Effect of Plant Community on Soil Organic Carbon in the Chewaucan River Basin

by

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Abstract

Variation in plant community composition has been shown to alter the concentrations of soil organic carbon (SOC) within the soil. Climate change, and anthropogenic disturbances have altered dominant plant communities across the globe, shifting them to new states of equilibrium, with important implications for SOC. The relationship between plant community and SOC is well understood in many regions; however, semi-arid ecosystems remain poorly represented in existing datasets liking above and belowground ecosystem properties. For example, in the Northwestern United States, ecosystems of the Great Basin have been both underfunded and understudied in terms of management and scholarship respectively. This is especially true within the state of Oregon, where a clear majority of research efforts are devoted to ecosystems West of the Cascades as opposed to the semi-arid ecosystems of the East. Ecosystems of eastern Oregon are undergoing rapid vegetation regime shifts that include woody encroachment, grass invasions, and large scale agricultural expansion simultaneously. To better understand the fundamental relationship between changes in plant communities and SOC within typical ecosystems of eastern Oregon, 14 plots were deployed to sample soil and vegetation across the Chewaucan

Basin. Sampling sites were chosen based on a priori vegetation community type that included sagebrush (*Artemisia tridentate, & Artemisia arbuscula),* western juniper (*Juniperus occidentalis),* ponderosa pine (*Pinus ponderosa*), western juniper/ponderosa pine, and Alfalfa (*Medicago sativa)*. Surface SOC was characterized for each vegetation category at a depth of 10 cm using the loss on ignition method. Woody vegetation communities showed substantially more SOC when compared to sagebrush and Alfalfa crops with plot average values ranging 67-83 Mg ha^{-1} to 33-47 Mg C ha^{-1} respectively. Sagebrush communities showed intermediate levels of SOC with 47 Mg C ha^{-1}, Undeveloped plots directly adjacent to agriculture with 33.5 Mg C ha^{-1}, and Alfalfa plots exhibited 43.5 Mg C ha^{-1}. Juniper plots showed 67.8 Mg C ha^{-1}, Pine dominated plots exhibited 83.6 Mg C ha^{-1}, and Juniper/Pine dominated displayed 67 Mg C ha^{-1}. This pilot study gives valuable insight into the current state of semi-arid ecosystems and provides the basis for future assessments of changes in vegetation cover and SOC concentrations.

Introduction

Climate change is part of a larger quite fundamental transformation of Earth's systems, affecting the composition of the atmosphere, biosphere, hydrosphere pedosphere and the cycling of carbon in and between them. It is well established that the carbon cycle is essential to the continuation and evolution of life on Earth (Wapner & Elver, 2016 and Farmer & Cook, 2012); however, many question remain unanswered regarding the evident disequilibrium between sources and sinks of carbon on land, with excess carbon leaving terrestrial systems. Due primarily to human activity, such disequilibrium has caused the concentration of atmospheric carbon dioxide (CO_2) to increase by nearly 40% since the start of the industrial revolution (Tian et al 2016). Furthermore, it reflects a basic shift in terrestrial ecosystem functions with potentially devastating consequences such as climatic instability and changes in composition and distribution of plant communities as they to reach ecological thresholds. An ecological threshold occurs when external factors, positive feedbacks, or nonlinear instabilities cause changes to propagate in a "domino-like" fashion that can in some cases be irreversible (Mavrommati et al 2016). Sudden changes to ecosystem composition and function are not fully understood, but they are unquestionably important if natural resource managers are to succeed in developing adaptation strategies in a changing world.

One example of predicted large scale ecosystem shifts associated with changes in the carbon cycle is the decline of savanna and grassland biomes across the globe. Savanna and grassland ecosystems cover approximately 3.5 million km^2 and are responsible for approximately 20 Pg C $year^{-1}$ of the globe's net primary productivity (Silva and Anand 2013). Specific alterations in vegetation cover associated with changes in climate and the carbon cycle include the expansion of woody vegetation into these grassland ecosystems due to CO_2

stimulation of tree growth (Santana 2017). The shifting dominance among herbaceous and woody vegetation within these ecosystems, alters primary production, plant allocation, rooting depth and soil faunal communities, potentially meters beneath the soil surface, and in turn are expected to alter nutrient cycling and carbon storage (Jackson et. al 2002).

In addition to CO_2-induced changes in vegetation cover, changes in temperature, precipitation, and deposition of biologically available nitrogen have increased across large regions, and further increases are almost certain in the future. (Farmer & Cook 2012 and Shaw et. al 2002). Disequilibrium within terrestrial ecosystems caused by the combined effect of CO_2 and nutrient pollution and changes in vegetation cover and climate have manifested all over the world as increasing humidity or aridity depending on the region and portion of the atmosphere studied (Bonan et al 2015); increased evaporation of ocean and fresh water and moisture from soils (Seneviratne et.al 2010, & Bates et. al 2008); increasing frequency and intensity of storms and unusual weather patterns (Mann et al 2017); melting glaciers and melting permafrost and release of methane to the atmosphere (Silva and Anand 2013; Silva et al 2016); decreasing snow cover in winter and temperature increases over land and sea; as well as increasing animal and plant species extinction (Farmer & Cook, 2012 and Glover, Ishee, & Collies, 2009). Taken together, these phenomena encompass the most important local and regional manifestations of global environmental change.

Crucial for planetary stability, forest biomes are known to control much of the global carbon budget and are generally considered to be net carbon sinks (Bonan et al 2015). Trees and forests are structurally and compositionally diverse and, thus, apt to dealing with at least some of the predicted effects of global environmental change (e.g., climate fluctuations). However, in many instances, the fast rate of human-induced environmental change has been found to impose severe limitations to the capacity of trees to adapt to new environmental conditions (Alcamo et. al, 2007). For example, when combined with land use and pressures and resulting habitat fragmentation, the effects of climate change are expected to place unique challenges on forest dynamics, such as tree growth decline due to drought and increasing temperatures, which can cause higher respiration rates while photosynthetic rates are reduced by dry conditions (Gomez-Guerrero 2013). Increasingly warm and dry conditions are also expected to increase the severity of drastic biotic and abiotic disturbances that can greatly impact forest ecosystems, such as those caused by insect infestation and fires (Collins et al 2012 & Harvey 2016). Thus, the impact of climate on a forest ecosystem will vary depending upon many other factors that limit tree growth and regeneration, including latitude and altitude limitations, and their inherent relationship with soils (Bravo, 2008).

A rough estimate suggests that approximately 18% of global carbon emissions are associated with deforestation, a major contributor to rising atmospheric CO_2 levels (Stern, 2006). To a large extent, the reduction of forest biomes is to blame. Tropical forests make up large areas of South America, Africa, and Indonesia are largely due to direct human activities such as agricultural expansion and deforestation, which a decade ago had already released approximately 1.6 (0.5–2.7) GtC yr^{-1} (Achard et. al, 2007). The tipping point of tropical rainforests due to increased CO_2 concentrations in the atmosphere could result in developments of savanna type ecosystems (Nobre & Borma, 2009). Temperate forest, bordered by boreal forest ecosystems to the North 55° N, and tropical ecosystems to the South 30°N, compromise approximately 25% of the worlds forest, and act as a sink of atmospheric CO_2, taking up 37% of the worlds terrestrial carbon uptake (Tyrell, Ross, & Kelty, 2011). Temperate forest that exhibit xeric moisture regimes are in part shaped by fire (National Interagency Fire Center, 2011); thus, as climate continues to warm, frequency, intensity, and duration of fire are expected to increase, altering successional stages of forest development and reducing the chances of forest reaching old growth status, the pinnacle of structural diversity (Tyrell, Ross, & Kelty, 2011 and Rapp, 2002). Boreal forests extend northward to about 68° N. These forests contain approximately 13% of the planets terrestrial biomass, and contains approximately 43% of the worlds soil carbon stocks within rich organic soils (Milakovsky, Frey, & James, 2011). Boreal forests are the most under represented biome in terms of research regarding their nutrient and energy networks, due to the intact nature of forest stands, and their immense area and narrow study window (Milakovsky, Frey, & James, 2011). Boreal forests and their soils are the most susceptible to climate warming and unstable precipitation, which can cause extensive forest dieback, increased fire disturbance, intense insect infestations, and severe permafrost de-thaw (Wooster and Zhang 2004, Kasischke et al. 1995, Malmstrom and Raffa 2000, Prokushkin et al. 2005). Interestingly, more rapid climate warming in the high latitude regions of the world can influence spatial borders between boreal forests and tundra ecosystem. Tundra ecosystems retain approximately 97% of their carbon within their soils, and current vegetation shifts northward can potentially release stored soil carbon (Billings, 1987 and Soja et. al, 2007).

Importance of arid ecosystems of eastern Oregon

As in other parts of the world, the forests of western Oregon are experiencing a process of "savannization" that will likely exacerbate climate warming and establish an overall drier water regime than that observed in the region today (Baldocchi, Chu, & Reichstein, 2018). In addition, as temperatures increase across the mixed coniferous forests, increased fuel loads and fire intensity are also promoting a shift toward increasing aridity west of the Cascades (Dalton et al, 2017). Similar albeit stronger eco-climatological regime shifts are expected to occur in arid ecosystems east of the Cascades. However, little research is being done to predict the consequences of climate changes within those dry systems, which encompass coniferous forest and adjacent sagebrush steppe ecosystems. Interestingly, divergent vegetation trajectories are juxtaposed in eastern Oregon, where savannaization and woody encroachment are occurring simultaneously (Rau 2011a, Rau 2011b Miller and Rose 1999, Miller et al 2013, Miller et al 2008), allowing a unique opportunity to study the relationship between plant cover and SOC in co-located yet contrasting ecological settings. To date, the few studies that have analyzed the effects of vegetation shifts in the semi-arid ecosystems of Oregon East of the Cascades, have failed to include land conversion to agriculture that totals approximately 13.8 million acres East of the Cascades (Oregon State Board of Agriculture 2017). The present study represents a first step towards addressing this limitation.

Arid and semi-arid ecosystems make up approximately 1/3 of the worlds continental land surface and face the most serious threats from ongoing and future climate change as major water regime shifts could occur on timescales of a few years to decades (Sadoff and Muller 2009, Cook et. al 2014, and Chambers and Wisdom 2009). Changes in water availability are predicted to alter vegetation communities of the sagebrush steppe ecosystems of the Great Basin in the western US (Miller et. al, 2013), which is the largest semi-arid ecosystem in the US totaling approximately 100 million hectares (Miller, 2010). It extends across southern Washington, eastern Oregon, northeastern California, southern Idaho, northern two-thirds of Nevada, and the western half of Utah, encompassing the Central Basin, Northern Basin, Columbia Basin, Snake River Plain, and Blue Mountain ecoregions (Miller et. al, 2013). Its topography is characterized by basins, mountains, and plateaus with elevations ranging from 400 to 3000 m but these majority falling between 760 to 2300 m. The sagebrush steppe represents vast areas of similar sagebrush/shrub vegetation states that defined as complex of a soil base and a suite of complex vegetation communities that are resistant to disturbances (Miller et. al, 2013 & Davies et. al, 2012). The sagebrush steppe has been identified as one of the most threatened land types in North America, and as much as half of this land type has already been lost in the Great Basin. Many of the plant communities that remain unaltered in this region are in poor conditions and are in need of restoration efforts. As stated above, climate change is threatening the sagebrush steppe and its functional dynamics; however, little research is being done to study the effects of a changing sagebrush steppe in relation to increased agriculture, and other economically productive activities, which are related to woody species encroachment or decline (Chambers and Wisdom 2009).

Ecological regime shifts within the sagebrush steppe include the rapid advance of western juniper and the envelopment of sagebrush communities, which in some regions has altered above and below ground carbon stocks (Rau 2011a, Miller and Rose 1999, Strand et al 2008, Miller et al 2008, Miller et al 2013, Brown et al 1997, Campbell et al 2012 Chambers and Wisdom 2009). According to Miller et al 2008, some areas have experienced as much as a 625% increase in juniper stands since at 1860 within the sagebrush steppe. Juniper encroachment is viewed as a problem, because woody encroachment occurs at the expense of sensitive grass and shrub lands already becoming vulnerable to agriculture and urbanization (Campbell et al 2012). Facilitation of juniper expansion at such a rapid pace within the sagebrush steppe is caused by intense livestock grazing, reduced frequency of fire, and optimal climate conditions during peak expansion periods (Miller and Rose 1999). Increased juniper expansion has consequences beyond loss of native shrubs and grasslands, as it changes nutrient cycles within the soil profile.

Scientists have suggested that increased woody expansion increases organic carbon stores above and below ground by as much as 0.13 Pg C $year^{-1}$, across vast regions of the west (Pacala et al, 2001). Specific analysis of below ground carbon accumulation has been measured at 5.1 Mg $year^{-1}$, due to increased litter accumulation in soils associated with higher densities of woody species like juniper (Rau 2011a). The importance of belowground SOC accumulation in arid and semi-arid landscapes is the proportion of mean annual temperature and precipitation amounts. Arid and semi-arid ecosystems in the west do not receive precipitation amounts in excess of 500mm, which is the threshold associated with increased microbial respiration and a loss of SOC (Rau 2011a). However, as density of woody cover increases, risk of severe fire also increases allowing for exponential losses of sequestered carbon stocks across the west allowing for potential and rapid cheatgrass invasion (Miller and Tausch 2001).

Self organizing non-woody vegetation states within the Great Basin are largely dependent on precipitation amounts with Desert shrub communities dominating in areas that receive fewer than 150 mm precipitation per year. Sagebrush communities are prominent in areas that receive 200-300 mm of precipitation that include, Wyoming big sagebrush, basin big sagebrush, low sagebrush and others. A mix of woody vegetation and sagebrush communities persist in upland areas that receive approximately 300-400 mm of precipitation and include mountain big sagebrush, low sagebrush, bitter brush, and western juniper. Lastly, areas that experience an average of 400 mm of precipitation per year contain mountain big sagebrush, bitterbrush, curled leaf mountain mahogany, Idaho fescue and others (Miller et. al, 2013). Sagebrush dominated ecosystems are often subject to disturbances such as fire, climatic changes, as well as anthropogenic influences. It is commonplace to see major vegetation community shifts after such disturbances have taken place such as woody juniper encroachment, cheatgrass intrusions, and other invasive species (Rau et. al 2011a, Rau et. al 2011b, Campbell et. al 2012, Miler and Rose 1999).

Within the sagebrush steppe ecosystem, documentation of invasive species is occurring at unprecedented rates, with at least some presence >10% in approximately 7 million hectares (Miller et. al 2008). In salt desert shrub, Wyoming sagebrush, and lower elevation mountain big sagebrush vegetation types, the annual grass-fire cycles in response to altered fire regimes, are resulting in progressive conversion of native shrub lands to homogenous grasslands dominated by nonnative invasive species and a loss of endemic species (Brooks and Pyke 2001 & Chambers and Wisdom 2009). The most dominant nonnative invasive grass is cheatgrass, which is thought to have increased fire intervals exponentially from historic 60-110 years to modern 3-5 years. The increased fire interval creates a negative feedback loop that perpetuates the continuation of cheatgrass establishment and invasion. (Brooks and Pyke 2001 & Miller et. al 2013). In contrast with woody species expansion, invasive grass invasion is supported by the increased resource availability in the first few years after fire disturbance, with above ground fuel loads estimated at 14.5kg/ha for pre-fire cheatgrass populations, growing to 732kg/ha in the second and third years post fire on average (Miller et al 2013). This conversion and increased fire interval can severely degrade ecosystems by reducing SOC and nitrogen stocks above the ground and slowing the infiltration rates of water into soils (Rau 2011b). Thus, the study of co-located open and woody vegetation physiognomies represents a critical step towards addressing land management and conservation questions in this region.

Vegetation Cover and Soil Organic Carbon

The paucity of studies regarding vegetation community structures and their relationship to below ground SOC creates a less robust picture of carbon cycling and storage and the relationships between soil, plants, and the atmosphere on a regional scale. Analyzing these relationships within the scope of the sagebrush steppe has the potential increase knowledge about carbon storage across a vast land area. However, it is important to define the characteristics of SOC to understand its importance to the overall ecosystem function. Current levels of SOC in an ecosystem represent a continuum of materials in varying states of decomposition with differing residence times following the deposition of plant biomass (Jastrow and Miller 1998). Generally, SOC is comprised of protected in soil aggregates within the soils matrix, as well as free particles and compounds adsorbed onto mineral surfaces, ranging from inaccessible to readily available substrate for microbial activity across a variety of exposure levels to oxygen (Hartemink and McSweeny 2014). Thus, the composition of SOC within the soil as well as the degree of its recalcitrance is highly dependent upon the molecular structure of the original carbon inputs, which are dependent on vegetation cover, allowing for differentiation of SOC within the soil across plant communities. Different forms of SOC are typically quantified based on the composition of the input and the timescale of SOC decomposition. Nevertheless, it is important to note that changes in climate and precipitation have significantly impacted the stability of SOC, especially as one travels upward through the soil matrix (Baldock 2007). Of the five forms of SOC the least stable is surface plant residue, composed of plant material and litter residing in the soils surface. The next least stable is Buried plant residue, consisting of plant material greater than 2mm in diameter. Third is Particulate organic matter (POC), which is semi-decomposed organic matter smaller than 2 mm and greater than 50 micrometers in diameter. Fourth is Humus which is well decomposed organic material smaller than 50 micrometers that is well incorporated into soil particles. Humus is very stable and is an important facilitator in soil structure. The most stable form of SOC is Resistant organic carbon (ROC) composed of charcoal or charred materials that result from the burning of organic matter (Hartemink and McSweeny 2014). Important functions of SOC include, nutrient cycling, soil architecture, soil fertility and yield production as well as dynamics in relation to climate change mitigation, management practices and modelling (Janzen 2006). Of these functions, climate change mitigation and soil functionality stand out as the focal points of soil carbon research and therefore are also the focus of this project.

Soil functionality in relation to SOC is a balance between sequestration of SOC stocks and how SOC is used within the soil matrix, as fuel for biological activity (Hartemink and McSweeny 2014). Increases of carbon within soils improves a soils quality, in reference to its water holding capacity, aggregate stability, fertility, and ion exchange buffering. However, for a soil to be functional the carbon sequestered into the soil must be broken down and in a state of decay, for it is in this state of decay in which the carbon can be accessed and used for biological processes (HH Janzen, J Six, Zvomuya et al, 2005). Although there is evidence that suggests soil functionality follows a broken stick regression model, where soil functionality and fertility improve to a certain point and flatline, in which soils can be both functional as well as a carbon sink. Moreover, SOC concentrations in regard to climate change mitigation is a current focal point for soil science research (Sommer & Basio 2014). There is increasing evidence that soils can mitigate climate change by reducing greenhouse gas emissions by way of soil carbon sequestration (SCS) i.e. the removal of CO2 from the atmosphere and storing it into a long lived carbon pools described above. Globally, SCS is recognized as a carbon sink due to the estimated size of the soil C pool estimated at 560 Gt of OC (Jobbágy and Jackson 2000). In addition, anthropogenic influences mainly land management, have induced changes in C that have resulted in an estimated loss of SOC of up to 78 Gt per annum (Lal 2004). Fortunately, research has been done that estimates of maximum SOC storage potential for a range of soil types and management regimes and has been estimated at 31 to 64 Gt in the next century in agricultural soils alone (Sommer & Basio 2014). Agricultural soils in particular have increased carbon sequestration potential due to their increased C:N ratio, as long as management practices reflect sustainable soil stewardship (Rau 2011a, & Wilman 2011). Agriculture in the western US alone encompasses approximately 300 million hectares, suggesting massive carbon sequestering potential (Office of Technology Assessment, 2004), which must be contrasted with tree-dominated ecosystems, which typically hold the highest levels of SOC across biomes (Duarte-Guardia et al., 2018).

A case of land cover and soil properties in eastern Oregon

The focus of this study is the relationship between shifting plant community composition and structure in agriculture and non-managed lands in the sagebrush steppe of eastern Oregon. Differences in the below ground SOC concentrations of the surface soil horizons were compared across multiple plant communities to tease apart the differences of surface SOC concentrations in in a variety co-located systems, which provide a robust perspective on vegetation effects than studying SOC separately at different locations. This study therefore captures a snapshot of the current state and potential changes occurring across such a number of climate-sensitive ecosystem in eastern Oregon.

Study Site

The Chewaucan River Basin in South central Oregon located within the Fremont National Forest, 8 km south of Paisley, was chosen as the study site based on previous land management regimes, on going vegetation changes, and current agricultural practices in close proximity. The study area encompasses approximately 5,000 ha, and elevation ranged between 1,450 and 1,875 m above the sea level. Topography is typified by highly dissected “benchland”s and “toe slopes” ranging from gentle to moderately steep (Miller and Rose 1999) with slope aspect oriented predominantly west to northwest. Soils across the landscape range from deep to moderately deep residual soils weathered from breccias and tuffs to shallow clayey soils (Wenzel 1979). The local climate is classified as cool and semi-arid, characteristic of the northern Great Basin, with long-term average precipitation of approximately 400 mm per year (Taylor 1993) received primarily as snow in November to January and as rain March through June. The vegetation is characterized by 2 predominant plant communities. On moderately deep soils, mountain big sagebrush (*Artemisia tridentata* Nutt.) with Idaho fescue (*Festuca idahoensis* Elmer) dominates. The low sagebrush (*Artemisia arbuscula* Nutt.)-sandberg bluegrass (*Poa sandbergii* Vasey) community occupies the stony shallow heavy clay soils on the benchlands. Associated with these plant communities are western juniper trees in varying levels of density, and Ponderosa pine (*Pinus ponderosa*) at the highest elevations.

Livestock was introduced to the Chewaucan River Basin in the late 1860s (Oliphant 1968). By the mid 1870’s a few thousand cattle were documented in the lower basin, with several thousand sheep moving in later (Miller and Rose 1999). By the turn of the century livestock numbers peaked and remained high until about 1915, with sheep populations declining form approximately 400,000 AUM (animal unit months) to less than 1,000 currently (Miller and Rose 1999). An AUM is the average amount of dry weight forage required by a lactating 1000-pound cow and her calf for one month, 30.4, days (Ogle et al 2009). Cattle populations since documented decline in 1915 have decreased from 95,000 AUM’s to about 60,000 AUM’s currently (Miller and Rose 1999). The inception of the US Forest Service Ranger Station in

Paisley in 1908 marks the beginning of fire suppression in the area, but little suppression effort was noted because of limited access to effected areas until the 1940's (Miller and Rose 1999).

<u>Methods</u>

Determination of sites within the study area was done based on dominant a priori vegetation community characterization that divided the landscape into the following categories: mountain big sagebrush (*Artemisia tridentata* Nutt.) and low sagebrush (*Artemisia arbuscula* Nutt.), western juniper (*Juniperus occidentalis*), western juniper-Ponderosa Pine, Ponderosa Pine, and alfalfa (*Medicago sativa*). Specific site selection was controlled by prioritizing a north by northwest aspect for all sampling locations where a 25 m transect tape (running North to South) was placed perpendicular to another 25 m transect (running East and West) creating four equally sized quadrants. A spherical photograph was taken at the plot center, along with a GPS reading using a Garmin GPSmap 60Csx. Samples were then collected at random within each quadrant of the transect using a metal collection cylinder with a volume of $88.09cm^3$. For samples to be analyzed singularly, the core(s) were taken from the top 10 cm of soil profiles and placed it in sample bags. Each sample bag was labeled with a clear and unambiguous identifier with a permanent marker. The sample identifier made reference to the land parcel, the plot, and the position of the sample with respect to the plot. This process was repeated over 14 separate sites yielding 55 usable soil samples. For drying and separating, a drying oven was heated to 60ºC and samples were transferred from plastic collection bags to paper bags, copying vegetation type and quadrant number.

Once samples were adequately dried, soil samples were separated into fine earth and coarse fragments using a dry sieving method. Samples were passed through a stack of sieves with a 4.75 mm sieve on the top and a 2.00 mm sieve below, and finally a collection pan at the bottom. Samples were dried and sieved for approximately 5 minutes each to adequately separate soil particles. Coarse fragments and fine earth materials were weighed independently and placed into separate bags. The fine earth portion of the samples were ground after sieving to ensure that homogenized samples were analyzed for SOC content. The steps of separation and weighing samples individually allows for a more comprehensive analysis of particle size analysis as well as bulk density used in SOC stocks calculations. It is important to note the coarse fraction of the soil was disregarded as a SOC store.

To calculate SOC levels, a loss on ignition (LOI) test was conducted using a Fisher Scientific Isotemp 650 series muffle furnace. Approximately 5 g of soil was obtained from each of the 55 samples collected were re-dried prior to ignition in the furnace. Samples were weighed before and after burning of carbon, utilizing a desiccator for transport within the lab space to account for reabsorption of atmospheric moisture, and SOC levels were determined based on mass differences (Heiri et al 1999). Variables including percent SOC, bulk density, and elevation were analyzed to determine the influence of plot type on SOC stocks at the top 10 cm depth of the soil profile. Statistical computing packages in R were used in the creation of all graphs and standard errors calculations. To create figures depicting the study area within the Chewaucan Basin (Figure 1), Google Earth 2018 was used with an NGS topographic overlay.

Results and Discussion

Results show significantly greater percent composition of SOC, on a mass basis, in tree-dominated sites compared to agriculture or shrublands (Figure 2). This trend is also reflected in a comparison of percent SOC across elevation, which in this region is a proxy for plant community type (Figure 3). Perennial grasses, forbs, and shrub species in the Great Basin tend to have roots concentrated in the top 20 cm of soil (Rau 2011), and lower percent SOC in the sagebrush communities (ARTRV and Und), is not what was expected given sampling was limited to the top 10 cm of the soil profile. However, similar studies in the Great Basin account for the increased SOC found in woody vegetation communities, (Juoc/Pipo, Juoc, Pipo), as increased litterfall from trees becomes incorporated into near surface soils (Rau 2011). The top 10 cm of soil contains a carbon pool that is highly subject to changes associated not only with vegetation changes mentioned previously, but fire disturbances as well. Research has shown that as fire return intervals increase due to invasive species invasion and increased fuel loads, the greater the magnitude of SOC lost from surface soils that readily burn (Muqaddas, Zhou, Lewis, Wild, & Chen, 2015, and Miller et. al, 2013). Seasonal precipitation and temperature regime fluctuations due to climate change could further alter fire return intervals and exacerbate the already changing sagebrush steppe ecosystem.

On average, agricultural sites had slightly higher SOC levels in the soil surface compared to shrub and bunchgrass communities. Agricultural soils typical of the Chewaucan Basin are void of a discernable O horizon, leaving little carbon flux into the soils surface layers. Studies that researched soil carbon retention determined that increased carbon content is positively correlated with increased nitrogen content, and a plant-soil system with a lower C:N ratio can efficiently increase SOC within the soil as the rate of below ground root decomposition is increased (Lal 2009). Agricultural soils have been predicted to act as a carbon sink if the correct management techniques are implemented. Management techniques such as no till agriculture are especially effective at preserving SOC in surface soils ass Carbon residue inputs initially enter the surface pool where the tillage regime influences their capture and retention. With no-tillage practices, the residue is protected and forms SOC, which is transferred to the deeper pool and sequestered over an extended period of time (Wilman, 2011). However, it has been shown that long term synthetic fertilization of crops (<100yrs) can reduce SOC, despite increased below ground crop residue growth (Khan, Mulvaney, Ellsworth, & Boast, 2007).

Extrapolation of soil samples from the Chewaucan Basin (Figure 4) suggests that mean SOC stocks under woody vegetation, (juoc, pipo, juoc/pipo) range from 67-83 Mg ha^{-1} on average. While mean SOC stocks under sagebrush and alfalfa communities contain approximately 33-47 Mg ha^{-1} on average. It is important to note that the sampling strategy of this study was to collect only the top 10 cm of soils from across the Chewaucan Basin. However, a significant portion of the soil profile remains unstudied. There is potential for different conclusions to be made if SOC stocks were measured across the entire depth of the soil profile (Figure 4.), as rooting depths for alfalfa and woody tree species are significantly deeper that sagebrush communities (Rau 2009).

It is expected that more extensive studies in this region would show that that woody vegetation types allocate SOC further below ground than shrub and grass communities, and woody encroachment in these areas has the potential to act as a stronger carbon sink that the several-fold increase observed here for SOC from open to woody physiognomies (Figure 2). Alfalfa communities should display increased SOC as alfalfa's rooting depth and nitrogen content of soil far surpasses those of shrub and grass communities. Furthermore, soils within semi-arid ecosystems could potentially sequester carbon for longer periods of time, as precipitation and temperature values in these areas seem to suppress microbial activity and lower decomposition rates. It is also important to note that soils depleted of vegetation cover, less than 50%, are at higher risk of desertification and large losses of soil SOC. Therefore, increased attention to maintaining increased vegetation cover through agriculture, restoration, or conservation of the sagebrush steppe is vital in maintaining SOC stocks in the study region.

Finally, semi-arid soils are said to have higher soil inorganic carbon (SIC) than other soil types, measurements of these SIC stocks could grant even more information about the nature of the soils of semi-arid ecosystems , thus future studies should also take inorganic pools into account.

It is well known that SOC sequestered in soils and plant biomass can provide a direct market value for conservationists and land managers. Indeed, trading C credits offers a new hope to resource poor and small land holders of the region prone to desertification, like the sagebrush steppe, by creating another income stream (Lal, 2009). The economic gains, through increase in production and C sequestration, have a potential to positively impact the economy and the environment. The choice of strategies to mitigate C abundance or capture C from the atmosphere is also linked to C trading, which is an emerging global market (Lal, 2008). Generating C credits, which can create extra income to provide incentives to farmers to adopt recommended land use and management options, need a well-defined strategy which is beyond the scope of this study. Accordingly, this pilot study has important social and ecological implications as it represents an analysis of naturally occurring and agriculture systems simultaneously, which creates a more robust mosaic of Oregon's land as well as much of the west, that adds to a plethora of research already completed in western Oregon (Figure 5). Most notably, Bonan et al 2002 describes different forest biomes and their respective carbon allocations across Oregon, but fails to capture the unique qualities within the biomes of central and eastern Oregon. Bonan's analysis of Oregon's dominant vegetation regimes in the West reinforces the bias of studies allocated to western Oregon, but also points to a promising area of future research. As for this study specifically, Bonan's research acts as a launchpad for more thorough study of sagebrush steppe ecosystems throughout Oregon and most of the west to include above ground carbon allocation, respiration, and net primary productivity. This pilot study serves as a starting point to map SOC throughout Oregon, if not the Great Basin and set a starting point to which a Carbon market could be established. Continued study is needed throughout the Great Basin especially with respect to its soils, which holds the key to understanding broader soil, plant, atmosphere interactions in Oregon and elsewhere.

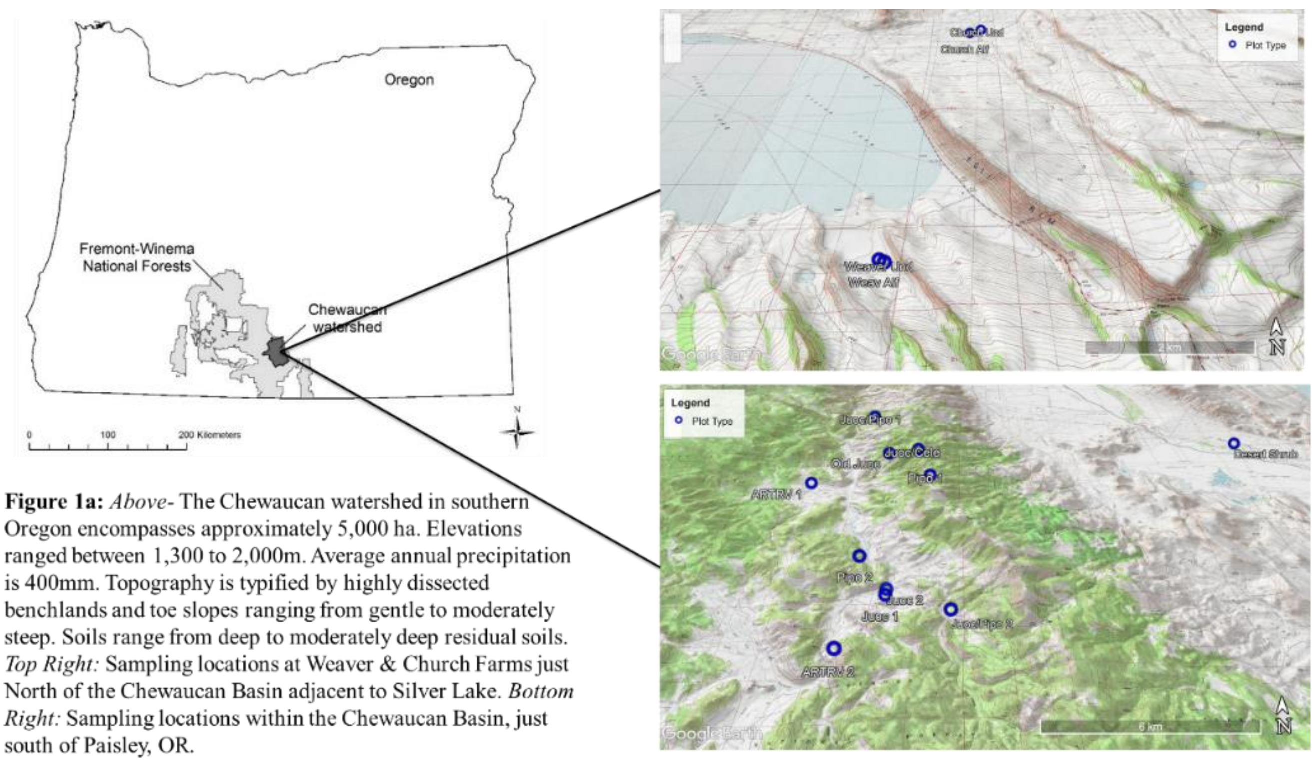


Figure 1a: *Above-* The Chewaucan watershed in southern Oregon encompasses approximately 5,000 ha. Elevations ranged between 1,300 to 2,000m. Average annual precipitation is 400mm. Topography is typified by highly dissected benchlands and toe slopes ranging from gentle to moderately steep. Soils range from deep to moderately deep residual soils. *Top Right:* Sampling locations at Weaver & Church Farms just North of the Chewaucan Basin adjacent to Silver Lake. *Bottom Right:* Sampling locations within the Chewaucan Basin, just south of Paisley, OR.




Figure 1b: Prominent vegetation communities within the Chewaucan Basin, used it determining soil sampling locations (Figure 1a). *Top Left:* Depicts a sagebrush community of mountain big sagebrush (*Artemisia tridentata* Nutt.) and low sagebrush (*Artemisia arbuscula* Nutt.) that denotes plot types Und and ARTRV. *Top Mid:* An old western juniper (*Juniperus occidentalis)* associated with plot types Juoc and Juoc/Pipo. *Top Right:* A large Ponderosa pine (*Pinus ponderosa*) representing plot types Pipo and Juoc/Pipo*Bottom. Left:* Alfalfa vegetation concomitant to dryland agriculture that signifies plot type Alf.

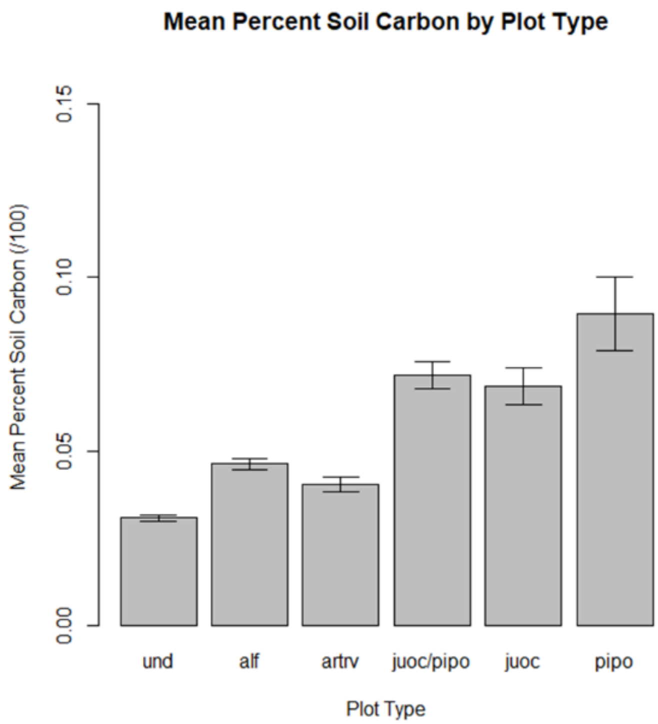


Figure 2: Means and standard errors of percent carbon content by plot type. SOC concentrations, and standard errors are calculated from the top 10cm of soil. % SOC= $\frac{predry\ weight - postdry\ weight}{predry\ weight}$ x 100

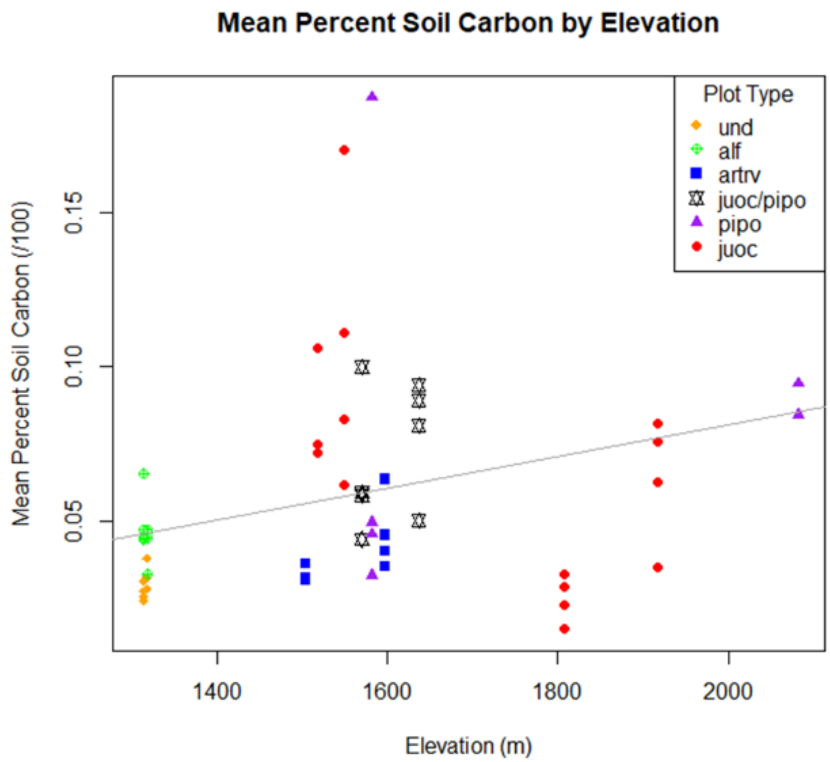
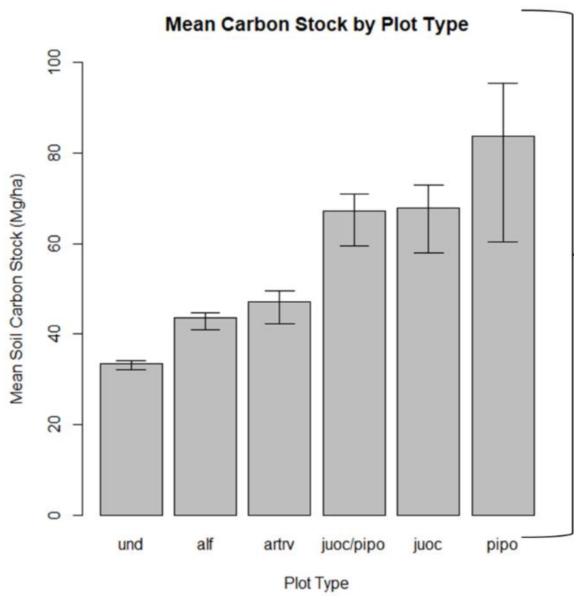


Figure 3: Scatterplot representing SOC values across elevation, and associations of plot type dependent upon elevation. SOC values are calculated from the top 10cm of soil. A regression line shows an increase of SOC with elevation and woody vegetation cover.



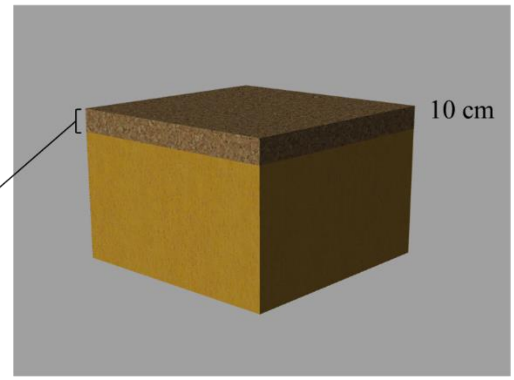


Figure 4: A bar plot representing mean carbon stocks within the top 10 cm of soil by plot type within the Chewaucan Basin. Denotation as to the scope of the study is represented in the 3D rendering on the right. Calculations of mean carbon stock are as follows.

$$\text{C Mg } ha^{-1} = \frac{bulk\ density\ g}{cm^3} \text{x} \frac{\%\ SOC}{100} \text{x} \frac{1{,}000{,}000{,}000 cm^3}{1\ ha} \text{x} \frac{1\ Mg}{1{,}000{,}000 g}$$

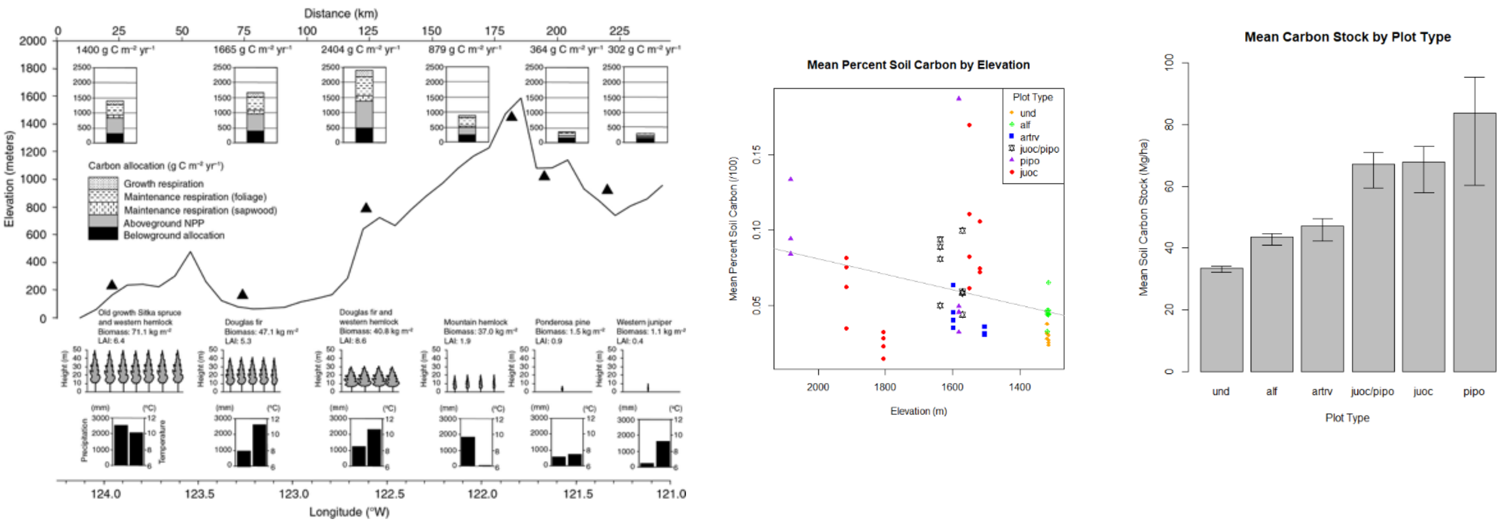


Figure 5: A East West transect of Oregon up to 225km inland, depicting forest zones and carbon allocations from Bonan et al 2002. In addition to SOC values in association with vegetation community types East of the Cascades from this study. It is important to note that allocations of Bonan's carbon allocations are displayed as a rate gC m^{-2} yr^{-1}, while this study presents carbon values as Mg C ha^{-1}.

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