AN ABSTRACT OF THE THESIS OF

<u>Christopher Miwa</u> for the degree of <u>Master of Science</u> in <u>Forest Resources</u>
presented on <u>December 6, 2007</u>.

Title: <u>Persistence of Western Juniper Resource Islands Following Canopy Removal.</u>

Abstract approved:_____

Ron Reuter

Western juniper (*Juniperus occidentalis*) cover has more than doubled within
the last century and currently occupies over 9 million acres in the Intermountain West.
Encroachment has altered the spatial distribution of soil nutrients and plants in these
systems, forming nutrient enriched 'resource islands,' under tree canopies. The
purpose of this study was to determine the persistence of resource island
characteristics after restoration treatment (tree removal).

The study site was a BLM grazing allotment in Eastern Oregon where trees
had been cut 1, 8 and 15 years ago. In each age class and in uncut western juniper
woodlands juniper stumps or trees were randomly selected for sampling. At each
bole three radial transects, set at 120º from each other, were marked and soil cores
were collected to 5-cm depth at distances of 50, 100, 150 and 300-cm from the bole
then combined to a single composite sample per distance class. Samples were
analyzed for total C and N, soluble P, K, Ca, Fe, Si, Al and Na, inorganic NH_4 and
NO_3, pH and gravimetric water content.

Fifteen years after canopy removal there was still strong evidence of western juniper resource islands. Canopy soils were generally significantly higher in C_{tot}, N_{tot}, NO_3^-, P, K and Ca compared to intercanopy soils in all treatments; however, accumulations for several elements appeared to degrade with time since canopy removal, and at different rates. We attribute resource island persistence to deep litter mats beneath relic canopies. Due to the strong zonal distribution of understory vegetation in juniper duff and bare zones these nutrient enriched areas will continue to influence long-term successional pathways and should be taken into account for restoration purposes, such as seeding of forbs and grasses.

Persistence of Western Juniper Resource Islands Following Canopy Removal

by
Christopher Miwa

A THESIS

submitted to

Oregon State University

in partial fulfillment of
the requirements for the
degree of

Master of Science

Presented December 6, 2007
Commencement June 2008

Master of Science thesis of Christopher Miwa presented on December 6, 2007.

APPROVED:

_____

Major Professor, representing Forest Resources

_____

Head of the Department of Forest Resources

_____

Dean of the Graduate School

_____

Christopher Miwa, Author

ACKNOWLEDGEMENTS

CONTRIBUTION OF AUTHORS

Dr. Ron Reuter helped me in all phases of field sampling including equipment gathering, experimental design and collection. Ron reviewed several thesis drafts and helped guide my interpretation of the results.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

1. Ecological Concerns on Arid Lands and Western Juniper Encroachment: A Review

1.1 Desertification of Arid Environments

Definition and Extent of Desertification

Arid lands which include semi-arid, arid and sub-humid zones currently cover 47 percent of the earth's surface (UNEP, 1997). These areas, commonly called 'drylands' are characterized by low annual rainfall, less than 250 mm, where plant uptake and evapotranspiration rates exceed precipitation during part of, or the entire year (Kassas, 1995). Only sparse vegetation can be supported in these regions; as such there are often very limited populations of humans and animals. Drylands can be found on every continent and include a diverse array of ecosystems including deserts, savannas and tropical dry forests. Human activity coupled by climate change has resulted in the degradation, or 'desertification,' of these lands which has become an increasing global concern.

The definition of desertification has evolved over time due to uncertainty about the causes and implication of this process. However, a commonly used description is that of The United Nations Convention to Combat Desertification (UNCCD) which defines land degradation as:

> "the reduction or loss, in arid, semi-arid and dry sub-humid areas, of the biological or economic productivity and complexity of rainfed cropland, irrigated cropland, or range, pasture, forest and woodlands resulting from land uses or from a process or combination of processes, including processes arising from human activities and habitation patterns, such as:
> (i) soil erosion caused by wind and/or water;
> (ii) deterioration of the physical, chemical and biological or economic properties
> of soil; and

(iii) long-term loss of natural vegetation" (UNCCD, 1994)

By this definition, desertification does not refer to the expansion of arid systems,

rather the degradation of them by over-exploitation and inappropriate land use.

Human activities commonly tied to desertification include fuelwood harvest,

agriculture, deforestation and land conversion to industry or urban growth (FAO,

1996; UNEP, 2002). In some circumstances the effected lands can no longer be

restored due to cumulative impacts or site vulnerability (Dregne, 1977; Belnap, 2004).

World leaders now recognize that this problem contributes to famine and poverty

(Gisladottir and Stocking, 2005) and puts at risk the over two billion people living in

drylands in more than one hundred countries (Adams and Eswaran, 2000). The Global

Environment Facility (GEF) is an independent financial organization dedicated to

helping source funds for projects to combat global environmental issues. The GEF

has been a key supporter of UNCCD initiatives; currently they are funding projects

totaling almost $350 million to combat land degradation at the regional, national and

global scale.

A New Definition of Desertification

In arid systems soil resources are relatively scarce due to high

evapotranspiration, infrequent rainfall, slow decomposition rates and an inherently low

nutrient content of the soil (Hadley and Szarek, 1981). Noy-Meir (1985) suggests

woody species survive this resource scarcity by altering the distribution of soil

nutrients. A patchy pattern of nutrients is commonly observed in shrubland systems

where nutrient distribution is associated with individual shrub canopies (Schlesinger et

al., 1990). Alternatively, in semi-arid grassland and steppe systems the distribution of

soil organic matter (SOM) and nutrient pools is at a much finer scale, which appears relatively homogenous compared to that found in shrublands (Hook et al., 1991). Shrub encroachment of grassland systems has been associated with reduced understory diversity and productivity, often due to grazing pressure (Brown and Archer, 1999; Holmgren, 2002) and increased soil erosion (Fryrear, 1995), thus Schlesinger et al., (1990) suggest that the process of desertification can be expressed as the change in scale of nutrient patchiness from fine in grasslands, to coarse in shrublands. In other words it can be viewed as the increase in soil resource heterogeneity in space and time due to shrubby and woody encroachment into grassland systems.

Shrubby and woody species that have encroached into grassland systems are predominantly indigenous and were not introduced from foreign lands or even distant locations (Van Auken, 2000). Instead, they are local or adjacent communities whose density and range has increased due to changes in local biotic and abiotic factors primarily due to human activity (Miller and Wigand, 1994). Thus the conversion of grassland to shrubland or forest should not be considered an invasion as is commonly termed for exotic species; the term 'encroachment' is commonly used instead.

Global Trends in Desertification

Overgrazing has contributed to the largest extent of land degradation worldwide with approximately 680 million ha considered degraded (20% of the world's pastures and rangeland) (FAO, 1996; UNEP, 2002). In the United States, rangelands make up nearly 164 million hectares, 21% of total land cover (USDA,

NRCS, 2006). The effects of overgrazing, fire suppression and climate change have been tied to the conversion of grassland to shrubland in the American southwest and other parts of the world (Grover and Musick, 1990; Schlesinger et al., 1990; Hook et al., 1991; Herman et al., 1995; Schlesinger et al., 1996; Schlesinger and Pilmanis, 1998; Reynolds et al., 1997; Bolling and Walker, 2002; Huenneke et al., 2002). Western juniper (*Juniperus occidentalis* Hook) encroachment into sagebrush-steppe systems of the Pacific Northwest (PNW) has resulted from similar causes except it is unique in that it involves the conversion of shrublands to woodlands. To date juniper encroachment has not fully entered the desertification discussion even though it increases soil resource patchiness beyond that of sagebrush (Roberts and Jones, 2000) and has led to equal if not greater ecological impacts.

1.2 Western Juniper

Western Juniper Expansion

The extensive expansion of western juniper (*Juniperus occidentalis*) throughout arid and semi-arid regions of the Pacific Northwest within the last century has been well documented (Miller and Wigand, 1994; Miller and Rose, 1995; Knapp and Soulé, 1998). Currently western juniper occupies 3.5 million acres in central and eastern Oregon (Miller et al., 2005), and over 9 million acres in the Pacific Northwest and Great Basin (Azuma et al., 2005), more than double the land area a century ago (Miller et al., 1987). In central and eastern Oregon western juniper forests have expanded especially fast, they now cover over five times the area they did as recently as the mid 1930s (Gedney et al., 1999).

The expansion of western juniper and other juniper species of the American West has been primarily attributed to anthropogenic factors including domestic livestock grazing and fire suppression, concurrent with human settlement in the 1870s (Miller and Tausch, 2001). Domestic grazing reduced fine fuel loads required to carry and sustain wildfires, a natural control of juniper, allowing juniper woodlands to expand into deeper, more productive soils of the sagebrush steppe (Miller and Rose, 1999). Abiotic factors including mild winters and increased precipitation between 1850 and 1916 (Graumlich, 1987) and rising atmospheric CO_2 levels have also been tied to juniper expansion (Knapp and Soulé, 1996; Knapp et al., 2001).

A Continuing Trend

Presently western juniper is continuing to expand into new areas in the PNW in addition to increasing its canopy cover in currently occupied regions. Miller et al., (2000) caution that the majority of the current expanse of western juniper in Oregon could still be in transition from shrub-steppe to juniper woodland. Gedney et al., (1999) reported that 65% of the then 0.9 million ha of western juniper in Oregon had less than 30% tree canopy cover.

Increases in western juniper density have the greatest impact on plant community composition and structure because of understory losses. Sites with shallow soils (40 to 60 cm) or southern aspects are especially vulnerable to these impacts (Miller and Wigand, 1994; Miller et al., 2000), areas with deeper soils tend to better withstand the influence of western juniper encroachment (Miller and Wigand, 2004). However, without fire to cycle the system back to sagebrush-steppe juniper

establishment has continued to expand. Johnson and Miller (2006) found that the rate of western juniper establishment and density increased along a continuum of elevation and exposure where increases were greatest on northerly exposures and at the highest elevation extent. They suggest that once the encroached site can no longer support fire, environmental factors (elevation, aspect) become important determinants of juniper forest structure and developmental rates.

Understory Diversity and Productivity in Juniper Woodlands

Since the late 1800s, large areas of sagebrush and aspen communities have been overtaken by pinyon and juniper woodlands (Miller and Tausch, 2001; Miller and Wigand, 1994). This has resulted in major shifts in plant and wildlife composition, consequently altering the ecological processes of the effected areas. Although juniper encroachment can eventually lead to undesirable ecological impacts the effects can vary depending on juniper dominance.

Plant communities containing western juniper can range from open stands with a diverse understory including shrubs and grasses to closed woodlands where the tree layer is the dominant control of ecological processes. Miller et al., (2005) classified three phases in this transition:

- Phase I, juniper trees are present but shrubs and herbaceous species are dominant and control ecological processes on the site;
- Phase II, juniper trees are codominant with the understory and all three vegetation layers influence ecological processes on the site;
- Phase III, trees are dominant and are the primary plant layer influencing ecological processes.

In western juniper forests, understory suppression is greatest in Phase III woodlands (closed stands) which can vary between 32 trees/ac on dry sites to 500 trees/ac on cool

moist sites (Miller et al., 2005). These phase III woodlands are reached between 70 and 90 years on cool, wet sites and between 120 and 170 years on dry, warm sites.

The effects of juniper encroachment on understory productivity are not linear. Miller et al., (2000) found that as western juniper cover increased to 50% of its maximum potential cover on plots in southeastern Oregon and northeastern California, mountain big sagebrush (*Artemesia tridentata*) cover was reduced by 80% of its maximum potential cover. As understory cover declines, soil erosion increases in these largely barren intercanopy zones (Rostagno, 1989; Wilcox, 1994). In central Oregon, Roberts and Jones (2000) found that juniper-sagebrush-grass plots had nearly 50% bare space compared to 30% in sagebrush-grass plots.

There is a threshold where these negative ecological impacts become manifested, which is reached as the role of western juniper moves from codominant to dominant (Phase II to Phase III) (Miller et al., 2005). This tipping point is marked by an altered fire regime due to changes in fuel loads and loss of native seed pools. Once a Phase III western juniper woodland has been reached the area will not revert to its original vegetation assemblage after juniper removal without considerable energy input.

Mechanisms of Understory Suppression

It is thought that western juniper competitively excludes understory vegetation by lateral root expansion and alteration of the soil environment. Tiedemann and Klemmedson (1995) recorded lateral roots up to four canopy radii to a depth of about 30 cm which may be able to exploit soil nutrients and moisture from these intercanopy

areas. Some studies suggest that canopy soil enrichment results in intercanopy reductions (Charley and West, 1975; Klopatek, 1987); however, Klemmedson and Tiedemann (2000) found that canopy soil enrichment did not result in significant intercanopy losses for several plant-critical nutrients incuding C_{org}, P, S, and K. They suggest that canopy gains may come in some measure from intercanopy soils, but other sources may be more important. Those other sources could include nutrient enriched stem and throughflow (Whitford et al., 1995) or physical mechanisms of sediment redistribution. Although understory declines have been attributed to allelopathy, to date there is no documented evidence of allelopathic chemicals produced by western juniper.

Hydrology

Ranchers have long accused juniper forests of reducing or eliminating streamflow on their land. This association is plausible considering that juniper trees take up large volumes of water, much of which is transpired. In southeastern Oregon Angell and Miller, (1994) found western juniper transpired as much as 141 mm of water annually (47% of total evapotranspiration, 300 mm). Bedell et al, (1993) estimated a single juniper (>45 cm in dbh) could transpire between 120 and 160 liters of water per day.

Rainfall water which reaches streams must be in excess of that lost by interception, evapotranspiration and soil column recharge. In this way, water yields generally cannot be increased by woody removal if annual rainfall is below 450-500 mm (Hibbert, 1983). Since western juniper can be found along a large precipitation

gradient (as low as 200 mm to greater than 500 mm annually) the net effect of removal will vary. While studies have generally supported the theory that juniper removal will increase streamflow they have largely been anecdotal (Hibbert, 1983; Brown, 1987). Empirical evidence has been lacking but is currently underway using watershed studies. Preliminary results from a paired watershed study conducted by Oregon State University scientists and the BLM in central Oregon suggests that removal of encroached juniper has increased streamflow and has recharged groundwater levels due to increased infiltration rates with conditions of herbaceous understory versus bare ground between junipers (Buckhouse, *personal communication,* October 24, 2007).

Wildlife

Shrub-steppe communities in Phases I and II of juniper encroachment generally contain a high degree of wildlife diversity and abundance due to greater vertical and understory diversity. Maser and Gashwiler (1978) catalogued 83 bird species and 23 mammal species which use these transitional phases. Although there have been no reported western juniper woodland obligates juniper berries are an important food source for many birds such as the mountain bluebird, cedar waxwing, Steller's jay and scrub jay (Lederer, 1977; Poddar and Lederer, 1982). Pre-settlement woodlands are considered to be old-growth and are a much smaller proportion of the current western juniper extent. These forests are typically lower in juniper density and also contain a high degree of structural diversity favorable to wildlife.

1.3 Resource Islands

Tree Layer Influence on Soil Properties and Herbaceous Vegetation

The influence of trees on soil properties has long received attention. Zinke (1962) was one of the first to specifically address the spatial distribution of plants and soil properties around individual trees in California. He observed a radial symmetry about tree trunks for several soil nutrients and pH and attributed these patterns to bark and leaf litter inputs. He notes that as forests become denser these patterns begin to merge, making individual tree signatures less discernable. Lodhi (1977) noticed large variations between the effect different tree species had on canopy soils in a lowland forest in Missouri. He suggested litter chemistry was an important contributor to these differences and speculated stem flow water contributed as well. While it is likely all woody species modify their soil environment, the impacts of tree species in arid regions is of particular concern because of their effect on ecosystem structure.

In arid zones there is a large discrepancy in soil fertility beneath and adjacent to woody canopies. While herbaceous densities and average height are often greater in the outer region of canopy soils compared to the inner region, the intercanopy zones are often much more sparse or devoid of such growth (Cooper and Stoesz, 1931; Bates et al., 1998). For this reason the spatial correlation of soil resources to shrubby and woody canopies of semi-arid and arid zones is commonly referred to as a 'resource island.' This pattern of soil resource distribution is more pronounced on drylands because tree density is typically lower than on more mesic sites where resource islands may overlap (Adams and Anderson, 1980; Miller et al., 2005).

Resource Island Characteristics

The encroachment of shrubby and woody species into grassland systems contribute to the development of patches of fertility known as "resource islands" or "fertile islands," which characterize shrub-desert ecosystems worldwide (Crawford and Gosz, 1982; Schlesinger and Pilmanis, 1998). Resource islands are described as the accumulation of soil nutrients and soil organic matter (SOM) under tree or shrub canopies compared to the interspace zones between them, which are relatively devoid of these nutrients (Stubbs and Pyke, 2005; Klemmedson and Tiedemann, 2000; Roberts and Jones, 2000; Noy-Meir, 1985). The distribution of soil organisms, including microbes (Gallardo and Schlesinger, 1992; Herman et al., 1995), microarthropods (Santos et al., 1978) and nematodes (Freckman and Mankau, 1986), follows this spatial pattern of resource availability.

In addition to soil nutrients and organisms there are also zonal trends in moisture content between canopy and intercanopy soils. Canopy soils receive less effective rainfall than intercanopy soils, due to canopy interception, and are more subject to plant uptake due to greater rooting density. However, these losses can be offset by the redistribution of water from intercanopy to canopy zones from runoff (Cornet et al., 1992; Seyfried and Wilcox, 1995), higher infiltration rates in duff zones due to better soil structure and lower impact energy of rain drops (Rostagno, 1989; Seyfried and Wilcox, 1995) and lower soil evaporation due to the altered microclimate created by the canopy structure. Nevertheless, in semi-arid environments, surface soil moisture is generally higher in intercanopy zones throughout most parts of the year,

winter months being the exception, potentially due to reduced plant uptake and evapotranspiration (Breshears et al., 1997a, 1997b; Breshears et al., 1998).

Optimal water utilization by plants requires adequate N-levels. Therefore N-availability in arid ecosystems, combined with limited water, can strongly influence plant growth (Ettershank et al., 1978). The enrichment of N in resource islands is thought to be the most significant consequence of resource island development (Lajtha and Schlesinger, 1986) because of its potential to limit net primary productivity (NPP) during periods when water is no longer limiting (Ettershank et al., 1978). Stubbs and Pyke (2005) reported higher levels of NO_3^- and NH_4^+ in canopy soils compared to intercanopy soils of a western juniper-sagebrush ecosystem in central Oregon for nearly all months of the year, even though concentrations of both fluctuated over time. The immobilization of N in litter and microbial biomass may help woody species preserve N within resource islands (Peterjohn and Schlesinger, 1991; Zaady et al., 1996). In addition to N, studies have found elevated concentrations of exchangeable Ca, Mg and K in soils beneath mature (>50 yrs) juniper canopies (Doescher et al., 1986; Klemmedson and Tiedemann, 2000). Generally nutrient concentrations in resource islands tend to be greatest at the soil surface and decrease with depth (Charley and West, 1975; Rostagno et al., 1991; Klemmedson and Tiedemann, 2000).

Resource Island Development and Persistence

Resource island formation is the result of both physical processes, which disperse nutrients, and biological factors which concentrate them. Physical processes include the movement of windblown sediments and plant debris that are caught in

woody canopies and then deposited beneath them, either as dry fall or part of rainfall.

This process can be accelerated by the loss of vegetation in interspace zones, which

increases soil erosion. Whitford et al., (1995) found total N, SO_4^{2-} and Ca

concentrations an order of magnitude higher in the stemflow of *Larrea tridentata*

(creosotebush) compared to bulk precipitation. They suggest captured sediments and

a microbial layer on the bark enrich precipitation as it travels through the shrub to the

ground beneath it. Biological processes can include nutrient uptake and organic

matter deposition beneath the canopy, N-fixation from root symbionts and microbial

decomposition of litter.

Garner and Steinberger (1989) theorized that if biotic processes were wholly

responsible, then only plant critical nutrients would be concentrated under shrub

canopies while non-critical elements would be distributed randomly. The results of an

experiment by Schlesinger et al., (1996), in the Chihuahuan Desert, to distinguish the

relative contributions of biotic and abiotic processes in nutrient distribution

demonstrated that both are important and vary for specific elements. In their study,

non-essential ions Na, Rb, Li and Sr were generally slightly more abundant in the

interspace zones of *Larrea tridentata*, suggesting physical processes localized these

nutrients. However, elevated concentrations of extractable N, P and K under the shrub

canopies demonstrated the importance of biotic factors as well.

There is a lack of data describing the persistence of resource island

characteristics following removal of the woody plant. When a plant dies or is

removed the associated biogeochemical cycle ceases allowing the dispersal of soil

nutrients from resource islands by physical processes (Schlesinger and Pilmanis,

1998). In addition the establishment of herbaceous plants can further homogenize soil resources (Hooke et al., 1991). Studies have demonstrated variable rates of resource island degradation which undoubtedly depends on several factors including species, time since shrub removal and removal technique, age of shrub at removal and climate. Burke et al., (1987) noticed few changes in soil chemistry fourteen years after eliminating *Artemisia tridentata* from a grassland site in Wyoming. Forty five years after the removal of sagebrush in a site in Reno, Nevada, Schlesinger et al., (1996) still found available N confined and autocorrelated over distances of 0 to 200 cm in the *Bromus tectorum* grassland that replaced it. In a study comparing nutrient availability before and thirteen years after mesquite removal, available N losses were substantial while only small losses of available P, K and S were observed (Klemmedson and Tiedemann, 1986). What is common between all these studies is the persistence of plant critical nutrients in former canopy areas long after plant death or removal. Currently, the only work done on western juniper resource island persistence following plant death has been short term (<3 years) and has generally shown elevated available N concentrations throughout the duration of the experiment (Bates et al., 2002; Stubbs and Pyke, 2005). Klemmedson and Tiedemann (2000) found a positive correlation between nutrient accumulations and aboveground biomass of western juniper. This suggests that the persistence of resource islands in western juniper communities is likely to be longer than in shrub communities considering juniper growth can continue past 600 years and can accumulate as much as 22.3 kg biomass tree^{-1} yr^{-1} during periods of its life cycle (Tiedemann and Klemmedson, 2000).

With time, resource islands can become favored sites for woody species regeneration (Schlesinger et al., 1990; Schlesinger and Pilmanis, 1998), perpetuating tree or shrub establishment in arid lands. In addition, the development of resource islands by trees and shrubs is likely to enhance resistance to environmental perturbation. Reynolds et al., (1997) suggest large shrubs with mature resource islands are able to draw from a larger pool of reserves during periods of drought. In the case of western juniper, a closed woodland system is considered to be in a stable state where the tree layer will continue to dominate without a disturbance event such as fire or mechanical removal.

Western Juniper Management

Livestock grazing in eastern Oregon reduces understory herbaceous productivity and diversity, and with the lengthening or elimination of fire return intervals, western juniper has been able to establish in these areas. As western juniper woodlands mature the intercanopy spaces become increasingly devoid of understory species. The loss of ground cover results in erosion, runoff and removal of seedbanks for native understory species (Campbell, 1999). Management of juniper woodlands has been focused on improving livestock and big game forage (McArthur and Young, 1998) and rehabilitating desired herbaceous vegetation and wildlife. Treatment has involved a variety of methods depending on objectives, site conditions and season, including: chaining, prescribed fire, herbicides, rollerchopping, harvest and thinning (Campbell, 1999).

A possible recovery threshold may exist after several years of juniper influence where understory response is not sufficient to support the recovery of ecological functions such as erosion control or nutrient and water capture. This could result from reduced seed bank reserves and viability over time; in this case seeding might be required (Bates et al., 2005).

Management on BLM lands in Central and Eastern Oregon has primarily been in the form of cutting and then burning slash or cutting and leaving the slash. These cutting treatments are directed at woodlands that have crossed an ecological threshold where the land can no longer carry a fire due to insufficient fuel loads and the removal of grazing is insufficient in itself to promote the recovery of previous vegetation (Miller et al., 2000). Recently there has been some success in partial cutting juniper woodlands to create fuel loads adequate to carry fire and then using a fall prescribed burn to remove the remaining juniper thereby minimizing the costs of cutting required for treatment (Bates and Miller, 2004).

Successional Pathways Following Juniper Cutting

Removal treatments of western juniper in eastern Oregon have been successful in promoting herbaceous repopulation. Bates et al., (2000, 2002) found that cutting western juniper woodlands on Steens Mountain, OR increased N availability and more importantly soil water, resulting in rapid increases in herbaceous biomass and cover within the first two years. The assemblage of understory species was similar to that prior to cutting; Bates et al., (2000) found that approximately 70% of the species present one year after cutting were existent prior to treatment.

The dominance of certain plant functional types can vary dramatically throughout succession following juniper removal. Bates et al., (2005) monitored successional changes on juniper cut plots for thirteen years on Steens Mountain, OR to determine long term species compositional changes. During the early seral stage, one to three years after cutting, perennial grasses including Sandberg's bluegrass dominated. Years four through six, cheatgrass invaded and was codominant with perennial grass. In this study and others (Young et al., 1985) it took approximately five years for herbaceous species to colonize open areas following juniper removal. By year twelve cheatgrass cover had dramatically receded while perennial grasses comprised nearly 90% of total herbaceous standing crop.

Cheatgrass response in treated juniper plots has been more rapid in other studies (Vaitkus and Eddleman, 1987; Evans and Young, 1985). Invasion in these cases was suggested to be tied to weather conditions as cheatgrass establishment tended to coincide with higher precipitation. During periods of drought, cheatgrass has been shown to decline due to perennial suppression (Bates et al., 2005).

Western juniper reestablishment can be rapid following cutting (Bates et al., 2005; Zophy, c2006). This indicates that treatments will need to be reapplied in the future to maintain desired understory characteristics. As part of deciding on the type of treatment to apply managers should consider the timing, current vegetation composition and risk of annual exotic invasion (Stubbs and Pyke, 2005; Zophy, c2006).

Following treatment, a key management concern is retaining nutrients on site. Patterns in soil nutrient availability are likely tied to patterns of seedling establishment

and plant biomass production due to plant-soil interactions (Halvorson et al., 1997). Since the investigation of soil nutrient content and spatial distribution on treated juniper woodlands has been limited to the short term there is a need for long-term research on soil nutrient dynamics in western juniper treated areas.

This study was designed to increase our understanding of the effects of juniper resource island degradation on soil nutrient and moisture availability. It explored the long-term expression of resource island and interspace nutrient and moisture distribution after juniper removal by cutting. The length of time in which western juniper resource island expression persists will have implications for restoring ecological diversity and site productivity as there is a demonstrated zonal distribution of understory vegetation in duff and non-duff areas (Bates et al., 1998).

2. Persistence of Western Juniper Resource Islands Following Canopy Removal

2.1 Introduction

The establishment of woody species in grassland systems contribute to the development of patches of soil fertility called 'resource islands' or 'fertile islands,' which characterize shrub-desert ecosystems worldwide (Schlesinger, 1998; Crawford and Gosz, 1982). Resource islands are described as the accumulation of soil nutrients and soil organic matter (SOM) under tree or shrub canopies compared to the interspace zones between them, which are relatively devoid of these resources (Noy-Meir, 1985; Klemmedson and Tiedemann, 2000; Roberts and Jones, 2000; Stubbs and Pyke, 2005). Key agents of resource island formation are higher levels of OM accumulation (Garner and Steinberger, 1989), microbial biomass (Gallardo and Schlesinger, 1992; Herman et al., 1995) and greater nutrient benefits from stem and through flow under woody canopies (Whitford et al., 1997). Some species, such as western juniper, form mutualistic associations with vesicular-arbuscular mycorrhizal (VAM) fungi which increase the efficiency of nutrient capture and sequestration by the host plant (Trappe, 1981). Resource island characteristics have been studied for several woody (Klemmedson and Tiedemann, 1986; Carrillo-Garcia et al., 2000) as well as shrubby species (Burke et al., 1987; Rostagno et al., 1991; Schlesinger et al., 1996; Halvorson et al., 1997; Stock et al., 1999; Bolling and Walker, 2002) around the world because of the ability of these plants to modify local soil chemical and physical properties, create safe sites and indicate desertification. In the Pacific Northwest

(PNW), one species which has had a strong influence on its environment in this way is western juniper (*Juniperus occidentalis*).

Over the past century, western juniper has expanded far beyond its historic range due to fire suppression policy and livestock overgrazing (Miller and Tausch, 2001). Current estimates of western juniper cover are 3.5 million acres in central and eastern Oregon (Miller and Tausch, 2001) and over 9 million acres in the Pacific Northwest and Great Basin (Azuma et al., 2005; Soule et al., 2004). Encroachment into sagebrush-steppe systems has resulted in diminished understory diversity and productivity (Bates et al. 2005) and accelerated soil erosion (Wilcox, 1994; Buckhouse and Mattison, 1980). The loss of neighboring herbaceous vegetation can be attributed to interspecies competition since the accumulation of soil resources by western juniper is at the expense of neighboring soils. The western juniper lateral rooting system can interact with bare soils as well as those of sage and grass (Roberts and Jones, 2000; Kramer et al., 1996) far beyond its canopy extent, roots can grow to four canopy radii (Tiedemann and Klemmedson, 1995). These spatial changes in nutrient distribution and losses of diversity are also indicative of the process of desertification (Schlesinger et al., 1990) where soil resource patchiness is enhanced by western juniper encroachment into sagebrush and grassland systems (Roberts and Jones, 2000).

In eastern Oregon, it is a common practice to cut western juniper woodlands in order to restore understory diversity and productivity (Bates et al., 1998, 2000). The cut trees are regularly left on site to reduce expenditures, thereby increasing total treatment acreage (Bates et al., 2002). While several studies have delineated post-cutting successional pathways (Bates et al., 1998, 2000, 2005, Bates, 2005) and soil

nutrient content of intact stands (Doescher et al., 1987; Tiedemann and Klemmedson, 1995, 2000, Klemmedson and Tiedemann, 2000; Roberts and Jones, 2000), little is known about changes in soil nutrient patterns following treatment. Currently only short-term data (<3 years) is available (Bates et al., 2002, 2007; Stubbs and Pyke, 2005) and is limited to only a few soil constituents (NO_3, NH_4, N-mineralization and nitrification).

Since there is a demonstrated zonal distribution of understory vegetation in juniper litter and bare soil zones (Bates et al., 1998, 2000; Yager and Smeins, 1999), the sustained presence of resource islands after canopy removal may have implications for restoration success. Juniper cutting has been an effective means of increasing understory productivity and diversity (Bates et al., 1998; Yager and Smeins, 1999; Bates et al., 2000; Bates et al., 2005); however, the treated site may not always be repopulated by a desirable understory assemblage (Evans and Young ,1985; Vaitkus and Eddleman, 1987). Bates et al., (1998) found that a qualitative distribution of understory herbaceous species in western juniper canopy and intercanopy zones was predictable following canopy removal based on pre-cut floristic patterns. Typically understory density and cover were greatest in the outer portion of the litter mats and decreased with proximity to the bole (Bates, 1996). Since changes in microclimate, created by the canopy, did not appear to influence post-cutting floristic patterns their results and others (Bates et al., 2005) suggest that soil resource availability and organic matter (OM) depth are important controls of vegetative distribution. Thus temporal dynamics of these two variables, following juniper canopy removal, may help to predict future vegetative assemblages where seedbanks are still present.

The objective of this study was to determine the persistence of western juniper resource islands for several plant critical and non-critical elements through fifteen years post canopy removal. We hypothesized that a resource island effect would be present for some soil variables in all treatments of this study due to similar work documenting sustained western juniper islands through two years (Bates et al., 2002) and mountain big sagebrush islands through nine years (Halvorson et al., 1997; Burke et al., 1987).

2.2 Methods

Site Description and Experimental Design

The study site was located in a 125-ha privately owned grazing allotment in southeast Oregon on a west-facing ridge at 1500 m elevation. Soils were weathered from basalt parent material and were primarily frigid Lithic Argixerolls. For a full site description see Bates (2005).

In June 1991, eight 0.9-ha blocks were established (Bates et al., 1998). Each block was divided into two 0.45-ha plots, oriented north and south of each other. One plot from each block was randomly assigned a cut treatment the other plot received a woodland (no cut) treatment. In the cut treatment plots, all western juniper trees and saplings > 20 cm in height were felled using chainsaws and left on-site in August 1991 (Bates et al., 2002).

A similar experimental design was setup during the summer of 1998, except four 1.8-ha blocks were created (Bates, 2005). One half of each block was randomly selected for cutting; the divisions were also oriented north and south of each other. At

the selected areas all juniper trees were cut during September and October 1998 and then 4-strand barbed-wire fence was built through the center of the block in April 1999 dividing it into east and west sections and creating four 0.45-ha plots (Bates, 2005). The juniper woodlands and 1991 and 1998 cuts (fifteen and eight years prior to sampling) were established on a single ridge that contained on average 80 year old western juniper trees. The control plots were indicative of a closed woodland where juniper trees were the dominant plant layer and understory suppression was evident (Miller et al., 2005). Barbed wire divided each block from the 1991 and 1998 sites into an east and west half, and was used to exclude cattle on the eastern side. All areas where sampling took place were on the western side and were subject to short periods of intense grazing in the spring of 1999 and 2000 when perennial grasses were in vegetative growth stages (Bates, 2005).

In addition to the 1991 and 1998 cuts, there was one plot located approximately 200 m northwest of the old study area where western juniper had been cut one year prior to sampling, in 2005. The one year cut plot was located at the footslope of the hill all other treatments were established on. While the 1991 and 1998 cuts were part of a block design which helped control for environmental variability there were no cuts less than eight years old. We felt it was important to include a younger cut treatment as some elements are likely to respond fairly rapidly to canopy removal.

Treatments were considered to be areas where western juniper was cut one, eight or fifteen years ago or had an intact stand of western juniper trees (control). Plots were randomly selected from those already established by Bates et al., (1998)

and Bates (2005) for the eight, fifteen and no-cut treatments. Each treatment had three

replicates except for the one-year old cut, which had one, and the control (no cut)

which had two. Tree ages on the control plots averaged 95 years.

Sampling was conducted between July 29 and July 31, 2006. Within each of

the control, eight and fifteen-year old cut plots, six western juniper trees or stumps

were randomly selected and nine stumps were selected in the one-year old cut plot. At

each selected western juniper stump or tree three radial transects, set at 120º from each

other, and angled to avoid interference with other stumps, felled trees or shrubs, were

created. Organic matter depth was measured if present, was then brushed away and

soil cores 5 cm in depth, 8 cm in length and 5 cm in width were collected at distances

of 50, 100, 150 and 300 cm from the edge of the bole along the transect. The 300 cm

class was chosen to represent the intercanopy zone as this distance was consistently

outside the canopy radius in the woodland plots.   A soil moisture measurement was

taken at each sample location using time-domain reflectometry (TDR). The moisture

rods (3 in) were placed into the excavated area so that moisture was measured at a

depth of 5 to 8 in. Soil samples collected at each distance class were sieved (2mm) in

the field and then combined into one composite sample (a total of four per tree).

Composite samples were refrigerated at 5°C until analysis.

Soil Analytical Measurements

All soil analytical measurements were conducted at the Central Analytical

Laboratory in the Crop and Soil Science Department at Oregon State University

(Corvallis, OR) except for pH and moisture.

Soil pH was determined using a 1:2 soil to water ratio (McLean, 1982).
Twenty grams of soil were added to 40 mL of distilled water and stirred thoroughly,
then allowed to stand for 15 minutes. The solution was stirred one additional time
then allowed to settle for 15 minutes before reading pH from the supernatant. Soil
moisture was determined as gravimetric water content (GWC) % using TDR.
Moisture content was corrected for soil type using a moisture curve built from a 700 g
composite soil subsample across all distance classes, plots and treatments.

Soil texture was determined for canopy and intercanopy soils. A composite
sample was created for the 50 and 300 cm distance class soils at each plot from 10 g
subsamples from each tree used at that plot. Soil texture was determined using the
hydrometer method (Gee and Bauder, 1986).

In the lab, each sample was measured for total C and N by drying 2.5 g
subsamples for 48 hr at 40°C and analyzing them using infra-red with a Leco CNS-
2000 automated combustion analyzer. Ammonium and NO_3^- concentrations were
determined using a method similar to that of Keeney and Nelson (1982), except it was
modified to extract from 8 g of soil with 30 mL of 2 M KCl. Available-N content was
determined colorimetrically; NH_4^+ content was analyzed using a flow solution auto-
analyzer (Astoria 305D) and NO_3^- content was analyzed with a cadmium reduction
flow solution auto-analyzer (Perstorp Analytical Environmental).

Soluble P, K, Ca, Fe, Al, Na and Mg were extracted from 2.5 g of soil using 25
ml of *Mehlich III* (M3) solution (Mehlich, 1984) and analyzed by inductively coupled
plasma (ICP) (Perkin Elmer Optima 2100 DV). The M3 extraction solution was
developed to include several plant critical nutrients in a single extraction and has been

used along with ICP to effectively and economically estimate soil nutrient availability (Baker et al., 2002). M3 generally extracts similar quantities of P, K, Ca and Mg compared to standard solutions; however, there can be some bias in certain metals. The *Mehlich III* solution extracts much more Fe than DTPA (Mallarino and Sawyer, 1999), leading to overestimates of plant availability. Also the use of ICP for determining nutrient concentrations in this manner can yield slightly different results compared to other standard analytical tools such as atomic-absorption and colorimetric analysis (Mallarino and Sawyer, 1999). However, for the purposes of this study an accurate assessment of plant available nutrient content is not necessary. The objective was to compare the relative amounts, in and around, western juniper resource islands and between treatments to determine trends in availability, which can be accomplished with these analytical tools.

Statistical Analysis

To determine the presence of a resource island effect we compared soil samples collected from the 50 cm distance class (close to bole) to those in the 300 cm distance class (intercanopy), both in and between treatments. If the value at the 50 cm distance class was significantly greater than that at 300 cm, a resource island effect was determined to be present. To test for significance, we used a mixed model with random and fixed effects with the statistical procedure MIXED from SAS version 9.1.3 (SAS, Cary, NC, USA).

The one-year old cut data was not included in the ANOVA due to confounding effects of biotic and environmental variability (juniper age, topographic position,

soils); data from this treatment is presented in Table 3 and Figures 1 and 2. The main

effect for each soil variable was distance and treatment. Data was tested for normality

using the UNIVARIATE option; data that was not normally distributed was log

transformed. Backtransformations represent the estimated multiplicative effect of

distance on the soil variable.

2.3 Results

Main Effects

The main effect of distance was significant for all soil variables except NH_4^+,

Fe, Na and Mg, a significant treatment effect was only observed for NO_3^- (Table 1).

There was a significant interaction between distance and treatment for K and Na.

Table 1. Summary statistics (ANOVA) including degrees of freedom (df), F-statistic
(F) and p-value (*P*) for all soil variables for the effect of treatment, distance and their
interaction

Total N	df	F	*P*		P	df	F	*P*		Al	df	F	*P*
Treatment (T)	2,5	1.17	0.3825		Treatment (T)	2,5	1.89	0.25		Treatment (T)	2,5	0.11	0.9
Distance (D)	1,69	89.44	<.0001		Distance (D)	1,70	8.43	0.005		Distance (D)	1,70	57.1	<.0001
T-by-D	2,69	0.55	0.579		T-by-D	2,70	0.03	0.97		T-by-D	2,70	0.25	0.78
Total C					**K**					**Na**			
Treatment (T)	2,5	2.15	0.2117		Treatment (T)	2,5	1.62	0.29		Treatment (T)	2,5	1.78	0.26
Distance (D)	1,69	107.4	<.0001		Distance (D)	1,70	50.05	<.0001		Distance (D)	1,9	1.82	0.21
T-by-D	2,69	1.39	0.2564		T-by-D	2,70	4.88	0.01		T-by-D	2,9	6.88	0.01
C/N					**Ca**					**Mg**			
Treatment (T)	2,5	0.86	0.48		Treatment (T)	2,5	0.28	0.77		Treatment (T)	2,5	0.1	0.91
Distance (D)	1,69	96.66	<.0001		Distance (D)	1,70	19.56	<.0001		Distance (D)	1,46	2.25	0.14
T-by-D	2,69	1.2	0.31		T-by-D	2,70	0.44	0.65		T-by-D	2,46	1.22	0.31
Nitrate-N					**Fe**					**pH**			
Treatment (T)	2,5	6.36	0.042		Treatment (T)	2,5	0.2	0.82		Treatment (T)	2,5	1.18	0.38
Distance (D)	1,70	48.85	<.0001		Distance (D)	1,70	3.42	0.069		Distance (D)	1,70	17.6	<.0001
T-by-D	2,70	0.41	0.66		T-by-D	2,70	0.51	0.61		T-by-D	2,70	1.54	0.22
Ammonium-N					**Gravimetric Moisture Content (%)**								
Treatment (T)	2,5	1.93	0.24		Treatment (T)	2,5	2.17	0.21					
Distance (D)	1,68	3.38	0.07		Distance (D)	1,70	20.95	<.0001					
T-by-D	2,68	0.35	0.7		T-by-D	2,70	0.7	0.5					

Resource Island Properties

Overall, the signature from relic western juniper canopies was still present fifteen years after cutting on our plots. Canopy soils were generally significantly higher in C_{tot}, N_{tot}, NO_3^-, P, K and Ca (p<0.02) compared to intercanopy soils in the control and post-cut treatments (Table 2). The exception was P in the control woodlands; mean P content did, however, increase towards the bole. Accumulations of S_{tot} were evident in all treatments (data not shown), but results could not be analyzed statistically due to an incomplete data set. Mean Mg concentrations decreased with distance from the tree bole in the woodland and one-year old cut treatments, but were not significant. Sodium content was significantly higher in canopy soils in the control and one-year cut treatments (p<0.01); this trend was not evident in older cuts (eight and fifteen years) (Table 2). In the juniper woodlands resource islands were most pronounced for C_{tot}, NO_3^-, K and Na which were generally at least twice as abundant under canopy compared to intercanopy soils (Figure 2). Total C was significantly correlated to N_{tot} for all treatments ($r^2 \geq 0.89$). The soil C/N ratio generally decreased with distance from the bole and was significantly lower in intercanopy soils for all treatments (p<0.0005).

While there were no significant trends for NH_4^+, mean concentrations were generally highest close to the bole in the juniper woodland and the eight and fifteen-year old cuts. While the available N fraction ($NO_3^- + NH_4^+$) was only a small fraction of the total N pool, available-N concentrations were two times higher under western juniper canopies compared to intercanopy soil.

Table 2. Estimated resource island effect (additive, multiplicative), t-statistic and *P*-value for each soil variable by treatment

Soil Variable	Control (woodland)			8 year cut			15 year cut		
	Estimate	t	*P*	Estimate	t	*P*	Estimate	t	*P*
Total C %	+1.98	4.21	<.0001	+2.5	6.44	<.0001	+2.98	7.78	<.0001
Total N%	+0.10	4.19	<.0001	+0.12	5.79	<.0001	+0.13	6.76	<.0001
C/N ratio	+2.39	3.95	0.0002	3.11	6.24	<.0001	3.58	7.29	<.0001
NO_3 mg kg^{-1}	x2.69	4.25	<.0001	x2.08	3.85	0.0003	x2.14	4.02	0.0001
NH_4 mg kg^{-1}	+0.27	0.28	0.78	+1.17	1.52	0.13	+1.25	1.63	0.1
P mg kg^{-1}	x1.24	1.6	0.11	x1.33	2.61	0.011	x1.32	2.57	0.012
K mg kg^{-1}	x2.05	5.95	<.0001	x1.41	3.47	0.0009	x1.28	2.48	0.02
Na mg k^{g-1}	+9.07	4.44	0.0016	-2.88	1	0.34	-0.02	0.01	0.99
Ca mg kg^{-1}	x1.25	2.29	0.03	x1.18	2.07	0.04	x1.31	3.4	0.001
Fe mg kg^{-1}	x0.79	1.66	0.1	x0.89	1	0.33	x1.05	0.44	0.66
Al mg kg^{-1}	x0.79	4.35	<.0001	x0.81	4.59	<.0001	x0.83	4.22	<.0001
Mg mg kg^{-1}	x1.21	1.54	0.13	x1.09	1.09	0.28	x0.97	0.34	0.74
Moisture %	-4.45	2.57	0.01	-4.97	3.52	0.0008	-2.69	1.9	0.06
pH	+0.33	3.38	0.001	+0.19	2.34	0.02	+0.11	1.37	0.18

There was a strong distance effect for Al and moisture content (Table 2) where values increased with distance from the tree bole (Figure 1). The trend of Fe and Al was mirrored by increases in soil acidity; Fe content was slightly correlated with pH ($R^2 = 0.51$). Soil pH was significantly higher in canopy soils for all treatments except the fifteen-year old cut (Table 2).

Soil moisture increased with distance from the bole and was significantly higher in intercanopy soils compared to canopy soils in the control, eight and fifteen year old cuts (Figure 1, Table 2). This trend was less evident in the one year old cut (Figure 1). Organic matter depth was greatest closest to the bole and decreased with distance from the bole and time since canopy removal (Table 3). There was very little variation in soil texture between distance classes and plots. Soils were predominantly clay loam/loam irrespective of shrub-intershrub position (Figure 3). There was no evidence of elevated clay or silt content in the canopy soils.
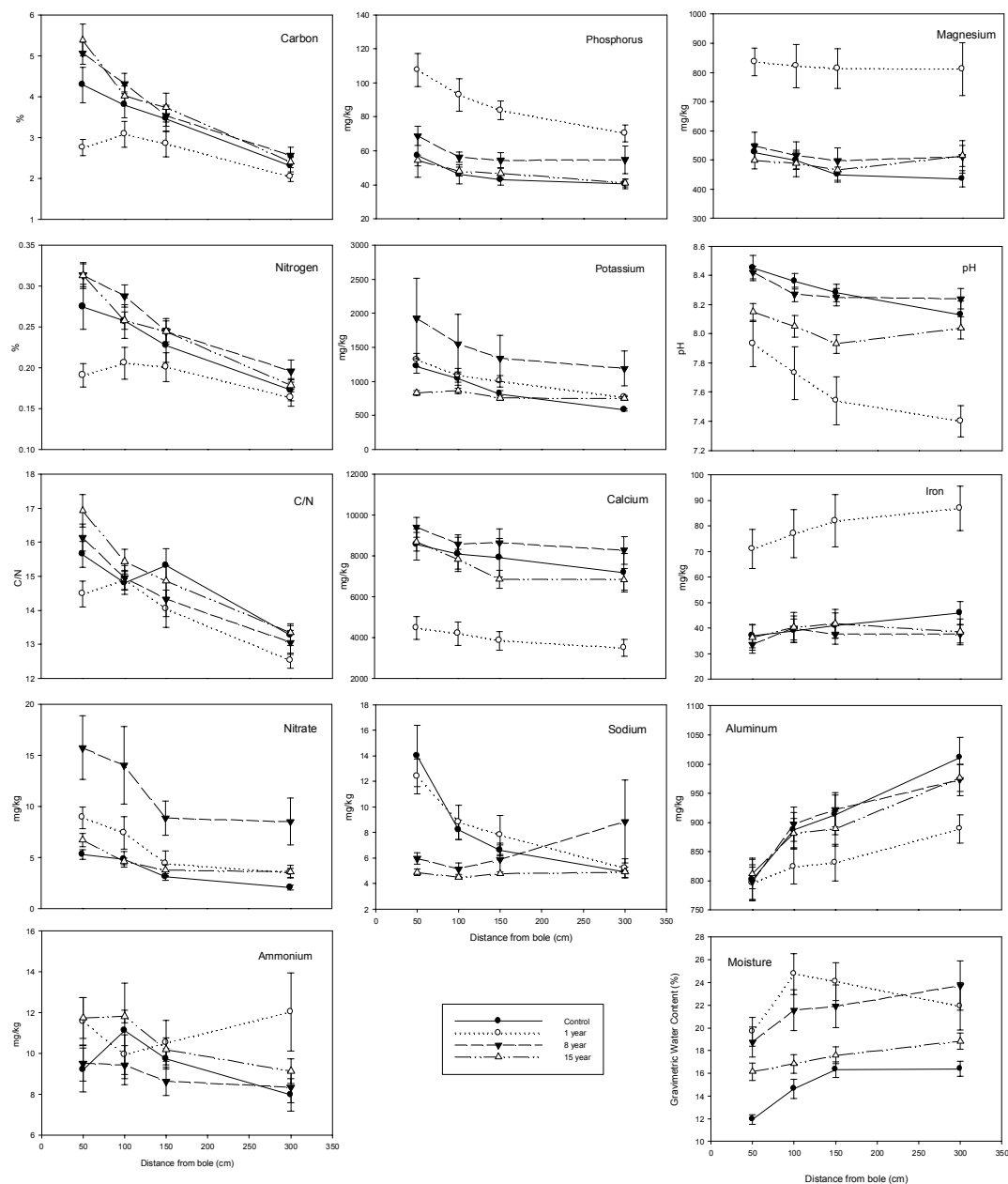
Figure 1. Mean and standard error values for each soil variable by distance from the bole for each treatment.
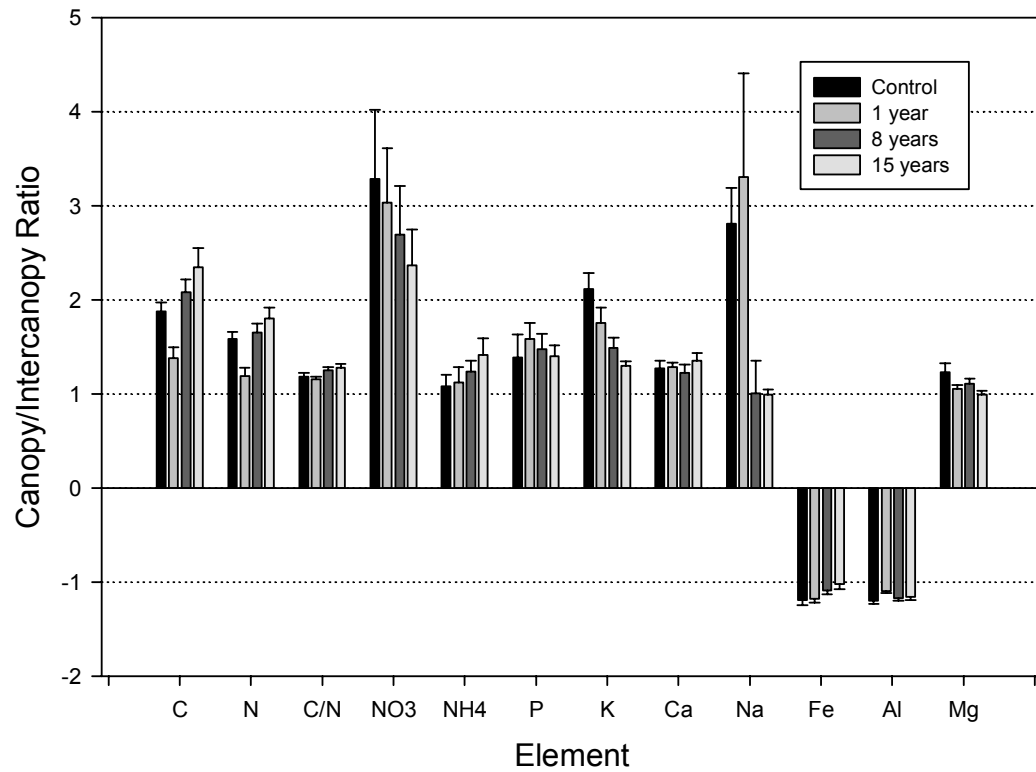
Figure 2. Ratio of the concentration of soil elements 50cm:300cm with error bars for all treatments

Soil Resource Variability Among Distance Classes and Treatments

In general, intercanopy soils had less variation in nutrient content than canopy soils. This was particularly true for NO_3^- and NH_4^+ which had standard deviations in canopy soil up to sixteen and ten times greater than intercanopy soil respectively. This variation was typically observed in the juniper woodland plots and less evident in the cut plots. This trend was not observed with concentrations of Ca, Fe or Al, which had very little to no difference in standard deviations between distance classes.

The trend in nutrient availability around juniper stumps in the one-year old cut was generally similar to that in other treatments; however, nutrient concentrations

Table 3. Mean litter layer depth and standard errors by distance class for each treatment

	Distance from bole (cm)			
Treatment	50	100	150	300
control	6.5 ± 1.0	3.6 ± 0.7	1.6 ± 0.5	0
1 year cut	4.2 ± 1.1	1.9 ± 0.5	0.6 ± 0.5	0
8 year cut	2.9 ± 0.6	1.4 ± 0.4	0.6 ± 0.3	0.1 ± 0.1
15 year cut	1.5 ± 0.4	0.6 ± 0.2	0.4 ± 0.1	0.1 ± 0.1

often were not. Soluble P, Ca, Mg and Fe concentrations were unique in the one-year
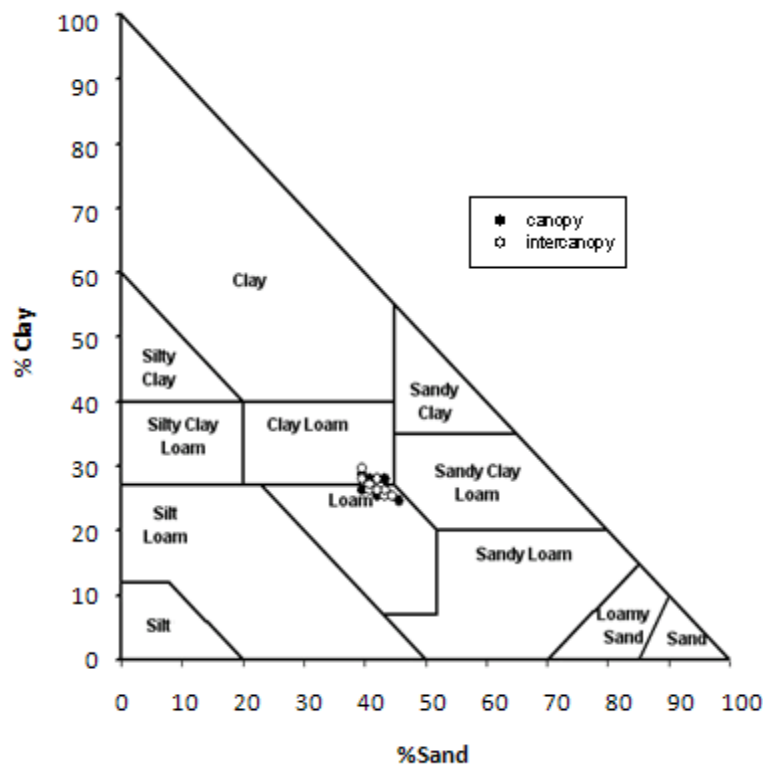
old cut compared to other treatments (Figure 1).

Figure 3. Range of soil textures observed at our study site including canopy and intercanopy zones

2.4 Discussion

Persistence of Western Juniper Resource Islands

When a plant dies or is removed the associated biogeochemical cycle ceases potentially allowing the dispersal of soil nutrients from resource islands by physical processes (Schlesinger and Pilmanis, 1998). In addition, the establishment of grasses and herbaceous plants can further homogenize soil resources (Hooke et al., 1991). Although total standing crop in the fifteen-year cut treatment had increased to ten times that of the woodlands (Bates et al., 2005) western juniper resource island effects were still evident for most soil variables measured.

We found elevated levels of C_{tot}, N_{tot}, NO_3^-, P, K and Ca in canopy soils of western juniper cut fifteen years prior. However, there was evidence that these resource islands were degrading, and at different rates. When resource islands were expressed as a unitless ratio, canopy (50 cm):intercanopy (300 cm) soil, NO_3^-, K and Mg accumulations appear to steadily decline with time since canopy removal (Figure 2). Sodium accumulations demonstrated a more rapid degradation likely due to leaching.

The resource island effect for NO_3^- in all cut treatments was surprising when compared to previous work. Although no long-term data exists for western juniper NO_3^- resource island persistence, studies of other semi-arid woody species (mesquite, sagebrush) suggest NO_3^- accumulations are more ephemeral following plant death (Klemmedson and Tiedemann, 1986; Halvorson, 1997). We attribute the sustained bioavailability of plant critical nutrients under relict western juniper canopies to the development of deep litter mats beneath juniper canopies, which are not found under

sagebrush or mesquite. In the control plots some juniper litter mats were up to 12 cm in depth and while litter depth decreased over time due to decomposition, distinct mats were still present in the fifteen-year old cut treatment. Therefore the retention of litter after canopy removal could maintain N-immobilizing microbial populations as long as they are present (Stubbs and Pyke, 2005). Although we did not survey litter chemistry, Klemmedson and Tiedemann (2000) found similar patterns in organic layer nutrients and accumulations in the soil.

Oddly, the strong resource island effect for NO_3^- in the woodland treatment of our study contrasted previous work done at this site. Bates et al., (2002) found NO_3^- levels of intercanopy soils consistently higher or equal to canopy soils in uncut western juniper plots during summer and fall months, though the cut plots did demonstrate a clear resource island trend during the same time period. They attributed their findings to higher net N-mineralization and nitrification rates in intercanopy soils. The discrepancy in our results is likely due to differences in sampling techniques. In their study, sampling in canopy zones took place at approximately ⅔ canopy radii and intercanopy sampling was done midway between tree canopy zones. We designated canopy soils as those taken at 50 cm from the tree bole which consistently had the highest levels of available-N; availability decreased dramatically by the mid canopy area (150 cm). This demonstrates the large spatial variability in nutrient availability just within canopy zones. For several soil resources including NO_3^-, P and K there was a fairly steep decline in availability moving from the bole, which appeared to level off close to ½ canopy radii (Figure 1). Klemmedson and

Tiedemann (2000) observed a similarly steep gradient in nutrients beneath juniper canopies.

Some studies have found higher soil moisture content beneath woody canopies in arid environments (Bates et al., 1998). We generally found soil moisture content increased with distance from the bole both pre- and post-cutting of western juniper (Figure 1). In the juniper woodland plots, surface moisture is lower in canopy soils primarily due to canopy interception and greater plant uptake than in intercanopy soils (Breshears et al., 1997); root biomass is typically greater closer to the bole. In the juniper cut plots this trend persisted probably because of soil hydrophobicity under juniper litter (Pierson et al., 2007); juniper litter mats can insulate the mineral soil from moisture. Also higher herbaceous establishment in juniper litter compared to interspace zones could lead to greater water uptake in these areas.

Our results originate from a single time point, and while they demonstrate clear trends in resource availability about juniper resource islands it is important to note these trends are dynamic seasonally and yearly (Bates et al., 1998; Bates et al., 2002). Our summer sampling period (July 29-31) occurred when resource island expression is likely to be most pronounced. When water is limiting plant uptake of nutrients is minimal thus availability is increased; in contrast during wet seasons plant critical nutrients such as N become limiting and resource island expression is dampened.

The presence of juniper litter in relic canopy zones was not surprising considering western juniper foliage has one of the lowest decomposition rates for trees and shrubs in semi-arid and temperate areas (Bates et al., 2007). The decrease in C/N ratio away from the tree bole was likely the result of these poor quality litter inputs.

Bates (1996) reports C/N values of 55:1 for juniper needles and 240:1 for twigs and branches; the decomposition rate of juniper foliage is six times slower than that of *Artemeisa tridentata* during the first two years (Murray, 1975; Bates et al., 2007). We consistently found the highest litter depth at the base of juniper trees and stumps which decreased with distance from the tree bole (Table 3). These large inputs of C-rich OM were great enough to offset elevated N-levels and increase the C/N ratio of the soil in these areas.

Our results are not unlike others for shorter-lived, semi-arid woody species which have left resource islands for certain soil nutrients several years after elimination. Halvorson et al., (1997) found only small losses of total organic carbon in canopy soils of *Artemesia tridentata* nine years after removal by fire; although N losses were substantial. In a study comparing canopy soils of mesquite removed thirteen years prior, Klemmedson and Tiedemann (1986) found small losses of available P, K and S while available-N losses were significant. However, it is likely that resource island persistence is much longer for juniper than other semi-arid species considering the relative longevity and biomass of western juniper trees. Klemmedson and Tiedemann (2000) found that organic layer mass and soil nutrient content increase with juniper size (age). In their study it appears this trend begins to level off at an average age of 100 years. Since juniper trees on our plots were similar in age it is possible that we captured western juniper resource island expression at its greatest on Steen Mountain. Other studies have also found that resource island expression for certain elements is less pronounced in juvenile trees (Tiedemann and Klemmedson, 1995; Doescher et al., 1987). Thus it will likely take much longer for dispersal

processes to degrade nutrient accumulations on out plots than it would for other shrubby species or junipers in different developmental stages.

Although we only observed the lateral nutrient dynamics of the top 5 cm of soil, redistribution may be occurring at lower depths. Burke et al., (1987) suggested that grasses translocated P and K from deeper soils to the surface following sagebrush removal and grassland takeover. A similar process may be occurring with the re-establishment of understory vegetation in our plots that could be influencing surface distribution of soil nutrients.

The Effect of Western Juniper Resource Islands on Soil Properties

There were noticeable accretions of Ca, Mg and K, although not all significant, in the juniper woodlands. The mining of these bases from lower soil depths and interspace zones and then deposition as litter likely contributed to increased alkalinity closer to the bole. This process may help the competitiveness of juniper trees by creating an unfavorable soil environment for herbaceous species. We did not analyze soil pH at the base of juniper boles (0 cm); however, pH may be lower here than at the 50 cm distance class because of the acid nature of bark litter (Zinke, 1962). Halvorson et al., (1997) found lower soil pH values at 0 cm from sagebrush boles compared to 20 cm; soil pH then became lower with increasing distance from the shrub.

Some studies suggest woody canopies in semi-arid regions are effective in capturing fine particles leading to finer soil textures in canopy soils (Coppinger et al., 1991; Parsons et al., 1992). This process could contribute to higher nutrient availability beneath shrubby and woody canopies compared to between them. We

found no difference between canopy and intercanopy soil textures; in fact there were very few differences in soil texture across zones, plots and treatments. Burke (1989) found similar results for three different shrubs species in Wyoming; she suggests a 5 cm sample depth may be too deep to detect any collection of soil fines at the surface.

Limitations of Interpreting Data from the One-Year Old Cut Treatment

The one-year old cut treatment was located outside of the area of the other treatments (control, eight and fifteen year old cuts); thus there were several factors confounding interpretation of this data. Trees in this treatment were likely part of a younger cohort as they were downhill from the other treatments and the needles had not yet dropped like they had on the other plots. Bates et al., (2007) estimate that it takes two years for 75% of the suspended litter to fall. The slow addition of fresh litter can influence decomposition rates and soil nutrient content. Soils varied between the different areas as well (Figure 4); the old study area (control, eight and fifteen year old cuts) had soils characterized as frigid Lithic Argixerolls with an argillic zone from 3 to 14 in, while the one-year old cut was underlain by frigid Cumulic Endoaquolls which are much deeper (60 in to bedrock vs. 24 in) and saturated during spring and early summer. These environmental differences probably contributed to the unique soil resource responses we observed in the one-year cut treatment compared to the other treatments (Figure 1).
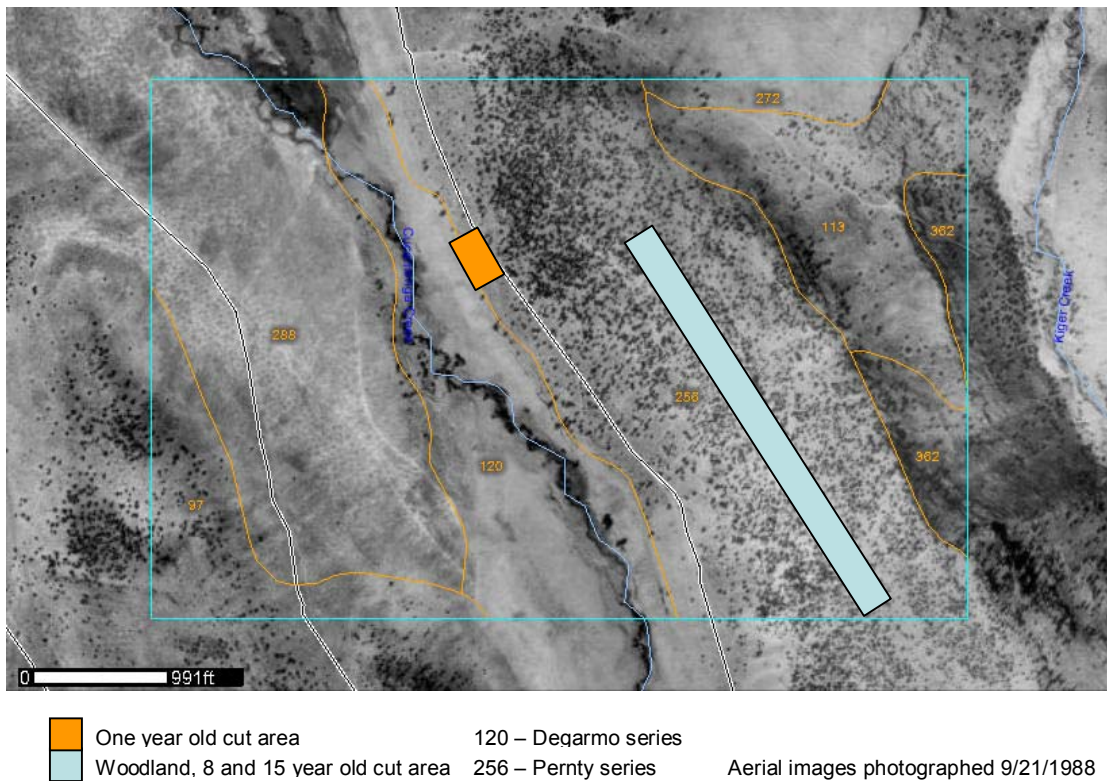
One year old cut area   120 – Degarmo series
Woodland, 8 and 15 year old cut area   256 – Pernty series   Aerial images photographed 9/21/1988

Figure 4. Locations of different study areas in the study site and related soil series (USDA-NRCS, 2007).

Management Implications

In semi-arid habitats western juniper has a dominant role in the capture and

retention of nutrients, both in biomass and in the organic and mineral soil layer

(Tiedemann and Klemmedson, 2000). Thus it would seem prudent to preserve these

accumulations to facilitate understory recovery following juniper control treatments.

Indeed plant density and cover are greatest in litter zones (Bates, 1996; Bates et al.,

1998); however, a critical concern is the response of desired and, perhaps more

importantly, undesired species following treatment. While a variety of juniper control

methods have been successful in restoring understory productivity and cover (Rose &

Eddleman, 1994; Yager & Smeins, 1999; Bates et al., 2005), there is a strong zonal

distribution of perennial and annual species between and within duff zones. Most

notably cheatgrass has been found to prefer the outer portion of juniper duff areas both before and after removal treatments, while germination of any species seems to be impeded by litter depth in regions closer to the stump (Bates et al., 1998). We also noticed rings of cheatgrass under relic juniper canopies in several of the post-cutting plots. The cessation of competition by the tree and resulting nutrient flush favors these ruderal species. Although results here do not have direct management implications as far as restoring desired species assemblages, the inherent connection between plants and the soil environment would indicate that resource island persistence can strongly influence future management success.

The sustained resource island effect for several plant critical nutrients fifteen years after removing western juniper canopies suggests that resource islands will have long-term implications for successional pathways. While the relative abundance of plant species can fluctuate greatly throughout post-removal succession (Bates et al., 2005) their zonal distribution generally does not (Bates, 1998). The management of juniper litter may be a useful tool in helping to prevent or curb establishment of undesirable species on sites where the risk of invasion is high.

The prevention of resource island development would be ideal considering their constancy in closed woodlands treated by cutting. Younger juniper stands should be given high importance for control treatments because resource island effects are less pronounced or nonexistent and the probability of successful restoration without added inputs such as seeding or follow up treatment is high.

Future work should focus on feasible management techniques of western juniper litter as part of or following control treatments. Bates and Svejcar (*In Review*)

found that burning juniper debris after cutting led to a quicker recovery of perennial bunchgrasses and reduced cheatgrass presence. These litter zones can occupy a substantial portion of the landscape; on our study site juniper canopy cover was more than 25% (Bates et al., 2005). As long as these litter zones remain, they will continue to provide favorable sites for annual establishment.

3. General Conclusions

Over the last century the encroachment of native shrubby and woody species has changed the structure and altered the ecological processes of semi-arid shrublands and grasslands. Western juniper expansion throughout the Pacific Northwest has attracted considerable attention due to the economic and ecological impacts on encroached lands. While western juniper was historically confined to higher elevation rocky outcrops, protected from fire events, a large part of juniper afforestation has occurred on deeper more productive soils downslope of these areas typically occupied by sagebrush (Soule et al., 2003). Over the last 7000 years western juniper cover has expanded and contracted; however, the recent rate of expansion is uncharacteristic of historical patterns. In the late 1800s, human settlement of the West has led to excessive livestock grazing and reduced fire frequency. These practices are considered to be the primary drivers of juniper and other woody species expansion during this time period (Humphrey, 1958; Archer et al., 1995; Bahre, 1995.

Advocates against western juniper control point to the aesthetic and rustic appearance of these forests as well as the potential carbon sink. In drier climates soil organic carbon (SOC) and plant biomass C increases with woody plant invasion (Jackson et al, 2002). Belsky (1996) notes that western juniper is a native species that has simply shifted in dominance under certain environmental conditions and that much of the criticism it has received is 'common knowledge' and largely unfounded scientifically. During certain stages of juniper woodland development (phase II, old growth) stand structure and wildlife diversity are indeed enhanced. Whether or not the concern over western juniper expansion is justified, there are multi-million dollar

programs aimed at removing juniper from encroached lands, both private and public. Ranchers have sought to restore forage for livestock and improve stream flow while public land managers, such as the BLM, have tried to improve ecological diversity and understory productivity.

The Eastern Oregon Agricultural Research Center (EOARC), Oregon State University, located in Burns, OR has been instrumental in directing research efforts towards successful and cost-effective methods of western juniper control. The western juniper woodland program was established by the EOARC with several long-term objectives related to western juniper ecology, biology and management. Currently study sites are located in Harney and Grant County in eastern Oregon and Modoc County in northeastern California; however sites continue to be added in order to cover all western juniper habitat types. Repeated observations by the EOARC and the Department of Rangeland Resources at Oregon State University have shown that without management western juniper will come to dominate encroached sites leading to reduced water yield and quality, as well as diminished productivity and biodiversity of the land (Bedell et al., 1999). Although our understanding of the ecological impacts posed by western juniper and possible treatment plans has increased significantly there are still several areas for future work. A comprehensive review of western juniper literature identified several knowledge gaps that still exist including the "long-term impacts of woodland succession and treatment on soil nutrients and development" (Miller et al., 2005). Our study was designed to contribute to this area and is complimentary to other work done at the same study site, Steens Mountain, OR.

The objectives of our study were first to identify trends for certain soil chemical and physical properties of western juniper resource islands; secondly we wanted to determine how long resource islands remained following western juniper cutting treatments. We found that western juniper resource islands were slow to degrade suggesting that this patchy distribution of soil resources will continue to influence the distribution of understory species long-term without treatment of litter zones. Currently little work exists documenting vegetative responses to resource island manipulations. However, there is evidence that burning juniper litter mats helps to reduce cheatgrass establishment (Bates and Svejcar, *In Review*).

Rangelands encompass vast tracks of land, thus treatment programs are often developed so that they affect the maximum acreage possible. Juniper cutting has been an economical method of restoring understory productivity and diversity on mature, closed woodlands. On these stands, increases in western juniper density have resulted in insufficient understory production to carry a fire. Although this method has been commonly used in the past, treatment strategies continue to evolve. Currently there are experimental treatments combining mechanical methods and fire where the stand is thinned in order to increase fuel loads for a future burn. In addition there is work being done documenting the effects of burning individual juniper canopies by drip torch. Ultimately the success of any control method will be determined by economic feasibility and efficiency in achieving the goals for the land. Where understory recovery is concerned, western juniper creates an interesting problem because over time it creates patches of soil resources beneath litter fall which are preferred zones by undesirable annuals. The challenge then is to retain as much of this nutrient capital as

possible for vegetative restoration while keeping invasive establishment to a minimum. The solution will likely involve a compromise where there is some type of alteration of the distribution or amount of juniper litter; currently there are still several possibilities for future research in this area.

LITERATURE CITED

Adams DE and Anderson RC. 1980. Species response to a moisture gradient in central Illinois forests. American Journal of Botany 67(3): 381-392

Adams CR and Eswaran H. 2000. Global land resources in the context of food and environmental security. In *Advances in Land Resources Management for the 20th century*, Gwande SP (ed.). Soil Conservation Society of India: New Delhi; 35-50

Archer S, Schimel DS, Holland EA. 1995. Mechanisms of shrubland expansion: land use, climate or CO_2? Climatic Change 29: 91-99

Azuma DL, Hiserote BA and Dunham PA. 2005. The western juniper resource of eastern Oregon, 1999. Resour. Bull. PNW-RB-249. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 18 p

Bahre CJ. 1995. Human impacts on the grasslands of southeastern Arizona. *In* McClaran MP, Van Devender TR, eds. 1995. The Desert Grassland. Tucson: University of Arizona Press pp. 230-264

Baker WH, Herron CG, Carroll SD, Henslee MA, Lafex DC and Evans EE. 2002. A comparative summary of the standard Mehlich 3 soil test with a modified Mehlich 3 dilution ratio procedure. Arkansas Agricultural Experiment Station, University of Arkansas. Special report 205

Bates JD. 1996. Understory vegetation response and nitrogen cycling following cutting of western juniper. Ph.D. dissertations, Oregon State University, Corvallis Oregon. 230 pp.

Bates JD. 2005. Herbaceous response to cattle grazing following juniper cutting in Oregon. Rangeland Ecology and Management 58: 225-233

Bates JD and Miller RF. 2004. Restoration of aspen woodland invaded by western juniper: applications of partial cutting and prescribed fire. In: Proceedings of the 16th Int'l Conferences, Society for Ecological Restoration, August 24-26, 2004, Victoria, Canada. 2004 CDROM.

Bates JD, Miller RF and Svejcar TJ. 1998. Understory patterns in cut western juniper (*Juniperus occidentalis* spp. *occidentalis* hook.) woodlands. Great Basin Naturalist 58 (4): 363-374

Bates JD, Miller RF and Svejcar TJ. 2000. Understory vegetation response following cutting of western juniper. Journal of Range Management 53: 119-126

Bates JD, Miller RF and Svejcar TJ. 2005. Long-term successional trends following western juniper cutting. Rangeland Ecology and Management 58: 533-541

Bates JD and Svejcar TJ. *In Review.* Herbaceous response to burning of western juniper debris.

Bates JD, Svejcar TJ and Miller RF. 2002. Effects of juniper cutting on nitrogen mineralization. Journal of Arid Environments 51: 221-234

Bates JD, Svejcar TJ and Miller RF. 2007. Litter decomposition in cut and uncut western juniper woodlands. Journal of Arid Environments 70: 222-236

Bedell TE, Eddleman LE and Deboodt T, Jacks C. 1993. Western juniper-its impact and management in Oregon rangelands. Oregon State Univ. Ext. Serv.

Belnap J. 2004. Surface disturbances: Their role in accelerating desertification. Environmental Monitoring and Assessment 37: 39-57

Bolling JD and Walker LR. 2002. Fertile island development around perennial shrubs across a Mojave Desert chronosequence. Western North American Naturalist 62(1): 88-100

Breshears DD, Rich PM, Barnes FJ and Campbell K. 1997a. Overstory-imposed heterogeneity in solar radiation and soil moisture in a semiarid woodland. Ecological Applications 7(4): 1201-1215

Breshears DD, Myers OB, Johnson SR, Meyer CW and Martens SN. 1997b. Differential use of spatially heterogeneous soil moisture by two semiarid woody species: Pinus Edulis and Juniperus Monosperma. The Journal of Ecology 85(3): 289-299

Breshears DD, Nyhan JW, Heil CE and Wilcox BP. 1998. Effects of woody plants on microclimate in a semiarid woodland: soil temperature and evaporation in canopy and intercanopy patches. International Journal of Plant Science 159(6): 1010-1017

Brown JR and Archer S. 1999. Shrub invasion of grassland: recruitment is continuous and not regulated by herbaceous biomass or density. Ecology 80(7): 2385-2396

Buckhouse JC and Mattison JL. 1980. Potential soil erosion of selected habitat types in the high desert region of central Oregon. Journal of Range Management 33(4): 282-285

Burke IC. 1989. Control of nitrogen mineralization in a sagebrush steppe landscape. Ecological Society of America 70(4): 1115-1126

Burke IC, Reiners WA and Sturges DL, Matson PA. 1987. Herbicide treatment effects on properties of mountain big sagebrush soils after fourteen years. Soil Science Society of America 51: 1337-1343

Campbell RB. 1999. Ecology and management of pinyon-juniper communities within the Interior West: Overview of the "ecological restoration" session of the symposium. USDA Forest Service Proceedings RMRS-P-9

Carrillo-Garcia A, Bashan Y and Bethlenfalvay GJ. 2000. Resource-island soils and the survival of the giant cactus, cardon, of Baja California Sur. Plant and Soil 218: 207-214

Charley JL and West NE. 1975. Plant-induced soil chemical patterns in some shrub-dominated semi-desert ecosystems of Utah. Journal of Ecology 63: 945-964

Cooper WS and Stoesz AD. 1931. The subterranean organs of *Helianthus scaberrimus*. Bulletin of the Torrey Botanical Club 58: 67-72

Coppinger KD, Reiners DL, Burke IC and Olson RK. 1991. Net erosion on a sagebrush steppe landscape as determined by cesium-137 distribution. Soil Science Society of America Journal 55: 254-258

Cornet AF, Montana C, Delhoume JP and Lopez-Portillo J. 1992. Water flow and the dynamics of desert vegetation stripes. pp. 327-345 *In* Hansen AJ and diCastri F, editors. Landscape boundaries: consequences for biotic diversity and ecological flows. Springer-Verlag, New York, New York, USA

Crawford CS and Gosz JR. 1982. Desert ecosystems: their resources in space and time. Environmental Conservation 9:181-195

Doescher PS, Elleman LE and Vaitkus MR. 1987. Evaluation of soil nutrients, pH, and organic matter in rangelands dominated by western juniper. Northwest Science 61(2): 97-102

Dregne HE. 1977. Desertification of arid lands. Economics Geography 53(4): 322-331

Ettershank G, Ettershank J, Bryan M and Whitford WG. 1978. Effects of nitrogen fertilization on primary production in a Chihuahuan Desert ecosystem. Journal of Arid Environments 1: 135-139

Evans RA and Young JA. 1985. Plant succession following control of western juniper (*Juniperus occidentalis*) with Picloram. Weed Science 33: 63-68

FAO. 1996. Our land, our future. UN Food and Agriculture Organization: Rome

Freckman DW and Mankau R. 1986. Abundance, distribution, biomass and energetics of soil nematodes in a northern Mojave desert ecosystem. Pedobiologia 29: 129-142

Fryrear DW. 1995. Soil losses by wind erosion. Soil Science Society of America Journal 59: 668-672

Gallardo A and Schlesinger WH. 1992. Carbon and nitrogen limitations of soil microbial biomass in desert ecosystems. Biogeochemistry 18: 1-17

Garner W and Steinberger Y. 1989. A proposed mechanism for the formation of 'fertile islands' in the desert ecosystem. Journal of Arid Environments 16: 257-262

Gedney DR, Azuma DL, Bolsinger CL and McKay N. 1999. Western juniper in eastern Oregon. USDA For. Ser. PNW-GTR-464

Gee GW and Bauder JW. 1986. Particle-size analysis. p. 383–411. *In* Klute A (ed.) Methods of soil analysis. Part 1. 2nd ed. Agronomy Monographs 9. ASA and SSSA, Madison, WI

Gisladottir G and Stocking M. 2005. Land degradation control and its global environmental benefits. Land Degradation and Development 16: 99-112

Graumlich LF. 1987. Precipitation variation in the Pacific Northwest (1675-1975) as reconstructed from tree rings. Annals of the Association of American Geographers 77(1): 19-29

Grover HD and Musick HB. 1990. Shrubland encroachment in southern New Mexico, U.S.A.: An analysis of desertification processes in the American Southwest. Climate Change 17: 305-330

Hadley NF and Szarek SR. 1981. Productivity of desert ecosystems. Bioscience 31(10): 747-753

Halvorson JJ, Bolton H and Smith JL. 1997. The pattern of soil variables related to *Artemisia tridentata* in a burned shrub-steppe site. Soil Science Society of America Journal 61: 287-294

Herman RP, Provencio KR, Herrera-Matos J and Torrez RJ. 1995. Resource islands predict the distribution of heterotrophic bacteria in Chihuahuan desert soils. Applied and Environmental Microbiology 61(5): 1816-1821

Hibbert AR. 1983. Water yield improvement potential by vegetation management on western rangelands. Water Resources Bulletin. 19: 375-381

Holmgren M. 2002. Exotic herbivores as drivers of plant invasion and switch to ecosystem alternative states. Biological Invasions 4: 25-33

Hook PB, Burke IC and Lauenroth WK. 1991. Heterogeneity of soil and plant N and C associated with individual plants and openings in North American shortgrass steppe. Plant and Soil 138: 247-256

Huenneke LF, Anderson JP, Remmenga M and Schlesinger WH. 2002. Desertification alters patterns of aboveground net primary production in Chihuahuan ecosystems. Global Change Biology 8: 247-264

Humphrey RR. 1958. The desert grassland: a history of vegetational change and an analysis of causes. Botanical Review 24:193-252

Johnson DD and Miller RF. 2006. Structure and development of expanding western juniper woodlands as influenced by two topographic variables. Forest Ecology and Management 229: 7-15

Kassas M. 1995. Desertification: a general review. Journal of Arid Environments 30: 115-128

Keeney DR and Nelson DW. 1982. Nitrogen – inorganic forms. p. 643-698. In A.L. Page, Miller RH, Keeney DR (eds.) Methods of soil analysis, part 2. Agron. Monogr. 9, Am. Soc. Agron., Madison, Wisconsin.

Klemmedson JO and Tiedemann AR. 1986. Long-term effects of mesquite removal on soil characteristics: II. Nutrient availability. Soil Science Society of America Journal 50: 476-480

Klemmedson JO and Tiedemann AR. 2000. Influence of western juniper development on distribution of soil and organic layer nutrients. Northwest Science 74(1): 1-10

Klopatek JM. 1987. Nitrogen mineralization and nitrification in mineral soils of pinyon-juniper ecosystems. Soil Science Society of America (51): 453–457

Knapp PA and Soulé PT. 1996. Vegetation change and the role of atmospheric CO_2 enrichment on a relict site in central Oregon: 1960-1994. Annals of the Association of American Geographers 86(3): 387-411

Knapp PA and Soulé PT. 1998. Recent *Juniperus occidentalis* (Western Juniper) expansion on a protected site in central Oregon. Global Change Biology 4(3): 347–357.

Knapp PA, Soulé PT and Grissino-Mayer HD. 2001. Post-drought growth responses of western juniper (Juniperus occidentalis var. occidentalis) in central Oregon. Geophysical Research Letters 28(13): 2657-2660

Kramer S, Miller PM and Eddleman LE. 1996. Root system morphology and development of seedling and juvenile Juniperus occidentalis. Forest Ecology and Management 86: 229-240

Lajtha K and Schlesinger WH. 1986. Plant response to variations in nitrogen availability in a desert shrubland community. Biogeochemistry 2(1): 29-37

Lederer RJ. 1977. Winter territoriality and foraging behavior of the Townsend's solitaire. American Midland Naturalist 97: 101-109

Lodhi MK. 1977. The influence and comparison of individual forest trees on soil properties and possible inhibition of nitrification due to intact vegetation. American Journal of Botany 64(3): 260-264

Mallarino A and Sawyer J. 1999. Interpreting Mehlich-3 soil test results. p. 11-14. *In* Integrated Crop Management Newsletter, IC-482(2). Iowa State University, Ames, IA

Maser C and Gashwiler JS. 1978. Interrelationships of wildfire and western juniper. 37-82. *In*: Martin RE, Dealy JE, and Caraher DL (eds.). Proc. Western juniper ecology and management workshop. USDA For. Ser. Gen. Tech. Rep. PNW-74.

McArthur ED and Young SA. 1999. Development of native seed supplies to support restoration of pinyon-juniper sites. In: Monsen SB, Stevens R, compilers. Proceedings: ecology and management of pinyon-juniper communities within the Interior West; 1997 September 15-18; Provo, UT. Ogden (UT): USDA Forest Service, Rocky Mountain Research Station. Proceedings RMRSP-9. p 327-330

McLean EO. 1982. Soil pH and lime requirement. P. 199-223. In Page AL., Miller RH and Keeney DR (eds.) Methods of soil analysis, part 2. Agron. Monogr. 9, Am. Soc. Agron., Madison, Wisc.

Mehlich A. 1984. Mehlich 3 soil test extractant: A modification of Mehlich 2 extractant. Commun. Soil. Sci. Plant Anal. 15: 1409-1416

Miller RF, Angell RF and Eddleman LE. 1987. Water use by western juniper. pp. 418–422 *In:* R.L. Everett, editor. Proceedings-Pinyon-juniper conference Gen. Tech. Rep. INT-215. USDA Forest Serv., Ogden, Ut.

Miller RF, Bates JD, Svejcar TJ, Pierson FB and Eddleman LE. 2005. Biology, ecology, and management of western juniper (*Juniperus occidentalis*). Corvallis: Oregon State University Agricultural Experiment Station Technical Bulletin 152. 77 p.

Miller RF and Rose JA. 1995. Historic expansion of *Juniperus occidentalis* (Western juniper) in southeastern Oregon. Great Basin Naturalist 55(1): 37-45

Miller RF and Rose JA. 1999. Fire history and western juniper encroachment in sagebrush steppe. Journal of Range Management 52: 550–559

Miller RF, Svejcar TJ and Rose JR. 2000. Impacts of western juniper on plant community composition and structure. Journal of Range Management 53: 574-585

Miller RF and Tausch RJ. 2001. The role of fire in juniper and pinyon woodlands: a descriptive analysis, p. 15–30. *In:* K.E.M.Galley and T.P. Wilson (eds.) Proc. Invasive Species Workshop: the Role of Fire in the Control and Spread of Invasive Species. Misc. Pub. No. 11, Tall Timbers Res. Sta. Tallahassee, Fla.

Miller RF, Tausch R and Waichler W. 1999. Old-growth juniper and pinyon woodlands. USDA Forest Service Proceedings RMRS-P-9

Miller RF and Wigand PE. 1994. Holocene changes in semiarid pinyon-juniper communities. Bioscience 44(7): 465-475

Noy-Meir I. 1985. Desert ecosystem structure and function, p. 93-103. *In* M. Evenari, I. Noy-Meir and D. Goodall (ed.), Hot deserts and arid shrublands. Elsevier, Amsterdam

Murray, R.B., 1975. Effects of Artemesia tridentata removal on mineral cycling. Ph.D. Dissertation. Washington State University, Pullman, Washington, 190 pp.

Parsons AJ, Abrahams AD and Simanton JR. 1992. Microtopography and soil-surface materials on semi-arid piedmont hillslopes, southern Arizona. Journal of Arid Environments 22: 107-115

Peterjohn WT and Schlesinger WH. 1991. Factors controlling denitrification in a Chihuahuan desert ecosystem. Soil Science Society of America Journal 55: 1694-1701

Pierson, F.B., Kormos, P.R., Williams, C.J., 2007. The influence of woodland encroachment on runoff and erosion in sagebrush steppe systems, Great Basin, USA. Eos Trans. AGU, 88(52), Fall Meet. Suppl., Abstract H31G-0750.

Poddar S and Lederer RJ. 1982. Juniper berries as an exclusive winter forage for Townsend's solitaires. American Midland Naturalist 108: 34-40

Reynolds JF, Virginia RA and Schlesinger WH. 1997. Defining functional types for models of desertification. In TM Smith, HH Shugart, Woodward FI, editors. Functional types. Cambridge University Press, Cambridge, England.

Roberts C and Jones JA. 2000. Soil patchiness in juniper-sagebrush-grass communities of central Oregon. Plant and Soil 223: 45-61

Rose JR and Eddleman LE. 1994. Ponderosa pine and understory growth following western juniper removal. Northwest Science 68: 79-85

Rostagno CM. 1989. Infiltration and sediment production as affected by soil surface conditions in a shrubland of Patagonia, Argentina. Journal of Range Management 42: 382-385

Rostagno CM, del Valle HF and Videla L. 1991. The influence of shrubs on some chemical and physical properties of an aridic soil in north-eastern Patagonia, Argentina. Journal of Arid Environments 20: 179-188

Santos PF, DePree E and Whitford WG. 1978. Spatial distribution of litter and microarthropods in a Chihuahuan desert ecosystem. Journal of Arid Environments 1: 41-48

Schlesinger WH and Pilmanis AM. 1998. Plant-soil interactions in deserts. Biogeochemistry 42: 169-187

Schlesinger WH, Raikes JA, Hartley AE and Cross AF. 1996. On the spatial pattern of soil nutrients in desert ecosystems. Ecology 77(2): 364-374

Schlesinger WH, Reynolds JF, Cunningham GL, Huenneke LF, Jarrell WM, Virginia RA and Whitford WG. 1990. Biological feedbacks in global desertification. Science 247: 1043-1048

Seyfried MS and Wilcox BP. 1995. Scale and the nature of spatial variability: field examples having implications for hydrologic modeling. Water Resources Research 31: 173-184

Soulé PT, Knapp PA and Grissino-Mayer HD. 2004. Human agency, environmental drivers, and western juniper establishment during the late Holocene. Ecological Applications 14:96-112

Stock WD, Dlamini TS and Cowling RM. 1999. Plant induced fertile islands as possible indicators of desertification in a succulent desert ecosystem in northern Namaqualand, South Africa

Stubbs MM and Pyke DA. 2005. Available nitrogen: A time-based study of manipulated resource islands. Plant and Soil 270: 123-133

Tiedemann AR and Klemmedson JO. 1995. The influence of western juniper development on soil nutrient availability. Northwest Science 69(1): 1-8

Tiedemann AR and Klemmedson JO. 2000. Biomass and nutrient distribution and system nutrient budget for western juniper in Central Oregon. Northwest Science 74(1): 12-23

Trappe JA. 1981. Mycorrhizae and productivity or arid and semiarid rangelands. *In* Advances in Food Producing Systems for Arid and Semiarid Lands. pp 581-599. Academic Press, New York

UNCCD (United Nations Convention to Combat Desertification). 1994. *United Nations Convention to Combat Desertification*, Geneva: United Nations.

UNEP (United Nations Environment Programme). 1997. World Atlas of Desertification. Edward Arnold: London.

UNEP (United Nations Environment Programme). 2002. Global Environment Outlook 3. United Nations Environment Programme: Nairobi

USDA (United States Department of Agriculture), NRCS (Natural Resources Conservation Service). 2006. National resources inventory: 2003 annual NRI

USDA (United States Department of Agriculture), NRCS (Natural Resources Conservation Service). Web soil survey. 5 June 2007 Available online at <http://websoilsurvey.nrcs.usda.gov>

Vaitkus M and Eddleman LE. 1987. Composition and productivity of a western juniper understory and its response to canopy removal: *In* Everett RL [ed]. Proceedings: Pinyon-Juniper Conference. Ogden, UT: Intermountain Research Station, USDA-Forest Service General Technical Report INT-215 pp. 456-460

Van Auken OW. 2000. Shrub invasions of North American semiarid grasslands. Annual Review of Ecology and Systematics 31: 197-215.

Whitford WG, Anderson J and Rice PM. 1997. Stemflow contribution to the 'fertile island' effect in creosotebush, *Larrea tridentate*. Journal of Arid Environments 35: 451-457

Whitford WG, Martinez-Turanzas G and Martinez-Meza E. 1995. Persistence of desertified ecosystems: explanations and implications. Environmental Monitoring and Assessment 37: 319-332

Wilcox BP. 1994. Runoff and erosion in intercanopy zones of pinyon-juniper woodlands. Journal of Range Management 47: 285-295

Yager LY and Smeins FE. 1999. Ashe juniper (Juniperus ashei: cupressaceae) canopy and litter effects on understory vegetation in a juniper-oak savanna. The Southwestern Naturalist 44(1): 6-16

Young JA, Evans RA and Rimby C. 1985. Composition and productivity of a western juniper understory and its response to canopy removal: *In*: RL Everett [ED.]. Proceedings: Pinyon-Juniper Conference. Ogden, UT: Intermountain Research Station, USDA-For. Ser. Gen. Tech. Rep. INT-215. p. 456-460

Zaady E, Groffman PM and Shachak M. 1996. Litter as a regulator of N and C dynamics in macrophytic patches in Negev desert soils. Soil Biology and Biochemistry 28(1): 39-46

Zinke PJ. 1962. The pattern of influence of individual forest trees on soil properties. Ecology 43(1): 130-133

Zophy KT. c2006. Spatial and age class analysis of managed western juniper (Juniperus occidentalis) woodlands in central Oregon. Thesis (MS)-Oregon State University, 2007