of these southwestern forests has drastically altered their historic structure and has made them extremely vulnerable to unnaturally severe stand-replacing fires (Covington and Moore 1994b; Swetnam et al. 1999), insect outbreaks, pine water stress (Kolb et al. 1998), and other deviations from historic conditions (Covington et al. 1997; Allen et al. 2002; Friederici 2003; Moore et al. 2004).

Fire management and intensive grazing have also contributed to the aggressive expansion of one-seed juniper (*Juniperus monosperma*) and piñon pine (*Pinus edulis*) woodlands (piñon-juniper [PJ] woodlands) into surrounding sagebrush shrublands and grasslands throughout arid southwestern regions (Young and Evans 1981; Evans 1988; Tausch and Tueller 1990; Davenport and Wilcox 1995; West and Young 2000; Brockway et al. 2002; Baker and Shinneman 2004). Like ponderosa pine communities, these PJ woodlands exhibit occupancy and structure that far exceed their natural range of variation (Gottfried et al. 1995). In many watersheds throughout the Southwest, over 90 percent of ponderosa pine woodlands are considered at high risk of crown fires because of dense structure, unnatural fuel bed depth, and accumulated fuels (Covington and Moore 1994a, 1994b; Allen et al. 2002). Similarly, a large percentage of PJ woodlands exhibit retrogressive characteristics such as limited understory vegetation and elevated soil erosion (Brockway et al. 2002).

Concerns over the degradation of forests throughout the nation and the frequency of major wildfire have led to increased interest in restoring western forests (Stone et al. 1999), a development that has been strengthened by a growing political view that restoration is necessary and urgent (Covington et al. 1997, Kloor 2000, Jenkins 2001, Allen et al. 2002). The U.S. National Fire Plan (2000), the U.S. Ten-year Strategy Plan (2001), the U.S. Healthy Forest Initiative (2002), and the Healthy Forest Restoration Act (2003) are all federal initiatives with a common message to mandate restoration of degraded forests and promote forest health. Millions of hectares of public lands have been targeted for restoration by federal, state, and local agencies (U.S. Forest Service [USFS] 2000).

There have been claims that restoration with the goal of improving forest and watershed health is ecologically inappropriate because of the difficulty of defining indicators of optimal health (Wicklum and Davies 1995). Despite this, managers need to have the ability to assess the health of a system in order to provide management goals. Kolb et al. (1998) suggest that healthy forests could be defined as having: (1) the physical environment, biotic resources, and trophic networks to support productive forests during at least some seral stages; (2) resistance to catastrophic change and/or the ability to recover from catastrophic change at the landscape level; (3) a functional equilibrium between supply and demand of essential resources (water, nutrients, light, growing space) for major portions of the vegetation; and (4) a diversity of seral stages and stand structures that provide habitat for many native species with all essential ecosystem processes.

An evolving idea that forest health encompasses ecosystem and utilitarian components has led to the acceptance of the following definition of forest health by the U.S. Forest Service (USFS) Southwestern Region: "Forest health is a condition wherein a forest has the capacity across the landscape for renewal, for recovery from a wide range of disturbances, and for retention of its ecological resiliency, while meeting current and future needs of people for desired levels of values, uses, products, and services" (Twery and Gottschalk 1996). The Internet websites: U. S. National Fire Plan (2005), U. S. Ten-year Strategy (2001), Western Governer's 10-year Implemental Plan (2002), U. S. Healthy Forest Initiative (2002), Healthy Forests and Rangelands (2008), New Mexico Healthy Forest Plan (2004), and the New Mexico Forest and Watershed Restoration Institute (2008), listed in the References Cited section below provide background and current information on various aspects of national and regional forest and watershed health relative to the scope of this monitoring project.

1.5.1 Southwestern Ponderosa Pine Woodlands and Piñon/Juniper Woodlands

A major goal of all ponderosa pine/mixed-conifer and PJ restoration is to renew ecosystem structure and function within a range of natural variability (Landres et al. 1999) and reverse unhealthy forest characteristics. Covington et al. (1997) provide a classification of unhealthy ponderosa pine as having: (1) decreasing soil moisture and nutrient availability; (2) decreasing growth and diversity of herbs and woody plants; (3) increasing mortality in the oldest age class; (4) decreasing stream and spring flows; and (5) increasing fire severity and size. PJ communities are frequently characterized by seasonal water deficits, low soil fertility, shallow rocky soils, and low species diversity (Gottfried et al. 1995); herbaceous cover is also known to decline as PJ crown cover increases (Arnold et al. 1964, Milton et al. 1994).

Restoration for both communities usually includes some combination of reducing high-density stands through thinning, reintegrating natural disturbance through prescribed burning, and increasing species diversity and abundance of native herbaceous understory (Covington et al. 1997; Brockway et al. 2002; Korb et al. 2003 Vigorous scientific debate has emerged, however, as to the potential effects of various restoration treatments on biotic and abiotic resources (Allen et al. 2002). The availability of funding for fuel reduction and the urgency with which many treatments are implemented has raised questions of whether treatments are ecologically sensitive (Allen et al. 2002).

In 2004 the Southwest Forest Health and Wildfire Prevention Act (2004) [PL 108-317] was passed by the U. S. Congress, outlining measures to be taken to reduce the risk of wildfire to forests and communities in the Southwest. Through thinning and prescribed fire treatments, this act aims to improve wildlife habitat and biodiversity of forest communities; increase tree growth and grass, forb, and shrub productivity; enhance watershed values; and provide a basis for economically and environmentally sustainable uses. To respond to concerns of thinning and to provide scientific support for restoration treatments, the Southwest Forest Health and Wildfire Prevention Act established Southwestern Ecological Restoration Institutes, at universities in three states: Northern Arizona University, Colorado State University), and New Mexico Highlands University.

At the state level the New Mexico Forest and Watershed Health Plan (2004) (NMFWHP), governed by the New Mexico Forest and Watershed Planning Committee (NMFWPC), was enacted to promote improved forest and watershed conditions in New Mexico through the promotion of ecological integrity, natural process, and long-term resiliency. Among the many goals of this plan was the development of a science-based range of flexible guidelines and protocols for ecosystem restoration practices that achieve effective and ecologically sound results. Integral to this process was promotion of effective long term monitoring to inform and modify restoration strategies and guide management.

Although there is much agreement among scientists as to the need for restoration of southwestern forests, the most appropriate management approach remains the subject of debate (Covington et al. 1997; Landres et al. 1999; Agee et al. 2000; Heinlein et al. 2000; Allen et al. 2002). Allen et al. (2002) developed 16 broad principles for the restoration of ponderosa pine woodlands, examples of which are reducing the threat of crown fire, utilizing existing forest structure, assessing cumulative effects of treatment, establishing monitoring and research programs, and implementing adaptive management. One of the most well established practices is for managers to strive to emulate the natural range of variability in conditions by including patchiness and forest diversity at multiple scales (Landres et al. 1999; Swetnam et al. 1999; Allen et al. 2002).

In many studies restoration entails reinstating frequent low intensity fires (Abella and Covington 2004) through a process-driven restoration approach (Heinlein et al. 2000). It is well accepted, however, that before widespread application of prescribed fire, mechanical treatments are needed to thin post-settlement-origin trees (<120 years) using what Heinlein et al. (2000) have called a "structural restoration approach." A common method for restoration thinning is a "thin from below" treatment, where stand density is reduced through the removal of smaller-diameter trees (up to 5 inches diameter at breast height [DBH that act as ladder fuels in the event of wildfire (Fulé et al. 2002). "Thin from below" treatment can be applied to reconfigure historic stand structures and spatial patterns (Covington et al. 1997), as this method retains both old-growth trees and appropriate diameter class distribution (Kaufman et al. 1998). No one prescription should be used in the restoration of ponderosa pine woodlands, however; instead, a variety of restoration approaches should be followed (Moore et al. 1999; Noss et al. 2005).

1.6 ENVIRONMENTAL IMPACTS OF SOUTHWEST FOREST RESTORATION/THINNING

1.6.1 SOILS

Although a number of studies have investigated how forest thinning affects aboveground ecosystem components, little is known about the effect of restoration on ecosystem function or biogeochemical constituents (fluxes of water, carbon, and other nutrients) (Kaye et al. 1999; Kaye et al. 2005). Understanding of these belowground processes is important because they ultimately influence site productivity and initial composition and trajectory of the understory community. Kaye et al. (2005) studied carbon, nitrogen, and phosphorus fluxes in the two years following thinning treatments in northern Arizona. A full restoration treatment was completed that involved whole-tree harvest of all post-settlement trees (post-1876). These plots were then compared to control sites. Stratified sampling by canopy cover was carried out within each plot. Soil carbon and nitrogen were measured using modified resin-core incubation (Binkley and Hart 1989) in 6-month intervals from May 1995 to May 1997. The authors found that restoration had little impact on the above fluxes at a landscape level. Lower pine foliage and fine root fluxes of carbon, nitrogen, and phosphorus in treated plots compared to controls were approximately balanced by higher fluxes of the above constituents in wood and herbaceous plants following treatment.

Increases in net carbon efflux and nitrogen pools following restoration of ponderosa pine have been noted by other authors (Covington and Sackett 1992; Kaye and Hart 1998). These studies were short term and attributed greater flux values to: (1) increased soil temperatures; (2) increased decomposition of roots following bole removal; (3) decreased competition between microbes and plants for soil resources; and (4) improved substrate due to the increase of high quality herbaceous litter inputs and reduction in low-quality pine litter (Hart et al. 2005). Grady and Hart (2006), however, suggest that over a longer period of 6–7 years post treatment, thinned stands have lower in situ annual rates of nitrogen mineralization than unmanaged stands. The sizes of soil, microbial carbon, and nitrogen pools generally declined with decreases in litter-fall, with thinned stands (particularly those then treated with prescribed fire) exhibiting the lowest values compared to controls. The authors and others acknowledge that long-term effects on nitrogen and carbon are variable between sites, being dependent upon the rate of recovery of vegetation inputs. Because microbial activity is closely coupled to aboveground inputs of carbon in southwestern ponderosa pine, treatments that reduce canopy cover and hence litter-fall would intuitively be expected to reduce the size and activity of the soil microflora (Deluca et al. 2002, Hart et al. 2005, Grady and Hart 2006). Grady and Hart (2006) suggest that the differences observed between their study and others could be a result of initial stand conditions and the intensity of the treatment, both of which would affect the magnitude and direction of the impact on soil processes. They recommend that restoration be applied to areas where there is currently low cover of herbaceous vegetation; in this way restoration would see increased herbaceous regeneration compared to pre-treatment levels and therefore increased substrate quality, microbial biomass, net soil carbon dioxide efflux, and rate of nitrogen mineralization and recovery of grass communities.

Covington et al. (1997) investigated the effects of thinning treatments (with burning) on vegetation and soils of an Arizona ponderosa pine woodland. Using 55 replicated circular plots distributed among four different overstory strata, the authors monitored species composition, cover, and productivity in treated and control stands. The authors found that treated plots exhibited higher soil moistures and temperatures than occurred in the control. These factors, which were attributed to less transpiration and less interception of precipitation in thinned plots, are thought to lead to increased fine-root production, litter decomposition, and nitrogen mineralization. The authors hypothesize, therefore, that treated stands (thinned with/without burning) would in time exhibit increased herbaceous production, increased tree growth, and greater resistance to disease and drought.

In addition to their effects on soil biochemistry, thinning treatments have also been cited as causing increased rates of soil erosion in southwestern forests (MacDonald and Stednick 2003). From a review of studies throughout the Southwest region, MacDonald and Stednick (2003) propose that reducing basal areas by thinning in ponderosa pine and mixed conifer can lead to increased sediment yield and soil erosion. Whicker et al. (2006) reaffirm that the removal of vegetation and disturbance of soils during thinning treatments can accelerate erosion of soils because of the non-linear relationship between ground cover and erosion rate for both wind and water erosion. This erosion, however, is thought to be short lived and will depend upon the re-establishment rate of herbaceous understory vegetation (Madrid 2005).

A growing number of studies also assess the impact of PJ removal on soil chemistry and stability. Nutrients in semi-arid woodlands, shrublands, and grasslands (typical of the Southwest) are generally clustered in association with vegetation in a stippled pattern across the landscape. In contrast, surrounding inter-canopy areas thought to be resource depleted have lower infiltration capacity (Schlesinger et al. 1990) and are prone to soil erosion (White et al. 1997, Brockway et al. 2002). The sparse understory that has become a growing characteristic of modern-day PJ woodlands has caused concern for the productivity of associated soils because of the risk of soil erosion and nutrient losses (Davenport et al. 1996). Blackburn (1975) and Roundy et al. (1978) both highlight studies showing that inter-canopy spaces generate significantly more runoff and erosion than canopy spaces. If a site becomes over-degraded through loss of cover, then greater connectivity between inter-canopy spaces and subsequent increases in runoff rates may result in system-wide gully formation (Davenport et al. 1996).

An important aspect of monitoring for erosion effects of thinning is the scale on which the study is focused as Wilcox and Breshears (1995) emphasized in a study of thinned PJ woodlands in northern New Mexico. On a within-slope scale, monitoring of runoff and erosion found substantial movement of sediment and water, but this movement was found to be negligible over a larger hillslope scale. This finding highlights the fact that even though treatments may cause major localized disturbance of soils or other ecosystem components, only minimal effects may be seen on the larger watershed scale. Other watershed studies have yielded similar findings demonstrating diminishing runoff and erosion from thinning treatments as spatial scale is increased (Davenport and Wilcox 1995).

1.6.2 HYDROLOGY

Ecological and hydrological processes are interrelated in water-limited environments (Middleton and Thomas 1997) such as the ponderosa forests and PJ woodlands of the Southwest. Ludwig et al. (2000) suggest that a positive feedback or self-reinforcing mechanism links water and vegetation in these environments and changes to canopy through different forms of disturbance, which in turn alters the hydrology of the system. Densely forested watersheds, characteristic of the degraded southwestern ponderosa pine woodlands, have been linked to decreasing total stream-flows, peak flows, base flows, and overall water yield (Trimble and Weirch 1987, Ffolliott et al. 1989, Madrid 2005). PJ invasion can also have significant impacts on the hydrological cycle by promoting soil compaction, decreasing infiltration, increasing surface runoff, increasing interception of precipitation, increasing evapotranspiration, and reducing soil moisture (Angell and Miller 1994).

Ponderosa pine woodlands, and to a lesser extent PJ woodlands, have for the last 20 years or so been the focus of restoration under the auspices of water resource management, where treatments are targeted at restoring pre-settlement hydrology and water yield (Stednick 1996). Many studies have attempted to quantify the hydrologic effects of such restoration, with varying results (Keppeler and Zeimer 1990; MacDonald and Stednick 2003; Ice and Stednick 2004).

Most water-yield studies have used a paired catchment approach to assess the effect of vegetation removal (Hibbert 1967; Burgy and Papazafiriou 1971). Time-trend studies have also been completed; however, they are often criticized as having no climatic control to separate vegetal cover effects from climatic effects (Whitehead and Robinson 1993). In 1967 Hibbert made the following observations regarding forest disturbance effects on water yield: (1) reduction of forest cover increased water yield; (2) establishment of forest cover (afforestation) decreased water yield; and (3) response to treatment is highly variable. More recent studies (over 94 in number) have had similar findings (Stednick 1996), with the additional claim that the magnitude of change in water yield is most strongly related to the amount of precipitation and the intensity of the treatment (Troendle et al. 2006). Water-yield increases in southwestern arid forests, for example, are lower than in northwestern forests following similar treatment intensities (Troendle et al. 2006).

Varying forest cover types have been found to strongly influence the degree of water yield post-harvest, with the greatest yield of water coming from coniferous forests (40 mm) with a 10% change in forest cover) compared to deciduous forest (25 mm with a 10% change in forest cover) after treatment (Bosch and Hewlett 1982). The intensity of the thinning is believed to govern hydrologic response post-treatment (Troendle and King 1987). The most significant (and more easily measured) increases in water yield in ponderosa pine and mixed-conifer were recorded following treatments where forest cover was reduced by more than 20% (Stednick 1996). Troendle et al (2006) go so far as to say that if this threshold in basal area reduction is not met, then effects of thinning on water yield will be negligible with maybe only slight increases observed during wet years.

One of the most researched sites for water-yield studies has been the Beaver Creek Watershed, Coconino National Forest, Arizona (Brown et al. 1974; Clary et al. 1974; Baker 1982, 1986). At this experimental forest site, researchers have found that thinning of ponderosa pine woodlands (with reductions in basal area up to 120 square feet/acre) generate stream-flow increases of 35% (Johnson 1996). From a review of studies throughout the Southwest, MacDonald and Stednick (2003) also report water-yield increases in ponderosa pine following thinning, but they caution that these peaks in forest hydrology are often short lived, decreasing as the forest floor becomes re-established. The persistence of increased water yield observed at Beaver Creek was found to vary by treatment intensity: completely cleared sites had statistically significant water yield increases for 7 years, declining with the onset of Gambel oak and herbaceous growth; light overstory removal maintained increased water yields for 6 years; and heavy overstory removal maintained increased water yields for 10 years. Water-yield increases were also more prolonged on slopes with southern exposures, compared to north-facing treatment areas (Baker 1986). MacDonald and Stednick (2003) suggest that changes to water yield from thinning are likely to be short lived in arid zones and that detectable changes would be significant only below the treatment area, becoming diluted in the downstream direction. This effect was also observed by Troendle and King (1987) in a Colorado study where stream flow increased significantly in a sub-basin below a treatment area but was barely detectable a few hundred meters downstream, as the harvest made up only a fraction (5.6%) of the wider watershed.

Changes to water yield following thinning have been attributed to a number of factors. In a study at the Fraser Experimental Forest in Colorado (Troendle and King 1985), 30% of observed water-yield increases were attributed to a decrease in interception and a resultant increase in water held in snowpack. Fifty percent of the increased yield was attributed to reduced evapotranspiration during the summer months and the corresponding reduction in melt-water used for soil recharge in the spring months. The remaining 20% of yield increase was attributed to reduced evapotranspiration losses during April and May. The results observed by Troendle and King (1985) are closely related to the climatic regime of the site, an observation also made by MacDonald and Stednick (2003). They say, from reviewing studies of paired catchments, that an annual rainfall threshold of 18–19 inches (450 mm–500 mm) is required in order to detect an increase in runoff as a result of removal of vegetation (Bosch and Hewlett 1982; Troendle et al. 2006).

Because of the huge variability in climate, it should be acknowledged that water-yield studies across states are not always transferable. Most New Mexico runoff is a result of intense summer thunderstorms of short duration (Wilcox et al. 1996; Reid et al. 1999; Wilcox et al. 2003), while Arizona, for example, experiences more winter precipitation and snowmelt (Davenport and Wilcox 1995). The relationship between precipitation, infiltration, storage, and runoff is complex and often site and vegetation specific. Lateral subsurface flow, for example, is rarely found in PJ communities, in contrast to findings in ponderosa pine communities in adjacent study areas (Wilcox et al. 1996; Wilcox and Breshears 1997). In areas such as New Mexico, where "infiltration excess overland flow" is the dominant runoff process, researchers warn against basing runoff predictions on precipitation measurements, as the relationship between precipitation and runoff is often likely to be poor. Infiltration is also linked largely to forest floor conditions, slash, and debris after thinning, as well as aspect and slope (Gifford 1975; Breshears et al. 1995; Davenport and Wilcox 1995).

Given the many variables involved in water-yield studies, some authors have depended on eco-hydrologic modeling to explain the hydrologic response of forests to thinning (Troendle and King 1987). Ludwig et al. (2000) developed the conceptual framework of “resource conserving” woodlands. This framework has been applied to PJ woodlands throughout the Southwest, including northern New Mexico (Wilcox et al. 2003). The basic premise of this concept is that in some semi-arid areas, resources (water and nutrients) are naturally distributed from source areas (bare patches or inter-canopy) to sink areas (vegetation patches) and stay maintained within the system. Any disturbance that occurs, however, may cause the system to become “leaky” and therefore less efficient at trapping runoff, leading to water and nutrient losses (Ludwig and Tongway 2000). This model is thought to be particularly appropriate for PJ woodland because there is often lateral variation in soil moisture governed by the presence/ absence of juniper and piñon canopies (Breshears 1993); inter-canopy soil receives increased precipitation, and the vegetative clusters access this inter-canopy water through runoff and root processes.

If the goal of restoration is to maintain water and nutrients within the vegetation community, Ludwig et al. (2000) propose that treatments be designed to create bands of vegetation that trap upslope runoff instead of increasing connectivity of inter-canopy spaces. In this way, restoration would create woodlands that help perpetuate the idea of a resource-conserving woodland.

1.6.3 VEGETATION

Ponderosa pine restoration and the treatment of PJ expansion in the Southwest have some common goals. Both communities have, since Euro-American settlement, undergone significant increases in density and structure beyond historic levels. As a result, understory vegetation has become sparse and is no longer sufficient as an agent and medium in the spread of low intensity surface fires that promote open stand characteristics with dense grassland. Restoration generally seeks to return both communities to conditions that would have been typical before the advent of fire suppression and domestic grazing, including the regeneration of rich herbaceous species diversity and ground cover. To obtain such results, some places can require drastic treatment of current stands. As an example, Fowler and Witte (1987) suggest that at least 2/3 of PJ overstory crown must be removed in order for any increase in herbaceous vegetation to occur in southwest PJ woodlands. Studies of ponderosa pine restoration are becoming increasingly common in forest management of the Southwest and other regions, and treatment of PJ woodlands is growing as a research topic amongst the rangeland and watershed disciplines.

In 2004, Abella and Covington studied the effect of thinning and burn treatments on southwestern ponderosa pine of Coconino National Forest, northern Arizona. The authors used point intercept sampling to monitor understory species percent cover and richness. There were twelve 14-hectare (ha) treatment areas within the study site, each with differing treatment intensities (thinned to varying stand densities). Twenty plots were set up in each treatment area and understory and substrate were sampled on a 50-meter (m) transect from the center of each plot using point intercept. From the study, the authors found the treatment areas to exhibit increasing frequency of exotic species with increased intensity of treatment. Increases in the frequency of native species, however, were only noted in the high intensity treatment areas. The authors suggest a treatment threshold has to be met before native understory species richness are increased by thinning. This study also acknowledged the importance of large plot size (>1m²) for increased species detection; this is especially significant where detection of rare plants is a focus of the study.

A similar study was carried out on thinning of PJ by Brockway et al. (2002) in the Mountainair ranger district of central New Mexico. This study sought to determine the effect of overstory removal of PJ and differing slash treatments on post treatment response by herbaceous vegetation. In 1996, 94–97% of (predominantly juniper) PJ woodland was removed from treatment areas. Residual slash was scattered throughout plots, clumped at old stump bases, or completely removed from the plot. Residual PJ density was 15 trees/ha and overstory cover was reduced from ~25% to <2%. In two seasons following treatment, grass cover increased from 9% to ~38%. The greatest increase in understory biomass occurred in plots where slash was clustered (215%), followed by plots with total slash removal (141%), and then plots with scattered slash (132%); control plots saw a biomass increase of approximately 70%. Blue grama (*Bouteloua gracilis*), black grama (*Bouteloua eripoda*), and broom snakeweed (*Gutierrezia sarothrae*) were the dominant understory species. Understory biomass increased from approximately 300/400 kg/ha to 900/1000 kg/ha. Since measurements of nutrient levels revealed little to no change post-treatment, the authors suggest that increased understory biomass following thinning could be attributed to increased availability of water. Other studies have also shown increased understory vegetation response to decreased competition from PJ to be largely water related (Clary et al. 1974; Miller and Wigland 1994). The increases observed at this site support well-recognized relationships between cover and understory growth and were similar to findings at other sites throughout the Southwest (Pieper 1995; Tausch and West 1995; White et al. 1997).

Korb et al. (2003) and Korb and Springer (2003) evaluated certain commonly used monitoring techniques for ponderosa pine restoration. They note that in monitoring vegetation it is important to have a sampling technique that will reliably and precisely detect change in understory herbaceous productivity. Understory vegetation is especially important in the restoration of ponderosa pine woodlands because of the role the understory plays in transmitting low severity wild fire, integral to restoring these communities. The authors selected four sampling techniques: point intercept, Daubenmire transects, belt transect, and modified Whitaker plots, which were tested on thinning treatments in northern Arizona. The study showed that each technique exhibited significant variation in species detection, with point intercept recording the lowest species diversity and modified Whitaker plots being the most robust at detecting species. The fact that the modified Whitaker plot captured the most species supports the concept that an increase in sampling area will increase species richness detection (Rapson et al. 1997). For large treatments, however, the time taken to complete a modified Whitaker plot could negate its use in terms of time and cost. Although point intercept plots have been found to have low detection of species in this (Korb et al 2003) and other studies (Kinsinger et al. 1960; Stohlgren et al. 1998; Etchberger and Krausman 1997), the use of contiguous point intercept has been shown to accurately measure species diversity following ponderosa pine restoration (Abella and Covington 2004).

In 2000 Lynch et al., using releve plots (vascular plants), belt transects (trees), point intercept (biotic and abiotic cover), and planar intercept (surface fuels) methods, recorded an overall decline in herbaceous plants on thinned and unburned stands of ponderosa pine. The thinned stands also supported fewer species of woody plants, forbs, and graminoids. Stands that underwent thinning followed by prescribed fire, however, exhibited increased forbs and grasses but declines in woody species. Although thinned stands exhibited lower stand densities, the volume of fine fuels was increased in thinned only stands, and therefore these stands had greater downed fuel loads, a fire risk factor. The authors claimed that thinning only did not improve forest floor conditions or increase diversity; they suggested that thinning alone cannot restore the ecological structure and processes that formerly characterized ponderosa pine woodlands of the Southwest and that restoration towards historic stand structure and diversity should include both thinning and burning.

Thinning treatments can also affect vegetation physiology. Kolb et al. 1998 suggested that under the current condition of southwestern ponderosa pine woodlands, there was growing evidence of increased pine water stress. In 1999 Stone et al. carried out a study of treated ponderosa pine in northern Arizona to determine the effects of thinning on pre-settlement residual tree vigor. This study assessed pre- and post-treatment soil volumetric water content, xylem water potential of 1-year-old needles, and total foliar nitrogen and phosphorous content. The results found treatment plots to exhibit increased water availability in the form of soil volumetric water content particularly in the upper horizons (1–15 cm) as compared to control plots. This was attributed to reduced tree root density in the upper horizons following removal of post-settlement trees, a variable that has been reported in previous studies (Aussenac and Granier 1988; Cregg et al. 1990). Furthermore, xylem water potential was found to be higher from foliar samples taken from thinned stands as compared to control plots. Increased canopy growth and increased uptake of water, nitrogen, and carbon are suggested indicators of greater tree vigor that result from thinning (Stone et al. 1999).

In general, the productivity of the understory of southwestern forests has been found to increase after dramatic tree thinning (Abella and Covington 2004; Huffman and Moore 2004). A study by Korb and Springer (2003) in southwestern ponderosa pine found treatments resulted in higher species richness, diversity, and cover on treated than untreated sites, though the degree to which understory vegetation responds to increased levels of light, water, and nutrients is significantly affected by year-to-year variability in climate (Korb et al. 2003). Davenport and Wilcox (1995) suggest that the relationship between overstory cover and understory density (i.e., increased cover leads to decreased herbaceous density) is especially true of xeric sites (south facing and shallow soil). Moreover, from studies in northern New Mexico, Davenport and Wilcox (1995) conclude that a moisture threshold exists below which PJ and ponderosa pine expansion is especially damaging for herbaceous vegetation. Monitoring must be continued over a long enough period to detect responses to the treatment (Sutter 1996). As discussed above, it is particularly difficult to detect change in foliar cover in a short-term study due to the natural variation in herbaceous response to yearly climate deviations (Korb et al. 2003). A long-term study helps to separate climate driven variability from treatment induced vegetation response.

1.6.4 ANIMALS

The degradation of ponderosa pine woodlands has been blamed for the decline of a number of vertebrate animal species because of declining habitat suitability (Dodd et al. 2006) Despite the obvious need to restore these degraded ecosystems, there is concern that ponderosa pine restoration may reduce the viability of metapopulations of sensitive species through habitat alteration and fragmentation (Converse et al. 2006). Changes in forest overstory and structure could for example alter forest microclimates, which are an important component of habitat quality for many wildlife species, influencing survival, reproductive success, and behavior (Meyer and Sisk 2001). Germaine and Germaine (2002) discuss the importance of thoroughly understanding the effects of restoration treatments on an entire ecosystem before large-scale implementation commences.

The majority of studies on this subject have focused predominantly on small mammals and birds (Dodd et al. 2006; Germaine and Germaine 2002; Converse et al. 2006). A number of studies have also focused on arthropod responses, for example butterfly abundance (Meyer and Sisk 2001; Waltz and Covington 2004). Common components of most studies are the indirect effect of thinning on wildlife through the response of understory vegetation. A common observation is that thinning treatments often favor a variety of wildlife, because of the increased availability of herbaceous understory plants immediately following treatment (Bock and Bock 1983; Harris and Covington 1983; Waltz and Covington 2004). Contrary to this is the effect that thinning may have on species that depend on dense security cover or are vulnerable to predation in open stands. For example two separate studies by Crocker-Bedford (1990 and 1995), showed declines in nesting success of Northern goshawk on thinned plots in Arizona ponderosa pine woodlands.

The monitoring of wildlife response to thinning has been promoted recently through the National Fire and Fire Surrogate Project funded by the USDI-USDA Joint Fire Science Program (http://www.fs.fed.us/ffs/). This program funds research into the effects of fire surrogates such as mechanical thinning on numerous ecosystem components. Research by Converse et al. (2006) on small mammal responses to thinning of ponderosa pine in the Jemez mountains of New Mexico is one example from the program. This study found that small mammal biomass generally responds positively to thinning, particularly where the pretreatment habitat was in poor condition, i.e., the greatest responses would occur in areas where stand structure is particularly dense prior to treatment (Converse et al 2006). As is noted in a number of studies, faunal responses to thinning are species specific (Bock and Bock 1983; Woolf 2003). For example, early seral species such as deer mice are consistently more abundant on disturbed forest sites (thinned versus not thinned) (Converse et al. 2006), whereas southern red backed voles often respond negatively to thinning as they require dense old growth forests (Woolf 2003).

Because some species undergo negative response to treatment, many scientists recommend incremental restoration to minimize impacts to sensitive fauna and flora (Allen et al. 2002). To retain varied habitat, restoration ecologists should strive to maintain a diverse forest structure of various aged trees and species composition (Dodd et al. 2006).

1.6.5 Fire Frequency and Intensity

Because of fire exclusion policies, wildfires in the Southwest are becoming more severe in stands that would have naturally undergone frequent but low intensity fire. A number of studies have been carried out throughout the Southwest that explore the impact of thinning on fire intensity and severity (Fulé et al. 2002, 2005; Pollet and Omi 2002) and all draw the common conclusion that untreated forests are at a higher risk of severe wildfire than treated areas. Pollet and Omi (2002) suggest that the removal of small diameter trees may be beneficial for reducing crown fire hazard in ponderosa pine sites. Prescribed fire may be effective at reducing these small diameter trees, but only after some form of mechanical thinning has occurred to prevent these mid-canopy trees transmitting fire to the overstory canopy. Agee and Skinner (2005) suggest that in order to make forests fire resilient, some form of forest thinning is required. They propose that forest treatments to reduce fire intensity in the case of a wildfire comprise four basic principles: (1) reduce surface fuels using mechanical methods or prescribed fire; (2) increase the height to the live crown; (3) decrease the crown density reducing the chance of crown fire spread; and (4) keep big trees of fire resistant species (e.g., ponderosa pine) that would reduce overall mortality in the case of fire and move the forest back to historical stand structures.

Ponderosa pine would have historically undergone frequent wildfire, with mean fire intervals of less than 10 years (Kaufmann et al. 1998; Allen et al. 2002). Full restoration of ponderosa pine woodlands therefore would require maintaining constant low intensity fires in pretreated stands, applying prescribed fire regularly enough to retard the growth of small diameter trees, while promoting a vigorous herbaceous community for greater forest biodiversity. Although there are concerns regarding the effects of fuel treatments on ecological and biological components of southwestern forests, these concerns need to be weighed against the impacts of a no-action response to forest management. High intensity stand replacing fires did not occur historically in southwestern ponderosa pine and mixed-conifer communities (Pyne 1982), so fauna and flora of these regions have evolved under very different stand conditions and fire regimes than are observed today. The ecological impacts of high intensity fires could therefore very well exceed impacts caused by thinning activities (Agee and Skinner 2005).

1.6.6 CONSIDERATIONS FOR CLIMATE CHANGE

Recent accelerated climate change is well documented as affecting both global and local environments and will likely have even more pronounced impacts in the foreseeable future. The following U .S. Government Accounting Office (GAO) report summary provides a clear statement of the problem relative to natural resource management in the U.S.: "Climate change has implications for the vast land and water resources managed by the Bureau of Land Management (BLM), Forest Service (FS), U.S. Fish and Wildlife Service (FWS), National Oceanic and Atmospheric Administration (NOAA), and National Park Service (NPS). These resources generally occur within four ecosystem types: coasts and oceans, forests, fresh waters, and grasslands and shrublands. GAO obtained experts' views on (1) the effects of climate change on federal resources and (2) the challenges managers face in addressing climate change effects on these resources. GAO held a workshop with the National Academies in which 54 scientists, economists, and federal resource managers participated, and conducted four case studies. GAO recommends that the Secretaries of Agriculture, Commerce, and the Interior develop guidance incorporating agencies' best practices, which advises managers on how to address climate change effects on the resources they manage and gather the information needed to do so. In commenting on a draft of this report, the three departments generally agreed with the recommendation and provided technical comments, which GAO has incorporated into the report as appropriate" (GAO 2007).

An important consideration relative to climate change is that our current knowledge about the status and function of Southwestern forest ecosystems is based on previous research and observations under different climates than those associated with recent global warming and associated changes in regional rainfall patterns. Harris et al. (2006) demonstrate the importance of considering climate change forecasts relative to ecological restoration activities in order to achieve desirable goals. Recent findings show that global warming is changing environments across western North America, including forest ecosystems. For example, snowmelt in the Rocky Mountains of Colorado and New Mexico is occurring earlier in the year (Hall et al. 2006; Gutzler 2007; Rahmstorf et al. 2007), and western North America forest wildfire frequency and intensity have recently increased (Veblen et al. 2000; Westerling et al. 2006), all of which have been linked to global warming. Since Southwestern forest environments and ecological processes are experiencing changes resulting from global warming, we need to be prepared to learn and understand changes in ecosystem processes and function, while employing adaptive management strategies to accommodate such changes over time.

Regional climate conditions affect wildfire timing, frequency, and intensity. Wildfires in cooler climates are more frequent and lower intensity than in warmer climates, and warmer climates promote large, stand replacing fires (Westerling et al. 2006). Pre-settlement climate data demonstrate that historical climates of the Rocky Mountains were generally cooler than today (Pierce et al. 2004), and current climate change is trending for a drier and warmer Southwest (Seager et al. 2007; Gutzler 2007), supporting low intensity fires during those times. Wildfires in the North American west were historically highly seasonal and occurred chiefly during the summer months when temperatures were highest and precipitation was relatively low. Westerling et al. (2006) found that wildfires dramatically increased in frequency and duration after 1980 in areas where spring and summer temperatures increased the most. Middle and high elevation forests (such as those in northern New Mexico) were found to have the greatest increase in large-scale, high intensity fires. They also concluded that increased fire risk due changes in climate may affect forest and fire management practices, rendering the standard methods of fuel reduction and restoration ineffective. New forest management strategies will need to be developed with climate change in mind. The effects of forest thinning activities today and in the future, will likely be different from those in the past.

1.7 THE ESTANCIA BASIN WATERSHED

1.7.1 GEOGRAPHY AND GEOMORPHOLOGY

The study area is located in Torrance County, New Mexico, on the east side of the Manzano Mountains (Figure 1.2). The Manzano/Sandia range traverses central New Mexico in a north-south direction on the east side of the Rio Grande. In Torrance County, the range becomes more elevated into high peaks of the Manzano Mountains and descends to a southern point near the town of Mountainair (Sivinski 2007). Elevation ranges from 6,000 to 11,000 feet (2,130 to 3,690 m). The numerous ridges and canyons in this region drain via intermittently flowing arroyos into the Rio Grande to the west and into the closed Estancia Basin to the east (Brockway et al. 2002; Allen and Anderson 2000). The surface geology is underlain by Abo Formation with thick bedrock of red-reddish brown cross-bedded siltstone and medium to fine grained sandstone overlying shale, siltstone, sandstone, and conglomerate (Brockway et al. 2002). The core of the Manzano Mountains dates to the Precambrian era (1.45 billion years old) and Precambrian granite is exposed along the western base (Sivinski 2007). More detailed descriptions of the geology of this region can be found in Bauer et al. (2003).

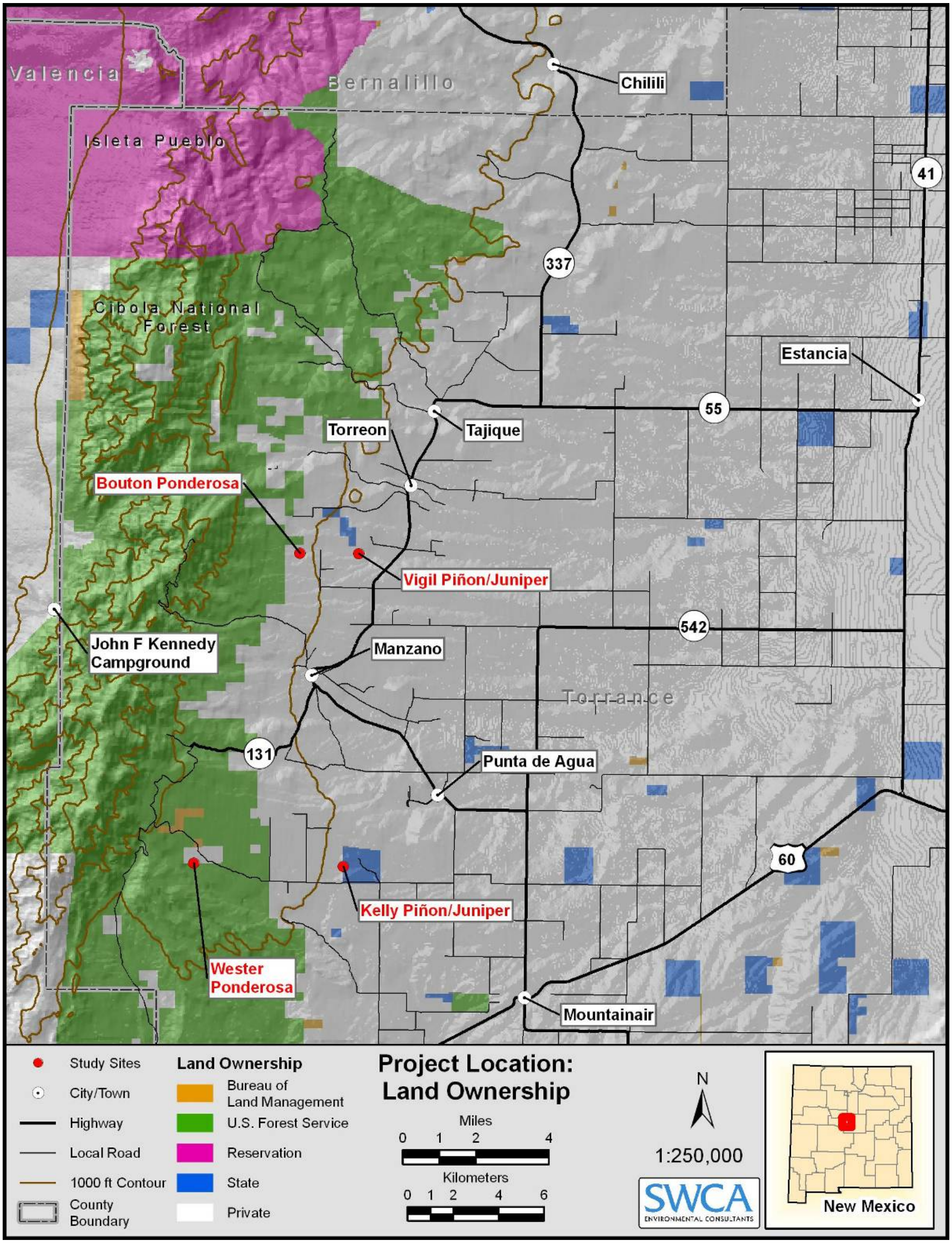

Figure 1.2. Land ownership in the project area.

1.7.2 CLIMATE

This area has a semi arid climate with highly variable annual precipitation with totals averaging approximately 360 mm (14.2 inches); average total snowfall is 617 mm (24.3 inches) (Figure 1.3). Sixty percent of the annual precipitation is received in the summer months during intense, short duration monsoonal thunderstorms (Bourlier et al. 1970). Average minimum and maximum monthly temperatures are 1.9°C (35.5 °F) and 19.5°C (67.5°F) respectively, with the highest temperatures generally occurring between May and September (Figure 1.4) (Western Regional Climate Center data (2007); period of record 1914-2006).

1.7.3 SOILS

A wide range of soil types exist in the study area due to the varied topography and elevation gradients. Sites located on shallower slopes are found on Washoe gravelly loams (Wb), and Wit loam (Wp) soils (1–9% slopes) (Bourlier et al. 1970; USDA-NRCS 2004). Wb soils occur on piedmont fan crests and side slopes of the southern and eastern foothills. The surface layer has a gravelly and granular structure that is highly permeable to water and plant roots. The subsoil has a more blocky structure that is less permeable and has a low to moderate water storage capacity. Surface runoff is medium to rapid and internal drainage is minimum, making the soil subject to severe water erosion when exposed. Wp soils occur on upland piedmont fans and are the major dry land soils in the area. They are quite heavily leached in some areas but have reasonable fertility and organic content. They are susceptible to wind and water erosion when exposed, but they have a greater water storage capacity and generate slightly less runoff than Wb soils (Bourlier et al. 1970).

Medium slope sites occur in the study area on Washoe cobbly loam (Wa) soils (9–25% slope) (Bourlier et al. 1970). These soils occur on mostly southern and eastern truncated slopes of piedmont fans. Wa soils are heavily cobbled and gravelly and as such are prone to drought and severe water erosion when not protected with adequate vegetation. Surface runoff is high to rapid and evapotranspiration losses are also high especially on south and west facing slopes. Sites on steeper slopes are found on Wilcoxson stone loam (Wf) and Fuera cobbly loam (Fu) soils (20–60% slopes). Wf soils occur on north and east facing slopes and ridges of the Manzano Mountains. This soil tends to have a thick layer of decomposing forest litter and the surface layer is a grayish-brown stony loam. The subsoil is limy and grades to limestone bedrock. This steep soil is prone to severe water erosion, particularly when exposed by fire or logging. However, runoff under normal conditions is usually slow because of a medium to rapid infiltration capacity and moderate water storage capacity; the stony surface generally controls erosion. Fu soils occur on side slopes of piedmont fans and are shallow and stony. Decomposing soil litter layers are generally thin but the surface layer is thicker than Wf soils and the subsoil has a heavy clay component. The soil is very droughty and supports limited plant growth. The soil readily absorbs water but has slow permeability; it generates medium runoff and is prone to moderate to severe erosion in the event of a fire or thinning (Bourlier et al. 1970). A soils map of the entire study area is too detailed to present here as a single map figure. Refer to USDA-NRCS (2004) for soils maps of the area.

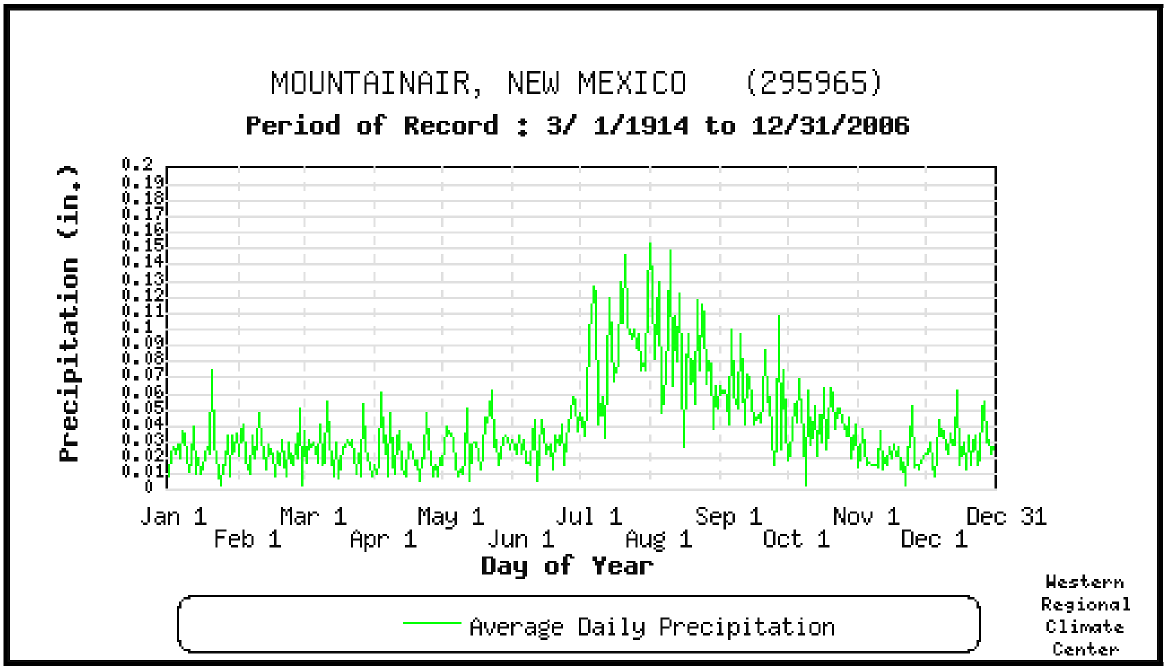

Figure 1.3. Average daily precipitation at Mountainair, NM. Source: Western Regional Climate Center (2007).

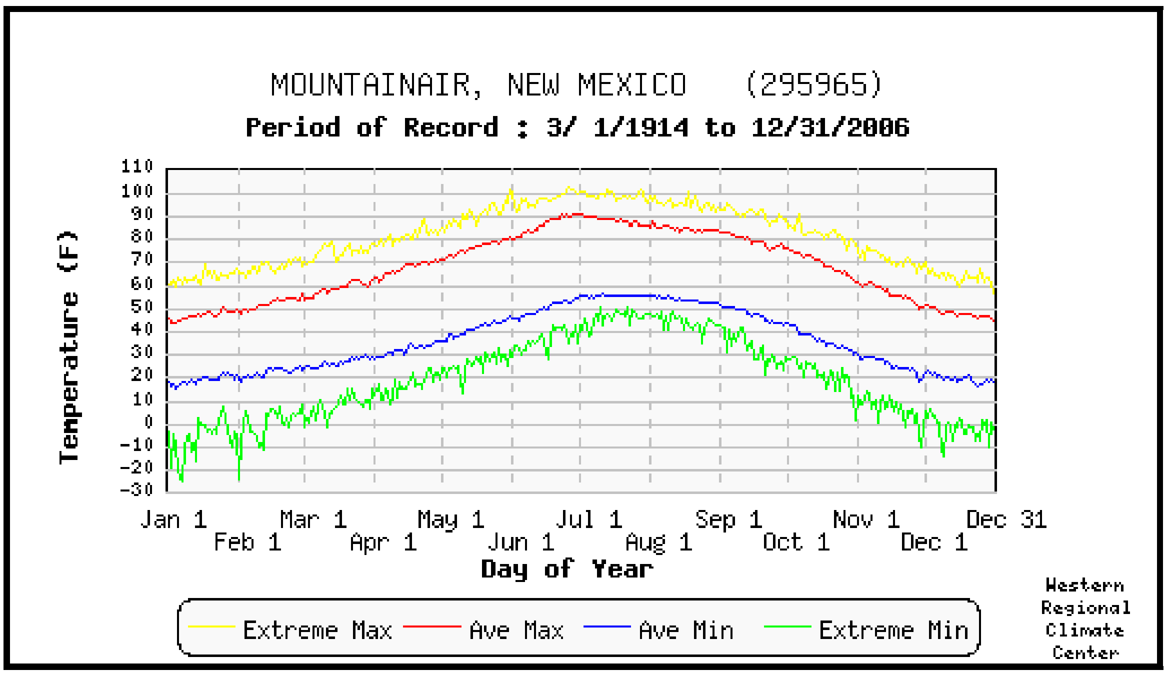

Figure 1.4. Average minimum and maximum temperatures at Mountainair, NM. Source: Western Regional Climate Center (2007).

1.7.4 VEGETATION

There is a relatively high plant species diversity in this area due to the positioning of the site near the confluence of four floristic regions: Southern Rocky Mountains to the north, Chihuahuan Desert to the south, Great Plains Short Grass Prairie near the east slope, and Colorado Plateau of the Great Basin to the west (Sivinski 2007; USFS 1994). Variations in the elevation, substrate, and exposure of the study plots also contribute to diverse habitat for a wide range of species. A USGS (2004) Gap Analysis Program (GAP) vegetation map of the Estancia Basin watershed, showing high-elevation mixed-conifer forests, mid-elevation ponderosa pine woodlands, piñon/juniper woodlands, and Great Plains grasslands is presented in Figure 1.5.

Sivinski (2007) produced a simple classification of plant community types of the Sandia and Manzano mountains. These were based on elevation and precipitation gradients as follows: foothill scrub at the arid base of the west face, PJ woodland, ponderosa pine woodland, mixed-conifer forest, and subalpine forest on the highest peaks. Lower elevation foothill scrub communities (on Wp soils) comprise blue grama (*Bouteloua gracilis),* galleta (*Pleuraphis jamesii*), sand dropseed (*Sporobolus cryptandrus),* ring muhly (*Muhlenbergia torreyi),* broom snakeweed (*Gutierrezia sarothrae),* and cane cholla cactus (*Cylindropuntia imbricate)* (Sivinski 2007, Bourlier et al. 1970). On Wb and Wa soils and at higher elevation these communities grade into PJ woodland of: piñon pine (*Pinus edulis),* alligator juniper (*Juniperus deppeana),* one-seed juniper (*Juniperus monosperma*), and at slightly higher elevation, Rocky Mountain juniper (*Juniperus scopulorum*) (Sivinski 2007). Understory associations of these PJ communities are similar to the desert scrub communities but also include western wheatgrass (*Elymus smithii),* mountain mahogany *(Cercocarpus montanus),* and Gambel oak *(Quercus gambelii).*

The steeper slopes (and Wf and Fu soils) are dominated by a ponderosa pine community with lesser amounts of PJ and Gambel oak. Creeping Oregon grape *(Berberis repens),* side oats grama *(Bouteloua curtipendula),* and other cool season grasses dominate the understory. Higher elevation mixed-conifer sites are dominated by Douglas fir *(Pseudotsuga menziesii),* white fir *(Abies concolor*), Rocky Mountain maple *(Acer glabrum),* and quaking aspen *(Populus tremuloides).* Mixed-conifer understory species include pine dropseed (*Blepharoneuron tricholepis),* Canada wildrye (*Elymus Canadensis),* western yarrow *(Achillea millefolium),* showy paintbrush *(Castilleja miniata),* and mountain parsley *(Pseudocymopterus montanus).* Finally, the sub-alpine high elevation forests are made up predominantly of Englemann's spruce (*Picea engelmannii),* cork bark fir *(Abies bifolia),* and limber pine (*Pinus flexilis*); fringed brome (*Bromus ciliatus),* western yarrow *(Achillea millefolium),* and Rocky Mountain Penstemon *(Penstemon strictus)* are common understory species.

Of the 937 plant taxa known to occur in the Sandia/Manzano range (Sivinski 2007), 11.5 % are non-native. The most threatening of these non-native species in the study area is Siberian elm (*Ulmus pumila*), which occurs along most canyons from the arid foothills up to mixed-conifer (Sivinski 2007). In terms of rare native plants, Sandia alumroot *(Heuchera pulchella)* is the only plant species known to be strictly endemic to the Sandia/Manzano range of mountains (Sivinski 2007), and tall bitter weed (*Hymenoxys brachyactis)* is an endemic nearly confined to the Manzano range. These and three other species, Santa Fe milk vetch (*Astragalus feensis*), Flint Mountains milk vetch (*Astragalus siliceous*), and Plank's campion (*Silene plankii*), are listed as New Mexico species of concern for Torrance County (New Mexico Rare Plants Technical Council, 2008).

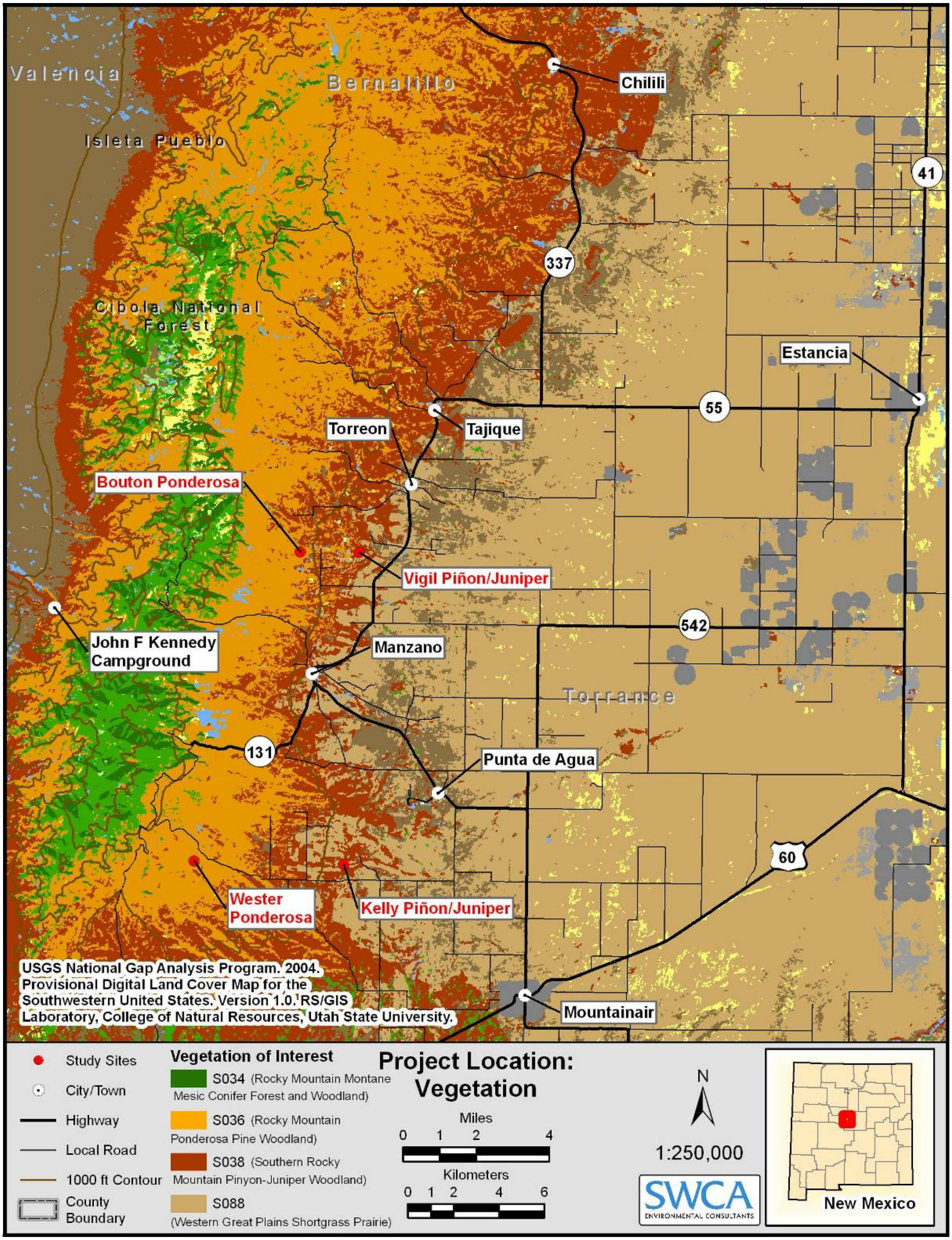

Figure 1.5. Vegetation in the project area (only dominant vegetation types are provided in legend).

1.7.5 FAUNA

Animals of the east slope of the Manzano Mountains are typical for the Southern Rocky Mountain forests and woodlands, as well as the southern Great Plains short-grass prairie (Brown 1982). The most comprehensive listing for animals of the Estancia Basin watershed area is reported in the BISON (2007) database. BISON (2007) lists six species of amphibians, 33 reptiles, 77 birds, and 64 mammals as occurring in Torrance County, many of which likely occur in our study area. BISON (2007) also lists over 200 species of invertebrates as documented from Torrance County; however, that number is far below the likely several thousand species of invertebrates that occur in the county, no invertebrate surveys have been conducted in the Estancia Basin watershed. No federal or state listed threatened or endangered animal species are known to occur in the watershed.

2.0 METHODS

2.1 STUDY AREA

Forest thinning practices in the Estancia Basin watershed follow strict protocols for different types of forests and woodlands based on project funding. This monitoring project addresses the effects of forest thinning in ponderosa pine woodland and in piñon/juniper woodlands. The thinning prescriptions for these particular types of forests are as follows verbatim from the original thinning guideline documents obtained from Dierdre Tarr, pers. comm., Claunch-Pinto Soil and Water Conservation District (CPSWCD) 2007):

2.1.1 PIÑON-JUNIPER:

1. Thin to 60 square feet of basal area.
2. Thin leaving a variety of size classes by each species.
3. Favor to leave alligator juniper. alligator juniper sprouts too much when cut. Cut it only when necessary to meet Firewise standards around homes.
4. Create random openings, do not space trees evenly; we are looking for an average of 60 over the whole stand.
5. Remove insect and diseased trees first then thin to 1, 2, 3, and 4 above.
6. No wood or chips to be stacked under the drip line of any trees.
7. Piñon firewood left on site will be solarized with at least a 6 mm clear plastic.
8. Chipping or mastication is preferred; chip depths are not to exceed an average of 2 inches deep and no greater then 6 inches in any single spot. Chips should not be accumulated under the drip lines of leave trees or within 10 feet of structures or woodpiles.
9. Where mulching operations are used 60% of the mulched material will be less than 3 inches in diameter and no longer than 3 ft in length and mulch depth will be less than 3 inches on average. No mulch depth will be allowed over 10 inches in any circumstance.
10. If any material (chips, slash, or firewood) is removed off site, above-mentioned measures should be identified to prevent insect, disease and fire problems.

2.1.2 PONDEROSA PINE:

1. Thin to 60 square feet of basal area.
2. Thin leaving a variety of size classes by each species.
3. Favor to leave ponderosa pine in groups of 2 to 7 trees, thinning out all trees under the drip line of the larger ponderosa pines.

4. Favor to leave Douglas fir. Cut it only when necessary to meet Firewise standards around homes or to meet 3 above.

5. Create random openings, do not space trees evenly; we are looking for an average of 60 over the whole stand.

6. Remove insect and diseased trees first then thin to 1,2,3,4 and 5 above.

7. No wood or chips to be stacked under the drip line of any trees.

8. Pine firewood left on site will not be solarized with at least a 6 mm clear plastic. Lop and scatter will be allowed but chipping or masticating.

9. Pile and burn can be allowed if projects are completed between November and March.

10. Chipping or mastication is preferred; chip depths are not to exceed an average of 2 inches deep and no greater then 6 inches in any singe spot. Chips should not be accumulated under the drip lines of leave trees or within 10 feet of structures or woodpiles.

11. Where mulching operations are used 60 percent of the mulched material will be less than 3 inches in diameter and no longer than 3 ft in length and mulch depth will be less than 3 inches on average. No mulch depth will be allowed over 10 inches in any circumstance.

2.1.3 General to Do and Do Not:

1. We are thinning to an <u>average</u> of 60 square feet of Basal area per acre, not 60 at every plot.

2. When there are forked trees remove both forks or leave both forks. Pruning or removing of multiple branches in juniper does not meet this requirement.

3. Porcupine forks are not considered diseased trees and can be left for character.

4. Multiple sharp forks, forking three or four times, represent genetic forks in ponderosa pine. These trees need to be removed.

5. In ponderosa pine, where there is heavy mistletoe throughout the tree, small clear cuts up to 3 acre in size are allowed to control the mistletoe.

6. It is best to leave juniper logs and slash on the ground for erosion control purposes. Try to avoid using piñon and or ponderosa pine. Piñon and ponderosa pine can attract bugs and rot to quick.

7. Material removed off site needs to address where, in what form (i.e., chips, slash or firewood) and why.

8. All thinning along roads should be done to create a good fire line. Thinning to 40 square feet of basal area for the first 50 feet then thin to 50 square feet of basal area for the

second 50 feet then thin to the 60 square feet of basal area. If room is an issue, try to thin to a lower Basal next to roads or meadows.

9. Send cost share calculations with prescriptions for approval (pers. comm. Dierdre Tarr, CPSWCD, 2007).

2.2 EXPERIMENTAL MONITORING DESIGN

The goal of this experimental monitoring design is to provide unbiased data representing information on the effects of the above forest thinning treatments on watershed and forest health in the Estancia Basin watershed over time. Due to costs and logistical constraints, such information must be derived from sampling the environmental effects of forest thinning from limited study sites, and then generalizing those results to the landscape level. Such generalizations assume that 1) the chosen study site environments are representative of the larger landscape of interest, 2) the imposed forest thinning treatments are typical for those of the region, 3) differences between treated and non-treated study plots are due to treatment effects alone, not to other unaccounted for background environmental factors, and 4) all treatments are the same across all treatment plots (for those that are treated) and that all measurements of response variables are the same across all study plots. Although the validity of these assumptions cannot be guaranteed, we have done everything possible to achieve these assumptions when designing the study.

The Estancia Basin watershed includes mixed-conifer forests from the highest elevation portions of the Manzano Mountains, extending east and downslope across ponderosa pine woodlands, PJ woodlands, juniper savanna, and Great Plains shortgrass prairie. However, the majority of Estancia Basin Watershed Health, Restoration and Monitoring Project forest thinning activities are in ponderosa pine woodland and PJ woodlands at upper and middleslope elevations. Therefore, we are focusing our monitoring efforts in those two forest type/elevation zones. Ponderosa pine woodlands and piñon/juniper woodlands occur at different elevations and in different climatic environments, so we have established monitoring sites in each zone within the same watersheds.

Any imposed treatment effects must be assessed by comparing the affected system or environment to the same system of environment that is not treated (Green 1979; Krebs 1998; Morrison et al. 2001; Gotelli and Ellison 2004). The goal of this monitoring study is to assess the impacts of forest thinning on response variables including soils, hydrology, plants, and animals. Therefore, we must pair treated forest locations with identical non-treated locations as close together on the same landscape as possible, assuming all background environmental features are the same across that landscape. Although the environments of adjacent locations are probably not identical, we have attempted to match them as closely as possible.

Since natural landscape environments do vary over space, one cannot be certain that even paired study plots located close to each other on the same landscape have the same background environmental conditions. The best way to ensure that monitoring data for a given treatment effect does in fact represent that treatment effect rather than pre-existing background environmental differences is to first measure all response variables to be monitored on both the treatment and control study areas prior to implementing the treatment (Green 1979; Gotelli and

Ellison 2004). Such an approach allows one to account for known pre-existing differences in response variables when assessing the effects of the treatment once monitoring of those response variables begins after the treatment has been imposed. We have installed study plots in both treatment and control areas that have not yet been treated, and we will measure response variables for at least one year prior to forest thinning treatments.

We have made efforts to design this monitoring study to address the above assumptions and problems associated with environmental monitoring of response variables on inherently variable natural landscapes.

2.2.1 Research Questions and Statistical Hypothesis Testing

As stated above, testing hypotheses for effects of forest thinning treatments on the various parameters measured will be based on comparisons of paired treatment (thinned) and control (not thinned) study plots. Research questions are the specific objective statements about the effects of forest thinning that we are addressing. Statistical hypotheses are based upon those research questions but take the form of a "null hypothesis" when applied to statistical testing (i.e., the statistical tests are testing for no difference between treatments and controls).

All research questions below are addressing one overall question: Does forest thinning affect (?):

Soil: 1) chemistry; 2) surface stability; 3) surface erosion; 4) surface water infiltration; 5) subsurface (0–15 cm) moisture; 6) subsurface (12 cm) temperature.

Hydrology: 1) sub-watershed (study site) surface runoff/infiltration; 2) watershed groundwater recharge.

Vegetation: 1) ground layer (herbaceous and shrubs) plant species, growth form; and life-history composition; 2) ground layer plant canopy cover; 3) ground layer plant foliage heights; 4) tree density; 5) tree diameter; 6) tree crown structure and heights; 7) tree trunk diameter; 8) tree insect damage; 9) down woody materials and leaf litter.

Animals: 1) bird species composition and relative abundance; 2) small mammal species composition and relative abundance.

Statistical null hypotheses for each of the above are that there is no difference in the above listed parameters between treatments and controls following thinning. Various statistical analyses of the data (e.g., the best analysis for each parameter and data type, parametric t-tests, analysis of variance, or non-parametric equivalents, etc.) will test those hypotheses. The results of statistical tests will provide definitive and objective "yes" or "no" answers to each of the above questions. In cases where the null hypothesis of no change is rejected, the answer to the question would be "yes." We will then need to provide expert interpretation to determine the environmental meaning of the confirmed change. Yes answers to change will include quantitative (measured) changes in both positive or negative directions, depending upon the parameter (e.g., soil erosion may decrease or increase, etc.). A positive or negative quantitative or measured change may have the opposite desired environmental or resource management outcome (e.g., a positive measured or quantitative change in soil erosion would mean more soil erosion, which is a negative environmental effect based on natural resource management objectives).

All of the above questions and hypotheses will also be relative to a time component over the years following the thinning treatments. Analyses will need to be performed for each year, and time-related analysis such as repeated measures tests for time effects and time-treatment interaction effects over time. Different parameters likely will show responses to thinning over different lengths of post-treatment time (lag-time responses). For example, soil surface erosion rates may change within one year of thinning, but forest tree density may not change for five to ten years following thinning treatments.

2.2.2 STUDY SITE SELECTION

Four study sites were selected among approximately 30 possible locations where CPSWCD forest thinning activities are planned. Study sites were selected on the basis of: 1) representing both upper watershed ponderosa pine woodland and lower watershed PJ woodland; 2) in locations adjacent to existing forest thinning projects where additional forest thinning was planned and where road access was reasonable; 3) on private land where land owners were in agreement to support the monitoring project; 4) on land with 5–20% slope in order to assess ground surface precipitation runoff; 6) on land that is not currently grazed by domestic livestock or has very light and occasional grazing; and 7) on relatively homogeneous landscapes large enough to contain both of the paired treatment and control sets of study plots 100 m apart, and each on approximately 10 acres of adjacent homogenous land. Homogeneity of study site landscapes included environmental features of elevation, geology, slope, aspect, surface hydrological drainages, soil type, vegetation type, history of human land use, and current and planned intensity of domestic livestock grazing. Study sites were ultimately located in two different watersheds, representing both upper watershed and lower watershed forest and woodlands.

2.2.3 STUDY SITES

The four study sites are located in two Estancia Basin sub-watersheds, the Mesteno Draw (Wester and Kelly sites), and the Arroyo del Cuervo (Bouton and Vigil sites) drainages (Figure 1.2 and Figure 1.5) One study site is located in the upper-elevation portion of each sub-watershed in ponderosa pine woodland, and one study site is located in the mid-elevation portion of each sub-watershed in piñon/juniper woodland. Note that both sub-watersheds are geographically nearby (10 km), separated by one sub-watershed, the Arroyo de Manzano, in which the community of Manzano is located. Figure 1.5 provides a vegetation map (USGS 2004) of the study region along with the study site locations. Note the locations of the study sites relative to the three major forest zones: Rocky Mountain montane mesic conifer forest, Rocky Mountain ponderosa pine woodland, and Southern Rocky Mountain piñon/juniper woodland. Also, note the location of sites relative to open grasslands. Figure 2.1 provides a schematic diagram of the study site and study plot design, including treatment and control study plots, discussed below.

Figure 2.1 and Figure 2.2 provide aerial images of each study site, showing the locations for the two sets of study plots, treatments and controls, and soils map overlays (USDA-NRCS 2004). Detailed soil descriptions corresponding to those soils codes may be found in Bourlier et al. (1970). Forest thinning treatments will be conducted on the treatment plots in 2008 or 2009 (at the earliest), and treatments will be randomly assigned to one of each pair of study plots. Note that the study plot images on the aerial images are 100 m in area, but actual study subplots (1. soils/vegetation subplot, 2. small mammal subplot) are smaller, about 50 m on each side. The actual subplots are located within each of the 100 m plot areas at the time that plots were installed. Figures 2.9–2.12 provide on-site ground level photographs of the center area of each of the four study sites to show vegetation structure.

The Bouton site (Figure 2.1 and Figure 2.2) is in a ponderosa pine woodland represented by relatively young trees typical for the region. Site elevations range from 7,080 to 7,160 feet (2,158-2,182 m) above sea level. A drainage transverses the site along the north side, and the site has a gentle slope into that drainage. Soils of the site are Wilcoxson stony loam (Wf), which occur on north-facing slopes of 20–50%, and contain stones. Pino loams (Pv) are the other associated soils of the site. The two sets of study plots are located 100 m apart on a W-E orientation to occupy areas with similar canopy cover and slope. There is no domestic livestock grazing on the site. Jim Bouton is the land owner, and he has agreed to support the monitoring study.

The Vigil site (Figure 2.3 and Figure 2.4) is in PJ woodland that is typical for the region. Site elevations range from 6,800 to 6,840 feet (2,072-2,085 m) above sea level.

Drainages transverse the site along the east and south sides, and the site has a gentle slope into the south drainage. Soils of the site are Witt loam (Wb), which are characteristic of 1–6% slopes and tend to have low gravel content. Washoe gravelly loam (Wp) is the other associated soil at the site. The two sets of study plots are located 100 m apart on a NW-SE orientation to occupy areas with similar canopy cover and slope. There is light domestic livestock grazing on the site. Ernie Vigil is the landowner, and he has agreed to support the monitoring study.

The Wester site (Figure 2.5 and Figure 2.6) is in a ponderosa pine woodland represented by relatively young trees typical for the region; this property is also known as the Thunderbird Ranch. Site elevations range from 7,480 to 7,520 feet (2,280-2,292 m) above sea level. A drainage transverses the site along the south side, and the site has a gentle slope into that drainage. Soils are Pino loam (Pv), typical of 1–12% slopes, and with few stones. Fuera cobbly loams (Fr and Fu) are the other associated soils of the site, both occurring on steeper slopes. The two sets of study plots are located 100 m apart on a SW-NE orientation to occupy areas with similar canopy cover and slope. There is no domestic livestock grazing on the site. Michael and Wayne Wester are the landowners and have agreed to support the monitoring study.

The Kelly site (Figure 2.7 and Figure 2.8) is in PJ woodland that is typical for the region. Site elevations range from 6,920 to 6,940 feet (2,109-2,115 m) above sea level. A drainage transverses the site along the north side, and the site has a gentle slope into that drainage. Soils of the site are Washoe gravelly loam (Wp), which is characteristic of 1–9% slopes and has a gravel content of 25–75%. Witt loam (Wb) is the other associated soil at the site. The two sets of study plots are located 100 m apart on a W-E orientation to occupy areas with similar canopy cover and slope. This is New Mexico State land, and there is light domestic livestock grazing on the site. Tim Kelly has a grazing lease on the land and has agreed to support the monitoring study.

Note that the Bouton ponderosa site is located at elevations around 7,100 feet (2,164 m) above sea level, while the Wester ponderosa site is located at elevations around 7,500 feet (2,286 m) above sea level. We were not able to locate potential ponderosa study sites at more similar elevations, and we feel that a 400-foot elevation difference is not as important as other matching environmental characteristics that we discovered, such as tree species and density. Also, note the soils of the Bouton site are Wf, while those of the Wester site are Pv. Both soils are similar physically and chemically, but typically found on slopes of different steepness. Pv soils occur on the ridge top above the Bouton site, but the vegetation of that ridge top is composed largely of PJ woodland, not ponderosa pine. Note also that the two subplot locations at each study site do vary within 100 feet of elevation. We do not believe that this small range of elevation difference will affect climates on the adjacent sets of plots, compared to differences of hundreds or thousands of feet in elevation. Again, we attempted to match as many important environmental characteristics as possible, given the limited number of available sites across the entire watershed. We believe that the subtle differences among pairs of sites and study plots within sites are well within the reasonable environmental variation that may be expected for paired sites and plots given the scope of this monitoring project. Representative ground level photographs of each of the four study sites are presented in Figure 2.9, Figure 2.10, Figure 2.11, and Figure 2.12.

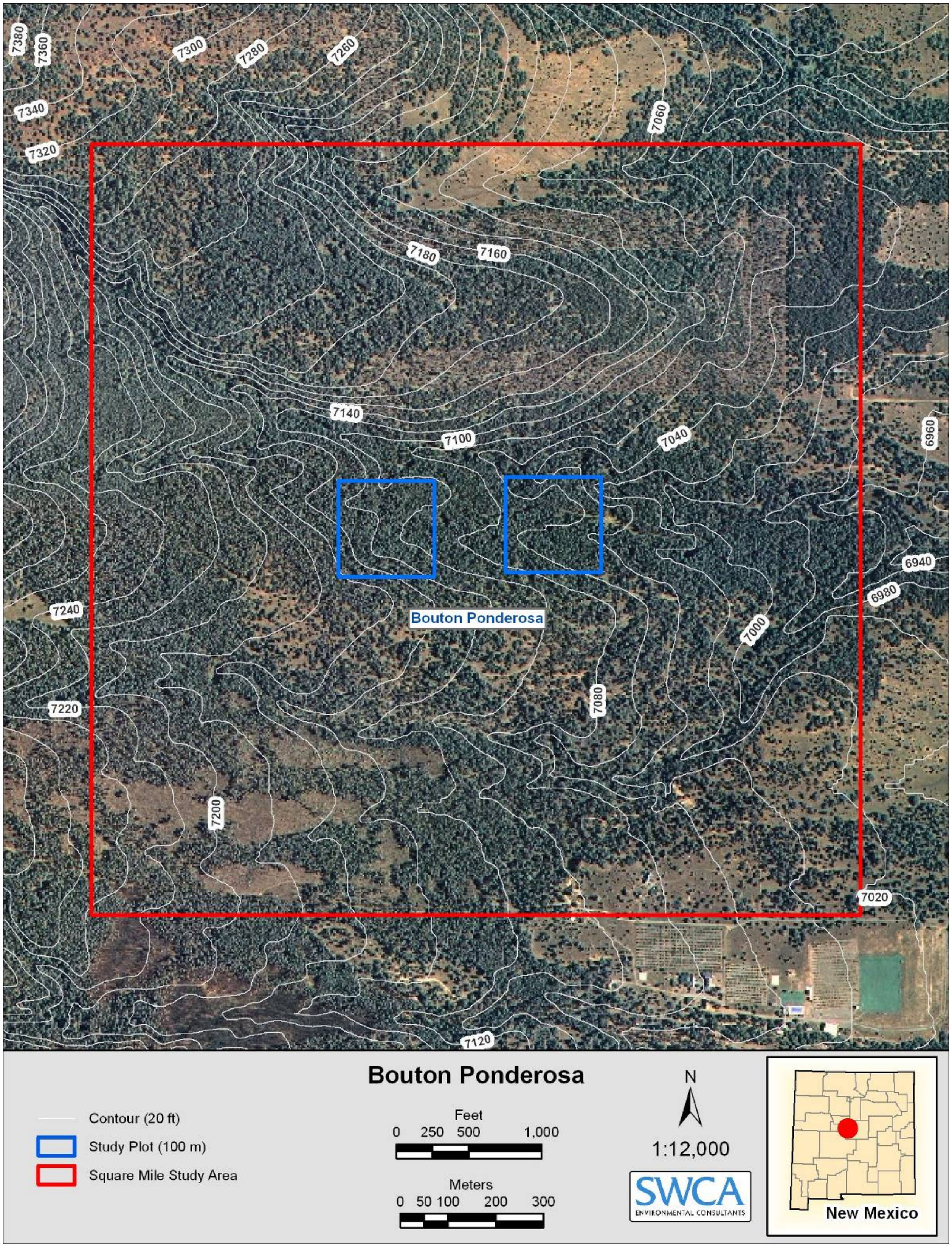

Figure 2.1. Bouton Ponderosa site map.

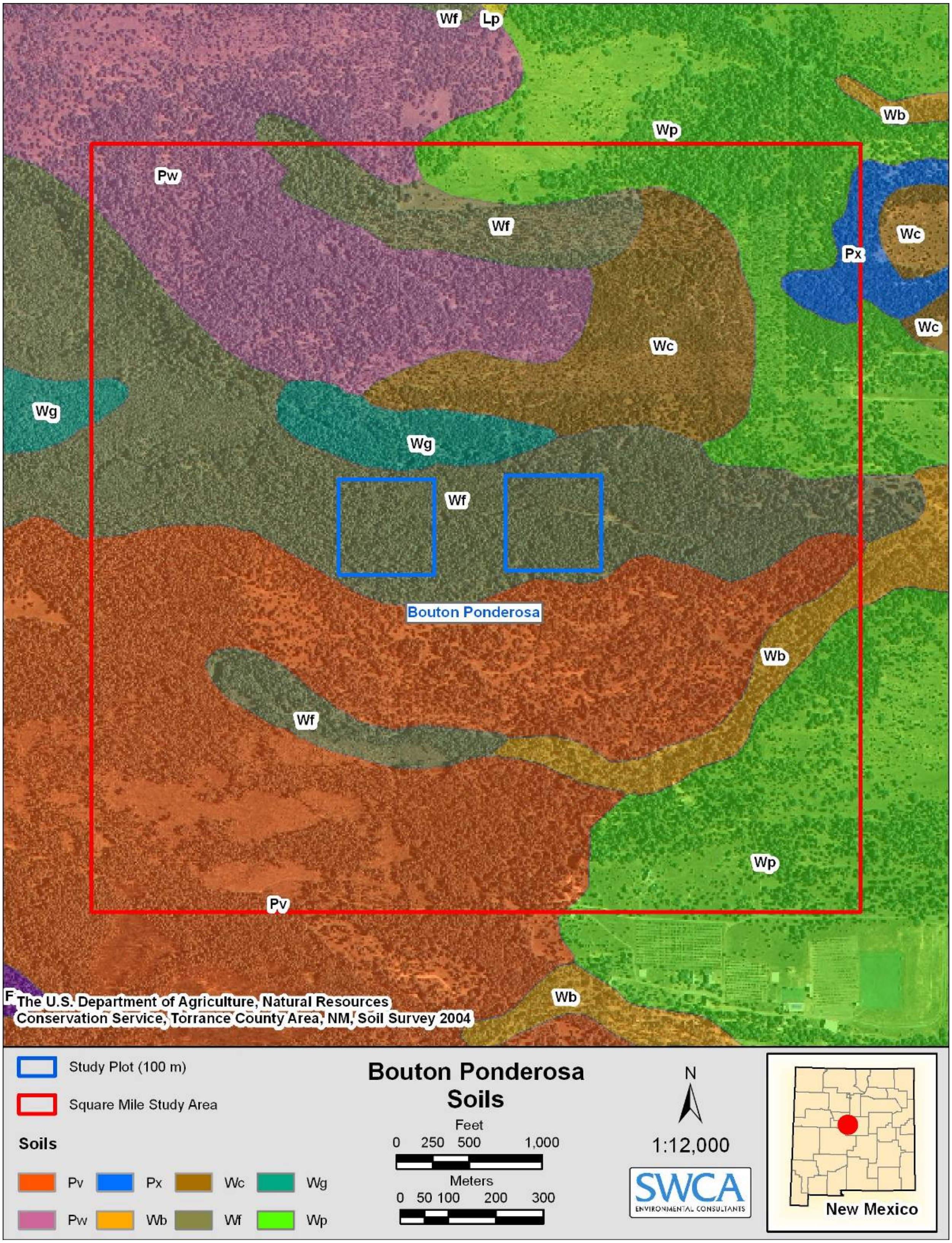

Figure 2.2. Bouton Ponderosa site soils map.

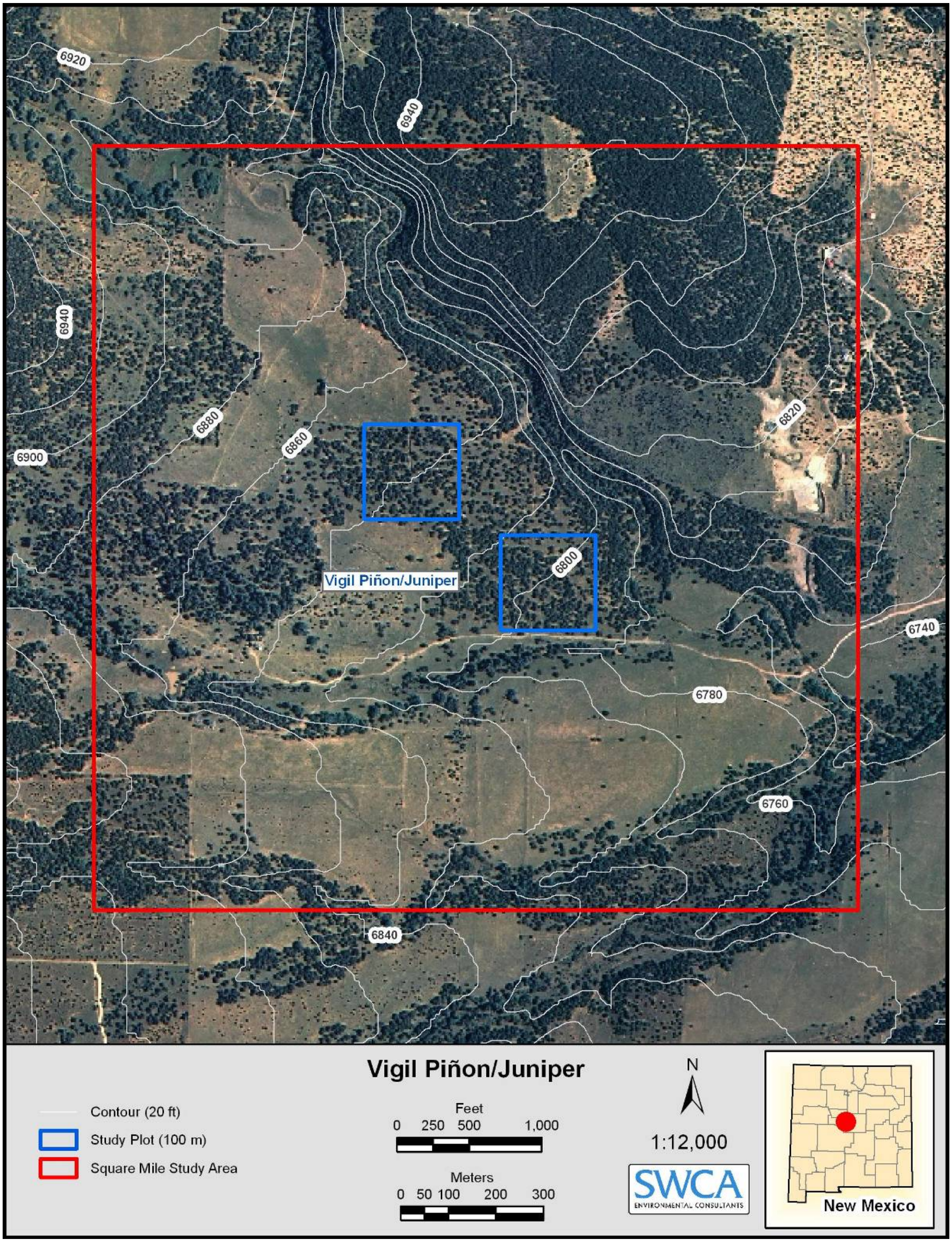

Figure 2.3. Vigil Piñon/Juniper site map.

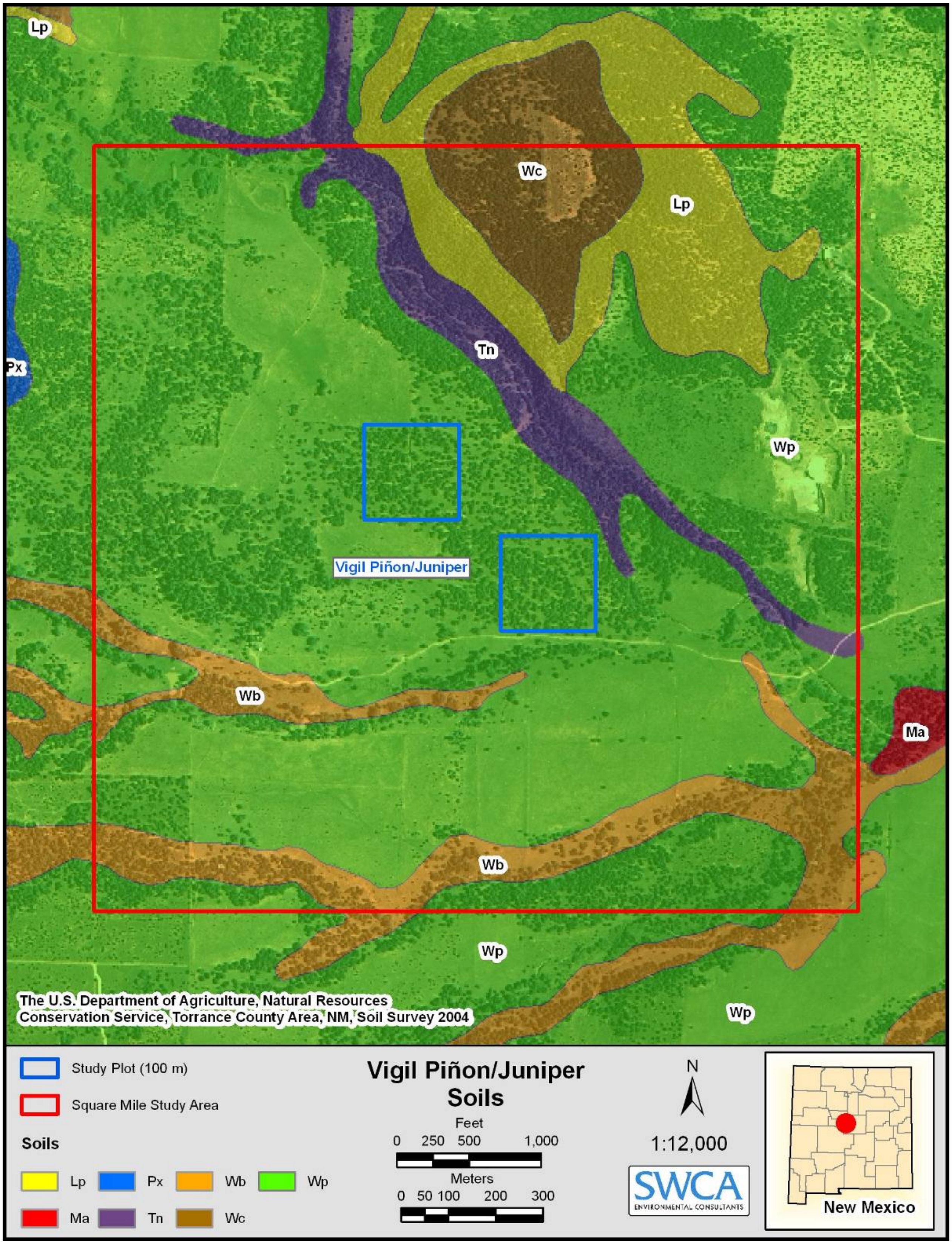

Figure 2.4. Vigil Piñon /Juniper soils map.

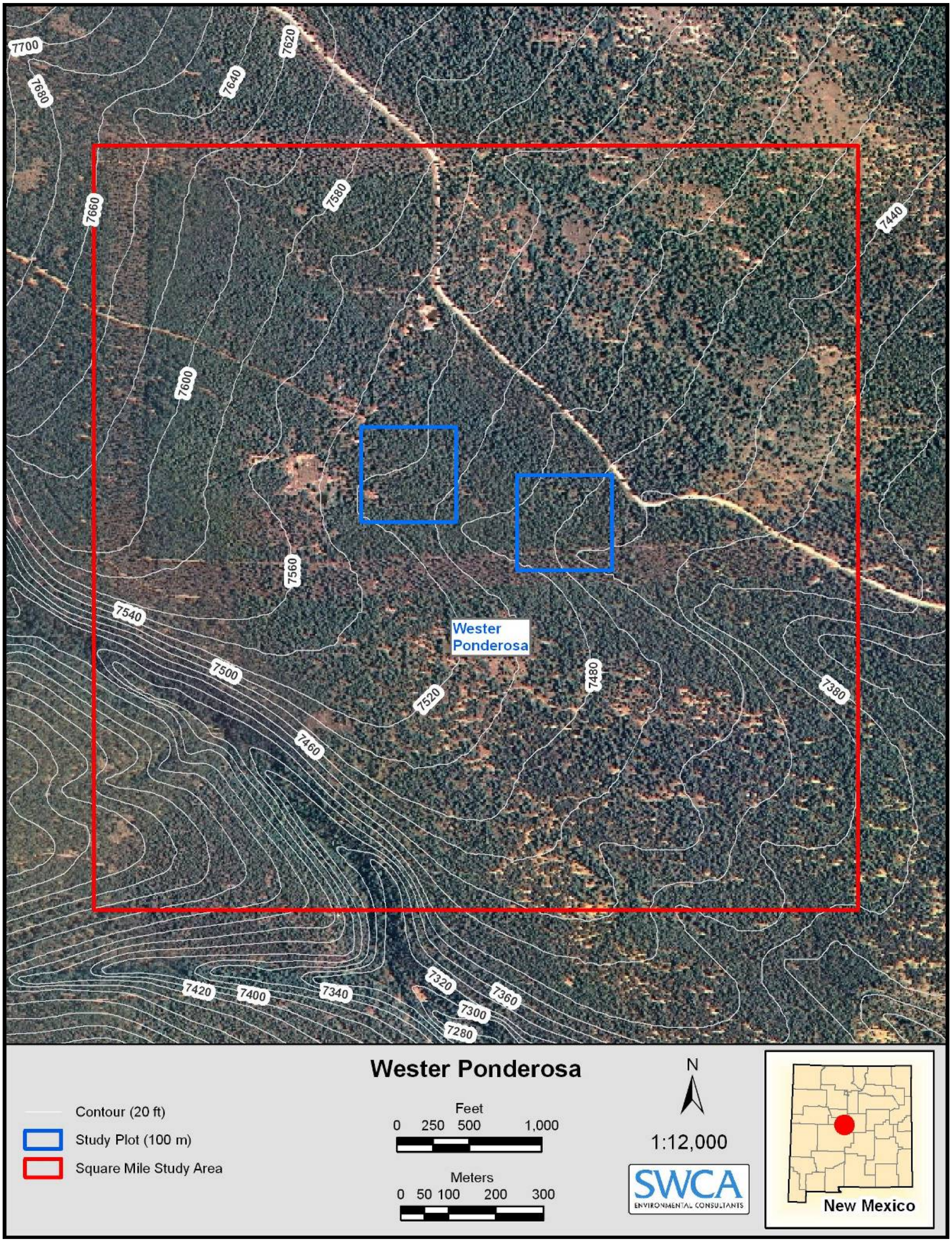

Figure 2.5. Wester Ponderosa site map.

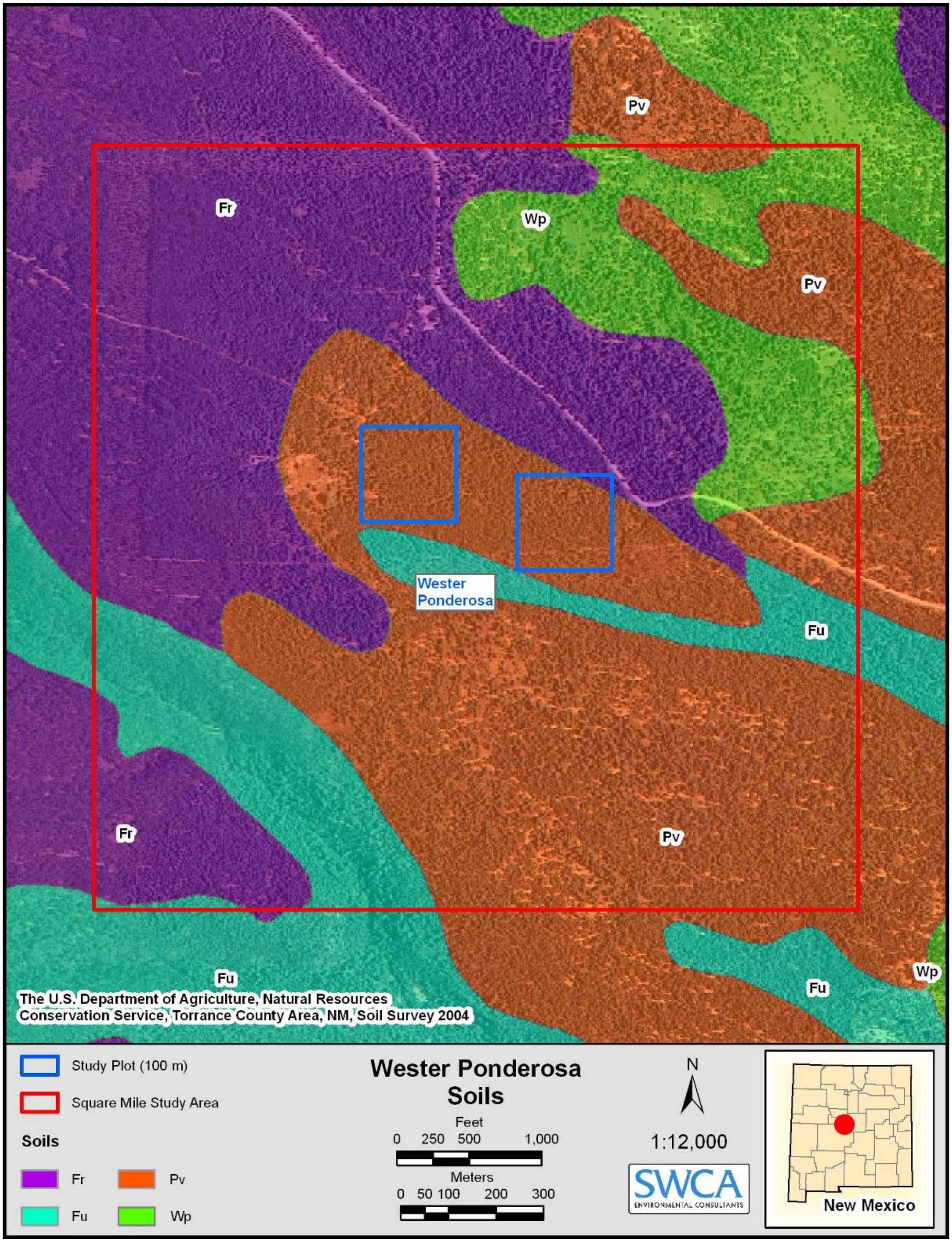

Figure 2.6. Wester Ponderosa site soils map.

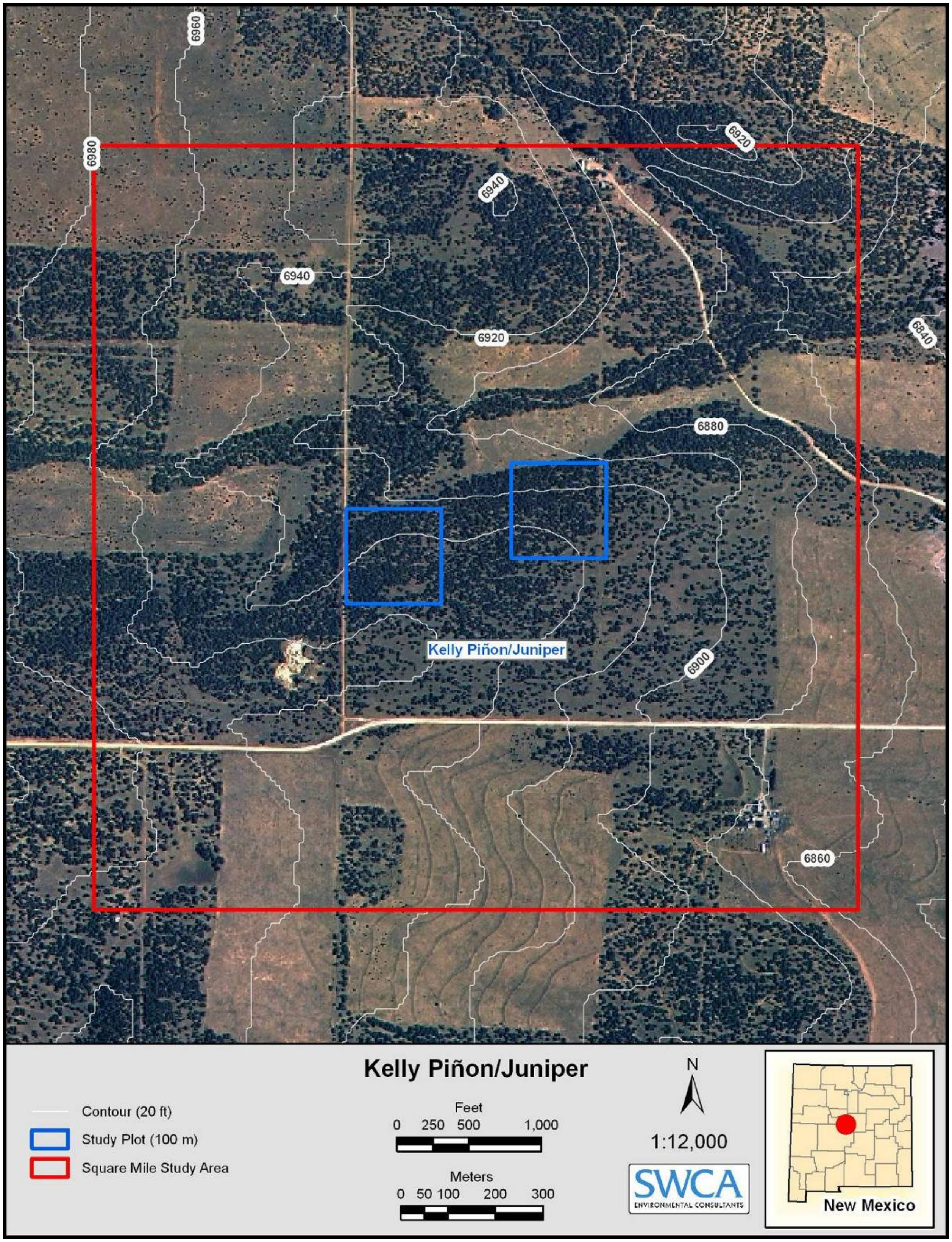

Figure 2.7. Kelly Piñon/Juniper site map.

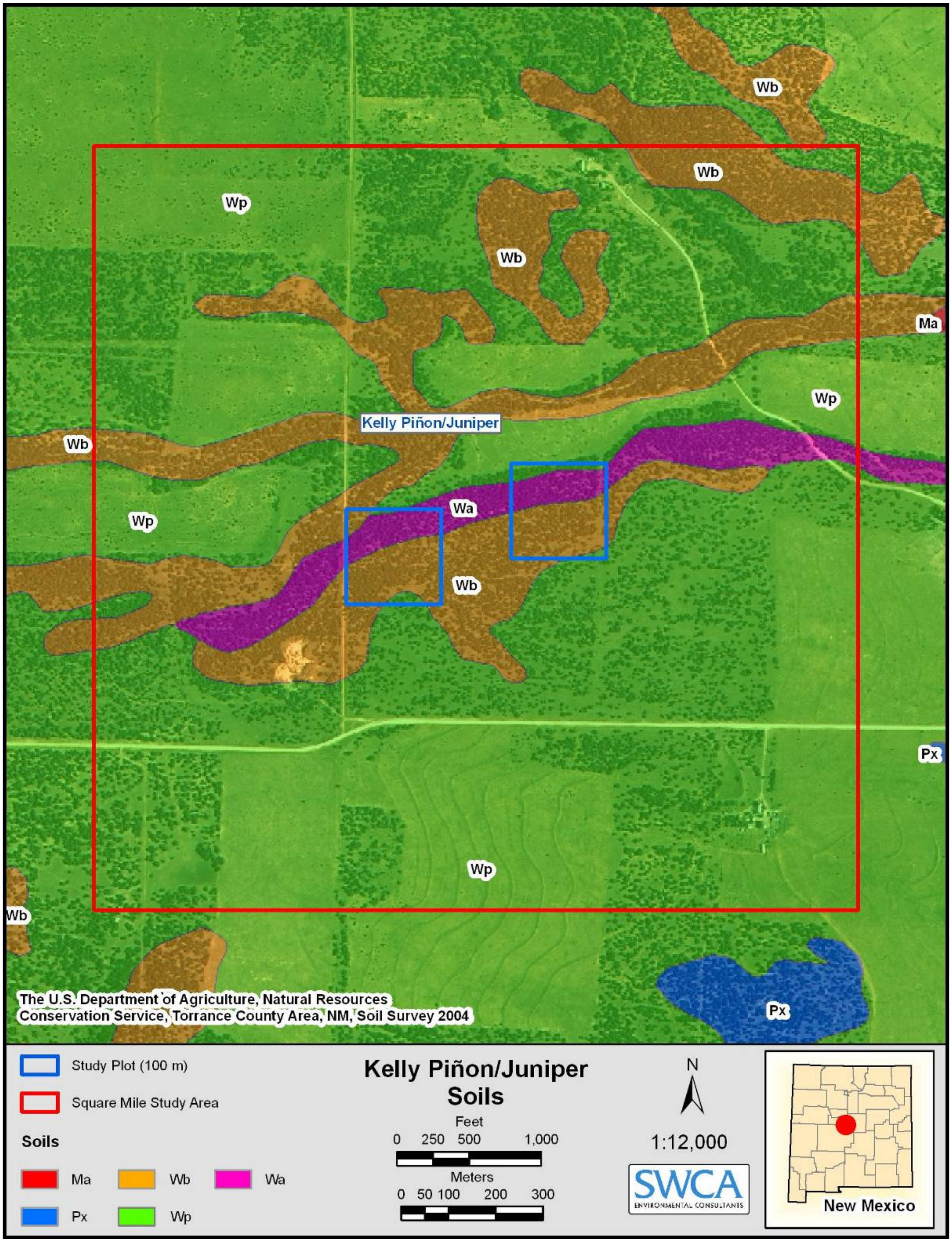

Figure 2.8. Kelly Piñon/Juniper soils map.

Figure 2.9. Bouton Ponderosa site.

Figure 2.10. Vigil Piñon /Juniper site.

Figure 2.11. Wester Ponderosa site.

Figure 2.12. Kelly Piñon/Juniper site.

2.2.4 Study Site Design

Forest thinning treatment effects on soils, hydrology, vegetation, and animals will be assessed and monitored over time by comparing treated locations to adjacent non-treated locations. We will collect one to two years of data from all study plots before thinning is conducted to assess any pre-treatment differences between treatment and control plots. Tree thinning will be done largely by hand use of chainsaws and minimal use of heavy equipment that would affect soils. Target trees for thinning are primarily one-seed juniper and piñon, and ponderosa pine trees as needed. Such treatments follow standard protocols for the Estancia Basin watershed as given above. Since the number of study sites for this monitoring project is small (4 total, 2 ponderosa, 2 piñon/juniper), knowing pretreatment environmental conditions on the plots to be treated is essential in accurately assessing treatment impacts on study plot environments. This study site/plot design is illustrated in Figure 2.13 (also refer to Figure 1.2 and Figure 1.5, and Figure 2.1-Figure 2.8 for the actual map locations of each study site). The pretreatment data will allow us to comparatively assess environmental differences or similarities between all pairs of treatment and control study plots. Knowledge of pre-treatment conditions will allow us to accurately determine the potential treatment impacts on study plot environments.

All study plots have been installed and permanently marked with steel rebar corner posts and permanent aluminum tags. Two study plots have been installed for each of the paired control and treatment locations. Geographic Positioning System (GPS) coordinates have been recorded for the corners of each study plot and subplot at sub-meter resolution. One plot will be used for measurements of soils, vegetation, and birds. The other plot will be used for sampling small nocturnal mammals. The repeated placement and monitoring of rodent traps by personnel would cause potential negative impacts to soil and vegetation monitoring locations. By locating the rodent and bird monitoring plots adjacent to the soil and vegetation monitoring plots on the same landscape, we assume that the soils and vegetation are similar on both plots at all study sites.

Each pair of treatment (“T”) plots (1. soils/vegetation/birds, and 2. mammals) are paired with non-treated control (“C”) plots. Each T and C pair is situated in the same general area, within 100 m of each other. T and C pairs are on the same landscape units (equivalent to U.S. Forest Service [USFS] Terrestrial Ecological Units) to control for as many environmental factors as possible, other than the forest thinning treatments. Elevation, geomorphology, soils, slope and aspect, and characteristic vegetation types have been standardized for each set of T and C pairs. Each T and C pair is situated on a north to south orientation at the same elevation position on the mountain slope, perpendicular to the east to west orientation of the general elevation gradient. All study plot T and C pairs represent the same type of forest-thinning treatment for either piñon/juniper or ponderosa pine (see thinning protocols above) conducted at approximately the same time (month and year). The T plot for each pair is situated approximately 50 m from the boundary of the treatment area, and the C plot is situated approximately 50 m in the opposite direction across the boundary of the treatment area. This arrangement should allow the paired study plots to be close enough (100 m) to be on similar landscapes, yet representative of their respective control and treatment environments and far enough away from the treatment boundary (100 m) to avoid ecological edge effects along the treatment boundary line.

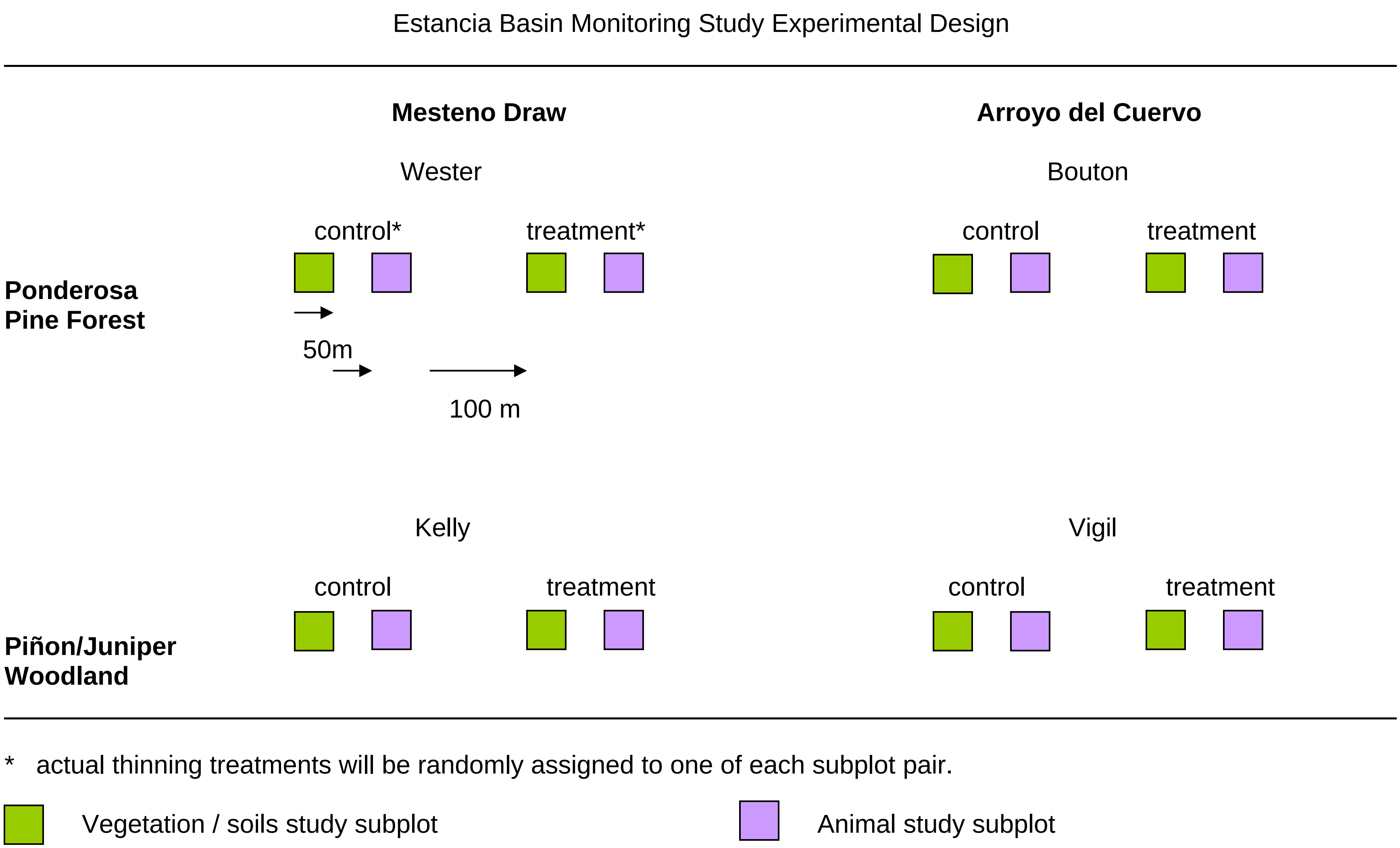

Figure 2.13. Schematic design of monitoring plot layout representing all four study sites.

Environmental monitoring study plot design for surface soils and vegetation follow the recommendations of Herrick et al. (2005) (http://usda-ars.nmsu.edu/JER/Monit_Assess/monitoring_main.php). Those monitoring recommendations are applicable to the study area because the relatively open understory soils and vegetation of ponderosa pine and piñon/juniper woodlands are similar to those of more open rangelands. This study design is similar to that recommended for forest monitoring by the USFS Forest Inventory and Analysis Guide (USFS 2005; http://www.fia.fs.fed.us/library/field-guides-methods-proc/), and measurements for trees on those study plots will follow the USFS recommendations.

Each of the T and C soils and vegetation study plots consists of a triangular spoke design of three 10 × 30 m rectangular subplots as recommended by Herrick et al. (2005) (Figure 2.14 and Figure 2.15). One 30-m vegetation sampling line bisects the center of each of the three subplots. Circular 14.6 m diameter tree monitoring plots are superimposed upon each soil and vegetation plot, with tree measurement circles situated at the center and the outside ends of each 30 m vegetation lines.

Each T and C mammal and bird-monitoring plot is a square plot 50 m on each side, and consists of a 6 by 6 grid of sample (rodent trap) points at 10 m intervals, for a total of 36 sample points or traps per plot (Figure 2.16).

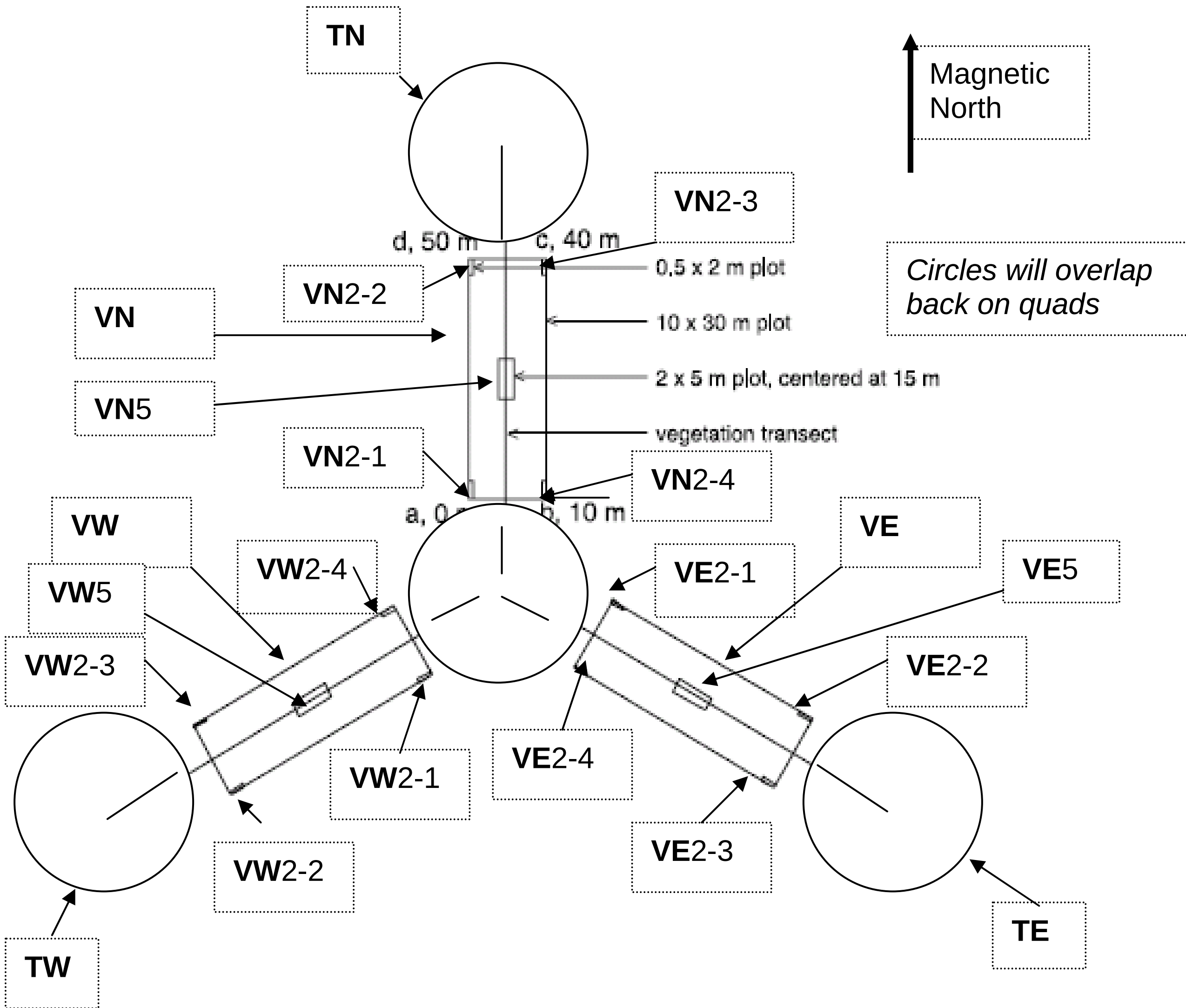

Figure 2.14. Vegetation, soils, and tree study plot design with subplot code names.

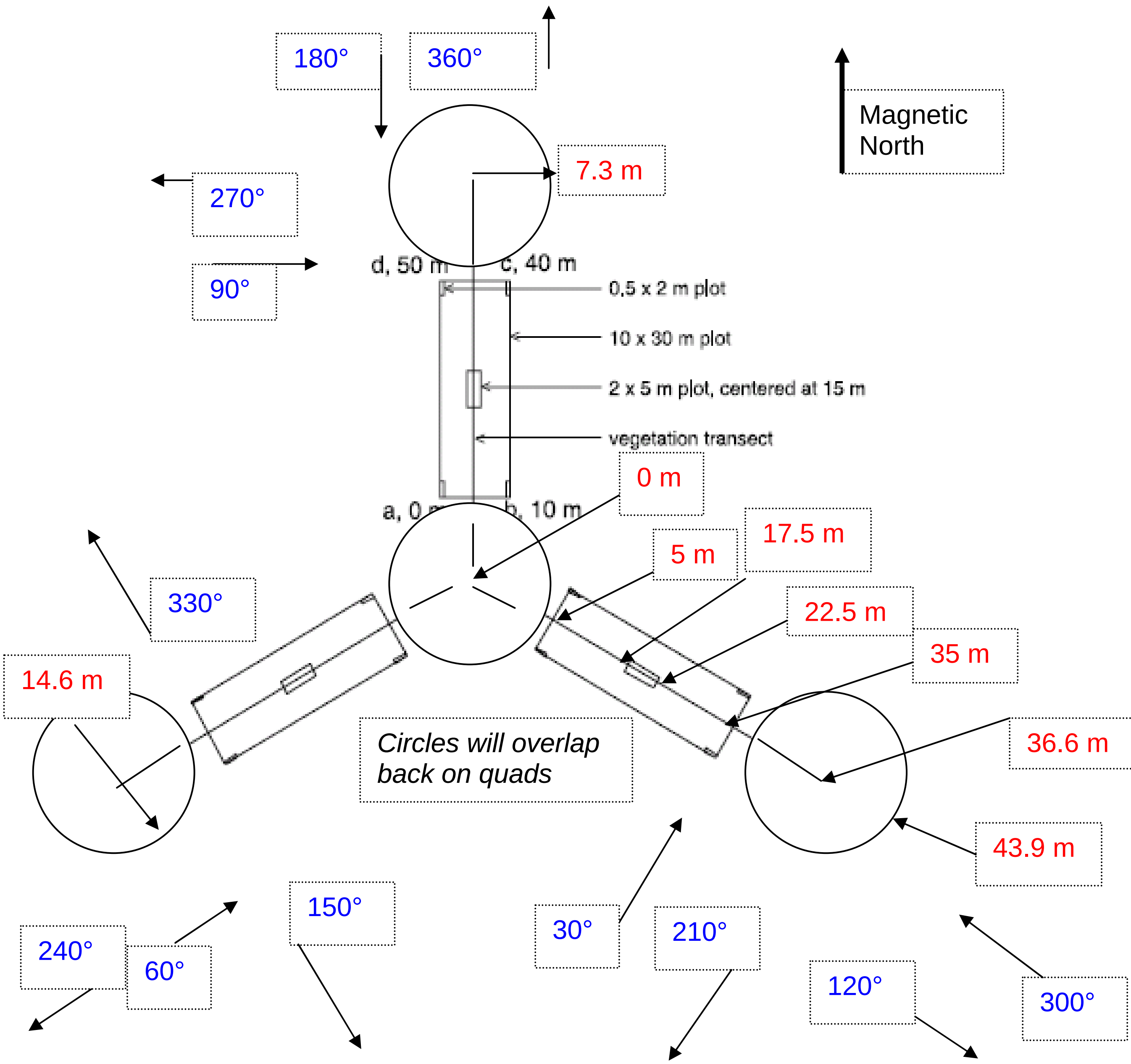

Figure 2.15. Vegetation, soils, and tree study plot design with subplot dimensions and orientation.

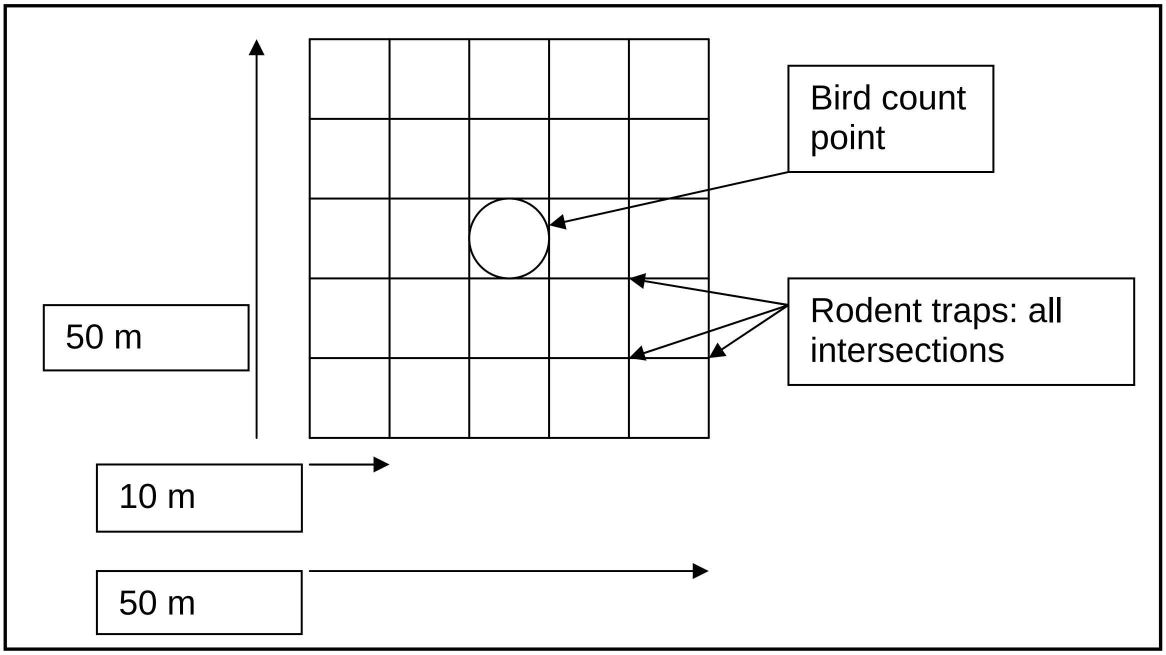

Figure 2.16. Small mammal trapping grid design and bird point count survey observer location.

2.2.5 FIELD SAMPLING SCHEDULE

Field measurements of weather variables and soil surface runoff are taken year-round by use of automated data loggers attached to weather stations and soil surface hydrology flumes. The effects of weather on the watershed occur throughout the year, so we must monitor weather over that time scale. Since soil surface water flow will occur at any time of the year significant rainfall occurs, we also must monitor surface water runoff year-round. Those automated systems were installed and began collecting hourly data in November of 2007.

The measurable effects of forest thinning practices on soil surface characteristics, plants, and animals will occur over longer periods. All of those measurements will begin in May and September of 2008. Soil surface characteristics, vegetation, and animals will be sampled during two field-sampling periods each year. Soil surface characteristics are best measured when surface soils are relatively dry, so those measurements will be taken during the typical dry season during the late spring and early summer, in May. Vegetation measurements will be taken during the end of the typical growing season in late summer/early autumn, in September. Animals (birds and small mammals) will be measured at both times of the year to provide additional data to monitor changes in relative densities and species composition of both, and to accommodate different seasonal bird faunas associated with spring breeding and fall migration.

The purpose of this project is to provide long-term (multiple years) monitoring data on soils, hydrology, plants, and animals. Annual "snap-shot" samples of soil surface characteristics and vegetation will be appropriate for this purpose. More frequent seasonal sampling would provide data on seasonal differences of soils and plants; however, that is not within the scope of this project. Furthermore, frequent sampling and walking on permanent study plots can create undesirable human researcher impacts on those same plots.

2.2.6 Data Collection

Field data from measurements on the various weather, soil, hydrology, and plant and animal response variables will be recorded by automated data loggers or on field data sheets by field crew personnel. Weather and hydrological flume data will be recorded on automated data loggers. Those data will be offloaded from data loggers on to laptop computers and then transferred to office computers for analysis. Soils, vegetation, and animal field data will be recorded on paper field forms. Those forms will be taken to the office and entered on Microsoft Excel spreadsheets. Those data will be stored in both spreadsheet format and as ASCII text files. Photographs from permanent photo points will be archived along with the data files.

2.2.7 Data Analysis and Interpretation

Measurement data will be summarized and analyzed using a variety of analytical techniques, depending on the type of data and the desired analysis. Details of proposed data analyses are provided under each category below. Statistical testing will follow analytical procedures found in Krebs 1998; Gotelli and Ellison 2004; McCune and Grace 2002; Ludwig and Reynolds 1988; and Thompson et al. 1998. We will use SigmaStat (Systat Software Inc. 2002) statistical analysis software to perform most statistical tests of data comparing parameters from treatment and control study plots. Separate paired t-tests will be run for ponderosa pine and PJ woodland sites. The same analytical statistical approach will be used for all data sets listed below, except as stated otherwise. Linear regression will also be used to examine relationships between variables or parameters.

2.2.7.1 Rainfall and Temperature Data

Hourly data for ambient temperature, rainfall, soil moisture, soil temperature will be summed to daily means, along with daily extreme minimum and maximum temperatures. Summary data will be graphed as calendar year annual charts. Data matrix values will be used for parametric statistical tests of mean measurement value differences between treatment and control plots.

2.2.7.2 Hydrology Data

Data on water levels collected in surface hydrology flumes will be recorded by automated data loggers and will be summarized as with weather data above. Surfacewater runoff volumes also will be correlated with rainfall amounts and soil surface characteristics.

2.2.7.3 Soil Data

Soil surface stability, erosion, and water infiltration measurement values for each plot will be stored in separate data matrices for each variable. Data analyses will be conducted as described above.

2.2.7.4 Vegetation Data

Vegetation line-intercept canopy cover data, tree counts and morphology, tree growth data, and tree insect pest data all will be stored as separate data matrices. Data analyses will be conducted as described above. Additionally, plant species community composition and species diversity data will be analyzed with non-parametric multivariate community similarity analyses using PC-ORD software (MjM Software Design 1999; McCune and Grace 2002).

2.2.7.5 Animal Data

Bird counts and small mammal abundance values for each plot will be stored in separate data matrices. Data analyses will be conducted as described above. Additionally, animal species community composition and species diversity data will be analyzed with non-parametric multivariate community similarity analyses using PC-ORD software (MjM Software Design 1999; McCune and Grace 2002).

2.2.8 Data Management and Dissemination

SWCA will handle and manage all data within our secured digital environment. SWCA staff will work closely with CPSWCD personnel to define the appropriate format for data and reports. All data will be available at any time for inspection or use by CPSWCD personnel on an SWCA ftp site. We will provide annual summary data sets and graphics to the New Mexico Forest and Watershed Restoration Institute (NMFWRI) (http://www.nmhu.edu/nmfwri/). NMFWRI will work with SWCA to post data summaries and findings from this monitoring study on their website for public access. SWCA will provide CPSWCD with annual summary reports (January of each year) on the previous year's findings, along with complete data files representing each year.

2.3 Sampling Design and Data Collection

2.3.1 Meteorological Data

Precipitation and ambient temperature are being recorded from the center of each study plot using eight meteorological stations. Each station is equipped with automated recording instruments that collect ambient temperature and precipitation on an hourly basis, year-round, and store the data on data loggers (Spectrum Technologies, Inc. (2007); Watchdog® data logger systems). Tipping-bucket rain gauges are used for the automated weather station, and they measure both snow and rain precipitation. Data are collected from field data loggers on a monthly basis and transported into the SWCA data management system.

Soil moisture is measured in hourly increments and data recorded on the same Watchdog® data loggers as weather data. Watermark® soil moisture probes have been installed at 10–15 cm (probes are 5 cm long) below the soil surface at each weather station on each study plot. A soil temperature sensor also has been installed 10 cm below the soil surface at each study plot weather station.

2.3.2 Study Plot Repeat Photo Points

Multiple photographs of each study plot will be taken each year in September from the same permanent points and orientation, capturing images of the same areas of the study. These repeat photos will provide valuable qualitative information on environmental features of each study plot over time, including vegetation structure and composition, relative amounts of open or nonvegetated areas and bare ground, and soil surface conditions. Permanent photo points will be located at the center points of each of the three circular tree sampling subplots located on each of the vegetation/soils study plots (see Figures 2.14 and 2.15). Photographs will be taken aimed toward the center of each study plot, for a total of 3 photographs per study plot, providing views across the lengths of each 30 × 10 m vegetation monitoring subplot. Photographs will be taken in September of each year when vegetation measurements are taken. A 2 m tall, white 1-inch diameter PVC pipe marked with black 0.5 meter increments will be placed vertically on the soil surface at the outside edge of each 5 × 2 meter vegetation monitoring subplot within each 30 × 10 m vegetation monitoring subplot to provide a reference scale for each photograph.

A digital camera with at least 5 megapixels of resolution will be used to take digital color images. The camera lens will be consistently set at about 30 mm to provide a slight wide angel view each time. This setting will be determined in the field when the first photos are taken, and maintained constantly throughout the remainder of the monitoring study. The outside 10 m edge of each 30 × 10 m vegetation monitoring subplot will occupy the bottom edge of each photograph, and the top of the 2 m tall reference pole will be situated just inside the top edge of each image, and the photograph will be centered on that same reference pole.

2.3.3 Soils

We will measure and monitor soil features that are indicators of watershed and forest health and those that may be disrupted following forest thinning treatments. Specifically we will measure features of the soil surface that relate to erosion and water infiltration, as well as subsurface soil moisture and temperature that also relate to water infiltration and changes in amounts of solar radiation reaching the soil surface. Soil surface characteristics will be measured on all soil/vegetation study plots once each year in May. Soil characteristics are best measured when the soil surfaces are dry; typically late spring and early summer is a dry season in New Mexico. A soil pit will be dug near the center of each study plot to document and characterize the A-horizon for each plot, and assess correspondence to the overall study area soils map (USDA-NRCS 2004; Bourlier et al. 1970).

Soil surface features are critical to terrestrial hydrological processes and ecosystem function and health. Soil surface characteristics will be assessed and monitored on each of the principal study plots from point locations that will not be impacted by other monitoring activities. Use of heavy equipment and mulching with wood chips could have significant impacts on soil surface characteristics. Sampling points are located at random distances along each of the vegetation measurement lines. Each sample point is 1 m away from, and consistently perpendicular to, one side of the vegetation lines, where soils will not be impacted by researchers measuring vegetation. The following soil surface characteristics will be measured and monitored over time.

2.3.3.1 Soil Classification, Morphology, and Chemistry

The depth and physical and chemical characteristics of the soil A-horizon are important to plant productivity, soil biota, and water infiltration. A small (approximately 1 m deep by 1 m wide) soil characterization pit will be dug adjacent to each study plot once during the first year of the study. The soil pits will be used to determine the soil classification (species), morphology, and texture, and depth of the A-horizon. Additional 20 cm deep by 4 cm diameter impact soil cores will be taken from points 2 m beyond the end-points of each vegetation transect, and from one point adjacent to the start point of each of the three transects near the center of each study plot, following the USFS Forest Inventory and Analysis Guide procedures (USFS 2005). These cores will provide additional samples of A-horizon depth around each study plot. The A-horizon portion of all four core samples from each plot will be pooled into one sample for laboratory chemical analysis. Soil analysis will measure total bulk density, water content, and coarse fragment [>0.08 inches (>2 mm)] content, pH in water and in 0.01 M CaCl2, total carbon, total organic carbon, total inorganic carbon (carbonates) (pH>7.5 soils only), total nitrogen, exchangeable cations (Na, K, Mg, Ca, Al, Mn), extractable sulfur and trace metals, extractable phosphorus (Bray 1 method for pH < 6 soils, Olsen method for pH > 6 soils. Core samples will be sent to the Soil, Water, and Agriculture Testing Laboratory, New Mexico State University, for analysis. Core samples will be repeated on an annual basis.

2.3.3.2 Soil Surface Stability

The soil surface stability test developed by Herrick et al. (2005) provides information on soil texture, the extent of soil structural development and resistance to erosion, and the biological integrity of the surface organic matter and soil biota. Soil surface stability reflects the presence of both abiotic and cryptobiotic surface crusts. The test measures the stability of the soil matrix when exposed to rapid wetting, such as occurs during intense rainfall. Unstable soil surfaces are prone to erosion when exposed to intense rainfall. Surface stability also indicates general stability of the soil surface when exposed to wind and other disturbances (Herrick et al. 2005). One sample point will be randomly located along each vegetation line, 1 m from and perpendicular to the line. The test will be repeated along the same lines, but not at the same points, once each year, during the dry season (May).

2.3.3.3 Soil Surface Erosion

Soil surface erosion is an important aspect of watershed and forest health. Soil surface erosion will be measured by use of soil erosion bridges (Shakesby 1993) on each of the study plots. The erosion bridges are similar to those used by Shakesby (1993) and White and Loftin (2000) and consist of two permanent 0.5 inch diameter steel rebar support posts and a portable aluminum square pipe bridge with a series of pin-drop holes, and 1 cm diameter by 60 cm long aluminum rod drop pins. The steel pipe support posts are 1.2 m apart, and support a 1.2 m portable bridge approximately 30 cm above the initial soil surface. 20 pin-drop holes are distributed at 5 cm intervals along the bridge, for a horizontal measurement area of 100 cm (1 m) across the soil surface. Repeat measurements will be made from the permanently positioned top of the bridge to the soil surface once each year in May.

2.3.3.4 Soil Water Infiltration

Water infiltration into the soil surface is an important component of rainwater availability to vegetation and groundwater, in contrast to the destructive effects of surface runoff and erosion. Water infiltration will be measured at one randomly located point along each vegetation line, each at a point 1 m from and perpendicular to the line. Water infiltration will be measured using the single-ring infiltrometer methods described by Herrick et al. (2005) during the dry season (May).

2.3.3.5 Soil Moisture

Soil moisture is critical to plant survival, growth, and species composition. Subsurface soil moisture varies as functions of surfacewater infiltration, soil particle water retention, and water loss through evaporation or uptake by plant roots. Input from infiltration and loss due to evaporation or plant uptake may change as a result of forest thinning resulting from soil disturbance and changes in plant canopy cover and composition.

We will use a portable time domain reflectometer (TDR) soil moisture meter to measure soil moisture on the study plots. TDR meters determine soil moisture by measuring the rate that an electromagnetic wave travels along a waveguide (the device rods) within the soil matrix. The speed of the wave through the soil is a function of the bulk dielectric permittivity of the soil, which in turn is a function of soil water content. The TDR converts dielectric permittivity to water content and provides a measure of soil volumetric water content. The TDR device is equipped with two 12 cm rods, which will be inserted to a depth of 12 cm into the soil at each measurement point. The TDR will provide an average water content of the soil for a cylinder of soil 9.3 cm across and 12 cm deep at each measurement point.

We will measure and monitor subsurface soil moisture and temperature from 12 systematically located points on each of the vegetation/soils monitoring plots. Each measurement point is located immediately outside of the outer-center of each of the small 0.5 × 2 m vegetation measurement subplots, four of which are in each corner of the three 30 × 10 m vegetation subplots. Measurements will be taken once every 2 months throughout the year, for a total of 6 readings each year (February, April, June, August, October, December), across the four seasons. The permanent soil moisture probe associated with the rain gauge on each study plot will provide continuous study site reference soil moisture data for depths of 10–15 cm below the soil surface to relate to the TDR interval data.

2.3.3.6 Soil Temperature

Soil temperature also is important to plant survival and growth, and affects soil water content by affecting evaporation and plant root uptake. Soil temperature also may change as a result of forest thinning because of reduced forest canopy cover and increased insulation. Portable, 10 cm digital soil moisture temperature probes will be used to measure soil temperature at 10 cm below the soil surface at the same locations and at the same times that soil moisture is measured as described above. The permanently placed soil temperature probe associated with the rain gauge on each study plot will provide continuous study site reference soil temperature data for depths of 10 cm below the soil surface, relative to the interval temperature data collected with the portable temperature probe.

2.3.4 HYDROLOGY

Previous forest studies suggest that forest treatment has a measurable effect on surface water runoff from treated watersheds. At each study site, the T and C study plots were placed on different areas with similar features (e.g., aspect, slope, vegetation). The study plots were selected to reflect a readily discernable point of concentration of runoff from each site. Permanent flumes were installed at the outlet of each of the paired sites to monitor surface flow from the T and C study plots.

2.3.4.1 Surface Flow/Runoff

Pre-fabricated fiberglass flumes were installed to measure precipitation runoff from the study plots. Pre-fabricated flumes can be obtained with various maximum flow ratings, ranging from approximately 0.2 to 20 cubic feet per second (cfs). Most flumes have the same minimum flow rating of 0.001 cfs. The maximum capacity of each flume will be estimated by calculating the expected runoff from each sub-watershed encompassing each study plot, using common a mathematical model such as the Rational method (Q = CiA).

Measurement of water flow in the flumes is automated by use of a stilling well built into the flume, and an integrated pressure transducer/datalogger placed into the stilling well. These dataloggers can automatically record the pressure on the instrument sensor, which mathematically correlates to depth of water above the sensor. Depth of water, in turn, mathematically correlates to flow through the flume. The In-Situ Level Troll 700 model pressure transducer/dataloggers are being used to monitor depth of water in each stilling well. Vented cables are used with the dataloggers to allow real-time download of data without disturbing the datalogger in the stilling well, while automatically correcting atmospheric pressure changes than can erroneously be recorded as water level changes.

None of the sub-watersheds study plots selected exhibit perennial or intermittent streamflow; all runoff to be monitored is expected to occur as separate finite runoff events, either from individual precipitation events (e.g., summer thunderstorms) or from snowmelt. Long periods with no flow in the flumes are expected. In order to adequately record these ephemeral events, dataloggers will collect data every hour. The dataloggers have a capacity of 100,000 data points; at this rate, the datalogger memory will fill in approximately 6 months. Downloads are expected to be conducted every 3 months. Once full, the datalogger will be set to start continually erase the earliest data, which will have already been downloaded.

2.3.4.2 Subsurface Water Table

Based on the preliminary site inspections and examination of geologic maps, the geology of the study area and study sites is not amenable to the installation of piezometers to monitor the local (adjacent to study plots) water tables. We expect precipitation to either be stored in the soil profile and/or lost through evapotranspiration, and/or flow off the study plots as surfacewater. Soil surfacewater runoff should then drain into ephemeral stream channels or arroyos, and infiltrate to the water table via subterranean conduits along those drainage channels (see Wilson and Guan 2004). Since the primary drainage channels are spatially removed from specific forest thinning projects and our monitoring study sites, direct links between the status of the water table along those drainages and their wider watersheds to specific thinning projects or our study plots on the landscape cannot be made. This study will provide surface hydrology monitoring data that will determine if forest thinning affects local soil surface runoff, and water recharge into landscape-scale drainage features, relative to infiltration into surface soils.

2.3.5 VEGETATION

Vegetation measurement and monitoring protocols will follow the methods developed by Herrick et al. (2005). Point line-intercept will be used to characterize the plant species composition and foliage canopy height profile up to 1 m above the ground surface. Gap line-intercept will be used to measure both plant canopy horizontal cover and soil surface cover, including bare soil, rocks, cryptobiotic crusts, leaf litter, and dead and down woody material. Animal (deer, livestock) tracks and scat also will be recorded along the lines. Total plant species lists will be compiled from the line-intercept data to provide species composition and diversity information. Tree seedlings and saplings less than 1 m in height will also be measured along the lines.

Monitoring percentage of cover can provide valuable insight on the response of vegetation to thinning practices and can aid in determining if the vegetation is providing (or making progress toward providing) adequate ground cover to protect the soil and/or arroyo banks from erosion. The point line-intercept method (Elzinga et al. 2001, Herrick et al. 2005) is SWCA's preferred methodology for measuring and monitoring percent ground, cover because it is both rapid and accurate. The measurements obtained during this study are related to water erosion, water infiltration, and the ability of a site to resist and recover from degradation.

Gap-intercept measurements, which will be taken at all permanent vegetation monitoring transect locations, will provide information about the proportion of land consisting of large gaps between plants. Special attention will be given to documenting and monitoring the occurrence of non-native invasive weeds. The analysis of the vegetative monitoring indicators, combined with the information gathered from the soil stability test, can help determine changes in erosive characteristics, such as loss or gain of plant cover, changes in the vegetation's spatial distribution, or reduced soil stability within study sites.

Trees (over 1 m in height) will be mapped on each of the three 14.6 m diameter tree subplots of each study plot. All trees will be identified to species and tagged for future reference. The height and crown dimensions of each tree will be measured annually using standard methods (USFS Forest Inventory and Analysis Guide, USFS 2005). Additionally, dendrometer bands will be placed on three randomly selected trees from each species from each subplot (or as many as available up to three), and growth measurements will be recorded once each year. Tree seedling abundance and survival will be measured along the vegetation sampling lines.

2.3.5.1 Plant Species, Growth-form, and Life-history Composition

The temporal dynamics of plant species diversity and relative proportions of different growth-forms and life-history strategies can indicate the effects of environmental impacts and the health and stability of plant communities. Lists of all plant species on the study plots will be compiled from line-intercept data. Additionally, we will visually survey each entire study plot to document all species, including those that may be outside of the vegetation measurement lines. A list of all plants including their growth-form (e.g., grasses, forbs, trees) and life histories (e.g., annual, perennial) will be constructed. Particular attention will be given to monitoring the presence and abundance of invasive weedy species that typically respond favorably to environmental disturbance and are negative indicators of forest and watershed health.

2.3.5.2 Plant Canopy Cover

Plant canopy cover is an important measure of plant productivity relative to biomass production, and usually is positively correlated with above ground net-primary production. Plant canopy cover will be measured from the gap line-intercept transects to provide measures of plant cover by species, growth-form, life history, and total plant foliage cover. Plant canopy cover will be measured once each year at the end of the growing season in September, and monitored over time.

2.3.5.3 Plant Foliage Heights

Plant foliage height is often an index of plant productivity, and health in taller plants of a given species in an area tends to produce more biomass. Point line-intercept data will provide height measures for all plants less than 1 m in height. Tree canopy measurements will provide heights for all plants greater than 1 m in height on the study plots. Plant heights will be measured once each year in September.

2.3.5.4 Tree Density

The key environmental impact that this monitoring study is addressing is the effect of tree thinning on forest and watershed health. We will measure and monitor the densities of trees on all study plots to: (1) document the actual reduction in tree density (following one year of pre-thinning measurements) and (2) monitor how tree densities then change following thinning treatments comparatively from both treated and control study plots. Tree densities will be determined by mapping all trees greater than 1 m in height on all study plots each year in September. Each mapped tree will be labeled and tagged, and the measurements of crown structure, diameter, and insect damage will be recorded for each tagged tree.

2.3.5.5 Tree Crown Structure and Heights

Tree crown structure has a great effect on the subcanopy environments by filtering sunlight and precipitation, as well as providing habitat for animals. Crown structure may also change as a result of thinning, as remaining tree crowns expand to utilize more available sunlight. Alternatively, a reduction in crown density from branch dieback may indicate poor tree health. Measuring and monitoring tree crown structure will be important to determine how the environments of our study plots change over time while indicating tree response to thinning. The crown structures of all trees greater than 1 m in height will be measured using standard USFS Forest Inventory and Analysis Program protocols (USFS 2005). Crown dimensions will be measured in September of each year.

2.3.5.6 Tree Diameter

Tree trunk diameters tend to be positively correlated with tree growth, and can be used to index tree production along with crown structure measurements. Tree trunk diameter at breast height (DBH) will be measured on all trees over 1 m in height on study plots once each year in September. Flexible dendrometer bands will be placed on three randomly selected trees of each species on each study plot. Band growth increments will be measured each year in September.

2.3.5.7 Tree Insect Damage

The populations of many forest insects are known to increase on conifer trees that are physiologically stressed, particularly relative to water status. Piñon and ponderosa pine are hosts to several species of bark beetle (especially *Ips confuses* and *Dendroctonus ponderosae*) and needle scale insects (especially *Matsucoccus acalyptus* and *Chionaspis pinifoliae).* The western spruce budworm (*Choristoneura occidentalis*) attacks a number of conifers at higher elevations.

Juniper is host to several branch tip insects and borers (twig pruners and pitch moths). The incidence of bark beetles, needle scales, spruce budworms, twig pruners, and pitch moths will be monitored on all conifer trees mapped and tagged on each of the study plots as an additional index of forest tree health. Presence of the bark beetle (wood dust/frass exuding from bark) and needle scales (dark scales on yellowing needles) and evidence of foliage mortality (pitch moths and twig borers) will be visually assessed each year when tree measurements are taken from the mapped/tagged trees on each of the subplots. Each tree will be scored on a scale of none (0), low (1–20%), moderate (21–50%), or high (>51%) incidence of insect occurrence (observable presence) and/or damage (observable tree tissue damage or senescence).

2.3.5.8 Down Woody Materials and Leaf Litter

Dead and down woody material, such as tree branches and logs, may have important roles in forest ecology and hydrology. Down woody materials can influence soil surface hydrology and infiltration by reducing surface runoff. Down woody materials also modify habitats for plants and animals, and provide fuel for fire. We will map all down woody materials greater than 1 inch in diameter on each of the soil/vegetation study plots and monitor those materials over time. We also will record all down and dead woody material encountered along the vegetation intercept lines when vegetation is measured.

2.3.6 ANIMALS

2.3.6.1 Bird Species Composition and Relative Abundance

Bird monitoring surveys are generally conducted by use of visual and acoustical observation surveys either along linear transects through an area or from single point locations within an area, and repeated over time (Thompson et al. 1998; Williams et al. 2001; Morrison et al. 2001). Transects are employed for large areas that are generally at least 1 km in length, while point counts tend to be used for smaller areas of less than 1 ha. Since this study focuses on plots less than 1 ha in size, point counts are the most appropriate method to monitor birds associated with local study plot environments. The number of replicate sample points varies with research goals and study design. We intend to characterize the bird communities associated with each vegetation/soils study plot, so we will center bird point counts on each of the eight study plots. Timing and frequency of bird counts varies with research goals and species of interest. Bird activity (and therefore, detectability) is greatest in the early morning, so we will conduct counts then. Multiple daily counts provide the most accurate data, so we will conduct bird point counts on each study plot for three consecutive mornings. Seasonally, birds of the Manzano Mountains establish territories and breed in the spring (April-June), so we will conduct breeding bird counts at that time of year. Fall migration occurs from August-October, so we also will conduct counts for migratory birds in September. This sampling design will allow us to determine how both breeding bird species and migratory bird species are utilizing the treated and non-treated woodland habitats.

Bird surveys will consist of point counts centered on each of the eight vegetation and soils study plots. An observer will record all birds visually observed and/or heard for one 20-minute period within 2 hours of sunrise. Bird counts will be conducted during the breeding season (May) and during the fall migration (September), at the same time that rodent trapping is conducted. Data produced will include species composition and total numbers of individuals recorded from each study plot for each morning. As described above for rodents, bird monitoring will focus on certain species that have good detection probabilities, have known habitat preferences, and may serve as indicators of environmental change.

2.3.6.2 Small Mammal Species Composition and Relative Abundance

Small mammals (i.e., species generally less than 300 grams body weight) are generally inventoried and monitored by use of small box traps that capture animals alive so they may be released again (Wilson et al. 1996; Thompson et al. 1998; Morrison et al. 2001). The traps have a spring plate mechanism that causes the open door to close when an animal enters the trap and releases the mechanism. The majority of small mammals in the Manzano Mountains are rodents, which are typically active at night, so trapping for this project will be conducted during the nighttime hours.

The number, spatial arrangements, and temporal sampling of small mammal trap arrays vary with research scope, environment, and biology of target mammal species. The goals of this monitoring project are to compare the rodent communities (species composition and relative abundance) of treatment and control sites of relatively small areas, so we will use grid arrays of trap lines of the same size and dimensions for all comparative plots. Capture-mark-recapture studies with associated mathematical density estimator algorithms are often used in order to determine recapture rates, estimate densities, and to monitor individual animals over time (Wilson et al. 1996; Thompson et al. 1998). Such approaches require rigorous sampling efforts over large or replicated areas, along with considerable repeat sampling over time in order produce data useful for density estimates. Such approaches also are subject to many mathematical and biological assumptions that often cannot be met or verified in the field (Wilson et al. 1996; Thompson et al. 1998; Williams et al. 2001). Since our sampling effort will be not be spatially or temporally extensive or intensive, and since we will not be able to verify assumptions relative to density estimation methods, we will simply document and monitor the species composition and relative abundances of all rodent species on the comparative study plots. Such an approach will provide useful information as to whether or not forest thinning practices have changed the small mammal communities and as to how those communities and species abundances change comparatively over time.

Small-mammal trapping will be conducted on a 50 × 50 m grid located adjacent to each main study plot. Each mammal-trapping grid consists of an array of 6 trap lines, each with 6 traps at 10-m intervals, by a perpendicular array of another 6 trap lines, each with 6 traps, making a 6 by 6 grid of 36 traps. Trapping will be conducted during one night without a moon, twice each year (May, September). All animals captured will be identified to species. Data produced will include species composition and total numbers of individuals captured representing each species. Relative abundance data will be useful for assessing changes within and between populations of species over time.

Entire community-relative abundance surveys tend to be biased for some species over others due to sampling biases, such as variation in detection or capture of different species, and may not provide an accurate estimate of the true relative abundance of all species present (Thompson et al. 1998). Certain rodent species that can be sampled with consistent bias, and that may be indicators of particular environmental conditions, will provide the most useful monitoring data. Likely species to focus on for monitoring changes in populations and rodent assemblages relative to environmental manipulations in the Manzano Mountains include the deer mouse (*Peromyscus maniculatus*), which prefers dense understory upland forest vegetation; the brush mouse (*P. boylii*), which prefers drier but brushy (e.g., oak) habitats; and the piñon mouse (*P. truei*), which prefers piñon woodlands.

2.4 ASSESSMENT OF FOREST AND WATERSHED HEALTH

Overall forest health will be evaluated by assessing and integrating the results of the soil, plant, and animal monitoring measurements. Soil stability, infiltration, and organic and nitrogen content will provide measures of soil health. Vegetation and animal monitoring will provide measures of plant and animal production, species diversity, stability of populations over time, and the status of non-native invasive weeds. Assessments of forest insect loads on conifer trees will provide a measure of tree physiological status, especially relative to water stress.

2.4.1 SOIL QUALITY

Several characteristics of surface soils are important indicators of ecosystem productivity and health. Soil quality not only affects hydrology, but also vegetation and animals. Assessment of comparative changes in soil nutrient status, surface stability, water infiltration, and subsurface soil moisture and temperature will provide us with the ability to determine how forest thinning may affect soil structure and productivity. Environmental impacts from forest thinning treatments, such as soil surface disturbance and mulching with wood chips will affect soil surfaces. However, negative impacts such as surface disturbance from equipment may be offset by positive impacts of mulching.

2.4.2 HYDROLOGY

Based on the results of previous studies conducted in the arid Southwest, significant changes to land cover, whether from intentional land treatment or inadvertent wildfire, have an effect on the hydrologic regime. These hydrologic changes are related to the expected changes in soil productivity, soil loss, canopy cover, and vegetation density; other hydrologic changes include changes in soil moisture and changes in the timing or seasonality, type (snow or rainfall), amount, duration, and frequency of runoff events.

Water falling on a watershed can only end up in several places: it can be intercepted by vegetation and eventually evaporate back to the atmosphere, it can reach the ground surface and infiltrate, or it can runoff. Once infiltrating, the water will stay in the soil profile, infiltrate deep enough to become part of the saturated aquifer, or be withdrawn for use by vegetation. The instrumentation for this study is designed to provide direct and indirect measurement of a subset of these parameters, in order to assess how the land treatment has affected the hydrologic cycle. With respect to assessing watershed health, there is usually a tradeoff between these different parameters, depending on what the desired outcome is.

Runoff will be monitored to assess the ability for land treatment to increase water yield. This will be assessed by two analyses. First, the runoff from the T watershed will be compared to the runoff from the C watershed. For this comparison, the volume will be weighted by acreage unless the T and C watersheds are identical in size. Second, the runoff on the T watershed will be compared before and after the treatment occurs. For this comparison, the volume will be weighted to account for differences in precipitation during the pre- and post-treatment periods.

Runoff will also be compared between the T and C watersheds, and the pre- and post-treatment T watershed with respect to the frequency of runoff events, when those events occur during the year, and how long they last. These parameters are important when assessing the impact of the runoff on the watershed as a whole. Short, flashy runoff events may carry a lot of water, but because of their short duration, they offer very limited opportunity for aquifer recharge and are more destructive. Longer events, as might occur during spring snowmelt, are generally more beneficial to the watershed.

Direct infiltration of precipitation to the aquifer after it falls on the treatment watershed is possible, but unlikely to occur. More likely, the soil profile will store the moisture for use by vegetation. Soil moisture will be measured directly; greater soil moisture generally would be considered a beneficial change with respect to watershed health. The amount of precipitation left in the soil profile will also be measured indirectly by watching vegetation responses, as vegetation density should improve with increased water availability.

2.4.3 Vegetation/ Forest

Monitoring of plant species diversity, plant foliage canopy cover, and height will provide us with comparative assessments of the effects of forest thinning on ecosystem productivity. Although we will not be measuring actual biomass production in the form of net primary production (NPP) in terms of biomass per area, canopy cover and height are generally correlated with NPP. Tree density and canopy and trunk diameter measurements will provide us with an assessment of tree productivity. Plant productivity and diversity are related to soil nutrient status and soil water availability to plant roots. Comparatively assessing vegetation/tree measurements with soil and hydrology data on treatment and control plots will allow us to directly assess the relationships between soils, hydrology, plant productivity, and forest thinning treatments. Furthermore, assessment of tree insect pest loads will provide us with additional measures of forest health, since insect pests of conifers tend to concentrate on physiologically stressed trees.

2.4.4 Animals

As with plants, animal species diversity and abundance are measures of ecosystem productivity or health. The presence and abundance of animals are a function of the availability of food and breeding habitat resources. Since plants provide the principal food and habitat resources for animals, we will assess the relationships between plant production, species, and physical diversity, and animal species diversity and abundance. Positive relationships between all of these factors will provide strong indications of ecosystem health and productivity. In addition, all should be directly related to soil health and hydrology.

2.4.5 Wildfire

Data on vegetation/tree cover, density, and structure, along with data on dead and down woody material, also will provide us with measures of potential fuel loads for wildfire.

2.4.6 CONSIDERATIONS FOR CLIMATE CHANGE

A warming climate and associated regional and seasonal changes in rainfall patterns will likely cause changes in the local weather and other environmental conditions, including associated changes of the flora and fauna of our study sites over time (see section 1.5.6 above). Since this is a monitoring study, we will account for such changes by relating our findings to regional climate and environmental trends. Any findings from this study must be interpreted relative to such overall environmental changes. Our non-treated control study plots will serve as reference sites for the affects of climate change relative to the paired thinned or treated study plots.

2.4.7 INTEGRATION

The most powerful aspect of this monitoring study is our ability to integrate responses of all the above ecosystem components to assess overall ecosystem function and health. Positive relationships between plant and animal diversity and productivity will be related to soil and hydrology function, as well as aspects of weather such as rainfall and temperature. Another important component of ecosystem health is stability in components over time. Monitoring all of the above variables over multiple years will allow us to comparatively assess the stability of soils, hydrology, vegetation, and animals relative to forest thinning. A variety of integrated ecosystem components monitored over time in a comparative way between thinned and non-thinned paired study plots will provide us with a powerful assessment of the effects of forest thinning on watershed and forest health.

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