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Herbicides Exert Weak and Variable Effects on Microclimate in Early-Seral Forests

Abstract

Ecosystem modification driven by anthropogenic land-use and land-cover change is one of the leading drivers of global biodiversity declines. Simultaneously, ongoing climate change is modifying ecosystems and will have far-reaching impacts on the structure and function of ecological communities. Rising surface temperatures are predicted to have negative effects on plants and animals, but such predictions are typically broad and poorly matched to the spatial scale at which most organisms experience the environment. Microclimate thermal regimes are often moderated by land-cover, and intensive forest management practices have the potential to either ameliorate or exacerbate climate change effects on biota. In this study, we examined the degree to which air temperature varied across an experimental gradient of herbicide application intensity within early-seral forests in the Coast Range mountains of western Oregon. We evaluated stand-level air temperatures in regenerating stands subjected to light, moderate, and intensive herbicide treatments, as well as a no-spray control. We examined whether daily temperature measurements (minimum, mean, and maximum) and their associated coefficients of variation were influenced by herbicide treatments. We found that herbicide treatments had some influence on mean and maximum air temperatures, but not on minimum temperature or on measures of temperature variability. However, temperature effects were small (< 0.5 °C), the direction of these effects were inconsistent, and pairwise contrasts often failed to detect significant differences after accounting for multiple comparisons. Our results suggest that post-harvest vegetation management has limited impacts on fine-scale air temperatures and is unlikely to either amplify or buffer the projected effects of climate change within early-seral forests.

Keywords: air temperature, early-seral forests, herbicides, intensive forest management, microclimate

Introduction

Much of the world's forests have been modified by anthropogenic changes, including ongoing modification that occurs through a wide range of intensive forest management practices (Rudel et al. 2005, Hooke et al. 2012). As part of these changes and in response to societal demands for wood products, there has been a global increase in the extent of planted forest area that is expected to continue well into the future (FAO 2015). Compared to naturally regenerated forests, planted forests may have reduced capacity to support diverse ecological communities, their component biodiversity, and the ecosystem processes they support (Foster et al. 1996, Spies 2004, Carnus et al. 2006, Thomas et al. 2006, Potapov et al. 2008).

Despite this, planted forests have been considered a potential opportunity for conservation (Hartley 2002, Brockerhoff et al. 2008) because of a growing recognition that the early-seral stage can provide habitat for many species of conservation concern (Swanson et al. 2011, DellaSala et al. 2014, King and Schlossberg 2014, Swanson et al. 2014).

Compositionally and structurally diverse early-seral forests are declining in some locations in the northern hemisphere (Angelstam 1998, Thomas et al. 2006) and the amount of early-seral forest is often considered below historic levels (Spies and Johnson 2007). This trend is of conservation concern because early-seral forests are associated with high food web complexity and species diversity (Hagar 2007, Swanson et al. 2011), and because population declines in a range of organisms have been linked to decreases in the availability of structurally and compositionally diverse early-seral forests (Litvaitis 1993, Hunt 1998). Particularly, intensive forest vegetation

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management can decrease wildlife habitat quality in early-seral forests (Swanson et al. 2011, Betts et al. 2013). Intensive forest management practices often use herbicides to suppress the growth of herbaceous and broadleaf vegetation that compete with crop trees in early-seral conditions (Fortier and Messier 2006), in turn changing the abundance, diversity, and composition of early-seral forest vegetation (Balandier et al. 2006). Herbicides mainly impact early-seral forests by altering the relative dominance of plant species within the community (Balandier et al. 2006) which, in turn, can have negative consequences for organisms that rely on early-seral forests during critical periods of the life cycle (Easton and Martin 1998, Betts et al. 2013).

In addition to management-related activities, climate is also recognized as an important modifier of natural ecosystems (Dawson et al. 2011), and changes in climate can affect organisms through direct and indirect means (Parmesan 2006, Bellard et al. 2012, Buckley et al. 2012). Consequently, it is expected that human modification of ecosystems and climate change are magnifying one another's effects on global biodiversity (Brook et al. 2008). Microclimatic shifts are one effect of climate and management-driven vegetation change that may alter habitat quality for organisms in early-seral forests. One component of microclimate often linked to habitat quality is air temperature (Huey 1991, Frey et al. 2016) as surface air temperatures and their variability are a primary driver of many bioecological processes, including nutrient and water cycles, organismal physiology, and species distributions (Bonan 2008a). Forest cover can exert a strong influence on local air temperature via altering biophysical characteristics such as albedo and surface roughness, ultimately influencing the amount of solar radiation intercepted by land surfaces (Bonan 2008b, Jackson et al. 2008, Anderson et al. 2010, Alkama and Cescatti 2016). However, previous work indicates that plant communities can also exert significant effects on forest microclimatic conditions through variation in species-specific phenologies (von Arx et al. 2012, Zhao and Jackson 2014, Naudts et al. 2016). For example, the albedo of broadleaf vegetation during summer can be 10% greater than coniferous forests (Anderson et al. 2010). This results in greater transpiration and reflectivity of broadleaf vegetation, in turn cooling air temperatures to a greater degree than coniferous forests. Indeed, von Arx et al. (2012) showed that daily maximum temperatures were two times lower in broadleaf forests than in those dominated by coniferous trees. The greater cooling effect of broadleaf vegetation suggests that forests with greater amounts of broadleaf cover are more likely to become decoupled from regional climates and act as "micro-refugia" for species under advancing global climate change (Ashcroft 2010, Dobrowski 2011).

To date, relatively few studies have examined the potential indirect effects of herbicide treatment on air temperature in forests (Proe et al. 2001, Devine and Harrington 2007, Parker et al. 2012). Importantly, these studies have examined mean air temperature alone without evaluating temperature variability despite the influence variability can have on organisms and ecological processes (Vasseur et al. 2014). Temperature variability may allow temperature-sensitive species to persist in localized areas with suitable microclimates (Sears et al. 2011); conversely, it may also impact early-seral species that prefer more stable microclimatic conditions (e.g., Checa et al. 2014). In this study, we examined patterns of microclimatic air temperature variability across a gradient of post-harvest vegetation control in the central and northern Oregon Coast Range. We predicted that experimental herbicide treatments would increase surface air temperatures by shifting stand composition towards predominantly coniferous species, and that temperature increases would be greater with increasing herbicide treatment intensity. We also predicted that herbicide-treated stands would exhibit greater air temperature variability relative to untreated stands (Baker et al. 2014, Hardwick et al. 2015) due to changes in the amount of vegetative cover that can moderate air temperature (Geiger et al. 2003). By examining a gradient in herbicide treatment intensity, our study provides a valuable step in determining how intensive management practices influence microclimatic conditions at the scale experienced by organisms, and whether changes in management practices

may alter expected climate change effects within early-seral forests.

Methods

Study Area

We conducted this study from May to August 2014 at five experimental study blocks located in the central and northern Oregon Coast Range. The maritime climate of this region is characterized by cool, wet winters and mild, dry summers (Franklin and Dyrness 1988). Mean annual precipitation ranges from 165 to 330 cm with most precipitation occurring October through March, and mean annual air temperature between 5 and 19 °C (Franklin and Dyrness 1988, Taylor and Hannan 1999). Soils are moderately deep to deep well-drained silt loam soil derived from basalts, sandstone, and siltstones. Topography is characterized by somewhat low, highly dissected mountains with slopes ranging from 0–90% (Knezevich 1982, Taylor and Hannan 1999). All stands in our study were found within the western hemlock (*Tsuga heterophylla* [Raf.] Sarg.) zone (Franklin and Dyrness 1988), and ranged in elevation from 165 to 765 m. Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco) dominates early-seral forest plantations in this region, with grand fir (*Abies grandis* [Douglas ex D. Don] Lindl.), western hemlock, noble fir (*Abies procera* Rehder), and western redcedar (*Thuja plicata* Donn ex D. Don) forming minor components. Dominant shrub and woody species include big-leaf maple (*Acer macrophyllum* Pursh), California hazelnut (*Corylus cornuta* Marshall *var. californica* [A. DC.] Sharp), cascara buckthorn (*Frangula purshiana* [DC.] A. Gray), common snowberry (*Symphoricarpos albus* [L.] S.F. Blake), oceanspray (*Holodiscus discolor* [Pursh] Maxim.), red alder (*Alnus rubra* Bong.), and vine maple (*Acer circinatum* Pursh). Smaller understory broadleaf species include Oregongrape (*Mahonia nervosa* [Pursh] Nutt.), salal (*Gaultheria shallon* Pursh), and *Vaccinium* spp. The herbaceous community is comprised of many native and non-native plants, especially swordfern (*Polystichum munitum* [Kaulf.] C. Presl) and brackenfern (*Pteridium aquilinum* [L.] Kuhn).

Study Design and Experimental Herbicide Treatments

Our study used a randomized complete block design in which 20 stands were selected in five separate blocks, with one stand within each study block randomly assigned to one of four experimental treatments (no spray control, and light, moderate, and intensive herbicide application). Our five experimental study blocks are a subset of sites (due to limitations associated with sampling logistics) of a broader study examining the effects of intensive forest management on early-seral forest ecology (see Betts et al. 2013 and Kroll et al. 2017 for full study description). All stands within each block were sited within 5 km of each other to ensure spatial independence of treatments while reducing within-block variation (e.g., slope, elevation). Stands were clearcut in fall 2009/winter 2010 and replanted in spring 2011 with Douglas-fir, the dominant conifer species in the region, at approximately 1100 trees per hectare. Herbicides and surfactants typically used by private industrial timber harvesting operations were applied to stands between 2009 and 2014 following current operational prescriptions and in order to create a gradient of management intensity. Stands in the control group received no herbicide application after harvest, and stands in the light herbicide treatment were aerially sprayed with an herbaceous spray in 2011 and a broadleaf release spray in 2012. A site preparation broadleaf vegetation spray was applied to the moderate treatment in 2010, as well as an herbaceous spray in 2011. The intensive treatment similarly had a site preparation broadleaf spray in 2010, with herbaceous sprays applied in 2011–2013, and a broadleaf release spray in 2012 and 2014. Follow-up backpack spraying was conducted on moderate and intensive treatments as needed to control deciduous stump sprouts. A full description of the amount and type of herbicides and surfactants used on stands can be found in Betts et al. (2013). We note that the moderate treatment was an approximation of current operational practices in our study area.

Air Temperature Sampling

We recorded air temperature during the local bird breeding season as part of a broader study evaluating the influence of temperature on songbird reproductive rates (Jones 2015). Air temperatures were measured with temperature loggers mounted at eight sampling points within each stand (total $n = 160$ points) to assess the effects of herbicide treatment on local temperatures experienced by many terrestrial organisms (Geiger et al. 2003, Potter et al. 2003). We located temperature sampling points with several considerations in mind: 1) points needed to be separated by a minimum of 50 m, 2) points had to evenly cover stands, and 3) walking distance between points needed to be < 30 min (for logistical considerations). Neither vegetation cover nor distance to edge were considered when siting temperature sampling points; thus, we measured both as covariates in our analysis because they have the potential to influence air temperature (Baker et al. 2014; see below).

At each air temperature sampling point, we mounted a single temperature logger (Thermochron iButton, Embedded Data Systems, Lawrenceburg, KY) on a metal stake approximately 1.5 m high off the ground (following conventional placement of meteorological instruments within meteorological stations; Geiger et al. 2003) with a wooden covering to protect the logger from direct solar radiation and precipitation. We placed each logger within a 10-cm diameter white PVC tube with ventilation holes to allow for airflow and minimize heat accumulation. We used two temperature loggers that varied slightly in their accuracy (iButton DS1921: ± 1.0 °C accuracy, $n = 71$ points; iButton DS1922L-F5: ± 0.5 °C accuracy, $n = 89$ points); the two models were distributed arbitrarily among sampling points so that estimated differences in air temperature between stands were within ± 1.0 °C of their true values. All loggers were validated against an independent digital thermometer (Omega HH609R, Omega Engineering, Stamford, CT) prior to deployment; only loggers that deviated from the digital thermometer by < 0.5 °C during validation procedures were used in this study.

We programmed temperature loggers to record temperature every 15 min so that we obtained data on the minimum (T_{min}), mean (T_{mean}), and maximum (T_{max}) air temperature at each sampling point across a 24-hr sampling period (i.e., starting at midnight and running until midnight on the successive day). We also used this 24-hr sampling period to calculate coefficient of variation measures for minimum temperature (T_{minCV}), mean temperature (T_{meanCV}), and maximum temperature (T_{maxCV}). Temperature data were initially inspected and assessed for errors, and we removed all extreme, atypical values (> 50 °C or < –10 °C) caused by instrument malfunction or when logging stations were disturbed by wildlife; this led us to remove < 5% of temperature values obtained.

Vegetation Sampling

We measured vegetation cover at each temperature sampling point to quantify changes in vegetation cover among the experimental herbicide treatments. At each sampling point, we measured vegetation on three subplots, each of which was circular in shape with a 3-m radius. We located one subplot centrally on the sampling station, and two peripheral subplots 25 m distant from the central subplot. The azimuth of the first peripheral subplot was chosen randomly, with the second peripheral subplot located 180° away from the first peripheral subplot. At each subplot, we visually estimated species cover (or to genus, in the case of *Rubus*, *Ribes*, and forbs) in each of three distinct vegetative layers (i.e., herbaceous [0–0.5 m], shrub [0.5–2.0 m], and canopy [> 2.0 m]). For analysis, we summed the total amount of 1) broadleaf cover (as defined by Ellis and Betts 2011), 2) conifer cover, and 3) vegetative cover for all plant species/functional group over all subplots. At each sampling point, we also measured three variables that have been shown to strongly influence microclimatic air temperatures (Dobrowski 2011): elevation, distance to stand edge, and aspect, which we transformed to "southwestness", providing an index ranging from –1 to 1 that is indicative of environmental aridity (Franklin et al. 2000, Huang et al. 2012).

Statistical Analysis

All models were fit using the lme function of the nlme package (Pinheiro et al. 2015) in 'R' v3.2.0 (R Development Core Team 2018). We used a randomized block analysis of covariance with one random effect (block) to evaluate whether mean temperature and variability differed among experimental herbicide treatments. We constructed models for air temperature response variables (i.e., T_{min}, T_{mean}, T_{max}) and their associated coefficient of variations (i.e., T_{minCV}, T_{meanCV}, T_{maxCV}) with herbicide treatment as a fixed effect (four levels: control, light, moderate, intensive). Our covariates (i.e., elevation, distance to stand edge, and southwestness) were averaged over all sampling points within a stand, and no covariates were correlated ($r < 0.25$ in all cases). We included elevation as a covariate because stands within a block could vary > 100 m due to the highly heterogeneous terrain of the Oregon Coast Range. Distance to stand edge and southwestness were included as covariates to account for any systematic variation in temperatures due to the location of sampling points (Geiger et al. 2003).

We conducted five pairwise comparisons using Bonferroni-corrected 95% confidence intervals and *P*-values in the 'estimable' function of the gmodels package in R (Warnes et al. 2013). We first compared values for the unsprayed control to each of the three herbicide treatments to evaluate how increasing management intensity influenced air temperature. In addition, we compared the moderate herbicide treatment to the light and intensive herbicide treatments (two comparisons) because the moderate treatment is closest to current operational practices on private industrial timber lands (Betts et al. 2013, Kroll et al. 2017).

We also used mixed model analysis to test whether vegetation cover differed significantly among herbicide treatments. We constructed models for three vegetation response variables (i.e., broadleaf, conifer, and total vegetation cover) that included fixed effects for herbicide treatment (four levels: control, light, moderate, intensive) and one random effect (block). All vegetation responses were log-transformed prior to analysis to meet assumptions of variance homogeneity

among herbicide treatments. We back-transformed our results so that values can be interpreted as the mean multiplicative increase or decrease in cover between treatments. Thus, a treatment contrast of one indicates that cover is equal between treatments. We report means and associated confidence intervals, and we considered effects significant at $P < 0.05$.

Results

Mean Air Temperature

Mean air temperature varied little among our study blocks (Figure 1), and was statistically indistinguishable among treatments for T_{min}, T_{mean}, and T_{max} (Figure 2a). We did not detect a treatment main effect on T_{min} ($F_{3,9} = 1.039$, $P = 0.421$). We detected a statistically significant treatment main effect on T_{mean} ($F_{3,9} = 7.655$, $P = 0.008$); however, this significance was driven largely by temperature increases and decreases in the control treatment ($\hat{\beta} = 18.6$, $SE = 1.0$, $t_9 = 18.6$, $P < 0.001$). Pairwise comparisons indicated no statistically significant differences in T_{mean} among treatments (Figure 2a). All pairwise differences in T_{mean} were ≤ 1.1 °C, with the majority < 0.5 °C (Figure 2a), and the direction of these differences varied relative to temperature measures (Figure 2a). We did detect a treatment main effect on T_{max} ($F_{3,9} = 5.34$, $P = 0.022$), with significant pairwise differences between the control and the light treatment ($\hat{\beta} = 1.1$, $SE = 0.4$, $t_9 = 2.67$, $P = 0.026$), and between the control and the moderate treatment ($\hat{\beta} = 1.0$, $SE = 0.4$, $t_9 = 2.34$, $P = 0.044$). Although T_{max} increased monotonically with increasing herbicide treatment intensity, all pairwise comparisons between herbicide treatments were not significant after correcting for multiple comparisons (Figure 2a).

Variability in Mean Air Temperature

Air temperature variability was similar among treatments (Figure 2b), and the majority of estimated differences among treatments for T_{minCV}, T_{meanCV}, and T_{maxCV} were ≤ 1.5% (Figure 2b). Compared to the control, there was less variability as measured by T_{minCV} in the moderate and intensive treatments, and more variability
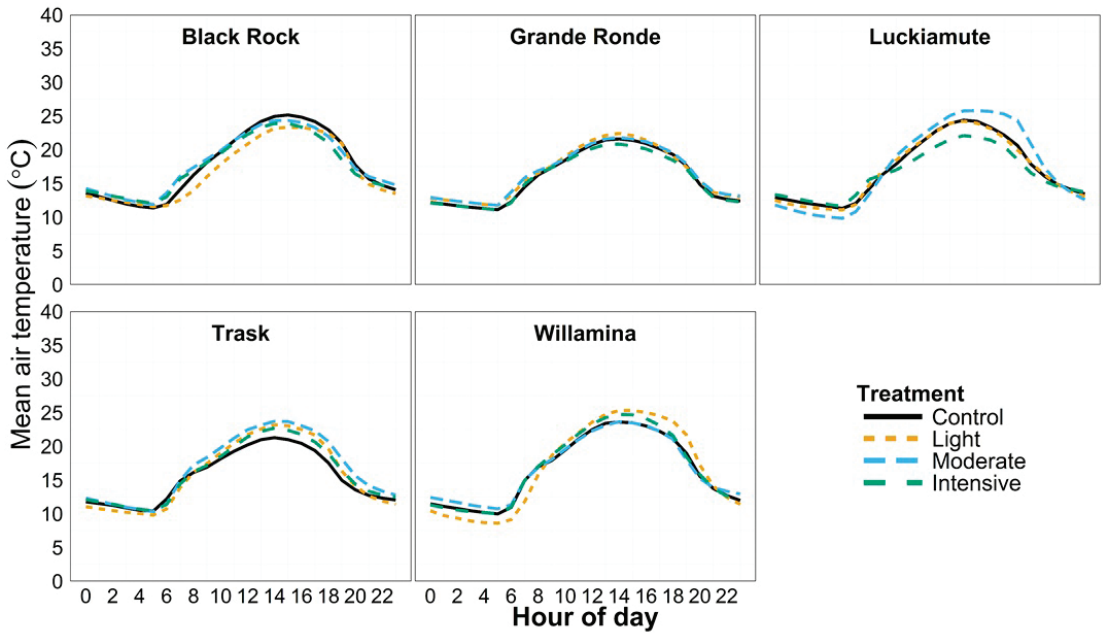
Figure 1. Average daily air temperature fluctuations among herbicide treatments at each experimental block in the Oregon Coast Range. Each line represents the mean of all sample points (8) taken during the May–August, 2014 sampling period.

in the light treatment. Among treatments, T_{minCV} was greater in the light and moderate treatments compared to the intensive treatment (Figure 2b). T_{maxCV} was lower in all herbicide-treated stands when compared to the control (Figure 2b). Among herbicide treatments, T_{maxCV} was greater in the light treatment compared to the moderate treatment and in the moderate treatment compared to the intensive treatment (Figure 2b), although none of these differences were statistically significant.

Vegetation

As expected, we found a significant main effect of herbicide on vegetation cover (Table 1), with broadleaf cover decreasing ($F_{3,11} = 18.68$, $P < 0.001$; Figure 3a; Figure 4) and conifer cover increasing as herbicide intensity increased ($F_{3,11} = 9.19$, $P = 0.003$; Figure 3b). We detected significant pairwise differences in broadleaf cover between the control and the moderate treatment ($t_{11} = 4.88$, $P < 0.001$, $\hat{\beta} = 1.88$ [0.68, 3.08]), the control and intensive treatment ($t_{11} = 5.04$, $P < 0.001$, $\hat{\beta} = 1.93$ [0.74, 3.12]) and the light and the moderate treatment ($t_{11} = 4.89$, $P < 0.001$,

$\hat{\beta} = 1.87$ [0.68, 3.05]; Figure 3a). For conifer cover, we detected a significant difference between the control and intensive treatment only ($t_{11} = -4.69$, $P < 0.001$, $\hat{\beta} = -0.95$ [–1.58, –0.32]; Figure 3b). We did not detect any significant differences in total vegetation cover among treatments ($F_{3,11} = 2.070$, $P = 0.162$; Figure 3c).

Discussion

Our study found that herbicide treatment intensity had little if any influence on air temperatures within early-seral forests, including no detectable differences in temperature among the three herbicide treatments. Similarly, we found that air temperature variability was not influenced by herbicide treatment. These results are surprising given that herbicide application in our study led to changes in the extent of broadleaf vegetation cover among treatments, and that previous work has found that broadleaf vegetation exhibits significantly cooler air temperatures relative to coniferous vegetation (von Arx et al. 2012, Zhao and Jackson 2014, Naudts et al. 2016). One explanation for the lack of relationship between broadleaf cover and air
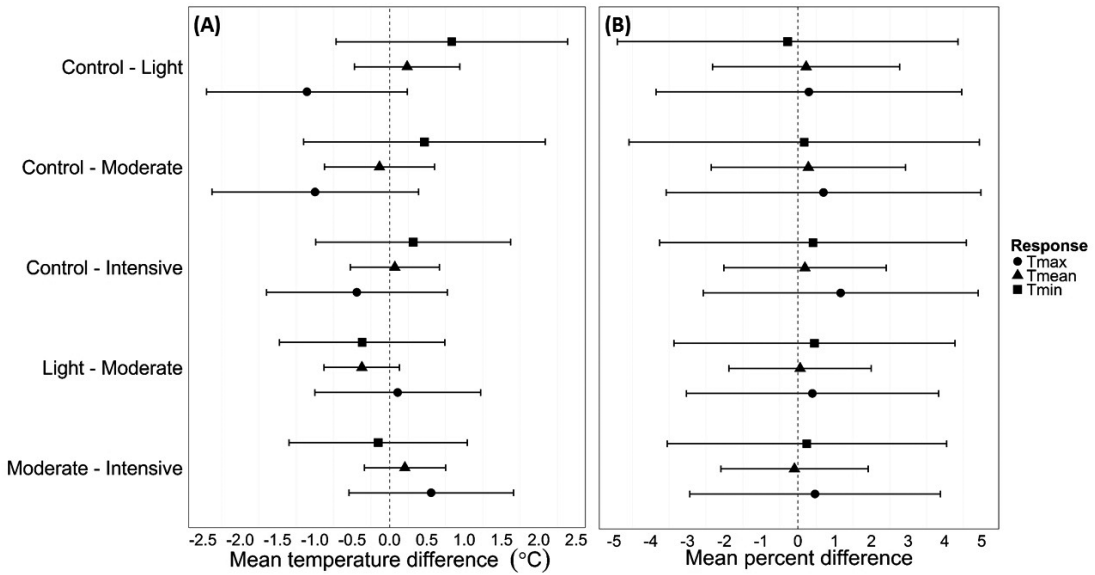
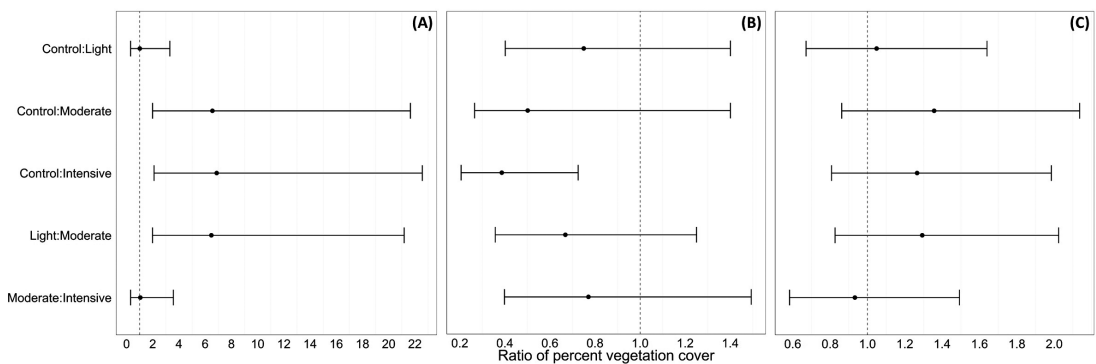Figure 2. Mean air temperature differences among control, light, moderate, and intensive herbicide treatments, with Bonferroni-adjusted 95% confidence intervals, for A) differences in mean air temperature and B) differences in air temperature variability (CVs). The dashed line at zero represents no treatment differences. Estimates on the right side of the dashed line correspond to decreases in air temperature, whereas estimates on the left side correspond to increases in air temperature.

temperature in our study is that temperatures may have been more dependent on total vegetation cover than on relative stand composition. Indeed, changes correlated with measures of total vegetation cover, such as leaf area index, can have the largest moderating influence on air temperature (Aussenac 2000). As we did not find a difference in total cover among herbicide treatments, it could be that stands subjected to different herbicide treatments were not differentiated enough to result in detectable temperature variation. This pattern could have been a function of rapid recovery of vegetation, which is characteristic of highly productive sites such as those in the Oregon Coast Range (Waring and Franklin 1979). We found that the forb cover was extensive in our stands, especially on sites with reduced broadleaf plant cover (Jones 2015), and this could have played a compensatory role in thermal regulation (Geiger et al. 2003). An additional and non-mutually exclusive explanation for our findings is that the differences in vegetation cover we observed between treatments were not large enough to produce a detectable cooling effect. Although

previous work (von Arx et al. 2012, Zhao and Jackson 2014) demonstrated that variation in broadleaf cover can result in small-scale changes in air temperature, the point at which changes in broadleaf vegetation results in detectable cooling effects is unclear. Furthermore, although stand age was not explicitly mentioned, the larger-scale focus of previous work such as von Arx et al. (2012) and study site descriptions given (e.g., tree basal area and height) suggests that examined stands were likely more mature than those included in our study—temperature differences driven by species-specific characteristics such as albedo and transpiration would therefore be more apparent. Lastly, elements of our sampling design may have limited our ability to detect air temperature differences—we measured air temperature in only one year occurring five years post-harvest. This may have limited our ability to detect potential temperature differences 0–4 years post-harvest driven by developing vegetation differences as our experimental treatments were differentiated by scheduled herbicide applications. Additionally, our placement of temperature loggers at the

TABLE 1. Stand level means (± 95% CI) for vegetation measurements and covariates relative to herbicide treatments in the Oregon Coast Range. Vegetation was measured June–August 2014 during the height of the growing season.

Treatment	Broadleaf cover (%)	Conifer cover (%)	Total cover (%)	Canopy cover (%)	Elevation (m)	Aspect (°)
Control	59.4 (22.8)	5.6 (2.6)	144.7 (8.8)	3.6 (4.8)	480.7 (247.60)	198.9 (75.5)
Light	59.0 (19.8)	7.8 (5.9)	140.0 (14.6)	4.2 (6.7)	452.8 (187.40)	188.5 (91.0)
Moderate	13.5 (17.7)	10.2 (3.0)	113.6 (19.9)	0.4 (0.4)	417.1 (183.26)	155.7 (42.4)
Intensive	9.0 (5.2)	15.0 (8.1)	115.8 (14.0)	0.9 (1.0)	526.4 (217.30)	138.9 (50.3)

Figure 3. Ratio of mean percent cover among control, light, moderate, and intensive herbicide treatments, with Bonferroni-adjusted 95% confidence intervals for A) broadleaf vegetation, B) conifer vegetation, and C) total vegetation cover. The dashed line at 1 represents no statistical difference. Note panels are not presented on the same scale.

conventional height of 1.5 m (Geiger et al. 2003) likely limited our ability to detect air temperature differences at smaller scales in the microclimatic layer, which may be relevant to some early-seral species, including various reptiles, invertebrates, and birds that nest and/or forage on the ground. Thus, several factors may have contributed to our inability to detect air temperature differences across our experimental herbicide treatment gradient.

We also did not detect significant vegetation-mediated effects of herbicide treatment on air temperature variability in this study. However, there were some broad patterns in air temperature variability among treatments worth noting. Contrary to expectations (Zhao and Jackson 2014), air temperature variability generally decreased in herbicide-treated stands compared to control stands. The greater air temperature variability observed in control stands was likely linked to spatial variability of regenerating vegetation

relative to stands treated with herbicides (Xu et al. 1997, Heithecker and Halpern 2006, Ma et al. 2010). The distribution and amount of understory cover also influences surface energy fluxes, where air temperatures are generally more tightly coupled to atmospheric conditions and thus more variable over bare ground compared to ground covered with vegetation (Geiger et al. 2003). A second reason why differences in air temperature variability among treatments may have been diminished was due to the matrix in which they occurred (Dobrowski 2011). Many factors, including the height of surrounding vegetation and cold air pooling (a climatic process that can occur in topographic depressions, resulting in a topographically-confined, stagnant air layer that is cooler than the air aloft [Whiteman et al. 2001, Geiger et al. 2003]) influence air temperature within forest clearings. Such matrix effects can reduce the effect of microclimatic differences from changes in surface roughness and albedo
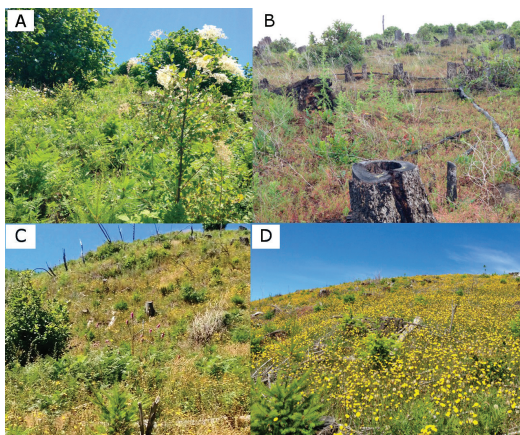
Figure 4. Representative examples of stands spanning the gradient in herbicide intensity for A) no-spray control, B) light, C) moderate, and D) intensive treatments in the Oregon Coast Range, 2014.

driven by decreased vegetation cover (Geiger et al. 2003, Bonan 2008a). An additional reason for the limited variability in temperature relative to our experimental treatments may be due to the rapid mixing of air masses that can occur within forest clearings (Geiger et al. 2003, Bonan 2008a). Regardless of the mechanism(s). responsible, the minor differences in temperature variability we observed may be unlikely to influence the persistence of temperature-sensitive species in four-year-old intensively managed Douglas-fir stands (Sears et al. 2011).

Our finding of generally weak and inconsistent effects of herbicide treatment on microclimatic air temperature in early-seral forests compliments the few other studies that have examined the effects of post-harvest vegetation management on microclimatic air temperatures. Proe et al. (2001) examined the effects of vegetation control practices on microclimate in post-clearcut Sitka spruce (*Picea sitchensis* [Bong.] Carrière) stands and found that herbicide-driven vegetation changes only influenced near-surface air temperatures when combined with other post-harvest treatments (fertilizer and whole-tree harvesting), presumably due to larger shifts in regenerating vegetation and woody material than herbicide application alone. Similarly, Parker et al. (2012) found minimal vegetation-mediated influences of herbicide on

air temperature when woody and herbaceous vegetation control was varied in pine-dominated shelterwood stands. In that study, woody vegetation control produced the only difference in air temperature among forest stands but this effect only occurred in one out of four growing seasons and was relatively small (~ 0.5 °C). Taken with the results of this study, it appears that herbicide use produces negligible differences in the local thermal environment in managed temperate coniferous forests of the central and northern Oregon Coast Range, which may have implications for early-seral forests in other geographic locations with similar characteristics. Moreover, our results indicate that this may be the case whether herbicide application is intensive or relatively light compared to commercial standards.

Microclimatic air temperature is an important component of habitat quality that has the potential to be altered by herbicide-mediated changes in forest vegetation (Lehtinen et al. 2003). Our results indicate that herbicide-mediated changes in the relative composition of broadleaf and coniferous vegetation cover did not result in detectable changes in air temperature in four-year-old intensively managed Douglas-fir stands. Our findings also indicate that post-harvest herbicide treatment has limited impacts on fine-scale air temperatures and is unlikely to either amplify or buffer the projected effects of climate change on biodiversity in early-seral forests (IPCC 2013). This suggests that management practices in early-seral forests may not alter air temperature in a way that impacts early-seral forest organisms during a critical period of the life cycle.

Acknowledgments

Literature Cited

Alkama, R., and A. Cescatti. 2016. Biophysical climate impacts of recent changes in global forest cover. Science 351:600-604.

Anderson, R. G., J. G. Canadell, J. T. Randerson, R. B. Jackson, B. A. Hungate, D. D. Baldocchi, G. A. Ban-Weiss, G. B. Bonan, K. Caldeira, L. Cao, N. S. Diffenbaugh, K. R. Gurney, L. M. Kueppers, B. E. Law, S. Luyssaert, and T. L. O'Halloran. 2010. Biophysical considerations in forestry for climate protection. Frontiers in Ecology and the Environment 9:174-182.

Angelstam, P. K. 1998. Maintaining and restoring biodiversity in European boreal forests by developing natural disturbance regimes. Journal of Vegetation Science 9:593-602.

Ashcroft, M. B. 2010. Identifying refugia from climate change. Journal of Biogeography 37:1407-1413.

Aussenac, G. 2000. Interactions between forest stands and microclimate: ecophysiological aspects and consequences for silviculture. Annals of Forest Science 57:287-301.

Baker, T. P., G. J. Jordan, E. A. Steel, N. M. Fountain-Jones, T. J. Wardlaw, and S. C. Baker. 2014. Microclimate through space and time: microclimatic variation at the edge of regeneration forests over daily, yearly and decadal time scales. Forest Ecology and Management 334:174-184.

Balandier, P., C. Collet, J. H. Miller, P. E. Reynolds, and S. M. Zedaker. 2006. Designing forest vegetation management strategies based on the mechanisms and dynamics of crop tree competition by neighbouring vegetation. Forestry 79:3-27.

Bellard, C., C. Bertelsmeier, P. Leadley, W. Thuiller, and F. Courchamp. 2012. Impacts of climate change on the future of biodiversity. Ecology Letters 15:365-377.

Betts, M. G., J. Verschuyl, J. Giovanini, T. S. Stokely, and A. J. Kroll. 2013. Initial experimental effects of intensive forest management on avian abundance. Forest Ecology and Management 310:1036-1044.

Bonan, G. 2008a. Ecological Climatology: Concepts and Applications. Cambridge University Press, Cambridge, UK.

Bonan, G. B. 2008b. Forests and climate change: forcings, feedbacks, and the climate benefits of forests. Science 320:1444-1449.

Brockerhoff, E. G., H. Jactel, J. A. Parrotta, C. P. Quine, and J. Sayer. 2008. Plantation forests and biodiversity: oxymoron or opportunity? Biodiversity Conservation 17:925-951.

Brook, B. W., N. S. Sodhi, and C. J. A. Bradshaw. 2008. Synergies among extinction drivers under global change. Trends in Ecology and Evolution 23:453-460.

Buckley, L. B., A. H. Hurlbert, and W. Jetz. 2012. Broad-scale ecological implications of ectothermy and endothermy in changing environments. Global Ecology and Biogeography 21:873-885.

Carnus, J. M., J. Parrotta, E. Brockerhoff, M. Arbez, H. Jactel, A. Kremer, D. Lamb, K. OHara, and B. Walters. 2006. Planted forests and biodiversity. Journal of Forestry 104:65-77.

Checa, M. F., J. Rodriguez, K. R. Willmott, and B. Liger. 2014. Microclimate variability significantly affects the composition, abundance and phenology of butterfly communities in a highly threatened neotropical dry forest. Florida Entomologist 97:1-13.

Dawson, T. P., S. T. Jackson, J. I. House, I. C. Prentice, and G. M. Mace. 2011. Beyond predictions: biodiversity conservation in a changing climate. Science 332:53-58.

DellaSala, D. A., M. L. Bond, C. T. Hanson, R. L. Hutto, and D. C. Odion. 2014. Complex early seral forests of the Sierra Nevada: What are they and how can they be managed for ecological integrity? Natural Areas Journal 34:310-324.

Devine, W. D., and C. A. Harrington. 2007. Influence of harvest residues and vegetation on microsite soil and air temperatures in a young conifer plantation. Agricultural and Forest Meteorology 145:125-138.

Dobrowski, S. Z. 2011. A climatic basis for microrefugia: the influence of terrain on climate. Global Change Biology 17:1022-1035.

Easton, W. E., and K. Martin. 1998. The effect of vegetation management on breeding bird communities in British Columbia. Ecological Applications 8:1092-1103.

Ellis, T. M., and M. G. Betts. 2011. Bird abundance and diversity across a hardwood gradient within early seral plantation forest. Forest Ecology and Management 261:1372-1381.

FAO. 2015. Global Forest Resources Assessment 2015: How have the world's forests changed? The Food and Agriculture Organization of the United Nations. Rome, Italy.

Fortier, J., and C. Messier. 2006. Are chemical or mechanical treatments more sustainable for forest vegetation management in the context of the TRIAD? The Forestry Chronicle 82:806-818.

Foster, D. R., D.A. Orwlg, and J. McLachlan. 1996. Ecological and conservation insights from reconstructive studies of temperate old-growth forests. Trends in Ecology and Evolution 11:419-424.

Franklin, J., P. McCullough, and C. Gray. 2000. Terrain variables used for predictive mapping of vegetation communities in Southern California. *In* J. P. Wilson, and J. C. Gallant (editors), Terrain Analysis: Principles and Applications. John Wiley and Sons, Inc. Pp. 331-353.

Franklin, J. F., and C. T. Dyrness. 1988. Natural Vegetation of Oregon and Washington. Oregon State University Press, Corvallis.

Frey, S. J. K., A. S. Hadley, S. L. Johnson, M. Schulze, J. A. Jones, and M. G. Betts. 2016. Spatial models reveal the microclimatic buffering capacity of old-growth forests, Science Advances 2:e1501392.

Geiger, R., R. H. Aron, and P. Todhunter. 2003. The Climate Near the Ground, 6th ed. Rowman and Littlefield Publishers, Oxford, UK.

Hagar, J. C. 2007. Wildlife species associated with non-coniferous vegetation in Pacific Northwest conifer forests: a review. Forest Ecology and Management 246:108-122.

Hardwick, S. R., R. Toumi, M. Pfeifer, E. C. Turner, R. Nilus, and R. M. Ewers. 2015. The relationship between leaf area index and microclimate in tropical forest and oil palm plantation: forest disturbance drives changes in microclimate. Agricultural and Forest Meteorology 201:187-195.

Hartley, M. J. 2002. Rationale and methods for conserving biodiversity in plantation forests. Forest Ecology and Management 155:81-95.

Heithecker, T.D., and C. B. Halpern. 2006. Variation in microclimate associated with dispersed-retention harvests in coniferous forests of western Washington. Forest Ecology and Management 226:60-71.

Hooke, R. L., J. F. Martin-Duque, and J. Pedraza. 2012. Land transformation by humans: a review. GSA Today 22:4-10.

Huang, C., G. P. Asner, and N. N. Barger. 2012. Modeling regional variation in net primary production of pinyon-juniper ecosystems. Ecological Modelling 227:82-92.

Huey, R. B. 1991. Physiological consequences of habitat selection. American Naturalist 137:91-115.

Hunt, P. 1998. Evidence from a landscape population model of the importance of early successional habitat to the American Redstart. Conservation Biology 12:1377-1389.

IPCC. 2013. The physical science basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK and New York, NY.

Jackson, R. B., J. T. Randerson, J. G. Canadell, R. G. Anderson, R. Avissar, D. D. Baldocchi, G. B. Bonan, K. Caldeira, N. S. Diffenbaugh, C. B. Field, B. A. Hungate, E. G. Jobbagy, L. M. Kueppers, M. D. Nosetto, and D. E. Pataki. 2008. Protecting climate with forests. Environmental Research Letters 3:044006.

Jones, K. N. 2015. Combined effects of intensive forest management and microclimate on reproduction in a cavity-nesting songbird. M.S. Thesis, Oregon State University, Corvallis.

Knezevich, C. A. 1982. Soil survey of Polk County, Oregon. United States Department of Agriculture, Soil Conservation Service, Washington DC.

King, D. I., and S. Schlossberg. 2014. Synthesis of the conservation value of the early-successional stage in forests of eastern North America. Forest Ecology and Management 324:186-195.

Kroll, A. J., J. Giovanini, J. Verschuyl, and M. G. Betts. 2017. Assembly dynamics of a forest bird community depend on disturbance intensity and foraging guild. Journal of Applied Ecology 54:1365-2664.

Lehtinen, R. M., J. B. Ramanamanjato, and J. G. Raveloarison. 2003. Edge effects and extinction proneness in a herpetofauna from Madagascar. Biodiversity and Conservation 12:1357-1370.

Litvaitis, J. A. 1993. Response of early successional vertebrates to historic changes in land use. Conservation Biology 7:866-873.

Ma, S., A. Concilio, B. Oakley, M. North, and J. Chen. 2010. Spatial variability in microclimate in a mixed-conifer forest before and after thinning and burning treatments. Forest Ecology and Management 259:904-915.

Naudts, K., Y. Chen, M. J. McGrath, J. Ryder, A. Valade, J. Otto, and S. Luyssert. 2016. Europe's forest management did not mitigate climate warming. Science 351:597-599.

Parker, W. C., D. G. Pitt, and A. E. Morneault. 2012. Influence of herbaceous and woody vegetation control on seedling microclimate, leaf gas exchange, water status, and nutrient relations of *Pinus strobus* L. seedlings planted in a shelterwood. Forest Ecology and Management 271:104-114.

Parmesan, C. 2006. Ecological and evolutionary responses to recent climate change. Annual Review of Ecology, Evolution, and Systematics 37:637-669.

Pinheiro J., D. Bates, S. DebRoy, and D. Sarkar 2015. nlme: Linear and Nonlinear Mixed Effects Models. R package version 3.1-122. Available online at https://cran.r-project.org/web/packages/nlme/index.html (Accessed 15 January 2014).

Potapov, P., A. Yaroshenko, S. Turubanova, M. Dubinin, L. Laestadius, C. Thies, , A. Aksenov, D. Egorov, Y. Yesipova, I. Glushkov, M. Karpachevskiy, A. Kostikova, A. Manisha, E. Tsybikova, and I. Zhuravleva. 2008. Mapping the world's intact forest landscapes by remote sensing. Ecology and Society 13:51.

Potter, K.A., H.A. Woods, and S. Pincebourde. 2013. Microclimatic challenges in global change biology. Global Change Biology 19:2932-2939.

Proe, M., J. Griffiths, and H. McKay. 2001. Effect of whole-tree harvesting on microclimate during establishment of second rotation forestry. Agricultural and Forest Meteorology 110:141-154.

R Core Team. 2018. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. Available online at https://www.R-project.org/ (accessed 23rd May 2018).

Rudel, T. K., O. T. Coomes, E. Moran, F. Achard, A. Angelsen, J. Xu, and E. Lambin. 2005. Forest transitions: towards a global understanding of land use change. Global Environmental Change 15:23-31.

Sears, M. W., E. Raskin, and M. J. Angilletta. 2011. The world is not flat: defining relevant thermal landscapes in the context of climate change. Integrative and Comparative Biology 51:666-675.

Spies, T. A. 2004. Ecological concepts and diversity of old-growth forests. Journal of Forestry 102:14-20.

Spies, T. A., and K. N. Johnson. 2007. Projecting forest policy and management effects across ownerships in coastal Oregon. Ecological Applications 17:3-4.

Swanson, M. E., J. F. Franklin, R. L. Beschta, C. M. Crisafulli, D. A. DellaSala, R. L. Hutto, D. B. Lindenmayer, and F. J. Swanson. 2011. The forgotten stage of forest succession: early-successional ecosystems on forest sites. Frontiers in Ecology and the Environment 9:117-125.

Swanson, M. E., N. M. Studevant, J. L. Campbell, and D. C. Donato. 2014. Biological associates of early-seral pre-forest in the Pacific Northwest. Forest Ecology and Management 324:160-171.

Taylor, G. H., and C. Hannan. 1999. The Climate of Oregon: From Rain Forest to Desert. Oregon State University Press, Corvallis.

Thomas, J. W., J. F. Franklin, J. Gordon, and K. N. Johnson. 2006. The Northwest Forest Plan: origins, components, implementation experience, and suggestions for change. Conservation Biology 20:277-287.

Vasseur, D. A., J. P. DeLong, B. Gilbert, H. S. Greig, C. D. G. Harley, K. S. McCann, V. Savage, T. D. Tunney, and M. I. O'Connor. 2014. Increased temperature variation poses a greater risk to species than climate warming. Proceedings of The Royal Society B 281:2013.2612.

Von Arx, G., M. Dobbertin, and M. Rebetez. 2012. Spatio-temporal effects of forest canopy on understory microclimate in a long-term experiment in Switzerland. Agricultural and Forest Meteorology 166:144-155.

Waring, R. H., and J. F. Franklin. 1979. Evergreen coniferous forests of the Pacific Northwest. Science 204:1380-1386.

Warnes, G. R, B. Bolker, T. Lumley, and R. C. Johnson. 2013. gmodels: Various R Programming Tools for Model Fitting. R package version 2.15.4.1. Available online at http://cran.r-project.org/web/packages/gmodels/index.html (Accessed 15 January 2014).

Whiteman, C. D., S. Zhong, W. J. Shaw, J. M. Hubbe, X. Bian, and J. Mittelstadt. 2001. Cold pools in the Columbia Basin. Weather and Forecasting 16:432-447.

Xu, M., J. Chen, and B. L. Brookshire. 1997. Temperature and its variability in oak forests in the southeastern Missouri Ozarks. Climate Research 8:209-223.

Zhao, K., and R. B. Jackson. 2014. Biophysical forcings of land-use changes from potential forestry activities in North America. Ecological Monographs 84:329-353.

Zurr, A. F., E. N. Ieno, N. J. Walker, A. A. Saveliev, and G. M. Smith. 2007. Mixed effects models and extensions with R. Springer Science+Business Media LLC, New York, NY.