Rice straw mulch for post-fire erosion control: assessing non-target effects on vegetation communities

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Abstract.   Straw mulch is commonly used for post-fire erosion control in severely burned areas but this practice can introduce non-native species, even when certified weed-free straw is used. Rice straw has recently been promoted as an alternative to wheat under the hypothesis that non-native species that are able to grow in a rice field are unlikely to establish in dry forested habitats. We investigated this hypothesis in the severely burned areas of the 2013 Rim Fire in the Sierra Nevada that were treated with rice straw post-fire. In 2014, we installed 134 plots in mulched and control areas with >95% tree mortality and re-measured a subsample in 2015. Mulched areas had significantly higher non-native forb cover, non-native graminoid cover and non-native species richness. In addition, 25 non-native species occurred exclusively in mulched areas; collectively, these responses contributed to more homogenous plant communities in mulched areas than in unmulched areas in 2015. In contrast, mulch had no effect on total plant cover, and conifer regeneration densities were generally unaffected with the exception of a slight positive effect on Douglas-fir. We recommend more stringent testing for weed-free certification and that funding for non-native species eradication be included with post-fire rehabilitation plans.

Introduction

With the increase in the occurrence, severity and extent of wildfires in the western US over the last few decades (Westerling *et al.* 2006; Miller *et al.* 2009) there has been a parallel increase in the application of post-fire emergency stabilisation treatments (Robichaud *et al.* 2014). These treatments are applied in severely burned areas where the loss of vegetation and ground cover significantly elevates the risk of post-fire soil erosion, which can result in increased flooding, water supply contamination, loss of soil, soil nutrients and seedbanks, as well as destruction of infrastructure such as roads, culverts and bridges (Robichaud *et al.* 2010*a*). Many treatments have been used to mitigate this risk, including seeding with native grasses and forbs, seeding with non-natives (sometimes sterile), and contour log felling and mulch application (agricultural wheat or rice straw, wood strands).

Straw mulch application has performed far better than other common erosion control treatments in terms of reducing soil erosion, runoff and sediment yield in surrounding streams (Groen and Woods 2008; Robichaud *et al.* 2013*b*; Robichaud *et al.* 2013*c*). This success is primarily due to the mulch acting as the ground cover that the fire removed, which can help to physically hold soil in place and reduce the force of raindrop impact. This performance advantage has led to an increase in mulch application in recent years (Robichaud *et al.* 2014). Of the commonly applied treatments, seeding and straw mulch have the greatest potential to affect native plant communities. To date, seeding has received a great deal of research attention (see review by Peppin *et al.* 2010), but the effects of straw mulch on plant communities are less well studied (but see Kruse *et al.* 2004; Dodson *et al.* 2010; Dodson and Peterson 2010; Morgan *et al.* 2014). The potential effects on native plant communities include changes in plant cover, shifts in response by life form or life cycle and the introduction of non-native species.

For example, mulch can potentially promote growth by stabilising the soil and increasing soil moisture, but it can also suppress growth by creating a physical barrier to seedling germination and establishment (Robichaud *et al.* 2010*a*; Dodson and Peterson 2010). In the northern Cascades of the US, Dodson and Peterson (2010) found that mulch cover was positively associated with higher plant cover, richness and conifer densities, except when mulch cover exceeded 70%. They also observed strong declines in plant responses when mulch depth exceeded 5 cm (Dodson and Peterson 2010). In contrast, Kruse *et al.* (2004) observed no facilitative effect on plant cover and negative effects on conifer regeneration. Morgan *et al.* (2014) also detected higher plant species richness with no clear trend in plant cover. These studies suggest that the effects of mulch on native plant communities are likely to be somewhat dependent on individual site and mulch characteristics.

The potential for non-native species introductions by mulch application is particularly problematic because severely burned landscapes are already at a higher risk of non-native species invasions. The disturbance alters dispersal dynamics and nutrient availability, reduces competition and increases light levels, which can favour non-native species that are well adapted to quickly colonise disturbed areas (Sutherland 2004; Martin *et al.* 2009). Altered disturbance regimes, such as the fire regimes in the Sierra Nevada (Parsons and DeBenedetti 1979), can further increase invasion risk (Moles *et al.* 2012). The non-native plant response to wildfire is highly variable and dependent on a complex interaction of propagule availability, propagule pressure and local site conditions (Lockwood *et al.* 2005); the long-term effects on ecosystems are equally variable (Rew and Johnson 2010). In some large, severe fires, post-fire abundance of non-native species has been low or ephemeral (Kuenzi *et al.* 2008; Wright and Tinker 2012), whereas in other cases, post-fire invasions have substantially altered post-fire landscapes and their subsequent disturbance regimes (D'Antonio and Vitousek 1992). Because of the potential for post-fire invasions to have such significant effects, post-fire management treatments that may further increase invasion risk are of concern.

In terms of invasion risk, the use of rice straw for erosion control has been promoted as a safe alternative to the more traditionally applied wheat straw based on the hypothesis that seeds from non-native species that are established in wet rice fields will not be able to establish in drier, forested habitats (Robichaud *et al.* 2000; Beyers 2004; USDA Forest Service 2015). To date, four studies have examined the effects of straw mulch on plant communities in the western US, and all detected associations between mulch treatments and non-native species (Kruse *et al.* 2004; Dodson *et al.* 2010; Dodson and Peterson 2010; Morgan *et al.* 2014). Three of these studies investigated the effects of wheat straw but only one study, in northern California, specifically investigated the effects of rice straw (Kruse *et al.* 2004). That study linked greater non-native species density with the rice mulch, but this was limited to one species that is commonly found in disturbed environments (*Cirsium vulgare* (Savi) Ten. Show (bull thistle)), which was also found in unmulched areas. We contribute to this literature by investigating prevailing hypotheses about rice mulch and non-native species, as well as the overall effects of mulch on native plant communities following the 2013 Rim Fire in the central Sierra

Nevada, California. Post-fire, the Burned Area Emergency Response (BAER) team prescribed rice straw mulch application on 1627 ha of severely burned forests, where tree mortality was >95%. To examine the effects of mulch on vegetation characteristics, we monitored field plots in rice-mulched and unmulched high-severity areas for the first 2 years post-fire, asking:

1. Did rice mulch affect non-native species richness or cover, overall plant cover, cover by life form (graminoid, forb, shrub or tree) or conifer regeneration density? Did this effect differ between the first and second year post-fire?
2. Where the rice mulch did have an effect, was the magnitude of the effect dependent on the amount of mulch cover or mulch depth that was applied in 2014?
3. What effect did rice mulch have on plant community species composition and abundance, and did this effect differ by year?

Methods

Study site

The 2013 Rim Fire burned 104 131 ha, making it the largest recorded wildfire in the Sierra Nevada, California. The fire burned through chaparral, oak woodlands, ponderosa pine, red fir and mixed conifer forests. Our sampling area was restricted to the Stanislaus National Forest portion of the fire, where rice straw mulch was applied immediately post-fire in the fall of 2013, using helicopters to drop bales on 1627 ha across 23 treatments units (Fig. 1). The rice straw was grown in California's Central Valley, and was applied at a rate of 3.4 Mg ha^{-1} (1 ton acre^{-1}), except two units where the application rate was 2.2 and 1.1 Mg ha^{-1} as part of a separate study on soil erosion. Treatment units were selected by land managers based on considerations of slope steepness, downstream values at risk and soil burn severity. Soil burn severity determinations are based on a combination of remotely sensed Burned Area Reflectance Classification (BARC) maps and ground-truthing. BARC maps are created by using the Normalised Burn Ratio, a ratio of the reflectance from Bands 4 and 7 in the Landsat satellite (Parson *et al.* 2010). Our study sites ranged in elevation from 930 m to 1930 m and included only areas that were mixed conifer forest pre-fire. These forests were dominated by ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson), sugar pine (*Pinus lambertiana* Douglas), Jeffrey pine (*Pinus jeffreyi* Grev. & Balf.), white fir (*Abies concolor* (Gordon & Glend.) Hildebr.), incense-cedar (*Calocedrus decurrens* (Torr.) Florin), coast Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco var. *menziesii*) and California black oak (*Quercus kelloggii* Newb.).

In 2014, we selected 46 sites of 0.2 km^2 and within each of them three to four plots were randomly installed, totalling 134 vegetation plots: 52 in mulched and 82 in unmulched control areas (Fig. 1). Thirty-five of these sites were co-located with a study focused on wildlife response to wildfire, which were selected based on known owl nest locations and limited to within 1 km of existing road for access efficiency. We ensured that these wildlife-specific sites were also representative of our target population, which we defined as herbaceous plant communities that occur after severe, stand-replacing fire (>95%
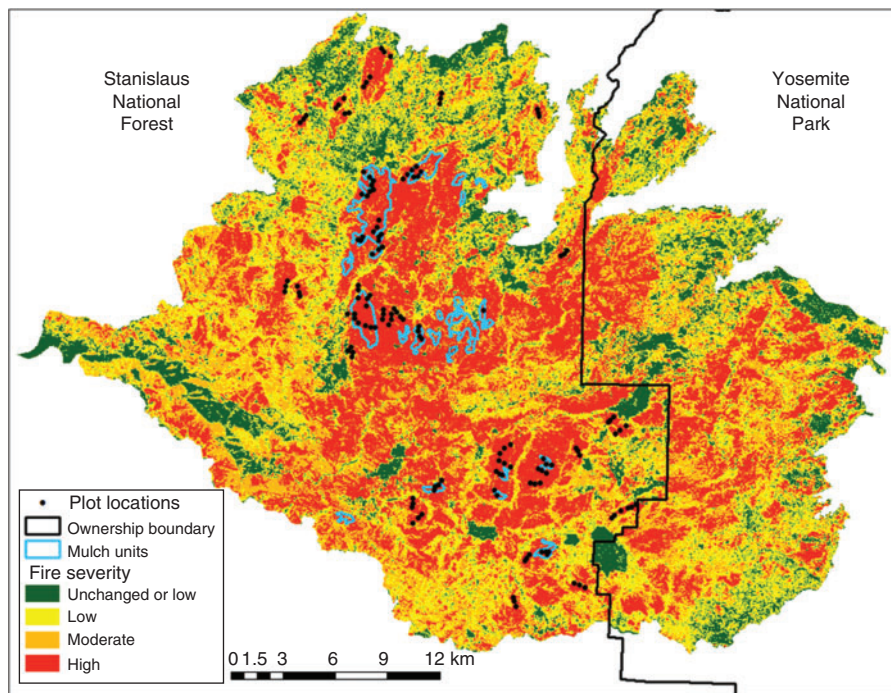
Fig. 1. Vegetation fire severity map of the Rim Fire generated with severity class thresholds described in Miller and Thode (2007). Map includes plot locations and mulch treatment polygons.

mortality) in mixed conifer forests. We confirmed that the sites co-located with owl nest locations were representative of the target population (mixed conifer that experienced >95% mortality from the fire) by assessing vegetation and burn severity variables in a GIS framework. Vegetation data were acquired from the United States Forest Service (USFS; USDA Forest Service 2014) and burn severity was classified with the Relativised Differenced Normalised Burn Ratio according to established thresholds (Miller and Thode 2007). The sites were then confirmed as representative in the field. An additional 11 sites (seven mulched and four unmulched) were randomly selected to increase sample size, using the same criteria for our target population and access by road. In 2015 we revisited 48 mulched plots (12 of these had been salvage logged after the 2014 measurements) and 49 unmulched plots (13 of these had been salvage logged after the 2014 measurements). All analyses were done on plots without salvage logging ($n_{mulched} = 36$, $n_{unmulched} = 36$), but we discuss some observed patterns based on all plots where appropriate.

Measurement

We measured understorey characteristics using two perpendicular, 32.1-m transects to create an 809 m^2 (1/5th acre) plot. Along each transect, we estimated plant canopy cover using cover classes that were adapted from Daubenmire (1959) (<1%, 1–5%, 6–25%, 26–50%, 51–75%, 76–95%, 96–100%) in 10 20 × 50-cm quadrats, for a total of 20 quadrats per plot (Daubenmire 1959). We recorded cover data by life form, species and native status, including trees shorter than 1.37 m. Where appropriate, we also estimated mulch cover and measured mulch depth at each quadrat to the nearest millimetre. To characterise total species richness,

we recorded all species occurring across the entire plot, identifying every plant to species according to the Jepson Manual of California (Baldwin *et al.* 2012). Where we did not have adequate plant parts to identify a plant to species, we identified it to the lowest possible taxonomic level, usually genus. Finally, we subsampled tree regeneration densities in a 60-m^2-circle using the USFS Region 5 post-fire regeneration protocol (Welch *et al.* 2016). We recorded distance to the nearest live tree of any species within 300 m of the plot. Where none were visible, we recorded 300 m as the minimum distance.

Statistical analyses

We used generalised linear mixed models (GLMMs) to assess the effects of rice mulch on univariate vegetation characteristics with the following distributions: total plant cover and cover by life form (β), richness (Poisson), exotic richness (negative binomial) and tree regeneration density by species (negative binomial). For all univariate variables, we created a base model that included known drivers of vegetation response: elevation, aspect and the year plots were measured (2014 or 2015). Because proximity to live trees is a known driver of conifer regeneration (Collins and Roller 2013; Welch *et al.* 2016), models for conifer seedlings additionally included distance from plot centre to the nearest live tree. Site was also included as a random effect and because some plots were measured twice, plot was also included as a random effect to account for repeated measures. We conducted initial tests with mulch as a categorical variable, year and their interaction; where there was an interaction, we subsequently tested all treatment contrasts. In addition, for variables where mulch had a significant effect, we then used conditional models on mulched plots only to test for linear

and nonlinear relationships with mulch cover, depth and their interaction as continuous variables. Nonlinear relationships were modelled by inclusion of a squared term. We selected the best conditional model based on the delta Akaike Information Criterion (dAIC), which sets the model with the minimum AIC to 0 and the remaining models are ranked by their difference in AIC. Because models with a dAIC <2 are considered only weakly distinguishable, we examined all models with a dAIC <2 for significant mulch cover or depth terms. dAIC and relevant P values for all conditional models evaluated are available as online supplementary material (see Table S1 in supplementary material available online). All univariate statistical analyses were performed using the glmmADMB package (Fournier *et al.* 2012; Skaug *et al.* 2013) in R.

To test for overall plant community composition and abundance differences by treatment we conducted PERMANOVA, a non-parametric permutation procedure, for both years in PC-ORD (McCune and Mefford 2011). Because this analysis requires equal sample sizes, we selected a subsample of the unmulched plots that most directly matched the mulched plots in terms of environmental setting characteristics (elevation and aspect) for both years. We used the Bray–Curtis dissimilarity measure with 9999 permutations, with $\alpha = 0.05$ (Anderson 2001). We then graphically examined community data in ordination space, using non-metric multidimensional scaling (NMDS). We conducted 500 runs with real data and 500 runs on randomisations from a random starting point, with an instability criterion of 0.00001. We allowed PC-ORD to choose the final number of axes based on relative stress levels and P values <0.05 (McCune and Grace 2002). The PERMANOVAs and ordinations were based on a subset of species that occurred in at least 5% of the plots to reduce the influence of very uncommon species (McCune and Grace 2002). We also used PC-ORD to identify indicator species by treatment, using the same subset of matched plots that were used for PERMANOVA tests. Species with an indicator value >25 and a P value (based on a Monte Carlo test) of <0.05 were considered indicator species for that treatment (Dufrene and Legendre 1997). Finally, we calculated Simpson's diversity index for each plot in PC-ORD and then tested for differences by treatment and year using the GLMMs in R as described above. Simpson's index represents the probability that two individuals drawn at random from a community are the same species: thus, higher values of the index represent lower diversity (greater dominance by fewer species) (McCune and Grace 2002).

Results

In the summer of 2014, mulch cover on plots in mulched units averaged 53.9% ($\pm 4.6\%$) but was highly variable across plots. Average mulch depth on plots ranged from 0.1 to 4.6 cm in 2014, and this averaged 1.1 cm across all plots (0.2 cm s.e.). Both remained highly variable but decreased substantially in 2015 (Fig. 2).

Plant cover

Mulch had no effect on total plant cover. There was significantly greater plant cover in the second growing season post-fire ($P < 0.001$; Fig. 3), and these results held when partitioned by
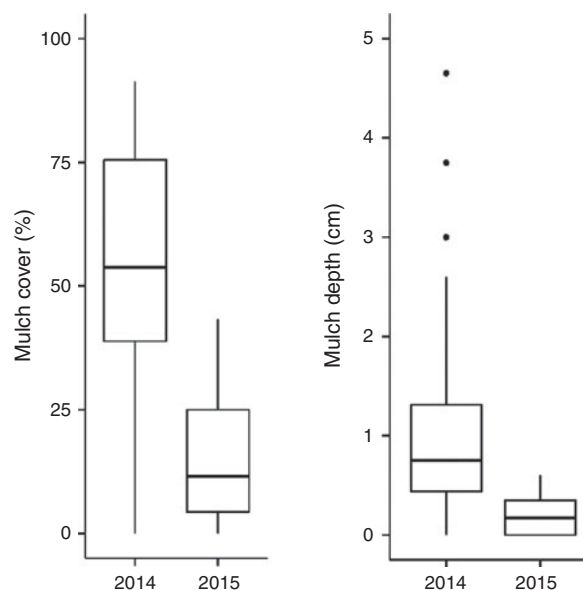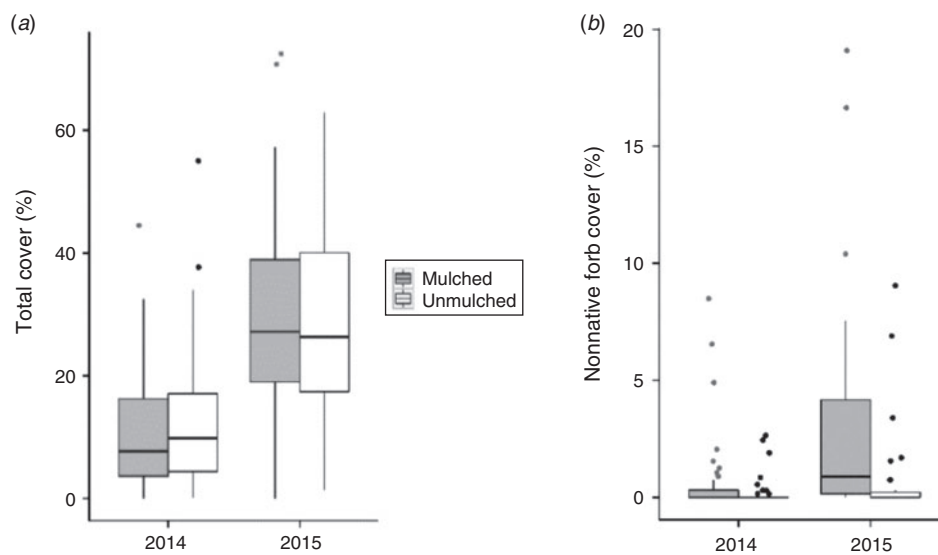

Fig. 2. Mulch cover and mulch depth on mulched plots in 2014 and 2015. The boxes define the 1st quartile (25th percentile) and 3rd quartile (75th percentile), and the line within the box defines the median. The whiskers extend to the highest value that is within 1.5 times the inter-quartile range and the dots are outliers that exceed this distance from the quartiles.

life form. There was a significant interaction for non-native forb cover ($P = 0.018$), and the treatment contrasts indicated a significant effect for mulch in both years (2014: $P = 0.029$; 2015 $P = 0.008$; Fig. 3) where the magnitude of the effect was greater in 2015. Non-native forb cover was higher in 2015 for both treatments (unmulched areas: $P < 0.001$; mulched areas: $P = 0.034$).

Non-native graminoid cover was variable but generally low ($\leq 1\%$), due in part to graminoid stature, across both treatments in both years. It was not affected by treatment ($P = 0.372$) but was significantly higher in 2015 ($P = 0.054$). At least 70% of plots detected zero cover at the quadrat level where the ocular estimates were made, but many had non-native graminoid species observed at the plot-level census (presence or absence). Given this discrepancy and the low-cover stature of many non-native graminoids, we also investigated the frequency of non-native graminoids observed. In 2014, at least one non-native graminoid was observed on 77% of mulched plots ($n = 40$) but only 40% ($n = 33$) of unmulched plots. In 2015, non-native graminoid frequency across plots was nearly equal (88.9% for mulched and 91.6% for unmulched plots).

Plant community composition

We observed a total of 331 species over the course of two post-fire growing seasons, of which 64 were non-native (Table 1). Thirty-five of the non-native species occurred across both treatments and four species occurred exclusively on unmulched plots (each occurring on ≤ 2 plots). The remaining 25 non-native species occurred exclusively on mulched plots at a wide range of frequencies (Table 1). Of the non-native species occurring on $\geq 10\%$ of the mulched plots in 2014, *Brassica nigra* (L.) W.D.J. Koch (black mustard) increased in frequency

Fig. 3. (*a*) Total plant cover and (*b*) non-native forb cover by treatment and year (note differences in the *y*-axes). The boxes define the 1st quartile (25th percentile) and 3rd quartile (75th percentile), and the line within the box defines the median. The whiskers extend to the highest value that is within 1.5 times the inter-quartile range and the dots are outliers that exceed this distance from the quartiles. For graph (*b*), the unmulched responses in particular are heavily zero-skewed, where only outliers are visible in 2014; note differences in the median and upper quartiles. Also note differences in *y*-axes.

by ~8% in 2015, *Persicaria maculosa* Gray (spotted ladys-thumb), *Erodium cicutarium* (L.) Aiton (redstem filaree), *Festuca perennis* (L.) Columbus & J.P. Sm. (Italian rye grass) and *Sorghum halepense* (L.) Pers. (Johnsongrass) increased by ≤5%; *Echinochloa crus-galli* (L.) P. Beauv. (barnyard grass) decreased by 11.1% (Table 1). When considering all plots measured, which includes those that were salvage logged between measurements, *Echinochloa crus-galli* had a lower decline (3.2%) and *Persicaria maculosa* and *Sorghum hale-pense* increased by <5% (data not shown). Two additional species with fidelity to mulched areas are not locally native, but native to California's Central Valley, where the rice mulch was grown: the grass *Leptochhloa fusca* (L.) Kunth var. *fascicularis* (Lam.) N. Snow (bearded sprangletop) and the forb *Symphio-trichium subulatum* (Michx.) G.L. Nesom var. *parviflorum* (Shinn.) s.d. Sundb (Eastern annual saltmarsh aster). *Leptochhloa fusca* decreased in frequency in 2015 whereas *Symphiotrichium subulatum* var. *parviflorum* increased ~5%. Despite being introduced from outside our study area, these species were treated as natives in all subsequent analyses, as we used the native status established for the entire California Floristic Province in the Jepson Manual (Baldwin *et al.* 2012).

In 2014, three species with complete fidelity to mulched areas were indicator species for mulched areas, one of which is not native to the US (*Echinochloa crus-galli*) and two that are native to California's Central Valley (*Symphiotrichium subula-tum* var. *parviflorum* and *Leptochhloa fusca*). The six other non-native species that were indicator species for mulched areas were also present in unmulched areas, where they occurred at lower frequencies and abundance. Native indicator species for mulched areas included two shrubs, one tree and four forbs (Table 2). Only the native forb *Dichelostemma multiflorum*

A. Heller (many-flowered brodeia) was an indicator for unmulched areas. In 2015, non-native indicators of mulched areas included *Lactuca serriola* L. (pricky lettuce) and *Sonchus asper* (L.) Hill (spiny sowthistle) – species that were also observed at lower abundances in unmulched areas. *Lactuca serriola* was observed on every mulched plot in 2015.

Species richness had a significant interaction between year and mulch treatment ($P = 0.001$). Subsequent contrast tests showed that mulch had a significant, positive effect ($P = 0.002$) in 2014 but not in 2015 ($P = 0.119$); richness was significantly higher overall in 2015 in both mulched ($P < 0.001$) and unmulched ($P < 0.001$) areas (Fig. 4). The best conditional model for richness in 2014 using dAIC included mulch cover and its squared term, but these were only marginally significant ($P = 0.073$ and 0.096 respectively). The higher richness in mulched areas was driven primarily by non-native species richness, which similarly had a significant interaction term ($P < 0.001$). Treatment contrasts indicated significantly higher non-native species richness in mulched areas in both years ($P < 0.001$) and higher non-native richness in 2015 across mulched and unmulched areas ($P < 0.001$) (Fig. 4). Like total richness, the best conditional model using dAIC for 2014 included mulch cover and its squared term, but these were more highly significant for non-native richness ($P = 0.023$ and 0.045 respectively; Table S1). This suggests a nonlinear effect; how-ever, the fitted line had exceptionally large confidence intervals, and this model was only weakly better than a model without a squared term in terms of dAIC. The best conditional model for 2015 was also nonlinear, but only mulch cover was marginally significant ($P = 0.064$). There was no difference in perennial richness between mulched and unmulched areas ($P = 0.186$), but perennial richness increased in 2015 ($P < 0.001$). Further

Table 1.   Frequency (number of plot occurrences) for all non-native species

Ratings indicate potential effect severity on wildlands from the California Invasive Plant Council (Cal-IPC) (California Invasive Plant Council 2016), and species with daggers (†) are rated noxious by the State of California. Species in bold occur only in mulched areas

Species by life form and life cycle	Cal-IPC Rating	Frequency			
		Mulched		Unmulched	
		2014	2015	2014	2015
Forbs					
Annuals					
Anthemis cotula L.		0.019	–	–	–
Brassica nigra (L.) W.D.J. Koch	Moderate	0.134	0.222	–	–
Capsella bursa-pastoris (L.) Medik.		0.058	0.027	–	0.278
Cardamine hirsuta L.		0.019	–	0.038	–
Cerastium glomeratum Thuill.		0.038	0.083	–	0.111
Chenopodium album L.		–	0.056	0.019	0.056
Dysphania botrys (L.) Mosyakin and Clemants		–	0.027	–	–
Dysphania pumilio (R.Br.) Mosyakin and Clemants		–	–	–	0.028
Erodium cicutarium (L.) Aiton	Low	0.135	0.167	–	–
Galium parisiense L.		–	0.194	–	0.056
Herniaria hirsuta L.		–	–	0.019	–
Lactuca salinga L.		0.135	0.278	0.038	0.139
Lactuca serriola L.		0.692	1.000	0.327	0.917
Logfia gallica (L.) Coss. and Germ.		–	0.028	–	0.111
Persicaria maculosa Gray		0.173	0.194	–	–
Polygonum aviculare L.		0.288	0.472	0.019	0.194
Pseudognaphalium luteoalbum (L.) Hillard and B.L. Burtt		0.135	0.222	0.038	0.250
Scleranthus annuus L. ssp. annuus		0.096	0.134	–	–
Senecio vulgaris L.		0.019	0.167		0.361
Sonchus asper (L.) Hill ssp. *asper*		0.519	0.889	0.038	0.528
Solanum nigrum L.		0.323	0.639	–	0.028
Spergularia bocconi (Scheele) Graebn.		0.038	0.027	0.038	–
Spergularia rubra (L.) J.S. Presl and C. Presl		0.017	0.139	–	–
Torilis arvensis (Huds.) Link	Moderate	0.077	0.361	0.058	0.028
Trifolium dubium Sibth.		0.019	0.278	–	0.361
Annuals or biennials					
Silene gallica L.		0.019	0.056	–	–
Sisymbrium altissimum L.		–	0.111	–	0.028
Tragopogon dubius Scop.		0.077	0.278	–	0.361
Biennials					
Cirsium vulgare (Savi.) Ten.†	Moderate	0.404	0.778	0.288	0.444
Verbascum blattaria L.		–	0.028	–	–
Verbascum thapsus L.	Low	0.019	0.167	–	0.028
Perennials					
Convolvulus arvensis L.†		0.481	0.472	0.635	0.417
Plantago lanceolata L.		0.115	0.139	0.019	0.278
Rumex acetosella L.	Moderate	0.115	0.167	0.077	0.028
Rumex crispus L.	Low	–	0.083	0.019	–
Silene latifolia Poir.		0.038	–	–	–
Taraxacum officinale F.H. Wigg.		0.019	0.028	–	0.028
Trifolium pratense L.		–	0.083	–	–
Grasses					
Annuals					
Aira caryophyllea L.		0.154	0.361	0.115	0.472
Avena barbata Pott ex Link	Moderate	0.019	0.056	–	–
Brachypodium distachyon (L.) P. Beauv.	Moderate	0.135	0.278	0.038	0.222
Briza minor L.		0.038	–	–	–
Bromus diandrus Roth	Moderate	0.077	0.083	–	–
Bromus hordeaceus L.	Low	0.115	–	0.038	0.056
Bromus madritensis ssp. rubens (L.) Husn.	High	0.038	–	–	–
Bromus sterilis L.		0.058	0.083	0.038	0.083
Bromus tectorum L.	High	0.038	0.444	0.154	0.472

Table 1.   (Continued)

Species by life form and life cycle	Cal-IPC Rating	Mulched 2014	Mulched 2015	Unmulched 2014	Unmulched 2015
Cynosurus echinatus L.	Moderate	0.019	0.25	0.038	0.222
***Digitaria sanguinalis* (L.) Scop.**		0.077	0.028	–	–
***Echinochloa crus-galli* (L.) P. Beauv.**		0.365	0.25	–	–
Festuca myuros L.	Moderate	0.154	0.583	0.346	0.556
***Festuca perennis* (L.) Columbus and J.P. Sm.**	Moderate	0.115	0.167	–	–
Hordeum marinum ssp. *gussoneanum* (Parl.) Thell.	Moderate	0.096	0.139	–	0.028
Hordeum murinum L.	Moderate	0.096	0.028	–	0.028
***Poa annua* L.**		0.038	–	–	–
***Secale cereale* L.**		–	0.028	–	–
***Triticum aestivum* L.**		0.038	–	–	–
Perennials					
***Brachypodium pinnatum* (L.) P. Beauv.**		0.019	–	–	–
Phleum pretense L.		–	–	–	0.028
Polypogon australis Brongn.		–	0.056	–	0.028
Poa bulbosa L.		0.135	0.139	–	0.056
Poa compressa L.		0.366	0.389	0.25	0.194
***Sorghum halepense* (L.) Pers.†**		0.135	0.167	–	–

investigation of the interaction for annual richness ($P < 0.001$) indicated that mulched areas had significantly higher annual richness in 2014 ($P < 0.001$) but not in 2015 ($P = 0.106$); similar to overall richness, annual richness was higher in 2015 for both treatments ($P < 0.001$ for both). There was only a weak relationship between annual richness and mulch cover ($P = 0.079$) in the best conditional model, which also included a squared term ($P = 0.139$).

Significant differences in plant communities were detected between mulched and unmulched areas in terms of species abundance in both 2014 and 2015 with PERMANOVA tests ($P = 0.017$, $P = 0.008$). In contrast, the NMDS ordinations on abundance data did not show any clear separation (data not shown). There was a significant interaction between treatment and year for Simpson's diversity index ($P = 0.029$), for which subsequent treatment contrasts indicated significantly higher indices in mulched areas in 2015 ($P = 0.003$) *v.* no difference through time for unmulched areas. Within year, mulched areas had significantly higher indices over unmulched areas in 2015 ($P = 0.001$), but in 2014 there was no difference ($P = 0.309$) (Fig. 5).

Conifer regeneration

In general, conifer regeneration was highly variable, where most plots had no regeneration across either treatment in either year (Fig. 6). For all conifer species together, regeneration was significantly less abundant in 2015 than in 2014 due to mortality of 2014 seedlings ($P < 0.001$; Table S1). When considering only plots that were measured in both years, mean mortality rates for 2014 germinants were higher in mulched areas than in unmulched areas for the more shade-tolerant Douglas-fir and white fir, but were lower in mulched areas for the remaining species, particularly for ponderosa and sugar pine. However, the only significant difference in mortality rates was higher white fir percentage mortality in mulched areas compared with unmulched areas

($P = 0.034$, Fig. 7). Combining surviving seedlings and new germinants for 2015, densities did not exceed 2014 values on mulched plots, and Douglas-fir and ponderosa pine only slightly exceeded 2014 densities on unmulched plots. For individual species, there was no effect of mulch or difference by year for the abundance of ponderosa pine, sugar pine or incense-cedar (P values were 0.937, 0.540, 0.756; Table S2). For Douglas-fir, there was a marginally significant interaction ($P = 0.069$); treatment contrasts indicated a marginally significant, positive effect in 2014 ($P = 0.072$) but no effect in 2015 ($P = 0.280$). The best conditional model for 2014 with dAIC included mulch depth, but this term was not significant ($P = 0.404$). In terms of differences by year, Douglas-fir regeneration was significantly higher in 2015 in unmulched plots ($P = 0.005$) but there was no difference for mulched plots ($P = 0.225$), suggesting there was no linear or nonlinear relationship between Douglas-fir density and the amount of mulch cover or depth. There was a significant interaction between mulch treatments and year for white fir ($P = 0.007$), but the only significant difference within treatment contrasts was the lower white fir abundance in 2015 for mulched plots ($P < 0.001$; Table S2). Summary data for regeneration density by species, treatment and year can be found in Table S2.

Discussion

Our results suggest that the increasing use of mulch for erosion control (Robichaud *et al.* 2014) may also increase the risk of unintended non-native species introductions after wildfire. We detected significantly greater non-native species richness and cover in mulched areas over unmulched areas, similar to other studies throughout the western US (Dodson and Peterson 2010; Dodson *et al.* 2010; Morgan *et al.* 2014), including one specifically examining rice straw mulch (Kruse *et al.* 2004). These studies documented greater occurrence of non-natives in mulched areas, but also detected the same non-native species in unmulched areas. We similarly found greater frequency and

Table 2. Indicator species for mulched and unmulched areas in 2014 and 2015

Daggers (†) indicate species exclusively observed in mulched areas

| Mulched areas	2014	2015	Life form
Non-native			
Cirsium vulgare (Savi) Ten.	X		forb
Echinochloa crus-galli (L.) P. Beauv.†	X		graminoid
Lactuca serriola L.	X	X	forb
Polygonum aviculare L.	X		forb
Poa compressa L.	X		graminoid
Sonchus asper (L.) Hill	X	X	forb
Solanum nigrum L.	X		forb
Native			
Abies concolor (Gord. and Glend.) Lindl. ex Hildebr.	X		tree
Acmispon americanus (Nutt.) Rydb. var. *americanus*			
Ceanothus integerrimus Hook. and Arn.	X	X	shrub
Epilobium brachycarpum C. Presl.		X	forb
Erigeron canadensis L.	X		forb
Leptochloa fusca var. *fascicularis* (Lam.) N. Snow†	X		graminoid
Pseudognaphalium beneolens (Davidson) Anderb.	X		forb
Ribes roezlii Regel	X	X	shrub
Symphyotrichum subulatum (Michx.) G. L. Nesom var. *parviflorum*(Shinn.) S. D. Sundb.†	X		forb
Viola lobata Benth.	X		forb
Unmulched areas			
Native			
Carex spp.		X	forb
Chamaebatia foliolosa Benth.		X	forb
Dichelostemma multiflorum A. Heller	X		forb

cover of many non-native species (e.g. *Cirsium vulgare*, *Sonchus asper*, *Lactuca serriola*) in mulched areas that also occurred in unmulched areas. However, this study is the first to document non-native species that were completely exclusive to mulched areas, providing robust evidence for non-native introductions by rice mulch. Of the 25 species found only in mulched areas, we acknowledge that many of the species with low frequency (one–two plots) could have already been present and the fidelity to mulched areas could have occurred by chance. However, we find it highly unlikely that the eight species that occurred on ≥10% of mulched plots occurred by chance, particularly as three are known associates in rice fields (*Leptochhloa fusca*, *Echinochloa crus-galli*, *Persicaria maculosa*) and two species are native to wet, marshy areas of California's Central Valley (*Leptochhloa fusca* and *Symphio-trichium subulatum* var. *parviflorum*), where the rice was grown. Certified weed-free rice mulch from a different supplier in a different county introduced many of the same species on the 2014 King Fire in northern California (Blake Engelhardt, Eldorado National Forest botanist, pers. comm., 2015).

This raises concerns over the process for weed-free certification requirements, which do not actually guarantee that the straw bales are free of any non-native species seed; rather, the standards only require that no federal or state-listed noxious species are observed in the supplier's agricultural fields by state inspectors (California Department of Food and Agriculture 2013). Our detection of *Sorghum halepense*, which is listed as noxious by the State of California (California Department of Food and Agriculture 2016), in mulched areas suggests that the current certification process is not adequate. Moreover, this

certification approach is inherently limited because state and federal noxious species lists tend to focus on species that are considered noxious in an agricultural setting, with less consideration for potential effects on wildlands. In addition to *Sorghum halepense*, we observed two species in mulched areas that are ranked as having high potential for affecting wildlands by the California Invasive Plant Council (Cal-IPC), and 12 that were ranked as moderate (Table 1, California Invasive Plant Council 2016). Because they are not designated as noxious, these species could be present during certification inspections but the supplier would still be certified 'weed-free'.

More broadly, the introduction of these non-native rice field associates in our sites violates the assumption that these species cannot establish in drier forested habitat, at least in the short term. However, we acknowledge that our monitoring is limited to just 2 years post-fire, and further monitoring will be required to see if these species will persist in the long term. These species may be ephemeral in upslope mixed conifer forest habitat, as many are adapted to wetter habitats (*Echinochloa crus-galli*, *Persicaria maculosa*, *Leptochhloa fusca*); however, they may be more likely to persist in riparian areas. In addition to increases in *Sorghum halepense* frequency for plots that were mulched and later salvaged, field crews observed generally higher densities of *Sorghum halepense* in salvaged areas, particularly in skid trails. Further research is needed to verify this observation, but it would not be surprising if an additional disturbance could facilitate further proliferations of these introduced species, particularly species such as *Sorghum halepense* that thrive in a variety of disturbed environments and can spread by both seed and rhizome fragments (Howard 2004). The introduction of this
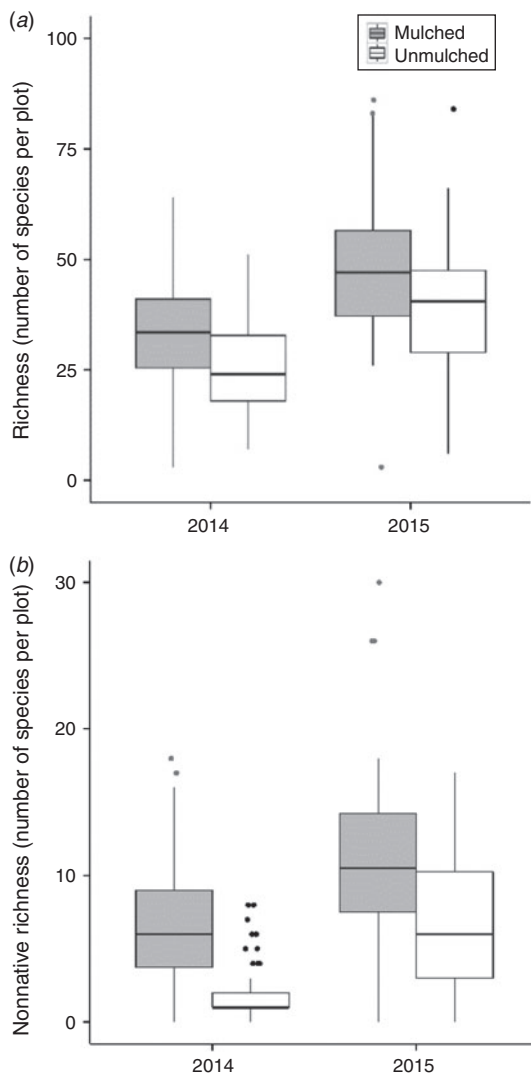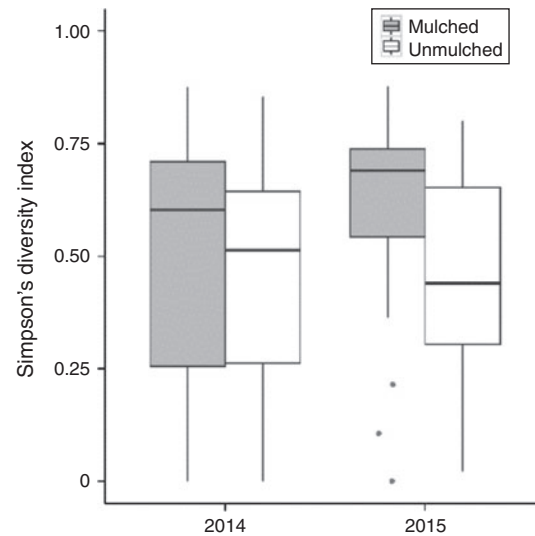
Fig. 4. (*a*) Overall species richness and (*b*) non-native richness by treatment and year. The boxes define the 1st quartile (25th percentile) and 3rd quartile (75th percentile), and the line within the box defines the median. The whiskers extend to the highest value that is within 1.5 times the inter-quartile range and the dots are outliers that exceed this distance from the quartiles. Note differences in *y*-axes.

Fig. 5. Simpson's diversity index by treatment and year. The boxes define the 1st quartile (25th percentile) and 3rd quartile (75th percentile), and the line within the box defines the median. The whiskers extend to the highest value that is within 1.5 times the inter-quartile range and the dots are outliers that exceed this distance from the quartiles.

species should be of some concern because of its noxious status in agricultural settings (California Department of Food and Agriculture 2016), which may suggest the potential for problems in wildland settings. In addition, several species that were detected only in mulched areas in 2014 were also detected on at least one unmulched plot in 2015 (Table 1), suggesting the potential for spread outside treatment areas.

One of the greatest concerns with non-native species introductions is the potential for a species to promote positive feedbacks that favour itself over other native species, eventually leading to state changes or disturbance regime changes (D'Antonio and Vitousek 1992; Suding *et al.* 2004). Of the species observed, *Bromus tectorum* L. (cheatgrass) and *Bromus madritensis* ssp. *rubens* (L.) Husn. (red brome) have been linked

to altered fire regimes (D'Antonio and Vitousek 1992) and both are ranked as potentially having large effects on wildlands by Cal-IPC (California Invasive Plant Council 2016). *Bromus madritensis* ssp. *rubens* occurred on only one mulched plot and *Bromus tectorum* occurred relatively equally across treatments, with dramatic increases in *Bromus tectorum* frequency across both treatments in 2015 (Table 1). For the remaining species, our understanding of the potential effects of their current levels of invasion is somewhat hindered by a relative lack of research on invasions in mountainous regions (Pauchard *et al.* 2009). If the remaining species do not cause state changes, some of the non-native response could be ephemeral and decline with increasing canopy closure by shrubs or trees in the future, as most are shade intolerant (Martin *et al.* 2009; Bohlman *et al.* 2016). Even in this case, there could be longer-term effects if these species are prevalent enough to alter seedbank dynamics, allowing for propagule pressure build-up that could result in a stronger response after another disturbance (Lockwood *et al.* 2005). Further, temporary displacement of native species could have cascading effects on other trophic levels.

We were encouraged that there were no detectable differences in overall plant cover by treatment. Our results are in contrast to those of Dodson *et al.* (2010), who detected a positive relationship between mulch cover and total plant cover; however we did not test continuous mulch cover because mulch as a categorical variable was not significant in the full model. The differences between these results may be partly due to our inclusion of plots explicitly chosen to be in control areas rather than relying on variation in mulch cover within mulched areas to serve as a baseline (Dodson and Peterson 2010). We were further encouraged that conifers were generally unaffected by mulch, with the exception of a marginally significant, positive mulch effect on Douglas-fir densities. We hypothesise this
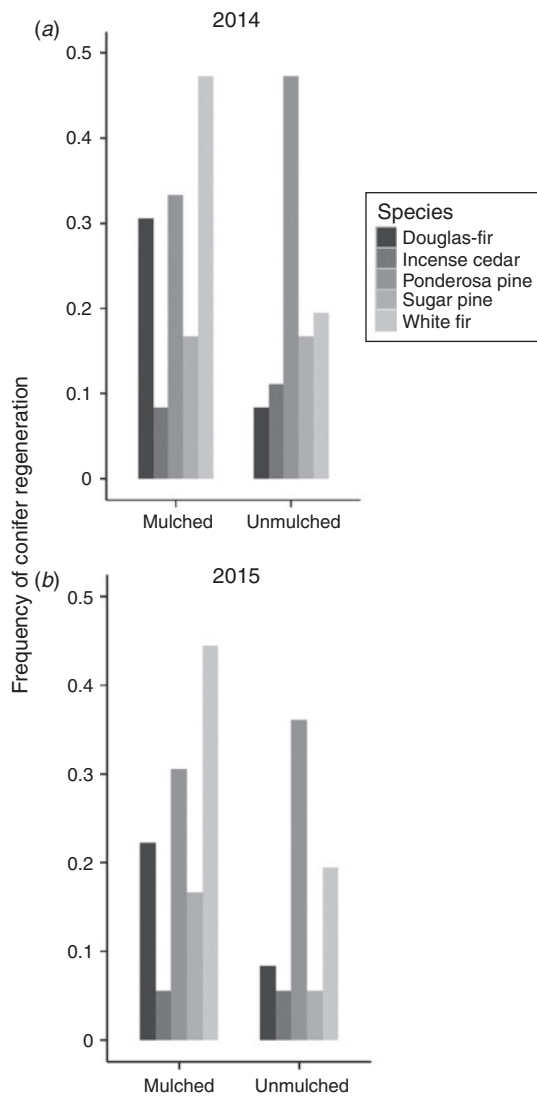
Fig. 6. Frequency of conifer regeneration across plots by treatment and species for (*a*) 2014 and (*b*) 2015.

effect is driven in part by increased soil moisture retention, but we would have expected this same effect on at least some of the other species as well. Investigation of mulch effects on these species on other fires is warranted. More broadly, our finding of no effect to a marginal, but positive, effect on conifer seedlings is similar to Dodson and Peterson's (2010) report of a positive effect of mulch cover <25% on lodgepole pine (*Pinus contorta* Loudon), with no effect at levels >25% in north-central Washington, US. However, this finding is in contrast to Kruse *et al.* (2004) who found a negative effect on conifer regeneration in forests that were more similar to ours, dominated by Douglas-fir, red and white fir pre-fire. This difference in response may be because Kruse *et al.* (2004) examined the relationship between mulch and conifers at the quadrat scale, whereas we examined conifer occurrence at the plot scale. It is possible that where we had deep mulch, we did not have conifers either, but that there is enough variability in mulch depth at our plot scale for some regeneration to occur.
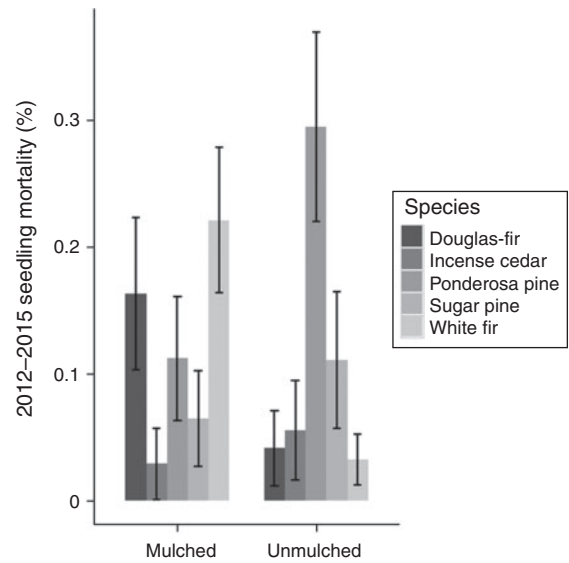
Fig. 7. Mean seedling mortality rates with standard error by treatment and species. Percentage mortality was calculated only for plots that were measured in both years ($n_{mulched} = 36$, $n_{unmulched} = 36$).

More generally, where mulch application did have an effect in at least 1 year (non-native forb cover, richness, non-native richness, annual richness, Douglas-fir density), we did not detect any relationship with mulch depth in conditional tests, and relationships with mulch cover were generally weak. Despite the weak signal, the nonlinear relationships between mulch cover and richness/non-native richness are interesting, because Dodson and Peterson (2010) also observed weak nonlinear relationships, with a similar peak in response at ~60% cover. Although weak, the nonlinear response may reflect a meaningful trend given the similarity between studies. In our study, the lack of effect for mulch depth and relatively weak effects for mulch cover on non-native richness indicates that the presence of any mulch may ultimately be more important than the amount of mulch applied.

We found our results for plant community composition and abundance by species somewhat more difficult to interpret. Because of significant differences between treatments with the PERMANOVA test and Simpson's diversity index, the lack of any visual separation observed with NMDS ordinations was surprising. We interpret this to mean that differences may be significant but not very large. Higher Simpson's indices in mulched areas indicate a less even community that is dominated by a few species, which is likely driven by the greater non-native response on mulched sites. Similar to the concerns raised above about non-native species, even if this homogenisation is ephemeral with canopy closure, it could have cascading effects on other trophic levels and alter seedbank dynamics.

Because mulch is currently considered the most efficacious treatment for post-fire erosion control (Robichaud *et al.* 2010; Robichaud *et al.* 2013*b*, 2013*c*, 2013*a*), the potential introduction of non-native species and homogenisation of plant communities presents a trade-off for managers. Stricter certification requirements or sterilisation of the straw by radiation or heat treatments could alleviate some of the invasion risk. Until such

strategies are implemented, BAER prescriptions for post-fire rice mulch treatments should also consider including plans and funding for non-native plant monitoring and eradication. This may be difficult to implement considering that the treatment is already more costly than others (Robichaud *et al.* 2014); however, given the potential for unintended consequences, such mitigations will be important for future post-fire rehabilitation efforts.

Considered more broadly, another way to reduce the potentially adverse effects of straw mulch or other rehabilitation treatments (Beschta *et al.* 2004) would be to reduce the need to apply them to begin with. Much of the forest that burned in the Rim Fire had dramatically changed in the last century as a result of fire exclusion and harvesting (Collins *et al.* 2011, 2015), making them much more vulnerable to high-severity fire (Lydersen *et al.* 2014). Fire hazard reduction and forest restoration treatments that reduce tree densities and surface fuels are very effective at reducing the intensity of wildfires (Fulé *et al.* 2012), suggesting that by mitigating fire severity, forest restoration treatments could reduce the need for rehabilitation. To that end, management strategies that enhance forest resilience and decrease the size of high-severity forest patches should remain an overarching management goal (Stephens *et al.* 2012, 2016).

Acknowledgements

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