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Net changes in aboveground woody carbon stock in western juniper woodlands, 1946–1998

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[1] Although regional increases in woody plant cover in semiarid ecosystems have been identified as a worldwide phenomenon affecting the global carbon budget, quantifying the impact of these vegetation shifts on C pools and fluxes is challenging. Challenges arise because woody encroachment is governed by ecological processes that occur at fine spatial resolutions (1–10 m) and, in many cases, at slow (decadal-scale) temporal rates over large areas. We therefore analyzed time series aerial photography, which exhibits both the necessary spatial precision and temporal extent, to quantify the expansion of western juniper into sagebrush steppe landscapes in southwestern Idaho. We established upper and lower bounds of aboveground woody carbon change across the landscape via two-dimensional spatial wavelet analysis, image texture analysis, and field data collection. Forty-eight 100-ha blocks across a 330,000-ha region were stratified by topography, soil characteristics, and land stewardship for analyses. Across the area we estimate aboveground woody carbon accumulation rates of 3.3 $gCm^{-2}yr^{-1}$ and 10.0 $gCm^{-2}yr^{-1}$ using the wavelet and texture method, respectively, during the time period 1946–1998. Carbon accumulation rates were significantly affected by soil properties and were highly dependent on the spatial and temporal scales of analysis. For example, at a 100-ha scale the aboveground carbon accumulation varied from -1.7 to 9.9 $gCm^{-2}yr^{-1}$, while at the 1-ha scale the range of variability increased to -11 to 22 $gCm^{-2}yr^{-1}$. These values are an order of magnitude lower than those previously suggested due to woody encroachment, highlighting the need for examining multiple spatial scales when accounting for changes in terrestrial carbon storage.

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1. Introduction

[2] Current and potential future changes in the Earth's climate system have stimulated great interest into further understanding the dynamics of the global carbon cycle. By quantifying sources and sinks of carbon across wide temporal and spatial scales, options for managing carbon sources and sinks can be quantitatively evaluated (http://www.usgcrp.gov). Although the majority of the increase in atmospheric CO_2 is considered due to fossil fuel combustion, a significant fraction of the increase arises from alterations in land cover and use [*Houghton and Goodale*, 2004]. These alterations include agricultural management, wood harvest, plantations, fire management, woody encroachment, and natural disturbances. Improved understanding of climate change requires further knowledge on global carbon pools and fluxes over the span of multiple decades, as although lag effects could exist, it is unlikely that pools and fluxes that have been relatively stable in recent history are the cause of changes in atmospheric CO_2 concentration [*Houghton et al.*, 1983]. However, detection of land cover and land use change over regional scales can be challenging because of (1) a dearth of historical records and (2) the large extent of affected land areas.

[3] Woody plant encroachment into lands previously covered by herbaceous or shrub-steppe vegetation is one land cover change that has been documented world-wide over the past 150 years [*Scholes and Archer*, 1997; *Van Auken*, 2000]. The biomass and density of native and/or exotic woody plant genera (e.g. *Quercus, Juniperus, Larrea, Prosopis, Acacia, Tamarisk* and *Yucca*) are currently increasing in many parts of the world in response to changes in environmental conditions, disturbance regimes and land use activities [*Archer et al.*, 1995]. Arid and semiarid lands cover approximately 45% of the Earth's land surface [*Bailey*, 1998] and the phenomenon of woody encroachment therefore has the potential to transform large areas of the Earth's terrestrial surface, potentially affecting the global carbon budget.

[4] At the continental scale, expansions of woody plants have been identified as contributing 0.06 $PgCyr^{-1}$ [*Houghton*, 2003], 1 Pg = 10^{15} g] to 0.12–0.13 $PgCyr^{-1}$

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[*Houghton et al.*, 1999; *Pacala*, 2001] to the U.S. carbon sink. These estimates are based on ecosystem process models and direct landscape inventories of on the ground carbon combined with reconstructions of land use change [*Pacala*, 2001]. Such estimates of carbon sinks and sources emphasize large uncertainties [*Pacala*, 2001], creating a need for regional-scale analyses across various ecosystems. Stand- to ecosystem scale carbon inventories characterizing woody plant expansions and thickening have recently been reported [*Pare and Bergeron*, 1995; *Tilman et al.*, 2000; *Archer et al.*, 2001; *Norris et al.*, 2001; *Asner et al.*, 2003; *Law et al.*, 2003; *Hicke et al.*, 2004; *Hughes et al.*, 2006]. Such studies typically quantify the aboveground carbon accumulation over a certain time period, sometimes along a successional gradient or chronosequence, via stand scale measurements and allometric equations.

[5] The study by *Asner et al.* [2003] additionally incorporated landscape scale topographic and edaphic variations and disturbance regimes and thereby allowed estimates to be extended from a local scale to a regional scale. *Asner et al.* [2003] estimated an average of 1.9 $gCm^{-2}yr^{-1}$ accumulation in aboveground woody plants from 1937 to 1999 in Texas rangelands encroached by honey mesquite (*Prosopis glandulosa*). Mesquite cover was quantified via textural analysis of aerial photography from 1937 and compared to 1999 estimates made via subpixel analyses of Landsat imagery. A local scale study at the site, however, estimated the aboveground carbon accumulation at a stand level to be approximately 35 $gCm^{-2}yr^{-1}$ over the 60-year time period [*Hughes et al.*, 2006]. Although *Asner et al.* [2003] emphasized that their regional scale estimate was likely to be low because riparian areas were excluded from the analysis, the difference between estimates made at the regional scale and the local scale is large, and underscores the need for regional scale analyses in order to discern area-wide trends. To facilitate such regional analyses, methodological developments are necessary, while to understand the contribution of regional-scale woody encroachment on continental- to global scale biogeochemical budgets, studies at additional sites are warranted.

[6] In response to these needs we utilized a new methodological approach to quantify woody plant cover dynamics over a 60-year time period in the western juniper/sagebrush steppe ecosystem of southwestern Idaho. We applied a recently established image analysis method, namely 2-D spatial wavelet analysis (SWA) [*Strand et al.*, 2005; *Falkowski et al.*, 2006; *Strand et al.*, 2006], to 1-m historic and current black and white aerial photography. In addition, to further evaluate and apply the SWA methodology at a regional scale, we compared the results from the SWA to the texture analysis remote sensing technique used by *Asner et al.* [2003] and to stand-scale field data. Our analyses were guided by a series of research questions relating to remote sensing methodology and biogeochemical cycling of carbon. The main methodological objectives of this research were twofold: (1) to estimate the minimum detectable juniper plant size in the application of SWA to 1-m panchromatic imagery and the level of canopy cover at which individual plants can no longer be detected due to tree clustering, and (2) to compare the carbon accumulation rates estimated via SWA and texture analysis. The biogeochemical science questions that we address are: (1) What is the change in aboveground woody carbon stock in the juniper zone of the Owyhee Plateau over a ~60-year time period?; (2) What are the effects of spatial scale on the estimate of changes in aboveground woody carbon stock?; and (3) What broad scale environmental and land use variables have significantly affected the establishment of western juniper across the region?

2. Methods

2.1. Study Region

[7] Our study is centered on a 400,000-ha area within the Owyhee Plateau of southwestern Idaho (116°W Long, 43°N Lat), an area characterized by western juniper woodlands (*Juniperus occidentalis* ssp. *occidentalis*) in a sagebrush (*Artemisia* spp.) steppe landscape. Western juniper encroachment into the sagebrush steppe is believed to have been occurring over the last 100–150 years [*Miller et al.*, 2005; *Bunting et al.*, 2007]. This site is representative of a much larger region across the American West, where juniper species have in the last century greatly expanded to the current condition of juniper and pinyon woodlands occupying over 30 million hectares [*West*, 1999]. Of this area, 3.6 million hectares are dominated by western juniper [*Miller et al.*, 2005], attesting to its significance as a species to observe. Although the total area affected by juniper has been estimated, actual encroachment rates are difficult to ascertain. Consequently, we are aware of only one study that quantifies long-term woody encroachment rates in western juniper woodlands [*Strand et al.*, 2006].

[8] Western juniper in the study area occurs as open savanna-like woodlands at various stages of succession, dissected by rocky canyons and riparian areas. The two dominant sagebrush species are low sagebrush (*Artemisia arbuscula*) and mountain big sagebrush (*Artemisia tridentata* ssp.*vaseyana*). The area encompasses three mountain ranges: the Silver City Range in the north, South Mountain, and Juniper Mountain in the south. The elevation where juniper occurs ranges from approximately 1400 to 2560 m with an annual average precipitation ranging from 250 mm at lower elevations to 1000 mm at the crest of the mountain range. Juniper cover becomes sparse at elevations above 2000 to 2100 m due to cold winter temperatures and harsh conditions [*Miller et al.*, 2005]. Aspen (*Populus tremuloides*), Douglas-fir (*Pseudotsuga mensiezii*) and small patches of wet meadows and mountain shrub are infrequent components in the juniper-dominated landscape. Common mountain shrub species are shiny-leaf ceanotus (*Ceanothus veluntinus*), mountain snowberry (*Symphoricarpos oreophilus*), bittercherry (*Prunus emarginata*), and chokecherry (*Prunus virginiana*). Altogether approximately 70,000 ha of the study area comprises these various vegetation types leaving 330,000 ha in juniper woodland/sagebrush steppe cover.

[9] Western juniper is a long-lived species with groups of individuals over 500 years old existing in areas of fire refugium [*Miller et al.*, 2005]. According to a landcover classification of juniper structural stages approximately 65% of the juniper cover on the Owyhee Plateau is composed of stand initiation and open young woodlands while another 16% is comprise of young multistory wood-

lands leaving only 17% in the mature juniper class [*Roth*, 2004].

[10] Soils that support sagebrush steppe and juniper woodlands on the Owyhee Plateau are dominated by xeric mollisols and alfisols [*USDA*, 1998] of igneous parent materials. Western juniper has encroached into many vegetation types; however, the encroachment rates can vary widely depending on several factors. Encroachment rates are relatively high on deeper soils supporting mountain big sagebrush, Idaho fescue (*Festuca idahoensis*) and aspen woodlands [*Young and Evans*, 1981; *Eddleman*, 1987; *Bunting et al.*, 2007], while slower rates are observed on soils dominated by low sagebrush. Low sagebrush occurs on soils that are, in general, shallow stony loams or silt loams while mountain big sagebrush occurs on deeper well-drained loams or sandy loams with a higher content of organic material. The distribution of soils are partially a result of topographic position where the stony shallow loams are located on ridges and other wind exposed areas where over the centuries wind and water erodes the top soil layer. The deeper, richer soils are found on protected side slopes, in swales and valleys where top-soil has been allowed to build via pedogenesis and site productivity with contributions from sediment in water runoff. Therefore, juniper encroachment rates can be very heterogeneous even across short distances.

[11] Euro-American settlement began when silver and other minerals were discovered in the Silver City Range in 1864. Before this era, the Piute, Shoshone and Bannock hunter and gatherer tribes inhabited the area [*Owyhee Canyonlands History*, 2003]. Silver City quickly became a booming mining town and during this era the mountains around Silver City were cleared of wood for use in the mines and for fire-wood. Silver City quickly went from ‘boom to bust’ and by 1920 it was in reality a ghost town. All three mountain ranges have a history of sheep and cattle grazing beginning as early as the 1860’s. The majority of the study region (71%) is comprised of public lands managed by the Bureau of Land Management. Remaining lands are managed by the State of Idaho (11%) or are in private ownership (18%). Today, the sparsely populated Owyhee Mountains are used primarily for summer range cattle grazing, hunting, camping, and outdoor recreation.

[12] Since the end of the mining in the 1920’s, most wood harvest in the juniper woodlands has been restricted to noncommercial wood-cutting with estimated minor impacts on the aboveground woody carbon pool. Fire atlas data from the Bureau of Land Management 1957–2002 show that approximately 10% of the study region has burned in wildfires within this time period. Overlay analysis in a GIS [*ESRI*, 2005] with a recently developed land cover map reveal that the majority of the wildfires occurred at lower elevations in the sagebrush steppe where juniper plants, if present, are small [*Roth*, 2004]. Only a few older juniper stands (1660 ha) have burned in wildfires, and thus the loss of woody carbon due to these fires can be considered negligible. Fire records were not available prior to 1957.

2.2. Analysis of Aerial Photography

[13] Remote sensing data are today available from a multitude of sensors covering a variety of spatial, spectral and temporal resolutions. For long-term analyses (>50 years) of landscape dynamics, current and historic aerial photography is available in many regions. Imagery with fine scale spatial resolution ($\sim$1 m) or subpixel analysis is required to accurately estimate levels of woody encroachment. For example, *Asner et al.* [2003] relied on subpixel analysis of Landsat 7 ETM+ imagery for estimates of current foliar cover of honey mesquite in Texas, while black and white aerial photography was only available for the historic assessment of the same area.

[14] We applied 2-D SWA to aerial photography [*Strand et al.*, 2006] to quantify the change in western juniper biomass from 1939–1946 to 1998–2004 on the Owyhee Plateau. Current aerial photography is available at 1-m pixel size [*USDA*, 2004] and historical photography at a scale of 1:27000 is available for most of the study area from the U.S. Geological Survey (http://eartheplorer.usgs.gov). For this detection of change we acquired ten black and white historical photos, each covering a 6x6 km area, from 1946 and three 6 $\times$ 5 km photos from 1939. The historical photos were georeferenced to the current imagery and resampled to 1-m pixel size, with an approximate RMSE of 10 m, in the ERDAS image processing software [*Leica Geosystems*, 2003].

[15] SWA stems from applications in medicine and astronomy and was recently adapted for landscape analysis within environmental remote sensing [*Falkowski et al.*, 2006; *Strand et al.*, 2006, 2007]. Within the field of medicine, wavelet analysis has been successful in object recognition in digital mammograms, magnetic-resonance and x-ray images [*Unser et al.*, 2003] while at the macro-scale wavelet analysis is capable of detecting galaxies, clusters and voids within astronomic images [*Slezak et al.*, 1992]. *Strand et al.* [2006] observed that a Mexican-hat 2-D wavelet can be convolved with remotely sensed imagery to quantify spatial patterns at multiple scales with the capability of automatically recording the diameter and location of individual objects, juniper plants in this specific application. The wavelet multiscale convolution was coded in MATLAB [*MathWorks*, 2004]. In summary, the input to the wavelet code is a panchromatic aerial photograph in ASCII format and the output is a list of x-y coordinates and the diameter of individual trees detected within the image. For a detailed description of the 2-D SWA technique we refer to *Strand et al.* [2006] and *Falkowski et al.* [2006].

[16] In addition, we processed the black and white imagery according to the texture method used by *Asner et al.* [2003]. Specifically, this texture analysis involved passing a 2 $\times$ 2 pixel filter across the image computing the mean, variance, and range within the kernel. *Hudak and Wessman* [1998] found that the ideal kernel size coincided with the size of the image objects to be detected (juniper plants in the present study). We repeated this moving window analysis with a kernel size of 5 m (the median crown diameter of the detectable trees based on the wavelet analysis was 4.75 m), for the historic and current aerial photography. Following the moving window analysis a supervised classification was performed on the three texture layers (mean, variance, and range) and the original image dividing the image into two classes; presence and absence of woody plants. Training data were derived from the aerial photographs.

[17] Although current imagery from various sensors (e.g. Landsat, SPOT, ASTER) is available for the study region we chose to use black and white aerial photography resampled to 1 m spatial resolution for both current (1998–2004) and historic (1939–1946) cover and carbon estimates. Choosing consistent imagery allows us to (1) assess errors that may be present utilizing these techniques because current imagery can be calibrated to field conditions, (2) minimize errors associated with utilizing two different analysis techniques in historic and current analyses [i.e., *Asner et al.*, 2003] and (3) improve spatial resolution over most freely available contemporary satellite sensors by using aerial photography.

2.3. Stand Scale Analysis: Field Validation

[18] Although past studies observed that juniper crown diameters derived via SWA exhibited a strong correlation with crown diameters derived via hand digitizing in a GIS in open canopy juniper woodlands [*Strand et al.*, 2006], comparison of this method's output to field measurements were clearly warranted. Furthermore, the collection of field-based data allows the evaluation of the methodological questions stated in the introduction.

[19] In the summer of 2005 we established twenty 60 × 60-m plots with juniper foliar cover ranging from 1.2–61.8%. Within the established plots we recorded the spatial location of each individual juniper tree larger than 1 m tall with a Trimble GeoXT global positioning system (GPS) unit with a spatial accuracy of <1 m. Plots were located such that trees were either in or out of the plot, i.e. no partial trees were present. Furthermore we recorded the maximum crown diameter, the crown diameter perpendicular to the maximum crown diameter, the height and the basal stem circumference of each juniper tree taller than 1 m within plots. Aerial photographs from 1998 and 2004 covering the field plots were then analyzed with 2-D wavelet analysis and crown diameters from the field and wavelet analysis were compared. We also compared the wavelet estimated juniper cover within the plots to field data and to cover estimated via a texture method.

2.4. Allometric Estimate of Biomass

[20] Western juniper plant biomass was estimated from allometric equations developed by *Gholz* [1980] relating stem basal circumference to stem, branch, and foliar biomass:

$$\mathrm{B} = \exp(m + n \ln \mathrm{X})$$

where B is the aboveground stem, branch or foliar biomass in kilograms, X is the stem basal circumference in centimeters and m and n are constants specific to western juniper stem, branch and foliar biomass in kilograms. *Gholz*'s [1980] allometric equations did not incorporate biomass accumulation in root biomass. Following *Law et al.* [2001] we assumed the belowground carbon storage to be 25% of aboveground woody carbon storage. We recognize that this estimate is associated with high uncertainty and therefore report above and belowground woody biomass separately. Biomass estimates were further converted to carbon by multiplication with 0.5 to account for the average carbon content of woody plant material [*Schlesinger*, 1997]. The carbon content in softwoods has been noted to vary from 0.47 to 0.55 g C per g biomass [*Lamlom and Savidge*, 2003], however no specific values are available for western juniper. Carbon contents of 52.14 ± 0.88% have been reported for eastern red cedar (*Juniperus viginiana*) and 52.47 ± 0.38% for ponderosa pine (*Pinus ponderosa*). We chose to report our findings as change in carbon, rather than change in biomass, to make our estimates comparable to those of other researchers, many of them cited in this paper, who are reporting carbon accumulation estimates in $\mathrm{gCm^{-2}yr^{-1}}$using the conversion factor of 0.5.

2.5. Landscape Scale Analysis

[21] Landscape scale assessments of change in woody carbon over time can be complicated by the lack of information of historic and current woody plant cover, the topo-edaphic heterogeneity of the land, the effects of land use history and management practices [*Asner et al.*, 2003], and variations in natural disturbance regimes. Other sources that contribute to the uncertainty are the large areal extent affected and the fine scale at which woody encroachment occurs [*Houghton and Goodale*, 2004]. Image processing of high spatial resolution, limited extent, remotely sensed data such as aerial photography for areas the size of the Owyhee Plateau is time consuming. Therefore, to avoid unrealistic efforts in image processing while still obtaining statistically sound estimates of change in woody carbon stock for the region we stratified the landscape and analyzed samples of the aerial photographs within these strata.

[22] Although attractive because of savings in effort and time, such stratification comes with its own challenges. Namely, it is important to select strata that are relevant to the ecological process observed (woody encroachment in this case), disturbance regimes, and management practices. With respect to expanding woodlands, studies have observed that structure and development are affected by land use practices [*Asner et al.*, 2003] and topography, with specific emphasis on elevation and site exposure [*Johnson and Miller*, 2005]. Accelerated successional rates have also been reported on deeper and richer soils supporting mountain big sagebrush compared to shallow soils where low sagebrush represents the vegetation potential [*Young and Evans*, 1981; *Eddleman*, 1987; *Bunting et al.*, 2007]. Following these and other prior studies [e.g., *Scott et al.*, 2002], we stratified the landscape based on elevation [*USGS*, 1999], aspect, soil type [*USDA*, 1998], and land stewardship. Another important ecological process in juniper woodlands affecting encroachment rates is the seed dispersal mechanisms, where seeds are adapted for spread by primarily berry eating birds and mammals [*Maser and Gashwiler*, 1977]. Therefore, proximity analysis in GIS (NEAR function in ArcGIS, ESRI, Redlands, CA) was used to evaluate how far current juniper plants had established from juniper plants present in historic photos to gain a better understanding of juniper encroachment rates across the landscape.

[23] The 1:27000 scale 1946 photographs each covered a 6 × 6 km area. We randomly selected 8 photographs stratified by elevation and spatial location. Within these photos we randomly selected six 100-ha (1000 × 1000 m) areas stratified by sagebrush type (representative of the soil characteristics), aspect, and land stewardship. Altogether,

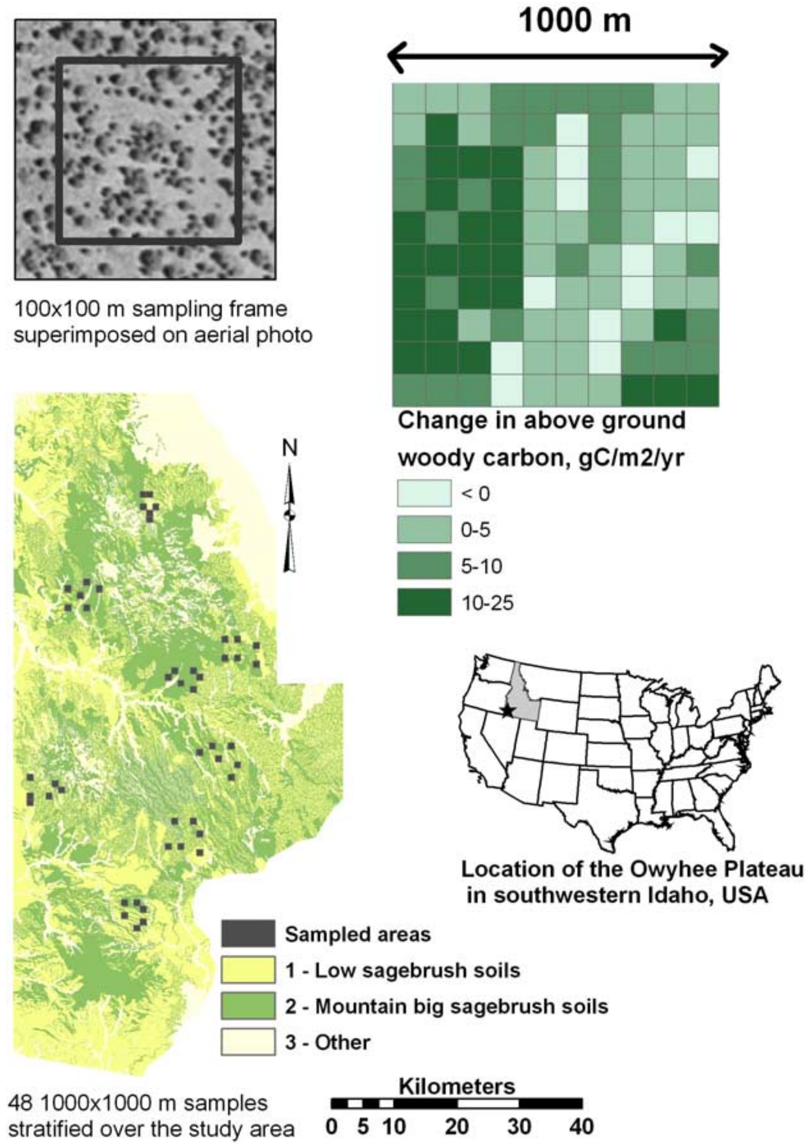

Figure 1. Sampling locations on the Owyhee Plateau. Estimates of carbon accumulation were performed at and three spatial scales: The total area of forty-eight 100-ha samples, individual 100-ha samples, and 1-ha.

net change in aboveground carbon stock was estimated for 4800 ha representative of the 330,000-ha area over the selected time period. Beyond estimating the change in biomass and carbon stock for the entire sampled area we also evaluated the change at two different spatial scales, 1000 $\times$ 1000 m and 100 $\times$ 100 m (Figure 1) using both SWA and textural analysis. To further evaluate which environmental variables affect juniper encroachment at a landscape scale, we statistically tested the effect of elevation, aspect, land stewardship, and sagebrush type (soil) with a single factor ANOVA with each of the 6 $\times$ 6 km photographs being the sampling unit (n = 8).

3. Results

3.1. Stand Scale Analysis

[24] Juniper crown diameters derived via the wavelet technique produced a strong unbiased correlation with crown diameters measured in the field (r = 0.86, n = 60), with a 19% omission error and 0% commission error (Figure 2). Plants with a crown diameter smaller than 2–3 m were not detected by the wavelet technique nor were they detectable in a GIS. However, even though these small plants comprised 55% of the number of juniper stems in field plots, they contain only 4% of the woody carbon across the sampled area (Figure 3).

[25] We further compared the SWA estimated juniper canopy cover to field data for 20 plots with juniper cover in the range 1.2–61.8 %. Cover estimates using SWA are accurate up to approximately 25% juniper cover, with increasing uncertainty in the range 25–55% due to crown clumping (r = 0.81, Figure 4). In the canopy cover range of 25–55%, SWA is biased towards underestimating foliar cover and above 55% cover the method is unreliable for analysis of aerial photography. Considering all of the 20 plots, SWA estimated 10.7% total plant cover compared to

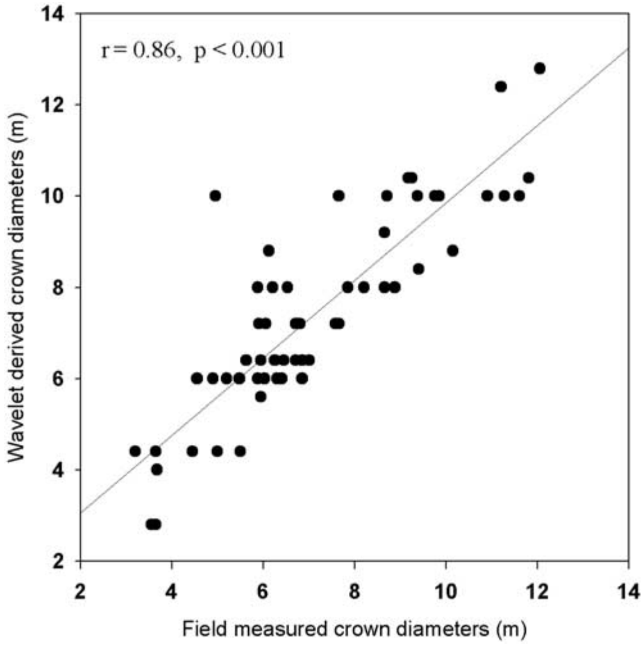

Figure 2. Comparison of wavelet derived and field measured crown diameters.

the field estimate of 13.9%, yielding a relative underestimation of 23% over the study area. We attribute approximately 50% of the error in the cover estimate to the inability to detect plants <2–3 m in size, and 50% of the error to misclassification due to clustering of trees in plots with >25% plant cover. The texture method over-estimated woody plant cover 1.8 times on average for the 20 field plots (Figure 4).

3.2. Estimate of Biomass and Carbon From Allometry

[26] We found a strong correlation ($r^2 = 0.86$, $n = 454$, $p < 0.001$) linking crown diameter to stem diameter for western juniper plants within the field data, which enabled us to directly compute aboveground biomass as a function of individual juniper plants identified using the SWA output of crown diameter (Figure 5).

3.3. Landscape Scale Analysis

[27] Within the sampled forty-eight 100-ha blocks situated across the area, the western juniper plant cover approximately doubled, from 5.3% to 10.4% total cover, during the time period 1939–1946 to 1998–2004. Juniper plant density (plants >3 m crown diameter) has increased by 128% with a higher proportion of the plant population in the smaller size classes compared to the size distribution 50–60 years ago. After image-based wavelet delineation of tree crown sizes, we computed the change in aboveground woody plant biomass and carbon stock between the two time periods using the allometric equations by *Gholz* [1980]. Overall 3.3 $gCm^{-2}yr^{-1}$ woody carbon accumulated over the sampled area above ground with an additional 0.8 $gCm^{-2}yr^{-1}$ estimated in root carbon stock. The variability in carbon accumulation rates increased with decreasing scale of analysis, as expected. At the 100-ha scale the aboveground woody carbon stock accumulation varied from -1.7 $gCm^{-2}yr^{-1}$ to 9.9 $gCm^{-2}yr^{-1}$ while at the 1-ha scale the variation was -11.0 $gCm^{-2}yr^{-1}$ to 22.1 $gCm^{-2}yr^{-1}$.

[28] Summarized over the entire sampled area the texture analysis yields an estimate of aboveground carbon accumulation of 10.0 $gCm^{-2}yr^{-1}$ compared to 3.3 $gCm^{-2}yr^{-1}$ for SWA. Figure 6 shows an example of a historic and a current photo (1000 × 1000 m) superimposed with juniper cover areas estimated via the wavelet and texture method. Using SWA over the area in Figure 6, we estimated juniper cover to be 2.5% in 1946 and 12.1% in 1998, while the texture method estimated 14.4% in 1946 and 61.1% in 1998. This figure illustrates how objects other than juniper plants and grayscale variations in the sagebrush steppe matrix can affect the texture based juniper cover estimate.

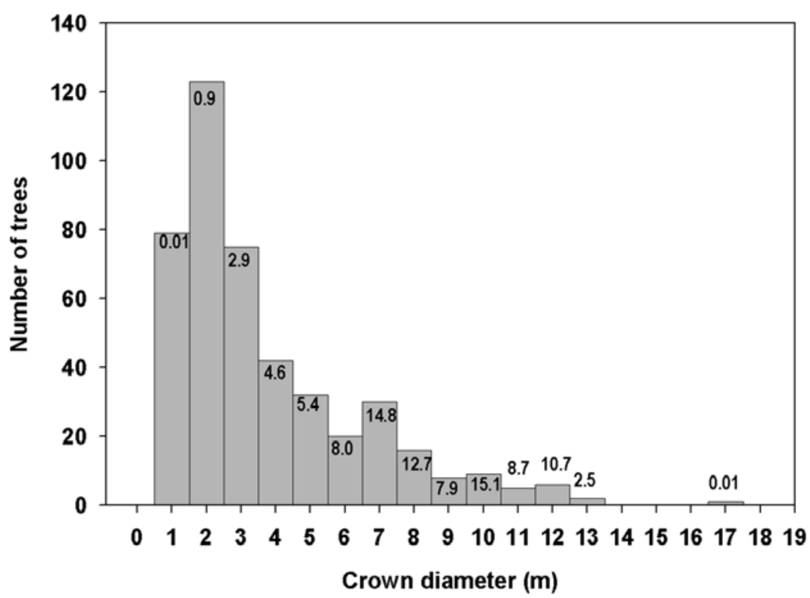

Figure 3. Size class histogram of juniper crown diameters and contribution to woody biomass. The numbers on the bars refer to the percent contribution of a size class to the total biomass in the 20 plots.

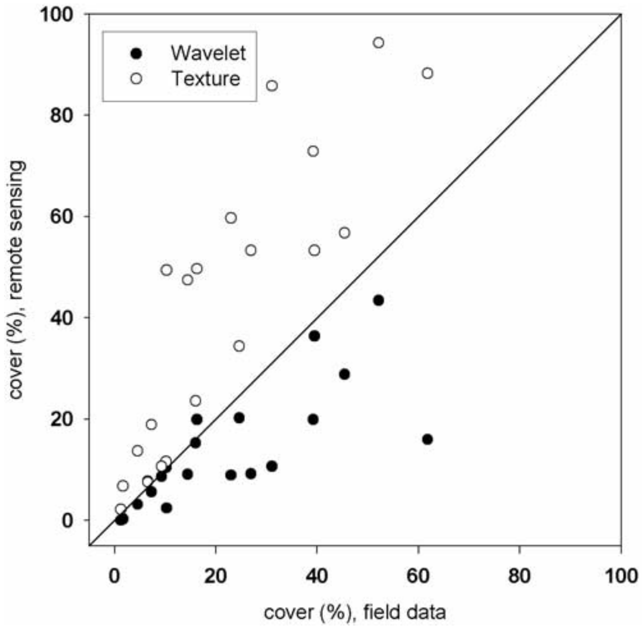

Figure 4. Juniper cover estimated via texture and 2-D wavelet remote sensing techniques compared to field estimates.

[29] Among the topographic, soil, and stewardship variables statistically tested, only soil type had a significant effect on the aboveground woody carbon accumulation rate ($p = 0.07$, $F = 3.81$, $n = 8$, Table 1) within the 1400–2560 m elevation range.

[30] The proximity analysis of current to historic juniper plants shows that western juniper plants in this area rarely establish farther away than ~50–100 m from an existing plant (Figure 7). When interpreting Figure 7 the reader should keep in mind that 44% of the plants in the current photo were present in the historic photo. The distance between a plant in the historic photo and the same plant in the current photo should be zero in the unachievable event of perfect georegistration; however, georegistration errors of up to 10 m are not uncommon. The graph does not include 1080 plants because they were located at distances 150–700 m from the closest 'historic' plant, representing approximately 1% of the recently established plants. The distance distribution in Figure 7 indeed represents the probability distribution of plant dispersal distance for western juniper.

4. Discussion

4.1. Comparison of 2-D Wavelet and Texture Analysis

[31] The 2-D wavelet analysis technique identifying plant size and spatial distribution has previously been identified as a remote sensing tool with the potential for analysis of current encroachment condition, rates of change, and ability to shed light on relationships between landscape patterns and ecological processes in arid and semiarid landscapes [*Strand et al.*, 2006, 2007]. *Strand et al.* [2006] found a strong correlation between crown diameters estimated via SWA and those derived via hand-digitizing in a geographic information system. Through this research we provide further evidence that the crown diameters and positions of individual western juniper plants accurately portray the plant distribution on the ground in open canopy woodlands. SWA is limited in detecting plants that are smaller than 2–3 times the image pixel size and when the canopy closure within the stand approaches 50%. The omission in detecting plants with a crown diameter smaller than 2–3 m in crown diameter is of minor concern considering that the biomass contained in these small but numerous plants only contribute only approximately 4% to the over all aboveground woody biomass across the landscape (Figure 3). In western juniper woodlands the commission errors of canopy cover were negligible while the omission errors amounted to 23% relative difference within the 20 field plots ranging from 1.2–61.8% in canopy cover. It must also be recognized that SWA is sensitive to the shape of the objects of interest as the

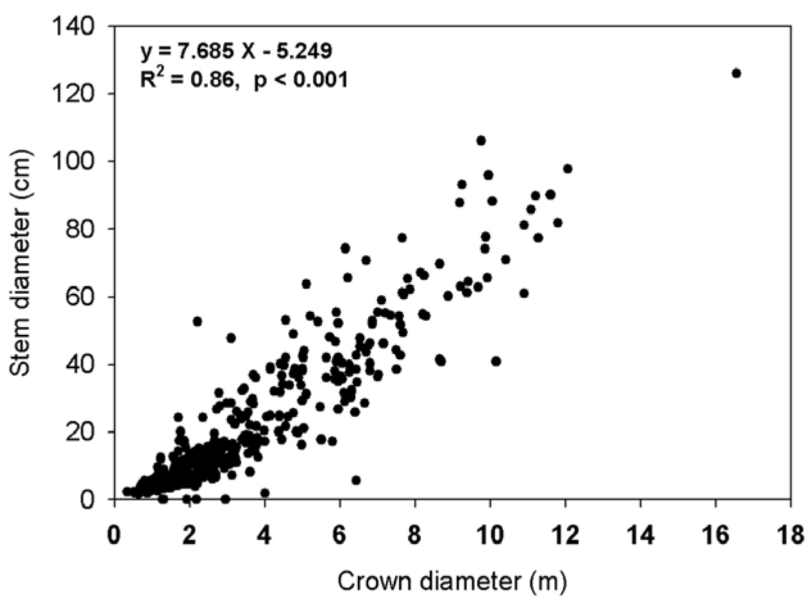

Figure 5. Stem diameter versus crown diameter, field measurement.

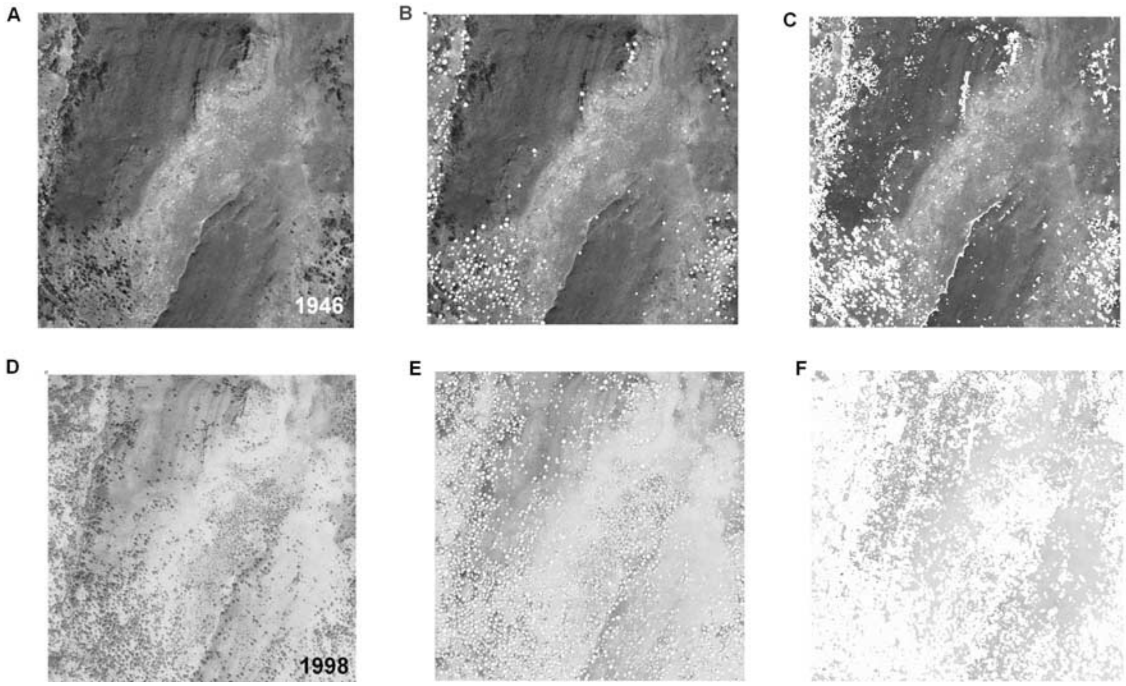

Figure 6. (top) (a) Historic photo from 1946. (b) Crown diameters estimated via wavelet analysis are superimposed in white on the photo. (c) Areas of estimated juniper cover via texture analysis are colored white. (bottom) (d) Photo from 1998. (e) Crown diameters estimated via wavelet analysis are superimposed in white on the 1998 photo. (f) Areas of estimated juniper cover via texture analysis are colored white.

wavelet function is convolved with the image objects, represented by pixels, to produce high scores where the shape and size of the two coincide. Western juniper plants are commonly symmetrical and their image brightness is well represented by the Mexican hat wavelet function dilated at different scales to detect plants of varying size. This method may not be suitable for plants or image objects with more irregular shapes.

[32] Texture analysis has previously been employed to estimate canopy cover [*Asner et al.*, 2003] and shrub density [*Hudak and Wessman*, 1998, 2001]. *Hudak and Wessman* [2001] found a weak ($r^2 = 0.2$) but significant relationship between image texture and woody plant canopy cover while the estimate by *Asner et al.* [2003] could not be validated because the analysis was performed only on historic (1937) data. Compared to field data the texture analysis technique in the case of western juniper overestimated canopy cover by a factor of 1.8 in the 20 field plots. This consistent overestimation of cover can be explained by the very nature of texture analysis in detecting edge. Both the variance and range components of the texture analysis methodology are extremely sensitive to edge, such as the edge between a juniper plant and the surrounding sagebrush steppe matrix. Edges are emphasized and in the supervised classification of the texture images these edges are classified as juniper plants, hence over-estimating the size of each plant. This phenomenon is apparent if the juniper cover map resulting from the texture analysis is laid over the aerial photograph in image processing software or GIS. Another shortcoming of the texture analysis methodology is its sensitivity to image brightness and the contrast between juniper plants and surrounding sagebrush steppe. Juniper plants with a light background (high bare ground proportion for example) are easily identified using texture analysis, however if the background is darker (dense sagebrush or a dense herbaceous understory for example) the juniper plant may not be identified by texture analysis, while the shape sensitive wavelet method accurately identifies these plants. Furthermore texture analysis does not discriminate between dark objects on a light background and light objects on a dark background. Hence such areas with fine-scale variability in

Table 1. Effect of Environmental Variables on Aboveground Woody Carbon Accumulation[a]

Variable (n = 8 blocks)	Mean ± std	Min	Max	p	F
Soil type				0.07	3.81
Supporting low sagebrush	2.0 ± 1.8	0.12	4.54		
Supporting mountain big sagebrush	3.9 ± 2.0	0.81	6.69		
Stewardship				0.29	1.33
BLM	3.3 ± 2.2	0.07	5.95		
Private	3.8 ± 2.0	1.36	6.31		
State	1.9 ± 1.7	0.41	4.66		
Aspect (north and south)				0.88	0.022
North	3.3 ± 1.9	0.76	6.00		
South	3.5 ± 2.0	0.51	6.01		
Elevation				0.65	0.56
1400–1600 m	2.9 ± 2.0	0.48	4.80		
1600–1800 m	3.1 ± 2.0	0.48	6.19		
1800–2000 m	4.1 ± 2.5	1.06	7.40		
>2000 m	2.0 ± 2.0	0.56	3.43		

[a]Units are in $gCm^{-2}yr^{-1}$. Analysis of variance with eight blocks (photographs).

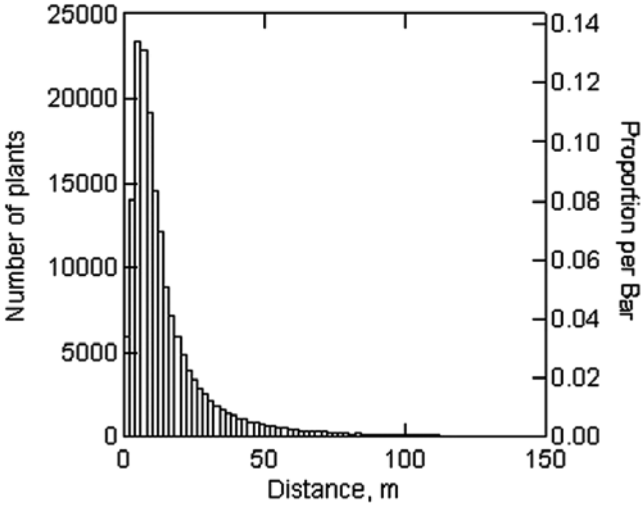

Figure 7. Histogram of the proximity of current (1998–2004) juniper plants to juniper plants present in the historic (1939–1946) photos. Plants that appear to be closer than ~10 m represents the same plant present in both photos (44% of the plants).

brightness may mistakenly be classified as areas with high juniper cover. On the other hand, texture analysis is superior to SWA in identifying clusters with juniper tree cover greater than approximately 50%, where the trees are too close together for adequate detection with the wavelet function.

[33] We conclude that SWA and texture analysis are complementary and powerful remote sensing techniques with the ability to accurately estimate western juniper cover in panchromatic current and historical aerial photography. The strengths and limitations of the two methods result in an underestimate of juniper canopy cover for the wavelet method and an overestimate for the texture method providing a respective lower and upper bound for the change in woody carbon over time. In a detection of change between two time-periods we emphasize the advantage of using the same method for cover estimates for both time-periods as each analysis technique has its own biases. However, temporal analysis of pools and fluxes with both an underestimating and overestimating method does improve the determination of the actual uncertainty within estimates and therefore we encourage the further application of such two-staged approaches to other aspects of biogeochemical cycling research.

4.2. Changes in Woody Plant Cover and Carbon Accumulation

[34] The research presented here confirms that western juniper over the past few decades has expanded into areas previously dominated by sagebrush steppe. The expansion occurs at significantly higher rates on deeper, well-drained soils capable of supporting mountain big sagebrush compared to shallow soils dominated by low sagebrush. Statistical analysis infers that within this study area there is no significant difference in management practices as they relate to juniper control between public, private and state land stewards nor does the topography represented by elevation and aspect significantly affect the juniper expansion at a 1000 × 1000-m scale. Although the effect of elevation is not statistically significant the highest rates of juniper expansion occurs at the 1800–2000 m elevation range compared to both lower and higher altitudes (Table 1). We can expect the encroachment rates to increase in areas of higher precipitation (i.e. higher elevation) while the drop in encroachment rates above 2000–2100 m can be explained by the cold winter temperatures and harsh conditions at these high altitudes [*Miller et al.*, 2005].

[35] Within the 4800-ha sampled area the wavelet and texture analyses estimate 3.3 $gCm^{-2}yr^{-1}$ and 10.0 $gCm^{-2}yr^{-1}$ accumulation rates of woody carbon over the ~60 year study period, respectively. Considering that field data shows that the wavelet method is underestimating cover and thereby biomass and carbon by ~23% and the texture method in average overestimates cover by 1.8 we can further constrain the estimate to the range 4.3–5.6 $gCm^{-2}yr^{-1}$ where the lower bound is 23% higher than the wavelet estimate and the higher bound is 1.8 times lower than the texture estimate. Approximating the root biomass to be 25% of the aboveground biomass increases this range to 5.4–7.0 $gCm^{-2}yr^{-1}$ for aboveground and belowground woody carbon.

[36] Expansion of juniper woodlands eventually leads to a loss of the shade intolerant sagebrush plants [*Bunting et al.*, 1999; *Miller et al.*, 2005]. The aboveground carbon in sagebrush steppe of the Great Basin has been estimated to 440 ± 180 gCm-2 [*Bradley et al.*, 2006]. Considering that the juniper cover averaged over the sampled 4800 ha area has increased from 5.3% to 10.4% replacing sagebrush steppe, we can calculate that over approximately 52 years (0.104–0.053)*440 = 22.4 gCm^{-2} sagebrush carbon has been lost averaging 0.43 $gCm^{-2}yr^{-1}$ or less than 10% of the woody carbon accumulation caused by the juniper expansion.

[37] The effect of woody plant encroachment on soil carbon pools and fluxes is uncertain. Honey mesquite (*Prosopis glandulosa*) encroachment in the semiarid lands of Texas has been reported to both decrease [*Jackson et al.*, 2002], increase [*Geesing et al.*, 2000; *Hibbard et al.*, 2001] and have no affect [*Hughes et al.*, 2006] on the soil carbon in the upper soil layer. While estimating the affect of woody encroachment in western juniper woodlands on soil carbon is beyond the scope of this study, the wavelet method conveniently provides a spatial point pattern that may serve as a covariate for further studies of the redistribution or accumulation of soil carbon between areas below woody plant canopies and in sagebrush steppe interspaces. It is reasonable to hypothesize that the carbon accumulation (or loss) around previously or recently established juniper plants is related to the distance from the center of the plant. Geostatistical analysis techniques could here be employed to create a soil carbon estimate over the landscape utilizing soil properties available in the county soil survey and the juniper point pattern produced by the wavelet analysis. The point pattern produced by the wavelet analysis provides opportunities to tie secondary landscape properties such as soil properties, water usage, plant competition, fire fuel distribution, seed dispersal etc. to the juniper plant distribution [e.g., *Strand et al.*, 2007].

Table 2. Effect of Spatial Scale on Carbon Accumulation Estimates in Three Locations

Vegetation Type	Scale	Aboveground Carbon Acc., g C/m^2/yr	Source
Western juniper Idaho, 1946–1998	Regional	3.3	This study
Western juniper Idaho, 1946–1998	1 ha scale	−11–22	This study
Mesquite (*Prosopis glandulosa*) Texas, 1937–1999	Regional	1.9	*Asner et al.* [2003]
Mesquite (*Prosopisglandulosa*) Texas	Plot scale (age 20–60)	35–50	*Hughes et al.* [2006]
Oak savanna Minnesota, includes soil and belowground C	Regional	16.9	*Johnston et al.* [1996]
Oak savanna Minnesota, includes soil and belowground C	Plot scale (age 0–59)	180	*Tilman et al.* [2000]

4.3. Need for Landscape Scale Analyses

[38] The spatial and temporal scale at which analyses are performed strongly influences estimates of woody carbon accumulation due to encroachment. We estimate the aboveground woody carbon accumulation using SWA at three spatial scales. Over all, 3.3 gCm^{-2}yr^{-1} aboveground woody carbon accumulated within the 4800 ha sampled area. At the 1000 × 1000 m scale the aboveground woody carbon stock accumulation varied from −1.7 to 9.9 gCm^{-2}yr^{-1} while at the 100 × 100 m scale the variation was −11.0 to 22.1 gCm^{-2}yr^{-1}. Similar results have been found in Texas where honey mesquite is expanding into grasslands. In a regional assessment *Asner et al.* [2003] estimated a 1.9 gCm^{-2}yr^{-1} aboveground woody carbon accumulation over 62 years using remote sensing technology while at a plots scale *Hughes et al.* [2006] reported aboveground woody carbon accumulations of 35–50 gCm^{-2}yr^{-1} over ~60 years (Table 2) in the same general area. Similarly, *Johnston et al.* [1996] estimated a regional carbon accumulation rate of 16.9 gCm^{-2}yr^{-1} in Minnesota oak savannas while *Tilman et al.* [2000] estimated the rate to be 180 gCm^{-2}yr^{-1} in plots within the same ecosystem (Table 2). The reason for the much higher estimates at a plot scale can likely be traced back to selection of plot areas. In a study of carbon accumulation due to woody encroachment plots are selected in areas where encroachment has occurred, neglecting the fact that encroachment has not occurred uniformly across the landscape. In agreement with *Asner et al.* [2003] we found that soil properties significantly affect the biomass production and woody encroachment rates creating heterogeneity in the potential carbon storage and accumulation rates across regions.

[39] Several researchers have explored temporal variations in biomass accumulation. *Law et al.* [2003] recorded remarkable differences in carbon accumulation rates along a chronosequence after a clearcut in ponderosa pine (*Pinus ponderosa*) where young regenerating stands were loosing carbon to the atmosphere, followed by an increase in carbon accumulation until the stands were 100–200 years of age when the carbon accumulation rates were reduced (Table 3). Similarly, *Pare and Bergeron* [1995] recorded higher carbon accumulation rates in Canadian boreal forest < 75 years of age while the rates were slower in the 75–200 year age stands. In Balsam fir (*Abies balsamea*) stands in New York, *Sprugel* [1984] observed carbon accumulation rates of 160 gCm^{-2}yr^{-1} for stands younger than 55 years while the rates were reduced to 58 gCm^{-2}yr^{-1} on average for older stands (Table 3). Assessments of carbon accumulation are commonly done at a stage in succession where the growth is likely to be the most rapid (20–100 year in age depending on the species). *Law et al.* [2003] show that both earlier and later in succession the carbon accumulation rates are lower in ponderosa pine systems and *Miller et al.* [2005] provide evidence that the most rapid biomass accumulation in western juniper stands occur between the onset of seed production (~30–50 years of age) and the time the trees reach maximum height (80–100 years depending on site conditions). The developing woodlands reach crown closure at 70–90 years on wet sites and 120–170 years on less productive sites [*Miller et al.*, 2005] at which point one could expect a decrease in biomass accumulation rates. *Canadell et al.* [2007] present evidence indicating that most biological carbon sinks will over time level off and eventually reach zero, a process referred to as sink saturation. At this time no further carbon is removed from the atmosphere and there is no net carbon accumulation in the terrestrial ecosystem.

[40] Landscapes are composed of a mosaic of vegetation types in patches of different age and structure classes, which are thought to be hierarchically nested [*Allen and Starr*, 1982; *Urban et al.*, 1987; *Turner et al.*, 1993]. Within a life zone the potential natural plant community at a given location is dictated by topo-edaphic and climatic variables while plant succession combined with natural or human induced disturbances create patches of structural variability within the potential vegetation types (PVT). Two PVT's dominate the Owyhee Mountains; the mountain big sagebrush steppe/western juniper and the low sagebrush steppe/western juniper [*Bunting et al.*, 2007]. Within PVT's disturbance regimes (especially fire) influence the western juniper patchwork of different structural stages ranging from stand initiation to mature woodlands in areas where the soil and climatic conditions allow juniper establishment and a seed source is available [*Bunting et al.*, 2007; *Miller et al.*, 2005]. The natural or historic range of variability is a useful concept, which describes the variability of ecosystem conditions and processes over time [*Morgan et al.*, 1994; *Swanson et al.*, 1994]. The range of variability can, for example, describe the landscape composition of different age and structural classes where the patch mosaic at a given time period is affected by interactions with disturbance regimes. When comparing carbon pools and fluxes within ecosystems at a landscape scale between time periods it is important to understand not only the ability for the ecosystem to accumulate (or release) carbon along successional gradients but also to estimate the expected range of variability in landscape composition for the

Table 3. Carbon Accumulation Rates in Various Ecosystems, Stand, and Regional Estimates

Plant Community	Time Since Last Stand Replacing Distrbance, years	Aboveground Carbon Accumulation, g C/m^2/yr	Source
Western juniper/Low sagebrush, Idaho	Regional estimate	2.0	This study
Western juniper/Mountain big sagebrush, Idaho	Regional estimate	3.9	This study
Mesquite (*Prosopis glandulosa*) regional estimate, Texas, 1937–1999	Regional estimate	1.9[a]	*Asner et al.* [2003]
Mesquite (*Prosopis glandulosa*) individual plots	20–60	35–50	*Hughes et al.* [2006]
Boreal forest, Canada	0–75	115	*Pare and Bergeron* [1995]
Boreal forest, Canada	75–200	~50	Estimate from *Pare and Bergeron* [1995]
Balsam fir (*Abies balsamea*) forest, New York	<55	160	*Sprugel* [1984]
Balsam fir (*Abies balsamea*) forest, New York	>55	58	*Sprugel* [1984]
Ponderosa Pine (*Pinus ponderosa*), Colorado	Stand age ~100 years	9–70	*Hicke et al.* [2004]
Ponderosa pine (*Pinus ponderosa*), central Oregon following clearcut	9–23 56–89 95–106 190–316	−124[b] 118[b] 170[b] 36[b]	*Law et al.* [2003]
Ponderosa pine (*Pinus ponderosa*), central Oregon	Regional estimate	70[b]	*Law et al.* [2003]
Ponderosa pine plantation, Patagonia, Argentina	0–20	361 roots included	*Laclau* [2003]
Aleppo pine (*Pinus halepensis*), Nevada	0–35	150[b]	*Grünzweig et al.* [2003]
Oak savanna, encroachment into old fields, Minnesota	Regional estimate	16.9[b]	*Johnston et al.* [1996]
Oak savanna, Minnesota	0–59	180[b]	*Tilman et al.* [2000]
Eastern red cedar (*Juniperus virginiana*), Kansas	35–70 years	130–230 ~ 400 inc. litter	*Norris et al.* [2001]
Tropical grassland/savanna, Venezuela	0–51	392[b]	*San Jose et al.* [1998]

[a]Shrub management was occurring within the study area.
[b]Includes belowground and soil carbon accumulation.

two time periods. Conceptually it is this change in range of variability that will explain the ecosystem carbon flux at the landscape scale. To better understand the change in carbon flux across regional extents samples must be collect randomly from a large enough area to adequately represent the structural variability within the landscape for the time periods of interest. Remote sensing technology provides avenues for mapping vegetative structure and disturbance events through time and is an invaluable asset in landscape scale assessments.

4.4. Carbon Accounting

[41] Woody encroachment has been identified as an ecological process that is contributing to the U.S. carbon sink [*Houghton and Goodale*, 2004]. Stand level analyses of carbon accumulation rates have been reported for many semiarid ecosystems [*San Jose et al.*, 1998; *Tilman et al.*, 2000; *Grünzweig et al.*, 2003; *Hicke et al.*, 2004; *Hughes et al.*, 2006, Table 3]; however, regional estimates are lacking [*Asner et al.*, 2003]. The results presented in this research and findings reported by *Asner et al.* [2003] show the importance of spatial scale in estimates of changes in carbon pools and flux. At a landscape scale, variability in soil productivity, topography, management objectives and natural disturbances creates a mosaic of vegetation structure with variable potential for carbon accumulation.

[42] Considering this temporal and spatial variability at a landscape scale, the values previously used to estimate the contribution of woody encroachment to the U.S. carbon sink may be too high. In a computation of the accumulation of woody carbon in juniper woodlands *Houghton et al.* [2000] used 65 gCm^{-2}yr^{-1} for temperate pine and juniper woodlands compared to our estimate of 5.4–7.0 gCm^{-2}yr^{-1} including aboveground and root woody carbon. We are aware that we are not including possible changes in soil carbon in our estimate and it is not clear whether they were included in the estimate by *Houghton et al.* [2000]. The seed dispersal mechanisms and distances are also clearly important, however not considered in previous estimates of the land area affected by woody encroachment. All non-forested noncultivated areas cannot be assumed to be at immediate risk of woody encroachment for two reasons. (1) All such areas do not have the topo-edaphic and climatic characteristics necessary to support woody plants. (2) All areas where topo-edaphic and climatic variable allow for woody

encroachment have not been exposed to seeds from these woody plants and may not be exposed for decades to come.

[43] In a thought experiment we could assume that our average regional scale carbon flux estimate within the bounds of the wavelet and the texture analysis estimate ($5.4-7.0\ gCm^{-2}yr^{-1}$) would apply to all pinyon-juniper woodlands in the Great Basin encompassing an area of 30 million ha [*West*, 1999] and calculate a $0.002\ PgCyr^{-1}$ contribution to the carbon sink proposed to be caused by woody encroachment ($0.06-0.13\ PgCyr^{-1}$ [*Pacala*, 2001; *Houghton*, 2003]).

[44] Another important consideration to make in the context of carbon accounting is the expected future of the recently created woodlands. Woody encroachment is in most areas undesirable from a management and conservation perspective and land stewards at all levels (private, state and federal) are likely to consider prescribed fire or other treatments to recover grassland or shrub steppe habitats by reducing the woody component. Certainly, before semiarid lands are considered to be suitable for long-term carbon storage via woody plant encroachment or plantations the consequences for future land use and land cover changes must be seriously evaluated. Grassland or steppe habitats where woody encroachment has been allowed to proceed will be extremely difficult to restore to their original state. These lands will most likely eventually burn in natural wildfires under extreme conditions or persist for centennial time periods in a low productive state, none of which will result in removal of carbon from the atmosphere.

5. Conclusions

[45] There is a need for landscape scale assessments to improve our understanding of the effect of ecological processes such as woody encroachment on the carbon fluxes between terrestrial ecosystems and the atmosphere. In such research it is important to capture the spatial and temporal variability over areas that are large enough to represent the ecosystem and its natural and managed disturbance regimes. Plot scale assessments cannot be directly extrapolated to large extents because of the natural variability in the topo-edaphic and climatic characteristics of the landscape, natural and human induced disturbance regimes, spread mechanisms of woody plant propagules, and temporal variability in carbon uptake along successional gradients. Remote sensing and geospatial analysis techniques are invaluable tools in such broad scale analyses. The 2-D wavelet analysis technique employed here using panchromatic aerial photography provides a powerful remote sensing tool because it enables changes to be detected at the scale of an individual plant that, when examined over broad extents, allows changes in aboveground woody carbon to be quantified in open canopy savannas and woodlands at multiple spatial scales. Using this method in combination with conventional image texture analysis we estimate a $3.3-10\ gCm^{-2}yr^{-1}$ woody carbon accumulated above ground in western juniper woodlands with an additional 25% estimated in root carbon stock from 1946 to 1998. This estimate is considerably lower than the $65-90\ gCm^{-2}yr^{-1}$ previously used to estimate the contribution of woody encroachment in western pine and juniper woodlands [*Houghton et al.*, 2000] to the U.S. carbon sink. Further research at regional to landscape scales across the West is warranted to further constrain the carbon cycle in western woodlands.

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