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Hydrologic response and recovery to prescribed fire and vegetation removal in a small rangeland catchment

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ABSTRACT

Prescribed fire can be used to return wild lands to their natural fire cycle, control invasive weeds, and reduce fuel loads, but there are gaps in the understanding of post-disturbance responses of vegetation and hydrology. The impact of a prescribed fire and subsequent aspen cutting on evapotranspiration (ET) and streamflow was assessed for the Upper Sheep Creek catchment, a 26-ha headwater catchment dominated by low sagebrush, mountain big sagebrush, and aspen within the Reynolds Creek Experimental Watershed. The 2007 prescribed fire consumed 100% of the mountain big sagebrush and approximately 21% of the low sagebrush. The aspen, which were mostly untouched by the fire, were cut in the fall of 2008. Post-disturbance ET and vegetation recovery were related to the loss of rooting depth. ET recovered within 2 years on the low sagebrush area with limited rooting depth, while that on the deeper-rooted mountain big sagebrush area took 4 years to recover. ET from the aspen trees, which can sprout from existing roots, recovered within 2 years. The influence of vegetation disturbance on streamflow was assessed using both empirical time trend analysis and process-based modelling. Although both approaches suggested approximately a 20% increase in streamflow during the 6 years post-disturbance, results from the empirical time trend analysis were marginally significant ($p = 0{\cdot}055$), while those from the process-based modelling were not statistically significant. Marginal streamflow response can be attributed to rapid post-disturbance recovery of the aspen where most of the streamflow originates. Published 2016. This article is a U.S. Government work and is in the public domain in the USA

KEY WORDS evapotranspiration; rangeland; root depth; streamflow; water balance

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INTRODUCTION

Wildland fire played a fundamental role in the development and maintenance of shrub-steppe plant communities in the Intermountain western USA and similar ecosystems worldwide. Prescribed fire is often used to return wildlands to their natural fire cycle, control invasive weeds, improve habitat, and reduce fuel loads (Ryan *et al*., 2013). Despite the evidence that fire is critical for maintaining the structure and function of shrub-steppe ecosystems, fire has not been restored as a fundamental process on much of the Intermountain West, resulting in widespread juniper encroachment and degradation of these ecosystems (Miller and Rose, 1999; Miller *et al*., 2005; Twidwell *et al*., 2013).

Numerous catchment studies have reported the hydrologic effects of vegetation change (Bosch and Hewlett, 1982; Brooks and Vivoni, 2008) with many giving conflicting results. Most studies in forested catchments report an increase in streamflow following removal of trees (Bosch and Hewlett, 1982; Stednick, 1996; Brown *et al*., 2005; Huang *et al*., 2006). However, a few studies, such as Guardiola-Claramonte *et al*. (2011) and Biederman *et al*. (2015), have shown decreased streamflow following tree die-off. Reasons given for this counter-intuitive observation is increase in understory cover and an increase in solar radiation reaching the understory resulting in increased sublimation and evapotranspiration (ET) (Biederman *et al*., 2014a, 2014b; Harpold *et al*., 2014).

Studies on juniper and shrubs have tended to focus on woody encroachment into grasslands. Although Wilcox *et al*. (2005, 2008) showed little to no increase in streamflow or baseflow after large increases in woody plants (mesquite and juniper) in areas with 450 to 710 mm of precipitation, Huang *et al*. (2006) found that streamflow increased 46 mm following juniper removal in a 900-mm precipitation zone. Wilcox (2002) pointed out that while numerous factors affect catchment response to shrub control, precipitation is the dominate factor, with little potential for increased streamflow from shrub control where annual precipitation is less than 500 mm. Despite the many studies in forested and juniper sites, there are very few studies of fire effects on catchment-scale water balance and streamflow in sagebrush-steppe ecosystems where juniper encroachment is increasing.

Studies on the effects of vegetation change on catchment hydrology often rely on a paired watershed approach

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(Brown *et al.*, 2005), which can prove problematic because it is often difficult to find watersheds that are truly ‘paired’ and climate variability can influence the apparent response (Zhang *et al.*, 2012; Biederman *et al.*, 2015; Burt *et al.*, 2015). Additionally, results are not always extendible beyond the area of interest because streamflow generating processes within the watershed are not directly addressed. An alternative approach is time trend analysis (Bosch and Hewlett, 1982; Huang *et al.*, 2006; Zhao *et al.*, 2010; Guardiola-Claramonte *et al.*, 2011; Biederman *et al.*, 2015), in which an empirical pre-disturbance model is developed by correlating hydrological processes with meteorological observations and post-disturbance observations are compared with predictions. If the empirical models fail to predict post-disturbance observations, one may assume there was a change in watershed relations, but little can be said about specific processes. An extension of the time trend analysis is to test and verify that a process-based model can simulate processes of interest, i.e. ET, soil water loss, streamflow, etc., for an extended period before vegetation change and then compare post-treatment observations with simulation results. The advantages of using a process-based model are that streamflow generating processes and partial area contributions can be addressed, useful information can be obtained even in arid areas when no run-off is observed, and sensitivity testing can extend knowledge gained beyond field observations.

The USDA Agricultural Research Service, Northwest Watershed Research Center (NWRC) conducted a series of studies and created a 24-year water balance for the Upper Sheep Creek catchment located in the Reynolds Creek Experimental Watershed (Chauvin *et al.*, 2011). Upper Sheep is similar to many mid-elevation to high-elevation watersheds of the semi-arid Intermountain West sagebrush steppe having ephemeral streamflow and dominated by snowmelt, ET, and relatively shallow subsurface water flow downslope to the stream. Taking advantage of the unique long-term data record and knowledge base developed for this catchment, NWRC conducted a prescribed fire within the Upper Sheep Creek catchment in September 2007 to investigate the effects of prescribed fire and vegetation removal on hydrologic response. Although the site was only in the initial stages of juniper encroachment and a prescribed fire may have been premature from a management standpoint, it afforded the opportunity to evaluate the impact of vegetation changes in sagebrush-steppe ecosystems. Thus, the objective of this paper was to assess the response and recovery of vegetation, rooting depth, ET, and streamflow to vegetation disturbance using a combination of observations, process-based model application, and time trend analysis. We hypothesized that, for a semi-arid catchment dominated by subsurface flow processes where precipitation is out of phase with transpiration demand and vegetation must rely on stored soil moisture through the growing season, ET would decrease and streamflow would increase in the years immediately following disturbance (Huxman *et al.*, 2005; Seyfried and Wilcox, 2006). Vegetation disturbance would likely cause reduced ET and thereby would reduce soil moisture deficit and increase percolation through the soil profile and subsurface flow to the stream (Flerchinger and Clark, 2003; Chauvin *et al.*, 2011). Overland flow and sediment production resulting from post-fire hydrophobicity and changes in post-disturbance snow drifting were not apparent in the watershed and are therefore not addressed.

BACKGROUND

Detailed studies of the Upper Sheep Creek Watershed were conducted by the USDA-ARS NWRC from 1984 through 1994. Numerous investigations were conducted to define the geology of the watershed (Winkelmaier, 1987; Mock, 1988; and Stevens, 1991) and to better understand the processes controlling hydrologic response (Cooley, 1988; Flerchinger *et al.*, 1992; Flerchinger *et al.*, 1993; Deng *et al.*, 1994; Flerchinger *et al.*, 1994; Unnikrishna *et al.*, 1995; Flerchinger *et al.*, 1996; Luce *et al.*, 1998). Chauvin *et al.* (2011) conducted a 24-year (1984 through 2007) water balance of the watershed that effectively characterized pre-fire hydrologic response. To account for spatial heterogeneity in hydrologic processes, the watershed was broken into three zones based on similarity in soils, vegetation, and snow accumulation (low sagebrush, mountain big sagebrush, and aspen), and a partial water budget was computed for each zone. Two approaches were taken to correlate streamflow to watershed processes. Area-weighted winter–spring precipitation and antecedent moisture conditions accounted for 83% of streamflow variability. Additionally, Chauvin *et al.* (2011) demonstrated that area-weighted percolation of water beyond the root zone simulated by the Simultaneous Heat and Water (SHAW) model correlated well with streamflow ($r^2 = 0{\cdot}85$). Subsequently, Flerchinger and Seyfried (2014) quantified ET for two vegetation types within the Upper Sheep Creek Catchment by comparing estimates from eddy covariance (EC), measured soil moisture profiles, and model simulations over a 6-years study period spanning the vegetation disturbance. They demonstrated that ET could be simulated accurately before and after vegetation treatments.

METHODS

Study site

The site is the Upper Sheep Creek study area, a semi-arid rangeland catchment located within the Reynolds Creek Experimental Watershed in the Owyhee Mountains of southwestern Idaho, USA (Marks, 2001; Slaughter *et al.*, 2001). It is a 0·26-km^2 headwater catchment with an elevation range of 1840–2036 m. A topographical map and instrument locations for the catchment are presented in Figure 1; locations within the watershed are referenced by the overlying 30-m grid. Average annual precipitation measured at D03 from 1984 through 2007 was 426 mm (Chauvin *et al*., 2011) and that measured at J10 was 572 mm, with approximately 60% occurring as snow. The site is underlain by basalt. Intermittent streamflow is generated almost entirely by subsurface flow of snowmelt, producing an average annual yield of approximately 44 mm.

Three landscape units within the catchment were identified based on similarity in vegetation, snow accumulation, and soils (inset of Figure 1; Flerchinger and Cooley, 2000; Chauvin *et al*., 2011). The southwest-facing slopes are sparsely vegetated with low sagebrush (*Artemisia arbuscula*) and some grasses. These exposed areas have little or no snow cover in the winter. Soils here are generally high in rock content (>50%) and shallow (~30 to 60 cm) and contain relatively high clay content (~25%) argillic horizons and thin (<10 cm) silt loam surface horizons. Mountain big sagebrush (*Artemisia tridentata vaseyana*), snowberry (*Symphoricarpos* spp.), and grasses/forbs covered the lower portions of the northeast-facing slopes prior to the prescribed fire. These areas typically accumulate about a metre of snow over the winter, and soils are deep loess-derived silt loam having low rock content. The upper portions of the northeast-facing slopes are predominantly vegetated by aspen (*Populus tremuloides*) and willow (*Salix* spp.) thickets. Large snow drifts (varying in depth from 1 m to typically 5 m) form annually in these areas. Soils here are virtually rock free and are very deep (>200 cm) loess-derived silt loam. These units are referred to as the low sagebrush, mountain big sagebrush, and aspen zones (inset of Figure 1) and comprise 58·9%, 26·6%, and 14·5% of the catchment, respectively. Wester juniper (*Juniperus occidentalis*) was observed to encroach into all three vegetation zones over the 20 years of observation within the catchment, with a few scattered trees growing to approximately 3 m (less than 10 within each zone) prior to treatment.

Vegetation disturbance

In order to maximize the hydrologic response and to observe vegetation recovery, the prescription for the 2007 fire within the Upper Sheep study area called for at least 50–75% of the watershed to be burned with almost total consumption of mountain big sagebrush and aspen. The boundary of the resulting prescribed fire shown in Figure 2 shows 100% consumption of the mountain big sagebrush zone and approximately 21% of the low sagebrush zone; only a few trees on the edge of the aspen thicket were scorched. Trees within the aspen thicket were therefore cut near ground level in September 2008 and left.

Field data

Leaf area index (LAI) and vegetation biomass sampling have been conducted annually at peak standing biomass

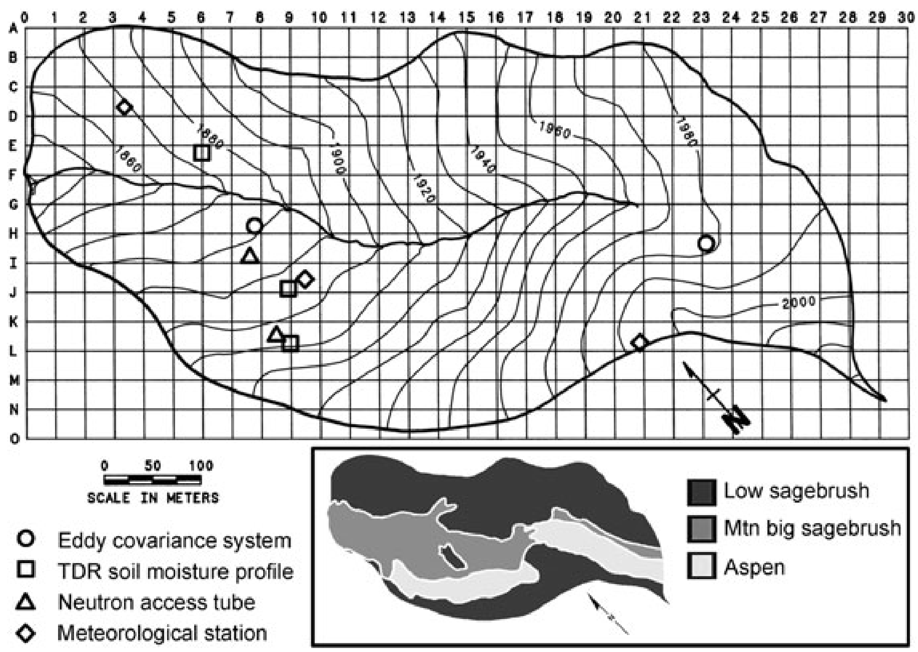

Figure 1. Upper Sheep Creek catchment orientation, elevation range, and instrumentation. Instruments locations are referenced by the overlying 30-m grid. The three landscape zones based on similarity in vegetation, snow accumulation, and soils are depicted in the inset.

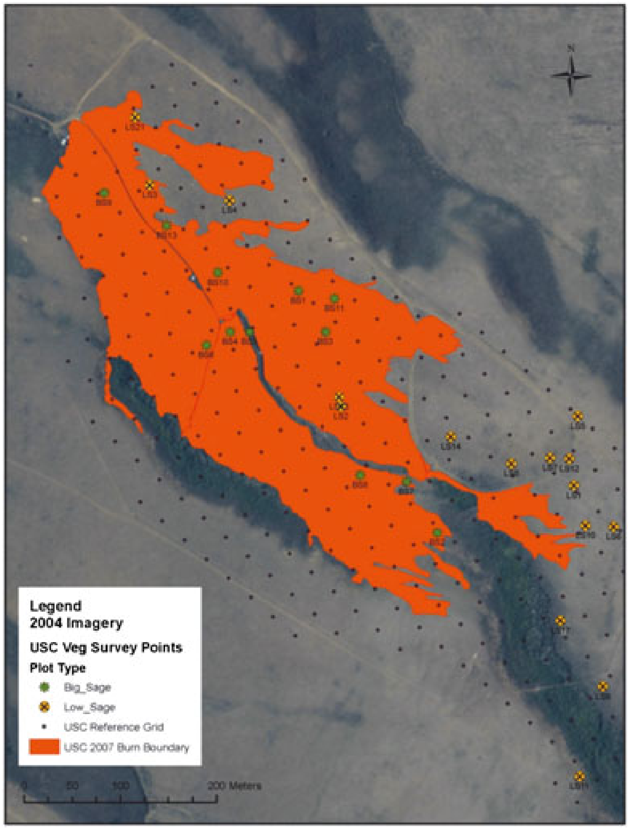

Figure 2. Burn boundary and locations of point-frame LAI measurement plots within the Upper Sheep Creek (USC) catchment.

since 2006 in the low sagebrush and mountain big sagebrush zones. LAI was measured using the point-intercept method (also referred to as point quadrat method; Clark and Seyfried, 2001) on 16 randomly selected 1-m^2 plots in the low sagebrush zone and 12 plots in the mountain big sagebrush zone (Figure 2). LAI of the aspen trees was estimated by taking the difference between measurements taken beneath the canopy at peak growth and after leaf-fall using a light interception instrument (LAI-2000, Li-Cor, Inc., Lincoln, Nebraska). LAI of the grass understory beneath the aspen was measured separately using the same light interception instrument.

Air temperature, precipitation, relative humidity, solar radiation, and wind speed measurements collected on the southwest-facing low sagebrush site and the northeast-facing mountain big sagebrush site (Figure 1) were used for SHAW model input. Hourly soil water content profiles were measured using a TDR 100 system (Campbell Scientific, Logan, Utah). Three-prong TDR rods (30·5 cm long, 0·5 cm diameter) were installed horizontally in a pit face at depths of 10, 30, 40, and 50 cm in the low sagebrush zone at grid location F6; 10, 30, 60, 90, and 120 cm in the mountain big sagebrush zone at grid location J9 (Figure 1); and 10, 30, 60, 90, 120, 150, and 180 cm in the aspen at L9. Six neutron access tubes were installed to varying depths in each of the mountain big sagebrush and aspen zones. The deepest tubes in the mountain big sagebrush and aspen, installed to approximately 225- and 270-cm depths and located at grid locations I8 and K9 (Figure 1), respectively, were used for analyses. Neutron tubes were read bi-weekly during the growing season (typically May through October) starting in July 2005. Readings were taken at 15- or 30-cm increments (Seyfried *et al*., 2001).

Evapotranspiration estimates from TDR based on a water balance of the soil column as computed by Flerchinger and Seyfried (2014) were used herein. ET for the TDR profiles was estimated directly from the change in storage within the profile (ΔS), assuming ET was zero on days with an increase in soil water storage, typically indicating precipitation for that day. This assumed that deep percolation of water beyond the measurement depth, net lateral flow of water into the soil profile, and net run-off/run-on from the soil surface are negligible. Thus, ET analyses were conducted only during the growing season after snowmelt when lateral flow and deep percolation beyond the root zone were minimal (Flerchinger and Seyfried, 2014); overland flow is seldom if ever observed in the catchment. Additionally, this approach assumes that the effective rooting depth did not extend beyond the measured profile. ET estimates commenced on calendar day 100 (April 10) for the low sagebrush zone and day 121 (May 1) for the mountain big sagebrush zone. For most years, ET estimates began on day 165 for the aspen zone but were delayed until day 175 for 2008 and day 180 for 2011 owing to late snowmelt and percolation occurring beyond the root zone of the aspen (Flerchinger and Seyfried, 2014). ET estimates were carried through day 330 (November 26) for the sagebrush zones. Flerchinger and Seyfried (2014) noted that for many of the years, ET measured by EC increased slightly in the early fall after day 290, while model simulations and TDR-measured soil water loss tapered off. Therefore, analysis of the aspen zone herein was limited to the period before leaf drop, taken as October 1 (day 274). Cumulative weekly and seasonal ET for the TDR profiles were taken as the sum of the daily ET values.

Open path EC systems as described by Flerchinger and Seyfried (2014) were installed in September 2004 above the aspen near grid location I23 and in August 2005 above the mountain big sagebrush (H8); these systems were subsequently run nearly continuously to monitor the surface energy balance. Sites were selected to optimize fetch in the predominant wind direction, which blows roughly parallel to the catchment drainage at approximately 330° from north. Maximum fetch for the aspen was approximately 150 m, and that for the sagebrush was over 200 m. Fetch for westerly winds was 65 m for the aspen, limited by the width of the aspen thicket, and 80 m for the sagebrush, limited by the distance to an aspen thicket.

Periods with a wind direction having improper fetch (i.e. coming directly over low sagebrush to the north and east) were flagged for filling of the latent heat flux data. Post-processing of the 30-min EC data included sonic temperature correction (Schotanus *et al.*, 1983), density correction (Webb *et al.*, 1980), and coordinate rotation (Kaimal and Finnigan, 1994). Gaps in the EC data, whether due to improper fetch or instrumentation problems, were filled using linear regression by correlating observations of the surrounding 14-day period to observed solar radiation. Latent and sensible heat fluxes were adjusted to force closure of the energy balance while maintaining the Bowen ratio, i.e. ratio of sensible to latent heat flux (Twine *et al.*, 2000). This was problematic when the Bowen ratio approached $-1{\cdot}0$. Therefore, whenever the magnitude of $H+LE$ was less than the error in the energy balance, H and LE were adjusted equally to compensate for the error and to force energy balance closure.

Model simulations

The SHAW model was applied to demonstrate treatment effects on rooting depth, ET, and streamflow by simulating pre-treatment and post-treatment vegetation conditions. The SHAW model has been tested and applied extensively over a range of vegetation types in semi-arid and arid environments, including previous studies within the Upper Sheep catchment (Flerchinger *et al.*, 1996, 1998, 2012; Chauvin *et al.*, 2011; Flerchinger and Seyfried, 2014). Version 3.0b of the model was used for this study, with modifications for radiation transmission and scattering within the canopy (Flerchinger and Yu, 2007; Flerchinger *et al.*, 2009b), incoming long-wave radiation (Flerchinger *et al.*, 2009a), and within-canopy turbulent transfer algorithms with correction for atmospheric stability (Flerchinger *et al.*, 2012). The SHAW model simulates a vertical, one-dimensional system composed of a vegetation canopy, snow cover (if present), plant residue, and soil profile. The surface energy balance, ET, and fluxes are simulated within a multi-species plant canopy using detailed physics of heat and water transfer through the soil–plant–atmosphere continuum, making it ideal for use in this study (e.g. Flerchinger and Cooley, 2000 and Chauvin *et al.*, 2011). A layered system is established though the model domain, with each layer represented by a node. Plant transpiration is computed by iteratively solving the following: the leaf energy balance within each layer of the multi-layer canopy, and water flux from the soil layers, through the roots, leaves, and stomatal openings to atmospheric humidity within the canopy. Plant and stomatal resistance parameters are defined for a given plant, and temporal variability in LAI, root depth, and plant height are input to the model.

Sensitivity analyses were conducted to test previous rooting depth assumptions for the low sagebrush zone made by Chauvin *et al.* (2011) and for post-fire rooting depth of the mountain big sagebrush zone made by Flerchinger and Seyfried (2014). The model was then applied and compared with pre-treatment years to demonstrate simulation accuracy of pre-treatment conditions. Simulated pre-treatment weekly ET was compared with weekly ET estimates from EC observations and TDR-measured soil water loss using two-tailed paired *t*-tests, thereby testing whether residuals between simulated and measured weekly ET were significantly different from zero. Post-treatment years were then simulated using average pre-treatment vegetation and actual post-treatment conditions, i.e. LAI and root depth; simulation results were compared with post-treatment observations of weekly ET and annual streamflow. Because the assumption is that post-treatment ET will decrease compared with pre-treatment conditions, single-tailed paired *t*-tests were used for comparing pre-treatment vegetation simulations with the post-treatment observations; thus the single-tailed *t*-tests examined whether the residuals between weekly measured ET and that simulated using pre-treatment conditions were significantly greater than zero. Differences were deemed not significant ($p>0{\cdot}10$), marginally significant ($p<0{\cdot}10$), significant ($p<0{\cdot}05$), or highly significant ($p<0{\cdot}01$).

The model was initialized each year with measured soil temperature and water profiles and used to evaluate the impact of vegetation disturbance on ET. Simulations for the aspen site were initiated shortly after snow ablation (23 May for 2004, 2005, and 2007 and 10 June for the remaining years) and continued through November as ET diminished and the seasonal snow pack developed. Simulations for the low and mountain big sagebrush sites were initiated on 10 April (day 100) and on 1 May (day 120), respectively, and also continued through November of each year. Vegetation parameters for aspen, sagebrush, and grasses/forbs used in the model were taken from previous studies in the area (Flerchinger *et al.*, 1996; Flerchinger and Cooley, 2000; and Chauvin *et al.*, 2011). Temporal variation in LAI was input to the model based on LAI measurements taken at peak standing biomass (Figure 3). Aspen leaves, grasses, and forbs were assumed to initiate growth after complete snowmelt at their respective sites; maximum LAI was assumed to occur in mid-June for the low and mountain big sagebrush site and 1 month after snow ablation at the aspen site (typically late June) based on site observations and previous studies (Flerchinger and Cooley, 2000; and Chauvin *et al.*, 2011). Burned and unburned plots existed post-fire for the low sagebrush zone, and actual LAI measurements were used each year for the respective plots. However, undisturbed plots did not exist post-disturbance for the mountain big sagebrush and aspen zones, so simulations were run using average pre-disturbance LAI for both pre-disturbance and post-disturbance years along with simulations using

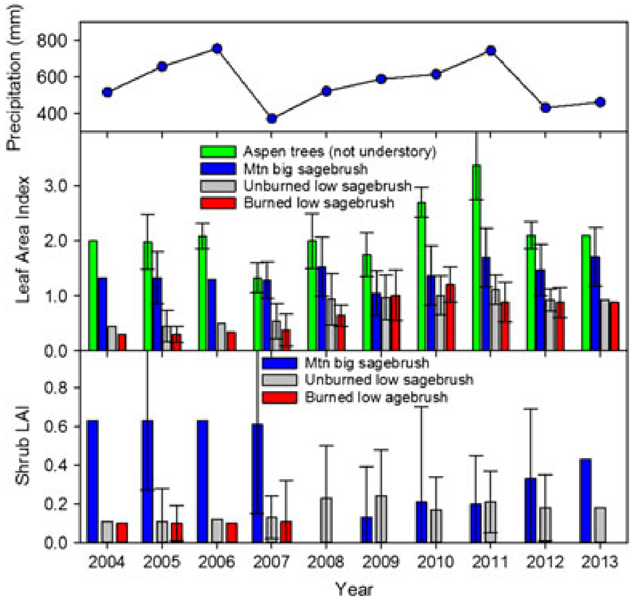

Figure 3. Water-year precipitation (defined as October through September) measured at J10 and leaf area index (LAI) for the study sites. Error bars on LAI plots indicate one standard deviation from the mean; LAI values without error bars were estimated. Sagebrush plots were burned after sampling in 2007, and aspen were cut after measurements in 2008. LAI of the grass understory beneath the aspen was approximately 1·0 throughout the study.

measured LAI for each year. Rooting depth for the aspen was taken from previous studies in the watershed (Flerchinger *et al.*, 1996) and assumed to remain unchanged after cutting as saplings quickly sprouted from the existing root stock. Root depth was set at 100 cm for the low sagebrush. Based on results from Flerchinger and Seyfried (2014), root depth of the mountain big sagebrush was set to 225 cm prior to the prescribed fire. Post-fire root depth of the recovering vegetation was estimated herein by a combination of the following: soil water extraction patterns from soil moisture profile measurements, and model sensitivity analyses comparing simulation results for a range of root depth to ET measurements from the EC system.

Continuous 31-year model runs (October 1982 through September 2013) were conducted for the analysis of vegetation disturbance on streamflow. This allowed 1 year for the model to ‘spin up’ and the assumed initial conditions to equilibrate with climatic conditions before the 1984–2013 analysis period. LAI prior to 2004 was taken as the average of the measured pre-disturbance LAI. Initiation of the growing season was adjusted each year based on date of complete snowcover ablation, but vegetation growth was not adjusted for yearly weather variations. Drift factors were applied to wintertime precipitation as described by Flerchinger and Cooley (2000) and Chauvin *et al.* (2011) to account for drifting of the snow, referred to as drift-adjusted precipitation. Snow drifts within the catchment are topographically driven, and no noticeable change in snow distribution was observed in the years following treatment as snow typically covered all vegetation during the winter, except in the immediate vicinity of the aspen EC tower. Therefore, catchment-scale changes in post-treatment sublimation due to changes in interception of precipitation and radiation (Veatch *et al.*, 2009; Biederman *et al.*, 2014a; and Harpold *et al.*, 2014) were not expected.

Streamflow analyses

Correlations between pre-disturbance streamflow and watershed processes established by Chauvin *et al.* (2011) were used as a starting point to investigate the influence of vegetation disturbance on streamflow. Chauvin *et al.* (2011) demonstrated strong correlations between streamflow and (1) simulated percolation beyond the root zone using the SHAW model and (2) a combination of soil moisture deficit at the beginning of the water year and area-weighted, winter–spring precipitation (i.e. through March for the low sagebrush, April for the mountain big sagebrush, and May for the aspen). However, the correlation developed using soil moisture deficit was problematic for the current analysis as it would implicitly include the influence of the vegetation disturbance, whereas the analysis requires a measure independent of vegetation influence. Therefore, various multivariate linear regressions were investigated using the previous year’s precipitation, temperature, vapour pressure deficit, and potential evaporation in lieu of soil moisture deficit. The regression models were calibrated to the period 1984 through 2001 and evaluated for the 6-year period immediately preceding disturbance (2002 through 2007). A one-sided *t*-test was used to determine whether the mean residuals during the evaluation period and post-disturbance period were significantly different from zero. A significant difference for the post-disturbance period would suggest that vegetation disturbance did indeed influence streamflow.

The regression between streamflow and simulated percolation beyond the root zone can be assumed to be independent of vegetation. Post-disturbance percolation was therefore simulated based on the actual vegetation and the average pre-disturbance vegetation. Streamflow was then estimated using the regression established for the pre-disturbance period. Presumably, post-disturbance residuals of estimated streamflow based on actual vegetation should not be significantly different from zero, whereas those using pre-disturbance vegetation would be significant if the disturbance had a significant influence on ET, percolation, and streamflow.

RESULTS

Vegetation response

Recovering post-fire vegetation for the low sagebrush zone was exclusively grasses and forbs; no shrubs were observed during the post-fire years as shown in Figure 3. Of the 16 low sagebrush point-intercept plots in the low sagebrush zone, four were consumed by the fire (Figure 2). Therefore, pre-fire and post-fire LAI values for the low sagebrush zone are separated into the plots that were burned and those that were not. A *t*-test comparing LAI of the burned and unburned plots in the low sagebrush zone at peak standing biomass indicated no significant difference in total LAI for any year. Not surprisingly, the year immediately following the fire did have the lowest *p*-value ($p = 0{\cdot}11$) although still not significant. The differences in shrub LAI between burned and unburned plots for the post-fire years were highly significant ($p < 0{\cdot}01$).

Data plotted in Figure 3 show an increase in post-fire LAI for the low sagebrush plots, even for the unburned plots. Unfortunately, there are only 2 years of data (2005 and 2007) prior to the fire. It is logical that LAI would be lower during the 2007 drought year; one reason that 2005 might be low is that these data were collected by different personnel than the remainder of the data. Nevertheless, the effect of the apparently low estimates of LAI during the pre-fire years is discussed in a subsequent section.

Shrubs, predominately mountain big sagebrush and snowberry, constituted about half of the pre-fire LAI in the mountain big sagebrush zone. LAI measured prior to the fire was approximately 1·3 (Figure 3), which compares well with that of 1·2 reported previously for the site (Flerchinger and Cooley, 2000); average vegetation height prior to the fire in 2007 was 0·9 m and grew to nearly 0·6 m by 2012. Unlike the low sagebrush zone, shrubs in the mountain big sagebrush zone did make some recovery but constituted only 25% of the LAI by the end of study period 6 years after the fire (Figure 3); post-fire shrub recovery was almost entirely snowberry, and little to no mountain big sagebrush was observed 6 years after the fire.

Leaf area index of the aspen trees was around 2·0 for most pre-fire years except for the drought year of 2007 (Figure 3). Aspen were quick to sprout from their existing root system after cutting and quickly recovered to their pre-cut LAI. By 2010, the second year after cutting, aspen LAI actually exceeded pre-cut levels and continued to increase until the relatively dry year of 2012. Aspen height at the EC system was approximately 4·5 m prior to cutting in 2008 and grew to 2·0 m by 2013. The grass understory was consistently around 1·0 throughout the study (data not shown).

Effect of root depth

Chauvin *et al*. (2011) assumed that the effective root depth extended beyond the shallow 50-cm soil of the low sagebrush zone to a depth of 100 cm through cracks and fissures of the rocky substrate. To assess the impact of this assumption, simulations for the low sagebrush zone were run for the study years herein with varying root depths and compared with the 50-cm TDR-measured soil water loss. Table I indicates very little difference in ET between simulations using 100- and 150-cm rooting depth, and these do not significantly differ ($p > 0{\cdot}10$) for most years from the 50-cm TDR-measured soil water loss based on two-tailed paired *t*-tests of weekly simulated ET and soil water loss. Thus, roots at this site extract relatively little water from deeper than the 50-cm profile. Unlike the mountain big sagebrush and aspen areas, snow does not accumulate through the winter in the low sagebrush zone, and there is not a big pulse of water into the soil profile in the spring. Therefore, the infiltrating water does not fully saturate and percolate beyond the soil profile except in very wet years, and the plants must rely on relatively shallow soil moisture. Indeed, the simulated wetting front reached the 100-cm depth only 2 of the 10 years (2006 and 2011). Therefore, results are relatively insensitive to the assumed rooting depth for the low sagebrush, and the 100-cm pre-fire root depth used by Chauvin *et al*. (2011) is quite reasonable. Full recovery of this relatively shallow root system is to be expected within the first year following the fire, as demonstrated for a similar site by Seyfried and Wilcox (2006).

Cumulative soil water loss by depth measured to 235 cm by soil neutron probe in the mountain big sagebrush zone plotted for each growing season through 2012 in Figure 4a suggests a loss of rooting depth after the fire. During the two pre-fire years (2006 and 2007), approximately 87% of the total water extracted came for the top 160 cm. However for 2008, the first year after the fire, soil water loss measured to 160 cm accounts for 98% of the soil water loss, and water loss to 190 cm accounts for all of it. Years 2009 and 2010 also extracted noticeably less water from deeper in the profile than the pre-fire years, but by 2011, the roots were extracting very similarly to the pre-fire year of 2006. Because 2007 was such a dry year with very little recharge of the soil profile, the vegetation relied heavily on soil water stored deeper in the profile, and Figure 4a would suggest that the roots were extracting water beyond the 235-cm measured neutron profile and the assumed rooting depth.

Figure 4a is contradictory to the analysis conducted by Flerchinger and Seyfried (2014) where post-fire ET for the mountain big sagebrush zone was simulated adequately using a rooting depth of 150 cm through 2011; however, they did not examine soil water extraction by depth. Thus, a sensitivity analysis of post-fire rooting depth on simulated ET was conducted for 2011 through 2013 as presented in Table II. Simulated ET for 2011 changes very little with increasing root depth, ranging from 502 to 509 mm for root depth spanning from 100 cm to 275 cm. Weekly simulated ET for 2011 compared well with measured ET regardless of root depth; *p*-values from the two-tailed paired *t*-test comparing simulated and measured

Table I. Cumulative evapotranspiration (ET) for each growing season (days 100 through 330) at the low sagebrush site computed by TDR-measured soil water loss within the 50-cm profile and simulated by the SHAW model using measured LAI and varying root depths.

Year	Precip. (mm)	ET from TDR (mm)	Burned with 100-cm roots		Burned with 150-cm roots		Unburned with 100-cm roots	
			Simulated ET (mm)	p(TDR)	Simulated ET (mm)	p(TDR)	Simulated ET (mm)	p(TDR)
2004	199	270	293	0·488	295	0·463	306	0·306
2005	316	357	367	0·639	370	0·580	379	0·384
2006	148	283	333	0·070	337	0·052	351	0·024
2007	154	248	228	0·362	279	0·334	215	0·107
2008	159	249	252	0·557	259	0·420	254	0·512
2009	265	374	325	0·334	327	0·353	327	0·330
2010	242	377	400	0·717	400	0·718	399	0·709
2011	179	324	398	0·055	401	0·048	431	0·013
2012	129	236	222	0·641	221	0·646	184	0·044
2013	206	297	312	0·803	359	0·292	313	0·765

Also shown is significance (p-values) of paired t-tests comparing weekly simulated ET with that estimated by TDR-soil water loss.
SHAW, Simultaneous Heat and Water; LAI, leaf area index; TDR, time domain reflectometry.

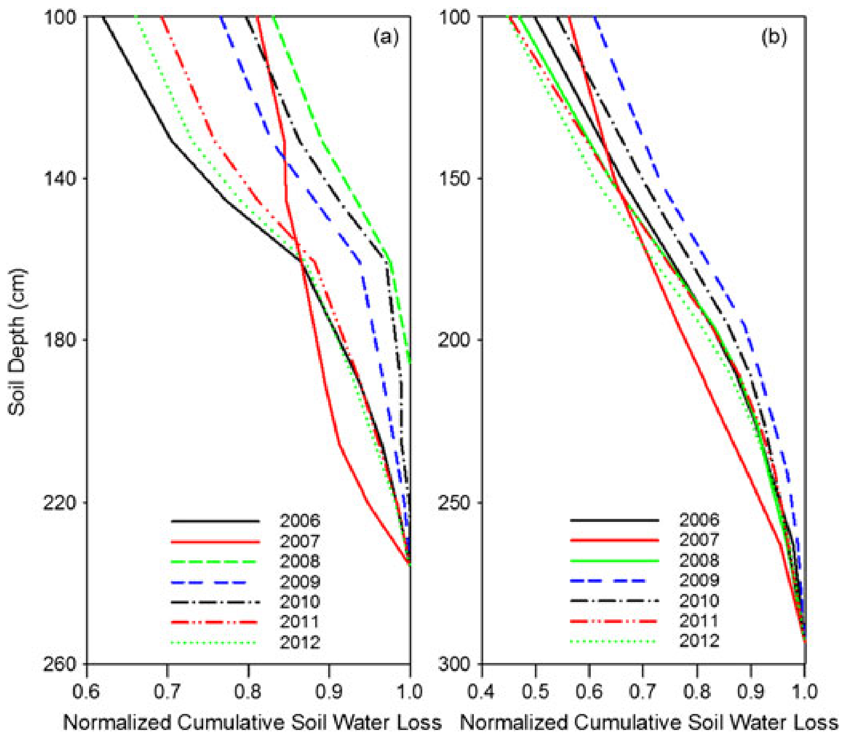

Figure 4. Soil water loss accumulated by soil depth and normalized for each growing season as measured in neutron access tubes for the (a) the mountain big sagebrush zone and (b) the aspen zone (profile is truncated above 100 cm to better focus on the deeper profile; solid lines indicate pre-treatment years).

weekly ET exceeded 0·49 for all root depths tested. Thus ET could be simulated adequately for 2011 using any reasonably selected root depth; although roots may have grown beyond 200 cm deep, there was sufficient precipitation (747 mm; Figure 3) and soil moisture that a rooting depth of 100 cm could meet ET demand. In contrast, simulated ET for 2012, which had only 432 mm of precipitation, is much more sensitive to root depth and ranges from 325 to 370 mm. Simulated ET for all root depths less than 225 cm during 2012 is significantly different from measured ET ($p < 0.05$). Thus, the evidence suggests that over time, roots had grown to at least 225 cm by 2012. Inspection of Table II indicates that a 200-cm root depth compares best with measurements for 2013, but as with 2011, none are significantly different from the measured ET. Therefore, based on Figure 4a and the

Table II. Cumulative growing season evapotranspiration (ET) for the mountain big sagebrush area measured by eddy covariance (EC) and simulated using root depth varying from 100 to 275 cm, and results of a paired *t*-test for comparing simulated and measured weekly ET values.

Year	ET from EC	Rooting depth							
		100 cm	125 cm	150 cm	175 cm	200 cm	225 cm	250 cm	275 cm
		Cumulative growing season ET (mm)							
2011	489	502	502	502	504	505	503	508	509
2012	377	325	329	331	333	342	355	363	370
2013	410	396	399	401	403	410	419	425	430
		Significance (*p*-value) of paired *t*-test							
2011	n/a	0·617	0·623	0·624	0·600	0·564	0·533	0·510	0·493
2012	n/a	0·002	0·004	0·005	0·007	0·024	0·151	0·377	0·681
2013	n/a	0·604	0·688	0·761	0·827	0·933	0·589	0·389	0·242

sensitivity analysis in Table II, post-fire root depth in the mountain big sagebrush zone was set to 150 cm for 2008 through 2010 and increased to 225 cm during 2011.

While it is clear that there was a loss of rooting depth for the mountain big sagebrush zone, Figure 4b indicates very little difference in the relative contribution of the deeper depths for the aspen between different years. Although the fraction of water extracted from the deeper depths was slightly less for the first and second years (2009 and 2010) after cutting the aspen compared with other years, the difference is not nearly as dramatic as for the mountain big sagebrush site. Both the mountain big sagebrush and aspen sites lost a significant amount of deep soil moisture during the drought year of 2007 (Figure 4).

Low sagebrush site ET

As expected, simulated ET for the low sagebrush zone plotted in Figure 5 is greater than TDR-measured soil water loss because the simulated root depth is greater than the 50-cm measured profile, even though it is not significantly different for most years (Table I). The only years that are statistically significant ($p < 0{\cdot}10$) are very wet years (2006 and 2011) when the simulated soil profile wetted well beyond 50 cm, and simulated soil water extraction by roots was beyond the measured profile. Simulated ET based on root depth, LAI, and plant composition of the burned plots is similar to that for the unburned plots for most years. Given the fact that there was no significant difference between total LAI between the burned and unburned plots, there is no real difference in simulated ET for the burned versus unburned areas within the low sagebrush zone. Indeed, cumulative ET is very insensitive to LAI for most years owing to the limited rooting depth. When the questionably low pre-fire LAI values mentioned previously were replaced with the higher values measured in 2008 (Figure 3), only a slight increase in ET (less than 3% in most years) was simulated.

Mountain big sagebrush site ET

Simulated pre-fire ET plotted in Figure 6 for the mountain big sagebrush zone changes very little whether using actual vegetation conditions or the average of all pre-fire years, and both compare favourably with ET measured using EC. During the first year after the fire (2008), ET from soil water loss compared much more favourably with EC measurements and simulations, after roots were no longer extracting water from the deeper soil depths. The SHAW simulation using pre-fire LAI and root depth had modestly more ET than that for post-fire vegetation conditions. By the second year after the fire (2009), ET estimated from soil water was less than for the other methods, suggesting that roots grew beyond the measurement depth (Flerchinger and Seyfried, 2014). Simulated ET using pre-fire vegetation remained higher than post-fire vegetation through the second year. Not much change is evident between the second and third years after the fire (2009 and 2010). Simulated ET tracks with the EC measurements reasonably well, while pre-fire vegetation still results in higher simulated ET. For the fourth and subsequent years after the fire, ET simulated using the actual vegetation approaches that simulated by pre-fire vegetation conditions. During the growing season immediately after the fire, simulated ET using pre-fire vegetation was 15% greater than simulations using measured post-fire vegetation conditions. This difference decreased to 4% for the pre-fire vegetation during the relatively wet 2011 growing season, and by 2013, simulated ET was 3% higher using post-fire conditions.

Consistent with the results plotted in Figure 6, weekly ET simulated using post-fire root depth and measured LAI was not significantly different from ET measured by EC, as shown for the post-fire paired *t*-tests in Table III. By contrast, weekly ET simulated using pre-fire vegetation was significantly different from EC measurements ($p < 0{\cdot}05$) for post-fire years 2008 and 2010, indicating that post-fire ET is significantly less as a result of the change in vegetation as

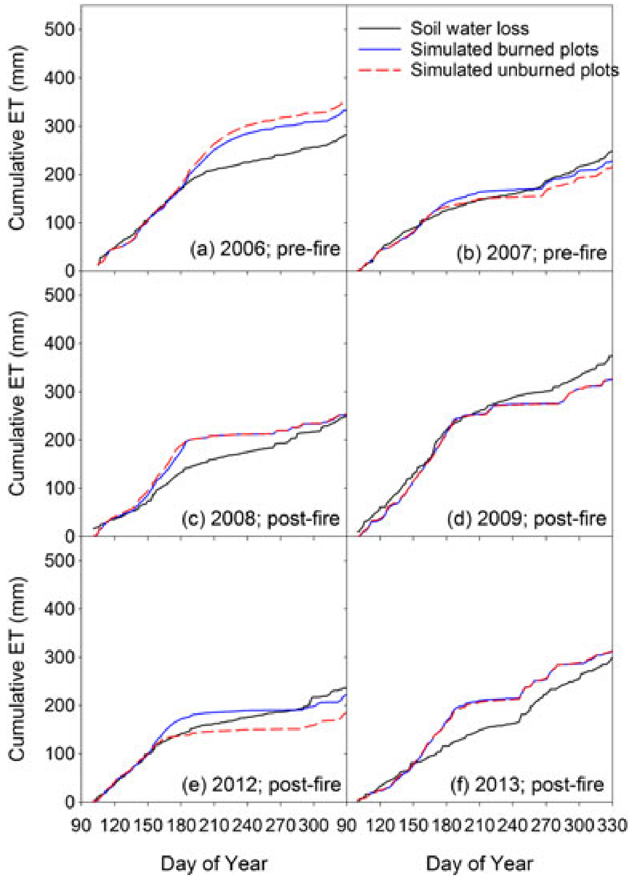

Figure 5. Cumulative daily evapotranspiration (ET) during selected years for the low sagebrush zone based on soil water loss within the 100-cm time domain reflectometry profile and model simulation based on leaf area index measured in each of the burned and unburned plots.

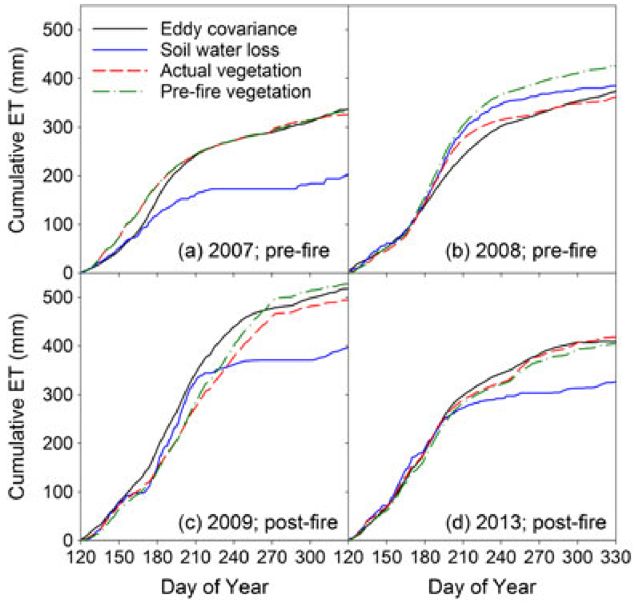

Figure 6. Cumulative daily evapotranspiration (ET) during selected years for the mountain big sagebrush zone computed from the following: eddy covariance; soil water loss within 125-cm time domain reflectometry profile; and model simulations based on average pre-fire vegetation (root depth and leaf area index) and actual vegetation. Comparisons between simulated and measured ET for additional years can be found in Flerchinger and Seyfried (2014).

indicated in Figure 7 for 2008. The paired *t*-tests indicate whether residuals are significantly different from zero, or alternatively, whether the data plot significantly above or below the 1:1 line. Simulated data using actual vegetation conditions for 2008 plot along the 1:1 line when compared with EC measurements; however, the 2008 pre-fire simulation consistently plot on or above the 1:1 line. Although cumulative ET for 2009 simulated with pre-fire vegetation was higher than the EC measurements, it was not significantly different. The reason that the 2009 pre-fire simulation is not significantly different from zero is largely related to the fact that the timing of simulated ET for either pre-fire or actual vegetation is off (Figure 6), resulting in more scatter around the 1:1 line, as shown for the 2009 pre-fire simulation in Figure 7. By 2012, cumulative ET simulated using pre-fire conditions is almost identical to that measured by EC and is actually lower during 2013 than either measured ET or that simulated using actual vegetation conditions.

Inspection of Table III indicates that soil water loss measured in the 120-cm TDR soil profile is less than simulated ET and ET measured by EC for all years except 2008, the year immediately following the fire. Soil water loss is not significantly less than simulated ET using actual vegetation for 2009 ($p = 0{\cdot}108$) due mostly to the incorrect timing of simulated ET as mentioned earlier. Based on the results in Figure 6 and Table III showing soil water extraction being significantly less than other methods of estimating ET, it is clear that vegetation extracted water from well beyond the 120-cm measured profile for all years except 1 or 2 years immediately following the fire, further confirming post-fire loss of rooting depth shown in Figure 4a discussed previously.

Aspen ET

There is very little difference between simulated ET for the years before aspen cutting whether using actual vegetation conditions or the average of all pre-cut years, as shown in Table IV. As noted by Flerchinger and Seyfried (2014), ET measured by EC during 2008 falls short of that based on TDR-measured soil water loss and simulated ET; this is due in part to limited fetch for the aspen, even under the best conditions, and the burned sagebrush surrounding the aspen area contributing to the flux measurements. EC measurements in 2006 may have experienced similar problems with fetch as well. Unfortunately, instrumentation difficulties caused large gaps in the 2007 EC data that could not reliably be filled. Height of the EC system was lowered post-cutting, thereby reducing the footprint of the observed fluxes and influence of the surrounding sagebrush.

Table III. Cumulative evapotranspiration (ET) for each growing season (days 120 through 330) at the mountain big sagebrush site: computed by eddy covariance (EC) and TDR-measured soil water loss within the 120-cm profile and simulated by the SHAW model using actual vegetation conditions and average pre-fire LAI and root depth.

Year	Precip. (mm)	EC (mm)	TDR (mm)	Actual vegetation (mm)	Pre-fire veg. (mm)	Actual vegetation		Pre-fire vegetation	
						p(EC)	*p*(TDR)	*p*(EC)	*p*(TDR)
2004	202	n/a	334	514	512	n/a	0·001	n/a	0·001
2005	298	n/a	402	473	471	0·166[a]	0·126	0·189[a]	0·132
2006	153	477	360	452	451	0·343	0·002	0·336	0·002
2007	133	337	204	326	333	0·645	<0·001	0·447	<0·001
2008	154	374	385	368	426	0·900	0·526	0·035	0·015
2009	255	517	397	498	528	0·585	0·108	0·372	0·024
2010	312	461	386	482	520	0·387	0·018	0·038	0·003
2011	159	489	426	503	524	0·607	0·086	0·163	0·031
2012	110	377	283	355	378	0·158	0·002	0·443	0·001
2013	214	410	326	419	405	0·612	0·002	0·427	0·005

Also shown is significance (*p*-values) of paired *t*-tests comparing simulated and measured weekly ET. Simulated root depth for the actual vegetation was to 150 cm for years 2008 through 2011 and 225 cm for all other years; simulated root depth for pre-fire vegetation was 225 cm.
SHAW, Simultaneous Heat and Water; LAI, leaf area index; TDR, time domain reflectometry.
[a] Denotes *p*-values for weekly ET after EC system was operational (day 229).

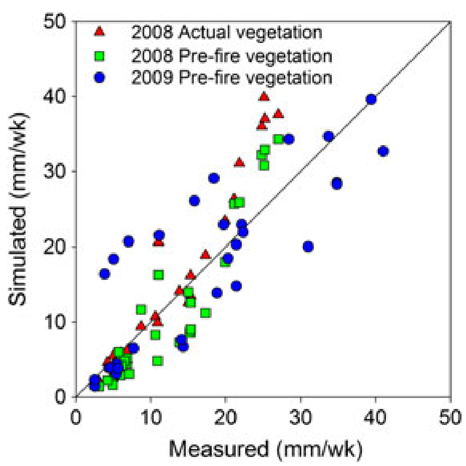

Figure 7. Measured weekly evapotranspiration (ET) for 2008 and 2009 versus simulated weekly ET using actual vegetation and average pre-fire vegetation (paired *t*-tests presented in the tables examine whether the data significantly plot above or below the 1:1 line).

The increase in ET may be a result of resurgence in grasses after fall precipitation, which was not accounted for by the model. Flerchinger and Seyfried (2014) suggested that the years that the TDR-measured soil water loss did not show an increase in water use may be attributed to the fact that the TDR-measured soil water loss was set to zero during days with precipitation. Also, the ET response may be from soil moisture shallower than the 10-cm TDR soil probe.

Simulated ET using cut aspen conditions compares well with ET measured by EC for all post-cutting years (Table IV). Simulated ET using pre-cut conditions also compares favourably with ET measured by EC, except being approximately 10% higher for 2009, which is marginally significant ($p = 0{\cdot}063$). By 2010, LAI of the new aspen shoots actually surpassed pre-cut conditions (Figure 3), and there is very little difference between simulations using actual vegetation conditions and pre-cut conditions (Table IV). Simulated ET using either actual vegetation or pre-cut conditions compares favourably with TDR-measured soil water loss for most years, with 2009 and 2011 being the exceptions; the difference between pre-cut simulated ET and TDR-measured ET for 2009 is highly significant ($p = 0{\cdot}002$). All told, the results suggest that cutting the aspen had little effect on ET except for the year immediately following cutting.

Streamflow

Stepwise regression analysis for 1984–2001 streamflow conducted using water-year precipitation measured at D03 and J10, precipitation adjusted for snow drifting, temperature, vapour pressure deficit, and potential evaporation confirmed the work of Chauvin *et al.* (2011) that drift-adjusted winter–spring precipitation is the single most important factor influencing streamflow in Upper Sheep Creek ($r^2 = 0{\cdot}68$, $p < 0{\cdot}01$ for winter precipitation > 300 mm; Figure 8a). Precipitation measured at J10 from the previous water year (lag-1 precipitation) was selected as the next most important independent variable ($p = 0{\cdot}01$), increasing the r^2 to 0·83. In the absence of lag-1 J10 precipitation, lag-1 average vapour pressure deficit and lag-1 potential evaporation were significant, but not in combination with it.

Dynamics of measured annual streamflow plotted in Figure 9 were captured by streamflow estimates from the multivariate linear regression analysis. Mean Residual streamflow (MR, i.e. model minus observed) during the 6-year evaluation period is −2·8 mm while that for the 6-year

Table IV. Cumulative evapotranspiration (ET) for each growing season (typically 165 through 274) at the aspen site: computed by eddy covariance (EC) and TDR-measured soil water loss within the 180-cm profile and simulated by the SHAW model using actual vegetation conditions and average pre-cut LAI.

Year	Precip. (mm)	EC (mm)	TDR (mm)	Actual vegetation (mm)	Average pre-cut (mm)	Actual vegetation		Pre-cut vegetation	
						p(EC)	*p*(TDR)	*p*(EC)	*p*(TDR)
2004	59	n/a	445	475	471	n/a	0·246	n/a	0·298
2005	60	n/a	438	469	465	n/a	0·996	n/a	0·928
2006	51	400	437	470	467	0·083	0·287	0·100	0·329
2007	40	n/a	262	282	284	n/a	0·646	n/a	0·794
2008	19	369	447	442	443	<0·001	0·684	<0·001	0·726
2009	70	435	395	434	478	0·951	0·062	0·063	0·002
2010	31	445	427	453	458	0·269	0·184	0·121	0·168
2011	14	437	463	423	425	0·254	0·028	0·273	0·046
2012	8	432	431	427	436	0·801	0·887	0·752	0·752
2013	147	465	429	463	483	0·806	0·431	0·430	0·100

Also shown is significance (*p*-values) of paired *t*-tests comparing simulated and measured weekly ET. Plots comparing simulated and measured ET can be found in Flerchinger and Seyfried (2014).
SHAW, Simultaneous Heat and Water; LAI, leaf area index; TDR, time domain reflectometry.

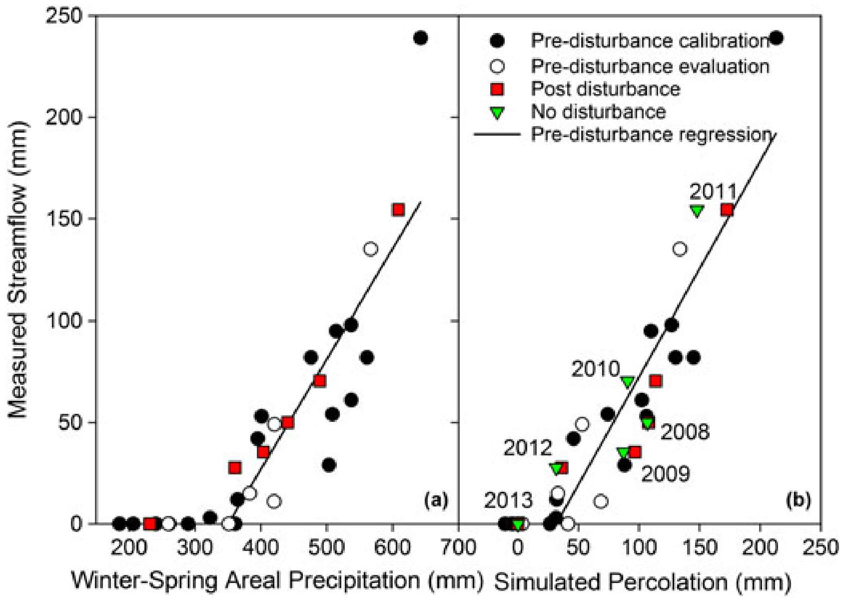

Figure 8. Measured streamflow versus (a) areal-weighted winter–spring precipitation and (b) simulated areal-weighted percolation beyond the root zone for pre-disturbance calibration years (1984–2001), pre-disturbance evaluation years (2002–2007), and post-disturbance years (2008–2013). Percolation was simulated using actual post-disturbance vegetation and average pre-disturbance vegetation (i.e. no disturbance).

post-disturbance period is −9·7 mm. Thus, average post-disturbance streamflow is approximately 20% higher than estimated. A *t*-test to discern whether the residuals are significantly different from zero yielded $p = 0{\cdot}300$ and $p = 0{\cdot}055$, respectively, indicating that vegetation disturbance is marginally significant. Inspection of Figure 9 and evaluation of the entire record indicated that the only 6-year period with residuals significantly greater than zero is 1994 through 1999 (MR = 15·4 mm, $p = 0{\cdot}047$). The reason that this period is significant might be attributed to recovery of the catchment following extended drought conditions; however, lag terms greater than 1 year were not found to be significant in the regression.

Although the aspen make up the smallest portion of the watershed, its contribution to streamflow is greatest as shown in Table V owing to the large snow drift; contribution from the low sagebrush and mountain big sagebrush zones is negligible except during wet years. Area-weighted average post-disturbance percolation beyond the root zone is approximately 12% higher using post-disturbance vegetation than using pre-disturbance vegetation (Table V). Not surprisingly, there is not much

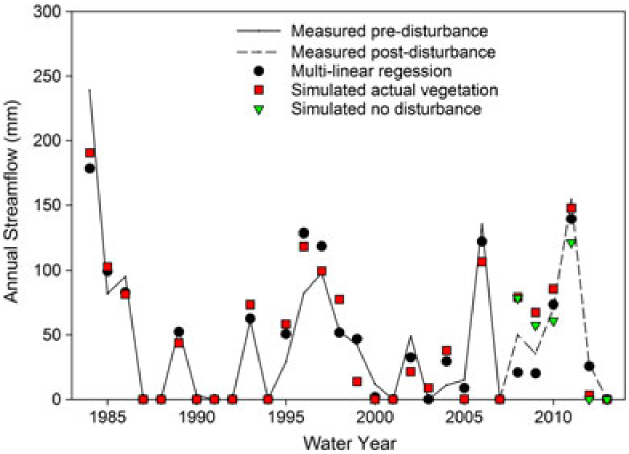

Figure 9. Measured annual streamflow along with estimated streamflow based on 1984–2001 regression analyses. Estimated streamflow are based on (1) multivariate linear regression, (2) simulated areal percolation using actual vegetation, and (3) and simulated areal percolation during post-disturbance years assuming no vegetation disturbance.

increase in simulated percolation for any of the sites immediately following the fire (2008) or aspen cutting (2009). A year of reduced ET resulting in reduced soil moisture deficit at the beginning of the water year is necessary for an increase in snowmelt percolation beyond the root zone (Flerchinger and Clark, 2003; Chauvin *et al*., 2011). Comparison of soil moisture storage on 1 October for the treated and untreated simulations 1 year post-treatment indicated that the cut aspen soil profile stored 4 cm more soil moisture, the burned mountain big sagebrush profile contained 3 cm more water, and the burned low sagebrush contained 1 cm more water. However, the simulated soil profile absorbed all of the 2009 snowmelt water in the low and mountain big sagebrush zones, so there was relatively little response in simulated percolation beyond the root zone until 2010 (Table V and Figure 8b).

Simulated areal average percolation from Table V was regressed with measured annual streamflow, as shown in Figure 8b. Mean Residuals (MR's) of the streamflow based on simulated percolation plotted in Figure 9 are −5·8 mm ($p = 0{\cdot}27$) for the evaluation period, 7·5 mm ($p = 0{\cdot}22$) for the post-disturbance using actual vegetation, and −3·3 mm ($p = 0{\cdot}38$) for post-disturbance using pre-disturbance vegetation. Thus, measured streamflow is greater than that predicted by the pre-disturbance vegetation, but not significantly so. However, the increase in estimated streamflow between post-disturbance and pre-disturbance vegetation is 10·8 mm (approximately 20%), which is consistent with the increased streamflow based on the multivariate linear time trend analysis mentioned previously. As with the time trend analysis, the only 6-year period where residuals are significantly different from zero immediately follows the extended drought period (1993–1998; MR = 17·3 mm, $p = 0{\cdot}02$).

It is possible that a more pronounced response might have been observed if the aspen and sagebrush had been burned in the fall of 2009 just prior to the two wetter years of 2010 and 2011. Simulations indicate that the predicted percolation response would have been lower for the 2010 run-off season (105 mm areal average) because of higher ET from an undisturbed 2009 growing season. Predicted percolation for 2011 (174 mm areal average) was almost identical to that based on the observed vegetation partly because the assumed root depth for the mountain big sagebrush zone was identical in both cases (150 cm). Although more percolation was predicted during 2011 for the aspen assuming a 2009 disturbance (912 mm), it covers

Table V. Measured streamflow (mm) and simulated percolation (mm) beyond the root zone based on actual vegetation and average of undisturbed vegetation conditions.

Water year	Streamflow	Simulated percolation beyond root zone: Actual vegetation				Undisturbed vegetation			
		Low sagebrush	Mountain big sagebrush	Aspen	Areal average	Low sagebrush	Mountain big sagebrush	Aspen	Areal average
2004	11	1	−6	480	69	1	−6	480	69
2005	15	1	6	215	33	1	5	218	33
2006	135	97	−5	519	134	120	−5	522	131
2007	0	−4	−6	50	3	−2	−6	50	3
2008	50	−1	−5	755	108	−1	−5	749	107
2009	35	0	0	664	97	0	−3	605	87
2010	70	0	62	669	114	0	21	585	90
2011	154	2	159	887	173	5	102	824	148
2012	28	0	−30	304	36	−1	−35	282	32
2013	0	0	−21	−36	0	0	−18	−36	0

Negative values of percolation indicate net water movement into the root zone from below.

only a small portion of the watershed, resulting in only a small increase in areal average percolation.

DISCUSSION AND CONCLUSIONS

The impact of a prescribed fire and subsequent aspen cutting on ET and streamflow was assessed for a small mountainous catchment by capitalizing on the unique long-term knowledge base developed for the Upper Sheep Creek Catchment, a 26-ha headwater catchment dominated by low sagebrush, mountain big sagebrush, and aspen within the Reynolds Creek Experimental Watershed. The 2007 prescribed fire consumed 100% of the mountain big sagebrush area and approximately 21% of the low sagebrush area. The aspen, which were mostly untouched by the fire, were cut in the fall of 2008.

Although no post-fire shrub recovery was observed 6 years after the fire in the low sagebrush zone, LAI (Figure 3) and root depth appear to have recovered in this zone within 1 year after the fire. Shrubs in the mountain big sagebrush zone were destroyed by the fire, resulting in a loss of rooting depth; no mountain big sagebrush was found within the LAI plots 6 years post-fire, but snowberry constituted approximately 25% of the peak LAI by the end of the study period. Based on observed soil water profiles (Figure 4a) and model simulations (Table II), root depth in the mountain big sagebrush zone was approximately 150 cm the first year post-fire and took 4 years to recover to its pre-fire depth of 225 cm. New vegetation in the mountain big sagebrush zone consisting of grasses and forbs replaced the deep-rooted shrubs; shrub recovery required time to re-establish roots deep within the profile, as reported for a similar site by Seyfried and Wilcox (2006). Aspen quickly sprouted from their existing roots after cutting and exceeded their pre-cut LAI levels 2 years after cutting the aspen (Figure 3).

Using the groundwork laid by Flerchinger and Seyfried (2014) and Chauvin *et al.* (2011), the effect of vegetation disturbance on ET and streamflow was assessed by comparing model simulations for pre-disturbance and post-disturbance vegetation conditions with measured soil water loss, ET, and streamflow. Hydrologic response of post-disturbance ET depended heavily on the loss of rooting depth caused by the disturbance. Because LAI and root depth in the low sagebrush zone recovered quickly after the fire, no significant difference was found in ET from the low sagebrush zone owing to the fire. ET from the mountain big sagebrush zone was estimated to be 15% less owing to the loss of vegetation and root depth the first year after the fire and decreased to 4% less by 4 years after the fire, at which time weekly ET simulated using pre-fire conditions was no longer significantly different from ET measured by EC.

Aspen trees, on the other hand, are able to sprout shoots from their roots after a disturbance, whether by fire or cutting, and can recover quickly. ET from the aspen was estimated to be approximately 10% less the first year after cutting compared with if it had not been cut. Model simulations using observed aspen conditions were shown to accurately simulate ET both before and after aspen cutting, giving confidence in the model's representation of changing site conditions. Indeed, while simulated weekly ET using observed aspen conditions compared well with TDR-measured soil water loss, simulated ET using pre-cut aspen conditions were significantly more during the first year after cutting. Thus, ET in the aspen recovered to pre-treatment conditions much quicker than in the mountain big sagebrush area. This is in stark contrast to observations made in much wetter climates (1000- to 2000-mm precipitation) by Keppeler and Ziemer (1990), who reported 5 years for water yield to recover to pre-logging conditions for coniferous forests, and Moore and Wondzell (2005) and Zhang *et al.* (2012) who reported as much as 10 to 20 years to return to pre-harvest conditions. Conversely, Biederman *et al.* (2015) showed no response in streamflow after coniferous tree mortality in multiple catchments receiving approximately 800-mm precipitation, and in some cases, streamflow was shown to increase.

The two analyses evaluating post-disturbance streamflow yielded mixed results. While time trend analysis based on multivariate linear regressions suggested marginally significant ($p = 0{\cdot}055$) increased streamflow during the six post-disturbance years, the analysis using a process-based model was not statistically significant. Both approaches however estimated approximately a 20% increase in post-disturbance streamflow compared with undisturbed conditions.

A marginal response in streamflow for this site is consistent with observations summarized by Wilcox (2002), who suggests that there is little potential for increased streamflow from shrub control where annual precipitation is less than 500 mm. The Upper Sheep Creek catchment is on the threshold of this boundary, and indeed, there is little potential for increased streamflow contribution from the sagebrush zone where the soil profile can absorb much of the precipitation pulse in most years. In such arid rangeland systems, plants tend to use all available water in the soil profile in most years (Wilcox, 2002; Huxman *et al.*, 2005; Wilcox and Thurow, 2006), and there is little potential to increase streamflow for more than a few years. While the aspen zone may have the potential to produce a more pronounced response in streamflow, its rapid post-disturbance recovery and relatively small coverage within the watershed make for a limited response.

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