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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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## Albedo induced radiative forcing from MPB outbreaks in Rocky Mountains

M. Vanderhoof et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures



Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion

## 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 ( $0.06 \pm 0.006$  and  $-0.8 \pm 0.1 \text{ W m}^{-2}$ , 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  $-1.6 \pm 0.2 \text{ W m}^{-2}$ ,  $-3.0 \pm 0.4 \text{ W m}^{-2}$ , and  $-1.6 \pm 0.2 \text{ W m}^{-2}$  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 composition 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

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-

## BGD

10, 11935–11968, 2013

### Albedo induced radiative forcing from MPB outbreaks in Rocky Mountains

M. Vanderhoof et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion





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.

## 2 Methods

### 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 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 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 regeneration 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 annual precipitation is 540 mm, while the average low and high temperature is  $-9^{\circ}\text{C}$  in January and  $25^{\circ}\text{C}$  in July, respectively. Topography and thus microclimate varies substantially within the study area. Locations of past fires (1984–2011) were derived from

**BGD**

10, 11935–11968, 2013

## Albedo induced radiative forcing from MPB outbreaks in Rocky Mountains

M. Vanderhoof et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion

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 kernel-driven 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 topography, 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  $\mu\text{m}$ ) 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 concur-

### Albedo induced radiative forcing from MPB outbreaks in Rocky Mountains

M. Vanderhoof et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures



Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



rent 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 m<sup>2</sup> plots (56 700 m<sup>2</sup> 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 m<sup>2</sup> 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 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

## BGD

10, 11935–11968, 2013

### Albedo induced radiative forcing from MPB outbreaks in Rocky Mountains

M. Vanderhoof et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures



Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion





## Albedo induced radiative forcing from MPB outbreaks in Rocky Mountains

M. Vanderhoof et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion

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 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:

$$\Delta F_{\text{TOA}} = \Delta \alpha \cdot K^\alpha \quad (1)$$

where  $\alpha$  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

After MPB outbreak, an annual increase in MODIS albedo peaked at 14–20 yr with snag fall ( $0.06 \pm 0.006$ ) before transitioning to a minor, non-significant change in albedo ( $-0.01 \pm 0.003$  and  $0.001 \pm 0.003$ ), 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-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

**Albedo induced  
radiative forcing from  
MPB outbreaks in  
Rocky Mountains**

M. Vanderhoof et al.

[Title Page](#)[Abstract](#)[Introduction](#)[Conclusions](#)[References](#)[Tables](#)[Figures](#)[⏪](#)[⏩](#)[◀](#)[▶](#)[Back](#)[Close](#)[Full Screen / Esc](#)[Printer-friendly Version](#)[Interactive Discussion](#)

0.06 ± 0.002, 0.10 ± 0.004, and 0.05 ± 0.003, for 4–13 yr, 14–20 yr and 20–30 yr post-outbreak, 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.

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 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).

These changes in albedo translated into sizeable TOA radiative forcing. Radiative forcing peaked during winter months, with an observed radiative cooling of  $-1.6 \pm 0.2 \text{ W m}^{-2}$ ,  $-3.0 \pm 0.4 \text{ W m}^{-2}$ , and  $-1.6 \pm 0.2 \text{ W m}^{-2}$  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  $-0.8 \pm 0.1 \text{ W m}^{-2}$ . Using only gray-attack dominated tree plot data (4–13 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  $0.2 \pm 0.03 \text{ W m}^{-2}$ ,  $-1.8 \pm 0.2 \text{ W m}^{-2}$  and  $-2.3 \pm 0.3 \text{ W m}^{-2}$  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  $\sim 40$ –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,  $R = 0.86$ , 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 re-growth, 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 (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 composition 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*).

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 ( $0.44 \pm 0.08$  stems  $m^{-2}$ ) relative to healthy plots ( $0.60 \pm 0.17$  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-

## BGD

10, 11935–11968, 2013

### Albedo induced radiative forcing from MPB outbreaks in Rocky Mountains

M. Vanderhoof et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion

## Albedo induced radiative forcing from MPB outbreaks in Rocky Mountains

M. Vanderhoof et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion

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 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).

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). Although 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 abundant. As herbaceous vegetation typically has a higher albedo than bare, dry soil (McCaughey, 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 severity 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.



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 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-

## BGD

10, 11935–11968, 2013

### Albedo induced radiative forcing from MPB outbreaks in Rocky Mountains

M. Vanderhoof et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures



Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



## Albedo induced radiative forcing from MPB outbreaks in Rocky Mountains

M. Vanderhoof et al.

[Title Page](#)

[Abstract](#)

[Introduction](#)

[Conclusions](#)

[References](#)

[Tables](#)

[Figures](#)

[⏪](#)

[⏩](#)

[◀](#)

[▶](#)

[Back](#)

[Close](#)

[Full Screen / Esc](#)

[Printer-friendly Version](#)

[Interactive Discussion](#)

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 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; 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, shade-intolerant 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

## Albedo induced radiative forcing from MPB outbreaks in Rocky Mountains

M. Vanderhoof et al.

[Title Page](#)

[Abstract](#)

[Introduction](#)

[Conclusions](#)

[References](#)

[Tables](#)

[Figures](#)

[⏪](#)

[⏩](#)

[◀](#)

[▶](#)

[Back](#)

[Close](#)

[Full Screen / Esc](#)

[Printer-friendly Version](#)

[Interactive Discussion](#)

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 lodgepole 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 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 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 explanations. Since seedling/sapling density data (unlike the percent understory vegetation) were derived from sub-plots ( $25 \text{ m}^2$ ) within the larger tree plots ( $900 \text{ m}^2$ ), 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 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 post-disturbance (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

## Albedo induced radiative forcing from MPB outbreaks in Rocky Mountains

M. Vanderhoof et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion

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.

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  $-1.9 \pm 0.3 \text{ W m}^{-2}$  at year four and  $-2.4 \pm 0.4 \text{ W m}^{-2}$  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 ( $-4 \pm 2 \text{ W m}^{-2}$  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 =  $-2.1 \pm 0.3 \text{ W m}^{-2}$ , annual =  $0.5 \pm 0.08 \text{ W m}^{-2}$ ) as reported here. This is because of a larger increase in winter albedo post-fire (change of  $\sim 0.4$ ) than that observed post-MPB outbreak (change of  $\sim 0.1$  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 carbon 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  $\text{CO}_2$  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  $340 \pm 130$  yr to  $630 \pm 410$  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 (Mikkelsen 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 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 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 consistently 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 in the winter season. This variation translated into large differences in winter radiative forcing for gray attack stands ( $0.2 \pm 0.03 \text{ W m}^{-2}$  for low severity outbreaks versus

**BGD**

10, 11935–11968, 2013

### Albedo induced radiative forcing from MPB outbreaks in Rocky Mountains

M. Vanderhoof et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



-2.3 ± 0.3 Wm<sup>-2</sup> 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 under-story 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.

*Acknowledgements.* This work was funded by the 2011–2013 NASA Earth and Space Science Fellowship (NESSF) (11-Earth11F-134 and 12-Earth12R-59). Additional financial support was received from the NASA ROSES09 Science of Terra and Aqua program (Grant #: NNX11AG53G) as well as the National Science Foundation (Grant #: 1262691). We thank Rocky Mountain National Park for their field support and Marcus Pasay for his assistance with field work. We also thank Brian Howell and Justin Backsen of the USFS Region 2, for their assistance with the historical ADS surveys.

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## Albedo induced radiative forcing from MPB outbreaks in Rocky Mountains

M. Vanderhoof et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures



Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



## Albedo induced radiative forcing from MPB outbreaks in Rocky Mountains

M. Vanderhoof et al.

[Title Page](#)

[Abstract](#)

[Introduction](#)

[Conclusions](#)

[References](#)

[Tables](#)

[Figures](#)

[⏪](#)

[⏩](#)

[◀](#)

[▶](#)

[Back](#)

[Close](#)

[Full Screen / Esc](#)

[Printer-friendly Version](#)

[Interactive Discussion](#)

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## Albedo induced radiative forcing from MPB outbreaks in Rocky Mountains

M. Vanderhoof et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion

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M. Vanderhoof et al.

[Title Page](#)

[Abstract](#)

[Introduction](#)

[Conclusions](#)

[References](#)

[Tables](#)

[Figures](#)

[⏪](#)

[⏩](#)

[◀](#)

[▶](#)

[Back](#)

[Close](#)

[Full Screen / Esc](#)

[Printer-friendly Version](#)

[Interactive Discussion](#)

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## Albedo induced radiative forcing from MPB outbreaks in Rocky Mountains

M. Vanderhoof et al.

[Title Page](#)

[Abstract](#)

[Introduction](#)

[Conclusions](#)

[References](#)

[Tables](#)

[Figures](#)

[⏪](#)

[⏩](#)

[◀](#)

[▶](#)

[Back](#)

[Close](#)

[Full Screen / Esc](#)

[Printer-friendly Version](#)

[Interactive Discussion](#)

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## Albedo induced radiative forcing from MPB outbreaks in Rocky Mountains

M. Vanderhoof et al.

[Title Page](#)

[Abstract](#)

[Introduction](#)

[Conclusions](#)

[References](#)

[Tables](#)

[Figures](#)

[⏪](#)

[⏩](#)

[◀](#)

[▶](#)

[Back](#)

[Close](#)

[Full Screen / Esc](#)

[Printer-friendly Version](#)

[Interactive Discussion](#)

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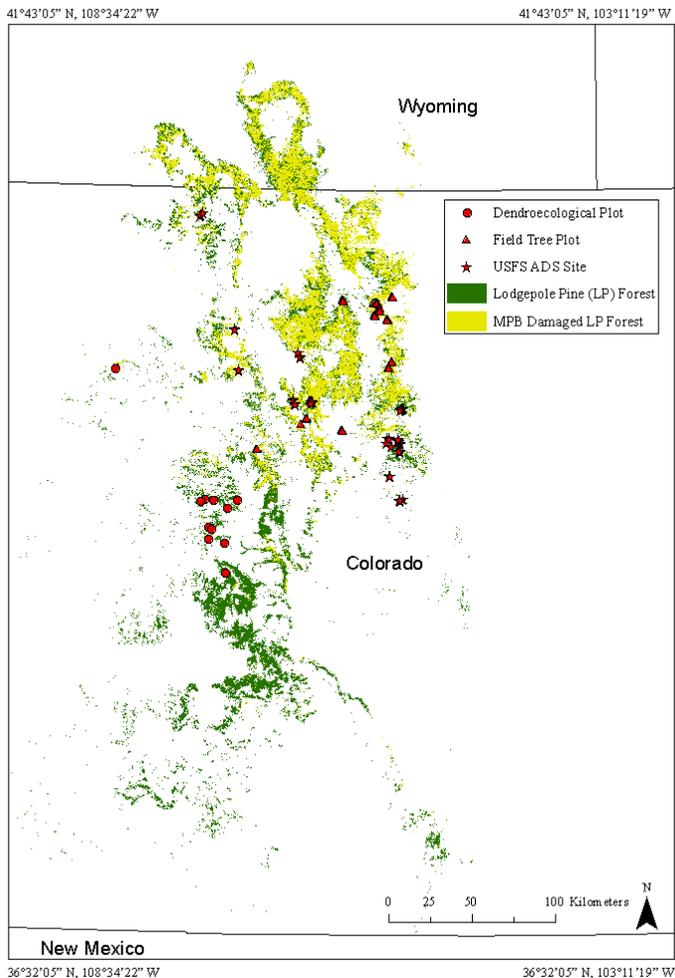
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**Fig. 1.** Plot locations for field tree plots, dendroecological plots and USFS ADS sites.

**Albedo induced radiative forcing from MPB outbreaks in Rocky Mountains**

M. Vanderhoof et al.

[Title Page](#)

[Abstract](#) | [Introduction](#)

[Conclusions](#) | [References](#)

[Tables](#) | [Figures](#)

[◀](#) | [▶](#)

[◀](#) | [▶](#)

[Back](#) | [Close](#)

[Full Screen / Esc](#)

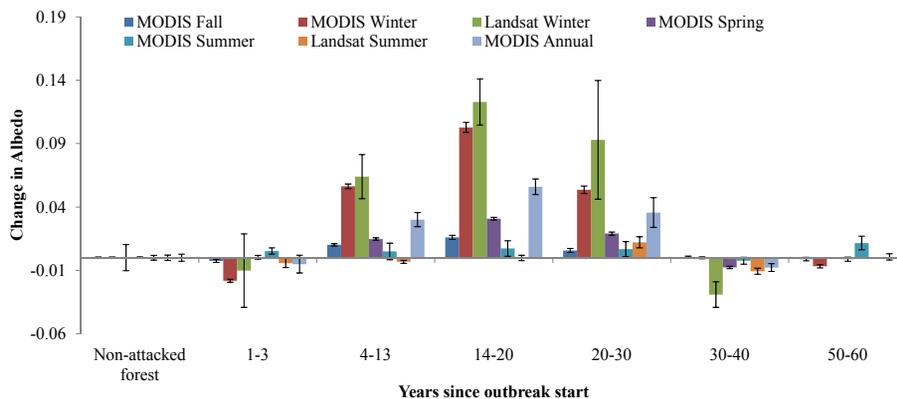
[Printer-friendly Version](#)

[Interactive Discussion](#)



## Albedo induced radiative forcing from MPB outbreaks in Rocky Mountains

M. Vanderhoof et al.



**Fig. 2.** Change in albedo by season, time since outbreak, and satellite source. Error bars indicate plus and minus standard error.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

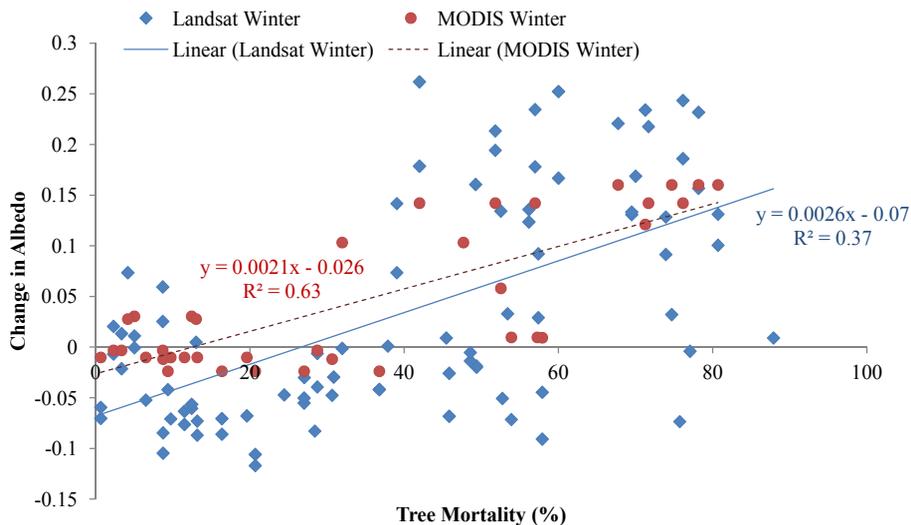
Full Screen / Esc

Printer-friendly Version

Interactive Discussion

**Albedo induced radiative forcing from MPB outbreaks in Rocky Mountains**

M. Vanderhoof et al.

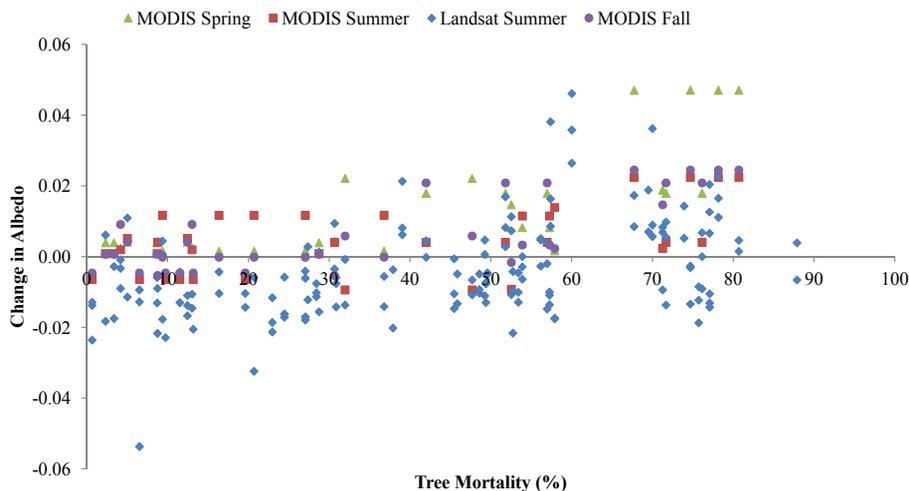


**Fig. 3.** Change in winter albedo with MPB outbreak severity, defined as percent tree mortality, for outbreaks 4 to 13 yr in age.

[Title Page](#)[Abstract](#)[Introduction](#)[Conclusions](#)[References](#)[Tables](#)[Figures](#)[◀](#)[▶](#)[◀](#)[▶](#)[Back](#)[Close](#)[Full Screen / Esc](#)[Printer-friendly Version](#)[Interactive Discussion](#)

**Albedo induced radiative forcing from MPB outbreaks in Rocky Mountains**

M. Vanderhoof et al.

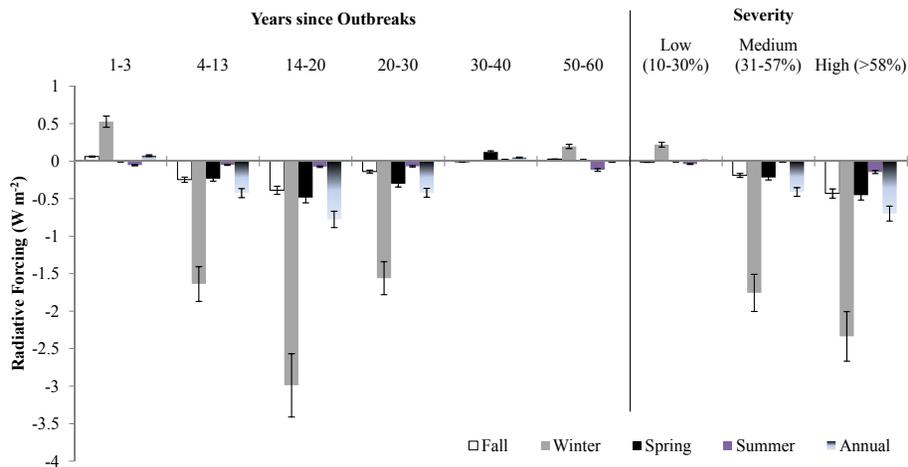


**Fig. 4.** Change in spring, summer and fall albedo with MPB outbreak severity, defined as percent tree mortality, for outbreaks 4 to 13 yr in age.

[Title Page](#)[Abstract](#)[Introduction](#)[Conclusions](#)[References](#)[Tables](#)[Figures](#)[⏪](#)[⏩](#)[◀](#)[▶](#)[Back](#)[Close](#)[Full Screen / Esc](#)[Printer-friendly Version](#)[Interactive Discussion](#)

## Albedo induced radiative forcing from MPB outbreaks in Rocky Mountains

M. Vanderhoof et al.



**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.

[Title Page](#)

[Abstract](#)

[Introduction](#)

[Conclusions](#)

[References](#)

[Tables](#)

[Figures](#)

⏪

⏩

◀

▶

[Back](#)

[Close](#)

[Full Screen / Esc](#)

[Printer-friendly Version](#)

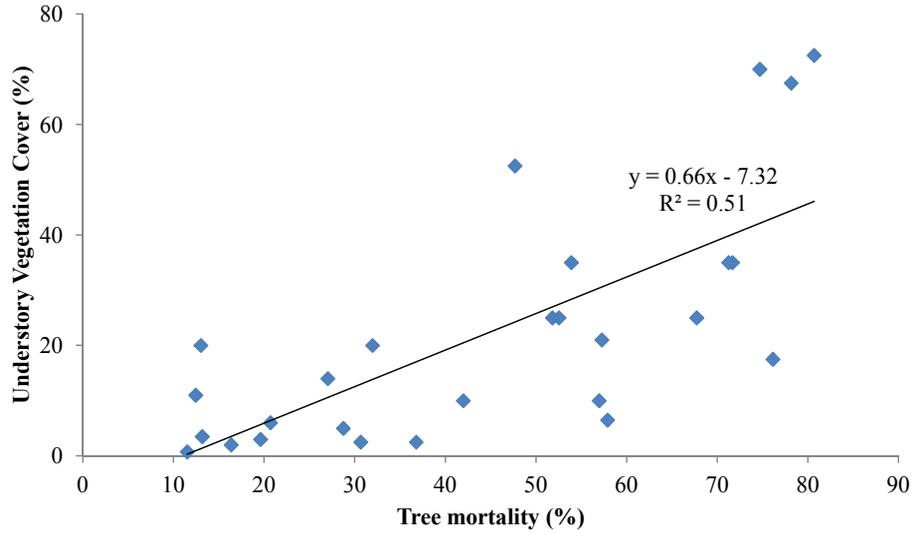
[Interactive Discussion](#)

# BGD

10, 11935–11968, 2013

## Albedo induced radiative forcing from MPB outbreaks in Rocky Mountains

M. Vanderhoof et al.



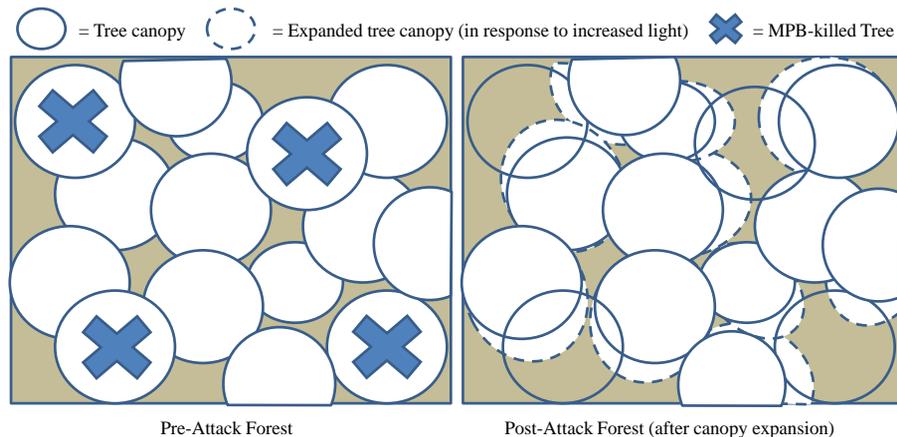
**Fig. 6.** Understory vegetation cover as a function of post-outbreak percent tree mortality, within gray-attack (4–13 yr since outbreak start) dominated field tree plots.

Title Page	
Abstract	Introduction
Conclusions	References
Tables	Figures
⏪	⏩
◀	▶
Back	Close
Full Screen / Esc	
Printer-friendly Version	
Interactive Discussion	



## Albedo induced radiative forcing from MPB outbreaks in Rocky Mountains

M. Vanderhoof et al.



**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.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

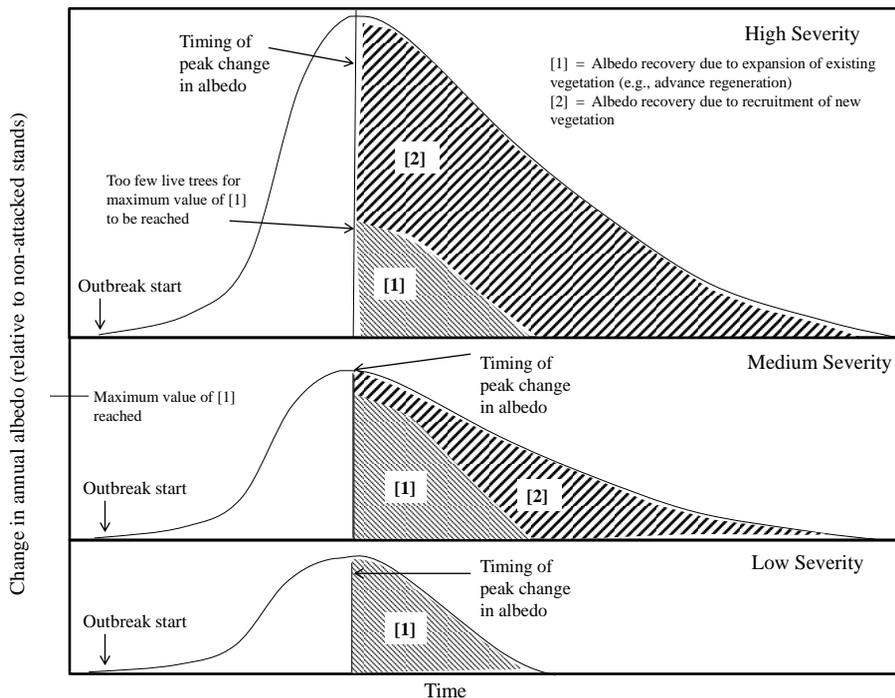
Full Screen / Esc

Printer-friendly Version

Interactive Discussion

## Albedo induced radiative forcing from MPB outbreaks in Rocky Mountains

M. Vanderhoof et al.



**Fig. 8.** The theoretical persistence of change to surface albedo due to outbreak severity and the resulting fractional contribution of each regrowth mechanism. It is assumed that an increase in canopy cover due to the expansion of existing vegetation will occur faster than due to the recruitment of new vegetation, but that some limit exists in the ability for existing vegetation to recover pre-attack canopy cover.

[Title Page](#)
[Abstract](#)
[Introduction](#)
[Conclusions](#)
[References](#)
[Tables](#)
[Figures](#)
[Back](#)
[Close](#)
[Full Screen / Esc](#)
[Printer-friendly Version](#)
[Interactive Discussion](#)