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Albedo-induced radiative forcing from mountain pine beetle outbreaks in forests, south-central Rocky Mountains: magnitude, persistence, and relation to outbreak severity

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Abstract

Mountain pine beetle (MPB) outbreaks in North America are widespread and have potentially-persistent impacts on forest albedo and associated radiative forcing. This study utilized multiple datasets, both current and historical, within lodgepole pine stands in the south-central Rocky Mountains to quantify the full radiative forcing impact of outbreak events for decades after outbreak (0 to 60 yr) and the role of outbreak severity in determining that impact. Change in annual albedo and radiative forcing peaked at 14–20 yr post-outbreak (0.06 ± 0.006 and -0.8 ± 0.1 W m⁻², respectively) and recovered to pre-outbreak levels by 30–40 yr post-outbreak. Change in albedo was significant in all four seasons, but strongest in winter with the increased visibility of snow (radiative cooling of -1.6 ± 0.2 W m⁻², -3.0 ± 0.4 W m⁻², and -1.6 ± 0.2 W m⁻² for 2–13 yr, 14–20 yr and 20–30 yr post-outbreak, respectively). Change in winter albedo and radiative forcing also increased with outbreak severity (percent tree mortality). Persistence of albedo effects are seen as a function of the growth rate and species compo-

sition of surviving trees, and the establishment and growth of both understory herbaceous vegetation and tree species, all of which may vary with outbreak severity. The establishment and persistence of deciduous trees was found to increase the temporal persistence of albedo effects. MPB induced changes to radiative forcing may have feedbacks for regional temperature and precipitation, which could impact future MPB outbreaks dynamics.

1 Introduction

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Current outbreaks of bark beetles (Coleoptera: Curculionidae, Scolytinae) in western North America are some of the largest and most severe in recorded history (Bentz et al., 2009). Since 1997, more than 41.7 million acres have been affected by bark beetles in the western United States and more than 6.6 million acres in Colorado alone (USFS, 2011). Outbreaks result in tree mortality, temporarily reducing live tree den-





sity and changing stand reflectivity of shortwave radiation. Tree mortality has multiple stages with the needles of a host lodgepole pine tree typically turning red within twelve months of being attacked (red attack stage) and falling off within three years (gray attack stage) (Mitchell and Preisler, 1998). In general, snags begin to fall five years after stand death, with snag fall peaking after ten to fifteen years, although this may vary 5 regionally (Mitchell and Preisler, 1998; Huggard and Lewis, 2007). In recent Colorado outbreaks (1996 to present), mountain pine beetles (MPB) (Dendroctonus ponderosae) have reduced the live basal area of lodgepole pine (*Pinus contorta*) dominated forests by 70%, on average (Klutsch et al., 2009). The current extent and high severity (percent tree mortality) of these recent MPB outbreaks has been attributed to warmer sum-10 mer and winter temperatures and drought conditions associated with regional climate change (Berg et al., 2006; Raffa et al., 2008). Temperature and precipitation trends have increased over-winter survival of beetle populations, increased the rate of reproduction and maturation, and created drought-induced stress in tree hosts (Berg et al.,

¹⁵ 2006; Raffa et al., 2008).

Terrestrial albedo directly influences the total shortwave energy input into the biosphere and is an important determinant of the earth's surface energy balance with attendant impacts on the biosphere and climate (Zhang et al., 1995). Reductions in canopy cover due to disturbance events cause changes in forest structure and compo-

- sition that can have profound impacts on surface albedo. These impacts can outweigh the carbon dioxide and volatile organic carbon (e.g. terpene) emission consequences of a mortality event resulting in an albedo-driven change in radiative forcing (Betts, 2000; Randerson et al., 2006; Spracklen et al., 2008). In the growing season, a site may experience a decline in albedo post-disturbance with the increased visibility of
- ²⁵ bare soil (Bourque et al., 1995). With the establishment of vegetation ground cover, albedo values are typically either elevated or similar to pre-disturbance levels (Mc-Caughey, 1987; Bourque et al., 1995; Gholz and Clark, 2002). Within boreal and high elevation regions, forest loss or tree mortality can result in a significant increase in yearly albedo due to the increased visibility of snow (Betts et al., 2000; Bala et al., 2000; Bala





2007). For evergreen coniferous forests that may be dominated by deciduous species in early successional stages, annual changes in albedo can persist for many more decades post-disturbance, relative to the establishment of evergreen species (Randerson et al., 2006). On an annual basis, an elevated albedo increases short-wave

- ⁵ reflectance and reduces net radiation, which may result in a local annual cooling effect (Betts, 2000; Claussen et al., 2001; Bala et al., 2007). In addition to more direct climate effects (e.g. radiative forcing), changes in albedo also have potential consequences for the hydrological cycle. Evapotranspiration and sublimation can be expected to respond to both a decrease in leaf area index (LAI) as well as an albedo-driven change to net
- radiation. Recent studies have shown a decrease in summer evapotranspiration at regional and watershed scales following bark beetle outbreak (Mikkelson et al., 2013; Maness et al., 2013), which could have implications for regional precipitation patterns (Bala et al., 2007).

Despite wide recognition of the large extent and severity of recent MPB outbreaks, an assessment of the decadal persistence of albedo and radiative forcing effects, as well as an exploration of the impact of outbreak severity on those effects, are both still lacking. Existing studies have documented a decrease in winter albedo in the early attack stages due to needle and litter accumulation on snow (Winkler et al., 2010; Pugh and Small, 2012) and an increase in winter albedo after needle fall, presumably due to the increased visibility of snow (O'Halloran et al., 2012; Vanderhoof et al., 2013). The persistence of winter albedo changes in existing studies has been limited

to ~ 15 yr post-outbreak when albedo change remains large. Furthermore, the variability in outbreak severity has largely been ignored. This study combines field data on tree attributes, historic USFS aerial surveys of insect damage and dendroecological

data to address the following research questions: How does albedo change with attack severity and season? What is the longevity of these albedo impacts and how do these impacts translate into radiative forcing? And can we use vegetation data to explore how outbreak severity may affect the temporal persistence of a change to albedo? Although the epidemic has slowed in many areas of Colorado and Wyoming, due potentially to





the depletion of available host trees, the temporal lag in needle loss, snag fall and forest re-growth implies a prolonged impact on forest structure, albedo, and potentially climate.

Methods 2

₅ **2.1** Study area

The study area was restricted to healthy and MPB damaged lodgepole pine dominated forests within the Colorado and southern Wyoming Rocky Mountains (Fig. 1). Lodgepole pine is found throughout the western United States and northwestern Canada on the lower slopes and valleys above the foothills at elevations between 2100 and 2900 m. It occurs in pure stands, either dense or open, and in mixed stands with other 10 species. Lodgepole pine trees are relatively shade intolerant and many have serotinous cones, adaptations that allow for establishment after a severe forest fire (Kaufmann et al., 2008). Lodgepole pine seedlings tend to grow faster than spruce or fir seedlings and can dominate stands in early successional stages, assuming favorable site conditions. If aspen is present, its sprouting ability can lead to a co-dominance of aspen 15 and lodgepole or to aspen dominating early successional stages, prior to giving way to slower-growing lodgepole pine. Without fire, older lodgepole pine stands can experience in-growth of other species including Engelmann spruce (Picea engelmannii) and subalpine fir (Abies lasiocarpa), if those species are present and if site conditions are favorable (Kaufmann et al., 2008). Non-serotinous cones also allow for the regenera-20 tion of lodgepole pine post-MPB outbreak (Axelson et al., 2010; Teste et al., 2011). Based on climatological records from Estes Park, CO (2365 m elevation), mean an-

nual precipitation is 540 mm, while the average low and high temperature is -9°C in January and 25 °C in July, respectively. Topography and thus microclimate varies substantially within the study area. Locations of past fires (1984-2011) were derived from 25

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Monitoring Trends in Burn Severity (MTBS) data and were masked out (Eidenshink et al., 2007).

2.2 Data

 Albedo was primarily derived from the MODIS Collection 5 BRDF/Albedo 16 day 500 m
 product (MCD43A3), which couples a semi-empirical, RossThick-LiSparse kerneldriven Bidirectional Reflectance Distribution Function (BRDF) model with multi-date, cloud-free, atmospherically-corrected surface reflectance to determine the anisotropy of global land surfaces at a 500 m spatial resolution (Schaaf et al., 2002). Steep to-pography, such as that in the south-central Rocky Mountains, has the potential to bias albedo values. To minimize this source of bias, white-sky albedo broadband was used as it is independent of view and solar angles and thus more comparable across both mountainous regions and temporally (Gao et al., 2005). The MODIS albedo data utilized were limited to 2012 and winter 2013, in order to minimize differences between the

- satellite spectral information and field observations (collected in the summer of 2012).
 To capture seasonal albedo variation, we examined shortwave (0.3–5.0 μm) broadband white sky albedo (WSA) for all 52 dates available between 1 January 2012 and 26 February 2013. Within each image, we rejected all observations with cloud cover, bad quality or fill values based on data quality flags. NIR white-sky broadband albedo was also tested but similarity in findings to SW white-sky broadband albedo led to its
- exclusion. The accuracy of the MODIS Collection 5 shortwave albedo is reported as 0.05 but is generally < 0.03 (Roman et al., 2009; Wang et al., 2010).

In addition to the MODIS albedo data, select images of albedo derived from Landsat-7 ETM+ were also utilized (p34, r32). Snow-free Landsat albedo images (DOY 170, 218 and 266 2012) were derived using the methodology described in Shuai et al. (2011)

which integrates land cover-based high quality BRDFs over the homogenous region, extracted from MCD43A operational products, with Landsat surface reflectance to calculate spectral and broadband white and black-sky albedo. Snow-cover Landsat shortwave broadband albedo images (DOY 012, 026, 2012) were derived in terms of concurrent MCD43A snow BRDFs and the albedo-to-reflectance ratios for each determined snow-cover pixel by MOD10 snow mapping algorithm (Hall et al., 2002). It should be noted that the snow-cover Landsat albedo methodology is still in development and that this was an experimental application of the method. Results derived using snow-cover Landsat albedo images are provided only for comparison to findings from the usage of

the MODIS albedo dataset. Data on locations of MPB outbreaks were derived from three sources: (1) field data

on tree attributes collected in plots (63 sites), (2) historic US Forest Service Aerial Detection Surveys (USFS ADS) (46 sites), and (3) field-based dendroecological plots (11

- sites) (Table 1, Fig. 1). Field-collected tree plot data were collected in lodgepole pine dominated stands from 18 June–20 July 2012, within 900 m² plots (56 700 m² total). Plots were targeted to be representative of (1) the surrounding 500 m × 500 m area, (2) a severity spectrum (from none to severe) of bark beetle outbreak, and (3) a spectrum of time since attack (green attack, red attack and gray attack stages). Although
- plots were not restricted to pure lodgepole pine stands, the plots were on average 93 % lodgepole pine. Within each plot, each tree was identified to the species level, its diameter measured (dbh > 7.5 cm), and recorded as un-attacked (no evidence of beetle presence), green attack stage (exit holes and pitch tubes present but needles still green), red attack stage (exit holes and pitch tubes present and needles red) or gray attack
- stage (exit holes and pitch tubes present and needles absent), or unknown dead (needles absent, but no evidence of beetle damage). Sapling and seedling (dbh < 7.5 cm) abundances and species composition were sampled in two 25 m² subplots within each plot. Mean canopy height was measured using a clinometer, canopy cover was measured using a densitometer, litter depth was measured using a ruler in soil pits, and
- ²⁵ understory cover (vegetation, litter, bare rock) was visually estimated. Plots were categorized by attack stage (healthy, red, gray), where attacked plots were categorized as red if the majority of attacked trees were in the red attack stage, or gray if the majority of the attacked trees were in the gray attack stage. The plots were also categorized by severity, using percent dead trees, where low (14 to 40% mortality), medium (41





to 57 % mortality) and high (> 58 % mortality) categories each contained an equivalent number of plots. Percent dead trees ranged from < 1 % to 88 % with a mean percent mortality of 41 %. A potential bias in the albedo signal due to differences in scale between the field data and MODIS data may occur. To correct for this potential bias, the

- MODIS pixel extent for each field plot was reviewed using 2011 National Agriculture Imagery Program (NAIP) imagery to check for consistency of forest cover and damage patterns. Plots within MODIS pixels that were < 75 % coniferous forest cover were removed from the analysis. Because no such scale issue occurred with the Landsat pixels, all field plots were included in the analysis.
- The primary source for historic (> 16 yr since outbreak) MPB outbreak locations was geo-referenced USFS Region 2 ADS maps (1956–1987). These maps were periodically created by expert surveyors to record "new damage" polygons during flown surveys. Although an absence of a polygon does not necessarily indicate true absence due to the incomplete spatial nature of these surveys, a drawn polygon provides high
- ¹⁵ confidence of the presence of an outbreak. Polygons of historic outbreaks were eliminated from consideration if they (1) overlapped with damage polygons identified in later USFS Region ADS maps (1994–2011), (2) did not occur within lodgepole-dominated forests as defined by multiple USFS forest layers, or (3) showed inconsistent (< 75 %) forest cover using aerial imagery (2011 USDA NAIP imagery). In addition, a dispropor-
- tionate number of downed trees, relative to a typical mature forest, were observed using aerial imagery within most historic damage locations, corroborating past outbreaks.

Additional historic MPB outbreaks were located using dendroecological methods that have been previously tested (Heath and Alfaro, 1990; Alfaro et al., 2010) and applied (Axelson et al., 2009; Kulakowski and Jarvis, 2011; Kulakowski et al., 2012) to lodge-

pole pine stands to reconstruct stand disturbance history in this and other regions. Eleven lodgepole pine-dominated sites were randomly selected for dendroecological reconstruction of MPB outbreaks (Kulakowski et al., 2012; Jarvis et al., 2013). In each stand increment cores were collected from living host and non-host and dead host trees. Dead host trees were only cored if MPB larval galleries were present under the





remaining bark. These cores were used to derive dates of stand-origin, tree mortality and releases (abrupt two-fold increases in ring width sustained for more than 10 yr). Additional methodological details are provided in Kulakowski et al. (2012). Data were collected in summer of 2008 (Kulakowski et al., 2012) and summer of 2009 (Jarvis et al. 2013). Similar to our tree plot data, the MODIS pixel extent for each dendroe-cological plot was reviewed using 2011 National Agriculture Imagery Program (NAIP) imagery to check for consistency of forest cover. Plots within MODIS pixels that were < 75 % forest were removed from the analysis. Plot locations were also checked for agreement with recent (2009–2011) outbreaks recorded in the USFS ADS data.

10 2.3 Analysis

Albedo was expressed as the change in albedo from the within scene "healthy" lodgepole pine forest. "Healthy" stands were derived from fourteen non-attacked 2012 tree plot locations. Change in albedo was averaged by time interval (1-3, 4-13, 14-20, 21-30, 31-40, and 50-60 yr since disturbance) and season (winter = December–February,

spring = March–May, summer = June–August, fall = September–November) to derive seasonal curves for change in albedo with time since outbreak start. The width of the time intervals was based on our confidence in deriving an accurate year of outbreak. The time intervals of 41–50 and > 60 yr since MPB outbreak were not included due to small sample sizes. To assess the role of attack severity, gray attack dominated tree plots were categorized by percent tree mortality, where thresholds between categories were determined using quantiles. Pearson pair-wise correlations and paired *t* tests were conducted using SPSS Statistics (IBM Corporation, Armonk, NY).

2.4 Radiative forcing

We defined the net radiative forcing as the mean annual change in reflected shortwave radiation at the top of the atmosphere (TOA) resulting from changes in surface albedo (Hansen et al., 1997). The TOA radiative forcing was calculated using the radiative ker-



nel technique, which represents changes to TOA fluxes caused by changes in monthly average surface albedo, relative to "healthy forest" values at 2.5° resolution, which was re-sampled to 1° resolution to better match the spatial extent of the study area (Shell et al., 2008). The kernel (*K*) was produced using the offline radiative transfer model
⁵ from the National Center for Atmospheric Research (NCAR) Community Atmospheric Model version 3 (Collins et al., 2006) as described in (Shell et al., 2008), and reported as the radiative forcing response to a 0.01 change in albedo, where:

 $\Delta F_{\rm TOA} = \Delta \alpha \cdot K^{\alpha}$

where *a* is the land surface shortwave albedo. A 5 % uncertainty (per product) was assumed for the MODIS albedo and the radiative kernels. We implemented a conservative assumption of non-random error propagation for which uncertainty was combined as the product of those for albedo and radiative kernels. This method of uncertainty propagation is analogous to an IPCC Tier 1 uncertainty (IPCC, 2000).

3 Results

- ¹⁵ After MPB outbreak, an annual increase in MODIS albedo peaked at 14–20 yr with snag fall (0.06 ± 0.006) before transitioning to a minor, non-significant change in albedo (-0.01 ± 0.003 and 0.001 ± 0.003), relative to non-attacked stands, by 30–40 yr and 50– 60 yr, respectively (Fig. 2). MODIS albedo of attacked stands was significantly higher than in non-attacked stands in all seasons from 4–30 yr post-outbreak (p < 0.01), with evidence of a slight decline in the first three years post-attack in fall, winter, and summer seasons (p < 0.01). No significant difference between non-attacked stands and attacked stands was seen 30–40 yr or 50–60 yr since attack, presumably due to the recovery of vegetation cover. The same was seen with Landsat data, for which winter and summer albedo was significantly higher 4–30 yr since disturbance relative to non-25 attacked stands (p < 0.01). The increase in albedo post-outbreak peaked during winter
- attacked stands (p < 0.01). The increase in albedo post-outbreak peaked during winter months, with the increased visibility of snow. Change to winter MODIS albedo was



(1)



 0.06 ± 0.002 , 0.10 ± 0.004 , and 0.05 ± 0.003 , for 4–13 yr, 14–20 yr and 20–30 yr postoutbreak, respectively (Fig. 2) (p < 0.01). Winter findings for Landsat albedo were not significantly different from those for MODIS for all time intervals except 30–40 yr, when Landsat showed a reduction in albedo, relative to non-attacked stands, not seen in MODIS (p < 0.01). This localized reduction in albedo, relative to non-attacked stands, could be due to high tree densities, a characteristic often observed in early successional stands before natural thinning occurs with tree and stand maturation.

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Using only gray-attack dominated tree plot data (4–13 yr since disturbance), change in albedo by season was examined as a function of percent tree mortality. A strong relationship was observed during winter months, with the increase in albedo rising with

- relationship was observed during winter months, with the increase in albedo rising with increased tree mortality for both MODIS (y = 0.0021x 0.026, $R^2 = 0.63$, p < 0.01) and Landsat (y = 0.0026x 0.068, $R^2 = 0.37$, p < 0.05) data sources (Fig. 3). The relationship between MODIS or Landsat albedo and percent tree mortality was weaker for spring, summer and fall seasons, but still statistically significant (p < 0.01) (Fig. 4).
- ¹⁵ These changes in albedo translated into sizeable TOA radiative forcing. Radiative forcing peaked during winter months, with an observed radiative cooling of $-1.6\pm0.2 \text{ Wm}^{-2}$, $-3.0\pm0.4 \text{ Wm}^{-2}$, and $-1.6\pm0.2 \text{ Wm}^{-2}$ for 2–13 yr, 14–20 yr and 20– 30 yr post-outbreak, respectively (Fig. 5). Annual radiative cooling peaked at 14–20 yr post-outbreak at $-0.8\pm0.1 \text{ Wm}^{-2}$. Using only gray-attack dominated tree plot data (4–
- ²⁰ 13 yr post-outbreak), seasonal and annual radiative forcing were quantified by attack severity. As would be expected, seasonal and annual radiative forcing increased with increasing attack severity. Winter radiative forcing was $0.2\pm0.03 \text{ Wm}^{-2}$, $-1.8\pm0.2 \text{ Wm}^{-2}$ and $-2.3\pm0.3 \text{ Wm}^{-2}$ for low, medium and high severity outbreaks, respectively (Fig. 5). A large increase in radiative cooling occurred between low (10–30% tree mortality)
- and medium (30–57% tree mortality) outbreak severity, with a minor change between medium and high (57–88% tree mortality) outbreak severity, suggesting that the relationship between radiative forcing and outbreak severity may exhibit a threshold rather than linear relationship. A shift in the relationship between albedo and percent tree mortality was observed at ~ 40–50% tree mortality in Fig. 3. This threshold could be





related to shadows of remaining live trees masking the change in albedo for low severity outbreaks.

For gray attack dominated tree plots (4-13 yr since outbreak) the post-outbreak change in MODIS albedo was most strongly explained by the percent tree mortality (winter, R = 0.87, fall, R = 0.83, p < 0.01) or number of dead trees (spring, R = 0.86, 5 summer, R = 0.63, p < 0.01) in a plot. Similarly, change in Landsat albedo was most strongly explained by the percent tree mortality (winter, R = 0.65, p = 0.02, summer, R = 0.55, p < 0.01). The persistence of this change in albedo, or the rate of forest regrowth, in turn, is typically a function of two mechanisms (which can co-occur): (1) the growth and reorganization of existing vegetation (e.g., advance regeneration), and/or 10

- (2) the recruitment of new vegetation (McCarthy, 2001). Given that the mortality event is targeted at lodgepole pine, the reorganization of existing vegetation can dramatically shift the forest species composition if other species are present. The remaining trees will temporarily dominate the canopy, and their size and growth rates will influence the
- rate of albedo recovery. Within the 2012 tree plots, a shift in live tree species composi-15 tion was observed between the non-attacked and attacked plots. Within healthy stands, lodgepole pine comprised an average of 93 % of trees, while within damaged plots this percent was reduced to 76%. The remaining trees (20%) within damaged plots were dominated by other coniferous species, principally Douglas fir (Pseudotsuga menziesii)

and ponderosa pine (Pinus ponderosa). 20

In addition to the species composition of existing vegetation, the species composition of newly recruited seedlings/saplings is also important to consider, as a stand's coniferous versus deciduous trajectory can shorten or prolong changes to winter albedo, respectively (Randerson et al., 2006). Within the 2012 tree plots, attacked plots exhibited a similar (non-significantly different) seedling/sapling stem density 25 $(0.44 \pm 0.08 \text{ stems m}^{-2})$ relative to healthy plots $(0.60 \pm 0.17 \text{ stems m}^{-2})$. Healthy and damaged plots exhibited similar proportions of lodgepole pine seedling/sapling composition (32-33%), but damaged plots showed a higher percent of quaking aspen (Populus tremuloides) seedlings/saplings (20%) relative to healthy stands (7%). In ad-





dition, high severity outbreaks more often had quaking aspen seedling/saplings present (50 % of plots) relative to non-attacked, low and medium severity outbreaks (12–17 % of plots). As aspen tends to be a shade intolerant tree species (Messier et al., 1999), this finding is not unexpected. The remaining seedlings/saplings were comprised of

- ⁵ other coniferous species, dominated by subalpine fir and Douglas fir. The presence of lodgepole pine seedlings, even in the absence of a fire event is consistent with the findings of others, although the success and rate of growth of these seedlings may depend on light availability and soil moisture (Stuart et al., 1989; Astrup et al., 2008; Teste et al., 2011).
- ¹⁰ We also explored whether the rate of new vegetation recruitment was dependent on outbreak severity, which influences light availability. The number of seedlings and saplings showed no relationship with outbreak severity or with change in albedo. Outbreak severity, however, was found to be strongly positively related to the percent understory non-tree vegetation cover (y = 0.66x - 7.32, R = 0.72, p < 0.01) (Fig. 6). Al-
- ¹⁵ though understory non-tree vegetation can be expected to be masked by snow during winter months, this phenomenon may be affecting the summer albedo change signal. If we control for outbreak severity, we find that the increase in summer albedo was positively correlated with percent understory vegetation for high severity outbreak sites (y = 0.0002x - 0.0004, R = 0.68, p < 0.01) where understory vegetation is most abun-
- dant. As herbaceous vegetation typically has a higher albedo than bare, dry soil (Mc-Caughey, 1987; Bourque et al., 1995; Gholz and Clark, 2002), the understory non-tree vegetation may be contributing to the change in summer albedo, at least for high severity outbreaks. Additionally, the annual increase in MODIS albedo decreased strongly with an increasing number of live trees remaining after outbreak, for medium sever-
- ity outbreaks (R = -0.85, p < 0.01). This finding suggests that, at least for medium severity outbreaks, in addition to the percent tree mortality, the absolute number of live remaining trees is also important in determining the magnitude and persistence of the albedo response.





4 Discussion

One of the advances this study made was to utilize historical data to document how albedo changes with time since a MPB outbreak. Both Vanderhoof et al. (2013) and O'Halloran et al. (2012) expected an increase in albedo past 14–15 yr post-outbreak

- the limit of data utilized in both studies). This expectation was observed in this study with change in albedo peaking 14–20 yr post-outbreak, presumably as snags fell and sapling/seedling re-establishment was still low and/or small. The timing of snag fall appears to be consistent with existing literature, which shows tree fall peaking after 10–15 yr (Mitchell and Preisler, 1998; Huggard and Lewis, 2007). Despite differences in the
- ¹⁰ number of images used and the scale of the data, MODIS and Landsat derived albedo produced very similar patterns of albedo with time since disturbance (Fig. 2), adding confidence to the overall characterization of albedo changes following outbreak. Larger error intervals observed for the Landsat derived change in albedo was a consequence of smaller sample sizes due to fewer images being used.
- ¹⁵ Although the scale of the albedo data (MODIS at 500 m, Landsat at 30 m) was found to have a minimal impact on results, the scale of the analysis (plot vs. landscape) was found to be relevant for the size of the change in albedo. In particular the increase in albedo for 4 to 13 yr since outbreak was greater when assessed at a plot scale than at a landscape scale, in both winter (77 % higher increase in albedo) and summer (166 %
- higher increase in albedo) (Vanderhoof et al., 2013). This difference can be attributed to using USFS ADS polygons for the landscape scale analysis. Although the USFS ADS dataset is currently the most comprehensive spatial source of MPB outbreak dynamics in the United States, the dataset tends to over-estimate the area attacked (Brian Howell, USFS, personal communication, 2013), leading to a dilution of the quantified change
- in albedo due to the inclusion of both attacked and non-attacked pixels, emphasizing the more regional impacts rather than the local responses specifically within areas of outbreak. As a side-note, this study maximized the site-based potential of the historic USFS ADS polygons by selecting points central to the polygons, therefore maximizing





the chance that an attack occurred at that location, as well as verifying both forest cover and evidence of a disproportionate number of downed trees using aerial imagery, as described in the Methods Section.

- Given that the magnitude of albedo change varies with outbreak severity (Fig. 3), it is logical to explore if the rate of albedo recovery also varies with outbreak severity. Unfortunately accurate severity data (% tree mortality) was lacking for the USFS ADS historical data as well as the dendroecological plots, thus we were unable to directly quantify this variation. We can, however, explore the role of regrowth mechanisms to increase canopy cover, reducing the change in albedo relative to non-attacked stands.
- Persistence of albedo change, or alternatively, the forest recovery from epidemic MPB outbreaks in lodgepole pine tends to occur primarily through the reorganization of existing vegetation (e.g., advance regeneration), with existing understory trees and vegetation being released, in response to increased light, moisture and nutrients, to form the new canopy (Cole and Amman, 1980; Greene et al., 1999). The presence of understory
- trees and vegetation can limit the recruitment of new tree seedlings through competitive exclusion, sometimes limiting seedling establishment for many decades (Wickman et al., 1986; Wohlgemuth et al., 2002). This may explain the low seedling/sapling stem density observed at sites 1–20 yr post-outbreak. In stands with limited understory trees and vegetation, recovery via the establishment of new seedlings can still be highly variable due varving site characteristics such as elevation, aspect, and soil type. (Mitchell
- able due varying site characteristics such as elevation, aspect, and soil type (Mitchell, 2005).

The primary re-growth mechanism to recover pre-disturbance albedo may also vary by outbreak severity. In low severity outbreaks, we may expect that gaps in tree canopy will be primarily filled through the growth release and canopy expansion of remaining live canopy and sub-canopy trees (Fig. 7). In high severity outbreaks, it is likely that

gaps will persist even after the growth release and canopy expansion of remaining live trees, due to (1) the limited capacity for the existing canopy to expand, and (2) the number of remaining live trees. In the case of high severity outbreaks, a full recovery of albedo will also require the recruitment and growth of new seedlings/saplings. If we as-

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sume that an increase in canopy cover due to the reorganization of existing vegetation will occur faster than due to the recruitment and growth of new vegetation (Griesbauer and Green, 2006), then the persistence of the change in albedo will depend on outbreak severity, which will influence the fractional contribution of each forest regrowth mechanism to albedo recovery (Fig. 8).

Secondarily, we may consider if the growth rate of both existing and new vegetation depends on outbreak severity. Current literature shows that the growth release for existing vegetation is typically related to light availability, which will vary based on outbreak severity (Mitchell, 2005; Messier et al., 1999; McCarthy, 2001). However, even following high severity outbreaks, standing snags will provide shade for approximately 15 yr

- ¹⁰ Ing high severity outbreaks, standing snags will provide shade for approximately 15 yr post-outbreak (Griesbauer and Green, 2006). In general, shade-tolerant species, such as Douglas fir, interior spruce (*Picea glauca*) and subalpine fir, tend to better utilize the gradual increases in light levels relative to shade-intolerant trees (e.g. lodgepole pine, ponderosa pine, quaking aspen, or paper birch (*Betula papyrifera*)) (Mitchell, 2005;
- ¹⁵ Messier et al., 1999; McCarthy, 2001). In addition, shade-tolerant trees can adjust their crown physiology and structure in response to different light regimes, even after long periods of suppression, which makes them highly responsive to gradual increases in light following an outbreak (Messier et al., 1999; McCarthy, 2001). In contrast, shadeintolerant species tend to have more fixed crown physiology and structure, and thus
- are less able to respond with a strong growth release (Messier et al., 1999; Williams et al., 1999). These species specific differences in growth release imply that Douglas fir and subalpine fir should play a substantial role in the recovery of the canopy, if these species are present at a given site. Shade-intolerant species, however, may still do well on edaphically limiting sites (Williams et al., 1999) or sites with large canopy gaps
 (McCarthy, 2001).

Following high severity outbreaks where light is less limiting, and in which we observed more frequent quaking aspen recruitment, the persistence of an albedo effect post-outbreak will also depend on the future species composition of the stand (coniferous versus deciduous). If the post-outbreak forest becomes dominated by deciduous





trees, we might expect a prolonged persistence of winter change in albedo due to winter leaf loss and therefore increased visibility of snow, relative to coniferous stands. In high light conditions and where aspen clones are present, aspen's sprouting ability can lead to its dominance in early successional stages, before slower-growing lodge-

- ⁵ pole pine trees out-compete patches of aspen (Kaufmann et al., 2008). Using aerial imagery, we found that most historic outbreak sites showed strong establishment of coniferous saplings which eventually created a closed-canopy environment, however, one historical dendroecological plot (Kulakowski et al., 2012), showed a persistent increase in winter albedo at 40–50 yr post-outbreak, similar to the change observed at
- 10 20–30 yr post-outbreak at other historic outbreak sites, due to the establishment and/or shift in species composition towards deciduous tree species. This observation indicates that the temporal persistence found in this study depends on the initial regeneration of evergreen coniferous species post-outbreak, and that the establishment of deciduous species post-outbreak can increase the temporal persistence of the albedo impact by 15 at least several decades (Randerson et al., 2006).

For new vegetation, we found a strong relationship between non-tree vegetation cover and percent tree mortality for recent outbreaks (< 14 yr) (Fig. 6), indicating that its establishment is influenced by light availability. We did not find a relationship between seeding/sapling recruitment and outbreak severity, for which there may be several ex-

- ²⁰ planations. Since seedling/sapling density data (unlike the percent understory vegetation) were derived from sub-plots (25 m²) within the larger tree plots (900 m²), the lack of correlation with outbreak severity may occur because the response or establishment of seedling/saplings is very local (at a tree-scale), leading to a potential disconnect between the seedling/sapling density and the percent tree mortality observed at the
- ²⁵ plot scale. In addition, the lack of correlation may occur because not enough time since disturbance has passed to reflect the expected pulse of seedlings/saplings postdisturbance (Wickman et al., 1986; Wohlgemuth et al., 2002). No significant difference between seedling/sapling density was observed between non-attacked, 4–13 yr or 14– 20 yr since attack. A future research step could be to collect additional vegetation data





within sites near the threshold of albedo recovery (20–40 yr post-outbreak) and correlate albedo change (or lack of albedo change, relative to non-attacked stands) with number of live mature trees, crown size of live trees, percent understory vegetation and number and size of saplings to directly determine the relative importance of factors to the persistence of albedo change post-MPB outbreak.

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The magnitude of MPB induced albedo-driven radiative forcing observed in this study agrees well with that reported by O'Halloran et al. (2012) (winter albedo radiative forcing of $-1.9\pm0.3 \text{ Wm}^{-2}$ at year four and $-2.4\pm0.4 \text{ Wm}^{-2}$ at year eleven). The long term radiative forcing of MPB attacks can also be compared to the impact of forest fires, another disturbance type common to the south-central Rocky Mountains. The long term albedo-driven radiative forcing of a severe fire event in boreal Alaska ($-4\pm2 \text{ Wm}^{-2}$ for years 0 to 80) reported by Randerson et al. (2006) was much higher than that of the long term (4–30 yr) albedo driven radiative forcing from a MPB outbreak (winter = $-2.1\pm0.3 \text{ Wm}^{-2}$, annual = $0.5\pm0.08 \text{ Wm}^{-2}$) as reported here. This is because of

- a larger increase in winter albedo post-fire (change of ~ 0.4) than that observed post-MPB outbreak (change of ~ 0.1 for 14–20 yr post-outbreak). However direct comparison may not be warranted given the difference in species composition, snow cover climatology, rates of regeneration, and other factors. Furthermore, the potentially smaller radiative cooling from MPB outbreaks may be somewhat compensated by the larger
 spatial extent of MPB outbreaks relative to fire within Colorado and southern Wyoming
- over the same timeframe. Fire extent (as derived from MTBS data) was 396 800 ha (all cover types) for 1994–2010 compared to 822 200 ha of MPB outbreak (1994–2010) within lodgepole pine forests.

A full radiative forcing analysis for MPB outbreaks must also consider committed car-²⁵ bon emissions, based on total tree mortality. However it must be noted that while the albedo radiative forcing component will occur immediately, as will a reduction in net ecosystem carbon uptake due to a stand reduction in LAI and associated productivity, a substantial lag will occur before killed trees fully decompose (the committed CO₂ emissions). The turnover time for lodgepole pine CWD in the Colorado Rocky Moun-





tains can be on the order of hundreds of years, ranging from 340 ± 130 yr to 630 ± 410 yr depending on temperature (Brown et al., 1998; Kueppers et al., 2004). This time lag between albedo and carbon radiative forcing is important to recognize.

Lastly, we acknowledge that bark beetle outbreaks have potential consequences for the hydrological cycle as well. Reductions in summer evapotranspiration post MPB outbreak (Mikkelson et al., 2013; Maness et al., 2013), could result in a positive feedback mechanism, in which a reduction in summer evapotranspiration results in reduced precipitation and increased drought conditions which may exacerbate current outbreaks that are stimulated by warmer, drier conditions. Alternatively, the documented decrease in a winter radiative forcing, due to increased snow visibility, could result in cooler

¹⁰ In a winter radiative forcing, due to increased snow visibility, could result in cooler wintertime air temperatures, potentially reducing beetle populations because of winter "cold snaps" known to kill beetles.

5 Conclusions

Combining multiple datasets, both recent and historical, we quantified the cumulative radiative forcing due to MPB outbreaks in lodgepole pine stands in the south central 15 Rocky Mountains. This is the first comprehensive estimate of the total albedo-induced radiative forcing from MPB outbreaks of which we are aware, and might be useful as a reference for other affected regions while recognizing likely variation with climate. edaphic, and biotic (species composition) factors. For MPB outbreaks in lodgepole pine stands in the south-central Rocky Mountains, albedo post-outbreak was consis-20 tently higher than in non-attacked stands from 4 to 30 yr since the start of outbreak and then returned to pre-outbreak levels. Seasonally, changes in albedo peaked during the winter season, with the increased visibility of snow, but significant differences in albedo, from non-attacked stands, were observed in all four seasons. Change in albedo also increased with increasing attack severity (% tree mortality), particularly 25 in the winter season. This variation translated into large differences in winter radiative forcing for gray attack stands $(0.2 \pm 0.03 \,\mathrm{Wm^{-2}}$ for low severity outbreaks versus



-2.3 ± 0.3 W m⁻² for high severity outbreaks). The magnitude of albedo response in the first decade following attack depends on the number of remaining live trees, their species composition, the size of the growth release, and the establishment of understory vegetation. In the longer term, the persistence of albedo effects depends also
on the rate of growth and species composition of sub-canopy seedlings and saplings. Prior to recovery, the albedo-induced radiative forcing may create seasonally specific temperature and precipitation feedbacks, which may contribute to the persistence or reduction of MPB outbreaks in the coming decades.

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Table 1. The number and source of MPB outbreak locations utilized by time interval or time since MPB outbreak.

Time Interval	Number of sites/Source	Number of sites/Source	Total #
Healthy	14/2012 tree plots		14
1–3 yr	10/2012 tree plots		10
4–13 yr	33/2012 tree plots		33
14–20 yr	6/2012 tree plots	14/USFS ADS sites	20
21–30 yr	10/2009 dendro plots	9/USFS ADS sites	19
31–40 yr	1/2008 dendro plot	18/USFS ADS sites	19
50–60 yr	5/USFS ADS sites		5
		Total	120









Fig. 1. Plot locations for field tree plots, dendroecological plots and USFS ADS sites.

















Fig. 4. Change in spring, summer and fall albedo with MPB outbreak severity, defined as percent tree mortality, for outbreaks 4 to 13 yr in age.





Fig. 5. Radiative forcing as derived from MODIS albedo, organized by time interval since outbreak start and outbreak severity (% tree mortality), where severity forcing is for gray attack stands (4–13 yr since outbreak start). Error bars represent forcing uncertainty.







Fig. 6. Understory vegetation cover as a function of post-outbreak percent tree mortality, within gray-attack (4–13 yr since outbreak start) dominated field tree plots.



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Pre-Attack Forest

Post-Attack Forest (after canopy expansion)

Fig. 7. Remaining canopy and sub-canopy trees can be expected to expand in response to increased light availability post-MPB attack, however we can anticipate that the capacity of this expansion will be limited.









