

¹ **Cross-scale interaction of host tree size and climate governs bark
2 beetle-induced tree mortality**

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²⁰ **Abstract**

²¹ The Californian hot drought of 2012 to 2015 created favorable conditions for unprecedented ponderosa pine
²² (*Pinus ponderosa*) mortality in the Sierra Nevada mountain range, largely attributable to the western pine
²³ beetle (*Dendroctonus brevicomis*; WPB). Climate conditions and forest density may interact to affect tree
²⁴ mortality, but density is a coarse gauge of forest structure that can affect WPB behavior in a number of
²⁵ ways. Measuring broad-scale climate conditions simultaneously with local forest composition and structure–
²⁶ the spatial distribution and size of trees– will refine our understanding of how these variables interact, but
²⁷ is generally expensive and/or labor-intensive. We use drone surveys over 32 distinct sites along a 350-km
²⁸ latitudinal and 1000-m elevational gradient in western slope Sierra Nevada ponderosa pine/mixed-conifer

29 forests and structure from motion (SfM) processing to segment and classify more than 450,000 trees over
30 9 km² of forest with WPB-induced tree mortality. We validated the segmentation and classification with
31 data from 160 coincident field plots (each 0.041 ha in area) throughout the 32 sites, with the caveat that
32 dead trees were all classified as ponderosa pine. We modeled the probability of ponderosa pine mortality as a
33 function of forest structure and composition and their interaction with site-level climatic water deficit (CWD),
34 accounting for spatial covariance using exact Gaussian processes. A greater local proportion of host trees
35 strongly increased the probability of host mortality, with greater host density amplifying this effect. Further,
36 we found a strong interaction between host size and CWD such that larger trees increased the probability of
37 host mortality at hot/dry sites, but smaller trees tended to drive mortality in cool/wet sites. Our results
38 demonstrate a variable response of WPB to local forest structure and composition across an environmental
39 gradient, which may help reconcile differences between observed ecosystem-wide tree mortality patterns and
40 predictions from models based on coarser-scale forest structure. Climate change adaptation strategies should
41 consider that future disturbance outcomes may depend on interactions between local forest structure and
42 broad-scale environmental gradients, with the potential for cross-scale interactions that challenge our current
43 understanding of forest insect dynamics.

44 Introduction

45 Bark beetles dealt the final blow to many of the nearly 150 million trees killed in the California hot drought
46 of 2012 to 2015 and its aftermath (USDAFS 2019). A harbinger of climate change effects to come, record
47 high temperatures exacerbated the drought (Griffin and Anchukaitis 2014), which increased water stress in
48 trees (Asner et al. 2016), making them more susceptible to colonization by bark beetles (Fettig 2012, Kolb
49 et al. 2016). Further, a century of fire suppression policy has enabled forests to grow into dense stands,
50 which can also makes them more vulnerable to bark beetles (Fettig 2012). This combination of environmental
51 conditions and forest structural characteristics led to tree mortality events of unprecedented size in the
52 driest, densest forests across the state (Young et al. 2017). The mechanisms underlying the link between
53 tree susceptibility to colonization by insects and hot, dry conditions are often directly attributed to tree
54 physiology (Bentz et al. 2010, Kolb et al. 2016), while the link to forest density is multifaceted (Fettig 2012).
55 Because forest density is a coarse metric of the forest features to which bark beetles respond (Raffa et al.
56 2008), our understanding of the connection between forest density and insect disturbance severity could
57 be enhanced with more finely-resolved measures of forest structure as well as explicit consideration of tree
58 species composition (Stephenson et al. 2019, Fettig et al. 2019). Finally, the challenge of simultaneously
59 measuring the effects of both local-scale forest features (such as structure and composition) and broad-scale

60 environmental conditions (such as climatic water deficit; CWD) on forest insect disturbance leaves their
61 interaction effect relatively underexplored (Seidl et al. 2016, Stephenson et al. 2019, Fettig et al. 2019).

62 The ponderosa pine/mixed-conifer forests in California's Sierra Nevada region are characterized by regular
63 bark beetle disturbances, primarily by the influence of western pine beetle (*Dendroctonus brevicomis*; WPB)
64 on its host ponderosa pine (*Pinus ponderosa*) (Fettig 2016). WPB is a primary bark beetle— its reproductive
65 success is contingent upon host tree mortality, which itself requires enough beetles to mass attack the host tree
66 and overwhelm its defenses (Raffa and Berryman 1983). This Allee effect creates a strong coupling between
67 beetle selection behavior of host trees and host tree susceptibility to colonization (Raffa and Berryman 1983,
68 Logan et al. 1998, Wallin and Raffa 2004). A key defense mechanism of conifers to bark beetle attack is to
69 flood beetle bore holes with resin, which physically expels colonizing beetles, can be toxic to the colonizers
70 and their fungi, and may interrupt beetle communication (Franceschi et al. 2005, Raffa et al. 2015). Under
71 normal conditions, weakened trees with compromised defenses are the most susceptible to colonization and
72 will be the main targets of primary bark beetles like WPB (Bentz et al. 2010, Boone et al. 2011, Raffa et al.
73 2015). Under severe water stress, many trees no longer have the resources available to mount a defense (Kolb
74 et al. 2016) and thus prolonged drought can often trigger increased bark beetle-induced tree mortality as
75 average tree vigor declines (Bentz et al. 2010). As the local population density of beetles increases due to
76 successful reproduction within spatially-aggregated weakened trees, as might occur during drought, mass
77 attacks grow in size and become capable of overwhelming formidable tree defenses such that even healthy
78 trees may be susceptible to colonization and mortality (Bentz et al. 2010, Boone et al. 2011, Raffa et al.
79 2015). Thus, water stress can be a key determinant of whether individual trees are susceptible to bark beetles
80 under many conditions, and this environmental condition may interact with beetle population dynamics to
81 drive tree susceptibility under extreme conditions (Bentz et al. 2010, Stephenson et al. 2019).

82 WPB activity is strongly influenced by forest structure— the spatial distribution and size of trees— and tree
83 species composition. Taking forest structure alone, high-density forests are more prone to bark beetle-induced
84 tree mortality (Fettig 2012) which may arise as greater competition for water resources amongst crowded trees
85 and thus average tree resistance is lower (Hayes et al. 2009), or because smaller gaps between trees protect
86 pheromone plumes from dissipation by the wind and thus enhance intraspecific beetle communication (Thistle
87 et al. 2004). Tree size is another aspect of forest structure that affects bark beetle host selection behavior with
88 smaller trees tending to have lower capacity for resisting attack, and larger trees being more desirable targets
89 on account of their thicker phloem providing greater nutritional content (Chubaty et al. 2009, Boone et al.
90 2011, Graf et al. 2012). Taking forest composition alone, WPB activity in the Sierra Nevada mountain range
91 of California is necessarily tied to the regional distribution of its exclusive host, ponderosa pine (Fettig 2016).

92 Colonization by primary bark beetles can also depend on the relative frequencies of tree species in a more
93 local area, akin to reduced oligophagous insect herbivory in forests comprising taxonomically-distinct tree
94 species compared to monocultures (Jactel and Brockerhoff 2007). The interaction between forest structure
95 and composition also drives WPB activity. For instance, high-density forests with high host availability
96 may experience greater beetle-induced tree mortality because dispersal distances between potential host
97 trees are shorter reducing predation of adults searching for hosts and facilitating higher rates of colonization
98 (Miller and Keen 1960, Berryman 1982, Fettig et al. 2007) or because high host availability reduces the
99 chance of individual beetles wasting their limited resources flying to and landing on a non-host tree (Moeck
100 et al. 1981, Evenden et al. 2014). Stand-scale measures of forest structure and composition thus paint a
101 fundamentally limited picture of the mechanisms by which these forest characteristics affect bark beetle
102 disturbance, but finer-grain information explicitly recognizing tree size, tree species, and local tree density
103 should more appropriately capture the ecological processes underlying insect-induced tree mortality (Kaiser
104 et al. 2013). Additionally, considering the effects of local forest structure and composition with the effects of
105 environmental conditions may help refine our understanding of tree mortality patterns in widespread events
106 such as during the recent California hot drought.

107 The vast spatial extent of tree mortality in the 2012 to 2015 California hot drought challenges our ability to
108 simultaneously consider how broad-scale environmental conditions may interact with local forest structure
109 and composition to affect the dynamic between bark beetle selection and colonization of host trees, and host
110 tree susceptibility to attack (Anderegg et al. 2015, Stephenson et al. 2019). Measuring local forest structure
111 generally requires expensive instrumentation (Kane et al. 2014, Asner et al. 2016) or labor-intensive field
112 surveys (Larson and Churchill 2012, Stephenson et al. 2019, Fettig et al. 2019), which constrains survey
113 extent and frequency. Small, unhumanned aerial systems (sUAS) enable relatively fast and cheap remote
114 imaging over dozens of hectares of forest, which can be used to measure complex forest structure at the
115 individual tree scale (Morris et al. 2017, Shiklomanov et al. 2019). Distributing such surveys across an
116 environmental gradient can overcome the data acquisition challenge inherent in investigating phenomena
117 with both a strong local- and a strong broad-scale component.

118 We used ultra-high resolution, sUAS-derived remote sensing data over a network of 32 sites in Sierra Nevada
119 ponderosa pine/mixed-conifer forests spanning 1000 m of elevation and 350 km of latitude (see Fettig et
120 al. 2019) and covering a total of 9 km² to ask how broad-scale environmental conditions interacted with
121 local forest structure and composition to affect the probability of tree mortality during the cumulative tree
122 mortality event of 2012 to 2018. We asked:

- 123 1. How does the proportion of host trees in a local area and average host tree size affect WPB-induced

- 124 tree mortality?
- 125 2. How does the density of all tree species (hereafter “overall density”) affect WPB-induced tree mortality?
- 126 3. How does environmentally-driven tree moisture stress affect WPB-induced tree mortality?
- 127 4. Do the effects of forest structure, forest composition, and environmental condition interact to influence
- 128 WPB-induced tree mortality?

129 **Methods**

130 **Study system**

131 We built our study coincident with 160 vegetation/forest insect monitoring plots at 32 sites established
132 between 2016 and 2017 by Fettig et al. (2019) (Figure 1). The study sites were chosen to reflect typical
133 west-side Sierra Nevada yellow pine/mixed-conifer forests and were dominated by ponderosa pine (Fettig
134 et al. 2019). Plots were located in WPB-attacked, yellow pine/mixed-conifer forests across the Eldorado,
135 Stanislaus, Sierra and Sequoia National Forests and were stratified by elevation (914-1219 m, 1219-1524
136 m, 1524-1829 m above sea level). In the Sequoia National Forest, the southernmost National Forest in our
137 study, plots were stratified with the lowest elevation band of 1219-1524 m and extended to an upper elevation
138 band of 1829-2134 m to capture a more similar forest community composition as at the more northern
139 National Forests. The sites have variable forest structure and plot locations were selected in areas with >35%
140 ponderosa pine basal area and >10% ponderosa pine mortality. At each site, five 0.041 ha circular plots
141 were installed along transects with 80 to 200m between plots. In the field, Fettig et al. (2019) mapped all
142 stem locations relative to the center of each plot using azimuth/distance measurements. Tree identity to
143 species, tree height, and diameter at breast height (DBH) were recorded if DBH was greater than 6.35cm.
144 Year of mortality was estimated based on needle color and retention if it occurred prior to plot establishment,
145 and was directly observed thereafter during annual site visits. A small section of bark (approximately 625
146 cm²) on both north and south aspects was removed from dead trees to determine if bark beetle galleries
147 were present. The shape, distribution, and orientation of galleries are commonly used to distinguish among
148 bark beetle species (Fettig 2016). In some cases, deceased bark beetles were present beneath the bark to
149 supplement identifications based on gallery formation. During the spring and early summer of 2018, all field
150 plots were revisited to assess whether dead trees had fallen (Fettig et al. 2019).

151 In the typical life cycle of WPBs, females initiate host colonization by tunneling through the outer bark and
152 into the phloem and outer xylem where they rupture resin canals.

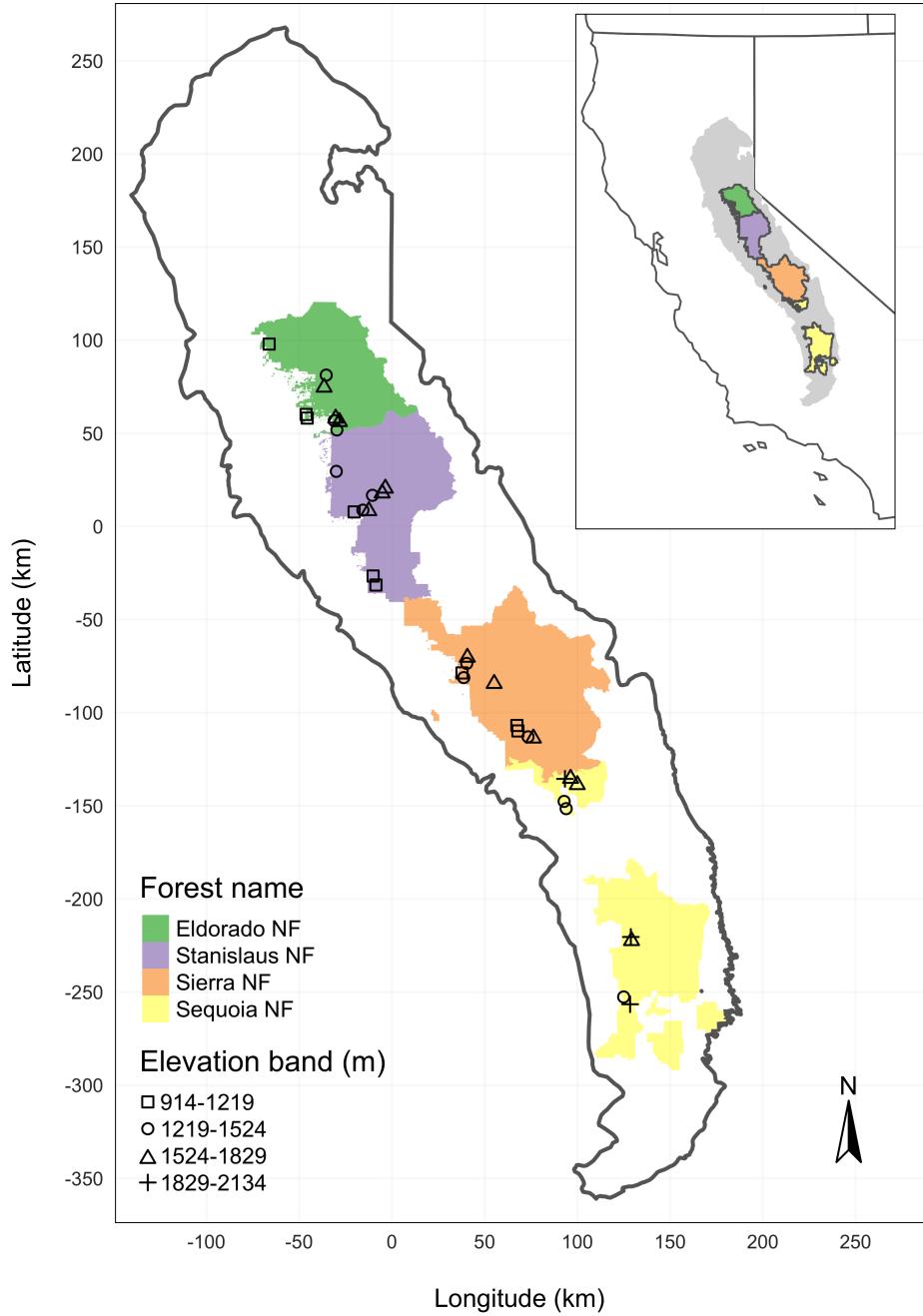


Figure 1: The network of field plots spanned a 350-km latitudinal gradient from the Eldorado National Forest in the north to the Sequoia National Forest in the south. Plots were stratified by three elevation bands in each forest, with the plots in the Sequoia National Forest (the southern-most National Forest) occupying elevation bands 305 m above the three bands in the other National Forests in order to capture a similar community composition.

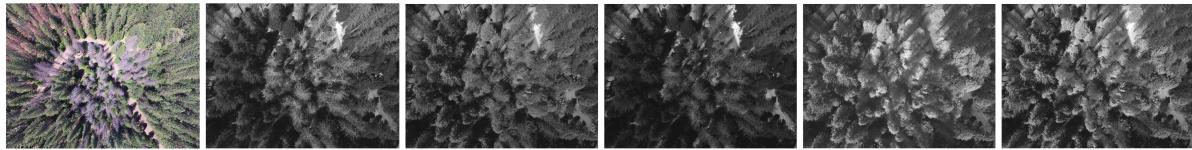
153 As a result, oleoresin exudes and collects on the bark surface, as is commonly observed with other bark beetle
154 species. During the early stages of attack, females release an aggregation pheromone component which, in
155 combination with host monoterpenes released from pitch tubes, is attractive to conspecifics (Bedard et al.
156 1969). An antiaggregation pheromone component is produced during latter stages of host colonization by
157 several pathways, and is thought to reduce intraspecific competition by altering adult behavior to minimize
158 overcrowding of developing brood within the host (Byers and Wood 1980). Volatiles from several nonhosts
159 sympatric with ponderosa pine have been demonstrated to inhibit attraction of WPB to its aggregation
160 pheromones (Shepherd et al. 2007, Fettig and Hilszczański 2015). In California, WPB generally has 2-3
161 generations in a single year and can often out-compete other primary bark beetles such as the mountain pine
162 beetle (*Dendroctonus ponderosae*), in ponderosa pines, especially in larger trees (Miller and Keen 1960).

163 **Aerial data collection and processing**

164 Nadir-facing imagery was captured using a gimbal-stabilized DJI Zenmuse X3 broad-band red/green/blue
165 (RGB) camera (DJI 2015a) and a fixed-mounted Micasense Rededge3 multispectral camera with five narrow
166 bands (Micasense 2015) on a DJI Matrice 100 aircraft (DJI 2015b). Imagery was captured from both cameras
167 along preprogrammed aerial transects over ~40 hectares surrounding each of the 32 sites (each of these
168 containing five field plots) and was processed in a series of steps to yield local forest structure and composition
169 data suitable for our statistical analyses. All images were captured in 2018 during a 3-month period between
170 early April and early July, and thus our work represents a postmortem investigation into the drivers of
171 cumulative tree mortality through the course of the hot drought.

172 Following the call by Wyngaard et al. (2019), we establish “data product levels” to reflect the image processing
173 pipeline from raw imagery (Level 0) to calibrated, fine-scale forest structure and composition information on
174 regular grids (Level 4), with each new data level derived from levels below it. Here, we outline the steps in the
175 processing and calibration pipeline visualized in Figure 2, and include additional details in the Supplemental
176 Information.

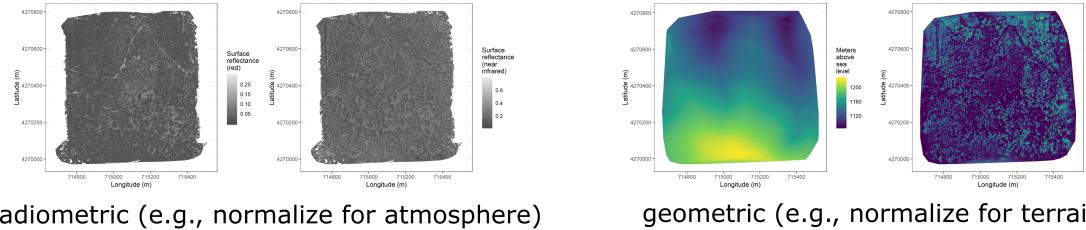
Level 0: raw data from sensors



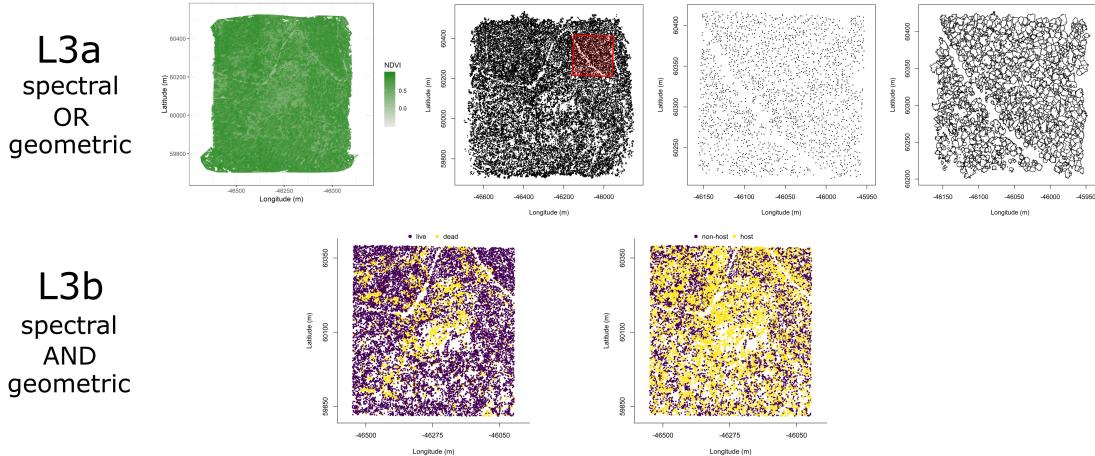
Level 1: basic outputs from photogrammetric processing



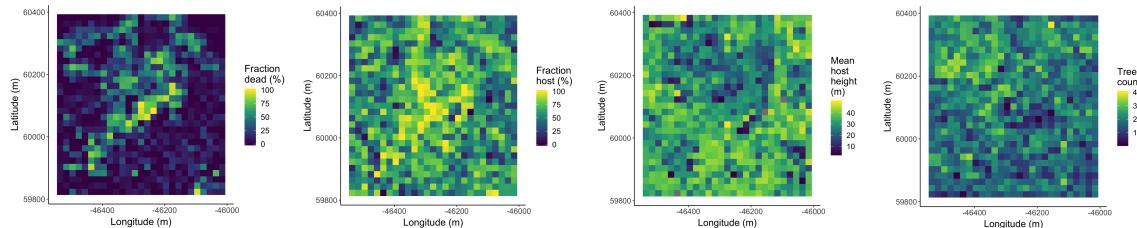
Level 2: corrected outputs from photogrammetric processing



Level 3: domain-specific information extraction



Level 4: aggregations to regular grids



178 Figure 2. Schematic of the data processing workflow for a single site with each new data product level derived
179 from data at lower levels.

180 Level 0 represents raw data from the sensors. From left to right: example broad-band RGB photo from
181 DJI Zenmuse X3 camera, example blue photo from Rededge3 (centered on 475nm), example green photo
182 from Rededge3 (centered on 560nm), example red photo from Rededge3 (centered on 668nm), example near
183 infrared photo from Rededge3 (centered on 840nm), and example red edge photo from Rededge3 (centered on
184 717nm).

185 Level 1 represents basic outputs from the photogrammetric workflow, in this case implemented with
186 Pix4Dmapper. From left to right: a dense point cloud visualized in CloudCompare (<https://www.danielgm.net/cc/>), an orthophoto generated from the RGB camera, and a digital surface model representing the
187 altitude above sea level (ground height + vegetation height) for every cell.
188

189 Level 2 represents outputs from photogrammetric processing that have been corrected radiometrically or
190 geometrically. From left to right: a radiometrically-corrected surface reflectance map of the red narrow band
191 from the Rededge3 camera, a radiometrically-corrected surface reflectance map of the near infrared narrow
192 band from the Rededge3 camera, a rasterized version of the digital terrain model derived by a geometric
193 correction of the dense point cloud, and a canopy height model derived by subtracting the terrain height
194 from the digital surface model.

195 Level 3 represents domain-specific information extraction from Level 2 products and is divided into two
196 sub-levels. Level 3a products are derived using only spectral or only geometric data. From left to right: a
197 reflectance map of Normalized Difference Vegetation Index (NDVI; Rouse et al. (1973)) derived using the red
198 and near infrared Level 2 reflectance products, a map of points representing detected trees from the canopy
199 height model with a red polygon highlighting the area presented in more detail for the next two images, a
200 close-up of points representing detected trees, and a close-up of polygons representing segmented tree crowns.
201 Level 3b products are derived using both spectral and geometric data. From left to right: a map of the point
202 locations of detected trees that have been classified as alive or dead based on the pixel values within each
203 segmented tree crown and a map of the point locations of detected trees classified to WPB host/non-host
204 using the same spectral information. Note that our study relies on the generation of Level 3a products in
205 order to combine them and create Level 3b products, but this need not be the case. For instance, deep
206 learning/neural net methods may be able to use both the spectral and geometric information from Level 2
207 simultaneously to locate and classify trees in a scene and directly generate Level 3b products without a need
208 to first generate the Level 3a products shown in this schematic (Weinstein et al. 2019, dos Santos et al. 2019).

209 Level 4 represents aggregations of Level 3 products to regular grids which might better reflect the grain size
210 of the data for which we have the best calibration and thus the most confidence or which might provide
211 new information not possible at an individual-tree level (e.g., average distance between trees in a small
212 neighborhood). From left to right: aggregation of live/dead classified trees as fraction of dead trees in a 20 x
213 20-m cell, aggregation of host/non-host classified trees as fraction of hosts in a 20 x 20-m cell, aggregation of
214 mean host height in a 20 x 20-m cell, and aggregation of tree count (including all species), in a 20 x 20-m
215 cell. In our case, the 20 x 20-m aggregation produces a grid cell with an area of 400 m², which most closely
216 matches the 404-m² area of the ground-based vegetation plots whose data we used in an aggregated form to
217 calibrate our derivation of Level 3 products.

218 **Level 0: Raw data from sensors**

219 Raw data comprised approximately 1900 images per camera lens (one broad-band RGB lens and five narrow-
220 band multispectral lenses) for each of the 32 sites (Figure 2; Level 0). Prior to the aerial survey, two strips of
221 bright orange drop cloth (~100 x 15 cm) were positioned as an “X” over the permanent monuments marking
222 the center of the 5 field plots from Fettig et al. (2019) (see Supplemental Information).

223 We preprogrammed north-south aerial transects using Map Pilot for DJI on iOS flight software (Drones-
224 MadeEasy 2018) at an altitude of 120 m above ground level (with “ground” defined using a 1-arc-second
225 digital elevation model (Farr et al. 2007)). The resulting ground sampling distance was approximately 5
226 cm/px for the Zenmuse X3 RGB camera and approximately 8 cm/px for the Rededge3 multispectral camera.
227 We used 91.6% image overlap (both forward and side) at the ground for the Zenmuse X3 RGB camera and
228 83.9% overlap (forward and side) for the Rededge3 multispectral camera.

229 **Level 1: Basic outputs from photogrammetric processing**

230 We used SfM photogrammetry implemented in Pix4Dmapper Cloud (www.pix4d.com) to generate dense point
231 clouds (Figure 2; Level 1, left), orthophotos (Figure 2; Level 1, center), and digital surface models (Figure 2;
232 Level 1, right) for each field site (Frey et al. 2018). For 29 sites, we processed the Rededge3 multispectral
233 imagery alone to generate these products. For three sites, we processed the RGB and the multispectral
234 imagery together to enhance the point density of the dense point cloud. All SfM projects resulted in a single
235 processing “block,” indicating that all images in the project were optimized and processed together. The
236 dense point cloud represents x, y, and z coordinates as well as the color of millions of points per site. The
237 orthophoto represents a radiometrically uncalibrated, top-down view of the survey site that preserves the

238 relative x-y positions of objects in the scene. The digital surface model is a rasterized version of the dense
239 point cloud that shows the altitude above sea level for each pixel in the scene at the ground sampling distance
240 of the camera that generated the Level 0 data.

241 **Level 2: Corrected outputs from photogrammetric processing**

242 **Radiometric corrections**

243 A radiometrically-corrected reflectance map (Figure 2; Level 2, left two figures; i.e., a corrected version of the
244 Level 1 orthophoto) was generated using the Pix4D software by incorporating incoming light conditions for
245 each narrow band of the Rededge3 camera (captured simultaneously with the Rededge3 camera using an
246 integrated downwelling light sensor) as well as a pre-flight image of a calibration panel of known reflectance
247 (see Supplemental Information for camera and calibration panel details).

248 **Geometric corrections**

249 We implemented a geometric correction to the Level 1 dense point cloud and digital surface model by
250 normalizing these data for the terrain underneath the vegetation. We generated the digital terrain model
251 representing the ground underneath the vegetation at 1-m resolution (Figure 2; Level 2, third image) by
252 classifying each survey area's dense point cloud into "ground" and "non-ground" points using a cloth simulation
253 filter algorithm (Zhang et al. 2016) implemented in the `lidR` (Roussel et al. 2019) package and rasterizing
254 the ground points using the `raster` package (Hijmans et al. 2019). We generated a canopy height model
255 (Figure 2; Level 2, fourth image) by subtracting the digital terrain model from the digital surface model.

256 **Level 3: Domain-specific information extraction**

257 **Level 3a: Data derived from spectral OR geometric Level 2 product**

258 Using just the spectral information from the radiometrically-corrected reflectance maps, we calculated several
259 vegetation indices including the normalized difference vegetation index (NDVI; Rouse et al. (1973); Figure
260 2; Level 3a, first image), the normalized difference red edge (NDRE; Gitelson and Merzlyak (1994)), the
261 red-green index (RGI; Coops et al. (2006)), the red edge chlorophyll index ($CI_{red\ edge}$; Clevers and Gitelson
262 (2013)), and the green chlorophyll index (CI_{green} ; Clevers and Gitelson (2013)).

Table 1: Algorithm name, number of parameter sets tested for each algorithm, and references.

Algorithm	Parameter sets tested	Reference(s)
li2012	131	Li et al. (2012); Jakubowski et al. (2013); Shin et al. (2018)
lmfx	30	Roussel (2019)
localMaxima	6	Roussel et al. (2019)
multichm	1	Eysn et al. (2015)
ptrees	3	Vega et al. (2014)
vwf	3	Plowright (2018)
watershed	3	Pau et al. (2010)

263 Using just the geometric information from the canopy height model or terrain-normalized dense point cloud,
 264 we generated maps of detected trees (Figure 2; Level 3a, second and third images) by testing a total of 7
 265 automatic tree detection algorithms and a total of 177 parameter sets (Table 1). We used the field plot data
 266 to assess each tree detection algorithm/parameter set by converting the distance-from-center and azimuth
 267 measurements of the trees in the field plots to x-y positions relative to the field plot centers distinguishable in
 268 the Level 2 reflectance maps as the orange fabric X’s that we laid out prior to each flight. In the reflectance
 269 maps, we located 220 out of 160 field plot centers while some plot centers were obscured due to dense
 270 interlocking tree crowns or because a plot center was located directly under a single tree crown. For each of
 271 the 220 field plots with identifiable plot centers— the “validation field plots”, we calculated 7 forest structure
 272 metrics using the ground data collected by Fettig et al. (2019): total number of trees, number of trees greater
 273 than 15 m in height, mean height of trees, 25th percentile tree height, 75th percentile tree height, mean
 274 distance to nearest tree neighbor, and mean distance to second nearest neighbor. For each tree detection
 275 algorithm and parameter set described above, we calculated the same set of 7 structure metrics within the
 276 footprint of the validation field plots. We calculated the Pearson’s correlation and root mean square error
 277 (RMSE) between the ground data and the aerial data for each of the 7 structure metrics for each of the 177
 278 automatic tree detection algorithms/parameter sets. For each algorithm and parameter set, we calculated its
 279 performance relative to other algorithms as whether its Pearson’s correlation was within 5% of the highest
 280 Pearson’s correlation as well as whether its RMSE was within 5% of the lowest RMSE. We summed the
 281 number of forest structure metrics for which it reached these 5% thresholds for each algorithm/parameter
 282 set. For automatically detecting trees across the whole study, we selected the algorithm/parameter set that

283 performed well across the most number of forest metrics (see Results).

284 We delineated individual tree crowns (Figure 2; Level 3a, fourth image) with a marker controlled watershed
285 segmentation algorithm (Meyer and Beucher 1990) implemented in the **ForestTools** package (Plowright
286 2018) using the detected treetops as markers. If the automatic segmentation algorithm failed to generate
287 a crown segment for a detected tree (e.g., often snags with a very small crown footprint), a circular crown
288 was generated with a radius of 0.5 m. If the segmentation generated multiple polygons for a single detected
289 tree, only the polygon containing the detected tree was retained. Because image overlap decreases near the
290 edges of the overall flight path and reduces the quality of the SfM processing in those areas, we excluded
291 segmented crowns within 35 m of the edge of the survey area. Given the narrower field of view of the
292 Rededge3 multispectral camera versus the X3 RGB camera whose optical parameters were used to define the
293 ~40 hectare survey area around each site, as well as the 35 m additional buffering, the survey area at each
294 site was ~30 ha (see Supplemental Information).

295 **Level 3b: Data derived from spectral AND geometric information**

296 We overlaid the segmented crowns on the reflectance maps from 20 sites spanning the latitudinal and elevation
297 gradient in the study. Using QGIS (<https://qgis.org/en/site/>), we hand classified 564 trees as live/dead
298 (Figure 3) and as one of 5 dominant species in the study area (ponderosa pine, *Pinus lambertiana*, *Abies*
299 *concolor*, *Calocedrus decurrens*, or *Quercus kelloggii*) using the mapped ground data as a guide. Each tree was
300 further classified as “host” for ponderosa pine or “non-host” for all other species (Fettig 2016). We extracted
301 all the pixel values within each segmented crown polygon from the five, Level 2 orthorectified reflectance
302 maps (one per narrow band on the Rededge3 camera) as well as from the five, Level 3a vegetation index
303 maps using the **velox** package (Hunziker 2017). For each crown polygon, we calculated the mean value of the
304 extracted Level 2 and Level 3a pixels and used them as ten independent variables in a five-fold cross validated
305 boosted logistic regression model to predict whether the hand classified trees were alive or dead. For just the
306 living trees, we similarly used all 10 mean reflectance values per crown polygon to predict tree species using a
307 five-fold cross validated regularized discriminant analysis. The boosted logistic regression and regularized
308 discriminant analysis were implemented using the **caret** package in R (Kuhn 2008). We used these models to
309 classify all tree crowns in the data set as alive or dead (Figure 2; Level 3b, first image) as well as the species
310 of living trees (Figure 2; Level 3b, second image). Finally, we estimated the basal area of each tree from their
311 photogrammetry-derived height using species-specific simple linear regressions of the relationship between
312 height and diameter at breast height as measured in the coincident field plots from Fettig et al. (2019).

313 Level 4: Aggregations to regular grids

314 We rasterized the forest structure and composition data at a spatial resolution similar to that of the field
315 plots to better match the grain size at which we validated the automatic tree detection algorithms. In each
316 raster cell, we calculated: number of dead trees, number of ponderosa pine trees, total number of trees, and
317 mean height of ponderosa pine trees. The values of these variables in each grid cell and derivatives from
318 them were used for visualization and modeling. Here, we show the fraction of dead trees per cell (Figure 2;
319 Level 4, first image), the fraction of host trees per cell (Figure 2; Level 4, second image), the mean height of
320 ponderosa pine trees in each cell (Figure 2; Level 4, third image), and the total count of trees per cell (Figure
321 2; Level 4, fourth image).

322 Note on assumptions about dead trees

323 For the purposes of this study, we assumed that all dead trees were ponderosa pine and thus hosts colonized
324 by WPB. This is a reasonably good assumption for our study area; for example, Fettig et al. (2019) found
325 that 73.4% of dead trees in their coincident field plots were ponderosa pine. Mortality was concentrated in
326 the larger-diameter classes and attributed primarily to WPB (see Figure 5 of Fettig et al. 2019). The species
327 contributing to the next highest proportion of dead trees was incense cedar which represented 18.72% of the
328 dead trees in the field plots. While the detected mortality is most likely to be ponderosa pine killed by WPB,
329 it is critical to interpret our results with these limitations in mind.

330 Environmental data

331 We used CWD (Stephenson 1998) from the 1981-2010 mean value of the basin characterization model (Flint
332 et al. 2013) as an integrated measure of temperature and moisture conditions for each of the 32 sites. Higher
333 values of CWD correspond to hotter, drier conditions and lower values correspond to cooler, wetter conditions.
334 CWD has been shown to correlate well with broad patterns of tree mortality in the Sierra Nevada (Young
335 et al. 2017) as well as bark beetle-induced tree mortality (Millar et al. 2012). The forests along the entire
336 CWD gradient used in this study experienced exceptional hot drought between 2012 to 2015 (Griffin and
337 Anchukaitis 2014, Robeson 2015). We converted the CWD value for each site into a z-score representing that
338 site's deviation from the mean CWD across the climatic range of Sierra Nevada ponderosa pine as determined
339 from 179 herbarium records described in Baldwin et al. (2017). Thus, a CWD z-score of 1 would indicate
340 that the CWD at that site is one standard deviation hotter/drier than the mean CWD across all geolocated
341 herbarium records for ponderosa pine in the Sierra Nevada.

342 **Statistical model**

343 We used a generalized linear model with a zero-inflated binomial response and a logit link to predict the
 344 probability of ponderosa pine mortality within each 20 x 20-m cell using the total number of ponderosa
 345 pine trees in each cell as the number of trials, and the number of dead trees in each cell as the number of
 346 “successes”. As covariates, we used the proportion of trees that are WPB hosts (i.e., ponderosa pine) in each
 347 cell, the mean height of ponderosa pine trees in each cell, the count of trees of all species (overall density) in
 348 each cell, and the site-level CWD using Eq. 1. Note that the two-way interaction between the overall density
 349 and the proportion of trees that are hosts is directly proportional to the number of ponderosa pine trees in
 350 the cell. We centered and scaled all predictor values, and used weakly-regularizing default priors from the
 351 `brms` package (Bürkner 2017). To measure and account for spatial autocorrelation underlying ponderosa pine
 352 mortality, we subsampled the data at each site to a random selection of 200, 20 x 20-m cells representing
 353 approximately 27.5% of the surveyed area. Additionally with these subsampled data, we included a separate
 354 exact Gaussian process term per site of the noncentered/nonscaled interaction between the x- and y-position
 355 of each cell using the `gp()` function in the `brms` package (Bürkner 2017). The Gaussian process estimates the
 356 spatial covariance in the response variable (log-odds of ponderosa pine mortality) jointly with the effects of
 357 the other covariates.

$$y_{i,j} \sim \begin{cases} 0, & p \\ Binom(n_i, \pi_i), & 1 - p \end{cases}$$

$$\text{logit}(\pi_i) = \beta_0 +$$

$$\beta_1 X_{cwd,j} + \beta_2 X_{propHost,i} + \beta_3 X_{PipoHeight,i} +$$

$$\beta_4 X_{overallDensity,i} + \beta_5 X_{overallBA,i} +$$

$$\beta_6 X_{cwd,j} X_{PipoHeight,i} + \beta_7 X_{cwd,j} X_{propHost,i} +$$

$$\beta_8 X_{cwd,j} X_{overallDensity,i} + \beta_9 X_{cwd,j} X_{overallBA,i} +$$

$$\beta_{10} X_{propHost,i} X_{PipoHeight,i} + \beta_{11} X_{propHost,i} X_{overallDensity,i} +$$

$$\beta_{12} X_{cwd,j} X_{propHost,i} X_{PipoHeight,i} +$$

$$\mathcal{GP}_j(x_i, y_i)$$

358 Where y_i is the number of dead trees in cell i , n_i is the sum of the dead trees (assumed to be ponderosa pine)
 359 and live ponderosa pine trees in cell i , π_i is the probability of ponderosa pine tree mortality in cell i , p is
 360 the probability of there being zero dead trees in a cell arising as a result of an unmodeled process, $X_{cwd,j}$

361 is the z-score of CWD for site j , $X_{propHost,i}$ is the scaled proportion of trees that are ponderosa pine in
362 cell i , $X_{PipoHeight,i}$ is the scaled mean height of ponderosa pine trees in cell i , $X_{overallDensity,i}$ is the scaled
363 density of all trees in cell i , $X_{overallBA,i}$ is the scaled basal area of all trees in cell i , x_i and y_i are the x- and
364 y- coordinates of the centroid of the cell in an EPSG3310 coordinate reference system, and \mathcal{GP}_j represents
365 the exact Gaussian process describing the spatial covariance between cells at site j .

366 We fit this model using the **brms** package (Bürkner 2017) which implements the No U-Turn Sampler extension
367 to the Hamiltonian Monte Carlo algorithm (Hoffman and Gelman 2014) in the Stan programming language
368 (Carpenter et al. 2017). We used 4 chains with 4000 iterations each (2000 warmup, 2000 samples), and
369 confirmed chain convergence by ensuring all **Rhat** values were less than 1.1 (Brooks and Gelman 1998) and
370 that the bulk and tail effective sample sizes (ESS) for each estimated parameter were greater than 100 times
371 the number of chains (i.e., greater than 400 in our case). We used posterior predictive checks to visually
372 confirm model performance by overlaying the density curves of the predicted number of dead trees per cell
373 over the observed number (Gabry et al. 2019). For the posterior predictive checks, we used 50 random
374 samples from the model fit to generate 50 density curves and ensured curves were centered on the observed
375 distribution, paying special attention to model performance at capturing counts of zero.

376 Software and data availability

377 All data are available via the Open Science Framework. Statistical analyses were performed using the **brms**
378 packages. With the exception of the SfM software (Pix4Dmapper Cloud) and the GIS software QGIS, all
379 data carpentry and analyses were performed using R (R Core Team 2018).

380 Results

381 Tree detection algorithm performance

382 We found that the experimental **lmf** algorithm with parameter values of **dist2d** = 1 and **ws** = 2.5 (Roussel
383 et al. 2019) performed the best across 7 measures of forest structure as measured by Pearson's correlation
384 with ground data (Table 2).

Table 2: Correlation and differences between the best performing tree detection algorithm (lmfx with dist2d = 1 and ws = 2.5) and the ground data. An asterisk next to the correlation or RMSE indicates that this value was within 5% of the value of the best-performing algorithm/parameter set. Ground mean represents the mean value of the forest metric across the 220 field plots that were visible from the sUAS-derived imagery. The median error is calculated as the median of the differences between the air and ground values for the 220 visible plots. Thus, a positive number indicates an overestimate by the sUAS workflow and a negative number indicates an underestimate.

Forest structure metric	Ground mean	Correlation with ground	RMSE	Median error
total tree count	19	0.67*	8.68*	2
count of trees > 15 m	9.9	0.43	7.38	0
distance to 1st neighbor (m)	2.8	0.55*	1.16*	0.26
distance to 2nd neighbor (m)	4.3	0.61*	1.70*	0.12
height (m); 25 th percentile	12	0.16	8.46	-1.2
height (m); mean	18	0.29	7.81*	-2.3
height (m); 75 th percentile	25	0.35	10.33*	-4

385 Classification accuracy for live/dead and host/non-host

386 The accuracy of live/dead classification on a withheld test dataset was 96.4%. The accuracy of species
 387 classification on a withheld testing dataset was 64.1%. The accuracy of WPB host/non-WPB-host (i.e.,
 388 ponderosa pine versus other tree species) on a withheld testing dataset was 71.8%.

389 Site summary based on best tree detection algorithm and classification

390 Across all study sites, we detected, segmented, and classified 452,413 trees. Of these trees, we classified
 391 118,879 as dead (26.3% mortality). Estimated site-level tree mortality ranged from 6.8% to 53.6%. See
 392 Supplemental Information for site summaries and comparisons to site-level mortality measured from field
 393 data.

394 Effect of local structure and regional climate on tree mortality attributed to western pine 395 beetle

396 We detected a positive main effect of CWD on the probability of ponderosa pine mortality within each 20 x
 397 20-m cell (Figure 4). We found a positive main effect of proportion of host trees per cell (effect size: 0.76;
 398 95% CI: [0.70, 0.82]), with a greater proportion of host trees (i.e., ponderosa pine) in a cell increasing the

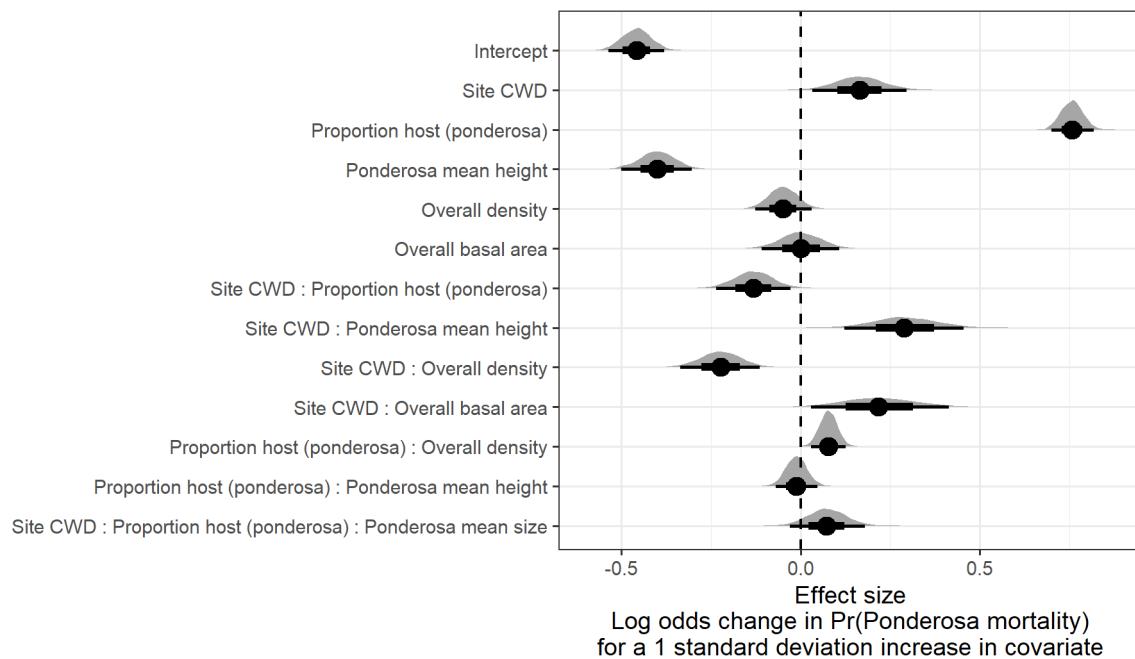


Figure 4: Posterior distributions of effect size from zero-inflated binomial model predicting the probability of ponderosa pine mortality in a 20 x 20-m cell given forest structure characteristics and site-level climatic water deficit (CWD). The gray filled area for each model covariate represents the probability density of the posterior distribution, the point underneath each density curve represents the median of the estimate, the bold interval surrounding the point estimate represents the 66% credible interval, and the thin interval surrounding the point estimate represents the 95% credible interval.

399 probability of ponderosa pine mortality. We detected no effect of overall tree density nor overall basal area
400 (i.e., including both ponderosa pine and non-host species; tree density effect size: -0.05; 95% CI: [-0.13, 0.03];
401 basal area effect size: 0.00; 95% CI: [-0.11, 0.11]).

402 We found a positive two-way interaction between the overall tree density per cell and the proportion of trees
403 that were hosts, which is equivalent to a positive effect of the density of host trees (effect size: 0.08; 95% CI:
404 [0.03, 0.13]; Figure 4).

405 We found a negative effect of mean height of ponderosa pine on the probability of ponderosa mortality,
406 suggesting that WPB attacked smaller trees, on average (effect size: -0.40; 95% CI: [-0.50, -0.30]). However,
407 there was a positive interaction between CWD and ponderosa pine mean height, such that larger trees were
408 more likely to increase the local probability of ponderosa mortality in hotter, drier sites (effect size: 0.29;
409 95% CI: [0.12, 0.46]; Figure 5).

410 We found weakly negative effects of the site-level CWD interactions with both the proportion of host trees
411 and overall tree density (CWD/proportion host interaction effect size: -0.13; 95% CI: [-0.23, -0.03]; Figure 4;
412 CWD/overall tree density interaction effect size: -0.22; 95% CI: [-0.34, -0.11]; Figure 4; Figure 5). We found
413 a positive effect of the interaction between CWD and total basal area (effect size: 0.22; 95% CI: [0.03, 0.42];
414 Figure 4; Figure 5).

415 Discussion

416 This study represents a novel use of drones to further our understanding of the simultaneous effects of
417 local forest structure and composition with broad-scale environmental gradients on tree mortality attributed
418 to WPB. We found strong positive effects (effect sizes >0.4) of both the proportion of host trees and the
419 interaction between site CWD and host tree mean size (height) on the probability of ponderosa pine mortality.
420 Conversely, we found a strong negative effect (effect size <-0.4) of mean height of ponderosa pine. Site-level
421 CWD exerted a positive, but relatively weak, main effect on the probability of ponderosa mortality (effect
422 size: 0.16; 95% CI: [0.03, 0.30]). To that end, we did not measure tree water stress at an individual tree
423 level as in other recent work (Stephenson et al. 2019), and instead treated CWD as a general indicator of
424 tree stress following results of coarser-scale studies (e.g., Young et al. 2017), which may have contributed to
425 our failure to detect a stronger CWD effect. Also, our entire study area experienced the same extreme hot
426 drought between 2012 and 2015 and the variation of mortality explained by a main effect of CWD may be
427 dampened when most trees are experiencing a high degree of water stress (Floyd et al. 2009, Fettig et al.
428 2019).

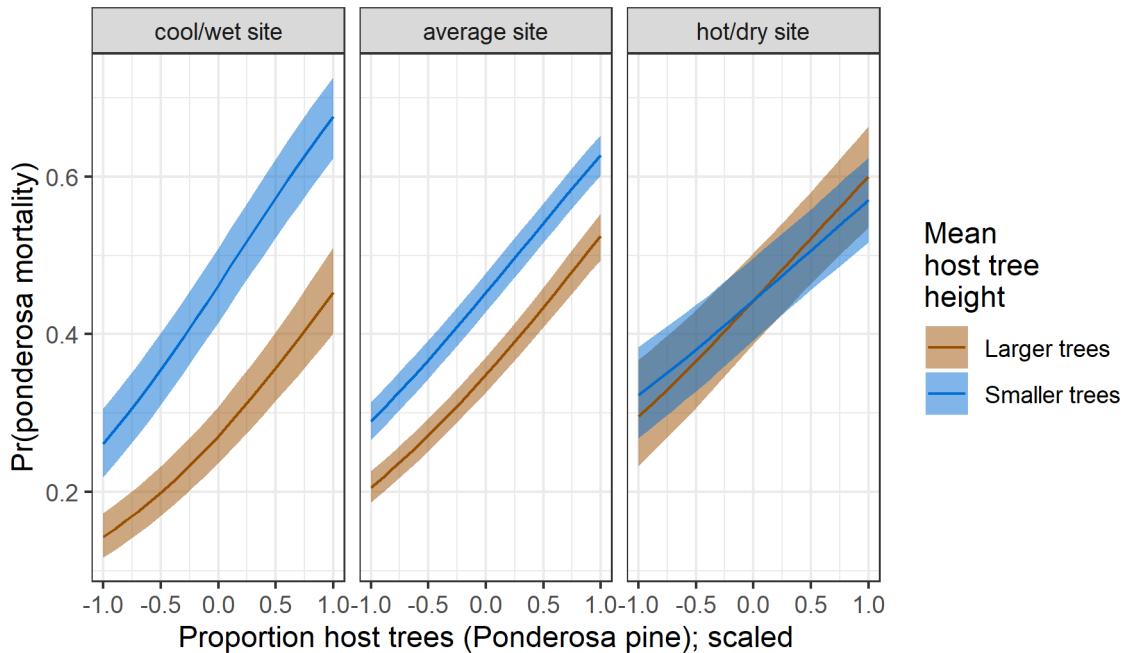


Figure 5: Line version of model results with 95% credible intervals showing primary influence of ponderosa pine structure on the probability of ponderosa pine mortality, and the interaction across climatic water deficit. The ‘larger trees’ line represents the mean height of ponderosa pine 0.7 standard deviations above the mean (approximately 24.1 m), and the ‘smaller trees’ line represents the mean height of ponderosa pine 0.7 standard deviations below the mean (approximately 12.1 m).

429 Positive effect of host density and a negative effect of overall density

430 A number of mechanisms associated with the relative abundance of species in a local area might underlie the
 431 strong effect of host proportion on the probability of host tree mortality. Frequency-dependent herbivory—
 432 whereby mixed-species forests experience less herbivory compared to monocultures (as an extreme example)—
 433 is common, especially for oligophagous insect species (Jactel and Brockerhoff 2007). Furthermore, it has been
 434 demonstrated that nonhost volatiles reduce attraction of several species of bark beetles to their aggregation
 435 pheromones (Seybold et al. 2018), including WPB (Fettig et al. 2005). To that end, combinations of nonhost
 436 volatiles and an antiaggregation pheromone have been used successfully to reduce levels of tree mortality
 437 attributed to WPB (e.g., Fettig et al. 2012). In general, Hayes et al. (2009) and Fettig et al. (2019) found
 438 that measures of host availability explained less variation in mortality than measures of overall tree density,
 439 but those conclusions were based on a response variable of “total number of dead host trees,” rather than
 440 the number of dead host trees conditional on the total number of host trees as in our study (i.e., a binomial
 441 response).

442 The negative relationship between overall tree density, a potential correlate of the local competitive envi-

443 ronment, and the probability of ponderosa pine mortality is counter-intuitive but corroborates findings of
444 coincident ground plots (Fettig et al. 2019, in their analysis using proportion of trees killed as a response)
445 and other work during the same hot drought (Restaino et al. 2019). The forest structure (in the absence of
446 management) is itself a product of climate and, with increasing importance at finer spatial scales, topographic
447 conditions (Fricker et al. 2019). Thus, the denser forest patches in our study may indicate greater local
448 water availability, more favorable conditions for tree growth and survivorship, and increased resistance to
449 beetle-induced mortality (Restaino et al. 2019). The negative two-way interaction between site CWD and
450 overall density that amplifies the negative overall density effect in hotter, drier sites (effect size: -0.22; 95%
451 CI: [-0.34, -0.11]) supports this explanation if greater local tree density implies especially favorable growing
452 conditions (and locally resistant trees) when denser patches are found in hot, dry sites.

453 The positive relationship between host density and susceptibility to colonization by bark beetles has been so
454 well-documented at the experimental plot level (e.g., Raffa and Berryman 1987, Oliver 1995) that lowering
455 stand densities through selective harvest of hosts is commonly recommended for reducing future levels of tree
456 mortality attributed to bark beetles (Fettig and Hilszczański 2015), including WPB (Fettig 2016). Greater
457 host density shortens the flight distance required for WPB to disperse to new hosts, which likely facilitates
458 bark beetle spread, however we calibrated our aerial tree detection to ~400 m² areas rather than to individual
459 tree locations, so our data are insufficient to address these relationships. Increased density of ponderosa pine,
460 specifically, may disproportionately increase the competitive environment for host trees (and thus increase
461 their susceptibility to WPB colonization) if intraspecific competition amongst ponderosa pine trees is stronger
462 than interspecific competition as would be predicted with coexistence theory (Chesson 2000). Finally, greater
463 host densities increase the frequency that searching WPB land on hosts, rather than nonhosts, thus reducing
464 the amount of energy expended during host finding and selection as well as the time that searching WPB
465 spend exposed to predators.

466 **Negative main effect of host tree mean size, but strong positive interaction with site CWD**

467 The negative main effect of host tree mean size was surprising, and appears to contradict long-standing
468 wisdom on the dynamics of western pine beetle in the Sierra Nevada. WPB exhibit a preference for trees 50.8
469 to 76.2 cm in diameter at breast height (Person 1928, 1931), and a positive relationship between host tree
470 size and levels of tree mortality attributed to WPB was reported by Fettig et al. (2019) in the coincident field
471 plots as well as in other recent studies (Restaino et al. 2019, Stephenson et al. 2019, Pile et al. 2019). Indeed,
472 Fettig et al. (2019) reported no mortality in ponderosa pine trees <10.0 cm DBH attributable to WPB and
473 found no tree size/mortality relationship for incense cedar or white fir in the coincident field plots. These

474 species represent 22.3% of the total tree mortality observed in their study, yet in our study all dead trees
475 were classified as ponderosa pine (see Methods) which could dampen positive effect of tree size on mortality.
476 Larger trees are more nutritious and are therefore ideal targets if local bark beetle density is high enough
477 to successfully initiate mass attack as can occur when many trees are under severe water stress (Bentz et
478 al. 2010, Kolb et al. 2016). In the recent hot drought, we expected that most trees would be under severe
479 water stress, setting the stage for increasing beetle density, successful mass attacks, and targeting of larger
480 trees. A possible explanation for our finding counter to this expectation is that our observations represent the
481 cumulative mortality of trees during a multi-year drought event and its aftermath. Lower host tree mean size
482 led to a greater probability of host mortality earlier in this drought (Pile et al. 2019, Stovall et al. 2019) and
483 that signal might have persisted even as mortality continued to accumulate driven by other factors. Finally,
484 tree growth rates may be a better predictor of susceptibility to WPB colonization than tree size per se, with
485 slower-growing trees being most vulnerable (Miller and Keen 1960). While slow-growing trees are often also
486 the largest trees, this may not be the case for our study sites especially given the legacy of fire exclusion
487 in the Sierra Nevada and its effect of perturbing forest structure far outside its natural range of variation
488 (Safford and Stevens 2017).

489 In hot, dry sites, larger average host size increased the probability of host mortality while smaller host sizes
490 increased the probability of host mortality in cooler, wetter sites. Notably, a similar pattern was shown
491 by Stovall et al. (2019) with a strong positive tree height/mortality relationship in areas with the greatest
492 vapor pressure deficit and no tree height/mortality relationship in areas with the lowest vapor pressure
493 deficit. Stovall et al. (2019) did not observe that this environmental dependence extended to a negative tree
494 height/mortality relationship (as we did) even at the lowest extremes of their vapor pressure deficit gradient,
495 perhaps because their entire study took place in the southern Sierra Nevada which represents a hotter, drier
496 portion of the more spatially extensive results we present here. Our work suggests that the WPB was cueing
497 into different aspects of forest structure across an environmental gradient in a spatial context in a parallel
498 manner to the temporal context noted by Stovall et al. (2019) and Pile et al. (2019), who observed that
499 mortality was increasingly driven by larger trees as the hot drought proceeded and became more severe.

500 All of our sites were considered in an “epidemic” population phase for WPB (>5 trees killed per hectare; see
501 Supplemental Information; Miller and Keen 1960, Hayes et al. 2009), but our results challenge the notion that
502 outbreak behavior by the WPB and subsequent tree mortality is always driven by greater tree size. Despite a
503 strong tree size/mortality relationship in coincident ground plots across our study area (Fettig et al. 2019),
504 our results from surveying the broader context surrounding those ground plots reveals different effects of
505 host tree size depending on CWD. Thus, it is possible that the massive tree mortality in hotter/drier Sierra

506 Nevada forests (lower latitudes and elevations; Asner et al. 2016, Young et al. 2017) during the 2012 to
507 2015 hot drought arose as a synergistic alignment of environmental conditions and local forest structure that
508 allowed WPB to successfully colonize large trees, rapidly increase in population size, and expand. It follows
509 that the unexpectedly low mortality in cooler/wetter Sierra Nevada forests compared to model predictions
510 based on coarser-scale forest structure data (Young et al. 2017) could be explained by a different WPB
511 response to local forest structure due to a lack of an alignment with favorable climate conditions.

512 **Limitations and future directions**

513 We have demonstrated that drones can be effective means of collecting forest data at multiple, vastly different
514 spatial scales to investigate a single, multi-scale phenomenon— from meters in between trees, to hundreds
515 of meters of elevation, to hundreds of thousands of meters of latitude. Some limitations remain but can be
516 overcome with further refinements in the use of this tool for forest ecology. Most of these limitations arise
517 from tree detection and classification uncertainty, and thus it was imperative to work with field data for
518 calibration and uncertainty reporting.

519 The greatest limitation in our study arising from classification uncertainty is in the assumption that all dead
520 trees were ponderosa pine, which we estimate from coincident field plots is true approximately 73.4% of
521 the time. Because the forest structure factors influencing the likelihood of individual tree mortality during
522 the hot drought depended on tree species (Stephenson et al. 2019), we cannot rule out that some of the
523 ponderosa pine mortality relationships to forest structure that we observed may be partially explained by
524 those relationships in other species that were misclassified as ponderosa pine using our methods. However,
525 the overall community composition across our study area was similar (Fettig et al. 2019) and we are able
526 to reproduce similar forest structure/mortality patterns in drone-derived data when restricting the scope
527 of analysis to only trees detected in the footprints of the coincident field plots with dramatically different
528 patterns observed when including data from the forest surrounding the coincident field plots (see Supplemental
529 information). Thus, we remain confident that the patterns we observed were driven primarily by the dynamic
530 between WPB and ponderosa pine. While spectral information of foliage could help classify living trees to
531 species, the species of standing dead trees were not spectrally distinct. This challenge of classifying standing
532 dead trees to species implies that a conifer forest system with less bark beetle and tree host diversity, such
533 as mountain pine beetle outbreaks in monocultures of lodgepole pine in the Intermountain West, should be
534 particularly amenable to the methods presented here even with minimal further refinement because dead
535 trees will almost certainly belong to a single species and have succumbed to colonization by a single bark

536 beetle species.

537 Some uncertainty surrounded our ability to detect trees using the geometry of the dense point clouds derived
538 with SfM. The horizontal accuracy of the tree detection was better than the vertical accuracy, which may
539 result from a more significant error contribution by the field-based calculations of tree height compared to
540 tree position relative to plot center (Table 2). Both the horizontal and vertical accuracy would likely improve
541 with better SfM point clouds, which can be enhanced with greater overlap between images (Frey et al. 2018)
542 or with oblique (i.e., off-nadir) imagery (James and Robson 2014). Frey et al. (2018) found that 95% overlap
543 was preferable for generating dense point clouds in forested areas, and James and Robson (2014) reduced
544 dense point cloud errors using imagery taken at 30 degrees off-nadir. We only achieved 91.6% overlap with
545 the X3 RGB camera and 83.9% overlap with the multispectral camera, and all imagery was nadir-facing.
546 While our live/dead classification was fairly accurate (96.4% on a withheld dataset), our species classifier
547 would likely benefit from better crown segmentation because the pixel-level reflectance values within each
548 crown are averaged to characterize the “spectral signature” of each tree. With better delineation of each
549 tree crown, the mean value of pixels within each tree crown will likely be more representative of that tree’s
550 spectral signature. Better crown segmentation might most readily be achieved through greater overlap in
551 imagery. Finally, we anticipate that computer vision and deep learning will prove helpful in overcoming some
552 of these detection and classification challenges (Gray et al. 2019).

553 Conclusions

554 Climate change adaptation strategies emphasize management action that considers whole-ecosystem responses
555 to inevitable change (Millar et al. 2007), which requires a macroecological understanding of how phenomena
556 at multiple scales can interact. We’ve shown that drones can be a valuable tool for investigating multi-scalar
557 phenomena, such as how local forest structure combines with environmental conditions to shape forest insect
558 disturbance. Understanding the conditions that drive dry western U.S. forest responses to disturbances such
559 as bark beetle outbreaks will be vital for predicting outcomes from increasing disturbance frequency and
560 intensity exacerbated by climate change. Our study suggests that outcomes will depend on interactions
561 between local forest structure and broad-scale environmental gradients, with the potential for cross-scale
562 interactions to challenge our current understanding of forest insect dynamics.

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