

- Rothermel, R. C. 1972. A mathematical model for predicting fire spread in wildland fuels. Research Paper INT-115. USDA Forest Service, Intermountain Forest and Range Experiment Station, Ogden, Utah, USA.
- Rothermel, R. C. 1991. Predicting behavior and size of crown fires in the Northern Rocky Mountains. Research Paper INT-438. USDA Forest Service, Intermountain Research Station, Ogden, Utah, USA.
- Scott, J. H. 2006. Comparison of crown fire modeling systems used in three fire management applications. Research Paper RMRS-RP-58. USDA Forest Service, Rocky Mountain Research Station, Ogden, Utah, USA.
- Scott, J. H., and R. E. Burgan. 2005. Standard fire behavior fuel models: a comprehensive set for use with Rothermel's surface fire spread model. General Technical Report RMRS-GTR-153. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado, USA.
- Scott, J. H., and E. D. Reinhardt. 2001. Assessing crown fire potential by linking models of surface and crown behavior. Research Paper RMRS-RP-29. USDA Forest Service, Rocky Mountain Research Station, Ogden, Utah, USA.
- Simard, M., W. H. Romme, J. M. Griffin, and M. G. Turner. 2011. Do mountain pine beetle outbreaks change the probability of active crown fire in lodgepole pine forests? *Ecological Monographs* 81:3–24.
- Van Wagner, C. E. 1977. Conditions for the start and spread of crown fire. *Canadian Journal of Forest Research* 7:23–34.
- Van Wagner, C. E. 1989. Prediction of crown fire behavior in conifer stands. Pages 207–212 in D. C. MacIver, H. Auld, and R. Whitewood, editors. *Proceedings of the 10th Conference on Fire and Forest Meteorology*. Environment Canada and Forestry Canada, Ottawa, Ontario, Canada.
- Van Wagner, C. E. 1993. Prediction of crown fire behavior in two stands of jack pine. *Canadian Journal of Forest Research* 23:442–449.
- Van Wagner, C. E. 1998. Modeling logic and the Canadian forest fire behavior prediction system. *Forestry Chronicle* 74:50–52.

Ecology, 93(4), 2012, pp. 941–946
 © 2012 by the Ecological Society of America

Do mountain pine beetle outbreaks change the probability of active crown fire in lodgepole pine forests?

Comment

W. MATT JOLLY,^{1,3} RUSSELL PARSONS,¹ J. MORGAN VARNER,² BRET W. BUTLER,¹ KEVIN C. RYAN,¹ AND COREY L. GUCKER¹

Manuscript received 4 August 2011; revised 3 October 2011; accepted 4 October 2011. Corresponding Editor: J. R. Thompson.

¹ US Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory, Missoula, Montana 59808 USA.

² Department of Forestry and Wildland Resources, Humboldt State University, Arcata, California 95521 USA.

³ E-mail: mjolly@fs.fed.us

An expansive mountain pine beetle (MPB) epidemic is currently impacting North American forests (Raffa et al. 2008). As beetle-attacked trees die, lose their needles, and eventually fall to the ground, there are substantial changes in stand structure. These fuel changes likely affect both surface and crown fire behavior, but there is not yet a consensus among experts regarding the nature and magnitude of these effects (Jenkins et al. 2008). Simard et al. (2011; hereafter referred to as SRTG) used linked crown fire models implemented in the NEXUS crown fire modeling system (Scott and Reinhardt 2001) to predict the occurrence of active crown fire across a chronosequence of increasing time since attack. They concluded that, under moderate fire weather conditions, recently attacked or red-stage stands had a lower occurrence of active crown fire than undisturbed stands. Here, we suggest that these conclusions are compromised because (1) the fire behavior modeling framework used has no mechanisms for considering highly heterogeneous fuels and (2) their use of this framework omitted critical aspects of how canopy and surface fuels change after an attack, particularly during the first two to three years.

Fire behavior modeling framework used was poorly suited to address MPB-wildfire interactions.—The NEXUS (Scott 1999, Scott and Reinhardt 2001) crown fire behavior modeling system calculates the occurrence of active crown fire by linking predictive models of surface fire spread and intensity (Rothermel 1972), crown fire initiation (Van Wagner 1977), and crown fire spread (Rothermel 1991). For the purposes of further discussion, we will refer to this collection of models as the linked crown fire model. It is important to note that these models are also implemented in a number of fire behavior modeling systems, such as the point-based BehavePlus fire modeling system (Andrews et al. 2008) or spatial fire behavior prediction systems such as FARSITE (Finney 2004), FlamMap (Finney 2002), and FVS-FFE (Rebain et al. 2010), and thus our discussions here are not limited to only NEXUS but rather, apply to the entire suite of fire modeling applications that implement these models. These systems are commonly used to support fire management decisions throughout the United States. To understand why these crown fire models may not accurately predict crown fire behavior in MPB attacked stands, some background into their components and the linkages between them is necessary.

These linked models predict active crown fire occurrence in two stages: (1) initiation, in which the fire ascends from the surface into tree crowns, and (2) spread, in which the fire is carried through the canopy. Surface fire intensity is simulated by the Rothermel surface fire spread model (Rothermel 1972), and crown fire spread rate is simulated by the Rothermel crown fire

spread rate model (Rothermel 1991), which is based on a correlation with the surface spread model. For crown fire initiation, surface fire intensity must surpass the model-predicted critical surface fire intensity threshold, which is determined by foliar moisture content (FMC) and effective canopy base height (CBH). When FMC or CBH values are large, higher surface fire intensities are required for crown fire initiation. Active crown fire spread occurs when the model-predicted crown fire spread rate exceeds the critical crown fire rate of spread threshold predicted by the Van Wagner model. Critical crown fire rate of spread is determined solely by canopy bulk density (CBD) or the amount of dry foliar biomass per unit volume of canopy. Critical crown fire rate of spread is sensitive to small changes in CBD, particularly when CBD is below $\sim 0.16 \text{ kg/m}^3$ (Fig. 1A).

Despite a legacy of fire behavior model use, and reliance on them for fire management decision making, there is growing recognition that crown fires are not well modeled by these existing linked model systems and that crown fires in general are poorly understood (see review by Cruz and Alexander [2010]). Additionally, it is recognized that these models are biased toward underpredicting the occurrence of active crown fires (Cruz and Alexander 2010). The fire spread model assumes homogenous, continuous fuels and steady-state flame spread, and these assumptions are inconsistent with the characteristics associated with actively spreading crown fires (Cohen et al. 2006, Finney et al. 2010). Fuel and flame spread assumptions of the fire models are even less applicable to beetle-killed stands, where the mixture of live and dead trees produces substantial spatial variability in fuel continuity, homogeneity, and moisture content. In recently beetle-killed stands, the red-needled foliage has low FMC values, which are far below the empirical data range used by the Van Wagner model (1977). Recognizing that these fire behavior models are likely mischaracterizing expected fire behavior in beetle-killed stands, it should be concluded that simulation results must be interpreted with caution.

Our two principal concerns with SRGT's use and interpretation of the linked crown fire models to predict active crown fire occurrence in beetle-killed stands are that their approach omitted critical aspects of how beetle attacks predictably alter surface and canopy fuels and that they interpreted model outputs without regard to the model's sensitivity to their choice of inputs. We demonstrate throughout the rest of this paper how model predictions and SRGT's conclusions may have been different had they considered the full range of surface and fuel structure changes that occur after a mountain pine beetle attack.

Critical aspects of canopy and surface fuel changes were omitted for stands recently attacked by MPB.—Choosing the linked crown fire models as a fire modeling

platform forced SRGT to make several assumptions about crown fuel dynamics following a beetle attack and these assumptions led to significant under-predictions of crown fire potential in red-stage stands. SRGT estimated crown foliar biomass for all trees, regardless of attack condition, using common allometric equations that relate tree diameter to foliar biomass. All red-needle trees were classified into two categories based on having more than or less than 50% of their original foliage remaining and then each tree's foliar biomass was reduced by 25% and 75% for the two foliar categories, respectively (see SRGT: 9). The modified crown biomass values were then used to estimate stand-level CBD. By applying a subjective biomass reduction to all red-needle trees, SRGT failed to account for stands with trees retaining most of their red needles. This early stage of attack where CBD remains unchanged and the foliar moisture of the trees declines rapidly represents a critical reference point when evaluating the impacts of fuel changes on crown fire potential (Fig. 1C). While foliar biomass loss is expected in beetle-killed trees, this reduction is not instantaneous and needles remain attached for up to three years after an MPB attack (British Columbia Ministry of Forests 1995). Though no research currently exists to show the timing of needle loss after an attack, it is reasonable to assume that most needles are retained for up to two years after an attack and then gradually lost over time until there are no needles remaining on the trees after the third year. These reductions in crown needle biomass would effect a continuous reduction of crown fire potential as an attack progresses rather than an instantaneous or discrete shift from one state (full needle retention) to another (25% or 75% needle loss). Thus, crown fire potential will be highest when needles are dessicated but not lost from the tree and then it will likely decrease as needles fall. The latter case would more closely match predictions made by SRGT, but the former case is the point in the post-attack time sequence where stands would have the highest active crown fire potential.

SRGT's average CBD of 0.16 kg/m^3 for undisturbed stands (estimated from SRGT: Fig. 6D) falls within a region of the crown fire model where small reductions in CBD correspond to large increases in predicted critical crown fire rate of spread (Fig. 1A). A 50% CBD reduction from this value corresponds to a doubling of the critical crown fire rate of spread criteria. Proper characterization of stand CBD is critical to correctly assess that stand's active crown fire potential. SRGT's categorical reduction of foliar biomass greatly reduced their calculated CBD values (0.07 kg/m^3 on average) and consequently reduced model-predicted active crown fire potential for any stand with red needle trees.

Foliar biomass reductions also influenced the canopy-average FMC values applied by SRGT. In a given stand,

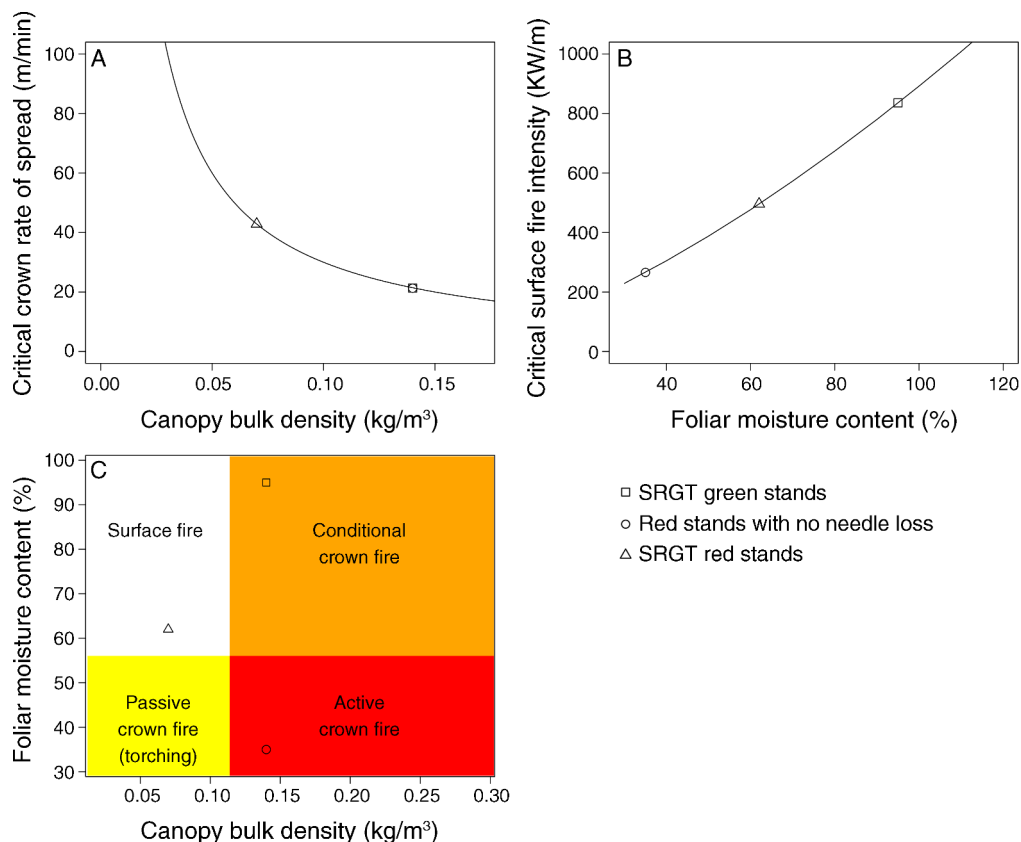


FIG. 1. Relationships between canopy bulk density (CBD), foliar moisture content (FMC), and crown fire criteria/potential (Van Wagner 1977). The square symbol shows the model predictions for a typical SRGT (short for Simard et al. [2011]) green stand with a crown bulk density of 0.16 kg/m^3 and an FMC of 95%. The triangle symbol shows a typical SRGT red-needle stand (CBD = 0.07 kg/m^3 , FMC = 63%). The open circle illustrates the case where an appropriately low FMC is used (35%) but no needles have fallen from the tree and thus CBD has not yet been reduced (CBD = 0.16 kg/m^3). Panel A shows the sensitivity of critical crown fire spread rates to changes in CBD, indicating that as CBD decreases from 0.16 kg/m^3 to 0.07 kg/m^3 , the criteria for active crown fire doubles. Panel B shows the sensitivity of critical surface fire intensity to changes in FMC, and it shows that, as FMC increases (from 63% to 95%), the crown fire initiation criteria nearly triples. Panel C shows that the model consistently predicts active crown fire if one considers a stand where attacked needles have a low FMC and yet have not lost needles (CBD is unchanged). (See Appendix A for more details.)

they used a foliar biomass-weighted average of green and red needle FMC as a linked crown fire model input. When low FMC, red-needle foliar biomass is reduced, a larger proportion of the needles in a given stand are high FMC, green needles. This would overestimate the canopy-average FMC. The lowest canopy-average FMC used by SRGT was 63% (see SRGT:14), which is more than 10 times the moisture content of red needles (as reported in SRGT: Table 2). While this simplification was required because the linked crown fire models only allow a single FMC input for a given stand, this simple averaging is unjustified, given the bimodal nature of green and red needle FMC. These uncharacteristically high FMC values would have erroneously suggested a reduction in the surface-to-crown transition potential of attacked stands relative to what might be expected for trees with very low foliar moisture contents. In recent

years, more detailed physics-based fire models have emerged, such as FIRETEC (Linn et al. 2002, 2005) and the Wildland–Urban Interface Fire Dynamics Simulator (WFDS; Mell et al. 2009) in which individual beetle-killed trees and healthy trees could have different fuel moistures, and fire behavior is sensitive to the geometry and arrangement of the fuel. The substantially higher detail with which both fuels and the physical processes of fire are represented in these models makes them more appropriate than the linked crown fire models used by SRGT when simulating fire behavior in highly heterogeneous, beetle-kill fuels.

Ultimately, inaccurate and insufficient characterization of the full range of foliar biomass possibilities following a MPB attack produced inaccurate CBD and FMC model inputs and an unrealistic decrease in the active crown fire potential of red-needle stands. Using

more representative CBD and conservative FMC values for red-stage stands in the early stages of beetle attacks (FMC = 35% and CBD = 0.016 kg/m³, as denoted by open circles in Fig. 1), we found that the model predicts an active crown fire (Fig. 1C). In fact, we found that any stand with a canopy-average foliar moisture content below ~75% and a CBD above 0.09 kg/m³ would support an actively spreading crown fire under the same environmental conditions used by SRGT (Fig. 1C). The failure by SRGT to recognize, account for, and appropriately characterize expected changes in CBD and FMC in the first two to three years following a MPB attack led to an under-prediction of crown fire potential.

SRGT's assumptions associated with surface fuel characterization also omitted key changes associated with beetle attacks, which reduced the chance of predicting active crown fire potential. In the linked crown fire models modeling system, surface fuels influence crown fire potential by influencing surface fire intensity, which must exceed a critical value for crown fire initiation to occur. Although they reported significant increases in litter depth between undisturbed and red-stage stands, these fuel changes were not included in the custom surface fuel models used by SRGT in their fire behavior calculations. Assuming a litter bulk density of 11 kg/m³ (consistent with reported literature [Brown 1981]), and the average litter depths reported in undisturbed (1.6 cm), red-stage (2.6 cm), and gray-stage (2.3 cm) stands, 1-hour fuel loads would increase by 1.76, 2.86, and 2.31 Mg/ha, respectively, in beetle-killed stands.

When we compared model-predicted surface fire intensities under SRGT's dry summer fuel moisture and open wind (20 km/h) scenario with and without litter included in the 1-hour fuel category, the incorporation of litter increased model-predicted surface fire intensities by 78%, 85%, and 95% for undisturbed, red-stage stands, and gray-stage stands, respectively (see Appendix B: Table B2). Increased surface fire intensities, as result of including litter as fine dead surface fuels, would increase surface fire intensity and thus would increase predicted crown fire initiation.

Space-for-time chronosequence sampling introduced fuels variability and resulted in an incomplete characterization of fuel changes that occur as time since MPB attack increases.—With the exception of an increase in litter depth between undisturbed and red-stage stands, SRGT found no significant changes in any surface fuel category over the time-since-beetle attack chronosequence. We suggest that the lack of significant surface fuel changes reflects the high variability introduced by sampling stands along a space-for-time chronosequence, rather than inconsequential changes over time in a given location as SRGT suggests.

If individual trees and their immediate surroundings were monitored over time following an attack, a very

predictable sequence of events would be observed, with needles falling after two to three years, followed by twigs, branches, and eventually large branches and tree trunks. This shedding of canopy fuels to the forest floor would increase surface fuel loads, particularly in the early years following a beetle attack. Beyond the initial years following beetle attacks, however, potential decomposition losses and rates are highly influenced by species characteristics (Cornwell et al. 2008) and biophysical setting (Keane 2008). Smaller diameter fuel components have faster decay rates (Keane 2008) and thus are less persistent over time.

In the SRGT study, expected patterns of surface fuel recruitment are not recognized. Significant losses in canopy fuels (SRGT: Fig. 6D and E) are not matched by increases in surface fuels (SRGT: Fig. 4), so some fuel components appear to vanish over time. Surface fuel abundance was not found to be significantly different between attacked and undisturbed stands (SRGT: Fig. 4B, C, D, and G). One potential explanation for the missing fuels is the intrinsic high variability in surface fuels (Woldendorp et al. 2004). High variability is problematic for fuels sampling in general; accuracy assessment studies of commonly used line intercept methods, such as those employed in this study, demonstrated that transect lengths must be an order of magnitude longer (>1000 m) than those employed by SRGT (100 m) in order to achieve confidence intervals below 25% (Pickford and Hazard 1978, Sikkink and Keane 2008). Shorter lengths may be adequate for litter (Sikkink and Keane 2008), which may explain why this variable alone was significant in SRGT's study. This high variability could result in significant inaccuracy in SRGT's fuel quantity estimates, which could substantially obscure effects over time, even if they had sampled the same stands over time. Additional issues likely arise from pitfalls associated with chronosequence sampling approaches, which include blurring small-scale heterogeneity by seeking plot-scale homogeneity or magnifying inherent site differences across heterogeneous space (Pickett 1989). Increasing surface fuel variability beyond what is already an intrinsic component of surface fuels could have resulted in substantial inaccuracy in SRGT's surface fuel load estimates and obscured fuel changes over time.

Conclusion

A robust understanding of the nature of fuel and fire behavior changes over time following beetle attack is essential for fire management, risk assessments, and successful long-term ecosystem management. Conclusions reported by SRGT, however, obscure our understanding of these changes because they fail to capture critical surface and crown fuel changes as a MPB attack progresses through a given stand. Use of a fire behavior

modeling framework that has no mechanism for considering highly heterogeneous fuels, omission of critical canopy and surface fuel changes in stands attacked by MPB, and potential increases in the inherent variability associated with fuels sampling in general and along a chronosequence, led to the predicted reduction in the crown fire potential in red-needle stands under the moderate conditions reported by SRGT. As we have demonstrated, if SRGT had chosen estimates of surface and canopy fuel inputs that better described stands during the early stages of a MPB attack, inevitably they would have found an increase in predicted crown fire potential in almost all cases, particularly during the first three years after an attack (Fig. 1C).

As the MPB epidemic continues in fire-prone North American forests, the scientific community must quantify expected changes in fire behavior in a manner that increases the understanding of pathogen and fire interactions. Some form of fire modeling will be associated with these studies. However, these models should be evaluated against observed fire behavior when possible (Alexander and Cruz 2006). In the absence of field-based fire behavior observations, fire models still play a role in predicting the potential impacts of any type of fuel modifications on expected fire behavior. However, great care is needed when choosing model inputs and evaluating model outputs, and the reliability and limitations of fire behavior model predictions must always be considered and explicitly discussed when reporting simulation results.

Literature cited

- Alexander, M. E., and M. G. Cruz. 2006. Evaluating a model for predicting active crown fire rate of spread using wildfire observations. *Canadian Journal of Forest Research* 36:3015–3028.
- Andrews, P. L., C. D. Bevins, and R. C. Seli. 2008. BehavePlus fire modeling system Version 4.0 User's Guide. General Technical Report, RMRS-GTR-106WWW Revised. U.S. Forest Service, Rocky Mountain Research Station, Ogden, Utah, USA.
- British Columbia Ministry of Forests. 1995. Bark beetle management guidebook. Forest Practices Branch, Victoria, British Columbia, Canada.
- Brown, J. K. 1981. Bulk densities of non-uniform surface fuels and their application of fire modeling. *Forest Science* 27:667–683.
- Cohen, J. D., M. A. Finney, and K. M. Yedinak. 2006. Active spreading crown fire characteristics: implications for modeling. *Forest Ecology and Management* 234 Supplement:S95.
- Cornwell, W. K., J. H. C. Cornelissen, K. Amatangelo, E. Dorrepaal, V. T. Eviner, O. Godoy, S. E. Hobbie, B. Hoorens, H. Kurokawa, and N. Pérez Harguindeguy. 2008. Plant species traits are the predominant control on litter decomposition rates within biomes worldwide. *Ecology Letters* 11:1065–1071.
- Cruz, M. G., and M. E. Alexander. 2010. Assessing crown fire potential in coniferous forests of western North America: a critique of current approaches and recent simulation studies. *International Journal of Wildland Fire* 19:377–398.
- Finney, M. A. 2002. Fire growth using minimum travel time methods. *Canadian Journal of Forest Research* 32:1420–1424.
- Finney, M. A. 2004. FARSITE: fire area simulator-model development and evaluation. Research Paper RMRS-RP-4. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ogden, Utah, USA.
- Finney, M. A., J. D. Cohen, I. C. Grenfell, and K. M. Yedinak. 2010. An examination of fire spread thresholds in discontinuous fuel beds. *International Journal of Wildland Fire* 19:163–170.
- Jenkins, M. J., E. Hebertson, W. Page, and C. A. Jorgensen. 2008. Bark beetles, fuels, fires and implications for forest management in the Intermountain West. *Forest Ecology and Management* 254:16–34.
- Keane, R. E. 2008. Biophysical controls on surface fuel litterfall and decomposition in the northern Rocky Mountains, USA. *Canadian Journal of Forest Research* 38:1431–1445.
- Linn, R., J. Reisner, J. J. Colman, and J. Winterkamp. 2002. Studying wildfire behavior using FIRETEC. *International Journal of Wildland Fire* 11:233–246.
- Linn, R., J. Winterkamp, J. J. Colman, C. Edminster, and J. D. Bailey. 2005. Modeling interactions between fire and atmosphere in discrete element fuel beds. *International Journal of Wildland Fire* 14:37–48.
- Mell, W., A. Maranghides, R. McDermott, and S. Manzello. 2009. Numerical simulation and experiments of burning Douglas fir trees. *Combustion and Flame* 156:2023–2041.
- Pickett, S. T. A. 1989. Space-for-time substitution as an alternative to long-term studies. Pages 110–135 in G. E. Likens, editor. *Long-term studies in ecology: approaches and alternatives*. Springer-Verlag, New York, New York, USA.
- Pickford, S. G., and J. W. Hazard. 1978. Simulation studies on line intersect sampling of forest residue. *Forest Science* 24:469–483.
- Raffa, K. F., B. H. Aukema, B. J. Bentz, A. L. Carroll, J. A. Hicke, M. G. Turner, and W. H. Romme. 2008. Cross-scale drivers of natural disturbances prone to anthropogenic amplification: the dynamics of biome-wide bark beetle eruptions. *BioScience* 58:501–517.
- Rebain, S. A., E. D. Reinhardt, N. L. Crookston, S. J. Beukema, W. A. Kurz, J. A. Greenough, D. C. E. Robinson, and D. C. Lutes. 2010. The fire and fuels extension to the forest vegetation simulator: updated model documentation. U.S. Department of Agriculture, Forest Service, Forest Management Service Center, Fort Collins, Colorado, USA.
- Rothermel, R. C. 1972. A mathematical model for predicting fire spread in wildland fuels. INT-115. U.S. Department of Agriculture, Forest Service, Ogden, Utah, USA.
- Rothermel, R. C. 1991. Predicting behavior and size of crown fires in the Northern Rocky Mountains. Research Paper INT-438. U.S. Department of Agriculture, Forest Service, Intermountain Research Station, Ogden, Utah, USA.
- Scott, J. H. 1999. NEXUS: a system for assessing crown fire hazard. *Fire Management Notes* 59:20–24.
- Scott, J. H., and E. D. Reinhardt. 2001. Assessing crown fire potential by linking models of surface and crown fire behavior. Research Paper RMRS-RP-29. U.S. Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado, USA.
- Sikkink, P. G., and R. E. Keane. 2008. A comparison of five sampling techniques to estimate surface fuel loading in montane forests. *International Journal of Wildland Fire* 17:363–379.
- Simard, M., W. H. Romme, J. M. Griffin, and M. G. Turner. 2011. Do mountain pine beetle outbreaks change the probability of active crown fire in lodgepole pine forests? *Ecological Monographs* 81:3–24.
- Van Wagner, C. E. 1977. Conditions for the start and spread of a crown fire. *Canadian Journal of Forest Research* 7:23–34.
- Woldendorp, G., R. J. Keenan, S. Barry, and R. D. Spencer. 2004. Analysis of sampling methods for coarse woody debris. *Forest Ecology and Management* 198:133–148.

SUPPLEMENTAL MATERIAL

Appendix A

Developing data to create Fig. 1 (*Ecological Archives* E093-082-A1).

Appendix B

Predicted fire behavior both with and without litter (*Ecological Archives* E093-082-A2).

Ecology, 93(4), 2012, pp. 946–950
© 2012 by the Ecological Society of America

Do mountain pine beetle outbreaks change the probability of active crown fire in lodgepole pine forests? Reply

MARTIN SIMARD,^{1,5} WILLIAM H. ROMME,² JACOB M. GRIFFIN,³ AND MONICA G. TURNER⁴

In a recent paper, we showed that contrary to conventional wisdom, outbreaks of the mountain pine beetle (*Dendroctonus ponderosae*) did not increase the predicted potential for active crown fires relative to comparable undisturbed stands in lodgepole pine (*Pinus contorta* var. *latifolia*) forests of Greater Yellowstone (Simard et al. 2011). Under certain intermediate weather conditions, the potential for active crown fires was actually predicted to decrease relative to comparable stands undisturbed by bark beetles. Our study was based on a rich data set of forest fuels sampled in a robust post-outbreak chronosequence of beetle-killed and undisturbed stands that were replicated in space and time and validated with dendrochronological reconstructions of pre-outbreak conditions. Moran and Cochrane (2012) and Jolly et al. (2012) have expressed concerns about certain aspects of our fire behavior modeling in recently killed stands. Although these authors do not refute the conclusions of our study, they do raise issues that are

highly debated among scientists and practitioners alike. Here we explain that these concerns are largely based on misconceptions about the dynamics of mountain pine beetle (MPB) outbreaks, and we provide complementary data that strengthen our conclusions.

Three misconceptions

Severity of beetle-caused tree mortality.—A first misconception about MPB outbreak dynamics by Jolly et al. (2012) and Moran and Cochrane (2012) is that many lodgepole pine forests attacked by the MPB sustain extremely high mortality—sometimes up to 100% of the basal area or number of trees within a stand—and that such areas of extremely high mortality are common across the landscape. Surprisingly, given how conspicuous beetle-killed trees are in affected landscapes, this notion is rarely substantiated by quantitative data. Mortality of 100% may be observed at very fine scales (a few square meters) but is rarely observed at scales > 1 ha (Rocca and Romme 2009), which are the scales relevant to consideration of fire behavior. In British Columbia, aerial surveys of the MPB outbreak, which peaked in 2005 and is now receding, show that cumulative (1999–2010) mortality of 100% (at a spatial resolution of 16 ha) is observed in <1% of the affected area, and mortality levels > 85% occurred on only 5% of the affected landscape (Walton 2011). Similarly, after reviewing published studies from the U.S. Rocky Mountains, we were unable to find study sites that sustained more than 85% mortality, either on a tree density or a basal area basis (Roe and Amman 1970, Amman and Baker 1972, Amman et al. 1973, Klutsch et al. 2009, Collins et al. 2010, 2011, Diskin et al. 2011). A corollary of these results is that the mortality levels of the stands that we studied (up to 82% basal area beetle killed) are within the range of mortality observed in 95% of MPB-affected areas.

Timing of beetle-caused tree mortality.—A second misconception evident in both comments is that infestations within a given stand occur over such a short period of time that beetle-caused tree mortality is synchronous. On the contrary, MPB infestations develop gradually, typically over several years, and tree mortality for any given year is always considerably less than the total mortality accumulated over the course of the outbreak. Data from the British Columbia outbreak (1999–2010) showed that annual mortality was >50% on <5% of the affected area during nearly the entire

Manuscript received 14 November 2011; accepted 23 November 2011. Corresponding Editor: J. R. Thompson.

¹ Department of Geography and Centre for Northern Studies, Université Laval, Québec City, Québec G1V 0A6 Canada.

² Natural Resource Ecology Laboratory, Colorado State University, Fort Collins, Colorado 80526 USA.

³ Biological Sciences Department, Edgewood College, Madison, Wisconsin 53711 USA.

⁴ Department of Zoology, University of Wisconsin, Madison, Wisconsin 53706 USA.

⁵ E-mail: martin.simard@ggr.ulaval.ca