AN ABSTRACT OF THE DISSERTATION OF

<u>Mohamed Abdallah</u> for the degree of <u>Doctor of Philosophy</u> in <u>Rangeland Ecology and Management</u> presented on <u>December 3, 2019</u>.

Title: <u>Ecosystem Processes in Treated and Untreated Western Juniper Systems in Semiarid Watersheds: A Paired Watershed Study in Central Oregon.</u>

Abstract approved:

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Western juniper (*Juniperus occidentalis)* encroachment has been associated with negative ecological and hydrological consequences including reductions in herbaceous production and diversity, deterioration for wildlife habitat, and higher erosion and runoff potentials. As a result, western juniper removal is a common and accepted rangeland management practice. Although studies evaluating the ecological repercussions and removal benefits of western juniper are increasing, quantitative evidence regarding the implications of juniper encroachment and removal on ecosystem carbon and nitrogen pools are lacking. In addition, it is unknown if the hydrologic system (transpiration in relation to soil moisture content) is altered by with or without juniper. This dissertation, as an addition to a central Oregon paired watershed study that began in 1993, addresses the effects of western juniper encroachment and removal on ecosystem processes: carbon, nitrogen, transpiration, and soil moisture. Our study site was a paired watershed in central Oregon where western juniper trees were eliminated in one watershed (treated, 116 ha) and were

left intact in an adjacent watershed (untreated, 96 ha). This research was unique because it involved a paired study approach to monitoring changes in these variables post western juniper removal.

The overarching goals of the study presented here were to: 1) determine ecosystem carbon stocks in an encroached juniper watershed and an adjacent watershed where juniper removal occurred 13 years prior to determinations; 2) quantify and compare major pools of nitrogen in an encroached juniper watershed and an adjacent watershed where juniper removal took place 13 years prior to quantification; 3) determine seasonal transpiration for western juniper in relation to soil moisture in one juniper-dominated watershed and in another watershed where juniper has been removed, with a major goal of improving scientific understanding of the effects of juniper encroachment and removal on hydrology.

Thirteen years after western juniper elimination, we quantified aboveground carbon stocks for western juniper trees, shrubs, grasses, and litter in both the treated and untreated watersheds. We also quantified belowground carbon stocks (roots and soil) in both watersheds at two soil depths (0-25cm and 25-50cm). Aboveground carbon stocks were 5.8 times greater in the untreated than in the treated watershed. On the other hand, root carbon stocks were 2.6 times greater in the treated than in the untreated watershed. Soil carbon stocks at both 0-25 cm and 25-50 cm depth were not affected by juniper cutting. Overall, total ecosystem carbon stocks (average 137.6 Mg C ha$^{-1}$) were not different between watersheds. Most carbon resided belowground (soil 0-50 cm and roots); 84% and 97% of the total ecosystem carbon, respectively, was found in the untreated and treated watershed.

Nitrogen stocks followed a similar pattern to that observed for carbon stocks. As a result of greater aboveground biomass, aboveground nitrogen storage in the untreated watershed (425.4 kg N ha$^{-1}$) was substantially greater than that in the treated watershed (61.9 kg N ha$^{-1}$). On the

other hand, root nitrogen storage was 3.1 times greater in the treated than in the untreated watershed due to the gain of understory root biomass associated with western juniper elimination. Soil nitrogen stores at both 0-25 cm and 25-50 cm depth were not affected by juniper removal. Overall, total ecosystem nitrogen storage (average 1283.2 kg N ha$^{-1}$) was not different between watersheds. Most nitrogen resided belowground (soil 0-50 cm and roots); 69% and 95% of the total ecosystem nitrogen, respectively, was found in the untreated and treated watershed.

We measured juniper transpiration using sap flow sensors for mature and juvenile growth stages in the untreated watershed and for saplings stage in the treated watershed where juniper trees were removed in 2005 but regrowth has occurred. Leaf water potentials were monitored for juniper trees to support the data of transpiration. We examined how seasonal transpiration is related to soil moisture. Results indicate that transpiration was greater in mature followed by juvenile and then saplings, supporting our expectation that water use consumption is a function of juniper's stage of development. The significant differences between predawn and midday leaf water potentials for all juniper stages indicate that some degree of water was lost over the course of the day. The maximum range between these values in July months corresponds with significantly higher transpiration for all juniper stages. Our findings also indicate that annual and seasonal precipitation was highly variable over the course of the study (2017-2019), which was reflected in the mean soil water content (0-80 cm). This resulted in considerable intra- and interannual variation in transpiration. Intra-annually there were two distinct seasonal pulses of transpiration by juniper: spring and summer. In years with heavy winter precipitation (2017 and 2019), juniper exhibited higher transpiration during summer seasons followed by spring and finally the fall months. In the year with a lower summer precipitation (2018), juniper was capable of greater transpiration during the moist period (spring).

This research serves as a basis to indicate the benefits of juniper removal can be attained without substantially affecting the potential for ecosystem carbon and nitrogen pools. Hydrologically, our data suggest that considerable amount of water can be saved in areas with juniper elimination after regrowth with respect to areas with intact mature juniper encroachment. In addition, our study highlights the sensitivity of western juniper woodlands to variations in seasonal precipitation and soil moisture availability.

Ecosystem Processes in Treated and Untreated Western Juniper Systems in Semiarid
Watersheds: A Paired Watershed Study in Central Oregon

By

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I understand that my dissertation will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my dissertation to any reader upon request.

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Mohamed Abdallah, Author

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CONTRIBUTION OF AUTHORS

Additional authors made significant contributions to this dissertation. Prof. Ricardo Mata-Gonzalez contributed to the initial design of the experiments, aided in field sampling methods and edited chapter 2. Prof. Carlos Ochoa contributed to the general design of the study in chapter 4 and contributed to the editing of chapter 2. Prof. Jay Noller contributed to the editing of chapter 2.

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*This dissertation is dedicated to the memory of my father,*
*Ahmed Belgasem*

**LITERATURE REVIEW**

## 1. Introduction

Woody plant encroachment, referred to as the establishment, development and spread of tree or shrub species (Hughes et al., 2006), is one change that has occurred in many ecosystems over the last century (Stevens et al., 2017; Wang et al., 2019). In the US, examples of encroaching species are (*Prosopis glandulosa*) in Texas (Liao et al., 2006), (*Prosopis velutina*) in Arizona (Wheeler et al., 2007; Throop and Archer, 2008), and (*Juniperus virginiana*) in Kansas (McKinley and Blair, 2008). Creosote bush (*Larrea tridentata*)/tar bush (*Flourensia cernua*) encroachment was observed in semi-desert grasslands within the Sonoran and Chihuahuan deserts (Barger et al., 2011). The caldén tree (*Prosopis caldenia*), a woody encroaching plant, was observed in South Africa (Coetsee et al., 2013), and central Argentina (González-Roglich et al., 2014). *Callitris glaucophylla* is observed encroaching species in Australia (Eldridge and Soliveres, 2015). So, woody plant encroachment is a widespread and worldwide phenomenon. The factors of woody plant encroachment are complex and variable by ecoregion, reflecting interactions among climate (e.g., changes in amount and seasonality of precipitation), land use (e.g., grazing by domestic livestock, reductions in fire frequency/intensity), and atmospheric chemistry (e.g., increased carbon dioxide $CO_2$ concentrations) (Archer et al., 1995).

Juniper (*Juniperus* spp) encroachment is one of the most large-scale changes that are occurring in North American rangelands (Baker and Shinneman, 2004; Sankey et al., 2010). The spatial distribution of juniper has increased between 30% and 625% since the mid-19th century throughout the Great Basin (Romme et al., 2009; Sankey et al., 2010) and encroachment rates have varied between 1.5% and 2% per year (Sankey and Germino, 2008; Sankey et al., 2010).

Western juniper (*Juniperus occidentalis* spp. *occidentalis* Hook.) is an encroaching species into the semiarid shrub-steppes of the western United States that occupies 3.6 million hectares in

central and eastern Oregon, northeastern California, southwestern Idaho and northwestern Nevada (Azuma et al., 2005) (Fig. 1.1). Over the last 130 years, western juniper woodlands have extended into huge regions of sagebrush steppe habitat across western north America (Miller et al., 2005; Davies et al., 2011) (Fig. 1.2). The expansion of western juniper has followed two complementary paths since 1870s: filling of juniper trees in areas where it was previously dominated by sagebrush

(*Artemisia tridentata* and similar species) and increases in the density of juniper in areas where it was previously relatively space (Caracciolo et al., 2017).
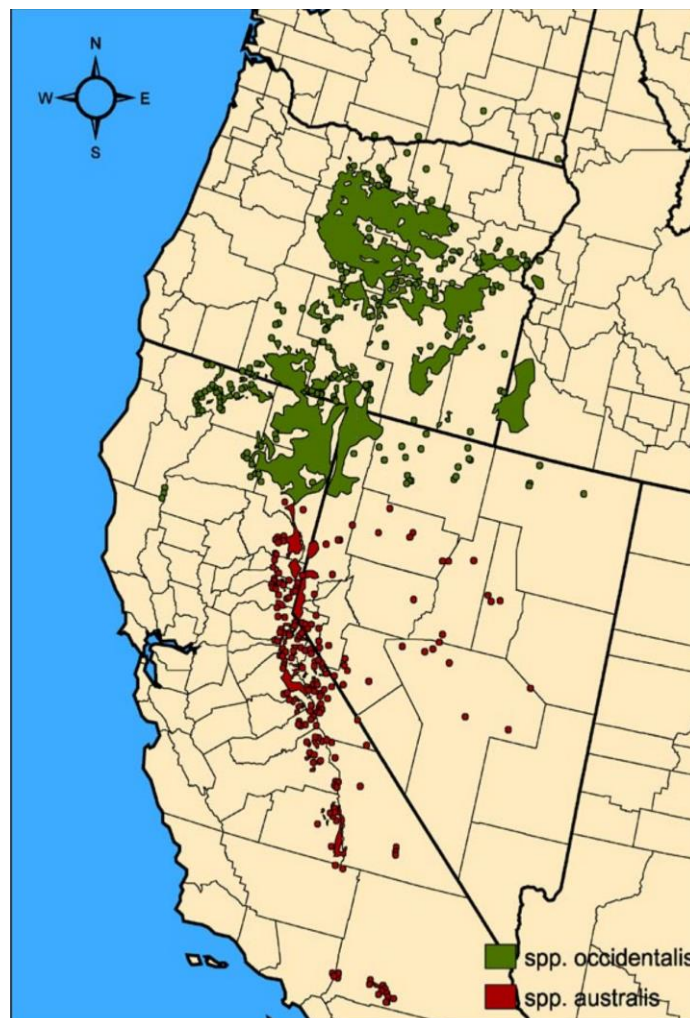
Figure 1.1. Distribution of western juniper (Miller et al., 2005).

The historic distribution and abundance of western juniper woodlands have been dramatically altered by land management practices as well as recent climatic changes (Gedney et al., 1999, Miller and Tausch, 2001). The most common methods for removing invasive juniper are prescribed fire and cutting with chainsaws (Miller et al., 2005), with the cutting treatment often followed-up by a slash removal/redistribution treatment to decrease wildfire risks and allow large animals to move across the site.
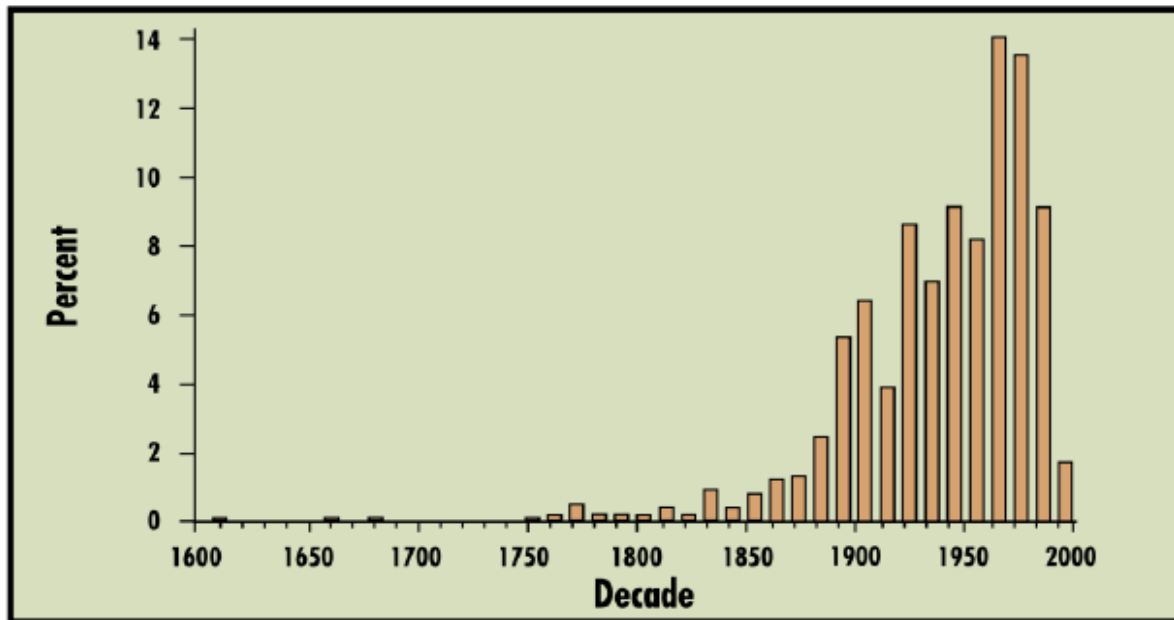
Figure 1.2. Encroachment of western juniper in the last century (Miller et al., 2005)

Western juniper grows on a great diversity of parent materials and soils (Driscoll, 1964). Rainfall fluctuates between 25 and 38 cm across most of the regions eventually occupied by western juniper (Gedney et al., 1999) and most of that precipitation occurs during the winter and spring (October through June) (Miller et al., 2005). Nevertheless, Western juniper is able to grow in areas receiving as little as 18 cm or exceeding 50 cm of rainfall annually (Miller et al., 2005). In most soils, western juniper exhibits a large extension of its lateral root system (Mollnau et al., 2014). The main lateral roots grow to radii that are at least equivalent to the height of a tree and extend beyond the dripline determined by the extension of the canopy (Young et al., 1984) but with large restriction of roots to surface soils (Miller et al., 2005). Western juniper roots are capable of infiltrating fractured basalt bedrock that allows it to occupy rock outcrops and soils less than 30 cm deep (Miller et al., 2005).

## 2. Causes of Western Juniper Encroachment

Factors such as fire suppression, livestock grazing, climate change, and increases in atmospheric $CO_2$ have caused the large encroachment of western juniper (Miller and Tausch, 2001).

### 2.1 Climate

During the early compared to last half of the 20th century, climatic conditions were relatively milder and wetter (Miller et al., 2008). This shift in climate beginning in the 1960s was noticeable in many regions of the world (Kerr, 2007). Mild wet climatic conditions enhanced juniper seedling establishment (Fritts and Wu, 1986). In Idaho for example, the early expansion of western juniper may be caused by a shift to milder temperatures and wetter conditions following the end of the little Ice age (Graumlich, 1987). That is in consistent with (Wigand, 1987) who reported that prior to the current encroachment, western juniper encroached only during wet periods. Western juniper growth exhibits great sensitivity to precipitation variability, but significantly less sensitivity to temperature variability (Grissino-Mayer, 1993; Grissino-Mayer et al., 1997).

### 2.2 Livestock Grazing

Domestic livestock likely stimulates western juniper by two ways (Soulé et al., 2004). First, grazing leads to a considerable decline in the amount of fine fuels (herbaceous biomass) needed to carry and sustain wildfires (Burkhardt and Tisdale, 1976; Miller and Rose, 1995, 1999). Fire-frequency intervals of 30-40 years would be adequate to keep western juniper from invading a sagebrush-grassland community (Burkhardt and Tisdale, 1976). Fire is believed to be the most

important factor limiting juniper expansion into shrub-grassland communities (West, 1999). A large decline in fire occurrence in Southeastern Oregon after large introduction of livestock in the late 1860'S (Miller and Rose, 1999) coincided with the initial expansion of western juniper (Miller et al., 2005) (Fig. 1.3). These findings are in agreement with those reported from ponderosa pine forests in the Pacific Northwest (Heyerdahl et al. 2001) and Southwest (Swetnam and Betancourt, 1998), where the occurrence of fire substantially declined between 1874 and 1900.
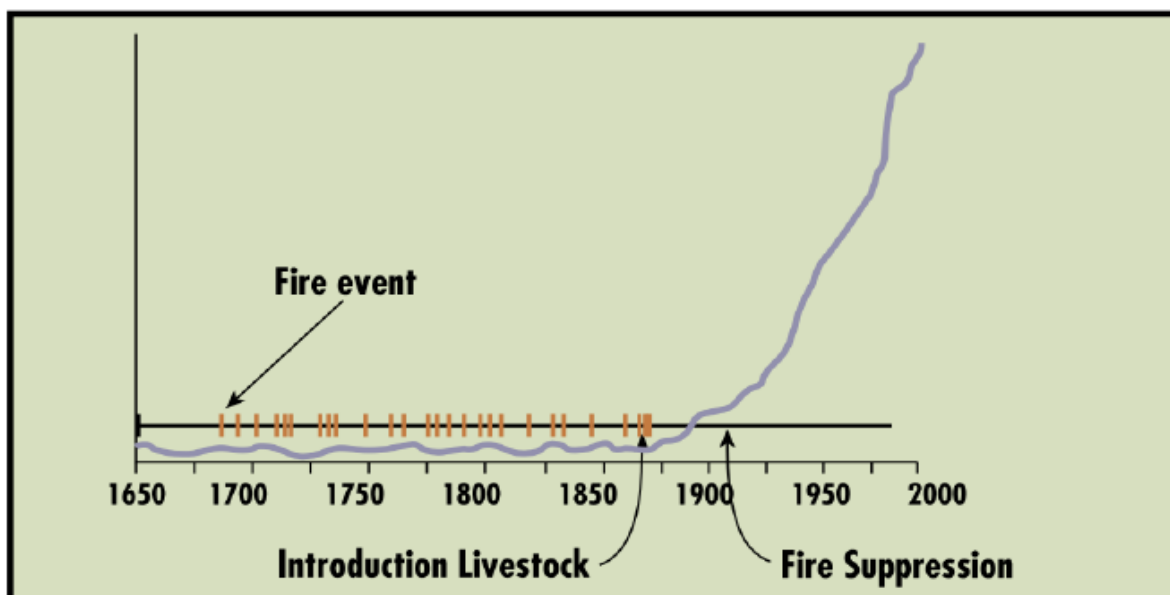


Figure 1.3. Encroachment of western juniper in response to fire suppression (Miller et al., 2005).

Second, grazing usually results in an increase in the woody species that serve as host or nurse-plant sites for seedlings (Evans, 1988). The increase in big sagebrush cover and density attributed to the lack of fire and decreased competition from grasses (Miller and Rose, 1995) may be related to juniper establishment (Knapp and Soule´, 1999). Increases in the big sagebrush created more favorable sites for juniper germination and establishment (Knapp and Soule´, 1999). Western juniper established typically beneath the canopy of big sagebrush plants (Burkhardt and

Tisdale, 1976; Eddleman, 1987; Knapp and Soulé, 1999). Also, it was indicated that big sagebrush acted as a nurse plant for juniper seedlings where the seedlings benefited from the microclimatic conditions (reduced temperature variability, reduced transpiration rates) and elevated nutrient status near the shrub base (Knapp and Soulé, 1999).

### 2.3 Rising Atmospheric $CO_2$

Greater atmospheric $CO_2$ levels benefited the seedlings of arid and semiarid woody species significantly (Polley et al., 1999; Hamerlynck et al., 2000; Maherali and DeLucia, 2000) due to increases in photosynthetic rates and/or greater drought tolerance. Also, higher atmospheric $CO_2$ may prompt juvenile trees to grow more quickly, therefore spending fewer years in the "topkill zone" where they are more vulnerable to periodic surface fires (Bond and Midgle, 2000). Increasing levels of $CO_2$ was the main cause of radial growth (given climate) of western juniper during the latter half of the 20th century because of the ameliorative impacts of increased water-use efficiency (Knapp et al., 2001a, b). With more atmospheric $CO_2$ available, radial growth of adult juniper may happen longer into the growing season and be less effected during drought conditions (Soulé et al., 2004). As a result, western juniper growth post-1950 increased significantly (mean increase of 23%) compared to the pre-1950 period, and the relative growth increases were especially pronounced during drought years (63% increases) (Knapp et al., 2001a). Elevated atmospheric $CO_2$ did not coincide with the initial peak periods of western juniper. Thus, elevated atmospheric $CO_2$ during the last half of the 20th century is a significant contributing factor in accelerating the tree canopy expansion and establishment (Soulé et al., 2004).

To summarize the factors contributing to its encroachment in North America, western juniper establishment during the past 130 years occurred under two different scenarios (Soulé et

al., 2004). The initial tree establishment phase was attributed to the combined impacts of favorable climatic conditions that coincided with decreased fire frequencies caused by extensive livestock grazing (Soulé et al., 2004). The second establishment phase, especially post-1950s, seems to be caused by mechanisms including atmospheric $CO_2$, increasing annual temperature, and increasing summer precipitation (Soulé et al., 2004). Therefore, climate change and increases in atmospheric $CO_2$ have been suggested as contributing to juniper spread.

Climate change is projected to influence the future expansion of western juniper. Under three scenarios of climate change, extended shifts were projected in vegetation types over the 21st century, with declining sagebrush steppe and expanding salt desert shrub probably by the end of the century in central Oregon (Creutzburg et al., 2015a). This rapid shift in vegetation types was caused by many extreme fire years that occurred under climate change scenarios (Creutzburg et al., 2015a, b). Increasing wildfire under climate change projections led to expansion of exotic grasses and reduced juniper encroachment relative to projections without climate change (Creutzburg et al., 2015a). That is consistent with (Gibson et al., 2011) who identified central and eastern Oregon as areas with great potential for juniper reduction under projected climate change. As a result, a combination of increasing climatic stress combined with increasing wildfire frequency may slow juniper expansion rates in the coming decades (Creutzburg et al., 2015a).

## 3. Ecological Repercussions of Western Juniper Encroachment

Western juniper encroachment has become problematic. In central Oregon an increase in bare ground and smaller, more widely spaced grass clumps on comparatively shallow depths (Roberts and Jones, 2000) and a great reduction in ground cover (Knapp and Soulé, 1998) were observed due to western juniper encroachment. Similar results were observed in southwestern

Idaho where herbaceous cover decreased in the mountain big sagebrush as western juniper encroached (Bunting et al., 1999).

In many junipers dominated area, the level of bare ground connectivity increases in tree interspace zones causing greater potential soil loss from increased overland water flow and erosion (Miller et al., 2005). As western juniper encroaches, soil erosion increases and consequently the potential offsite loss of nutrients via sediment would finally lead to a decline in community productivity (Miller et al., 2005). Thus, western juniper encroachment is concerning because as tree cover increases, sagebrush is lost, forage production and diversity decline, and runoff and erosion potential increase (Miller et al., 2000; Bates et al., 2005; Pierson et al., 2007; Davies et al., 2014).

Precipitation interception by western juniper encroachment through leaf interception minimizes the effective precipitation over areas encroached by these woodlands (Miller et al., 2005). The chance of rain to be intercepted is higher in a western juniper community than in a shrub-steppe community (Eddleman et al., 1994). For instance, when measured directly under the tree canopy, up to 74% of rainfall was intercepted in central Oregon (Eddleman, 1986; Larsen, 1993). According to Ochoa et al. (2018), juniper woodlands intercepted up to 46% of total precipitation, altering soil moisture distribution under the canopy and in the interspace. Also, with 9.25 inches of precipitation 42% of that amount was intercepted by the juniper canopy (Young et al., 1984). Therefore, the amount of rain reaching the soil surface can be reduced by western juniper canopy. In addition, western juniper can influence infiltration rates, sediment loss, and soil water storage and depletion rates (Miller et al., 2005). Higher runoff, sediment yields, and rill formation in uncontrolled compared to controlled treatments of juniper were reported (Pierson et al., 2003).

Western juniper encroachment into the sagebrush steppe can potentially alter composition, structure, and productivity of understory vegetation (Miller et al., 2005). The understory plants decline at the time western juniper starts to control the plant community (Bates et al., 2000; Miller et al., 2000). Also, soil resources become less available as western juniper encroaches (Bates et al., 2002). Moreover, due to the reduction in forage base and the change in natural habitat, rangelands being encroached might be less capable of supporting livestock and native wildlife (Miller et al., 2005).

Wildlife abundance, diversity, and species richness decrease as consequence of western juniper encroachment (Miller et al., 2005). For example, Baruch-Mordo et al. (2013) and Dittel et al. (2018) reported that the reduction in sagebrush in response to western juniper encroachment was harmful to sagebrush-associated wildlife, especially sagebrush-obligate wildlife species. In central Oregon, sage grouse (*Centrocercus urophasianus*) avoided western juniper communities for nesting and winter use (BLM, 1994). As western juniper densities increase, sage grouse habitat declines particularly in mountain big sagebrush habitat below 7000 feet (Miller et al., 2005).

Western juniper dominance leads to the decline of mule deer populations (Miller et al., 2005). The digestibility and levels of available proteins are low in western juniper (Miller et al., 2005). Increased western juniper - dominated areas across the landscape cause browse resources to decline (Miller et al., 2000; Schaefer et al., 2003). So, in northeastern California the reduction of mule deer populations in the late 1960s may be partially related to the concurrent expansion of western juniper and the decline in shrubs (Schaefer et al., 2003). Additionally, the great reduction in mule deer populations in southwestern Idaho in late 1950s and 1960s coincided with juniper transition from phase II to III, leading to the rapid reduction in shrub cover (Miller et al., 2005). In brief, western juniper has significantly increased in density and distribution in North America

since the late 1800s and if left unremoved can have considerable effect on soil resources, plant community structure and composition, water and nutrient cycling, and wildlife habitat (Miller et al, 2005). Consequently, the elimination of western juniper has been a main concern of land management since the early 1960's (Miller et al., 2005).

## 4. Removal Benefits of Western Juniper Woodlands

Western juniper removal was reported to increase grass and shrub presence (Eddleman, 2002; Coultrap et al., 2008; Bates et al., 2017; Dittel et al., 2018; Ray et al., 2019) with immediate favorable results for livestock (Miller et al., 2005; Bates et al., 2005). Soon after western juniper was eliminated, productivity of forage plants and forage quality increased in some areas (Young et al., 1985; Vaitkus and Eddleman, 1987; Bates et al., 2000). For example, crude protein levels of forage plants used by livestock and wildlife were 50% larger in cutting compared to uncutting treatments of western juniper woodlands (Bates et al., 2000). So, western juniper removal does improve forage productivity. In addition, western juniper elimination improved wildlife habitat (Miller et al., 2005; Baruch-Mordo et al., 2013). Greater capture rates of small mammals were reported in cutting compared to uncutting treatments of western juniper woodlands (Willis and Miller, 1999).

Western juniper removal led to an increase in soil water content due to an increase in soil water recharge and reduction in transpiration and interception rates (Bates et al., 2000; Deboodt, 2008; Mollnau et al., 2014). Therefore, western juniper cutting in encroached systems can lead to a rapid increase in herbaceous production and cover (Bates et al., 2000; Bates et al., 2005), increasing available forage for grazing plants and improving site processes including nutrient and hydrologic cycling (Bates et al., 2002).

**5. Research Justification**

This research, in fulfillment of a Doctor of Philosophy Degree extends the Camp Creek-Paired Watershed Study (CCPWS) in central Oregon, which was initiated in 1993 as a long-term collaborative research project aimed at quantifying the effects of large-scale western juniper elimination on biological and physical characteristics of semiarid watersheds. Phase I of the project (1993 – 2004) completed pre-treatment data collection. During this time, monitoring parameters such as the vegetation composition, soils and soil erosion, channel morphology, terrain indices, geology, streamflow, and climate were described (Fisher, 2004). Phase II (2004 – Present) was the juniper treatment phase involving measurements of hydrological processes such as streamflow, spring flow, groundwater, and soil water content; physical features such as hillslope and channel morphology, and; biological components such as change in plant community and species composition. This dissertation details an additional research contribution which was aimed at investigating and elaborating on several of the key findings resulting from Phase I and II efforts.

Considerable work has been done evaluating the ecological repercussions and removal benefits of western juniper as well as a clear perception by ranchers that juniper encroachment represents a serious threat (Johnson et al., 2011). As a result, western juniper removal is a common and accepted rangeland management practice (Campbell et al., 2012). However, little is known about the implications of juniper encroachment and removal on ecosystem carbon and nitrogen storage. In addition, it is unknown if the hydrologic system (transpiration in relation to soil moisture content) is altered by with or without juniper. Understanding the influence of western juniper removal on these measured variables; carbon, nitrogen, transpiration and soil moisture may prove valuable for planning and maintaining juniper cutting programs. This research was unique because it involved a paired study approach to monitoring changes in these variables post western

juniper cutting. The merit of a paired watershed study is that the impacts of the treatment can be compared to the untreated watershed (Deboodt, 2008).

## 6. Research Objectives

The overreaching goal of this research is to provide a documented analysis using paired watersheds to determine the effects of western juniper encroachment and removal on ecosystem processes. Within this goal are the following objectives:

(1) Determine ecosystem carbon stocks in an encroached juniper watershed and an adjacent watershed where juniper removal occurred 13 years prior to determinations.

(2) Quantify and compare major pools of nitrogen in an encroached juniper watershed and an adjacent watershed where juniper removal took place 13 years prior to quantification.

(3) Determine seasonal transpiration for western juniper in relation to soil moisture in one juniper-dominated watershed and in another watershed where juniper has been removed, with a major goal of improving scientific understanding of the effects of juniper encroachment and removal on hydrology.

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# ECOSYSTEM CARBON IN RELATION TO WOODY PLANT ENCROACHMENT AND CONTROL: JUNIPER SYSTEMS IN OREGON, USA

**Abstract**

The encroachment of western juniper (*Juniperus occidentalis*) trees represents a substantial problem in Oregon rangelands because of the displacement of understory vegetation of importance to wildlife and livestock. Therefore, the control of this species is a common ecological restoration practice. However, western juniper control may also affect the carbon sequestration capacity for an area, although this effect is not well understood. Our study site was a paired watershed in central Oregon where western juniper trees were cut in one watershed (treated, 116 ha) and were left intact in another (untreated, 96 ha). Thirteen years after control, we quantified aboveground carbon stocks for western juniper trees, shrubs, grasses, and litter in both the treated and untreated watersheds. We also quantified belowground carbon stocks (roots and soil) in both watersheds at two soil depths (0-25cm and 25-50cm). Aboveground carbon stocks were 5.8 times greater in the untreated than in the treated watershed. On the other hand, root carbon stocks were 2.6 times greater in the treated than in the untreated watershed. Soil carbon stocks at both 0-25 cm and 25-50 cm depth were not affected by juniper control. Overall, total ecosystem carbon stocks (average 137.6 Mg C ha$^{-1}$) were not different between watersheds. Most carbon resided belowground (soil 0-50 cm and roots); 84% and 97% of the total ecosystem carbon, respectively, was found in the untreated and treated watershed. Juniper control represents benefits such as habitat restoration for native wildlife, increased forage for livestock, and restoration of hydrological functions. Our study provides basis to suggest that the benefits of juniper control can be attained without substantially affecting the potential for ecosystem carbon sequestration.

**1. Introduction**

Woody plant encroachment has been documented worldwide over the past 150 years in many ecosystems (Archer et al., 2017). The expansion of woody plants into grasslands and shrublands has important implications for wildlife habitat, fire regimes, forage and livestock production, hydrology and soil erosion, and biodiversity (Archer, 2010; Baruch-Mordo et al., 2013; Eldridge et al., 2011; Ochoa et al., 2018). Woody plant encroachment may also have an impact on carbon pools worldwide by modifying aboveground and belowground net primary productivity and modifying rooting depth, biomass and distribution (Hughes et al., 2006; Boutton et al., 2009). Woodland encroachment commonly results in aboveground carbon stock increases (Barger et al., 2011; Shackleton and Scholes, 2011; Fernandez et al., 2013), but increases in total ecosystem carbon stocks have also been reported (Daryanto et al., 2013; González-Roglich et al., 2014; Pellegrini et al., 2014). González-Roglich et al. (2014) found that an ecosystem encroached by the woody plant *Prosopis caldenia* produced three times greater total ecosystem carbon than an herbaceous-dominated ecosystem. However, thicket encroachment into South African grasslands did not represent significant gains in total ecosystem carbon pools (Coetsee et al., 2013).

Juniper (*Juniperus* spp) encroachment is one of the most large-scale changes that are occurring in North American rangelands (Baker and Shinneman, 2004; Sankey et al., 2010). The spatial distribution of juniper has increased between 30% and 625% since the mid-19th century throughout the Great Basin (Romme et al., 2009; Sankey et al., 2010) and encroachment rates have varied between 1.5% and 2% per year (Sankey and Germino, 2008; Sankey et al., 2010). Western juniper (*Juniperus occidentalis* spp. *occidentalis* Hook.) is an encroaching species into the semiarid shrub-steppes of the western United States that occupies 3.6 million hectares in central and eastern Oregon, northeastern California, southwestern Idaho and northwestern Nevada

(Azuma et al., 2005). These woodlands have expanded significantly over the last 130 years due to a combination of factors including changes in climate, increases in the atmospheric carbon dioxide, introduction of livestock, and reduction of fire occurrences (Soulé et al., 2004; Miller et al., 2005). Documented implications of western juniper encroachment into shrublands and grasslands include a reduced forage base for livestock and habitat deterioration for wildlife species of concern such as the greater sage grouse (*Centrocercus urophasianus*) (Baruch-Mordo et al., 2013; Dittel et al., 2018). Elimination of juniper has resulted in an increase of soil moisture and streamflow compared to encroached areas (Ochoa et al., 2018; Ray et al., 2018).

There is a relative abundance of studies evaluating the ecological repercussions and control benefits of western juniper as well as a clear perception by ranchers that juniper encroachment represents a serious threat (Johnson et al., 2011). As a result, juniper control is a common rangeland management practice (Campbell et al., 2012). However, little is known about the implications of juniper encroachment and control on ecosystem carbon pools. Existing work indicates that western juniper encroachment increases aboveground carbon stocks with respect to non-encroached conditions (Campbell et al., 2012; Throop and Lajtha, 2018). Western juniper encroachment into a sagebrush community increased carbon stocks (aboveground, roots, litter and soil carbon at 0-10 cm soil depth) from 13.5 to 30.2 Mg C $h^{-1}$, but understory vegetation, such as grasses and shrubs, were not included in total carbon stocks calculation (Throop and Lajtha, 2018). Several studies indicate an increase in surface soil carbon (up to 10 cm depth) associated with western juniper encroachment (Bates et al., 2002; Miwa and Reuter, 2010; Throop and Lajtha, 2018). However, Rau et al. (2011) reported no gains in soil carbon resulting from woody plant (including western juniper) encroachment in the Great Basin of North America. Except for Throop and Lajtha (2018), information on carbon stocks as affected by western juniper control does not exist.

Our study site involved paired watersheds in central Oregon. In one of the watersheds (the treated watershed) western juniper trees were eliminated 13 years prior to sampling whereas in the other watershed the western juniper trees have been left intact (the untreated watershed). Following western juniper control on the treated watershed, changes in vegetation composition have been reported, including western juniper regrowth and greater presence of shrubs and grasses (Ray et al., 2019). Because of the vegetation changes following western juniper control, the potential impact of this management practice on carbon accumulation is difficult to forecast. Evaluations of ecosystem carbon stocks require an understanding of how both aboveground stocks (including understory vegetation) and belowground stocks respond to the presence and control of western juniper trees. Carbon stock evaluations as affected by rangeland management practices are essential in ecological studies, given the significance of promoting carbon sequestration and the extent of rangelands worldwide (Barger et al., 2011; Bikila et al., 2016; Archer et al. 2017). The objective of this study was to determine ecosystem carbon stocks in an encroached juniper watershed and an adjacent watershed where juniper control occurred 13 years prior to determinations. We hypothesized that the treated watershed, after 13 years of mature western juniper control, would store less aboveground and belowground carbon than the untreated watershed, implying lower capacity for soil carbon sequestration resulting from juniper control.

## 2. Materials and methods

### 2.1. Study area

This study was conducted at the Camp Creek Paired Watershed Study site (lat 43.96 N, long 120.34 W) in Crook County, central Oregon, USA (Fig. 2.1). The study site comprises an area of approximately 212 ha and includes two adjacent watersheds, one treated (116 ha) and the

other untreated (96 ha) with elevations ranging from 1370 m to 1524 m (Ochoa et al. 2018). In the treated watershed, approximately 90% of the western juniper trees were cut in 2005 using chain saws, leaving only old-growth trees intact (Ray et al., 2019). The felled trees and debris that resulted from juniper cutting were scattered and left on the ground. The average slope for each watershed is around 25% with similar distribution of aspects (Fisher, 2004). Prior to juniper elimination from the treated watershed, juniper occupied 27% cover in the whole area (Ray et al., 2019), which is near the 30% cover described for Phase (III) juniper sites (Miller et al., 2005). In addition, prior to juniper elimination there were no statistically significant differences in vegetation cover (including juniper, shrubs, and grasses) between the treated and untreated watersheds (Fisher, 2004). After elimination, juniper cover in the treated watershed was 1% (Ray et al., 2019).

The average annual precipitation of the study site is 358 mm. The study area comprises mostly three major soil series; Westbutte, Madeline, and Simas; Westbutte very stony loam and Madeline loam, the two major soil types, were found to comprise approximately 70% to 74% of the study area (Fisher, 2004). Simas, gravelly silt loam accounts for the final portion with additional soil series occupying <1%. The Westbutte series is classified as loamy-skeletal, mixed, superactive, frigid Pachic Haploxerolls. The Madeline series is classified as clayey, smectitic, frigid Aridic Lithic Argixerolls. The Simas series is classified as fine, smectitic, mesic Vertic Palexerolls. The untreated watershed is primarily composed of 48% Madeline, 26% Westbutte, and 21% Simas series while the treated watershed is composed of 50% Westbutte, 20% Madeline, and 3% Simas series (Fisher, 2004).

The most common tree in the area is western juniper (*Juniperus occidentalis*). The most common shrub species are mountain big sagebrush (*Artemisia tridentata, spp vaseyana*), antelope bitterbrush (*Purshia tridentata*), rubber rabbitbrush (*Ericameria nauseosa*), and green rabbitbrush

(*Chrysothamnus viscidiflorus*). For the purposes of this study, western juniper is considered a tree, not to be confused with shrubs in any context or determination. The most common perennial grasses of the area are Idaho fescue (*Festuca idahoensis*), bluebunch wheatgrass (*Pseudoroegneria spicata*), Sandberg bluegrass (*Poa secunda*), prairie junegrass (*Koeleria macrantha*), and Thurber's needlegrass (*Achnatherum thurberianum*). Forbs are not common, representing <3% of plant cover in the area (Ray et al., 2019).

## 2.2. Field sampling layout

The treatments of study were: 1) untreated = no manipulation of vegetation, i.e., western juniper trees, shrubs, and grasses were left intact in the untreated watershed and 2) treated = western juniper trees were cut in 2005 (shrubs and grasses were left intact). The trees were manually cut with chainsaws to ground level and the resulting debris was scattered and left on the ground. Cattle grazing has occurred in both watersheds before and after juniper elimination in the treated watershed. In each watershed (treated and untreated), 20 plots (replications) of 20 m X 20 m were established (Fig. 2.2). The 20 plots were systematically randomized in a 4 X 5 grid (sensu Keith, 2017) trying to represent most of the terrain within each watershed (Fig. 2.1). The samples were first located on a digital map and then the plots were found on the terrain with the help of a GPS unit. The sampling was random because we did not control the specific areas where the plots were located. The distance between plots within the predefined grid was 130 m among columns and 180 m among rows. The 20 m X 20 m plots were used to sample western juniper trees. Then, a 10 m X 10 m plot within each 20 m X 20 m plot was established for estimating shrub biomass (Fig. 2). In addition, four plots of 2 m X 2 m within each 20 m X 20 m plot were established for grass and litter biomass evaluations.
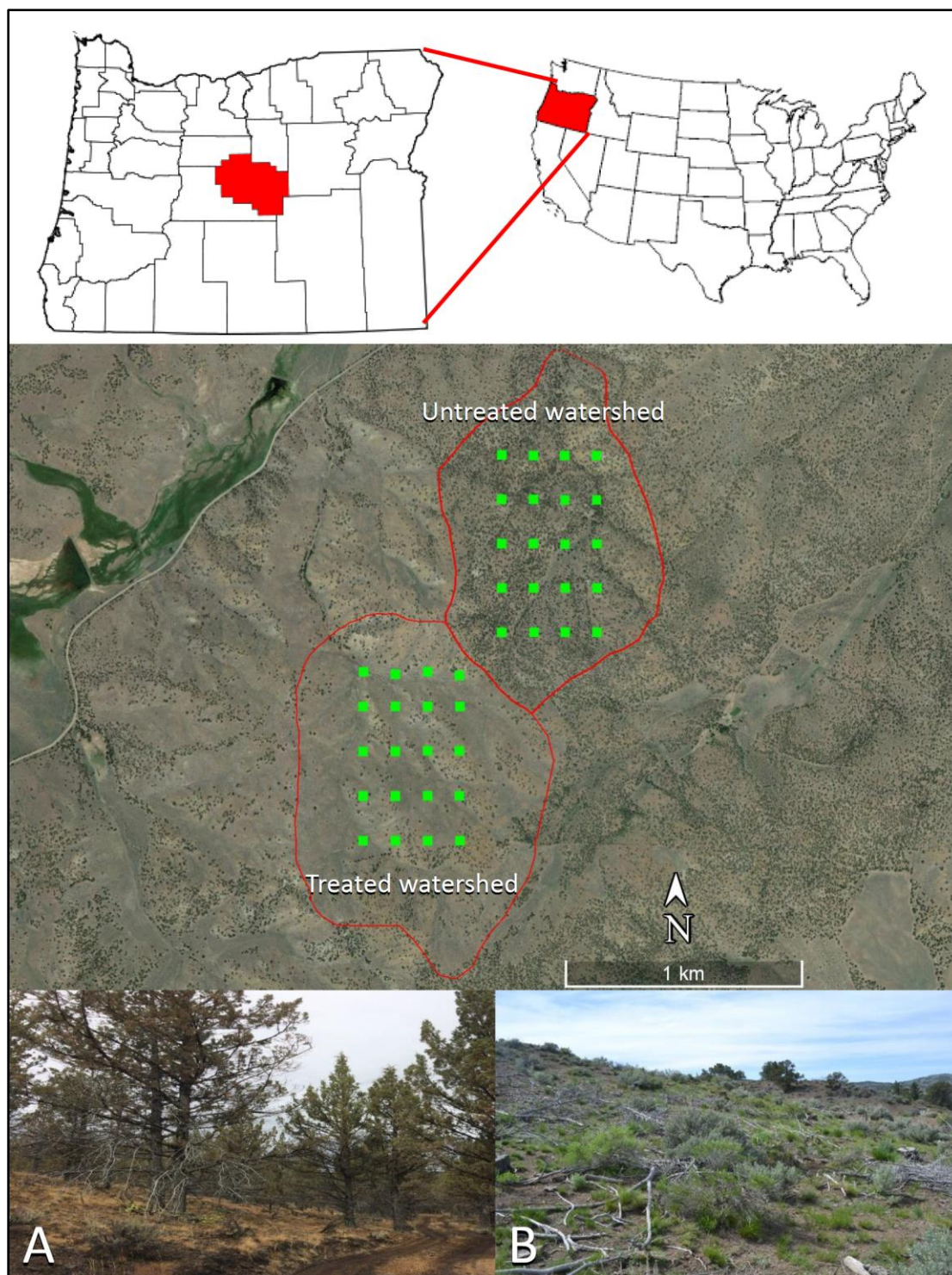
Figure 2.1. Map of the study area showing untreated and treated watersheds, indicating locations of systematically randomized plots used in this study. Photograph A shows intact western juniper (*Juniperus occidentalis*) trees in the untreated watershed, while photograph B shows cut western juniper trees in the treated watershed.
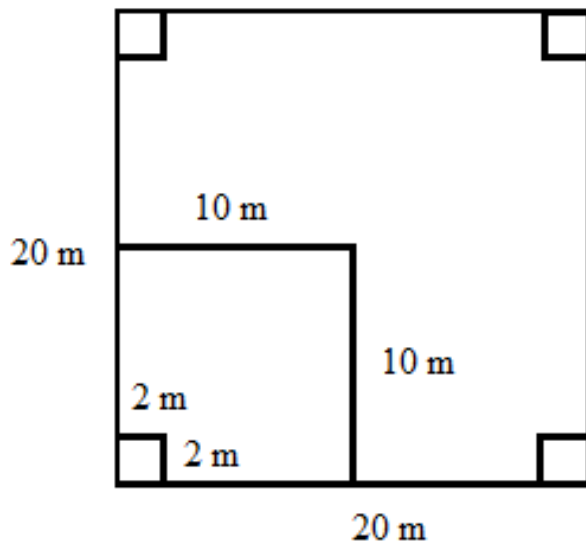
Figure 2.2. Plots used in the study for biomass evaluations. The 20 m X 20 m plots were used for western juniper (*Juniperus occidentalis*) sampling, the 10 m X 10 m plots were used for shrub sampling, and the 2 m X 2 m plots were used for grass and litter sampling.

*2.3. Data collection*

We followed the Intergovernmental Panel on Climate Change (IPCC) guidelines for developing greenhouse gas (GHG) emission inventories for forestlands. We estimated carbon content in western juniper and understory biomass using a component of the tiered approach to develop carbon inventories suggested by the IPCC: Tier 3 estimates carbon biomass taking into consideration management practices and long-term effects (Eggleston et al., 2006).

*2.4. Aboveground biomass and litter sampling*

Aboveground biomass for western juniper trees was estimated in each 20 m X 20 m plot of the untreated watershed using allometric equations developed by Sabin (2008) for western juniper trees of comparable dimensions in eastern Oregon. Based on the best-reported fitting equation, canopy diameter was recorded in two opposite directions for all trees within each plot

(Sabin, 2008). Then, canopy area was calculated using the average diameter for each tree. Subsequently, aboveground biomass (kg) was estimated using the equation $y = 9.7164x + 37.506$, where x is the tree canopy area ($m^2$). The allometric equation that we used has been previously used by Campbell et al. (2012) to estimate aboveground biomass in other areas of Oregon.

Aboveground biomass for regrowth western juniper trees in the treated watershed was obtained by counting the number of individuals within a given plot, cutting and collecting a representative individual from within a given plot, obtaining its dry weight at the lab, and extrapolating biomass weight by area. When mature trees were found inside the treated watershed plots (since 10% of the mature juniper trees were left intact), they were also included in the carbon quantification. The biomass of those trees was calculated as in the untreated plots. The shrub biomass (in both watersheds) was obtained in a similar manner as regrowth juniper trees, except that for shrubs one representative individual was collected for each shrub species found within the sampling plot. For grasses, eighty 2 m X 2 m plots were established in each watershed. Grass aboveground biomass was estimated in both watersheds by harvesting all live standing tissue for dry matter analysis. Non-grass herbs were not common, but when they were present, we sampled them as we did grasses and their biomass was lumped into that of grasses. Litter was sampled from the same plots that were used for grass sampling. All dead lying tree, shrub, and grass materials were considered litter. Because all the litter was dry and detached from the soil, we picked it all to obtain its weight it and returned it to its original place.

*2.5. Belowground biomass sampling*

Root biomass was estimated by the trench method, as in Komiyama et al. (1987). In the untreated watershed, root biomass was estimated for random stands of mature western juniper

trees, shrubs, and grasses, while in the treated watershed, it was estimated for random stands of regrowth juniper, remaining tree stumps, shrubs, and grasses. Three trenches for each plant type in each watershed were dug to a depth of 50 cm using an excavator (Bobcat Inc. West Fargo, North Dakota, USA). The trench width was 61 cm while the trench length was about 3 m (the exact length for each trench was measured and recorded). Twenty-one trenches (9 in the untreated watershed and 12 in the treated watershed) were dug in total. The trenches were dug first for the top 25 cm and then for the subsequent 25 cm, maintaining the soil from each depth separated. After obtaining the substrate from each trench and soil depth, the roots were carefully separated from the soil using a sieve prior to subsequent rinsing.

## 2.6. Plant dry matter and carbon concentration determinations

Dry matter of aboveground and belowground biomass was obtained by placing biomass samples in an oven at 60°C until constant weight. The carbon concentrations for aboveground biomass were determined in five samples for western juniper and the main shrub species found in the area. Five samples were considered enough because carbon concentration variation is considered low in relation to that of other nutrients (Martin et al., 2015). Evidence that carbon concentration variation is low and that our sampling protocol was adequate was the low variation obtained in all the plant carbon measurements (Table 2.1). Carbon concentrations for root samples were determined for western juniper and the shrub species without distinguishing shrubs species (it was difficult to separate shrub roots by species). Likewise, carbon concentrations were also determined for samples of grasses and litter without distinguishing species. These determinations were conducted at the Central Analytical Laboratory of the Crop and Soil Science Department at

Oregon State University (Corvallis, OR) using a CNS automatic analyzer (Elementar Vario MMARCO CNS, Elementar Analysen Systeme GmbH, Hanau, Germany).

*2.7. Soil sampling*

Soil samples were collected from all systematically randomized plots (20 plots in each watershed) (Fig. 2.1) using a cylindrical soil step probe of 1.6 cm inner diameter (AMS, Inc; American Falls, Idaho, USA). Soil samples were collected from under the canopy of mature western juniper trees in the untreated watershed and near stumps of cut western juniper trees (areas that would have been under canopy prior to tree cutting) in the treated watershed. Our sampling was done from areas 50 cm to the trunk or stump but we did not follow any specific direction within the canopy because a previous study found no differences in soil organic matter in relation to soil sample orientation under tree canopies (Rossi and Villagra, 2003). Soil samples were also obtained from interspaces of both watersheds. Thus, there were two sampling contexts in each plot, under-canopy/near stumps and interspace. Two soil samples were collected from each sampling context of each plot at two soil depths (0-25 cm and 25-50 cm). In total, eight soil samples were collected for each plot (2 samples x 2 conditions x 2 depths). For the whole study, we analyzed 320 soil samples for carbon at the Central Analytical Laboratory in the Crop and Soil Science Department at Oregon State University, Corvallis, OR, USA. The soil samples were sieved (2 mm mesh) to avoid gravel or rocks, oven-dried at 40 °C for 48 h, weighed, and analyzed using a CNS automatic analyzer (Elementar Vario MMARCO CNS, Elementar Analysen Systeme GmbH, Hanau, Germany). Although we analyzed total soil carbon, the soils of the study area contain very little to no inorganic carbon, especially in the upper layers (Soil Survey Staff, 2019), therefore, our analysis reflect soil organic carbon.

*2.8. Soil bulk density and carbon calculation*

Soil cores for determining bulk density were obtained from four plots selected from the middle of the 20 plots in each watershed. A soil core sampler was used to collect core samples (5 cm diameter x 7.5 cm length) in under-canopy zones of mature western juniper trees in the untreated watershed, near-stump zones of cut western juniper trees in the treated watershed, and in interspace zones of both watersheds. We tried to avoid rocky areas for the bulk density sampling because those were not representative of the whole area. Because of this, the samples that were obtained did not contain large pebbles and our samples were not sieved. One core sample was obtained in each zone of each plot at two soil depths (0-25 cm and 25-50 cm). Soil core samples were oven-dried at 105°C for 48 h and weighed. Bulk density was calculated as the ratio of the mass of oven-dried soil sample to core volume (g cm$^{-3}$). The soil carbon mass per area (Mg C ha$^{-1}$) was computed by the following formula: Soil C (Mg ha$^{-1}$) = BD (g cm$^{-3}$) x C % x d (cm). Where: d = soil depth (cm), BD = bulk density in g cm$^{-3}$, and C % = percentage carbon content of the sample.

In determining soil carbon stocks per plot, we adjusted for the areas under canopy cover (or near tree stumps) and interspace because the soil carbon concentrations were demonstrably different between those areas. The cover of under-canopy and interspace zones were determined for each 20 m X 20 m plot of the untreated watershed. Canopy cover was calculated based on individual tree canopy area estimates in each plot using the canopy diameter measurements. The sum of all tree canopy areas was divided by the total plot area to determine plot-scale canopy cover. The interspace cover at plot scale was estimated by subtracting tree canopy area from the total plot area. We considered that the under-canopy cover in the treated watershed prior to tree elimination was similar to that of the untreated watershed because this was previously evaluated

for our study area (Fisher, 2004). Mean soil carbon concentration (Mg C ha$^{-1}$) for each plot zone-cover (under-canopy/near-stump and interspace zones for both watersheds) was multiplied by the total areas of the relevant plot zone-cover in order to estimate zone-cover specific soil carbon stocks (Edmondson et al., 2014). Finally, the values for both zones were summed for each plot to represent soil carbon stocks per plot.

*2.9. Data analysis*

Single factor analysis of variance with two-sample t-tests (treated vs. untreated) were applied to test for differences in each analyzed variable using the R Statistical Software (R Core Team 2019). The number of sample replications by treatment varied by analyzed variable; for aboveground biomass of juniper and shrubs n=20, for grasses and litter n=80, for root biomass by plant type n=3, for soil carbon n=20. Because of the large-scale (around 100 ha watersheds) nature of our study, we only had one treated and one untreated area. Our comparisons are valid, however, because we properly randomized our sampling, have a high number of replications, and have pre-treatment information (Wester, 1992).

**3. Results**

*3.1. Carbon concentration of biomass samples*

The carbon concentrations that we obtained were used to calculate carbon pools and they were not intended to test for carbon content differences among species or plant parts. Yet, it was evident that grasses had less carbon than woody plants (Table 2.1).

Table 2.1. Mean (±SE) of carbon concentration (%) obtained by laboratory analysis of western juniper (*Juniperus occidentalis*), main shrub species, grasses, and litter of the study area, central Oregon, USA. Root carbon analysis by species was not performed on shrubs.

| Biomass samples | Aboveground carbon % | Root carbon % |
|---|---|---|
| Mature western juniper | 51.2(0.50) | 44.4(1.10) |
| Regrowth western juniper | 51.1(0.44) | 45.2(0.87) |
| Shrubs (average) | 46.2(0.97) | 38.8(1.92) |
| *Artemisia tridentata* | 47.7(0.35) | |
| *Purshia tridentata* | 46.7(0.10) | |
| *Eriogonum fasciculatum* | 43.3(0.71) | |
| *Ericameria nauseosa* | 47.0(0.43) | |
| Grasses | 39.7(0.58) | 37.6(1.61) |
| Western juniper stumps | - | 45.0(1.26) |
| Litter (mainly western juniper debris) | 46.4(0.50) | - |

*For all live species (juniper, shrubs, and grasses), the aboveground carbon analysis was made in leaves and stems or twigs.

### 3.2. Tree density and cover

Tree density was evaluated for the determination of tree carbon stocks. Tree density was greater ($P < 0.05$) in the untreated (327 individuals ha$^{-1}$) than in the treated (210 individuals ha$^{-1}$) watershed (Fig. 2.3). The density values on the treated watershed mainly reflect small regrowth

trees that resulted after juniper control. In the calculation of soil carbon stocks at the watershed scale, juniper cover and interspace cover in the untreated watershed were evaluated. Interspace cover was 68.5% while western juniper canopy cover was 31.5%, similar to that reported by Ray et al. (2019).
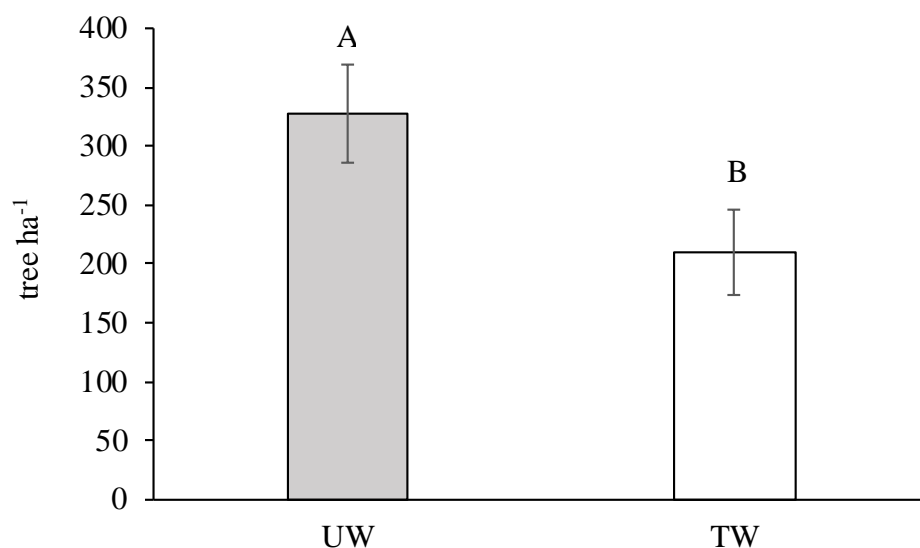


Figure 2.3. Tree density (tree ha$^{-1}$) across management practices (UW= untreated watershed, mature trees; TW= treated watershed, regrowth trees after 13 years of control). The data are means (± standard error) by management practice based on 20 sampling plots.

### 3.3. Aboveground carbon stocks

Aboveground tree carbon stocks were greater ($P < 0.05$) in the untreated watershed than in the treated watershed (Table 2.2). Mature western juniper trees contained 21 times more carbon than regrowth trees. Similarly, grass carbon stocks were 50% greater ($P < 0.05$) in the untreated watershed than in the treated watershed. In contrast, the shrub and litter carbon stocks were greater ($P < 0.05$) in the treated watershed than in the untreated watershed. Shrubs and litter had 8.1 and 6.2 times more carbon in the treated than the untreated watershed, respectively. Sagebrush was the main shrub species in both watersheds.

Table 2.2 Mean (±SE) aboveground carbon stocks (Mg C ha$^{-1}$) for plant groups (western juniper [*Juniperus occidentalis*] trees, shrubs, grasses) and litter by management practice in central Oregon, USA. The management practices are 1) Untreated (western juniper intact) and 2) Treated (western juniper cut). The trees in the Treated management watershed are regrowth western juniper after 13 years of juniper removal.

| Management Practice | Trees | Shrubs | Grasses | Litter |
|---|---|---|---|---|
| Untreated | 21.98(2.77)[a] | 0.10(0.02)[b] | 0.06(0.006)[a] | 0.32(0.05)[b] |
| Treated | 1.05(0.71)[b] | 0.81(0.12)[a] | 0.04(0.004)[b] | 1.98(0.24)[a] |
| *P* value | <0.05 | <0.05 | <0.05 | <0.05 |

Different lowercase letters (a, b) along columns indicate significant differences between management practices for a given plant group or litter ($P < 0.05$).

### 3.4. Belowground carbon stocks

Root carbon stocks (0-50 cm soil depth) for mature trees (intact vs. stumps) and shrubs were not significantly different ($P > 0.05$) between treated and untreated watersheds (Table 2.3). It was noticeable that root carbon stock for regrowth juniper on the treated watershed was only slightly lower (14%) than in mature intact trees. Root carbon for grasses was eight times greater ($P < 0.05$) in the treated than the untreated watershed.

Table 2.3. Mean (±SE) root carbon stocks at 0-50 cm soil depth (Mg C ha$^{-1}$) for plant groups (mature juniper [*Juniperus occidentalis*] trees, regrowth juniper trees, shrubs, and grasses) by management practice in central Oregon, USA. The management practices are 1) Untreated (western juniper intact) and 2) Treated (western juniper cut). The comparison of mature trees included root determination near existing juniper trees (untreated watershed) and near juniper tree stumps (treated watershed).

| Management Practice | Mature trees | Regrowth trees | Shrubs | Grasses |
|---|---|---|---|---|
| Untreated | 1.50(0.09)[a] | - | 1.49(0.10)[a] | 0.58(0.10)[b] |
| Treated | 1.92(0.24)[a] | 1.29(0.20) | 1.26(0.09)[a] | 4.76(0.84)[a] |
| *P* value | 0.70 ns | | 0.47 ns | <0.05 |

Different lowercase letters (a, b) along columns indicate significant differences between management practices for a given plant group ($P < 0.05$). ns = not significant.

More soil carbon ($P < 0.05$) was stored under mature western juniper tree canopies of the untreated watershed and near western juniper tree stumps of the treated watershed than the corresponding interspaces in each watershed (Fig. 2.4). Under-canopy zones of mature western juniper and near-stump zones of cut western juniper had about twice the soil carbon stocks of the interspaces. Soil carbon stocks were similar under the canopies of mature western juniper and near-stump zones of cut western junipers. Likewise, soil carbon stocks were similar in untreated and treated watershed interspaces.
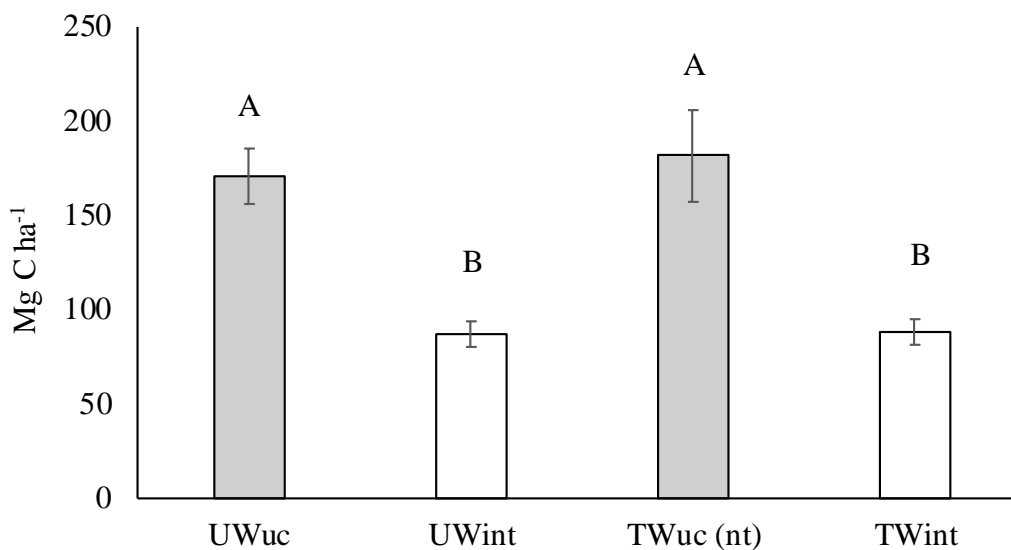
Figure 2.4. Soil carbon stocks at 0-50 cm soil depth (Mg C ha$^{-1}$) by watershed and treatments (UWuc = untreated watershed and under tree canopies; UWint = untreated watershed and interspaces beyond tree canopies; TWuc (nt) = treated watershed and under tree canopies (near the tree stumps); TWint = treated watershed and interspaces beyond the tree stumps. Data are means ± standard error.

### 3.5. Total carbon stocks by management practice

Total aboveground carbon stocks differed by watershed ($P < 0.05$) (Table 2.4). The untreated watershed had 5.8 times more total aboveground carbon than the treated watershed. Root carbon stocks were also significantly different ($P < 0.05$) by watershed. Yet, contrary to aboveground biomass, roots in the treated watershed stored 2.6 times more carbon than those in the untreated watershed.

Table 2.4. Mean (±SE) total aboveground, total belowground carbon stocks (0-50 cm soil depth) and total carbon stocks (Mg C ha$^{-1}$) by management practices in central Oregon, USA. The management practices are 1) Untreated (western juniper [*Juniperus occidentalis*] intact) and 2) Treated (western juniper cut). Root carbon stocks in the untreated watershed represent the sum of root carbon in mature western juniper, shrubs, and grasses, while root carbon stocks in the treated watershed represents the sum of regrowth western juniper, western juniper stumps, shrubs and grasses.

| Management practices | Root carbon | Soil carbon | Total belowground carbon | Total aboveground carbon | Total carbon stocks |
|---|---|---|---|---|---|
| Untreated | 3.57(0.30)[b] | 117.08(9.60)[a] | 120.65(9.60)[a] | 22.46(2.77)[a] | 143.11(11.53)[a] |
| Treated | 9.23(0.83)[a] | 119.06(11.85)[a] | 128.29(11.85)[a] | 3.88(0.66)[b] | 132.17(12.12)[a] |
| *P* value | <0.05 | 0.90 ns | 0.62 ns | <0.05 | 0.52 ns |

Different lowercase letters (a, b) along columns indicate significant differences between management practices for a given variable ($P < 0.05$). ns = not significant.
*Soil carbon stocks were calculated by adjusting the amount of surface under canopy cover (or near tree stumps) and interspace because the soil carbon concentrations were demonstrably different between those areas.

In contrast to the total aboveground carbon stocks, total belowground carbon stocks (root and soil) did not differ ($P >0.05$) by watershed. Moreover, total belowground carbon stocks were 5.4 and 33.1 times greater than their corresponding total aboveground carbon stocks in the untreated and treated watersheds, respectively. Total carbon stocks, including both total belowground and aboveground carbon, were not significantly ($P >0.05$) different between the untreated and treated watersheds.

*3.6. Belowground carbon stocks by soil depth*

Root carbon stocks for western juniper mature trees, stumps, and regrowth trees did not vary significantly ($P >0.05$) by depth (Table 2.5). In contrast, root carbon stocks for shrubs in both watersheds were 3.5 to 10.5 times greater ($P <0.05$) in the top soil layer (0-25 cm depth) than in

the bottom soil layer (25-50 cm depth). In a similar fashion, root carbon stocks for grasses in the treated watershed were 19 times greater ($P$ <0.05) in the top than in the bottom soil layer. In the untreated watershed, grass root carbon did not vary significantly ($P$ >0.05) by depth.

Soil carbon stocks in under-canopy zones were about 70% and 60% greater ($P$ <0.05) in the top than in the bottom soil layer for untreated and treated watershed areas, respectively (Table 5). Soil depth did not produce significant difference ($P$ >0.05) in interspace soil carbon stocks in both watersheds.

Table 2.5. Mean (±SE) belowground carbon stocks (Mg C ha$^{-1}$) from roots of different plant types and from soil from different areas by soil layer depth and management practices in central Oregon, USA. The management practices are 1) Untreated (western juniper [*Juniperus occidentalis*] intact) and 2) Treated (western juniper cut). Regrowth trees were only evaluated in the treated watershed.

| Compartment | | Soil layer depth (cm) | Untreated Watershed | Treated Watershed |
|---|---|---|---|---|
| Root | Mature Juniper Tree* | 0-25 | 0.68(0.11)$^{Aa}$ | 1.36(0.17)$^{Aa}$ |
| | | 25-50 | 0.82(0.02)$^{Aa}$ | 0.56(0.16)$^{Aa}$ |
| | Regrowth Juniper Tree | 0-25 | - | 0.97(0.07)$^{A}$ |
| | | 25-50 | - | 0.32(0.14)$^{A}$ |
| | Shrub | 0-25 | 1.36(0.07)$^{Aa}$ | 0.98(0.06)$^{Aa}$ |
| | | 25-50 | 0.13(0.04)$^{Ba}$ | 0.28(0.05)$^{Ba}$ |
| | Grass | 0-25 | 0.57(0.10)$^{Ab}$ | 4.52(0.84)$^{Aa}$ |
| | | 25-50 | 0.02(0.004)$^{Ab}$ | 0.24(0.05)$^{Ba}$ |
| Soil*** | Under canopy** | 0-25 | 106.8(10.44)$^{Aa}$ | 111.4(15.12)$^{Aa}$ |
| | | 25-50 | 63.8(5.93)$^{Ba}$ | 70.3(9.98)$^{Ba}$ |
| | Interspace | 0-25 | 47.4(3.66)$^{Aa}$ | 48.0(4.13)$^{Aa}$ |
| | | 25-50 | 39.7(4.47)$^{Aa}$ | 40.5(2.83)$^{Aa}$ |

Different capital letters (A, B) along columns indicate significant differences between soil depths for a given management practice and plant type or soil area ($P < 0.05$).

Different lowercase letters (a, b) along rows indicate significant differences between management practices for a given soil depth and plant type or soil area ($P < 0.05$).

*Roots from mature trees in the treated watershed were extracted from the base of stumps of cut juniper.

**Soil carbon under tree canopies in the treated watershed was determined from areas near the base of stumps of cut juniper.

***Soil carbon was not adjusted by the amount of surface under canopy cover (or near tree stumps) and interspace because the purpose of this table is to show the absolute differences between these two areas.

**4. Discussion**

This study reports quantitative estimates of ecosystem carbon stocks in encroached juniper systems following juniper control. Given the importance and extent of juniper control as a common rangeland management practice in western USA, it is essential to shed light on the environmental implications of such practice. The vegetation differences between treated and untreated watersheds that we report are attributed to juniper control because prior to it there were no statistically significant differences in vegetation cover (including juniper, shrubs, and grasses) between the same treated and untreated watersheds (Fisher, 2004)

*4.1. Effects of management practices on aboveground carbon stocks*

As hypothesized, the treated watershed had less aboveground carbon after 13 years of juniper control than the untreated watershed. This was attributed to the elimination of mature western juniper trees, which represented the bulk of the aboveground carbon. Thirteen years after mature juniper control, the regrowth juniper trees were abundant on the treated watershed, as seen by their high density, but their aboveground biomass accumulation was only about 5% of that of mature trees. Therefore, the contribution of regrowth trees to the carbon accrual was small. Juniper control caused a clear increase in shrub presence, as it has been previously reported in different areas of Oregon (Bates et al., 2017; Dittel et al. 2018; Ray et al., 2019), but similar to regrowth juniper trees, the contribution of shrub aboveground biomass was small because of the dwarfing effect of mature juniper trees. As expected, litter carbon stocks were much higher in the treated than in the untreated watershed due to the juniper cut-and-leave operation. Litter carbon in the treated watershed was the largest pool, accounting for 51% of the total aboveground carbon in it. An unexpected result was that grass carbon stocks were lower in the treated watershed than in the

untreated watershed; previous reports have found higher grass presence after juniper cutting (Bates et al., 2017; Dittel et al. 2018; Ray et al., 2019). It is possible that the lower grass stocks at the treated watershed resulted from uneven cattle grazing, heavier on the treated watershed, during the year of evaluations.

Aboveground carbon stocks usually increase due to woody species encroachment through time while elimination of woody vegetation would decrease those stocks (Barger et al. 2011; Eldridge et al., 2011). Such a response was observed in our juniper study site, despite the positive response of shrubs and the litter accumulation resulting from juniper control. Throop and Lajtha (2018) also reported a decrease in aboveground carbon stocks following western juniper control.

The greater total aboveground carbon stocks observed in the untreated watershed were within the ranges of aboveground biomass carbon stocks (10–65 Mg C ha$^{-1}$) reported for southern Great Plains encroached by mesquite (*Prosopis* spp) (Hibbard et al., 2001; Knapp et al., 2008); Great Basin encroached by western juniper (Tiedemann, and Klemmedson, 1995); and Great Plains encroached by *Juniperus virginiana* (Knapp et al., 2008; McKinley and Blair, 2008). Compared to mature western juniper trees, the contribution of shrubs, grasses and litter to total aboveground carbon stocks in the untreated watershed was minimal. Combined, the total aboveground carbon stocks of those three pools was 0.48 Mg C ha$^{-1}$, representing 2.2% of the tree contribution in the untreated watershed.

*4.2. Effects of management practices on belowground carbon pools*

Regardless of soil depth, this study showed a clear increase (2.6-fold) in root carbon stocks 13 years post juniper control in the treated watershed. The change was mainly due to the large increase in grass roots following juniper control, despite the observed slight decrease in grass

aboveground biomass. Other studies have found grass biomass production surges post juniper control (Bates et al., 2017; Dittel et al. 2018; Ray et al., 2019) but grass root responses have not been previously documented. Because of their typically large root:shoot ratio, perennial grasses can store large amounts of carbon belowground (Evans et al., 2013). In our study, grass roots contributed slightly more than the other plants types (juniper and shrubs) combined to the total root carbon stocks in the treated watershed. The important contribution of grass roots to carbon stocks was previously reported by Sharrow and Ismail (2004) who found that soil organic carbon in pastures was greater than in tree plantations and agroforests.

Soils under-canopy and near-stump zones of cut western juniper had soil carbon stocks twice as great as those of interspace zones across the watersheds, which is consistent with previous studies on woody plant canopies (Throop and Archer, 2008; Neff et al., 2009; Miwa and Reuter, 2010; DeMarco et al., 2016; Zhou et al., 2017). Our results also indicate that soil carbon near-stump zones of cut western juniper remained elevated even 13 years following tree elimination and did not differ from soil carbon under-canopy zones of mature western juniper of the untreated area. Another study has found that soil carbon remains elevated under mesquite (*Prosopis velutina*) canopies 8 years after cutting (DeMarco et al., 2016). However, 40 years after mesquite cutting, a loss of soil carbon was observed (McClaran et al., 2008) indicating the recalcitrant nature of woody biomass in arid soils (Zhang and Wang, 2015).

Total soil carbon stocks at 50 cm depth, including both under-canopy and interspace areas, remained unchanged 13 years after juniper elimination with an average of 118 Mg C ha$^{-1}$. This is similar to results by Throop and Lajtha (2018) who suggested that the lack of degradation of organic material in western juniper settings might reflect stabilized carbon pools. Our results did not support our hypothesis of lower soil carbon following juniper control. It is possible that after

a longer time (40 more years) of woody plant control a decrease in soil carbon may occur, as has been reported in other systems (McClaran et al., 2008; Neff et al., 2009; DeMarco et al. 2016). Yet, that decrease would be reflected mainly on the areas under the juniper canopy, which amount to about 30% of the whole area. The other 70% of the area, the juniper interspaces, would be much less affected by the juniper control. In fact, an increase in soil carbon is more likely in these areas because of the increase in grass and shrub roots following juniper control. Thus, the replacement of mature juniper roots by those of understory vegetation might compensate potential losses of soil carbon in the long term. As stated by Barger el al. (2011), the control of woody vegetation might shift the vegetation growth to more dynamic, younger, and more productive plant populations (grasses and shrubs in our case).

Soil organic carbon stocks integrate long-term contribution from roots and aboveground plant matter; thus, it is expected that soil carbon stock would be negatively affected by elimination of vegetation, i.e., woody plant control (Archer et al., 2017). However, changes in soil carbon are much slower than those aboveground, which explains that soil carbon is negatively affected by woody plant control only after >40 years of the practice in the Sonoran Desert (McClaran et al. 2008). The aridity level in our study area of central Oregon is lower than in the Sonoran Desert. Therefore, it is more likely to expect carbon sequestration following woody plant control and long-term plant succession in our study area than in the Sonoran Desert (Archer et al., 2017). Although our study is still of short duration (13 years post juniper control), our plant succession results, with positive biomass gain in understory vegetation, led us to hypothesize that juniper control is not likely to result in significant net carbon losses.

*4.3. Total carbon stocks and management practices*

Total carbon stocks in both management practices did not differ and showed an average carbon level of 137.6 Mg C ha$^{-1}$. The encroachment of western juniper increased aboveground carbon stock, but the belowground carbon stock (0-50 cm soil depth) was not affected by the management practices of untreated and treated watersheds. These findings partially supported our hypothesis, indicating that the treated watershed stored less aboveground carbon relative to the untreated watershed. Even though woody encroachment in globally extensive arid environments usually increases aboveground carbon stocks, its impact on belowground carbon stock is uncertain, varying with spatial scale, species composition, and environmental conditions (Barger et al., 2011; Eldridge et al., 2011; DeGraaff et al., 2014). In our study, the gain of understory plant roots associated with western juniper cut led to partially counteract the losses of aboveground carbon in the treated watershed.

In both treated and untreated watersheds, the large majority of the total carbon pool was stored belowground (roots and soil). According to the present study, 84% and 97% of the total carbon stocks in the untreated and treated watershed, respectively, are allocated belowground (0-50 cm soil depth). Our results are comparable to those of Sharrow and Ismail (2004) who reported that 88% and over 90% of the total carbon stocks were stored in the soil (0-45 cm depth) for agroforest and grasses-dominated pastures, respectively. In addition, rangeland ecosystems were reported to contain 85.8% of the total carbon stocks in the soils (0-30 cm depth) (Bikila et al., 2016). Cold desert ecosystems such as the sagebrush steppe in North America generally have very low ratios of aboveground to belowground biomass, and belowground herbaceous biomass tends to substantially contribute to belowground carbon pools (Jackson et al., 2000).

Most organic carbon in terrestrial ecosystems is contained in the soil (Schlesinger, 1997). We found that soil carbon was the single greatest carbon pool in both watersheds, representing 81% (untreated watershed) and 90.1% (treated watershed) of the total ecosystem carbon stocks. Consequently, any anthropogenic activities that might have negative impacts on soils, such as grazing-induced erosion (Carbajal-Moron et al., 2017) would have major implications in reducing carbon stocks in these systems. Even without considering erosion, inadequate management practices such as overgrazing may lead to carbon losses in the upper soil layers (Daryanto et al., 2013; Bikila et al., 2016).

The aboveground biomass in arid and semi-arid woodlands is viewed as an unstable organic carbon pool because of the frequency of wildfire in these systems (15-90 years), the threat of exotic grass invasion, and the poor recovery of important shrubs such as sagebrush following fires (Rau et al., 2011; Canadell and Raupach, 2008; Reed-Dustin et al., 2016; Mata-Gonzalez et al., 2018). Therefore, it is imperative to protect the stable belowground carbon pool in juniper-managed areas to mitigate climate change and global warming. Degradation of the belowground carbon pool will eventually result in a reduction of total carbon stocks in the system.

Elimination of juniper encroachment has well documented benefits such as habitat restoration for native wildlife, increased forage base for livestock, increased soil moisture and restoration of watershed hydrological functions (Baruch-Mordo et al. 2013; Dittel et al., 2018; Ochoa et al., 2018; Ray et al., 2019). Our study provides basis to suggest that the benefits of juniper control can be attained without substantially affecting the potential for carbon sequestration of these systems.

*4.4. Effects of soil depths on belowground carbon within management practices*

The shrubs and grasses in our study area had generally greater root carbon stocks in the top soil layer (0-25 cm) than in the bottom soil layer (25-50 cm). In fact, about 95% of grass root carbon stocks in the treated watershed was concentrated in the top soil layer. This is in agreement with reports for Great Basin vegetation (Rau et al., 2009) and in general with estimations of global carbon stock distribution with soil depths (Jobbagy and Jackson, 2000). Grasses have a dense, fibrous root system of shallow depth in the top 20-30 cm of the soil profile, where water and nutrients are at maximum concentrations (Archer et al., 2017). In contrast to shrubs and grasses, root carbon stocks for juniper trees were more homogeneously distributed with depth down to 50 cm, coinciding with results by Young et al. (1984). Western juniper develops a large extension of lateral roots (Mollnau et al., 2014) with their greatest proportion found within 0-50 cm soil depths (Young et al., 1984). The main lateral roots grow to radii that are at least equal to the height of a tree and extend beyond the dripline determined by the extension of the canopy (Young et al., 1984) but with large restriction of roots to surface soils (Miller et al., 2005). Woody species roots are typically more lignified and deeper rooted than the grasses they displace (Boutton et al., 1999; Barger et al., 2011).

The top soil layer contained more soil carbon in both under-canopy and near-stumps zones of western juniper. The mechanism responsible for the increase in soil carbon at 0-25 cm was likely the concentration of organic matter inputs from litterfall and the incorporation and redistribution of soil carbon into near surface soils (Jobbagy and Jackson, 2000; Eggleston et al., 2006; Zhou et al., 2017). Although it decreased with depth, substantial soil carbon under-canopy and near-stumps zones of western juniper existed below 25 cm soil depth. That was consistent with results of soil carbon sequestration occurring deeper than 30 cm following woody

encroachment (Chiti et al., 2017; Zhou et al., 2017). Our results showed that 38% of the soil carbon stocks in under-canopy and near-stump zones was present in the 25-50 cm soil layer. In contrast, the interspace areas did not show difference in carbon stocks with soil depth, reflecting the lower organic matter inputs that these areas receive.

Carbon stocks of surface soil layers are more prone to be affected by management or vegetation changes than deeper layers (Bikila et al., 2016; Throop and Lajtha, 2018). Our evaluation of total carbon stocks as affected by juniper control included the 0-50 cm soil layer because juniper roots may influence changes at that depth (Young et al., 1984). However, there were no differences in soil carbon stocks due to juniper control (between watersheds) at either the 0-25 cm or the 25-50 cm soil layers.

*4.5. Limitations of the Study*

This study had the advantage of analyzing juniper elimination at the whole watershed (approximately 100 ha each) scale. Yet, because of its large scale, the replication of the study is limited to one watershed per treatment and the interpretation and extrapolation of the results should be done cautiously. In addition, our vegetation and soil sampling occurred in the 20 replicated plots scattered through the watersheds. However, root sampling was done in only three trenches per vegetation type in each watershed because of the difficulty of moving an excavator throughout the whole study area (more than 200 ha). Root sampling is always a difficult task and in some cases, roots are only estimated based on biomass modeling.

## 5. Conclusions and Implications

Juniper encroachment is considered an important contributor to rangeland degradation in North America and thus its control is a common restoration practice with ecological benefits to native wildlife and livestock. Yet, the implications of juniper control in terms of ecosystem carbon stocks are largely unknown. This motivated our study. We hypothesized that a treated watershed, after 13 years of mature western juniper elimination, would store less aboveground and belowground carbon than an untreated watershed, implying lower capacity for soil carbon sequestration resulting from juniper control. Although aboveground carbon stock was reduced in the treated western juniper area, the belowground carbon stock was not, rendering no significant effects on total carbon stocks (aboveground and belowground) caused western juniper control. A greater root carbon accumulation in the treated area than in the untreated area partially offset the losses in aboveground carbon due to juniper control.

The greatest ecosystem carbon accumulation resides belowground (over 90%). Therefore, changes in the 10% aboveground biomass can be of less relative significance in the short term. However, our 13-year post treatment study, is still of short duration to contemplate soil carbon changes. It is not known if juniper control may result in lower soil carbon stock in decades to come but our vegetation succession results do not anticipate that.

Protecting the belowground carbon source is paramount. This study indicates that cutting western juniper did not affect belowground carbon pools, at least after 13 years of cutting. Western juniper elimination facilitates the recovery of shrubs and grasses and permits the restoration of watershed hydrological functions. Therefore, evidence supports that juniper control can help to improve the habitat quality for wildlife such as sage grouse and the forage productivity for moderate cattle grazing while maintaining the carbon sequestration potential of the system. We

would like to emphasize, however, that studies of longer duration and mechanistic models are needed to forecast and better understand carbon stocks as affected by land management in these dynamic systems.

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# ECOSYSTEM NITROGEN AS A CONSEQUENCE OF JUNIPER ENCROACHMENT AND REMOVAL IN CENTRAL OREGON, USA

**Abstract**

In recent decades, substantial areas of North America grasslands have been lost because of the establishment and encroachment of woodlands, including those dominated by western juniper (*Juniperus occidentalis*). This shift in dominant plant life form may be accompanied by changes in standing stocks of biomass with subsequent implications on the nitrogen pools for an area. Our study site was a paired watershed in central Oregon where western juniper trees were eliminated in one watershed (treated, 116 ha) and were left intact in an adjacent watershed (untreated, 96 ha). Thirteen years after elimination, we quantified aboveground biomass and nitrogen stores for western juniper trees, shrubs, grasses, and litter in both the treated and untreated watersheds. We also quantified belowground nitrogen stores (roots and soil) in both watersheds at two soil depths (0-25cm and 25-50cm). As a result of greater aboveground biomass, aboveground nitrogen storage in the untreated watershed (425.4 kg N ha$^{-1}$) was substantially greater than that in the treated watershed (61.9 kg N ha$^{-1}$). On the other hand, root nitrogen storage was 3.1 times greater in the treated than in the untreated watershed due to the gain of understory root biomass associated with western juniper cutting. Soil nitrogen stores at both 0-25 cm and 25-50 cm depth were not affected by juniper removal. Overall, total ecosystem nitrogen storage (average 1283.2 kg N ha$^{-1}$) was not different between watersheds. Most nitrogen resided belowground (soil 0-50 cm and roots); 69% and 95% of the total ecosystem nitrogen, respectively, was found in the untreated and treated watershed. Cutting juniper, a common ecological restoration practice in Oregon, is documented to restore understory plants and improve hydrological functions of systems. Our study indicates that juniper removal does not affect substantially the nitrogen ecosystem pools.

**Keywords:** Aboveground nitrogen store, belowground nitrogen store, *Juniperus occidentalis,* soil nitrogen, watershed management, woody plant encroachment.

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## 1. Introduction

The widespread replacement of grasses by woody plants has occurred in many ecosystems (Barger et al., 2011; Eldridge et al., 2011; Li et al., 2016; Stevens et al., 2017; Wang et al., 2019). This geographically worldwide vegetation change, woody encroachment, has been shown to change the ecosystem structure by altering the vegetation community composition, hydrology, and the spatial distribution and fluxes of nutrients (Boutton et al., 2009; Archer, 2010; Michaelides et al., 2012; Chen et al., 2015). Following vegetation shift from grass to woody plant dominance, the quantities and qualities of aboveground and belowground residue inputs as well as the soil microbial biomass are altered (Filley et al., 2008; Liao and Boutton, 2008; Godey et al., 2010; Barger et al., 2011; Li et al., 2017; Zhou et al., 2017), with subsequent implications on carbon and nitrogen pools (Booth et al., 2005; Hughes et al., 2006; Throop and Archer, 2008; McKinley and Blair, 2008). Nitrogen is considered the growth-limiting nutrient in arid and semi-arid regions (Gebauer and Ehleringer, 2000) and almost in all western ecosystems (Rau et al., 2009), yet the effects of woody plants on nitrogen accumulation require further research. Woody plant encroachment resulted in increases (Archer et al., 2004; Liao et al., 2006; Boutton and Liao 2010), declines (Li et al., 2012; Yusuf et al., 2015), as well as no significant change (McCarron et al., 2003) in soil nitrogen storage. Moreover, the total nitrogen pool in grasslands experiencing woody plant encroachment tended to increase (Liao et al., 2008; Boutton and Liao, 2010) and decrease (Jackson et al., 2002).

In the past 150 years, juniper (*Juniperus* spp.) and piñon (*Pinus* spp.) coniferous woodlands have increased 2 to 10-fold across the Intermountain region of the western United States (Omernik, 1987; Romme et al., 2009). Nearly, 95% of the expansion has occurred in the sagebrush (*Artemisia* spp.) steppe communities (Miller et al., 2011). Among the array of encroaching juniper species, western juniper (*Juniperus occidentalis* spp. *occidentalis* Hook.) is estimated to cover about 4 million hectares in semi-arid regions of the inland northwest (Azuma et al., 2005). The greatest concentrations of western juniper are found in eastern Oregon, and northeast California (Bates, 1996). Several factors are associated with juniper encroachment, including changes in climate, increases in the atmospheric carbon dioxide, overgrazing, and reduced fire frequencies (Burkhardt and Tisdale, 1976; Miller and Wigand, 1994; Soulé et al., 2004; Miller et al., 2005). The encroachment of juniper is reported to cause major declines in understory productivity and diversity (Miller and Wigand, 1994; Miller et al., 2000; Miller et al., 2014a, Miller et al., 2014b; Roundy et al., 2014), change the spatial distribution of nutrients litter and soils beneath juniper canopies (Doescher et al., 1987; Klemmedson and Tiedemann, 2000; Miwa and Reuter, 2010), increase soil erosion (West, 1984; Miller et al., 2005), and increase interception of precipitation (Larsen, 1993; Eddleman et al., 1994). Elimination of juniper stands has resulted in significant improvements compared to encroached areas (Pierson et al., 2014; Ochoa et al., 2018; Ray et al., 2019).

Although studies pertaining the western juniper ecosystem are increasing, quantitative evidence regarding the implications of juniper encroachment and removal on ecosystem nitrogen pools is lacking. Studies indicated that succession to western juniper woodland increased nitrogen and other soil nutrients in juniper biomass, litter mats, and canopy influenced soils, but understory vegetation such as grasses and shrubs, were not included in total nitrogen stocks calculation

(Doescher et al., 1987; Tiedemann and Klemmedson, 1995; Klemmedson and Tiedemann, 2000; Bates, 1996; Bates et al., 2002). Elimination of western juniper (cut treatment) increased grass nitrogen biomass to 6.19 kg N ha$^{-1}$ compared to 0.59 kg N ha$^{-1}$ in woodland treatment in the second year post-cutting, but neither the aboveground juniper biomass nor belowground biomass was determined (Bates et al., 2000). Furthermore, previous studies were primarily concerned with the impacts of juniper encroachment on topsoil layers ($\leq$ 24 cm), so there is a lack of information about the influence of western juniper encroachment on deep soil nitrogen stocks. Understanding the effects of transition from grassland to woodland on total nitrogen has significant implications for global change, whole ecosystem dynamics, land resource use and ecosystem management (McKinley et al., 2008; Wei et al., 2009; Yusuf et al., 2015). It is also essential to understand the effects of juniper cutting on ecological processes such as nitrogen because cutting is expected to increase in the near future for land management purposes (Bates et al., 2002).

Our study site involved paired watersheds in central Oregon. In one of the watersheds western juniper trees have been eliminated for 13 years (the treated watershed) whereas in the other watershed the western juniper trees have been left intact (the untreated watershed). Following western juniper elimination on the treated watershed, changes in vegetation composition have been reported, including western juniper regrowth and greater presence of shrubs and grasses (Ray et al., 2019). Because of the vegetation changes following western juniper removal, the potential influence of this management practice on nitrogen accumulation is difficult to forecast. Evaluations of ecosystem nitrogen stores require an understanding of how both aboveground stores (including understory vegetation) and belowground stores respond to the presence and removal of western juniper trees. The goal of this study was to quantify and compare major pools of nitrogen in an encroached juniper watershed and an adjacent watershed where juniper removal took place

13 years prior to quantification. We hypothesized that 13 years after cutting mature western juniper, aboveground and belowground nitrogen would have decreased in the treated watershed compared to the untreated watershed, implying lower soil nitrogen pools due to juniper removal.

## 2. Methods and materials

### 2.1. General site description

This study was carried out in the Camp Creek-Paired Watershed Study (CCPWS) site, located 27 km northeast of Brothers, Oregon (43.96° Lat.; −120.34° Long). The CCPWS site comprises an area of around 212 ha and includes two adjacent watersheds, one treated (116 ha) and the other untreated (96 ha) (Fig. 3.1) with elevations ranging from 1370 m to 1524 m. The average slope for each watershed is approximately 25% with similar distribution of aspects (Fisher, 2004). The untreated watershed is dominated by western juniper. Juniper canopy cover averaged 31.5% and tree density averaged 327 trees/ha in the untreated watershed (Abdallah et al, 2020). In the treated watershed, approximately 90% of the western juniper trees were cut in the fall of 2005 using chainsaws, leaving only old-growth trees intact and a big sagebrush (*Artemisia tridentata, spp vaseyana*) as dominant overstory vegetation. Prior to juniper elimination from the treated watershed, tree canopy cover was estimated to cover 27% of the whole area (Ray et al., 2019) and there were no statistically significant differences in vegetation cover (including juniper, shrubs, and grasses) between both watersheds (Fisher, 2004). After cutting, juniper cover was estimated to account for 1% in the treated watershed (Ray et al., 2019).

The understory was dominated by Idaho fescue (*Festuca idahoensis*), bluebunch wheatgrass (*Pseudoroegneria spicata*), Sandberg bluegrass (*Poa secunda*), prairie junegrass (*Koeleria macrantha*), and Thurber's needlegrass (*Achnatherum thurberianum*). In addition to *A.*

*tridentate,* more species such as antelope bitterbrush (*Purshia tridentata*), rubber rabbitbrush (*Ericameria nauseosa*), and green rabbitbrush (*Chrysothamnus viscidiflorus*) were common in the whole study site. The average annual precipitation (2009-2017) at the study site was 358 mm (Ochoa et al., 2018). Soils in both watersheds are classified as Westbutte very stony loam, Madeline Loam, and Simas gravelly silt loam, where Westbutte and Madeline define around 70% to 74% of the study site and Simas makes up the rest with more soil series occupying <1% (Fisher, 2004). Both Westbutte and Madeline series are formed of colluvium derived from basalt, tuff and andesite, whereas Simas is formed of loess and colluvium derived from tuffaceous sediments (OSDs). The Westbutte series is classified as loamy-skeletal, mixed, superactive, frigid Pachic Haploxerolls. The Madeline series is classified as clayey, smectitic, frigid Aridic Lithic Argixerolls. The Simas series is classified as fine, smectitic, mesic Vertic Palexerolls. The untreated watershed is mainly made of 48% Madeline, 26% Westbutte, and 21% Simas series, while the treated watershed consists of 50% Westbutte, 20% Madeline, and 3% Simas series (Fisher, 2004).
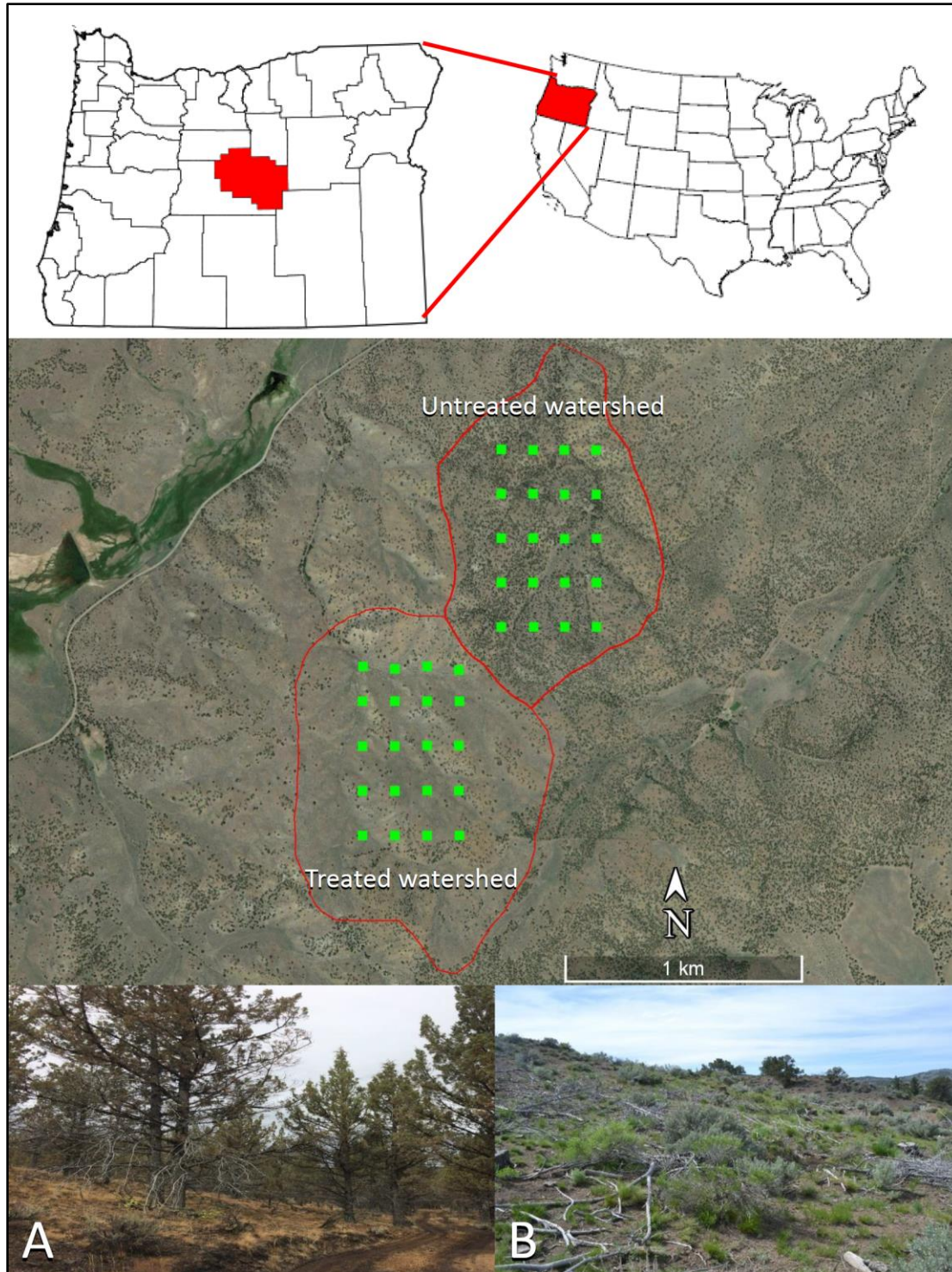
Figure 3.1. Map of the study area showing untreated and treated watersheds, indicating locations of systematically randomized plots used in this study. Photograph A shows intact western juniper (*Juniperus occidentalis*) trees in the untreated watershed, while photograph B shows cut western juniper trees in the treated watershed.

*2.2. Experimental design*

The two management practices (treatments) used for this study were; 1) untreated = juniper trees and all vegetation species were kept intact and maintained in the untreated watershed and 2) treated = juniper trees were manually removed with chainsaws to ground level in 2005 and other vegetation species were left intact in the adjacent treated watershed. The felled trees and debris as a consequence of juniper removal were scattered and left on the ground. Both watersheds were used for cattle grazing before and after juniper elimination. A random sampling procedure was followed in this study. Twenty replicate sites (plots), each of 20 m X 20 m were established in each watershed. The 20 plots were systematically randomized in a 4 X 5 grid trying to represent varying characteristics of slope and aspect within each watershed (Fig. 3.1). The coordinates of the sampling locations were randomly generated, located on a digital map, and then the plots were found on the terrain with the help of a GPS unit. The distance between plots was 130 m among columns and 180 m among rows. The 20 m X 20 m plots were used to sample western juniper trees. Then, a 10 m X 10 m plot within each 20 m X 20 m plot was established for estimating shrub biomass (Fig. 3.2). In addition, four plots of 2 m X 2 m within each 20 m X 20 m plot were established for grass and litter biomass evaluations.
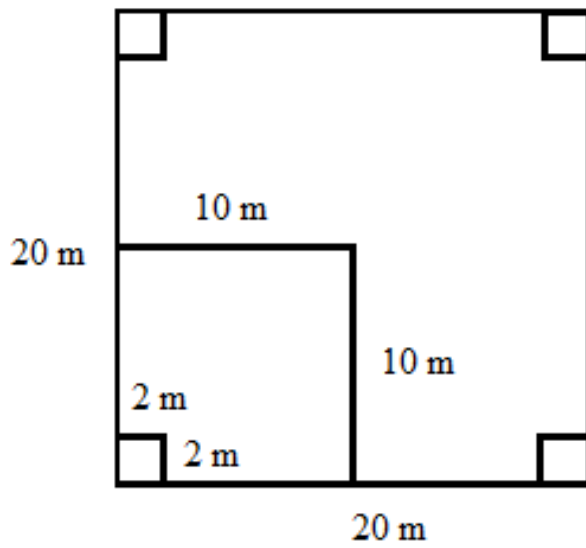
Figure 3.2. Plots used in the study for biomass evaluations. The 20 m X 20 m plots were used for western juniper (*Juniperus occidentalis*) sampling, the 10 m X 10 m plots were used for shrub sampling, and the 2 m X 2 m plots were used for grass and litter sampling.

*2.3. N Stores: Aboveground biomass and litter*

The allometric equations developed by Sabin (2008) was employed to estimate aboveground biomass of all juniper trees in each 20 m X 20 m plot of the untreated watershed. For each individual juniper tree within a given plot, canopy area was calculated by averaging the tree diameters from two different directions. Then, canopy area ($x=m^2$) was transformed into aboveground biomass ($y=$ kg) using the equation $y = 9.7164x + 37.506$ ($R^2=0.86$). For the regrowth juniper in the treated watershed, aboveground biomass was estimated by counting all trees within a given plot, cutting and collecting a representative sample, determining its dry weight at the lab, and extrapolating biomass weight by area. Likewise regrowth juniper trees, the shrub biomass in both watersheds was estimated, but one representative sample was collected for each shrub species found within the sampling plot, 10 m X 10 m. Aboveground grass biomass was measured in both

watersheds by clipping and collecting all live standing tissue for dry matter analysis within each 2 m X 2 m plot (80 plots in each watershed). In addition, litter sampling was applied to the same grass plots where all dead lying tree, shrub, and grass materials considered litter were picked and weighed. The nitrogen concentrations (%) for aboveground biomass were determined in five samples for western juniper, grasses, the main shrub species found in the area, and litter. Each sample was dried at 60°C until constant weight, finally ground, and analyzed for total nitrogen using a CNS automatic analyzer (Elementar Vario MMARCO CNS, Elementar Analysen Systeme GmbH, Hanau, Germany) at the Central Analytical Laboratory of the Crop and Soil Science Department at Oregon State University (Corvallis, OR).

### 2.4. N Stores: Root biomass

The trench method by Komiyama et al. (1987) was applied to measure root biomass in both watersheds. In the treated watershed, root biomass was measured for random stands of regrowth juniper, remaining tree stumps, shrubs, and grasses, while in the untreated watershed, it was measured for random stands of mature western juniper trees, shrubs, and grasses. Three trenches for each vegetation type in each watershed were dug to a depth of 50 cm using an excavator (Bobcat Inc. West Fargo, North Dakota, USA). The trench width was 61 cm while the trench length was about 3 m (the exact length for each trench was measured and recorded). Twenty-one trenches (9 in the untreated watershed and 12 in the treated watershed) were dug in total. The trenches were dug first for the top 25 cm and then for the subsequent 25 cm, keeping the soil from each depth separated. The roots from each trench and soil depth were carefully separated from the soil using a sieve, rinsed, oven-dried, and weighed. Similar to aboveground biomass, five root samples were obtained from each vegetation type for nitrogen concentration determinations. Since it was

difficult to separate shrub roots by species, root nitrogen for shrub was determined without distinguishing shrubs species. The dry root tissues were subsequently ground for lab analysis. Total nitrogen of roots was determined in a similar manner as aboveground biomass and with the same analyzer. For aboveground and belowground biomass, percent of total nitrogen was multiplied by each fraction's mass to obtain the mass of total nitrogen per area.

*2.5. N concentrations: Soil sampling*

Soil samples were obtained from all systematically randomized plots (20 plots in each watershed) (Fig. 3.1) with a cylindrical soil step probe of 1.6 cm inner diameter (AMS, Inc; American Falls, Idaho, USA). Soil sampling was done from areas 50 cm to the trunk (under the canopy of mature western juniper trees in the untreated watershed) and stump (near stumps of cut western juniper tree in the treated watershed). In addition, soil samples were obtained from interspaces of both watersheds. Therefore, there were two sampling contexts in each plot, under-canopy/near stumps and interspace. Two soil samples were collected from each sampling context of each plot at two soil depths (0-25 cm and 25-50 cm). In total, eight soil samples were collected for each plot (2 samples x 2 conditions x 2 depths). Each sample was sieved to avoid gravel or rocks, oven-dried at 40 °C for 48 h, weighed, and analyzed using the same analyzer used for vegetation pools.

*2.6. N stores: Soil bulk density and nitrogen calculation*

A soil core ring with a radius of 2.5 cm was used to determine bulk density for plots selected from the middle of the 20 plots in each watershed. One core sample was obtained in each zone of under-canopy of mature western juniper trees in the untreated watershed, near-stump of

cut western juniper trees in the treated watershed, and in interspaces of both watersheds at two soil depths (0-25 cm and subsequent 25-50 cm). Soil core samples were oven-dried at 105°C for 48 h and weighed. Bulk density was calculated as the ratio of the mass of oven-dried soil sample to core volume (g cm$^{-3}$).

The soil nitrogen mass per area was calculated by the following equation:

$$Ns = (STN \times BD \times D)/10 \text{ (Deng et al., 2016)}$$

where, Ns is soil nitrogen stock (Mg N ha$^{-1}$), BD is soil bulk density (g cm$^{-3}$), STN is soil total nitrogen content (%) and D is the thickness of the sampled soil layer (cm). Soil nitrogen mass per plot area were determined by multiplying each plot zone cover (under-canopy/near-stump and interspace zones for both watersheds) by the total areas of the relevant plot zone-cover in order to estimate zone-cover specific soil nitrogen stocks. Areas of juniper canopy cover and interspaces were measured in a previous study (Abdallah et al., 2020) in the same site. Finally, the values for both zones were summed for each plot to obtain soil nitrogen stocks per plot. All nitrogen values for aboveground and belowground biomass were presented in units, kg N ha$^{-1}$.

## 2.7. Statistical analysis

Differences in each analyzed variable were determined by single factor analysis of variance (ANOVA) with two-sample t-tests (treated vs. untreated). All analyses were performed with the R Statistical Software (R Core Team 2019). The number of sample replications by treatment varied by analyzed variable; for aboveground biomass of juniper and shrubs n=20, for grasses and litter n=80, for root biomass by plant type n=3, for soil nitrogen n=20.

# 3. Results

## 3.1. Nitrogen concentration of biomass samples

The purpose of obtaining the nitrogen concentrations of biomass samples was to calculate nitrogen stores and they were not intended to test for nitrogen content differences among species or plant parts. Yet, it was evident that shrubs especially *Purshia tridentate* had more aboveground nitrogen and grasses had more root nitrogen than woody plants (Table 3.1).

Table 3.1. Mean (±SE) of nitrogen concentration (%) obtained by laboratory analysis of western juniper (*Juniperus occidentalis*), main shrub species, grasses, and litter of the study area, central Oregon, USA. Root nitrogen analysis by species was not performed on shrubs.

| Biomass samples | Aboveground nitrogen % | Root nitrogen % |
| --- | --- | --- |
| Mature western juniper | 0.97(0.03) | 0.41(0.09) |
| Regrowth western juniper | 0.79(0.05) | 0.55(0.07) |
| Shrubs (average) | 1.03(0.11) | 0.85(0.12) |
| *Artemisia tridentata* | 1.06(0.12) | |
| *Purshia tridentata* | 1.31(0.10) | |
| *Eriogonum fasciculatum* | 0.82(0.08) | |
| *Ericameria nauseosus* | 0.92(0.03) | |
| Grasses | 0.61(0.03) | 0.97(0.09) |
| Western juniper stumps | - | 0.60(0.12) |
| Litter (mainly western juniper debris) | 0.62(0.12) | - |

*For all live species (juniper, shrubs, and grasses), the aboveground nitrogen analysis was made in leaves and stems or twigs.

### 3.2. Aboveground nitrogen stores

The aboveground biomass for the trees and grasses was greater ($P < 0.05$) in the untreated watershed than the treated watershed (Table 3.2). However, the aboveground biomass for total shrubs (average) and litter was greater ($P < 0.05$) in the treated watershed than in the untreated watershed. Consequently, aboveground tree nitrogen stores were greater ($P < 0.05$) in the untreated watershed than in the treated watershed (Table 3.2). Mature western juniper trees contained around 26 times more nitrogen than regrowth trees. Similarly, grass nitrogen stores were 60% greater ($P < 0.05$) in the untreated watershed than in the treated watershed.

Table 3.2. Means (±SE) of aboveground biomass and N content parameters for plant groups (western juniper [*Juniperus occidentalis*] trees, shrubs, grasses) and litter by management practice in central Oregon, USA. The management practices are 1) Untreated (western juniper intact) and 2) Treated (western juniper cut). The trees in the Treated management watershed are regrowth western juniper after 13 years of juniper removal.

| Parameters/various components | Biomass (t ha$^{-1}$) | | $P$ value | Biomass N (kg N ha$^{-1}$) | | $P$ value |
|---|---|---|---|---|---|---|
| | Untreated | Treated | | Untreated | Treated | |
| Trees | 43.0(5.41)$^a$ | 2.06(1.39)$^b$ | <0.05 | 417.5(52.5)$^a$ | 16.3(11.0)$^b$ | <0.05 |
| Shrubs (average) | 0.25(0.05)$^b$ | 1.72(0.25)$^a$ | <0.05 | 2.68(0.57)$^b$ | 17.4(2.62)$^a$ | <0.05 |
| *Artemisia tridentata* | 0.13(0.03)$^b$ | 1.31(0.24)$^a$ | <0.05 | 1.37(0.30)$^b$ | 13.90(2.54)$^a$ | <0.05 |
| *Purshia tridentata* | 0.08(0.03)$^b$ | 0.18(0.03)$^a$ | <0.05 | 1.05(0.34)$^b$ | 2.36(0.39)$^a$ | <0.05 |
| *Eriogonum fasciculatum* | 0.02(0.01)$^b$ | 0.12(0.02)$^a$ | <0.05 | 0.16(0.16)$^b$ | 0.97(0.17)$^a$ | <0.05 |
| *Ericameria nauseosus* | 0.01(0.005)$^a$ | 0.01(0.003)$^a$ | 0.42 ns | 0.09(0.05)$^a$ | 0.07(0.02)$^a$ | 0.42 ns |
| Grasses | 0.16(0.01)$^a$ | 0.11(0.01)$^b$ | <0.05 | 0.98(0.09)$^a$ | 0.66(0.06)$^b$ | <0.05 |
| Litter | 0.69(0.11)$^b$ | 4.28(0.53)$^a$ | <0.05 | 4.27(0.71)$^b$ | 26.51(3.26)$^a$ | <0.05 |

Different lowercase letters (a, b) within the same parameter along rows indicate significant differences between management practices for a given plant group or litter ($P < 0.05$). ns = not significant. t ha$^{-1}$= ton of biomass per hectare. kg N ha$^{-1}$= kilogram of nitrogen per hectare.

In contrast, the shrub and litter nitrogen stores were greater ($P < 0.05$) in the treated watershed than in the untreated watershed. Shrubs and litter had 6.5 and 6.2 times more nitrogen in the treated than the untreated watershed, respectively. All the main shrub species had greater ($P < 0.05$) aboveground nitrogen stores in the treated watershed than in the untreated watershed, except for the shrub, *E. nauseosus* where aboveground nitrogen stores were the same between the two areas. Since it was the dominant shrub species in both watersheds, *A. tridentata* stored 51% and 80% of the total shrub nitrogen in the untreated and treated watersheds respectively.

### 3.3. Belowground nitrogen stores

The root biomass (0-50 cm soil depth) for mature western juniper trees (intact vs. stumps) and shrubs were not significantly different ($P > 0.05$) between treated and untreated watersheds (Table 3.3). Root biomass for grasses was significantly ($P < 0.05$) greater in the treated watershed than in the untreated watershed. Although the root biomass for the mature trees was the same in both watersheds, the stump roots in the treated watershed stored about 1.8 more nitrogen compared to the uncut tree roots in the untreated watershed. The stump roots in the treated watershed had 1.5 times more nitrogen concentration compared to the intact tree roots in the untreated watershed (Table 3.1). It was noticeable that root nitrogen store for regrowth juniper on the treated watershed was slightly higher (13%) than in mature intact trees. Root nitrogen stores for the shrubs were not significantly different ($P > 0.05$) between the watersheds. Root nitrogen for grasses was eight times greater ($P < 0.05$) in the treated than the untreated watershed.

Table 3.3. Means (±SE) of root biomass and N content parameters at 0-50 cm soil depth for for plant groups (mature juniper [*Juniperus occidentalis*] trees, regrowth juniper trees, shrubs, and grasses) by management practice in central Oregon, USA. The management practices are 1) Untreated (western juniper intact) and 2) Treated (western juniper cut). The comparison of mature trees included root determination near existing juniper trees (untreated watershed) and near juniper tree stumps (treated watershed).

| Parameters/various components | Biomass (t ha$^{-1}$) | | *P* value | Biomass N (kg N ha$^{-1}$) | | *P* value |
|---|---|---|---|---|---|---|
| | Untreated | Treated | | Untreated | Treated | |
| Mature juniper trees | 3.38(0.20)$^a$ | 4.27(0.52)$^a$ | 0.19 ns | 13.9(0.8)$^b$ | 25.6(3.1)$^a$ | <0.05 |
| Regrowth juniper trees | - | 2.85(0.44) | - | - | 15.7(2.4) | - |
| Shrubs | 3.85(0.25)$^a$ | 3.25(0.22)$^a$ | 0.15 ns | 32.7(2.1)$^a$ | 27.6(1.9)$^a$ | 0.15 ns |
| Grasses | 1.55(0.27)$^b$ | 12.66(2.24)$^a$ | <0.05 | 15.1(2.6)$^b$ | 122.8(21.7)$^a$ | <0.05 |

Different lowercase letters (a, b) within the same parameter along rows indicate significant differences between management practices for a given plant group or litter ($P < 0.05$). ns = not significant. t ha$^{-1}$= ton of biomass per hectare. kg N ha$^{-1}$= kilogram of nitrogen per hectare.

More soil nitrogen ($P < 0.05$) was stored under mature western juniper tree canopies of the untreated watershed and near western juniper tree stumps of the treated watershed than the corresponding interspaces in each watershed (Fig. 3.3a). Under-canopy zones of mature western juniper and near-stump zones of cut western juniper contained approximately 1.7 times more soil nitrogen than the interspaces. Soil nitrogen stores were not significantly ($P > 0.05$) different under the canopies of mature western juniper and near-stump zones of cut western junipers. Likewise, soil nitrogen stores were similar in untreated and treated watershed interspaces. However, considering the areas of zone cover, soil nitrogen stores were not different ($P > 0.05$) between the watersheds or across the zones (Fig. 3.3b).
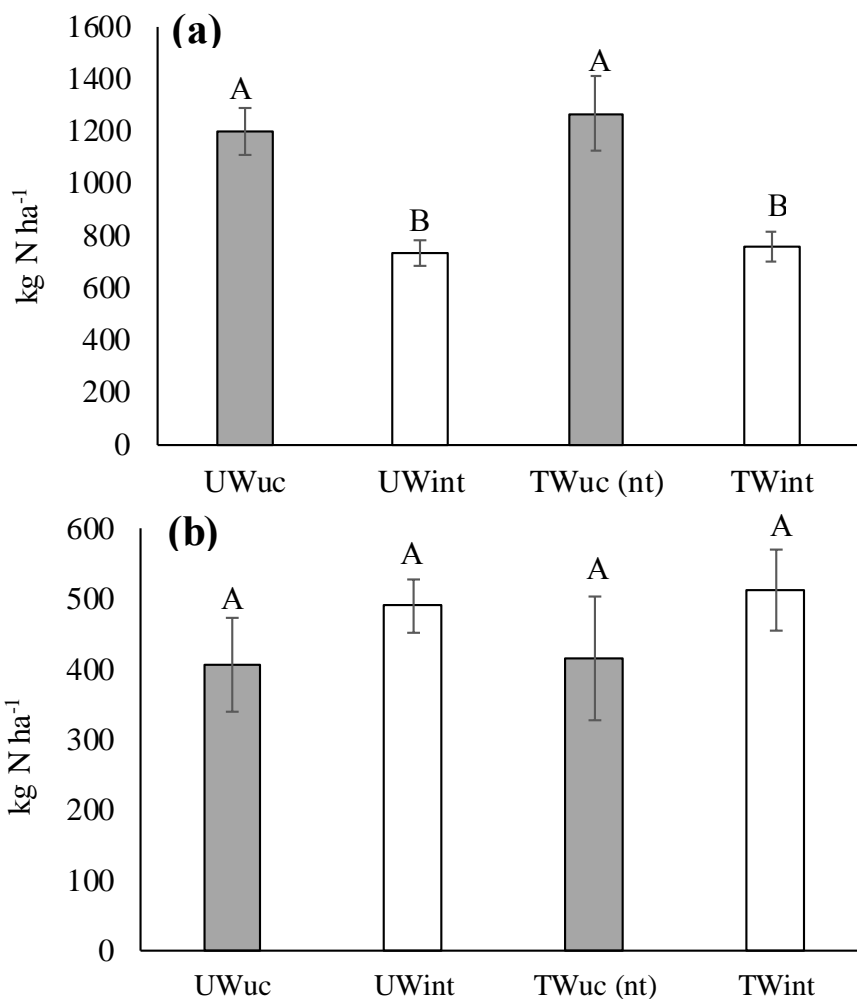
Figure 3.3. Soil nitrogen stores at 0-50 cm soil depth (a) not adjusted by the amount of surface under canopy cover (or near tree stumps) and interspace to show the absolute differences between the two watersheds and (b) adjusted by the amount of surface under canopy cover (or near tree stumps) and interspace to show zone-cover specific soil nitrogen stores. The watershed and treatments are (UWuc = untreated watershed and under tree canopies; UWint = untreated watershed and interspaces beyond tree canopies; TWuc (nt) = treated watershed and under tree canopies (near the tree stumps); TWint = treated watershed and interspaces beyond the tree stumps. Data are means ± standard error.

## 3.4. Total nitrogen stores by management practice

Total aboveground nitrogen stores differed by watershed ($P < 0.05$) (Table 3.4). The untreated watershed had 6.9 times more total aboveground nitrogen than the treated watershed.

Root nitrogen stores were also significantly different ($P$ <0.05) by watershed. Yet, contrary to aboveground biomass, roots in the treated watershed stored 3.1 times more nitrogen than those in the untreated watershed. Soil nitrogen stores were not significantly ($P$ >0.05) different between the untreated and treated watersheds. In contrast to the total aboveground nitrogen stores, total belowground nitrogen stores (root and soil) did not differ ($P$ >0.05) by watershed. Moreover, total belowground nitrogen stores were 2.3 and 18.1 times greater than their corresponding total aboveground nitrogen stores in the untreated and treated watersheds, respectively. Total nitrogen stores, including both total belowground and aboveground nitrogen, were not significantly ($P$ >0.05) different between the untreated and treated watersheds.

Table 3.4. Mean (±SE) total aboveground, total belowground nitrogen stores (0-50 cm soil depth) and total nitrogen stores (kg N ha$^{-1}$) by management practices in central Oregon, USA. The management practices are 1) Untreated (western juniper [*Juniperus occidentalis*] intact) and 2) Treated (western juniper cut). Root nitrogen stores in the untreated watershed represent the sum of root nitrogen in mature western juniper, shrubs, and grasses, while root nitrogen stores in the treated watershed represents the sum of regrowth western juniper, western juniper stumps, shrubs and grasses.

| Management practices | Root nitrogen | Soil nitrogen | Total belowground nitrogen | Total aboveground nitrogen | Total nitrogen stores |
|---|---|---|---|---|---|
| Untreated | 61.6(5.0)[b] | 897.3(58.9)[a] | 958.9(58.9)[a] | 425.4(52.3)[a] | 1384.5(100.7)[a] |
| Treated | 191.7(22.4)[a] | 928.5(84.4)[a] | 1120.2(84.4)[a] | 61.9(10.1)[b] | 1181.9(87.8)[a] |
| $P$ value | <0.05 | 0.76 ns | 0.13 ns | <0.05 | 0.14 ns |

Different lowercase letters (a, b) along columns indicate significant differences between management practices for a given variable ($P$ < 0.05). ns = not significant.

*3.5. Belowground nitrogen stores by soil depth*

Root nitrogen stores for the uncut mature western juniper did not vary significantly ($P$ >0.05) by depth (Table 3.5). In contrast, root nitrogen stores for western juniper stumps and regrowth trees in the treated watershed were 2.4 and 3.0 times respectively greater ($P$ <0.05) in the top soil layer (0-25 cm depth) than in the bottom soil layer (25-50 cm depth). Root nitrogen stores for shrubs in both watersheds were 3.5 to 10.3 times greater ($P$ <0.05) in the top soil layer (0-25 cm depth) than in the bottom soil layer (25-50 cm depth). Additionally, root nitrogen stores for grasses in both watersheds were about 19 to 35 times greater ($P$ <0.05) in the top than in the bottom soil layer. Soil nitrogen stores in under-canopy zones were about 60% and 58% greater ($P$ <0.05) in the top than in the bottom soil layer for untreated and treated watershed areas, respectively (Table 3.5). In the untreated watershed, soil nitrogen store in the interspace zone was 55% greater ($P$ <0.05) in the top than in the bottom soil layer. However, soil depth did not produce significant difference ($P$ >0.05) in interspace soil nitrogen in the treated watershed.

Table 3.5. Mean (±SE) belowground nitrogen stores (kg N ha$^{-1}$) from roots of different plant types and from soil from different areas by soil layer depth and management practices in central Oregon, USA. The management practices are 1) Untreated (western juniper [*Juniperus occidentalis*] intact) and 2) Treated (western juniper cut). Regrowth trees were only evaluated in the treated watershed.

| Compartment | | Soil layer depth (cm) | Untreated Watershed | Treated Watershed |
|---|---|---|---|---|
| Root | Mature Juniper Tree* | 0-25 | 6.32(0.98)$^{Ab}$ | 18.11(2.25)$^{Aa}$ |
| | | 25-50 | 7.55(0.17)$^{Aa}$ | 7.49(4.55)$^{Ba}$ |
| | Regrowth Juniper Tree | 0-25 | - | 11.82(0.82)$^{A}$ |
| | | 25-50 | - | 3.91(1.70)$^{B}$ |
| | Shrub | 0-25 | 29.83(1.48)$^{Aa}$ | 21.53(1.33)$^{Ab}$ |
| | | 25-50 | 2.89(0.86)$^{Ba}$ | 6.09(1.05)$^{Ba}$ |
| | Grass | 0-25 | 14.64(2.69)$^{Ab}$ | 116.61(21.59)$^{Aa}$ |
| | | 25-50 | 0.42(0.12)$^{Bb}$ | 6.15(1.25)$^{Ba}$ |
| Soil*** | Under canopy** | 0-25 | 721.0(62.4)$^{Aa}$ | 731.2(83.4)$^{Aa}$ |
| | | 25-50 | 480.1(31.4)$^{Ba}$ | 530.6(62.7)$^{Ba}$ |
| | Interspace | 0-25 | 405.3(29.0)$^{Aa}$ | 402.3(33.2)$^{Aa}$ |
| | | 25-50 | 327.7(20.8)$^{Ba}$ | 358.9(28.3)$^{Aa}$ |

Different capital letters (A, B) along columns indicate significant differences between soil depths for a given management practice and plant type or soil area ($P < 0.05$).
Different lowercase letters (a, b) along rows indicate significant differences between management practices for a given soil depth and plant type or soil area ($P < 0.05$).
*Roots from mature trees in the treated watershed were extracted from the base of stumps of cut juniper.
**Soil nitrogen under tree canopies in the treated watershed was determined from areas near the base of stumps of cut juniper.
***Soil nitrogen was not adjusted by the amount of surface under canopy cover (or near tree stumps) and interspace because the purpose of this table is to show the absolute differences between these two areas.

**4. Discussion**

This study reports quantitative estimates of ecosystem nitrogen stores in encroached juniper systems following juniper elimination. Also, to our knowledge this the first study to examine the influence of western juniper encroachment and removal on deep belowground (root and soil) nitrogen stores below 25 cm soil depth. Considering the ecological benefits of juniper elimination as common rangeland management practice in western USA, it is extremely important to investigate the effects of such practice on ecosystem services such as nitrogen. Prior to this study, vegetation cover (including juniper, shrubs, and grasses) did not differ significantly between the same treated and untreated watersheds (Fisher, 2004). Thus, the vegetation differences between the watersheds reported in this study are due to juniper elimination in the treated watershed.

*4.1. Effects of management practices on aboveground nitrogen stores*

After 13 years of juniper elimination in the treated watershed, we estimated the mean aboveground biomass in the untreated watershed to be 44.1 t ha$^{-1}$, which was approximately 5.4 times greater than that of the adjacent treated watershed (8.17 t ha$^{-1}$). The total aboveground biomass is comparable to that of oak-pine forest which was estimated to be about 61.9 t ha$^{-1}$ including both overstory and understory biomass (Hubbard et al., 2004). Western juniper trees comprised approximately 97% of the total aboveground biomass in the untreated watershed and 25% in the treated watershed. This resulted in the aboveground nitrogen to be substantially greater in the untreated watershed compared to the treated watershed, supporting our hypothesis. Cutting mature western juniper in the treated watershed caused the aboveground nitrogen to decline considerably compared to the untreated watershed where the uncut mature juniper represented the bulk of aboveground nitrogen. Thirteen years after elimination, the regrowth juniper trees were

already abundant on the treated watershed, but their aboveground nitrogen accumulation was 16.3 kg N ha$^{-1}$ and only 4% of that of mature western juniper. This is comparable to Blair et al. (1998) who reported 40 kg N ha$^{-1}$ for a loblolly pine (*Pinus taeda* L.) plantation 18 years after the establishment. Cutting juniper caused shrub species mainly *A. tridentate*, *P. tridentate*, and *E. fasciculatum* to establish and increase in biomass in the treated than the untreated watershed. The greater shrub biomass in the treated watershed translated into greater biomass nitrogen compared to the untreated watershed. Other studies measured a rapid increase in shrub presence following juniper cutting in different areas of Oregon (Bates et al., 2017; Dittel et al. 2018; Ray et al., 2019).

Additionally, litter accumulation in the treated watershed, contained on average 26 kg N ha$^{-1}$, was substantially greater than the untreated watershed due to the juniper cut-and-leave operation. Litter nitrogen in the treated watershed was the largest pool, accounting for 43% of the total aboveground nitrogen in it. Unpredictable results were that grass biomass and nitrogen stores were lower in the treated than in the untreated watershed. In western juniper and pinyon-juniper woodlands, increases in grass presence following juniper elimination have been measured (Clary, 1989, Bates et al., 2005, 2016, 2017; Dittel et al., 2018; Ray et al., 2019). It is possible that the lower grass biomass accumulation at the treated watershed was attributed to uneven cattle grazing, heavier on the treated watershed, during the year of measurements. The total aboveground nitrogen in the treated watershed, estimated to be 61.9 kg N ha$^{-1}$, is around 50% higher than 30 kg N ha$^{-1}$ measured for grassland area adjacent to *Juniperus virginiana* forest encroached area (McKinley and Blair, 2008).

Mature western juniper-dominated area, untreated watershed stored relatively considerable quantities of aboveground nitrogen, even with the positive response of shrubs and the litter accumulation resulting from juniper elimination in the treated watershed. The greater total

aboveground nitrogen stores observed in the untreated watershed were within the ranges of aboveground nitrogen stores (370-620 kg N ha$^{-1}$) measured in areas encroached by the shrub *Cornus drummondii* (Lett et al., 2004); and *J. virginiana* (Norris et al., 2001; McKinley and Blair, 2008). Compared to mature western juniper trees, the contribution of shrubs, grasses and litter to total aboveground nitrogen stores in the untreated watershed was minimal. Combined, the total aboveground nitrogen store of those three pools was 7.93 kg N ha$^{-1}$, representing 1.9 % of the tree contribution in the untreated watershed.

*4.2. Effects of management practices on belowground nitrogen stores*

This study demonstrated a considerable increase (3.1-fold) in root nitrogen stores 13 years post juniper elimination in the treated watershed, regardless of soil depth. The increase was predominately attributed to the large increase in grass root biomass following juniper cutting, even with the observed slight decrease in grass aboveground biomass. Other studies have shown greater grass biomass production following juniper elimination (Bates et al., 2017; Dittel et al. 2018; Ray et al., 2019), but grass root responses have not been previously investigated. Within the treated watershed, grass roots compared to the combined juniper and shrub roots were the greatest nitrogen pools, accounting for 64% of the total root nitrogen stores in it. The significant contribution of grass roots to nitrogen stores was previously documented by Sharrow and Ismail. (2004) who found that soil organic nitrogen in pastures was higher than in tree plantations and agroforests.

Regardless of zonal cover areas, nitrogen was more concentrated (1.7-fold) in soils under-canopy and near-stump zones of cut western juniper than their corresponding interspace zones in both watersheds. Greater soil fertility in juniper mounds compared to interspaces (McDaniel and Graham, 1992; Davenport et al., 1996; Kramer and Green, 1999) may contribute to the higher soil

nitrogen under-canopy zones of western juniper. Young et al. (2014) found that tree mounds of Utah juniper (*Juniperus osteosperma* Torr.) had greater concentrations of total nitrogen than adjacent interspaces. Our findings agree with previous studies that determined greater concentrations of total nitrogen in soils under woody canopies relative to intercanopy soils (Doescher et al., 1987; Vetaas, 1992; Jurena and Van Auken, 1998; Klemmedson and Tiedemann, 2000; Wheeler et al., 2007; Miwa and Reuter, 2010; Turnbull et al., 2010; Law et al., 2012; Michaelides et al., 2012; Turpin-Jelfs et al., 2019). In addition, our findings indicate that soil nitrogen near-stump zones of cut western juniper remained elevated even 13 years following tree cutting and did not differ from soil nitrogen under-canopy zones of mature western juniper of the untreated area. Western juniper resource islands can persist for 15 years following canopy removal as a result of deep mounds (Miwa and Reuter, 2010). A study measuring nutrient availability for western juniper in Oregon has found that soil inorganic nitrogen in canopy zones, while decreasing overtime, remains dynamic with small differences between cut and uncut treatments lasting into the sixth year posttreatment (Bates and Davies, 2017). Another study measured higher soil inorganic nitrogen in burned canopy litter layers under sagebrush and pinyon juniper trees compared to unburned areas 4 years post fire (Rau et al., 2007).

Considering the zonal cover areas and since soil nitrogen stores at 50 cm depth are not statistically separable among zones across the watersheds, total soil nitrogen stores, including both under-canopy and interspace areas, remained unchanged 13 years post juniper elimination with an average of 913 kg N ha$^{-1}$. That did not support our hypothesis of lower soil nitrogen post juniper elimination. Unchanged total soil nitrogen between forest and grazing lands was measured in Ethiopia (Yimer et al., 2006; 2007). Our findings contrasted with observations reporting increased total soil nitrogen following woody encroachment into grasslands (Liao et al., 2006; Sitters et al.,

2013; Berihu et al., 2017; Zhou et al., 2018; Li et al., 2019). The site-level concentrations of total soil nitrogen remained unchanged with or without the presence of mature western juniper as losses from intercanopy zones were proportional to increases in soils under-canopy/near stump zones (data not shown).

Although not statistically different, soil nitrogen in the interspace zone of the treated watershed was about 1.3 times greater than that of under-canopy/near-stump zones (Fig. 3.3b), which may be explained by the increased plant uptake of nitrogen in mature western juniper compared to the interspaces. Juniper elimination reduces competition for belowground resources and releases additional soil water and nitrogen for uptake by understory plants (Bates, 1996), which was indicated by the higher aboveground biomass nitrogen of understory (combined grasses and shrubs, excluding juniper) and higher root nitrogen in the treated compared to the untreated watershed. The likelihood for nitrogen loss as a consequence of juniper removal appears to be influenced by the ability of the understory to respond to cutting. However, that loss would be reflected mainly on the areas under the juniper canopy, which amount to about 30% of the whole area. The other 70% of the area, the juniper interspaces, would be much less affected by the juniper cutting. In fact, an increase in soil nitrogen is more likely in these areas because of the increase in grass and shrub roots post juniper elimination. Therefore, the replacement of mature juniper roots by those of understory vegetation might compensate potential losses of belowground nitrogen in the long term. Although our study is still of short duration (13 years post juniper removal), our plant succession results, with positive biomass gain in understory vegetation, led us to hypothesize that juniper cutting is not likely to cause significant net nitrogen losses.

*4.3. Total nitrogen stores and management practices*

Total nitrogen stores in both management practices (untreated and treated watersheds) did not differ and showed an average nitrogen level of 1283.2 kg N ha$^{-1}$. Despite substantial increases in aboveground nitrogen with western juniper encroachment in the untreated watershed, we did not see significant changes in belowground nitrogen pools (0-50 cm soil depth) between both watersheds (Fig. 3.4). These findings partially supported our hypothesis, indicating that the treated watershed stored less aboveground nitrogen relative to the untreated watershed. According to Hughes et al. (2006), the stand development of *Prosopis glandulosa* (honey mesquite) increased total aboveground nitrogen including *P. glandulosa* biomass, herbaceous biomass, and litter (~1860 kg N ha$^{-1}$), but with no significant change in the upper 10 cm soil depth accompanying the encroachment in a temperate savanna. Our findings contrast with other observations where significant nitrogen accumulations occurred as a consequence of woody encroaching species, *P. glandulosa* (Liao et al., 2006), *Prosopis velutina* (Wheeler et al., 2007), and *J. virginiana* (McKinley and Blair, 2008) into grasslands. This inconsistency may be attributed to differences in land use/disturbance history, soil physical properties, and climate (Tiedemann and Klemmedson, 2004; Williams et al., 2004; Wessman et al., 2005). In our study, the gain of understory plant roots associated with western juniper removal led to partially counteract the losses of aboveground nitrogen in the treated watershed. Greater root biomass enhances nitrogen cycling and accumulation (Fornara and Tilman, 2008).

In both management practices, the large majority of the total nitrogen pool was stored belowground (roots and soil). According to the present study, 69% and 95% of the total nitrogen stores in the untreated and treated watershed, respectively, are allocated belowground (0-50 cm soil depth) as compared to the aboveground nitrogen pools (Fig. 3.4). Our results are comparable

to those of Sharrow and Ismail (2004) who reported that about 99% of the total nitrogen stores were contained belowground (0-45 cm depth) for both agroforest and grasses-dominated pastures. Moreover, an encroached area by *J. virginiana* stands was reported to contain 89% of total ecosystem nitrogen belowground (0-10 cm soil depth) (McKinley and Blair, 2008). Our results indicated that the single largest pool of nitrogen in the CCPWS site was the soil, accounting for 65% (untreated watershed) and 78.5% (treated watershed) of the total ecosystem nitrogen stores. Therefore, any anthropogenic activities that might have adverse effects on soils will have significant implications in reducing nitrogen stores in western juniper systems. Degradation of the belowground nitrogen pool will eventually lead to a reduction of total nitrogen accumulation in the system.

Juniper removal is associated with several ecological benefits such as habitat restoration for native wildlife, increased understory plants, increased soil moisture and restoration of watershed hydrological functions (Baruch-Mordo et al. 2013; Dittel et al., 2018; Ochoa et al., 2018; Ray et al., 2019). Our study provides basis to suggest that the benefits of juniper elimination can be achieved without considerably disturbing nitrogen pools of these systems.
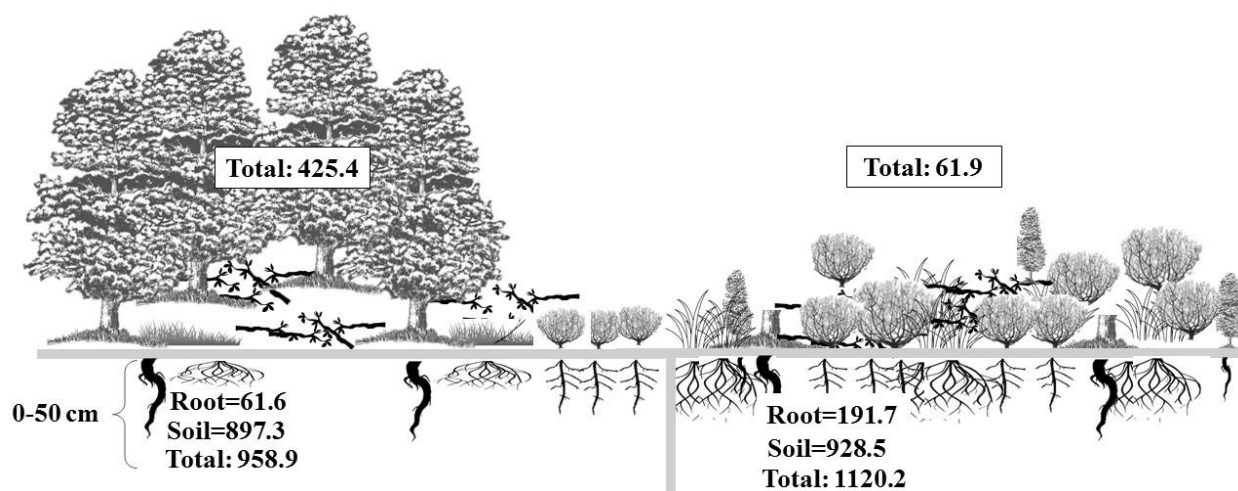
Figure 3.4. Major pools (kg/ha) of nitrogen in mature uncut western juniper area (untreated watershed) and comparable cut western juniper area (treated watershed), including nitrogen values for aboveground biomass and belowground biomass (0-50 cm soil depth).

## 4.4. Effects of soil depths on belowground nitrogen within management practices

In both watersheds, the shrubs and grasses had greater root nitrogen stores in the top soil layer (0-25 cm) than in the bottom soil layer (25-50 cm). In fact, over 95% of grass root nitrogen store was concentrated in the top soil layer for both watersheds. Also, the top soil layer contained about 91% of the total shrub nitrogen in the untreated watershed and 78% of that in the treated watershed. Distribution of shrub and grass root biomass was not evenly along soil depths (0-25 vs 25-50 cm), with significantly greater biomass distribution in the top soil layer than the bottom soil layer (data not shown). While shrubs have a greater root depth in the upper 20-30 cm of the soil profile, grasses also have a dense fibrous root system of this limited depth, well suited to exploit soil resources, where water and nutrients are at peak concentrations (Briske, 2017). Contrary to shrubs and grasses, root nitrogen stores for mature juniper trees in the untreated watershed were

more homogeneously distributed with depth down to 50 cm. Western juniper develops a large extension of lateral roots (Mollnau et al., 2014) with their greatest distribution found within 0-50 cm soil depths (Young et al., 1984). The main lateral roots grow to radii that are at least equal to the height of a tree and extend beyond the dripline determined by the extension of the canopy (Young et al., 1984) but with large restriction of roots to surface soils (Miller et al., 2005). Woody plant roots are typically more lignified and deeply rooted than the grasses they displace (Boutton et al., 1999; Barger et al., 2011). In the treated watershed, root nitrogen for the stump and regrowth juniper did not distribute evenly along depths with concentrations in the upper soil layer, approximately 70-75% greater than the bottom soil layer. Juniper root biomass in the treated watershed was not equally proportioned along soil depths, with substantially greater biomass proportion in the top soil layer compared to the bottom soil layer (data not shown).

Soil nitrogen was considerably held in the top soil layer in both under-canopy and near-stumps zones of western juniper. On the whole soil depth (0-50 cm) in under-canopy zone of western juniper in the untreated watershed, about 60% of the soil nitrogen was held in the top soil layer and the remaining 40% in the lower layer. On the other hand, in near-stumps zone of western juniper in the treated watershed about 58% and 42% of the soil nitrogen was held in the top and subsoil layer, respectively. The increase in soil nitrogen at the top soil layer (0-25 cm depth) was likely due to the fact that the greatest concentrations of soil organic matter inputs from litterfall are allocated in the top layer (0-30 cm soil depth) (Jobbagy and Jackson, 2000). In addition, nitrogen mineralization tends to be greater in upper soil surfaces, decreasing with soil depth (Charley and West, 1977). Soil nitrogen in the top soil layers could increase as a result of more intensive exploration of the soil by woody plant roots (Boutton and Liao, 2010). The deep-rooted woody plants can uplift and concentrate nutrients from deeper to shallow soil horizons (Jobbagy

and Jackson, 2000; 2004). Thus, the effect of western juniper trees on soil nitrogen is more pronounced in the top soil layer.

The interspace area in the untreated watershed also showed difference even though small in nitrogen stores with soil depth. In contrast, the interspace area in the treated watershed did not show difference in nitrogen stores with soil depth, reflecting the lower organic matter inputs that this area receives. Our evaluation of total nitrogen stores as affected by juniper elimination included the 0-50 cm soil layer because juniper roots may influence changes at that depth (Young et al., 1984). However, there were no differences in soil nitrogen stores as a result of juniper elimination (between watersheds) at either the 0-25 cm or the 25-50 cm soil layers.

## 5. Conclusions and Implications

Western juniper is rapidly increasing in cover and displacing grasslands in North America as a consequence of fire suppression. Juniper trees are commonly eliminated to restore the sagebrush steppe plant community. However, the implications of juniper cutting in terms of ecosystem nitrogen stores are largely unknown. We hypothesized that 13 years after cutting mature western juniper, aboveground and belowground nitrogen would have decreased in the treated watershed compared to the untreated watershed, implying lower capacity for soil nitrogen accumulation due to juniper cutting. We compared the ecosystem biomass and nitrogen allocation in an uncut juniper area (untreated watershed) with that in an adjacent cut juniper area (treated watershed). After 13 years of juniper removal, we observed that the aboveground biomass was greater in the untreated watershed and the belowground biomass was greater in the treated watershed. As a result, the aboveground nitrogen store was reduced in the treated western juniper area, but the belowground nitrogen store was not, resulting in no significant changes in total

nitrogen stores (aboveground and belowground). A greater root nitrogen accumulation in the treated area than in the untreated area partially counterbalanced the losses in aboveground nitrogen due to juniper cutting.

Ecosystem nitrogen was dominated by the belowground pool in both management practices. The greatest ecosystem nitrogen accumulation resides belowground (over 80%). Thus, changes in the 20% aboveground biomass can be of less relative importance in the short term. However, our 13-year post treatment study, is still of short duration to contemplate soil nitrogen changes. More research of longer duration about effects of western juniper encroachment and elimination on soil nitrogen is needed to further support findings from this study. Protecting belowground nitrogen source is of great importance. This study concludes that western juniper removal did not impact belowground nitrogen pools, at least 13 years post elimination. The restoration attempts of western juniper in central Oregon appear to represent more immediate favorable results for livestock and watershed hydrological functions. Thus, it is evident that juniper removal can help to improve the habitat quality for wildlife such as sage grouse and the forage productivity for moderate cattle grazing without changes in nitrogen accumulation.

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# WESTERN JUNIPER WATER UPTAKE AND SOIL MOISTURE RELATIONSHIPS: A PAIRED-WATERSHED STUDY IN CENTRAL OREGON, USA

**Abstract:**

In this study, we examined the interannual and seasonal variation in transpiration in relation to soil moisture with implementation of western juniper (*Juniperus occidentalis*) removal in two adjacent juniper-dominated experimental watersheds, wherein one watershed received treatment (treated) and the other served as untreated watershed. Juniper trees were instrumented with sap flow sensors to monitor plant water use in mature and juvenile growth stages in the untreated watershed as well as in a sapling stage in the treated watershed where juniper trees were removed in 2005 but regrowth has occurred. Leaf water potentials were monitored for juniper trees to support the data of transpiration. Results indicate that transpiration was greater in mature followed by juvenile and then saplings, supporting our expectation that water use consumption is a function of juniper's stage of development. For example, we found that mature juniper used around 43 and 72 times more water than juvenile and saplings, respectively during summer 2018. Predawn and midday leaf water potentials ranged from -1.0 to -2.6 MPa and -1.9 to -3.4 MPa, respectively during the study period (July 2018-September 2019) for all juniper stages. The significant differences between predawn and midday leaf water potentials for all juniper stages indicate that some degree of water was lost over the course of the day. The maximum range between these values in July months for 2018 and 2019 years corresponds with significantly higher transpiration for all juniper stages. Our findings also indicate that annual and seasonal precipitation was highly variable over the course of the study (2017-2019), which was reflected in the mean soil water content (0-80 cm). This resulted in considerable intra- and interannual variation in transpiration. Intra-annually there were two distinct seasonal pulses of transpiration by juniper: spring and summer. In years with heavy winter precipitation (2017 and 2019), juniper exhibited higher transpiration during summer seasons followed by spring and finally the fall months. In the year

with a lower summer precipitation (2018), juniper was capable of greater transpiration during the moist period (spring). Our results suggest that considerable amount of water can be saved due to western juniper management. In addition, our study highlights the sensitivity of western juniper woodlands to variations in seasonal precipitation and soil moisture availability.

**Keywords:** Central Oregon, Rangelands, Soil moisture, Transpiration, Water potential, Watershed, Western juniper.

## 1. Introduction

The replacement of grassland vegetation by encroaching woody species in rangelands is a global phenomenon occurring during the past century (Andela et al., 2013; Elkington et al., 2014; Archer et al., 2017). This progressive shift from grasses to woody species has altered ecosystem processes such as rates of soil carbon sequestration (Zhou et al., 2017), soil nutrient cycles (Hibbard et al., 2003) and hydrological processes (Huxman et al., 2005). Juniper (*Juniperus* spp) encroachment is one of the most large-scale changes that are occurring in North American rangelands (Baker and Shinneman, 2004; Sankey et al., 2010). The spatial distribution of juniper has increased by an order of magnitude since the mid-19th century throughout the Great Basin (Romme et al., 2009). Reported increases in areal extent have varied between 30% and 625% (Romme et al., 2009), and encroachment rates have varied between 0.4% and 4.5% per year (Sankey and Germino, 2008). The expansion of juniper is most frequently attributed to a mix of biophysical and anthropogenic factors including overgrazing, increases in the atmospheric carbon dioxide, and fire suppression (Soulé et al., 2004; Miller et al., 2005). Since the late 1890s western juniper (*Juniperus occidentalis* spp. *occidentalis* Hook.), the widespread juniper species has been

actively encroaching into grassland and sage steppe ecosystems of Oregon and has increased from 170,000 ha in 1936 (Cowlin et al., 1942) to more than 1.4 million ha (Azuma et al., 2005).

Western juniper dominance in sagebrush steppe has several negative consequences, including reductions in herbaceous production and diversity (Bunting et al., 1999; Ray et al., 2019), deterioration for wildlife habitat (Baruch-Mordo et al., 2013), and higher erosion and runoff potentials (Miller et al., 2005; Coultrap et al., 2008). Juniper expanding into rangeland ecosystems can limit the growth of understory vegetation, by outcompeting them for light, soil moisture and soil nutrients (Vaitkus and Eddleman, 1987). Juniper may increase evapotranspiration or reduce recharge by canopy interception, adversely affecting water yield; some studies attempting to quantify these effects have indicated negative effects of these woodlands (Eddleman et al., 1994; Thurow and Hester, 1997; Owens et al., 2006; Pierson et al., 2007; Petersen and Stringham, 2008; Mollnau et al., 2014; Ochoa et al., 2018) with others finding negligible effects of juniper on water yield (Dugas et al., 1998; Wilcox et al., 2005; Wilcox et al., 2006). Multiple studies indicate that western juniper elimination could have positive results on several hydrologic processes (Bates et al., 2000; Pierson et al., 2003; Mollnau et al., 2014; Ochoa et al., 2018). None of these studies address the effect of juniper reduction on water use.

In arid and semiarid regions, water is the major ecological resource that limits plant growth, and hydrological processes determine the ecological functioning of soil-plant systems (Li, 2011). A full understanding of the relationships and coupling mechanisms among soil, water and plants interactions are important for predicting the land surface development processes in the dry lands (Wang et al., 2012). The water potential of plants is the major driving force that moves the water through plants (Skaloš et al., 2014). Values of leaf water potential are translated in relation to the volume flux of water through plants and the pathway characteristics of water transfer from soil to

leaf (Jarvis, 1976). When leaf water potential is near zero, the driving force inducing water movement in the xylem is low, but at extremely negative water potentials cavitation strongly reduces hydraulic conductance, thereby in both cases transpiration is limited (Manzoni et al., 2013). Existing work on western juniper indicates that the daily average stand-level values of transpiration for these woodlands is 0.4 mm during the summer months (Mollnau et al., 2014) and 0.09-0.21 mm during the growing season (leaf level model, Miller et al. 1987). Different juniper species, mature Ashe juniper is found to transpire approximately 150 L day$^{-1}$ during the growing season, which is estimated to be equivalent to 400 mm year$^{-1}$ (Owens and Ansley, 1997). However, it is unknown if the hydrologic system (transpiration) is modified by with or without western juniper.

Specifically, water use by juniper growth stages, which potentially possess different rooting depth (McCole and Stem, 2007), has not been adequately studied. Because of this, it is not clear how much water are land managers saving by cutting juniper and how much of this savings are curtailed by the regrowth of sapling juniper trees that happens after years of juniper removal. When trees progress from juvenile to young mature stages, roots may reach increasingly soil moisture at deeper layers (Irvine et al., 2002). According to (West et al., 2007; Mollnau et al., 2014), mature juniper trees draw soil water from deeper soil depths because of their deep root system. Juniper trees remaining on a site after mature juniper elimination possess the potential to rapidly consume residual water resources due to reduced competition with mature trees (Moore and Owens, 2006). In addition, studies evaluating soil moisture dynamics in areas with and without western juniper continue to increase (Ochoa et al., 2018, Ray et al., 2019), but soil moisture depletion in relation to transpirational water loss patterns under different management practices of treated and untreated western juniper is unknown.

Sap flow (SF) methods are increasingly used to quantify whole-plant water use, especially in woody plants (Smith and Allen, 1996; Wullschleger et al., 1998). Both water balances and micrometeorology can provide estimates of evapotranspiration, which includes soil evaporation and plant water use, while SF techniques measure transpiration directly (Allen and Grime, 1995; Chabot et al., 2005; Kool et al., 2014). The accurate monitoring of plant water uptake in arid and semiarid environments considering the inherently variable nature in moisture availability (Loik et al., 2004), requires long-term, continuous studies of plant transpiration (West et al., 2008). SF techniques are robust in monitoring transpiration continuously from component species of an ecosystem over several years, with an adequate frequency to detect responses to environmental conditions (Granier et al., 1996).

This study aimed to determine seasonal transpiration for western juniper in relation to soil moisture in one juniper-dominated watershed and in another watershed where juniper has been removed, with a major goal of improving scientific understanding of the effects of juniper encroachment and removal on hydrology.

## 2. Materials and Methods

### 2.1. Description of Study Site

This study took place in the Camp Creek-Paired Watershed Study (CCPWS) site, 27 km northeast of Brothers, Oregon. The CCPWS site is a long-term collaborative research project located (43.96° Lat.; −120.34° Long) in central Oregon (Deboodt, 2008). The CCPWS site encompasses an area of approximately 212 ha and includes two adjacent watersheds, one treated (116 ha, Mays WS) and the other untreated (96 ha, Jensen WS) (Fig. 4.1) with elevations ranging from 1370 m to 1524 m. Dominant overstory vegetation in Jensen WS is western juniper. In the

fall of 2005, approximately 90% of the western juniper trees were cleared in Mays WS using chain saws, leaving only old-growth trees intact and a big sagebrush (*Artemisia tridentate*) as dominant overstory vegetation. Following western juniper removal on Mays WS, changes in vegetation composition have been reported, including western juniper regrowth (sapling juniper trees) and greater presence of shrubs and grasses (Ray et al., 2019). The average slope for each watershed is around 25% with similar distribution of aspects (Fisher, 2004). Prior to juniper elimination from Mays WS, juniper occupied 27% cover in the whole area, which is near the 30% cover described for Phase (III) juniper sites (Miller et al., 2005).

The average annual precipitation (2009-2017) at the study site is 358 mm (Ochoa et al., 2018). Most precipitation in the area occurs as a mix of rain and snow between October and March, with sporadic rainfall events occurring during spring and summer. The two watersheds were instrumented to monitor multiple hydrologic variables including precipitation, soil moisture, and transpiration. Total precipitation (mm) for the days corresponding to soil moisture data collection was recorded by onsite instrumentation (Ochoa et al., 2018).

Three (3) primarily soil series, Westbutte, Madeline, and Simas were found to define most of the soil types in both watersheds. Westbutte very stony loam and Madeline loam, the two main soil types, were found to comprise about 70% to 74% of the study area (Fisher, 2004). Simas, gravelly silt loam accounted for the final portion with additional soil series occupying <1%. The Westbutte series is classified as loamy-skeletal, mixed, superactive, frigid Pachic Haploxerolls. The Madeline series is classified as clayey, smectitic, frigid Aridic Lithic Argixerolls. The Simas series is classified as fine, smectitic, mesic Vertic Palexerolls. The Westbutte and Madeline series consist of moderately shallow to deep, well-drained soils, formed of colluvium derived from basalt, tuff and andesite. The Simas series consists of very deep, well drained soils, formed of loess

and colluvium derived from tuffaceous sediments (OSDs). The Jensen WS mainly consists of 48% Madeline, 26% Westbutte, and 21% Simas series, while the Mays WS consists of 50% Westbutte, 20% Madeline, and 3% Simas series (Fisher, 2004).
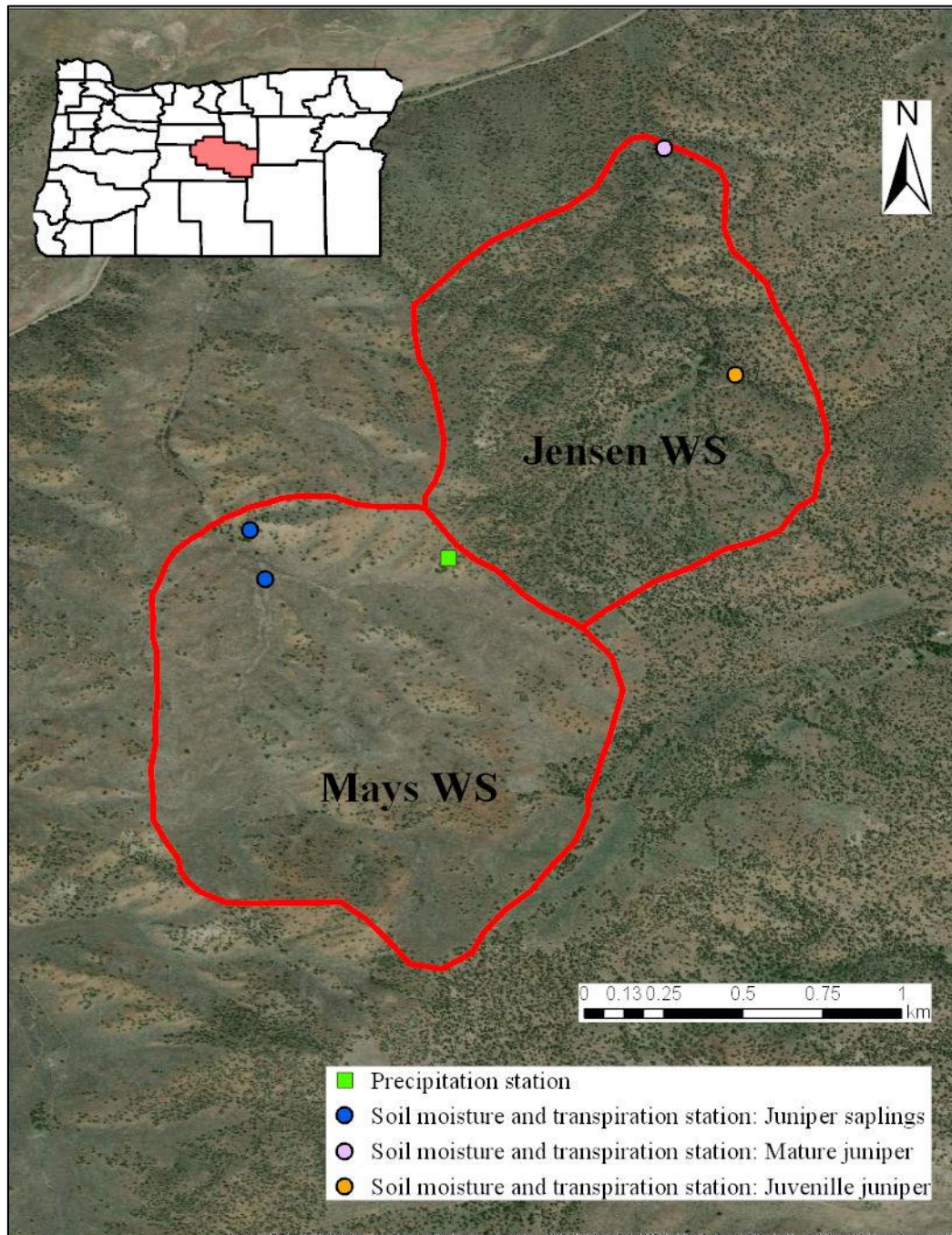


Figure 4.1. Map of the study area showing Mays and Jensen WS, indicating the locations of different monitoring instrumentation used in this study.

*2.2. Field Data Collection*

2.2.1. Soil Physical Properties

In May 2019, soil samples for characterizing soil texture, water content, bulk density and porosity were collected from a close distance to the soil moisture sensor installation in both watersheds. At each sensor depth (Fig. 4.2), three soil cores for bulk density, water content and porosity, and one loose soil sample for textural classification were obtained. Soil texture was determined using the hydrometer method described by Gee and Bauder (1986). Soil cores were weighed, *oven*-dried at 105°C for 48 hours and reweighed to determine gravimetric water content (*GWC*). Bulk density (*BD*) was calculated as the ratio of the mass of oven-dried soil sample to core volume (Mg m$^3$). The volumetric water content (*VWC*) was computed using the formula;

$$VWC = (GWC)(BD) \tag{1}$$

Soil porosity (*%PS*) was determined using the formula:

$$\%PS = 100\left(1 - \frac{BD}{PD}\right) \tag{2}$$

where

*PD*= particle density, value is very near to 2.65 Mg m$^3$.

The three soil cores used for bulk density and water content determinations were thoroughly mixed to produce a representative sample for each sensor depth and location. The characteristic water retention curve of each soil mix was determined at the Soil Physics Laboratory of Oregon State University. Soil matric potential was determined for each representative sample using a pressure plate apparatus and dew point meter (Gardner, 1965; Bittelli and Flury, 2009). Two important soil water capacity parameters, soil moisture at field capacity ($\theta$ FC) and permanent wilting point ($\theta$ PWP) were determined to understand the potential of soil water availability for

plant growth. Then, we determined the available water content (AWC) from $\theta$ FC- $\theta$ PWP (Waller and Yitayew, 2016) (Table A1).
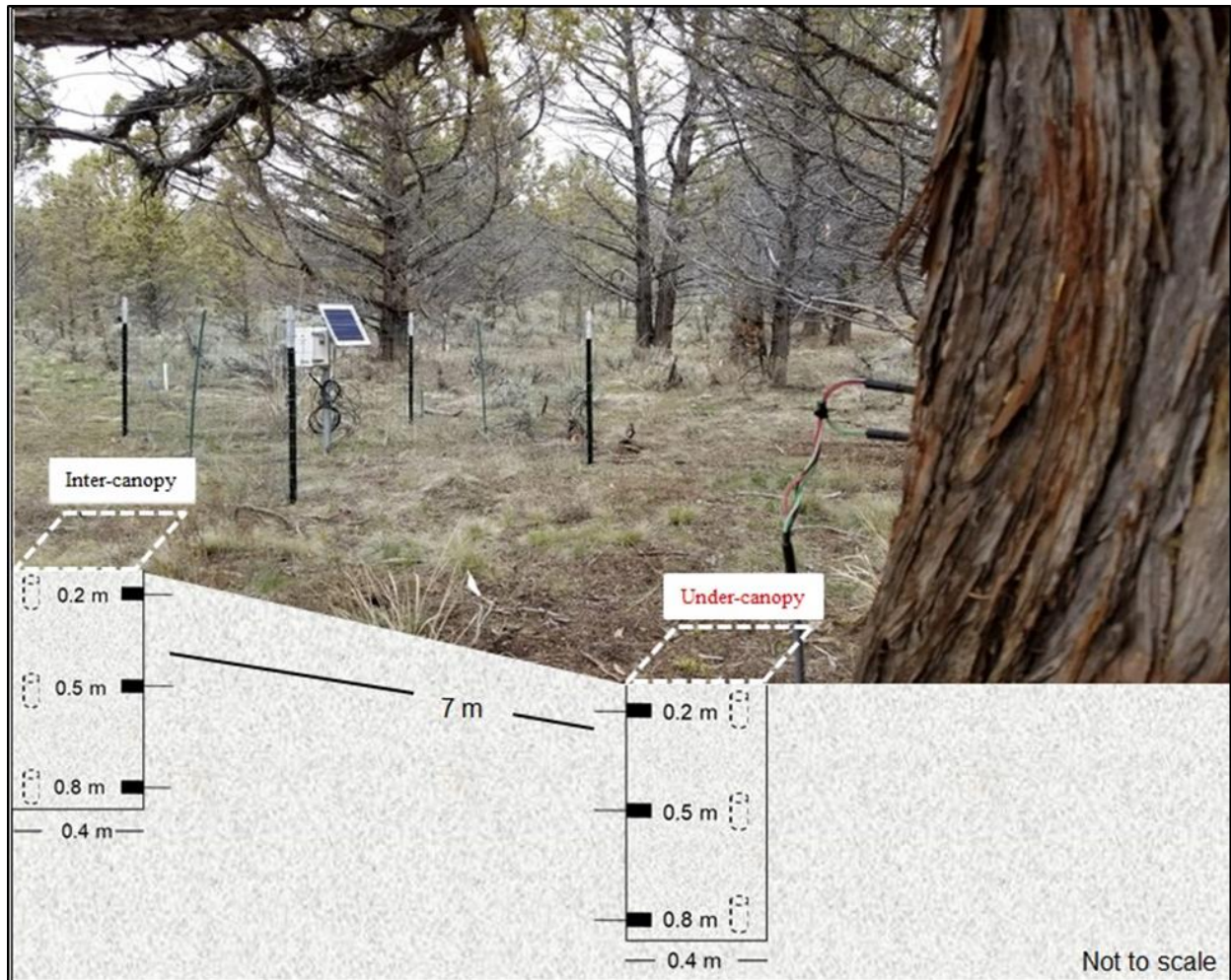


Figure 4.2. Schematic illustrating soil moisture sensor placement, soil sampling depth at under-canopy and inter-canopy locations, and a thermal dissipation probe installation for transpiration measurement in the valley location in Jensen WS.

2.2.2. Sap Flow Measurements and Transpiration Estimates

The two watersheds were instrumented to monitor western juniper sap flow. Sap flow was determined in two manners: 1) for saplings in the (Mays-East) and (Mays-West) locations, and medium trees in size (juvenile) juniper trees in the (Jensen-Upslope) location using the stem heat

balance (SHB) technique (Baker and Van Bavel, 1987; Smith and Allen, 1996) and 2) for large mature juniper trees using the thermal dissipation probe (TDP) technique (Smith and Allen, 1996; Bladon et al., 2006; Davis et al., 2012) in the (Jensen-Valley) location. Sap flow data were obtained for nine different periods for sapling juniper including trees in (Mays-East) and (Mays-West) locations from late June 2017 through October 2019. Sap flow data for juvenile trees were obtained for five different periods in (Jensen-Upslope) location from April 2018 through September 2019. For mature juniper, sap flow was obtained for eight different periods late May 2017 to September 2019. There were numerous days with incomplete data collection because of battery failure.

Four (4) sapling trees were equipped with dynamax sap flow gauges (Chabot et al., 2005) in the Mays WS. Two SHB gauges (Models SGB16, SGB19, Dynamax, Houston, Tx, USA) were equipped each in saplings (S1 and S2 trees) respectively in selected branches to measure sap flow (Grime and Sinclair, 1999) in the (Mays-East) location. To scale sap flow from a branch to whole tree, we used reference unit technique by (Kirmse and Norton, 1985). The branches containing sap flow gauges were designated as reference units and their lengths were determined. We determined and added the lengths of each individual branch including the main stem. We estimated the percentage of the reference branch to the whole tree. The other two saplings (S3 and S4 trees) were equipped each with SHB gauge (Model SGB25, Dynamax, Houston, Tx, USA) in their main stems in the (Mays-West) location.

In addition, two (2) juvenile (J1 and J2 trees) were equipped each with SHB gauge (Model SGB35, Dynamax, Houston, Tx, USA) in their main stems in the (Jensen-Upslope) location. The diameters of the equipped stems/branches and related parameters are reported in (Table. 4.1). The SHB gauges comprising a flexible heater that itself is encircled by foam insulation were wrapped around branches/main stems (Smith and Allen, 1996). Gauges were wrapped in aluminum foil to

shield from rain and direct solar radiation. The SHB instrument description and theory of operation are described in detail by Steinberg et al. (1989), Lascano et al. (1992) and Smith and Allen. (1996). Measurements were collected every 15 minutes and recorded in dataloggers (CR10X Campbell Scientific, Inc., Logan, Utah). The sap flow rate ($F$) in the stems/branches in (g h$^{-1}$) was calculated using the following equation (Grime and Sinclair, 1999).

$$F = \frac{3600(Q_f)}{C_p(T_d - T_u)} \tag{3}$$

where:

$Q_{f\,=}$ The convective heat flux carried by the sap in Watt;

$C_p$ = The specific heat capacity of xylem sap, assumed to be that of water (4.186 J g$^{-1}$ k$^{-1}$);

$T_d$ and $T_u$ = Temperatures in Celsius measured at an equal distance downstream ($d$) and upstream ($u$) from the heated stem.

Table 4.1. Characteristics of the sapling and juvenile tree samples used for sap flow determination.

| Tree no | SHB Sensor Model | Tree Height (m) | Maximum Width (m) | Equipped Stem/Branch Diameter (mm) | Equipped Stem/Branch Area (mm²) |
|---------|------------------|-----------------|-------------------|-------------------------------------|----------------------------------|
| S1 | SGB16 | 1.55 | 0.70 | 15 | 177 |
| S2 | SGB19 | 1.30 | 1.05 | 21 | 346 |
| S3 | SGB25 | 1.50 | 0.80 | 28 | 615 |
| S4 | SGB25 | 1.40 | 0.52 | 28 | 615 |
| J1 | SGB35 | 2.90 | 0.89 | 40 | 1256 |
| J2 | SGB35 | 3.75 | 1.14 | 40 | 1256 |

Measurements of sap flow for the mature juniper trees in the (Jensen-Valley) location were made using commercially available TDP probes (Model TDP-30, TDP-50, Dynamax, Inc., Houston, TX). Initially, seven (7) mature (M1 to M7 trees) were used for sap flow measurements (2 trees installed with TDP-50 and 5 trees installed with TDP-30). Thermocouple needles 1.2 and 1.65 mm in diameter (TDP-30 and TDP-50, respectively) were installed as a vertically aligned pair 40 mm apart within the sapwood of each tree (Dynamax manual). Sets of some probes were implanted on the north and south side of each tree at around 1-2 m above the ground. Plastic putty was installed around the needles for water protection and foam quarter-spheres were tightly secured on both sides of the needles to protect the wire from bending stress and to provide thermal insulation to the needles (Gazal et al., 2006). Reflective bubble wrap was installed around the tree boles for additional insulation. When necessary, probes were relocated to different trees (West et al., 2008). The general methodology is described by (Köstner et al., 1998; Lu, 1997; Lu et al., 2004). Sap flow data were sampled at a 5-minutes interval and recorded by a data logger (CR10X, Campbell Scientific, Logan, UT, USA) (Elkington et al., 2014). The raw data output was graphically assessed for individual sensors.

Practically, probe pair (two needles) linked via thermocouples were installed in the xylem. The upper probe was connected to a power source with a current constantly inducing heat into the sap stream while recording the temperature of the passing sap. The lower probe remained unheated. The temperature difference ($dT$) between the heated and unheated probes reflects the amount of water passing the probes (Mollnau et al., 2014). When sap flow occurs, the sap transports part of the heat and therefore the temperature of the heated probe drops (Ringersma et al., 1996). When no flow occurs, the $dT$ reaches a maximum value (Ringersma et al., 1996). The value of maximum temperature difference ($dTm$), necessary for calculating sap flow is determined for each probe pair

from the *dTm* measured during the night period when transpiration is assumed to equal zero (Ringersma et al., 1996). Dimensionless parameter ($K$) was calculated using the following equation (Granier, 1985; 1987).

$$K = \frac{(dTm - dT)}{dT} \tag{4}$$

Average sap flow velocity (cm s$^{-1}$) was calculated using the following equation (Gazal et al., 2006).

$$V = (0.0119)(K^{1.231}) \tag{5}$$

Sap flow velocity then was converted to sap flow rate (g hr$^{-1}$) using the equation.

$$F = (V)(SA)(3600) \tag{6}$$

where:

$SA$ = Sapwood cross-sectional area (cm$^2$).

Leaf area was calculated for each tree using the following pre-established equation developed by Miller et al. (1987).

$$Y = -35.036 + 2.296(X) \tag{7}$$

where:

$Y$ = Leaf area (m$^2$);

$X$ = Basal circumference (m).

Then, sapwood area (Table 4.2), which is a necessary parameter to calculate whole-tree water use, was estimated using the following equation (Miller et al., 1987).

$$Y = 8.145 + 0.155(X) + 0.00005(X^2) \tag{8}$$

where:

$Y$ = Leaf area (m$^2$);

$X$ = Sapwood area (cm$^2$).

The sap flow data was calculated daily in terms of volume (L day$^{-1}$). Multiple comparasion analyses (Bonferroni t-test) were performed to compare the transpiration means of two trees or more within each juniper growth stage and across various time periods.

Table 4.2. Characteristics of the mature tree sample used for sap flow determination.

| Tree no. | TDP Sensor Model (mm) | Basal Diameter (m) | Basal Circumference (m) | Tree Diameter at Sensor Placement (m) | Leaf Area (m$^2$) | Sap Wood Area (m$^2$) |
|---|---|---|---|---|---|---|
| M1 | 50 | 0.87 | 2.85 | 0.70 | 619.3 | 0.1118 |
| M2 | 50 | 0.75 | 2.47 | 0.60 | 532.5 | 0.1023 |
| M3 | 30 | 0.52 | 1.74 | 0.38 | 363.8 | 0.0811 |
| M4 | 30 | 0.37 | 1.27 | 0.28 | 256.1 | 0.0649 |
| M5 | 30 | 0.52 | 1.74 | 0.37 | 364.5 | 0.0812 |
| M6 | 30 | 0.52 | 1.73 | 0.43 | 361.0 | 0.0807 |
| M7 | 30 | 0.40 | 1.39 | 0.30 | 284.1 | 0.0694 |

2.2.3. Leaf Water Potential Measurements

Leaf water potential ($\Psi$) measurements were used to predict water use for juniper. Lower water potential is an index of soil water availability in the rooting zone (Bates et al., 2000). Plant water potential measurements were made for 2-3 days in the middle of each month from July 2018 to September 2019 using a Scholander pressure chamber (Scholander et al. 1965) (PMS Instruments, Corvallis, OR). The measurements were made predawn ($\Psi_{pd}$; between 03:00 - 05:00 am and completed before sunrise) and midday ($\Psi_{md}$; approximately 12:30 pm). The measurements

were taken from the trees monitored with sap flow sensors. Vegetative stems (n=3-5) were clipped from each tree using small scissor during each sampling period. The measurements were made in the field immediately after clipping the stems. Vegetative stems for two mature trees were not accessible, so only five mature trees were measured. Differences between $\Psi_{pd}$ and $\Psi_{md}$ for all juniper growth stages were evaluated with a paired Student's t-test.

2.2.4. Soil Moisture Content

In Jensen WS, two monitoring stations, one was installed for the location of juvenile trees in the upslope (Jensen-Upslope) and the other one in the location of the mature trees in the valley (Jensen-Valley), were installed to collect hourly soil moisture content ($\theta$) data. One network was placed in the under-canopy zone and one in the inter-canopy zone. The under-canopy $\theta$ sensors were located 1 m northwest of the trees in the (Jensen-Upslope) location and 1 m north of the mature tree#1 in the (Jensen-Valley) location. The inter-canopy $\theta$ sensors were located 7 m north of the under-canopy $\theta$ sensors in both juvenile and mature juniper locations.

In Mays WS, two monitoring stations, one was installed for the location of sapling trees in the Mays valley east-facing slope (Mays-East) and the other one in the location of sapling trees in the Mays valley west-facing slope (Mays-West), were installed to collect hourly $\theta$ data. One network was placed in the inter-canopy zone. The $\theta$ sensors in each network were installed at 0.2, 0.5, and 0.8 m soil depth. For the sapling and juvenile juniper locations, model CS655 sensors (Campbell Scientific, Inc., Logan, UT, USA) were used. For the mature juniper location, HydraProbe sensors (Stevens Water Monitoring Systems, Inc., Portland, OR, USA) were used. A Kruskal-Wallis One Way Analysis of Variance (ANOVA) on Ranks test was performed to assess $\theta$ variability between inter-canopy and under-canopy zones in each location in the Jensen WS. Daily-averaged $\theta$ values

at each sensor depth were used in this analysis. SigmaPlot® version 14.0 (Systat Software, Inc., San Jose, CA, USA) was used for all statistical analyses.

*2.3. Transpiration and Soil Moisture Relations*

To better understand $\theta$ control on the temporal dynamics of SF, we simultaneously measured SF for western juniper and $\theta$ at three depths (0.2, 0.5, 0.8 m) in the locations of trees containing SF sensors. Sensor outputs for SF were scaled to daily flow rates (L d$^{-1}$). VWC (m$^3$/m$^3$) at the three depths was measured hourly using three soil moisture probes. In the analysis, we used daily averages of VWC from the three soil moisture probes. All depths were averaged to determine mean soil water content for the entire soil profile ($\theta_{tot}$). We used multiple polynomial regression analyses to understand the responses of juniper water uptake to soil $\theta_{tot}$ at the three juniper growth stages (sapling, juvenile, and mature).

**3. Results**

*3.1. Transpiration—Juniper Growth Stages Scale*

3.1.1. Sapling (Regrowth) Juniper in Mays WS

For the sapling trees in (Mays-East) location, the percentages of the reference branches to the whole trees were estimated to be 10.3% and 16.6% for S1 and S2 respectively and sap flow was scaled from branches to the whole trees. In the onset recorded period of sap flow (the summer 2017 months), transpiration did not differ ($P = 0.06$) between S1 and S2 trees with an average of 2.76 L d$^{-1}$ ($\pm$ 0.07) (Table 4.3). Following that, a significantly sharp decline in transpiration occurred during early fall 2017 with an average of 0.43 L d$^{-1}$ ($\pm$ 0.07). In 2018, there was no transpirational difference ($P = 0.06$) between spring and summer seasons with an average of 1.00

L d$^{-1}$ ($\pm$ 0.068) and 0.79 L d$^{-1}$ ($\pm$ 0.03) respectively. In the summer 2018, transpiration was 1.3 times greater for saplings in (Mays-East) location than saplings in (Mays-West) location, with S2 having significantly ($P \leq 0.05$) greater water use compared to the other saplings. From October 2018 through March 2019, what can be considered the wet season, transpiration was significantly the lowest with no significant ($P \geq 0.05$) differences between fall 2018 and winter 2019 seasons. In spring 2019 (April-June), trees became active and transpiration was observed to be significantly ($P \leq 0.05$) greater than the wet season, with an average of 0.74 L d$^{-1}$ ($\pm$ 0.04). Transpiration was 1.4 times significantly ($P \leq 0.05$) greater in the summer (1.02 $\pm$ 0.04 L d$^{-1}$) compared to spring for the year 2019. In early fall 2019, transpiration declined significantly reaching the minimum values, with an average of 0.42 ($\pm$ 0.04).

Table 4.3. Sapling juniper transpiration results in Mays watershed, showing the average water uptake and the number of observed days from late June 2017 through October 2019. N/A = Data not available. Means with different lowercase letters (a, b, c) along rows indicate significant differences ($P \leq 0.05$) in transpiration. Means with different capital letter (A, B, C, D, E) along columns indicate significant differences ($P \leq 0.05$) in the average transpiration among different time periods.

| Period of Record | | | | | Average (L d$^{-1}$) |
|---|---|---|---|---|---|
| Sapling Juniper Trees | S1 | S2 | S3 | S4 | |
| 27 June to 22 September 2017 | 2.59(43) a | 2.93(30) a | N/A | N/A | 2.76(51) A |
| 8 October to 13 November 2017 | 0.26(22) b | 0.60(27) a | N/A | N/A | 0.43(28) D |
| 15 April to 30 June 2018 | 0.70(31) b | 1.30(29) a | N/A | N/A | 1.00(32) BC |
| 1 July to 30 September 2018 | 0.46(72) c | 1.32(71) a | 0.68(51) b | 0.71(52) b | 0.79(82) C |
| 1 October to 31 December 2018 | 0.12(56) c | 0.29(54) ab | 0.21(65) bc | 0.30(68) a | 0.23(81) E |
| 1 January to 31 March 2019 | 0.12(81) c | 0.23(84) ab | 0.23(79) b | 0.28(81) a | 0.22(88) E |
| 1 April to 30 June 2019 | 0.55(39) bc | 1.13(87) a | 0.74(80) b | 0.53(73) c | 0.74(87) C |
| 1 July to September 2019 | 0.79(78) c | 1.55(85) a | 0.98(82) b | 0.75(63) c | 1.02(92) B |
| 1 to 31 October 2019 | 0.14(30) b | 0.70(31) a | N/A | N/A | 0.42(31) D |

The results from the ANOVA test showed that transpiration was significantly ($P \leq 0.05$) greater in S2 compared to the other trees for most time periods. Across seasonal periods, transpiration was significantly ($P \leq 0.05$) the greatest in summer 2017 followed by spring 2018 and summer 2019, and significantly ($P \leq 0.05$) lowest in fall 2018 and winter 2019.

3.1.2. Juvenile (Medium Growth Stage) Juniper in Jensen WS

Likewise, as observed in saplings, transpiration was not different ($P = 0.33$) between spring and summer 2018 seasons with a daily average of 1.61 L d$^{-1}$ ($\pm$ 0.09) and 1.33 L d$^{-1}$ ($\pm$ 0.08) respectively (Table 4.4). Transpiration decreased suddenly and bottomed out, 0.27 L d$^{-1}$ ($\pm$ 0.02) during fall 2018 corresponding with a reduction in evaporative demand. In late spring 2019 (May to June), transpiration by juvenile increased with a daily average of 1.76 L d$^{-1}$ ($\pm$ 0.15). During summer 2019, the increase in transpiration was maintained, averaging 2.00 L d$^{-1}$ ($\pm$ 0.11) with no statistical differences ($P = 0.58$) compared to spring 2019. Similar to the saplings, water consumption was not different ($P = 1.00$) in spring season for the years 2018 and 2019.

Table 4.4. Juvenile juniper transpiration results in Jensen watershed, showing the average transpiration and the number of observed days from late April 2017 through September 2019. N/A = Data not available. Means with different lowercase letters (a, b) along rows indicate significant differences ($P \leq 0.05$) in transpiration. Means with different capital letter (A, B, C, D) along columns indicate significant differences ($P \leq 0.05$) in the average transpiration among different time periods.

| Period of Record | | | Average (L d$^{-1}$) |
|---|---|---|---|
| Juvenile Juniper Trees | J1 | J2 | |
| 22 April to 30 June 2018 | 1.79(52) a | 1.42(53) b | 1.61(55) BC |
| 1 July to 17 September 2018 | 1.39(66) a | 1.26(67) a | 1.33(67) C |
| 16 October to 16 December 2018 | 0.25(29) a | 0.28(29) a | 0.27(29) D |
| 17 May to 28 June 2019 | 1.69(42) a | 1.83(42) a | 1.76(42) AB |
| 14 July to 15 September 2019 | 1.66(49) b | 2.35(51) a | 2.00(52) A |

The results from the ANOVA test showed there was no difference ($P \geq 0.05$) in transpiration between juvenile trees for most time periods.

### 3.1.3. Mature Juniper in Jensen WS

In the late spring 2017 (May to June), the daily-averaged transpiration was 89.6 L d$^{-1}$ ($\pm$ 5.80) and was significantly different ($P \leq 0.05$) between the first two sap flow monitored trees (Table 4.5). Similar to the saplings, water consumption by mature was significantly ($P \leq 0.05$) the greatest in the summer (July to September 2017) with a daily average of 115.2 L d$^{-1}$ ($\pm$ 3.56).

Table 4.5. Mature juniper transpiration results in Jensen watershed, showing the average transpiration and the number of observed days from late May 2017 through September 2019. N/A = Data not available. Means with different lowercase letters (a, b, c, d, e) along rows indicate significant differences (P $\leq$ 0.05) in transpiration. Means with different capital letter (A, B, C, D, E) along columns indicate significant differences (P $\leq$ 0.05) in the average transpiration among different time periods.

| Period of Record | | | | | | | | Average (L d$^{-1}$) |
|---|---|---|---|---|---|---|---|---|
| Mature Juniper Trees | M1 | M2 | M3 | M4 | M5 | M6 | M7 | |
| 28 May to 30 June 2017 | N/A | N/A | 75.1(30) b | 104.0(32) a | N/A | N/A | N/A | 89.6(32) B |
| 1 July to 30 September 2017 | 93.7(13) b | N/A | 117.4(90) b | 134.5(91) a | N/A | N/A | N/A | 115.2(91) A |
| 1 to 20 October 2017 | 95.7(16) a | N/A | 36.9(15) b | 52.9(16) b | N/A | N/A | N/A | 61.8(16) CD |
| 22 April to 30 June 2018 | 138.9(46) a | N/A | 52.5(65) cd | 78.1(67) b | 34.1(16) d | 66.3(69) b | 66.5(39) bc | 72.7(69) BC |
| 1 July to 30 September 2018 | 83.5(38) a | 73.9(46) a | 37.4(84) b | 71.2(90) a | 44.2(73) b | 47.1(90) b | 42.0(83) b | 57.0(90) D |
| 1 to 14 October 2018 | 48.7(4) a | N/A | 8.3 (2) c | 17.7(6) bc | 23.7(6) b | 16.4(6) bc | 11.0(2) bc | 21.0(6) E |
| 10 April to 30 June 2019 | 106.4(43) a | N/A | 37.4(59) cd | 77.3(60) b | 44.2(52) c | 31.7(60) de | 24.1(58) e | 53.5(61) D |
| 1 July to 27 September 2019 | N/A | N/A | 44.4(82) c | 91.9(71) a | 63.1(81) b | 41(78) c | N/A | 60.1(85) D |

The average amount of water use significantly ($P \leq 0.05$) dropped toward the progression of early fall 2017. In contrast to sapling and juvenile trees, water uptake by mature was about 1.3 times significantly ($P \leq 0.05$) greater in spring (72.7 $\pm$ 2.59 L d$^{-1}$) than summer (57.0 $\pm$ 2.49 L d$^{-1}$) for the year 2018. Similar to juvenile, transpiration sharply declined and was significantly ($P \leq 0.05$)

the lowest in fall 2018 across the observed seasonal periods. In addition, mature juniper used same amount of water in spring ($53.5 \pm 2.15$ L d$^{-1}$) and summer ($60.1 \pm 1.42$ L d$^{-1}$) for the year 2019, similar to juveniles and different than saplings. The results from the ANOVA test showed that transpiration was significantly ($P \leq 0.05$) greater in M1 compared to the other trees for most time periods, which may be explained by the greater leaf area (Table 4.2).

## 3.2. Leaf Water Potential

For all juniper stages, $\Psi_{pd}$ gradually increased (more negative) from July-September 2018 (Fig 4.3), reaching as high as -2.6 MPa ($\pm 0.18$) for the juvenile trees, reflecting lower water availability. According to Miller and Shultz. (1987), there was a gradual increase in $\Psi_{pd}$ with the progression of summer drought, even though soil water remained available in the lower depths, and juniper roots were observed to be well distributed throughout the soil profile. Due to freezing or near-freezing air temperatures, $\Psi_{pd}$ measurements were not taken during the period from October 2018 to April 2019. In the year 2019, $\Psi_{pd}$ tended to significantly ($P \leq 0.05$) decline (less negative) from May-July for all juniper stages, indicating a gradual increase in transpiration peaking in July. Following that, a significantly steep rise in $\Psi_{pd}$, accompanied with less water consumption occurred from July-September.

Midday water potentials for the saplings ranged from a high (more negative) of -2.9 MPa in September for both years ($\pm 0.05$ and $\pm 0.03$; 2018 and 2019 respectively) to a low (less negative) of -1.9 MPa ($\pm 0.06$) in May 2019. In contrast to the saplings, $\Psi_{md}$ for juvenile ranged from a high of -3.7 MPa ($\pm 0.23$) in November 2018 to a low of -1.9 MPa ($\pm 0.01$) in February 2019. Like the juveniles, $\Psi_{md}$ for mature was most negative in November 2018, -2.9 MPa ($\pm 0.08$), but least negative in June 2019, -2.0 MPa ($\pm 0.03$). In general, $\Psi_{md}$ for all juniper growth stages

tended to be highest during late summer and early fall, when western juniper constantly started to use less water. The general trend of decline in $\Psi_{md}$ (less stress) in late winter and spring 2019 is an indication of juniper activity to transpire.
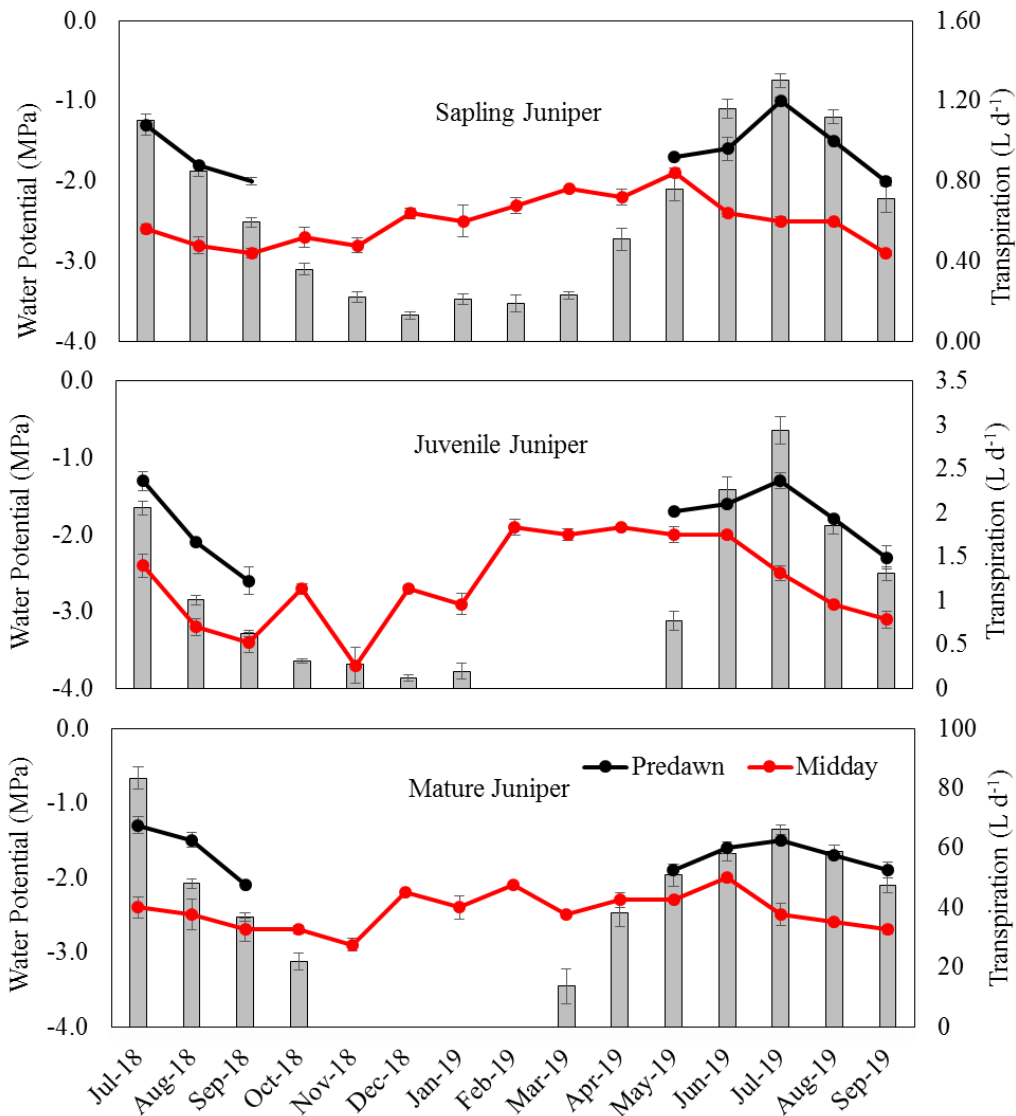


Figure 4.3. Monthly leaf water potential at predawn (least water stressed) and midday (most water stressed), and transpiration estimate for western juniper growth stages; sapling, juvenile, and mature. Data points are means ± standard error.

Paired t-tests showed that there were significant differences between $\Psi_{pd}$ and $\Psi_{md}$ for all juniper stages, indicating some degree of water loss over the course of the day. This is consistent

with West. (2006) who found similar findings for *Juniperus osteosperma*. Western juniper exhibited a comparable range in $\Psi_{pd}$ and $\Psi_{md}$ throughout the study. During a part of the growing season (May-July), $\Psi_{pd}$ and $\Psi_{md}$ values gradually diverged corresponding with an increase in transpiration peaking in July (Fig 4.3). However, these values started to diverge less during summer (July-September) when transpiration started to gradually decline.

*3.3. Soil Moisture Variability- Watershed Scale*

3.3.1. Mays WS

Figure 4.4 illustrates the seasonal pattern of daily-averaged $\theta$ fluctuations collected from the monitoring stations installed in Mays WS at two locations. In (Mays-East) location, $\theta$ values were generally greater during spring and lower during the dry seasons, reaching their lowest values in fall and early winter. For (Mays-East) location, the highest $\theta$ values at 0.2 m depth (~30-32%) were recorded in the months of March for all years. In addition, the highest $\theta$ values at sensor depths 0.5 and 0.8 m were recorded in the months of March but just for years 2017 and 2019. In the year 2018, the month of April had the highest $\theta$ values for 0.5 m depth (26%) and for 0.8 m depth (17.4%). In the (Mays-East) location, there was no significant difference ($P = 0.56$) in $\theta$ in the first two sensor depths in winter 2017 (January-March). The top soil depth (0.2 m) had significantly ($P \leq 0.05$) the greatest $\theta$ in winter seasons (22.5 $\pm$ 0.52% and 22.5 $\pm$ 0.58%, respectively) for the years 2018 and 2019. Throughout spring and summer seasons (April-September) in all years, there were more spatial variations in $\theta$ with significantly ($P \leq 0.05$) greater values at 0.5 m depth and lower values at 0.8 m depth. In the fall (October-December), there were less spatial variations among $\theta$ sensor depths with significantly ($P \leq 0.05$) greater values also at 0.5 m depth and apparent lower values at 0.2 m depths.

In the (Mays-West) location, the surface depth and deeper depth had equally greater $\theta$ values with no statistical difference ($P = 0.61$) in fall 2018 (Fig 4.4). Likewise (Mays-East) location, in winter 2019 the top soil had significantly ($P \leq 0.05$) the greatest $\theta$ (15.5 $\pm$ 0.51%). The subsurface depths, 0.5 and 0.8 m responded equally to precipitation, with no significant difference ($P = 1.00$) in winter 2019. By April to early fall 2019, $\theta$ values were significantly ($P \leq$ 0.05) greater at 0.2 m with a remarkable difference in October.
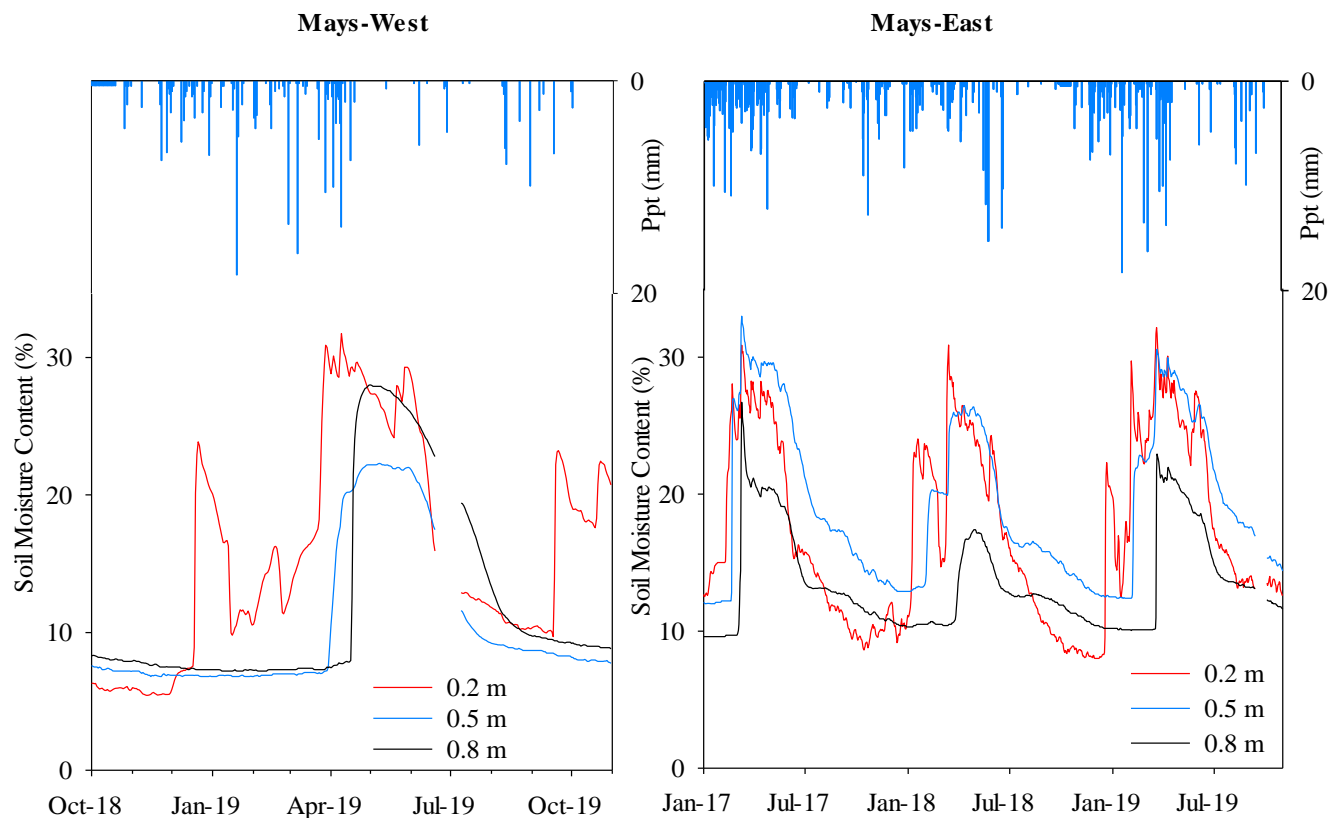


Figure 4.4. Daily-averaged preceipitation and soil moisture content fluctuations at different soil depths at inter-canopy zones in both locations in Mays WS. The (Mays-East) location is during January 2017 through October 2019 and the (Mays-West) location is during October 2018 through October 2019.

3.3.2. Jensen WS

Similar to Mays WS, $\theta$ levels responded to precipitation variability seasonally in both locations in the Jensen WS (Fig 4.5). The dynamics of $\theta$ followed similar patterns to Ochoa et al. (2018) with greater $\theta$ values recorded in the spring and lower $\theta$ values recorded in the fall and early winter seasons for both locations. The significantly ($P \leq 0.05$) lower $\theta$ in all the times in the deeper soil profile (0.8 m) at the under-canopy zone compared to the corresponding depth at the inter-canopy zone in the (Jensen-Valley) location may indicate the potential impacts of mature juniper water uptake on $\theta$ at that depth. Additionally, in the (Jensen-Valley) location the greater $\theta$ levels recorded at the inter-canopy zones for both the 0.5 and 0.8 m sensor depths compared to the corresponding depths in the under-canopy zones may be as a result of the significantly ($P \leq 0.05$) greater sand content values at these depths (Table 4.A1). Greater $\theta$ levels at the 0.2 m depth were generally observed most of the time for the under-canopy zones in both locations, probably because of higher porosity in the upper soil layers (Table 4.A1). The results from the ANOVA test showed there are significant $\theta$ differences ($P \leq 0.05$) between under-canopy and inter-canopy zones for most sensor depths in both locations. However, at the (Jensen-Upslope) location there was no significant difference ($P = 0.42$) in $\theta$ at 0.8 m depth between inter-canopy and under-canopy.
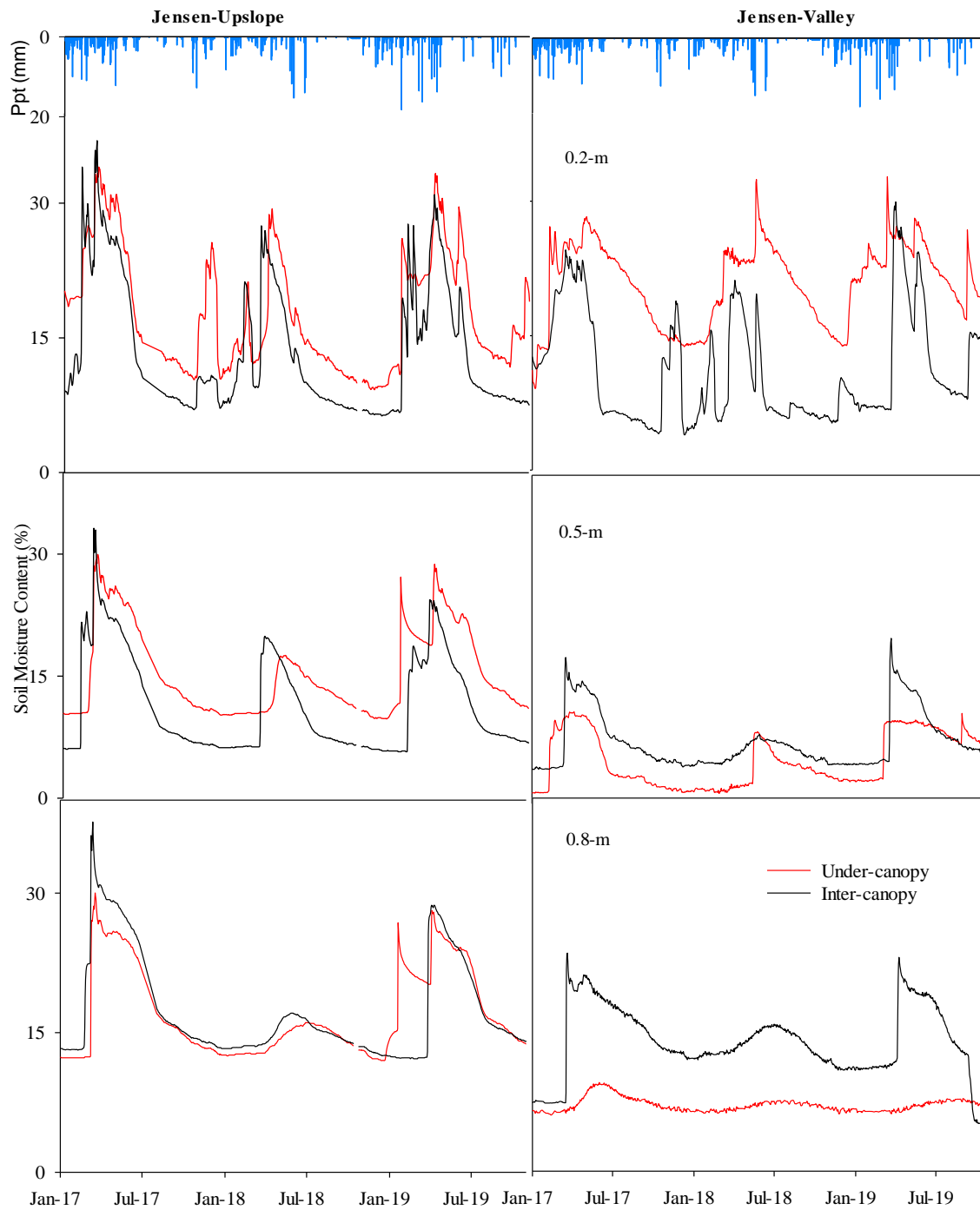
Figure 4.5. Daily-averaged preceipitation and soil moisture content for the three sensor depths (0.2, 0.5, and 0.8 m) at inter-canopy and under-canopy zones in both locations in Jensen WS during January 2017 through October 2019.

*3.4. Transpiration and Soil Moisture Relations—Watershed Scale*

3.4.1. Mays WS

Figure 4.6 shows the daily transpiration and $\theta_{tot}$ relationships in response to precipitation in Mays WS. There was no ($P = 0.3$) significant differences in $\theta_{tot}$ in the onset (January-March) of the three years (2017-2019) for the (Mays-East) location. The values of $\theta_{tot}$ peaked in late winter for the years 2017 and 2019, whereas reached a maximum value in early spring 2018. The significantly greater $\theta_{tot}$ recorded in spring season for the year 2017 compared to 2018 (paired t-test, t = 27.5, df = 90, p $\leq$ 0.001), which may be accounted by heavy winter precipitation resulted in saplings transpiration to be 3 times greater in the summer 2017 compared to 2018. Total winter precipitation was 2.8 times significantly ($P \leq 0.05$) greater in the year 2017 compared to 2018 (data not shown). When $\theta_{tot}$ dropped below 15% in the end of the summer and through the progression of the fall, transpiration accordingly declined for both years 2017 and 2018. The soil water content at 0-80 cm depth was fully replenished in 23 March 2019, around the day that coincided with a gradual increase in transpiration. The significantly ($P \leq 0.05$) greater transpiration in summer 2019 than 2018 was also because of higher recorded $\theta_{tot}$ in spring 2019.

In the other location, (Mays-West) the same relationship was observed from fall 2018 through summer 2019 for the saplings (Figure 4.6). Except three extreme values recorded in both trees at the same days, transpiration was maintained at rate of less than 0.60 L d$^{-1}$ during the winter season in (Mays-East). In the spring 2019 season, there was a gradual increase in transpiration related to rainfall-fed $\theta_{tot}$. It can be noted that transpiration for the saplings in Mays WS peaked during spring/summer seasons and remarkably declined through late summer to fall in concert with $\theta_{tot}$ drying, indicating the time at which soil moisture became limiting.

Figure 4.6. Juniper daily-averaged transpiration and mean soil water content, 0-80 cm ($\theta_{tot}$) response to precipitation inputs at inter-canopy zones in Mays WS for the saplings in the (Mays-East) location during January 2017 through October 2019 and in (Mays-West) location during October 2018 through September 2019.

## 3.4.2. Jensen WS

The daily transpiration and $\theta_{tot}$ relationships in response to precipitation in Jensen WS are illustrated in (Fig 4.7). In the (Jensen-Valley) location, peak values of $\theta_{tot} \approx 14$ to 16% were recorded in spring for the three years, whereas minimum values of $\theta_{tot} < 11\%$, indicative of extreme soil drought, occurred during the end of the fall. There was significantly ($P \leq 0.05$) more $\theta_{tot}$

available at the onset (January to March) of the years 2017 and 2019 (with no difference, $P = 0.11$) compared to the year 2018. In addition, spring $\theta_{tot}$ did not significantly $(P = 0.98)$ differ between the same years and was significantly $(P \leq 0.05)$ greater than 2018. This indicates that the year 2017 was wetter and maybe warmer than 2018, and this may have accounted for mature juniper transpiration to be 2.0 times significantly $(P \leq 0.05)$ greater in summer for the year 2017 than 2018 (Table 4.5). The day at which maximum transpiration was recorded indicated the point at which $\theta_{tot}$ became limiting. This point differed between the years 2017 and 2018, probably because of the higher amount of $\theta_{tot}$ available from winter and spring precipitation in 2017, and occurred in 18 August, 2017 than in 22 June, 2018. From this point, transpiration decreased in concert with $\theta_{tot}$.

In year 2019, $\theta_{tot}$ responding to heavy winter precipitation gradually increased and reached a maximum value of 15% in middle of May. Consequently, transpiration peaked early June and then declined, fluctuating slightly, until it reached the low for the year in late September the same time as $\theta_{tot}$ was recorded a minimum. The limiting $\theta_{tot}$ point where transpiration started to decline was about 10 days more extended in 2019 than 2018, which may have caused transpiration to be greater during spring 2018 than 2019.

In the other location, (Jensen-Upslope), the same relationship was observed from January 2018 through September 2019 for juvenile juniper (Fig 4.7). A similar pattern to the other locations was observed. Following precipitation, $\theta_{tot}$ rose to about 19% (VWC = 0.19 18 $m^3$ $m^3$) in late April and then started to decline gradually until reached its minimum, 10% in mid-December. Transpiration, which was first recorded in 22 April, had immediate response to $\theta_{tot}$. Transpiration rose sharply and peaked 3.6 L $d^{-1}$ on 2 June, then fluctuated above and below 2.7 L $d^{-1}$ until 8 July when it began a steady decline and bottomed out to 0.09 L $d^{-1}$ on 16 December. So, when $\theta_{tot}$ replenished after rainfall events, transpiration quickly increased during spring and early summer,

then transcription decreased as an indication of marked dry down in $\theta_{tot}$ over the course of the summer.
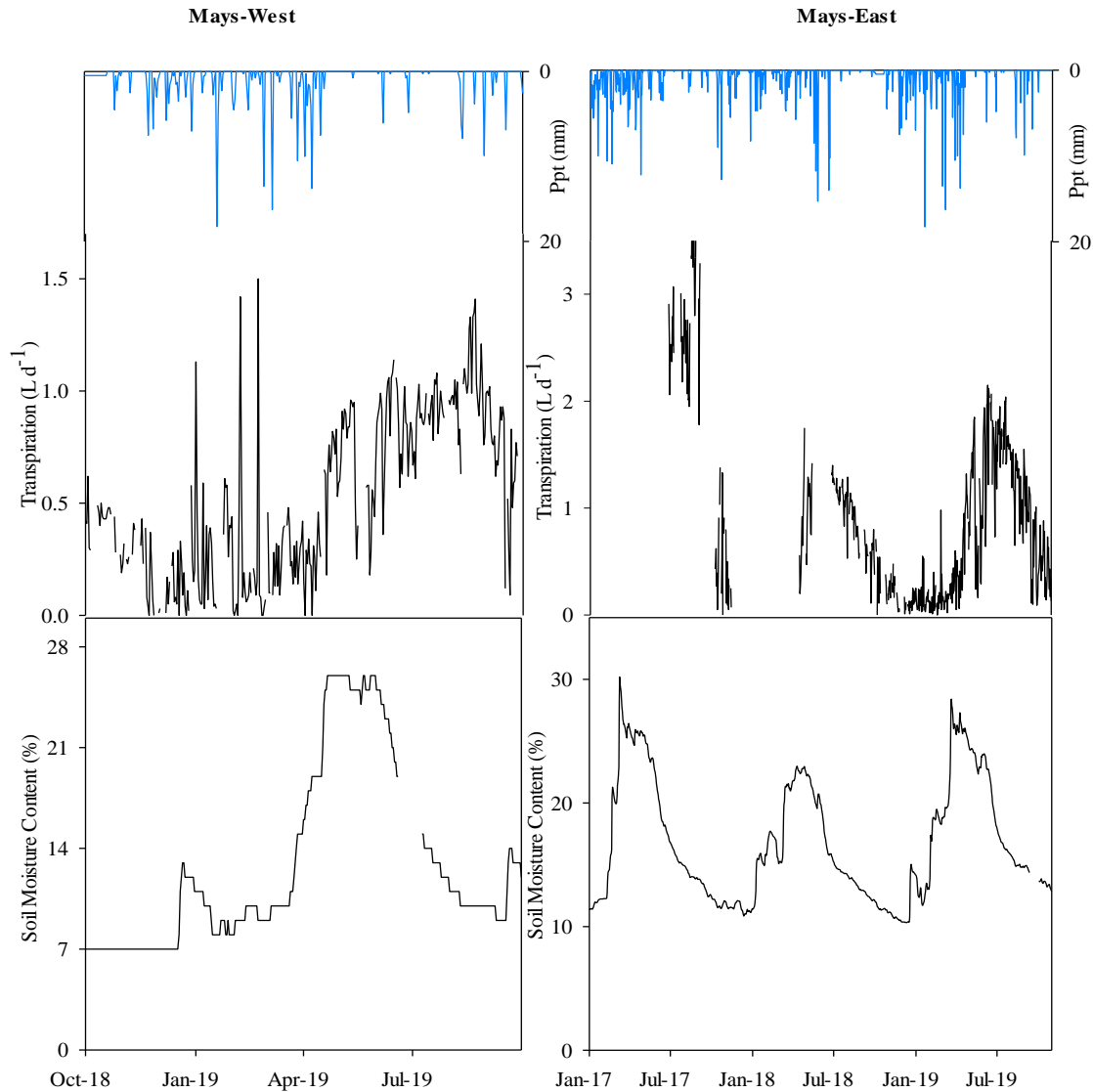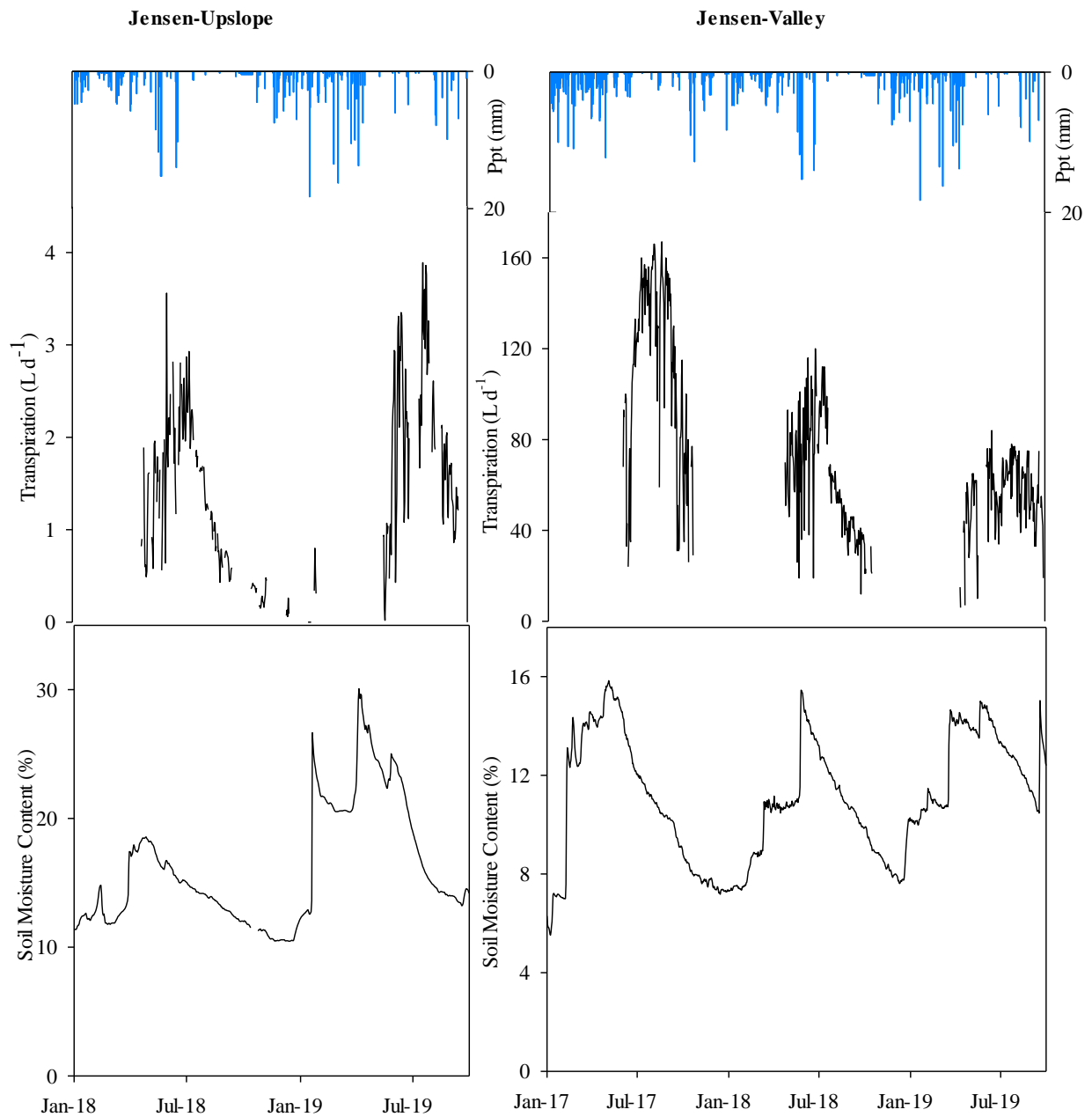


Figure 4.7. Juniper daily-averaged transpiration and mean soil water content, 0-80 cm ($\theta_{tot}$) response to precipitation inputs at under-canopy zones in Jensen WS for mature juniper in the (Jensen-Valley) location during January 2017 through September 2019 and for juvenile juniper in the (Jensen-Upslope) location during January 2018 through September 2019.

In early winter to mid-spring 2019, there was fluctuated rise in $\theta_{tot}$ which was followed by a steeper increase in transpiration during the spring ranging from 0.02 to the peak, 3.89 L d$^{-1}$ on 20 July. By then, there was a noticeable decline in transpiration during the progression to the late summer. The significantly ($P \leq 0.05$) greater $\theta_{tot}$ in spring 2019 than 2018 due to the relatively considerable winter precipitation inputs caused summer transpiration to be 1.5 times greater in 2019 than 2018.

In this study, $\theta_{tot}$ was shown to control sap-flow and therefore juniper transpiration for all locations. Considerable variation in precipitation and available $\theta$ was observed over the study (2017-2019). Precipitation varied seasonally over the three years of this study (data not shown). Heavy winter precipitation recorded for the 2017 and 2019 years was reflected in the $\theta_{tot}$ with high levels of $\theta$ being available for a longer portion of the growing season in these years compared to the 2018 year. Thus, transpiration by western juniper varied seasonally and tended to be highest during summer seasons followed by spring and finally the fall months for the years 2017 and 2019. The summer precipitation was considerably variable over the three years, highlighting the sensitivity of the semiarid ecosystem to summer rain. In the least rainy year (2018), summer rain was very minimal leading juniper daily transpiration to rapidly decline after peaking in spring, with a clear trend for mature juniper. In addition, the daily-averaged water consumption by juniper tended to be greater in the year 2017, followed by 2019 and then 2018, which is likely because of the higher precipitation registered in 2017 and 2019. Higher precipitation likely caused larger soil water reserves (Mollnau et al., 2014) that were signifcantly ($P \leq 0.05$) available for plant uptake in 2017 and 2019 compared to 2018.

The relationships between soil water availability and transpirational water loss were examined (Fig 4.8) for the three years. The relative response of juniper transpiration to soil water content exhibited similar patterns over the three years, although the nonlinear regression relations

are different. The close coupling between the $\theta_{tot}$ in the upper 0.8 m of soil and transpirational loss for the saplings in Mays WS (Fig 4.8) indicated that most transpired water was coming from the surface rather than subsurface. However, a relatively lower correlation between $\theta_{tot}$ in the upper 0.8 m depth and transpired water for mature juniper was indicative of ample water use from lower soil depths. Juvenile juniper seemed to rely less on the water in the top 0.8 m of the soil profile compared to the saplings. In all years, $R^2$ was greatest for saplings, intermediate for juvenile, and least for mature juniper.
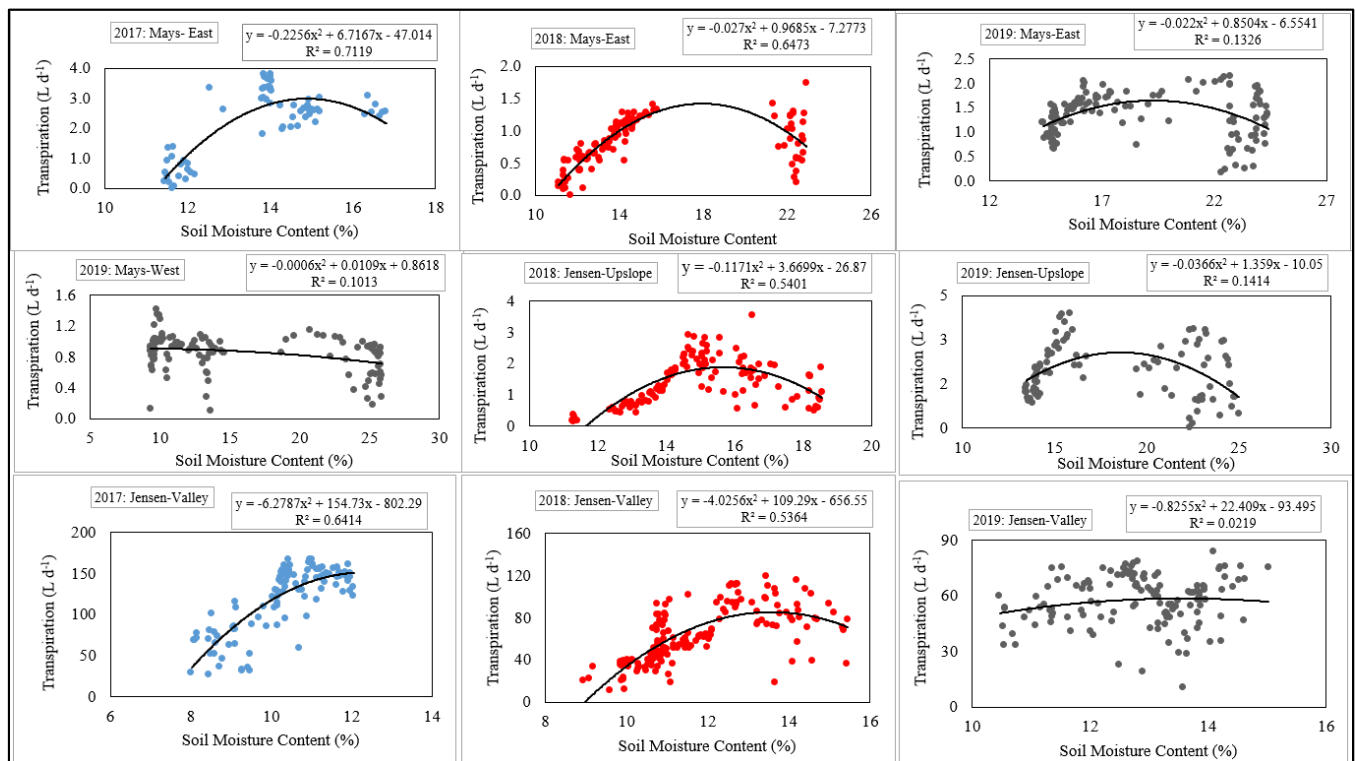


Figure 4.8. Polynomial regression analyses performed to determine relationships between daily transpiration (L d$^{-1}$) and daily mean soil water content for the entire soil profile, 0-80 cm ($\theta_{tot}$) for western juniper growth stages at different years with corresponding periods of time. The analyses were done from (July-October), (April-October), and (May-September) for the years 2017 (blue dots), 2018 (red dots), and 2019 (black dots) respectively.

## 4. Discussion

This study shows that strong seasonal relationships between transpiration and soil moisture content exist in rangeland watershed systems of semiarid central Oregon. This case study added information pertaining to these existing relations for different juniper growth stages.

All studied trees showed similar patterns of seasonal transpiration, with generally maximum values in late spring/early summer and acute reductions as the summer drought progressed and toward early fall. During winter, cold soil temperatures restrict transpiration for these woodlands (Miller and Schultz, 1987). As soil temperatures augment in March, juniper begins to actively transpire and grow (Miller et al., 2005). The decline in leaf water potential (less negative) through time for western juniper and the range between predawn and midday water potentials supported our findings that transpiration peaked during the period (June-August) and declined with soil drying. Juniper woodlands were reported to maintain low transpiration while leaf water potentials continuously increased with soil drying (Miller and Schultz, 1987; McDowell et al., 2008; West et al., 2008). The daily maximum values of transpiration estimated for the mature western juniper during the study were within the ranges of maximum transpiration (80-170 L d$^{-1}$) reported for a mature *Juniperus ashei* (Owens and Ansley, 1997), *Abies amabilis* (Martin et al., 1997), *Pinus pinaster* (Granier et al. 1990), *Pinus radiata* (Edwards 1986), Eucalyptus *loxophleba* (Barrett et al. 1996), and *Ficus insipida* (Goldstein et al. 1998).

Soil water content under treated and untreated western juniper management practices was investigated. The great levels of juniper encroachment, which has around 31% canopy cover (Ray et al., 2019) in the untreated watershed had large effects on soil moisture. We found that soil moisture was considerably greater under juniper canopies than in open areas dominated by herbaceous species for all sensor depths except for the 0.5 and 0.8 m depths in (Jensen-Valley)

location where greater soil moisture content levels were found at inter-canopy versus under-canopy zones. That is consistent with Ochoa et al. (2018). This could suggest that soil moisture depletion by mature juniper root abstraction was higher in those depths. Juniper is an evergreen with extensive lateral and deep roots, and it has physiological adaptations which enable it to maximize water extraction from deeper depths (Thurow and Hester, 1997). According to Mollnau et al. (2014), water uptake by western juniper was not only exploited in shallow soil layers, it also occurred in deeper depths. Deep roots of trees may increase the supply of hydraulically lifted water (Fernandez et al., 2008). Another juniper species, *J. osteosperma* was reported to extract water from 1 m in the Great Basin (Leffler et al., 2002). We found that the upper (0.2 m) soil layer had greater soil water content levels in under-canopy vs. inter-canopy zones in (Jensen-Valley) location. Our findings are similar with previous in the same site (Ochoa et al., 2018; Ray et al., 2019) and can be partially caused by the shading provided by the trees that may have led to less soil evaporation. Our results are different from a study carried out in western juniper in Idaho, where no difference in topsoil water content levels were found under canopy and inter-canopy zones (Niemeyer et al., 2016).

We observed that there were connections between $\theta_{tot}$ and transpiration for western juniper, as indicative of highest $\theta_{tot}$ in spring seasons corresponding with active transpiration by juniper. Transpiration declined quickly as $\theta_{tot}$ was depleted. Seasonal variaiton in $\theta_{tot}$ had an considerable impact on juniper transpiration at our site. Heavy winter precipiotation provided the majority of the moisture for transpiration over the growing season in the years 2017 and 2019. During the summer, transpiration was greater in wetter years (2017 and 2019) relative to a least rainy year (2018) at the same $\theta_{tot}$. In the year 2018, western juniper effectively transpired in spring season. The annual variations in $\theta_{tot}$ availability influenced transpiration activities of western juniper. Our

findings of year to year variations in soil moisture effects on transpiration were consistent with other studies with different woody species in arid and semiarid lands (Loik et al. 2004, Schwinning and Sala 2004, West et al., 2008) and highlight the need for long-term, continuous measurements of transpiration in order to accurately capture the dynamics of these systems. The woody species, mesquite was found to transpire about 30 to 205 L d$^{-1}$ (Ansley et al. 1991; Ansley et al. 1994; Ansley et al. 1998), but when soil moisture was limited, transpiration rates considerably declined (Dugas et al., 1992).

The following conclusions can be drawn from this research: 1) transpiration by western juniper is driven by precipitation and soil moisture, which in turn varies by year, 2) the period of the higher transpiration by juniper is summer in wet years like 2017 and 2019 and in spring during dry year like 2018, 3) leaf water potential measurements are good indicators of juniper water status where less negative values corresponds with the occurrence of transpiration, and 4) Juniper removal results in considerable water savings. Even after a 13-year period of juniper regrowth, the amount of water used by single mature trees can be about 70 times higher than the amount of water used by re-growing saplings. This study provides compelling evidence that western juniper elimination has great potential for increasing water yield in the Camp Creek watershed, which could support livestock grazing by increasing forage production. Furthermore, results from this study contribute to the knowledge of soil moisture and transpiration relations at different management practices in rangeland ecosystems.

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## Appendix

### Soil Physical Properties

Table 4.A1. Soil physical properties for the paired watersheds within the study area, (a) Jensen WS and (b) Mays WS, showing the mean and standard error (n = 3) of soil bulk density, soil particle distribution of sand, silt, and clay, volumetric water content and soil porosity at each soil depth. Also, mean $\theta$ values at permanent wilting point ($\theta$PWP), field capacity ($\theta$FC), and available water content (AWC) are identified for each soil depth. The Jensen WS location illustrates data collected at under-canopy and inter-canopy area in upslope and valley settings. Means with the same letter superscripts (a, b) along columns indicate no significant differences for soil depths ($P < 0.05$).

| Soil Depth | BD | Sand | Silt | Clay | VWC | PS | $\theta$FC | $\theta$PWP | AWC |
|---|---|---|---|---|---|---|---|---|---|
| | (Mg m$^{-3}$) | (%) | (%) | (%) | (%) | (%) | (%) | (%) | (%) |
| (a) Jensen WS | | | | | | | | | |
| Upslope-Under-canopy | | | | | | | | | |
| 0.2 m | 1.22 ± 0.04 b | 54.4 ± 2.80 a | 25.1 ± 0.35 a | 20.51 ± 2.45 a | 31.10 ± 4.80 a | 53.97 ± 1.52 a | 27 | 10 | 17 |
| 0.5 m | 1.35 ± 0.03 ab | 56.3 ± 2.40 a | 25.9 ± 3.08 a | 17.91 ± 0.81 a | 40.95 ± 12.20 a | 49.05 ± 1.06 ab | 27 | 9.8 | 17.2 |
| 0.8 m | 1.49 ± 0.03 a | 53.7 ± 1.04 a | 24.4 ± 0.23 a | 21.91 ± 1.27 a | 33.68 ± 9.12 a | 43.61 ± 1.05 b | 28 | 10 | 18 |
| Upslope-Inter-canopy | | | | | | | | | |
| 0.2 m | 1.33 ± 0.04 b | 54.3 ± 0.67 a | 26.6 ± 1.94 a | 19.1 ± 1.85 a | 25.51 ± 0.74 a | 49.62 ± 1.30 a | 29 | 9.7 | 19.3 |
| 0.5 m | 1.43 ± 0.015 b | 49.9 ± 2.73 a | 26.6 ± 1.20 a | 23.5 ± 1.58 a | 26.76 ± 0.23 a | 46.21 ± 0.51 a | 27 | 10 | 17 |
| 0.8 m | 1.63 ± 0.015 a | 54.4 ± 1.47 a | 22.3 ± 1.33 a | 23.3 ± 0.47 a | 29.02 ± 2.21 a | 38.49 ± 0.62 b | 25 | 9.9 | 15.1 |
| Valley-Under-canopy | | | | | | | | | |
| 0.2 m | 1.19 ± 0.042 b | 50.3 ± 0.73 c | 33.6 ± 0.70 a | 16.1 ± 0.07 a | 21.45 ± 0.63 a | 55.20 ± 1.57 a | 42 | 10.7 | 31.3 |
| 0.5 m | 1.46 ± 0.061 a | 58.2 ± 0.60 b | 24.2 ± 0.24 b | 17.7 ± 0.37 a | 24.05 ± 1.27 a | 44.92 ± 2.26 b | 30 | 9.8 | 20.2 |
| 0.8 m | 1.28 ± 0.067 ab | 63.7 ± 1.16 a | 18.6 ± 0.95 c | 17.7 ± 0.82 a | 19.46 ± 5.48 a | 51.63 ± 2.43 ab | 26 | 9.3 | 16.7 |
| Valley-Inter-canopy | | | | | | | | | |
| 0.2 m | 1.43 ± 0.118 a | 60.5 ± 1.07 b | 20.6 ± 0.35 a | 18.9 ± 1.12 a | 27.33 ± 1.13 a | 46.02 ± 4.46 a | 39 | 8.9 | 30.1 |
| 0.5 m | 1.52 ± 0.027 a | 68.7 ± 1.62 a | 14.5 ± 1.29 b | 16.9 ± 1.79 a | 20.71 ± 0.49 a | 42.55 ± 0.93 a | 27 | 9.5 | 17.5 |
| 0.8 m | 1.64 ± 0.072 a | 73.8 ± 0.31 a | 10.5 ± 1.00 b | 15.7 ± 0.70 a | 23.00 ± 5.07 a | 38.21 ± 2.71 a | 24 | 4.9 | 19.1 |

(b) Mays WS

East Facing Slope-Inter-canopy

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| 0.2 m | 1.52 ± 0.009 b | 60.1 ± 1.43 b | 22.1 ± 0.74 a | 17.9 ± 0.70 a | 22.85 ± 4.05 a | 42.65 ± 0.34 a | 33 | 9.3 | 23.7 |
| 0.5 m | 1.59 ± 0.006 a | 61.7 ± 0.71 ab | 20.1 ± 0.79 ab | 18.2 ± 0.42 a | 34.57 ± 4.57 a | 40.04 ± 0.27 b | 28 | 9.5 | 18.5 |
| 0.8 m | 1.57 ± 0.012 a | 65.9 ± 0.29 a | 17.5 ± 0.29 b | 16.5 ± 0.47 a | 24.83 ± 5.58 a | 40.88 ± 0.44 b | 25 | 3.8 | 21.2 |

West Facing Slope-Inter-canopy

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| 0.2 m | 1.31 ± 0.012 a | 58.1 ± 2.27 a | 24.5 ± 1.53 a | 17.5 ± 0.74 a | 24.85 ± 4.05 a | 50.48 ± 0.38 a | 39 | 10.1 | 28.9 |
| 0.5 m | 1.33 ± 0.055 a | 53.3 ± 0.55 a | 28.7 ± 3.65 a | 18 ± 3.22 a | 19.77 ± 1.01 a | 49.68 ± 2.07 a | 35 | 10.1 | 24.9 |
| 0.8 m | 1.38 ± 0.019 a | 47.9 ± 0.41 a | 34.5 ± 1.43 a | 17.6 ± 1.47 a | 27.85 ± 1.52 a | 48.10 ± 0.74 a | 36 | 9.9 | 26.1 |

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# GENERAL CONCLUSIONS

This dissertation examined the effects of western juniper encroachment and removal on ecosystem processes in central Oregon. Two semiarid paired watersheds, of approximately 110 ha each, were used to quantify ecosystem carbon and nitrogen accumulations. Also, they were equipped with hydrologic data sensors to measure precipitation, soil water content, and transpiration.

The first task of this research compared field data collection and determined ecosystem carbon pools of one treated watershed (most juniper removed) and one untreated watershed (juniper is dominant overstory vegetation). Biomass samples for juniper trees, shrubs, grasses, and litter were collected. Additionally, soils were collected and brought to a laboratory for carbon concertation determination. We found that the treated watershed stored less aboveground carbon 13 years post juniper removal than the untreated watershed. This was attributed to the elimination of mature western juniper trees, which represented the bulk of the aboveground carbon. Root carbon has increased since juniper removal, which was mainly due to the large increase in grass roots in the treated watershed. Thus, total ecosystem carbon did not significantly ($P = 0.52$) differ and showed an average of 137.6 Mg C ha$^{-1}$. The greatest ecosystem carbon accumulation resides belowground (over 90%). Therefore, changes in the 10% aboveground biomass can be of less relative significance in the short term.

The second portion of this research had the same ground data techniques to the first task with an aim of quantifying ecosystem nitrogen storage. Similarly, after 13 years of juniper elimination we found that the aboveground biomass was greater in the untreated watershed resulting in significantly ($P < 0.05$) more aboveground nitrogen accumulations in the untreated

watershed than the treated watershed. A greater root nitrogen accumulation in the treated area than the untreated area partially counterbalanced the losses in aboveground nitrogen due to juniper elimination. Total nitrogen stores in both management practices (untreated and treated watersheds) did not differ ($P = 0.14$) and had an average nitrogen level of 1283.2 kg N ha$^{-1}$. Ecosystem nitrogen was dominated by the belowground pool in both management practices. The greatest ecosystem nitrogen accumulation resides belowground (over 80%). Therefore, changes in the 20% aboveground biomass can be of less relative importance in the short term.

The first two tasks suggest it is crucial to protect belowground for carbon and nitrogen stores. They conclude that western juniper removal did not impact belowground carbon and nitrogen pools, at least 13 years post removal. However, our 13-year post treatment study, is still of short duration to contemplate soil carbon and nitrogen changes. More studies of longer duration about effects of western juniper encroachment and removal on these soil nutrients are suggested to further support findings from this research. The restoration programs of western juniper in central Oregon has been associated with favorable results for livestock and watershed hydrological functions. In addition, these tasks provide evidence that western juniper management does not significantly impair with the capacity of ecosystem for storing soil carbon and nitrogen.

The third task of this research sought to measure western juniper transpiration at different growth stages/sizes (saplings, juvenile, and mature) and to investigate relationships between transpiration and soil moisture at treated and untreated watersheds. All juniper stages had similar patterns of seasonal transpiration with maximum values around (June-August) and lower values during fall. Mature juniper in the untreated watershed substantially used more water that the saplings that occurred 13 years post juniper elimination in the treated watershed. That indicates that a considerable area of dominated-juniper land would need to be treated for water savings.

Seasonal variaiton in the mean soil water content at 0-80 cm ($\theta_{tot}$) largely affected juniper transpiration at our site. Heavy winter precipiotation recorded for the years 2017 and 2019 provided the majority of the moisture for transpiration over the growing season. Additionally, considering that summer precipitation was less pulsed in the year 2018 (least rainy year) compared to wet years (2017 and 2019) $\theta_{tot}$ availability was not the same across the three years (2017-2019). Thus, juniper transpiration tended to differintiate inter-annually. During the summer, transpiration was greater in the wetter years relative to the least rainy at the same $\theta_{tot}$. In the year 2018, western juniper effectively transpired in spring season. The polynomial regression analyses demonstrated that transpiration was less strongly coupled to $\theta_{tot}$ for mature juniper compared to saplings. It is likely that mature juniper had access to deeper moisture below the range of our sensors (80 m). Mature juniper has a larger proportion of roots in deeper layers. The extensive and long-term field data collected throughout this research can be utilized as baseline information for future western juniper restoration projects.
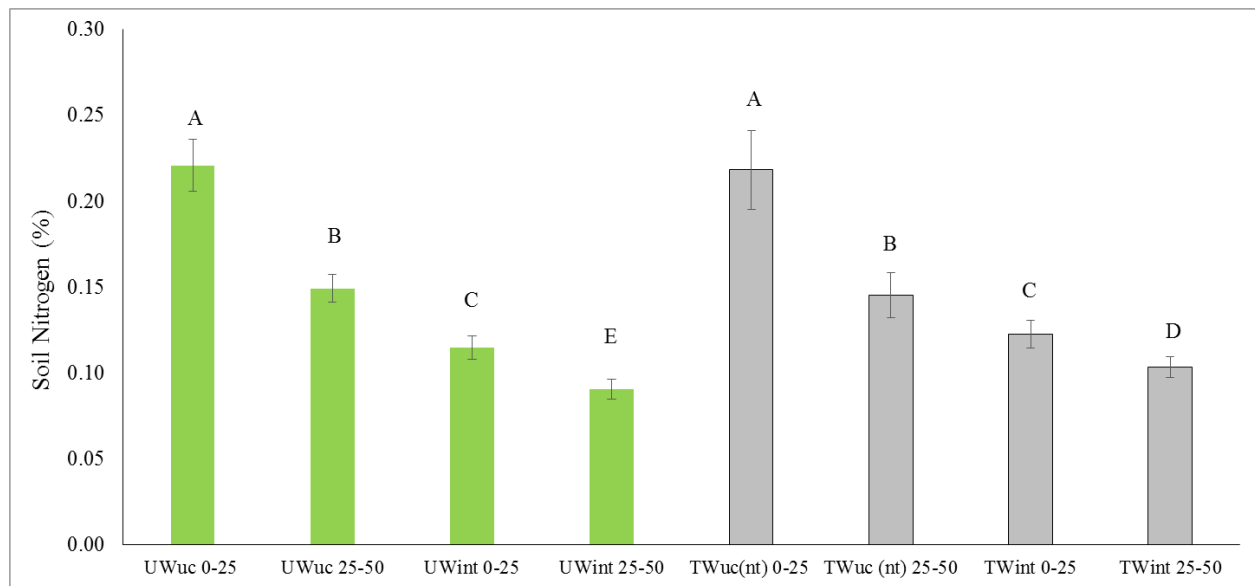
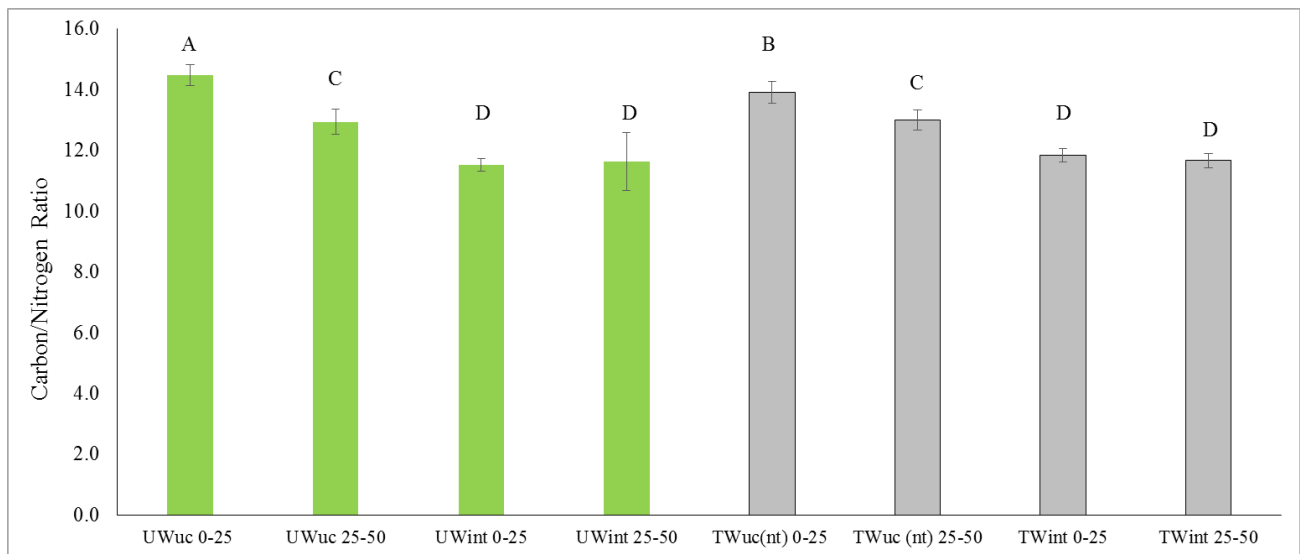# APPENDICES

List of Appendix Figures
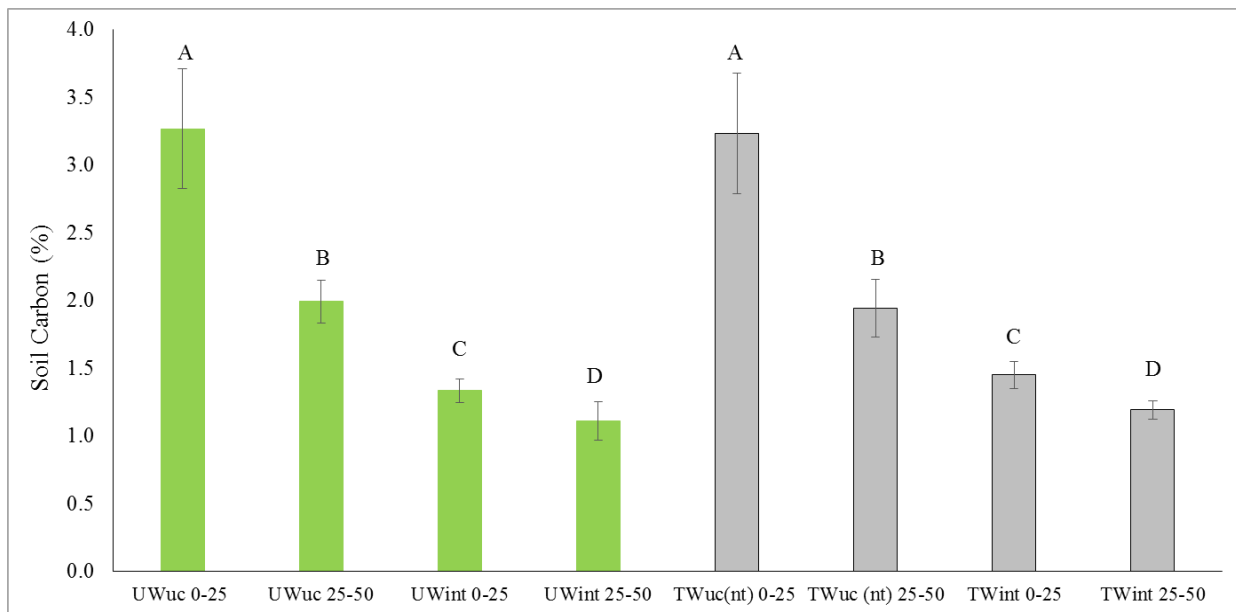


Figure A2.1. Soil carbon concentration (%) at (0-25 and 25-50 cm) soil depths by watershed and treatments   (UWuc = untreated watershed and under tree canopies; UWint = untreated watershed and interspaces beyond tree canopies; TWuc (nt) = treated watershed and under tree canopies (near the tree stumps); TWint = treated watershed and interspaces beyond the tree stumps. Data are means ± standard error. Different capital letters (A, B, C, D) along indicate significant differences ($P \leq 0.05$) across management practices.
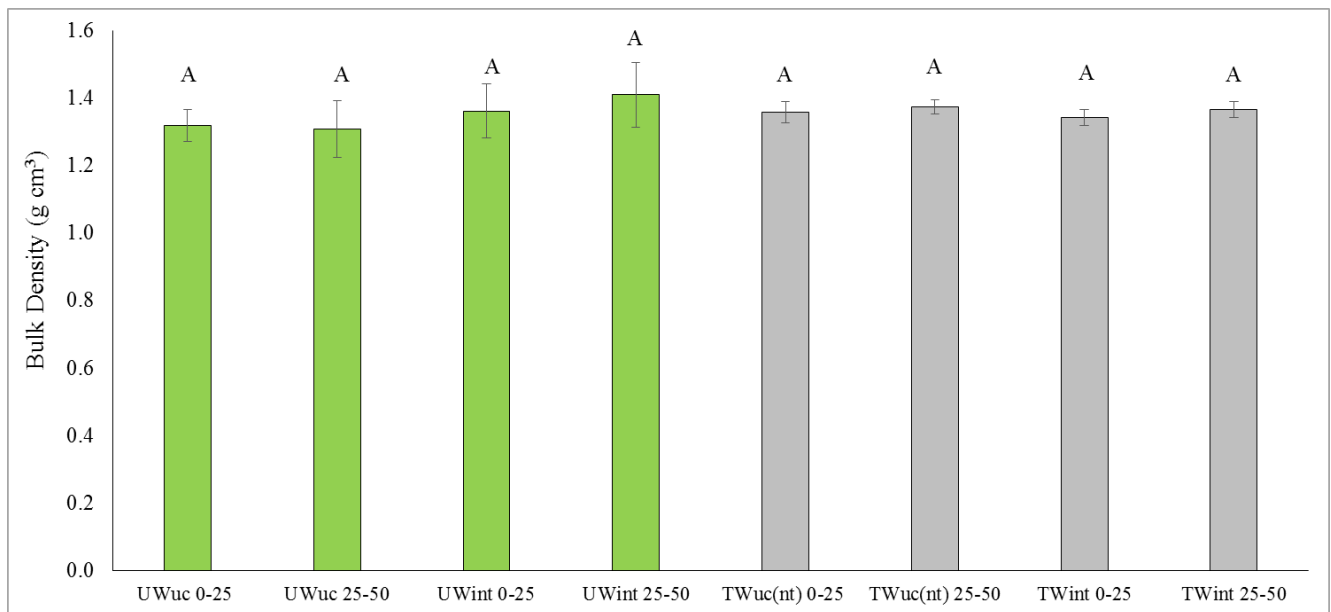
Figure A3.1. Soil nitrogen concentration (%) at (0-25 and 25-50 cm) soil depths by watershed and treatments (UWuc = untreated watershed and under tree canopies; UWint = untreated watershed and interspaces beyond tree canopies; TWuc (nt) = treated watershed and under tree canopies (near the tree stumps); TWint = treated watershed and interspaces beyond the tree stumps. Data are means ± standard error. Different capital letters (A, B, C, D, E) along indicate significant differences ($P \leq 0.05$) across management practices.

Figure A3.2. Soil carbon to nitrogen ratio at (0-25 and 25-50 cm) soil depths by watershed and treatments (UWuc = untreated watershed and under tree canopies; UWint = untreated watershed and interspaces beyond tree canopies; TWuc (nt) = treated watershed and under tree canopies (near the tree stumps); TWint = treated watershed and interspaces beyond the tree stumps. Data are means ± standard error. Different capital letters (A, B, C, D) along indicate significant differences ($P \leq 0.05$) across management practices.

Figure A3.3. Soil bulk density (g cm$^3$) at (0-25 and 25-50 cm) soil depths by watershed and treatments (UWuc = untreated watershed and under tree canopies; UWint = untreated watershed and interspaces beyond tree canopies; TWuc (nt) = treated watershed and under tree canopies (near the tree stumps); TWint = treated watershed and interspaces beyond the tree stumps. Data are means ± standard error.

Figure A4.1. Installation of automated water use equipment in the field for the area (treated watershed) where small sapling trees that result after regrowth dominate.

Figure A4.2. Installation of automated water use equipment in the field for the area (untreated watershed) dominated by mature juniper.
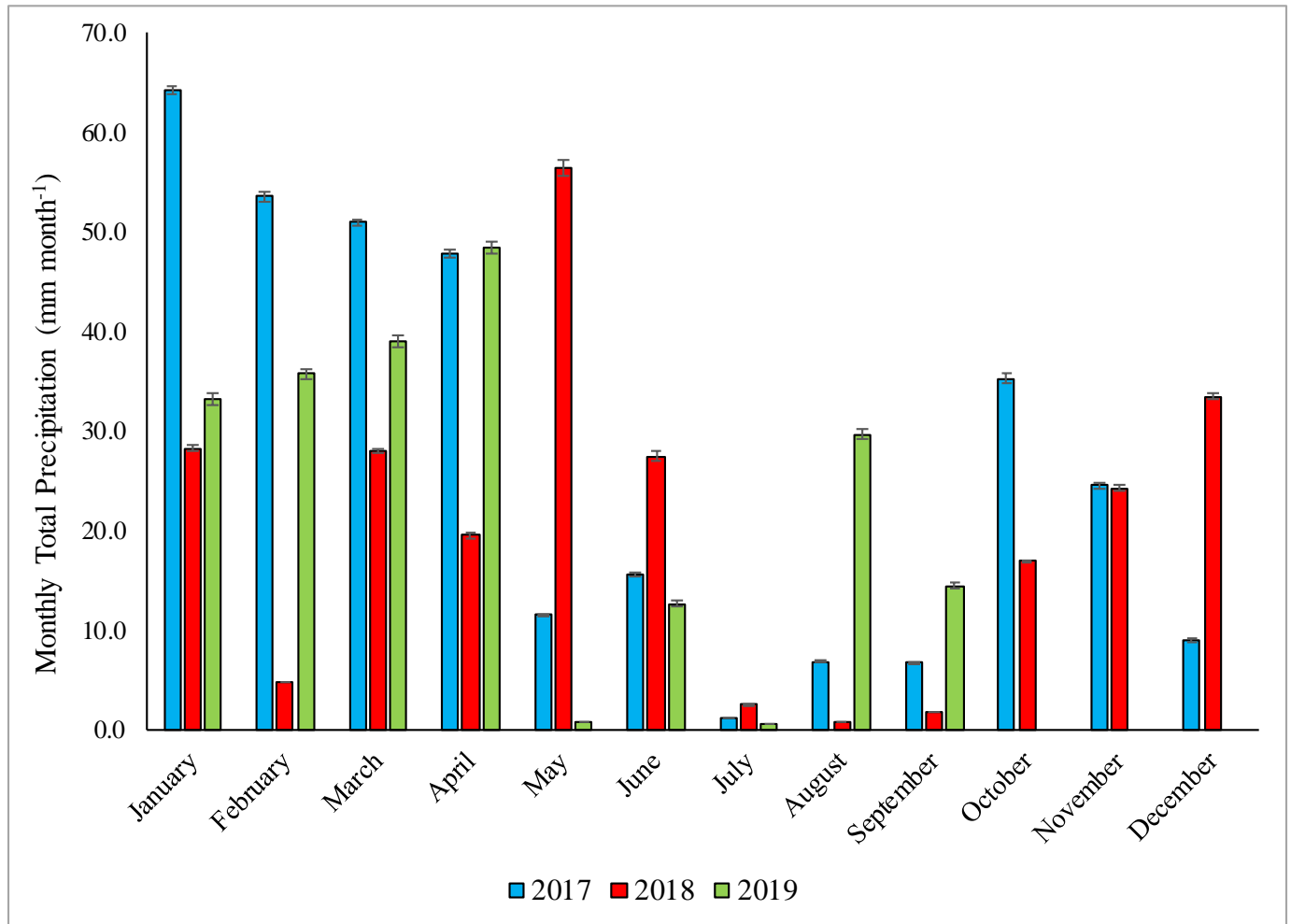
Figure A4.3. Monthly total precipitation (mm month$^{-1}$) for three years (2017-2019) recorded by onsite instrumentation.
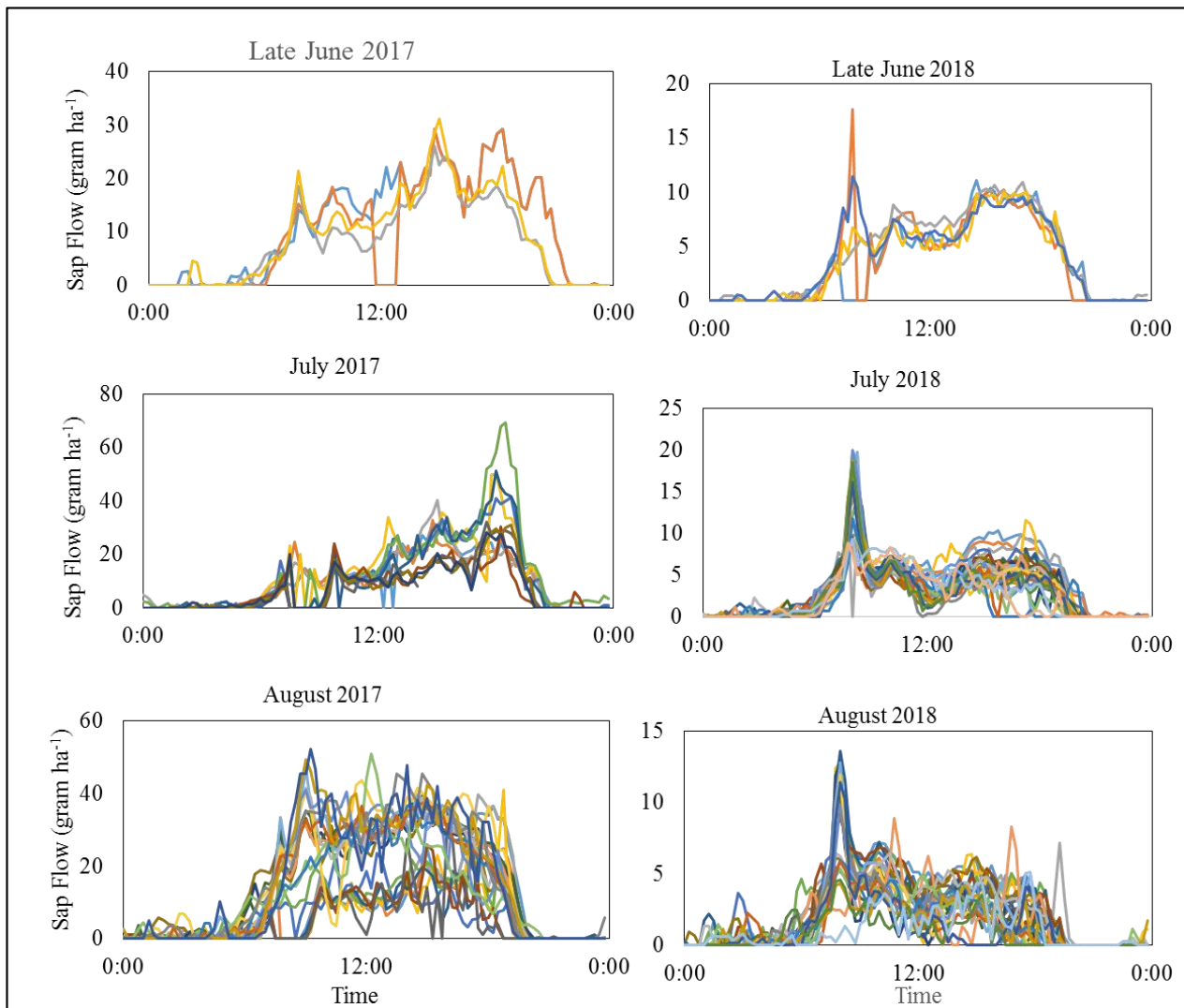
Figure A4.4. Hourly sap flow (gram h[-1]) for sapling (S1) instrumented with SHB (SGB16) sensor for selected months in years 2017 and 2019. These values represent branch sap flow before scaling to the whole tree.
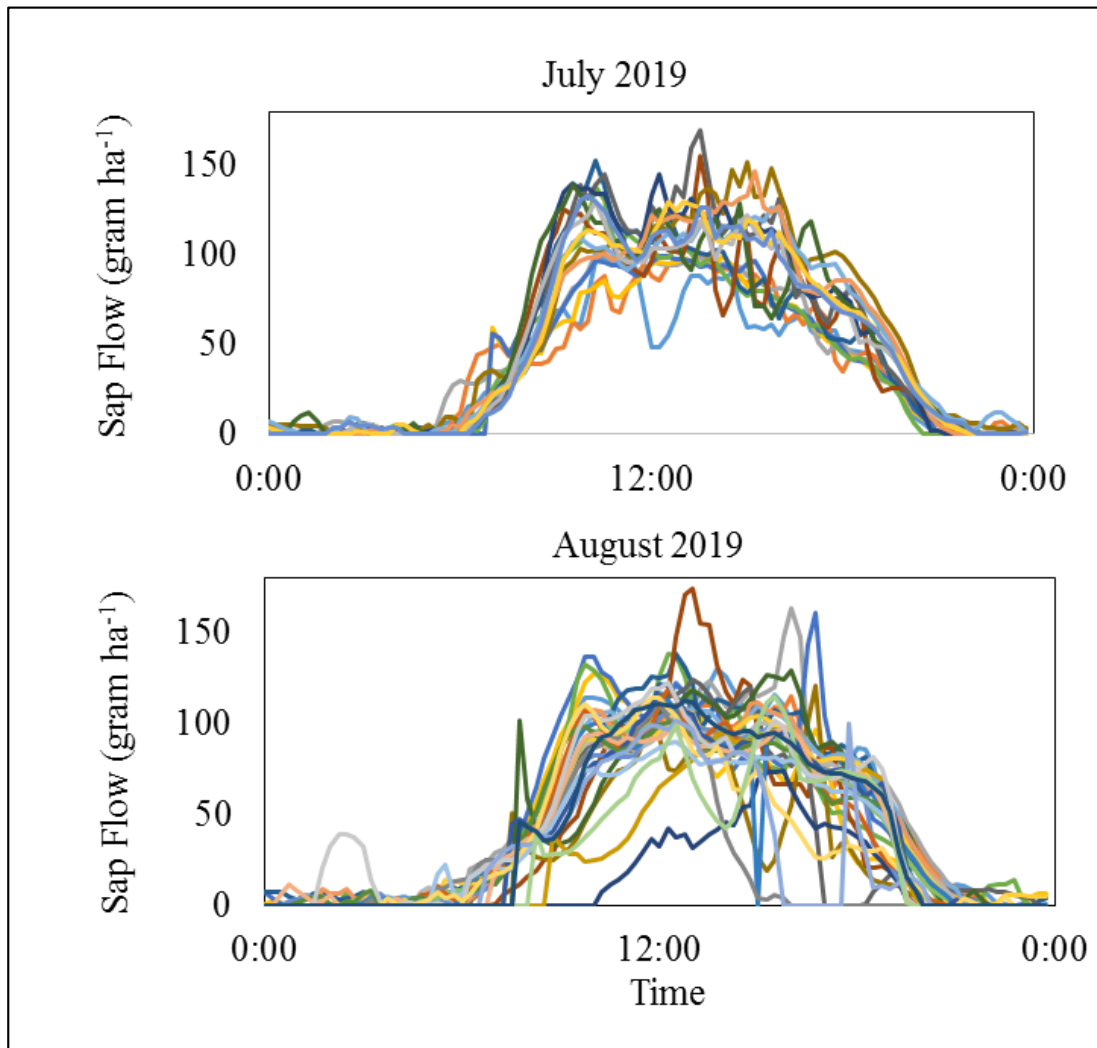
Figure A4.5. Hourly sap flow (gram h[-1]) for sapling (S1) instrumented with SHB (SGB25) sensor for July and August 2019. These values represent the main stem sap flow. So, there was no need for scaling from branch to whole tree.
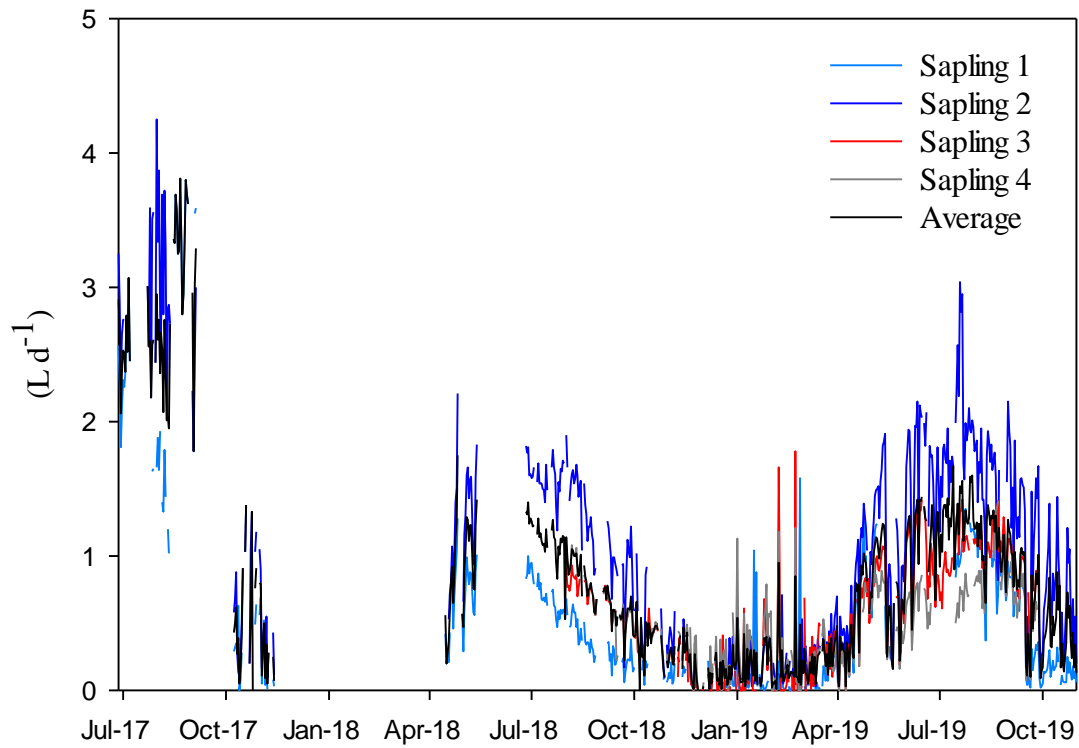
Figure A4.6. Daily-averaged transpiration for the saplings 1, 2 at (Mays-East) location, the saplings 3, 4 at (Mays-West) location, and the average in the Mays WS. Missing data are due to lightening-induced equipment failure or battery failure.
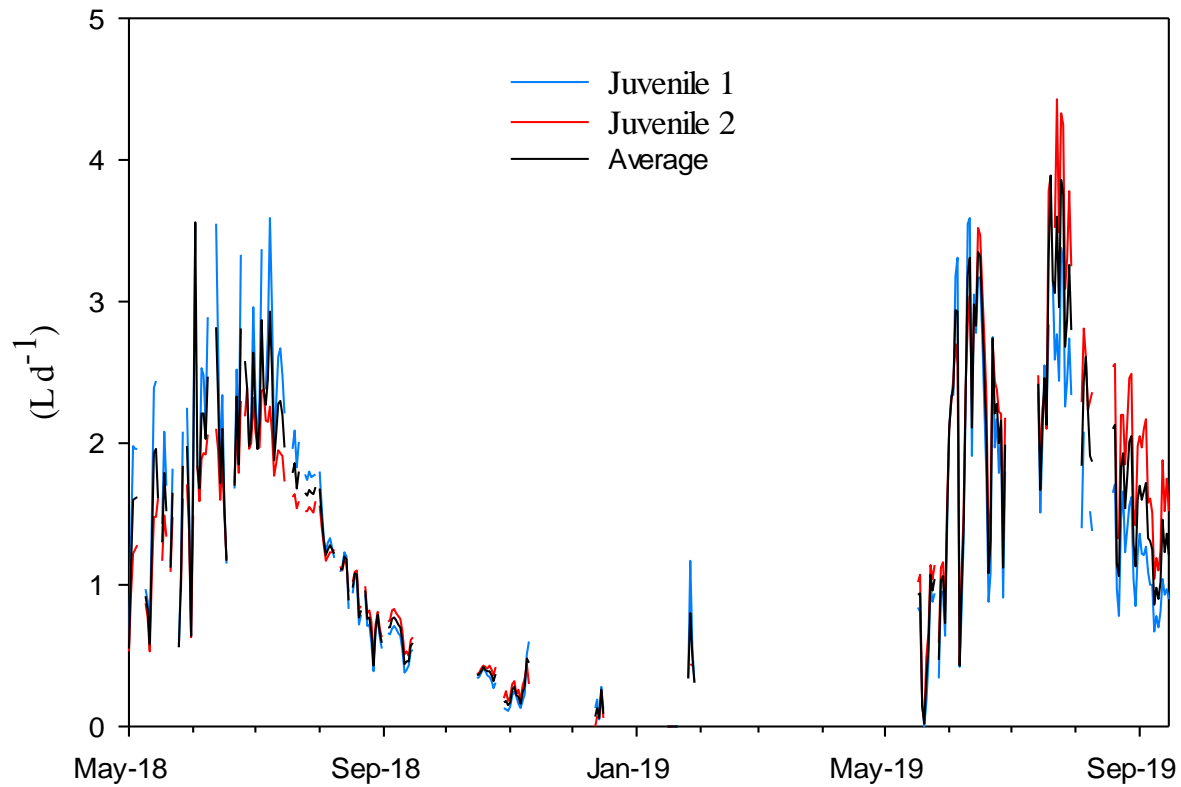
Figure A4.7. Daily-averaged transpiration for the juvenile trees 1, 2 and the average at (Upslope - Jensen) location in the Jensen WS. Missing data are due to lightening-induced equipment failure or battery failure.
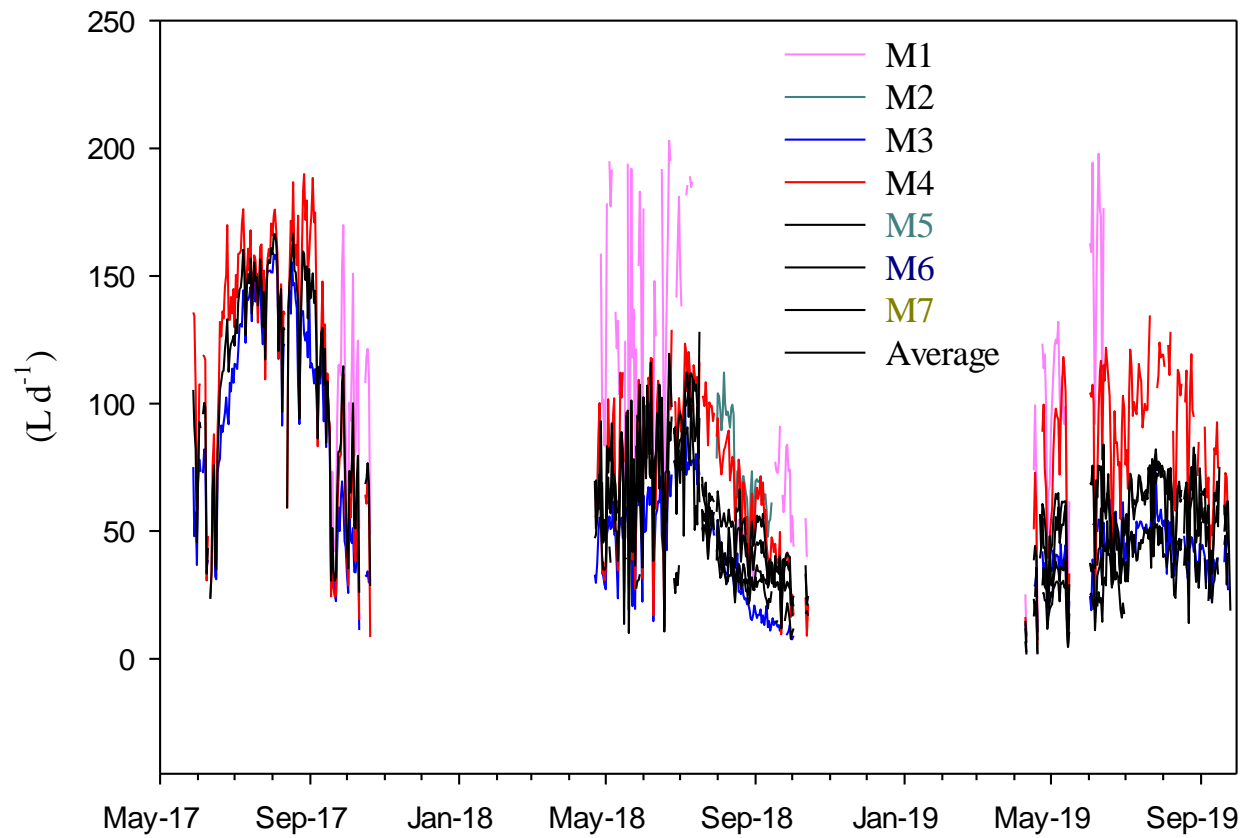
Figure A4.8. Daily-averaged transpiration for the mature trees (M1-M7) and the average at (Valley Jensen) location in the Jensen WS. Missing data are due to lightening-induced equipment failure or battery failure.
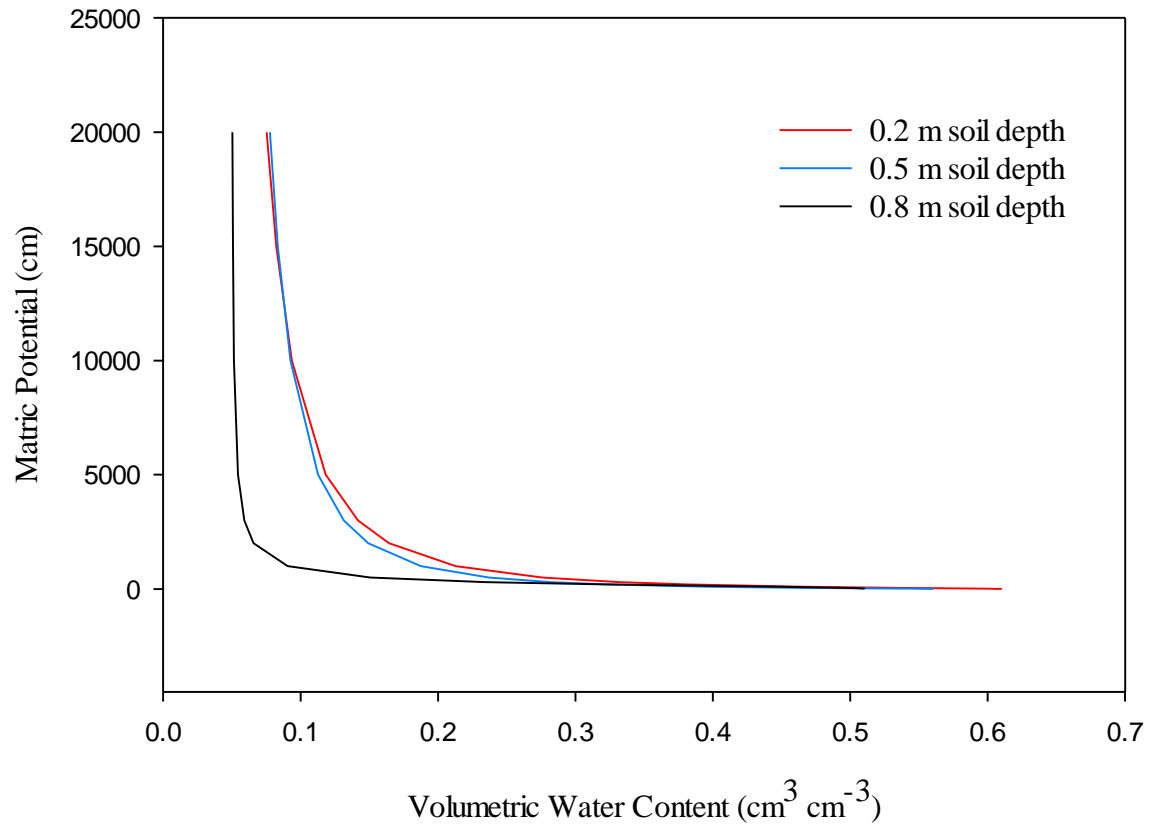
Figure A4.9. Relationship between volumetric water content and matric potential in each of the three depths in the (Mays-East) location in Mays WS.
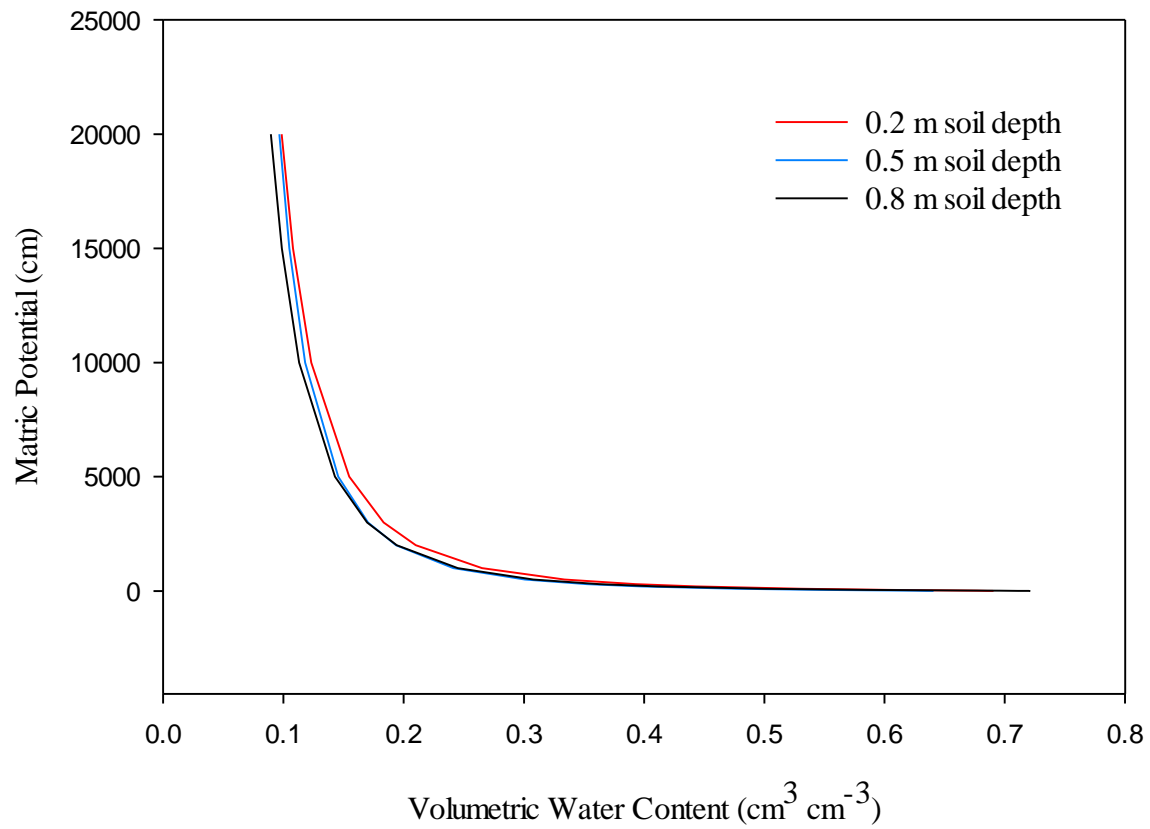
Figure A4.10. Relationship between volumetric water content and matric potential in each of the three depths in the (Mays-West) location in Mays WS.
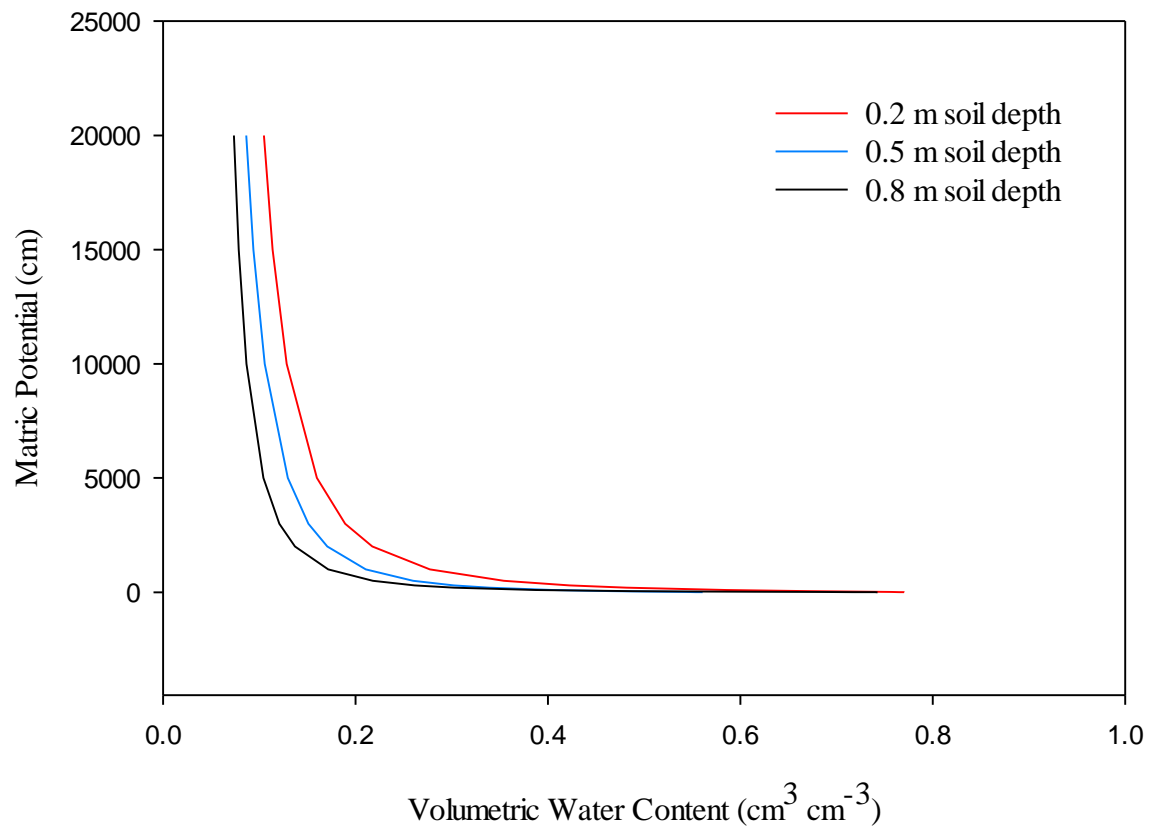
Figure A4.11. Relationship between volumetric water content and matric potential in each of the three depths in the (Jensen-Valley) location at under-canopy zone in Jensen WS.
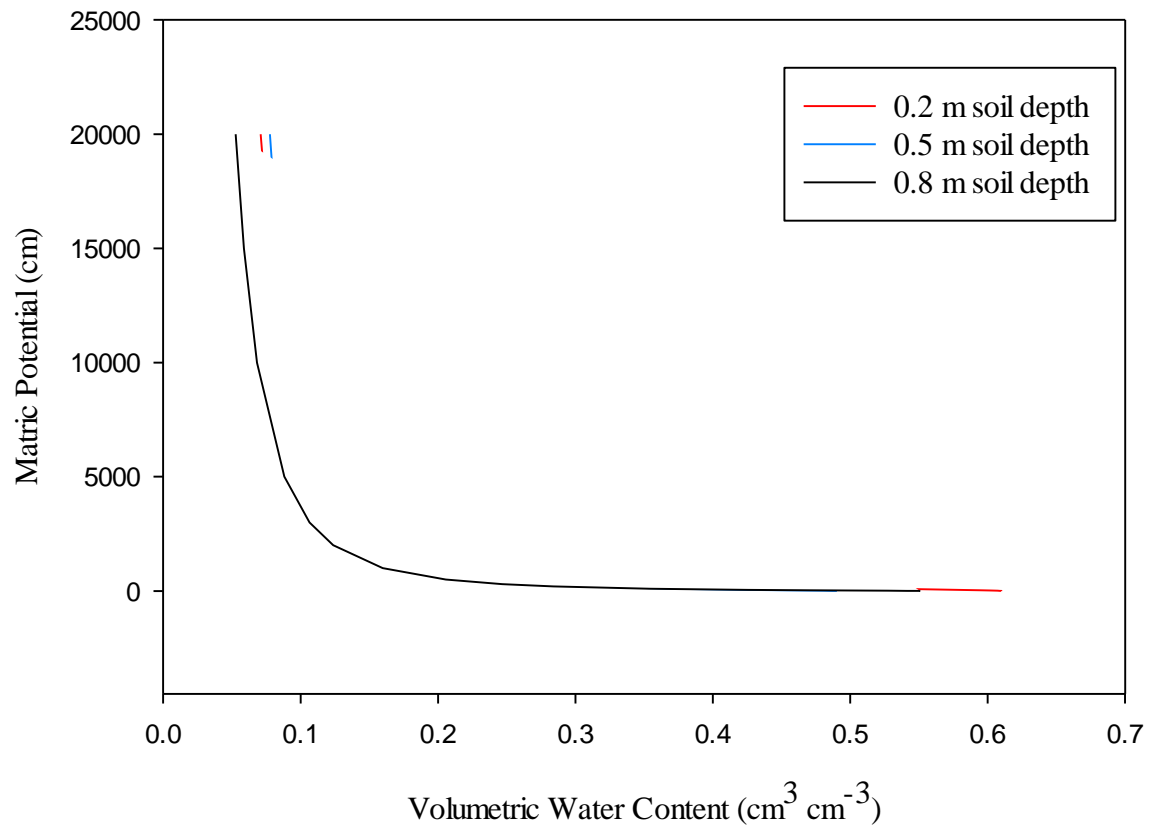
Figure A4.12. Relationship between volumetric water content and matric potential in each of the three depths in the (Jensen-Valley) location at inter-canopy zone in Jensen WS.

Figure A4.13. Relationship between volumetric water content and matric potential in each of the three depths in the (Jensen-Upslope) location at under-canopy zone in Jensen WS.
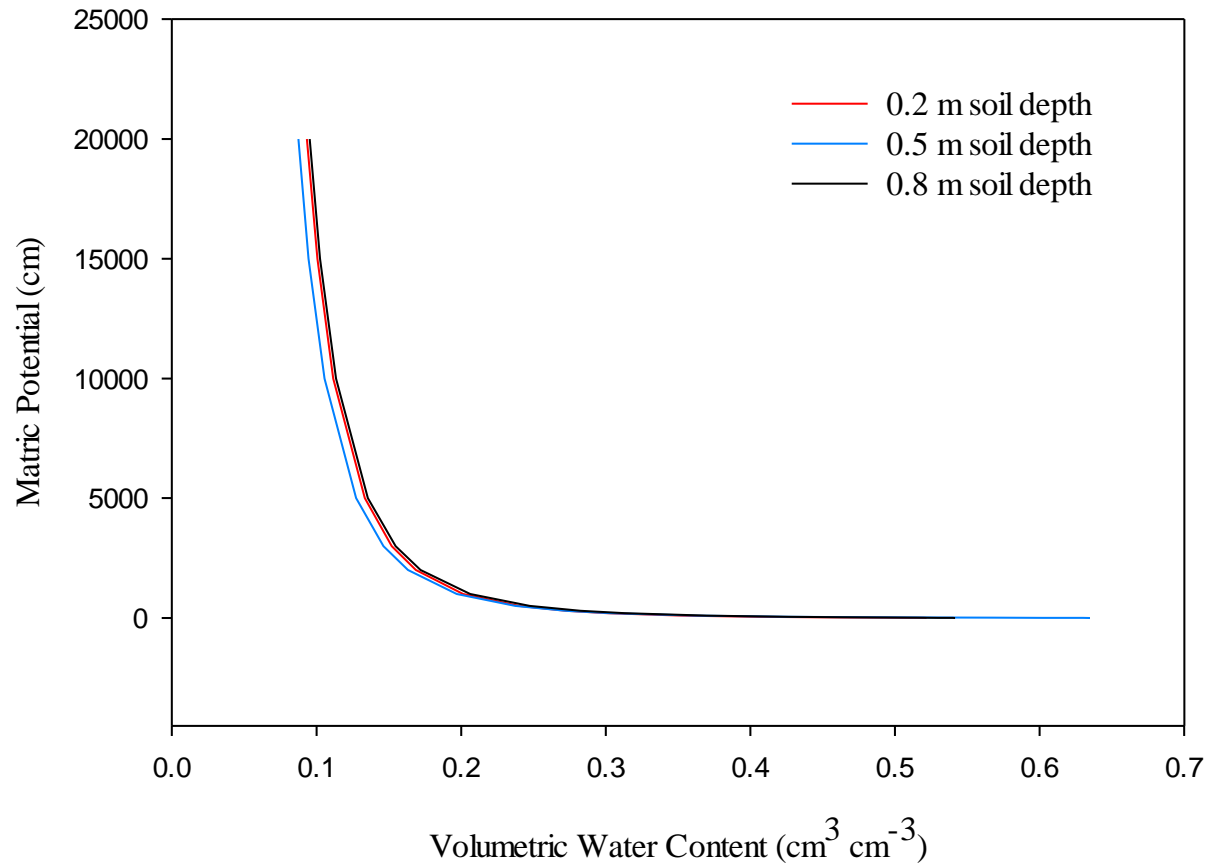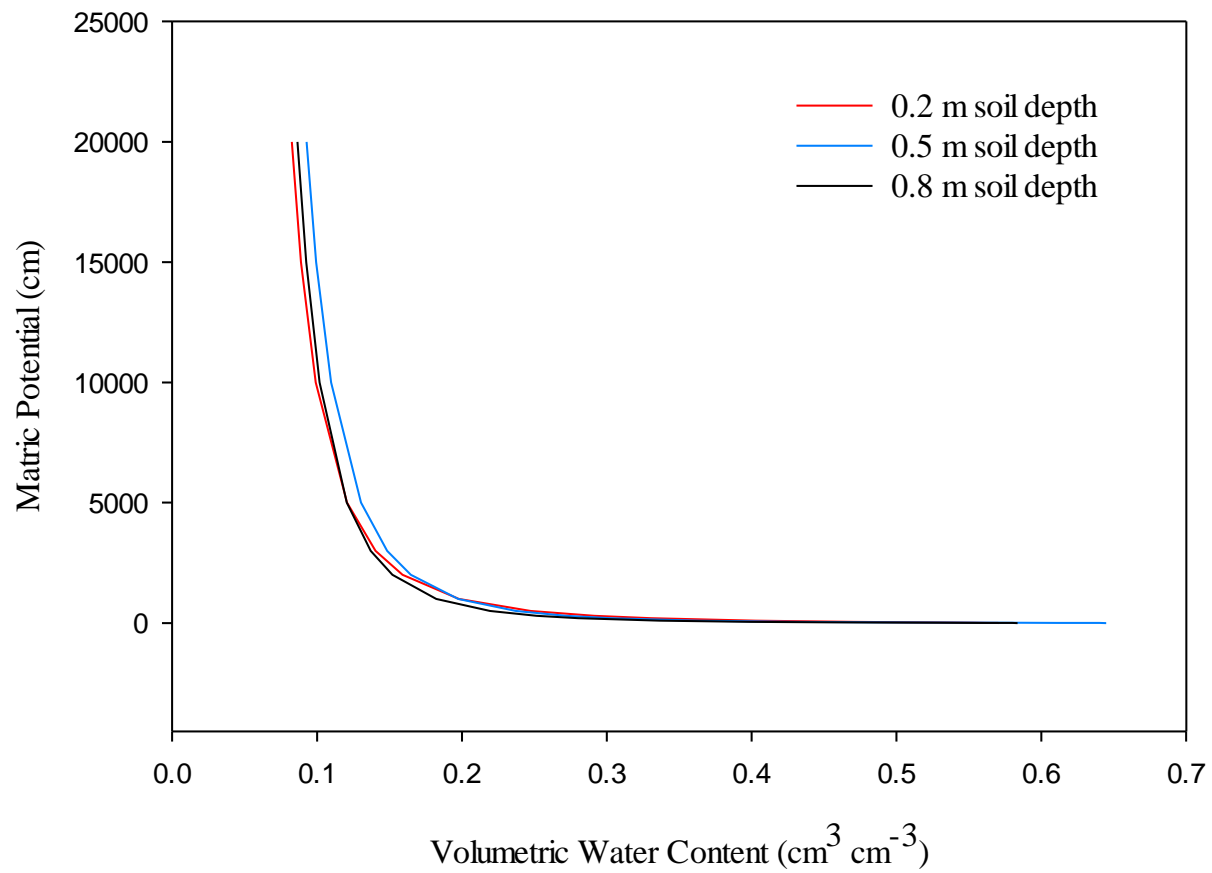
Figure A4.14. Relationship between volumetric water content and matric potential in each of the three depths in the (Jensen-Upslope) location at inter-canopy zone in Jensen WS.
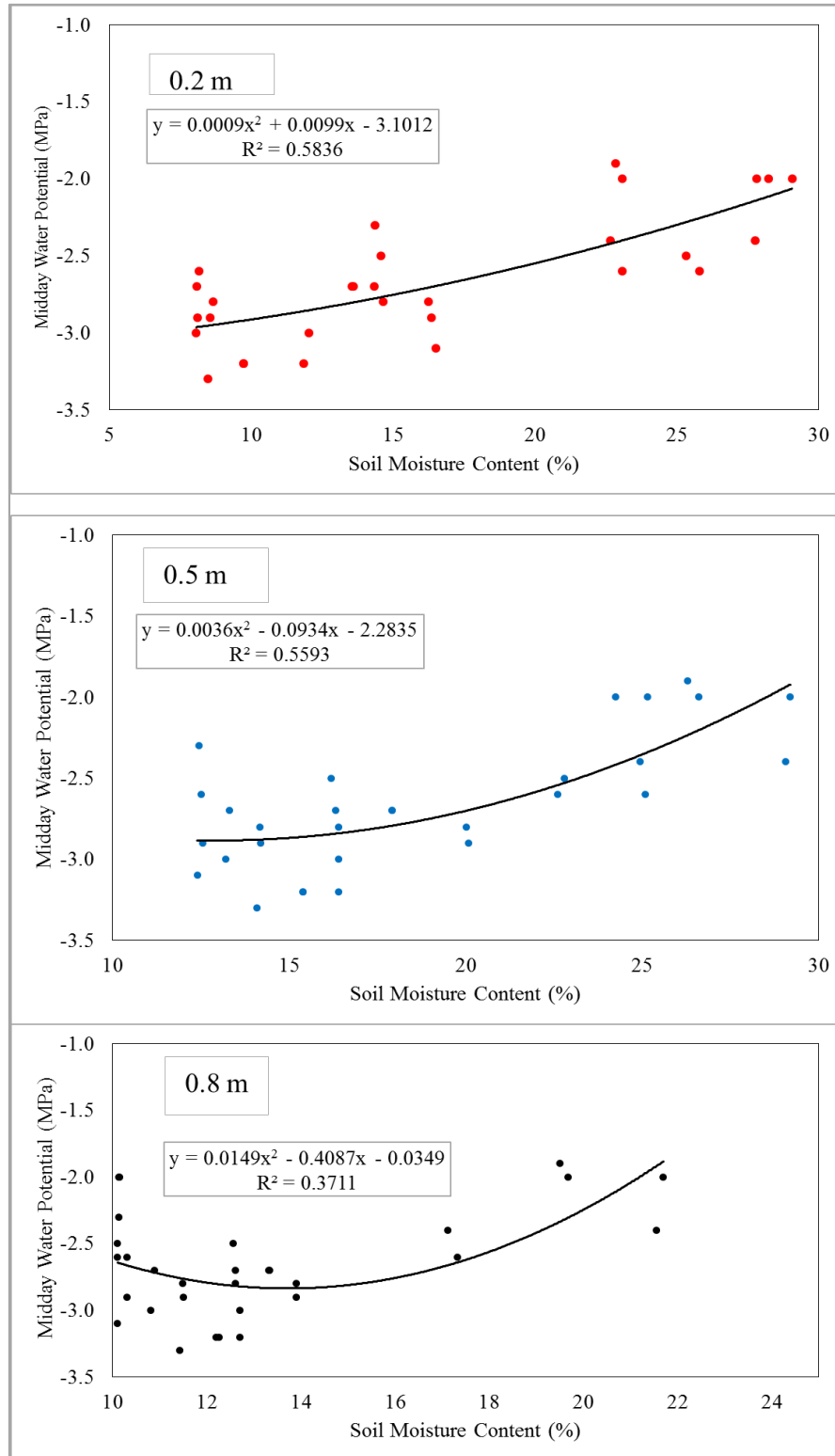
Figure A4.15. Polynomial regression analyses performed to determine relationships between saplings midday leaf water potential (MPa) and soil moisture content (%) in each of the three depths in the (Mays-East) location in Mays WS.

Figure A4.16. Polynomial regression analyses performed to determine relationships between saplings midday leaf water potential (MPa) and soil moisture content (%) in each of the three depths in the (Mays-West) location in Mays WS.
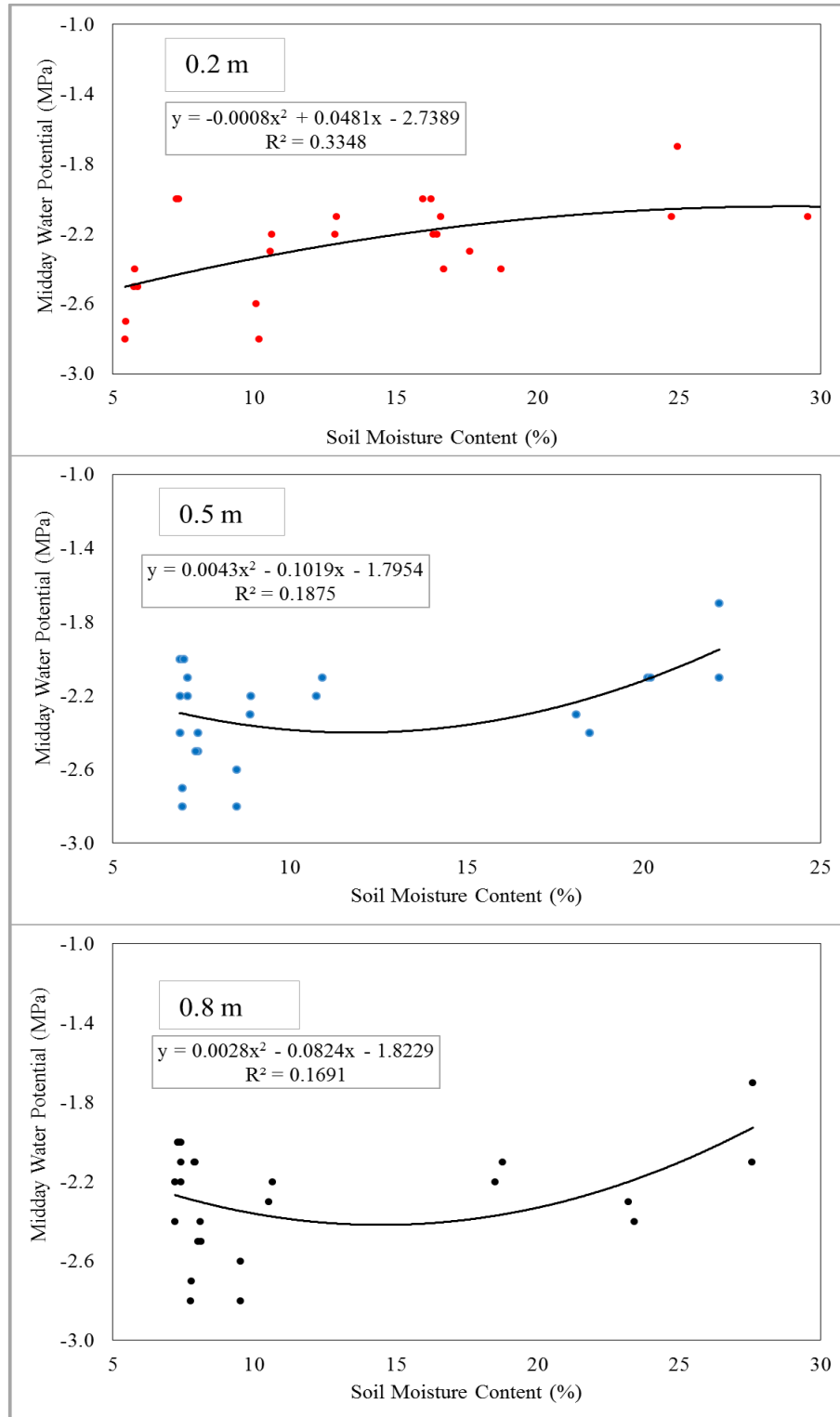
Figure A4.17. Polynomial regression analyses performed to determine relationships between mature midday leaf water potential (MPa) and soil moisture content (%) in each of the three depths in the (Jensen-Valley) location in Jensen WS.
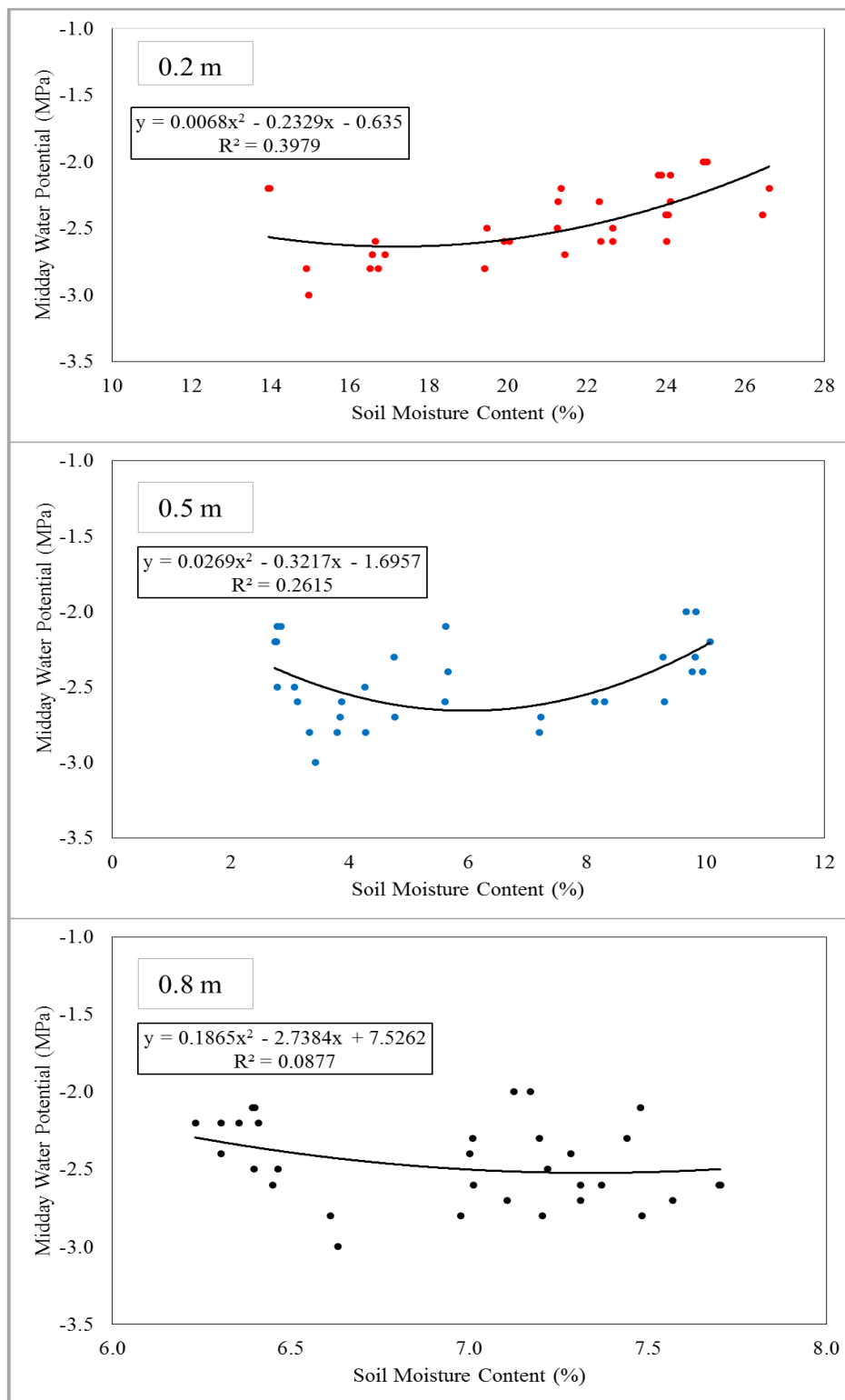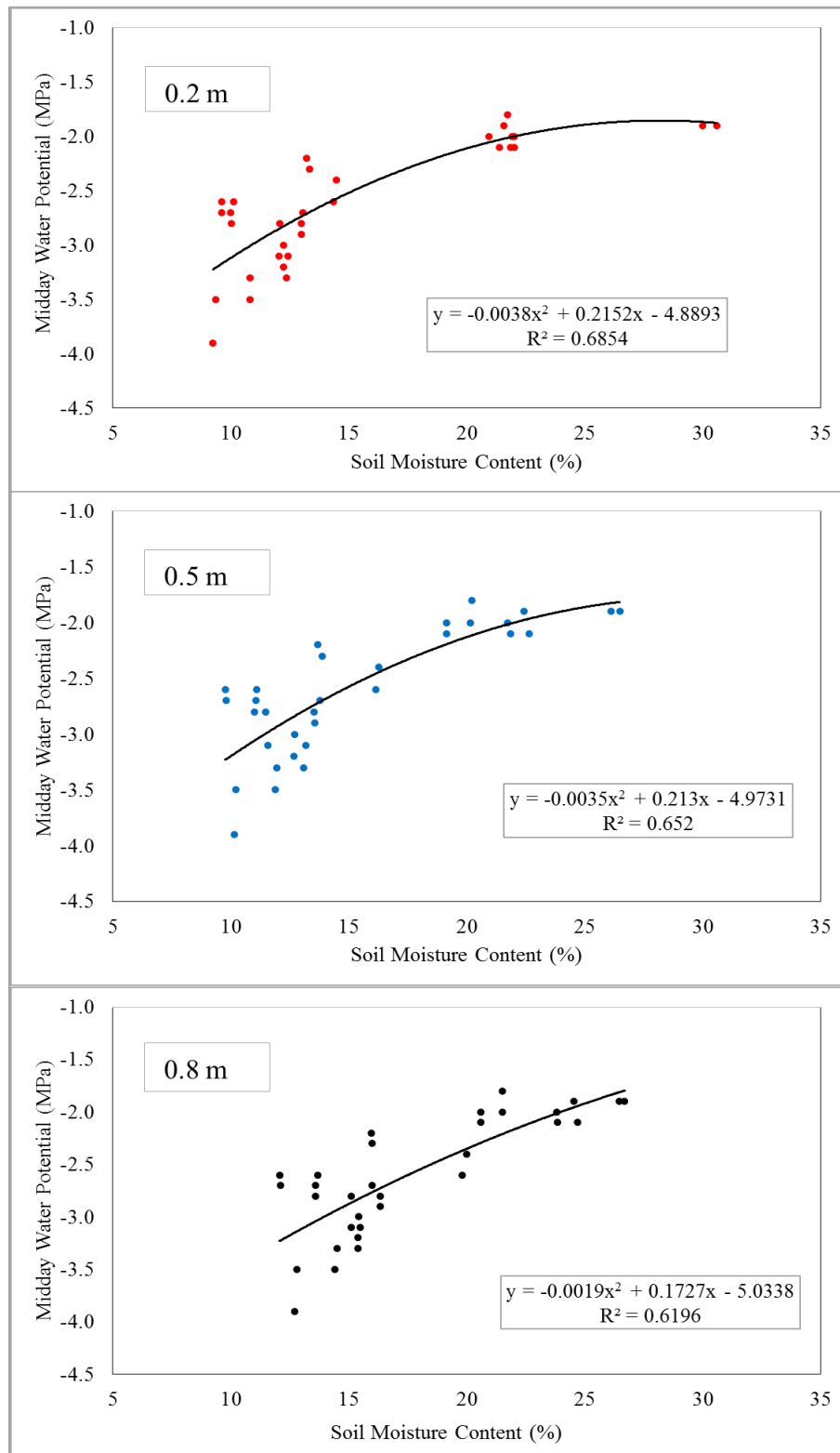
Figure A4.18. Polynomial regression analyses performed to determine relationships between juvenile midday leaf water potential (MPa) and soil moisture content (%) in each of the three depths in the (Jensen-Upslope) location in Jensen WS.

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Table A2.1. Aboveground carbon and nitrogen concentrations (%), and C/N ratio obtained by laboratory analysis of western juniper (*Juniperus occidentalis*), main shrub species, grasses, and litter of the study area, central Oregon, USA.

| Biomass samples | C (%) | N (%) | C/N ratio |
|---|---|---|---|
| Mature western juniper | | | |
| | 51.9 | 0.97 | 53.5 |
| | 49.5 | 0.91 | 54.9 |
| | 52.5 | 1.09 | 48.3 |
| | 51.0 | 0.90 | 56.6 |
| | 51.2 | 0.97 | 53.0 |
| Average | **51.2** | **0.97** | **53.0** |
| Regrowth western juniper | | | |
| | 52.6 | 0.81 | 64.9 |
| | 50.7 | 0.69 | 74.0 |
| | 50.4 | 0.96 | 52.6 |
| | 50.2 | 0.72 | 70.0 |
| | 51.5 | 0.76 | 68.0 |
| Average | **51.1** | **0.79** | **65.0** |
| *Artemisia tridentata* | | | |
| | 48.3 | 0.95 | 50.7 |
| | 48.5 | 1.54 | 31.4 |
| | 46.5 | 0.93 | 50.1 |
| | 47.5 | 0.86 | 55.5 |
| | 47.9 | 1.01 | 47.3 |
| Average | **47.7** | **1.06** | **45.1** |
| *Purshia tridentata* | | | |
| | 46.5 | 1.25 | 37.1 |
| | 46.9 | 1.60 | 29.4 |
| | 46.9 | 1.31 | 35.9 |
| | 46.6 | 1.42 | 32.7 |
| | 46.4 | 0.97 | 47.7 |
| Average | **46.7** | **1.31** | **35.6** |
| *Eriogonum fasciculatum* | | | |
| | 44.7 | 1.10 | 40.8 |
| | 44.1 | 0.77 | 57.2 |
| | 43.4 | 0.86 | 50.4 |
| | 43.9 | 0.70 | 63.2 |
| | 40.6 | 0.69 | 59.3 |
| Average | **43.3** | **0.82** | **52.8** |
| *Ericameria nauseosa* | | | |
| | 48.1 | 1.02 | 47.1 |
| | 46.2 | 0.91 | 50.6 |
| | 48.0 | 0.91 | 52.7 |
| | 46.5 | 0.88 | 53.1 |
| | 46.3 | 0.88 | 52.5 |
| Average | **47.0** | **0.92** | **51.2** |

| Grasses | | | |
|---|---|---|---|
| | 41.1 | 0.52 | 79.4 |
| | 39.8 | 0.62 | 64.6 |
| | 41.0 | 0.58 | 70.1 |
| | 38.4 | 0.73 | 52.4 |
| | 38.5 | 0.61 | 62.8 |
| Average | **39.7** | **0.61** | **65.9** |
| Litter (mainly western juniper debris) | | | |
| | 44.8 | 1.06 | 42.5 |
| | 45.7 | 0.65 | 70.2 |
| | 47.3 | 0.51 | 93.7 |
| | 47.2 | 0.47 | 101.4 |
| | 47.2 | 0.42 | 112.9 |
| Average | **46.4** | **0.62** | **75.0** |

Table A2.2. Root carbon and nitrogen concentrations (%), and C/N ratio obtained by laboratory analysis of western juniper (*Juniperus occidentalis*), shrubs, and grasses of the study area, central Oregon, USA.

| Biomass samples | C (%) | N (%) | C/N ratio |
|---|---|---|---|
| Mature western juniper | | | |
| | 45.9 | 0.51 | 90.2 |
| | 44.6 | 0.18 | 242.2 |
| | 46.3 | 0.36 | 128.0 |
| | 44.8 | 0.68 | 65.9 |
| | 40.2 | 0.32 | 126.3 |
| Average | **44.36** | **0.41** | **108.0** |
| Western juniper stumps | | | |
| | 42.5 | 0.66 | 64.2 |
| | 44.7 | 0.34 | 133.5 |
| | 42.1 | 0.53 | 80.1 |
| | 46.7 | 1.02 | 45.6 |
| | 48.7 | 0.48 | 102.6 |
| Average | **44.95** | **0.60** | **74.4** |
| Regrowth western juniper | | | |
| | 46.8 | 0.45 | 103.2 |
| | 44.8 | 0.50 | 89.5 |
| | 47.5 | 0.36 | 132.8 |
| | 44.1 | 0.70 | 62.8 |
| | 42.8 | 0.72 | 59.6 |
| Average | **45.19** | **0.55** | **82.7** |
| Shrubs | | | |
| | 39.2 | 1.00 | 39.1 |
| | 36.4 | 1.09 | 33.3 |
| | 40.7 | 0.51 | 80.5 |
| | 44.5 | 0.60 | 74.1 |
| | 33.2 | 1.06 | 31.2 |
| Average | **38.80** | **0.85** | **45.5** |
| Grasses | | | |
| | 37.6 | 0.85 | 44.5 |
| | 31.8 | 0.93 | 34.4 |
| | 39.6 | 1.13 | 34.9 |
| | 41.4 | 0.73 | 56.9 |
| | 37.6 | 1.21 | 31.1 |
| Average | **37.61** | **0.97** | **38.8** |

Table A4.1. Seasonal precipitation (mm month$^{-1}$) for three years (2017-2019) recorded by onsite instrumentation.

| Season | Year 2017 | Year 2018 | Year 2019 |
|--------|-----------|-----------|-----------|
| Winter | 168.9 | 61.0 | 108.0 |
| Spring | 75.0 | 103.5 | 61.9 |
| Summer | 14.6 | 5.0 | 44.7 |
| Fall | 68.8 | 74.7 | N/A |

Table A4.2. Mean differences in transpiration for all juniper growth stages accumlated at a seasonal scale during July 2017 through September 2019. Some periods were not included due to limited data for at least one growth stage.

| Period of Record | Stage 1 | Stage 2 | Mean Difference (1-2) | Significance | 95% Confidence Interval | |
|---|---|---|---|---|---|---|
| | | | | | Lower Bound | Upper Bound |
| 1 July to 30 September 2017 | Mature | Sapling | 124.8* | <0.001 | 117.72 | 131.87 |
| 1 October to 13 November 2017 | Mature | Sapling | 61.2* | <0.001 | 50.31 | 72.06 |
| 15 April to 30 June 2018 | Mature | Juvenile | 73.4* | <0.001 | 68.22 | 78.58 |
| | Mature | Sapling | 74.1* | <0.001 | 68.88 | 79.23 |
| | Juvenile | Sapling | 0.66* | <0.001 | 0.43 | 0.88 |
| 1 July to 30 September 2018 | Mature | Juvenile | 54.3* | <0.001 | 49.35 | 59.25 |
| | Mature | Sapling | 54.79* | <0.001 | 49.85 | 59.94 |
| | Juvenile | Sapling | 0.49* | <0.001 | 0.32 | 0.66 |
| 1 October to 31 December 2018 | Mature | Juvenile | 21.6* | <0.001 | 14.21 | 29.00 |
| | Mature | Sapling | 21.6* | <0.001 | 14.24 | 29.03 |
| | Juvenile | Sapling | 0.03 | 0.284 | -0.026 | 0.088 |
| 1 April to 30 June 2019 | Mature | Juvenile | 49.3* | <0.001 | 44.96 | 53.59 |
| | Mature | Sapling | 50.2* | <0.001 | 45.92 | 54.53 |
| | Juvenile | Sapling | 0.96* | <0.001 | 0.64 | 1.27 |
| 1 July to September 30 2019 | Mature | Juvenile | 56.1* | <0.001 | 53.29 | 58.97 |
| | Mature | Sapling | 57.1* | <0.001 | 54.29 | 59.96 |
| | Juvenile | Sapling | 1.00* | <0.001 | 0.76 | 1.24 |

Table A4.3. Analysis of variance (ANOVA) for monthly mean (± SE) soil water content, 0-80 cm ($\theta_{tot}$) (%) by year in (Mays-East) location in Mays WS. N/A = Data not available. Means with the same lowercase letters (a, b, c) are not significantly different ($P \geq 0.05$).

| Month | Year 2017 | Year 2018 | Year 2019 |
|---|---|---|---|
| January | 11.9 (± 0.06) c | 14.5 (± 0.30) a | 12.8 (± 0.12) b |
| February | 15.7 (± 0.65) b | 16.9 (± 0.15) b | 18.3 (± 0.22) a |
| March | 25.5 (± 0.53) a | 19.3 (± 0.50) c | 22.6 (± 0.65) b |
| April | 22.5 (± 0.07) a | 22.5 (± 0.07) b | 25.8 (± 0.11) a |
| May | 23.9 (± 0.20) a | 21.1 (± 0.17) b | 23.6 (± 0.12) a |
| June | 18.7 (± 0.23) b | 17.1 (± 0.29) c | 21.4 (± 0.36) a |
| July | 15.7 (± 0.11) b | 14.6 (± 0.05) c | 16.6 (± 0.11) a |
| August | 14.2 (± 0.07) b | 13.6 (± 0.07) c | 15.1 (± 0.06) a |
| September | 13.1 (± 0.10) b | 12.5 (± 0.06) c | 14.7 (± 0.06) a |
| October | 11.9 (± 0.05) b | 11.5 (± 0.04) c | 13.5 (± 0.05) a |
| November | 11.7 (± 0.04) a | 10.8 (± 0.05) b | N/A |
| December | 11.3 (± 0.06) b | 12.1 (± 0.37) a | N/A |

Table A4.4. Analysis of variance (ANOVA) for monthly mean (± SE) soil water content, 0-80 cm ($\theta_{tot}$) (%) by year in (Jensen-Valley) location in Jensen WS. N/A = Data not available. Means with the same lowercase letters (a, b, c) are not significantly different ($P \geq 0.05$).

| Month | Year 2017 | Year 2018 | Year 2019 |
|---|---|---|---|
| January | 6.6 (± 0.12) c | 7.4 (± 0.01) b | 10.3 (± 0.04) a |
| February | 11.3 (± 0.53) a | 8.3 (± 0.10) b | 10.8 (± 0.04) a |
| March | 13.6 (± 0.15) a | 10.1 (± 0.16) c | 12.2 (± 0.32) b |
| April | 14.5 (± 0.08) a | 10.7 (± 0.02) c | 14.2 (± 0.03) b |
| May | 15.2 (± 0.06) a | 11.7 (± 0.33) c | 14.2 (± 0.10) b |
| June | 13.1 (± 0.14) b | 11.9 (± 0.33) c | 14.0 (± 0.09) a |
| July | 11.6 (± 0.06) c | 12.4 (± 0.07) b | 13.0 (± 0.04) a |
| August | 10.6 (± 0.04) c | 11.2 (± 0.06) b | 12.0 (± 0.07) a |
| September | 9.7 (± 0.09) c | 10.3 (± 0.06) b | 12.0 (± 0.25) a |
| October | 8.3 (± 0.05) c | 9.2 (± 0.07) b | 11.9 (± 0.07) a |
| November | 7.7 (± 0.02) b | 8.3 (± 0.05) a | N/A |
| December | 7.3 (± 0.01) b | 8.5 (± 0.18) a | N/A |

Table A4.5. Analysis of variance (ANOVA) for monthly mean (± SE) soil water content, 0-80 cm ($\theta_{tot}$) (%) by year in (Jensen-Upslope) location in Jensen WS. N/A = Data not available. Means with the same lowercase letters (a, b, c) are not significantly different ($P \geq 0.05$).

| Month | Year 2017 | Year 2018 | Year 2019 |
|---|---|---|---|
| January | 14.0 (± 0.03) b | 12.1 (± 0.07) b | 17.0 (± 1.01) a |
| February | 15.6 (± 0.22) b | 12.8 (± 0.19) c | 21.2 (± 0.08) a |
| March | 26.4 (± 0.90) a | 12.8 (± 0.23) c | 20.8 (± 0.08) b |
| April | 27.1 (± 0.09) a | 17.9 (± 0.09) b | 27.1 (± 0.31) a |
| May | 24.8 (± 0.21) a | 17.1 (± 0.14) c | 23.8 (± 0.15) b |
| June | 20.4 (± 0.22) b | 15.6 (± 0.10) c | 22.1 (± 0.31) a |
| July | N/A | 14.4 (± 0.05) b | 16.6 (± 0.22) a |
| August | 14.4 (± 0.07) a | 13.8 (± 0.06) b | 14.3 (± 0.05) a |
| September | 13.4 (± 0.08) b | 12.4 (± 0.06) c | 13.9 (± 0.07) a |
| October | 12.7 (± 0.15) b | 11.6 (± 0.06) c | 14.2 (± 0.15) a |
| November | 15.3 (± 0.18) a | 10.8 (± 0.06) b | N/A |
| December | 12.1 (± 0.26) a | 10.8 (± 0.10) b | N/A |

Table A4.6. VanGenuchten equation parameters for soils collected from different locations and depths from both watersheds. SCL= sandy clay loam soil, SL= sandy clay, and L= loam.

| Location | Soil Depth (m) | Soil Type | α | N | $\theta_S$ | $\theta_r$ | measured $\theta_r$ | M |
|---|---|---|---|---|---|---|---|---|
| Jensen-Upslope UC | 0.2 | SCL | 0.04033 | 1.25711 | 0.52184 | 0 | 10 | 0.204525 |
| | 0.5 | SL | 0.07436 | 1.27148 | 0.63486 | 0 | 9.8 | 0.213515 |
| | 0.8 | SCL | 0.03505 | 1.28005 | 0.54156 | 0.0105 | 10 | 0.218781 |
| Jensen-Upslope IC | 0.2 | SL | 0.02161 | 1.37462 | 0.57077 | 0.02644 | 9.7 | 0.272526 |
| | 0.5 | SCL | 0.0742 | 1.28907 | 0.64482 | 0.01666 | 10 | 0.224247 |
| | 0.8 | SCL | 0.06478 | 1.29827 | 0.58364 | 0.01998 | 9.9 | 0.229744 |
| Jensen-Valley UC | 0.2 | L | 0.01453 | 1.4105 | 0.77058 | 0.0327 | 10.7 | 0.291032 |
| | 0.5 | SL | 0.0221 | 1.32589 | 0.56085 | 0.01075 | 9.8 | 0.24579 |
| | 0.8 | SL | 0.04808 | 1.42507 | 0.74277 | 0.03527 | 9.3 | 0.29828 |
| Jensen-Valley IC | 0.2 | SL | 0.00636 | 1.5563 | 0.61007 | 0.03182 | 8.9 | 0.35745 |
| | 0.5 | SL | 0.01601 | 1.3904 | 0.49041 | 0.02914 | 9.5 | 0.280783 |
| Mays-East | 0.8 | SL | 0.02806 | 1.37058 | 0.55101 | 0 | 4.9 | 0.270382 |
| | 0.2 | SL | 0.01356 | 1.42971 | 0.61039 | 0.0223 | 9.3 | 0.300557 |
| | 0.5 | SL | 0.01957 | 1.41027 | 0.5606 | 0.03201 | 9.5 | 0.290916 |
| | 0.8 | SL | 0.00593 | 2.35654 | 0.51058 | 0.04968 | 3.8 | 0.575649 |
| Mays-West | 0.2 | SL | 0.01524 | 1.3592 | 0.69059 | 0.01147 | 10.1 | 0.264273 |
| | 0.5 | SL | 0.01811 | 1.35384 | 0.6407 | 0.01939 | 10.1 | 0.26136 |
| | 0.8 | L | 0.02437 | 1.33721 | 0.72125 | 0 | 9.9 | 0.252174 |

VanGenuchten equation parameters for soils collected from different locations and depths from both watersheds. SCL= sandy clay loam soil, SL= sandy clay, and L= loam. The equation;

$$\theta = \theta_R + \frac{(\theta_s - \theta_R)}{\left[1 + (\alpha h)^N\right]^M}$$

where;

$\theta$ = the volumetric soil water content, (cm$^3$ cm$^{-3}$);

$\theta_s$ = the saturated soil water content, (cm$^3$ cm$^{-3}$);

$\theta_r$ = the residual soil water content, (cm$^3$ cm$^{-3}$);

α = parameter corresponding approximately to the inverse of the air-entry value, (cm$^{-1}$);

h = the capillary pressure (cm);

M and N = dimensionless parameters where $M = 1 - \frac{1}{N}$

Table A4.7. Polynomial regression analyses performed to determine relationships between daily transpiration (L d$^{-1}$) and soil water content levels for western juniper during the period July throughout October 2017.

| Juniper Growth Stages | Soil Depth (m) | Polynomial Equation | $R^2$ |
|---|---|---|---|
| (Mays-East) Saplings | 0.2 | $y= -0.1277x^2 +3.399x - 19.589$ | 0.638 |
| | 0.5 | $y= -0.1274x^2 + 4.763x - 41.543$ | 0.697 |
| | 0.8 | $y= -0.9379x^2 + 24.226x -153.62$ | 0.701 |
| Mature Juniper | 0.2 | $y= -1.4303x^2 + 69.232x - 687.92$ | 0.657 |
| | 0.5 | $y= -7.2293x^2 + 109.07x - 145.71$ | 0.568 |
| | 0.8 | $y= -42.365x^2 + 713.21x - 2853.3$ | 0.554 |

Table A4.8. Polynomial regression analyses performed to determine relationships between daily transpiration (L d$^{-1}$) and soil water content levels for western juniper during the period April throughout October 2018.

| Juniper Growth Stages | Soil Depth (m) | Polynomial Equation | $R^2$ |
|---|---|---|---|
| (Mays-East) Saplings | 0.2 | $y = -0.0103x^2 + 0.3743x - 2.125$ | 0.655 |
| | 0.5 | $y = -0.0308x^2 + 1.2882x - 11.849$ | 0.575 |
| | 0.8 | $y = -0.1049x^2 + 3.103x - 21.493$ | 0.430 |
| Juvenile Juniper | 0.2 | $y = -0.0302x^2 + 1.0599x - 7.2433$ | 0.473 |
| | 0.5 | $y = -0.1228x^2 + 3.7073x - 26.08$ | 0.526 |
| | 0.8 | $y = 0.2388x^2 - 6.4038x + 43.418$ | 0.537 |
| Mature Juniper | 0.2 | $y = -0.6515x^2 + 36.184x - 418.44$ | 0.577 |
| | 0.5 | $y = 2.1827x^2 - 18.188x + 90.301$ | 0.274 |
| | 0.8 | $y = 111.4x^2 - 1576.1x + 5628.3$ | 0.112 |

Table A4.9. Polynomial regression analyses performed to determine relationships between daily transpiration (L d$^{-1}$) and soil water content levels for western juniper during the period May throughout September 2019.

| Juniper Growth Stages | Soil Depth (m) | Polynomial Equation | $R^2$ |
|---|---|---|---|
| (Mays-East) Saplings | 0.2 | $y = -0.0085x^2 + 0.3343x - 1.7374$ | 0.078 |
| | 0.5 | $y = -0.0181x^2 + 0.8047x - 7.3581$ | 0.117 |
| | 0.8 | $y = -0.0453x^2 + 1.464x - 10.171$ | 0.162 |
| (Mays-West) Saplings | 0.2 | $y = -0.0004x^2\ 0.0018x + 0.9693$ | 0.194 |
| | 0.5 | $y = -0.0072x^2 + 0.209x - 0.4138$ | 0.165 |
| | 0.8 | $y = -0.0027x^2 + 0.0911x + 0.211$ | 0.139 |
| Juvenile Juniper | 0.2 | $y = -0.0082x^2 + 0.2978x - 0.5316$ | 0.057 |
| | 0.5 | $y = -0.0561x^2 + 1.9709x - 14.514$ | 0.153 |
| | 0.8 | $y = -0.1015x^2 + 4.0607x - 37.368$ | 0.347 |
| Mature Juniper | 0.2 | $y = -0.1452x^2 + 6.5011x - 14.614$ | 0.007 |
| | 0.5 | $y = -3.5532x^2 + 63.29x - 222.06$ | 0.036 |
| | 0.8 | $y = 16.017x^2 - 224.67x + 841.43$ | 0.037 |