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Using structural sustainability for forest health monitoring and triage: Case study of a mountain pine beetle (*Dendroctonus ponderosae*)-impacted landscape

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ABSTRACT

Heavy disturbance-induced mortality can negatively impact forest biota, functions, and services by drastically altering the forest structures that create stable environmental conditions. Disturbance impacts on forest structure can be assessed using structural sustainability—the degree of balance between living and dead portions of a tree population's size-class distribution—which is the basis of baseline mortality analysis. This analysis uses a conditionally calibrated reference level (i.e., baseline mortality) against which to detect significant mortality deviations without need for historical references. Recently, a structural sustainability index was developed by parameterizing results of this analysis to allow spatial and temporal comparisons within or among forested landscapes. The utility of this index as a tool for forest health monitoring programs and triage decisions has not been examined. Here, we investigated this utility by retrospectively analyzing the structural sustainability of a mountain pine beetle (*Dendroctonus ponderosae*)-impacted, lodgepole pine (*Pinus contorta*)-dominated landscape annually from 2000 to 2006 as well as among watersheds. We show that temporal patterns of structural sustainability at the landscape-level reflect accumulating beetle-induced mortality as well as beetle-lodgepole pine ecology. At the watershed-level, this sustainability spatially varied with 2006 beetle-induced mortality. Further, structural sustainability satisfies key criteria for effective indicators of ecosystem change. We conclude that structural sustainability is a viable tool upon which to base or supplement forest health monitoring and triage programs, and could potentially increase the efficacy of such programs under current and future climate change-associated disturbance patterns.

1. Introduction

Global forest health is influenced by natural and anthropogenic disturbances (Gauthier et al., 2015; Millar and Stephenson, 2015). The impact and prevalence of some biotic (e.g., insect and disease outbreaks) and abiotic (e.g., drought and wildfire) disturbances are predicted to increase in severity and frequency in coming decades in response to climate change factors (Allen et al., 2015, 2010; Gauthier et al., 2015; McDowell and Allen, 2015; Millar and Stephenson, 2015; Wotton et al., 2010). While some level of disturbance is necessary for maintaining forest communities, more frequent or severe disturbances may surpass the resistance—the capacity to endure disturbances without substantial change to the existing community—and resilience—the ability of a community to recover from disturbance-induced change—thresholds of affected forests, potentially resulting in substantial mortality. This mortality inevitably alters forest structure as trees of certain sizes are killed (Pommerening, 2006). However, forest structure is integral in defining the forest environment because dominant trees are foundation species (Ellison et al., 2005). This environment can change in response to altered forest structure, potentially causing cascading positive or negative effects on wildlife and plant populations as habitat suitability changes (Cale et al., 2013; Foster et al., 2002). Further, altered structure can affect forest ecosystem functions

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and services. For example, recent climatic shifts have facilitated unprecedented mountain pine beetle (*Dendroctonus ponderosae* Hopkins; Coleoptera, Curculionidae; MPB) outbreaks in western North America, resulting in substantial changes in forest structure and function including reduced carbon sequestration and storage, altered watershed hydrology, and hindered forest regeneration potential by reducing fungal-mutualist diversity (Bearup et al., 2014; Kurz et al., 2008; Treu et al., 2014). These qualities of a healthy forest are maintained by stable environments created by a lack of disturbance, or natural disturbance regimes (Ellison et al., 2005; Pommerening, 2006). Thus, significant changes to these qualities could be predicted to occur following major disruptions, such as from novel disturbance events or regimes, to forest structure.

Teale and Castello (2011) define a healthy forest as one that satisfies ecological and/or economic management objectives and is structurally sustainability while ultimately considering the ecology of the species evaluated. Structural sustainability is the degree of balance between living and dead portions of a tree population's size-class distribution, and conceptually predicts that biotic and abiotic components are stable in forests where mortality and regeneration/growth are balanced (i.e., structurally sustainable) because demographic development and turn-over occurs unhindered (Castello et al., 2011; Duchesne et al., 2005; Manion and Griffin, 2001; Manion and Rubin, 2001; Teale and Castello, 2011). This balance is important as forest mortality increases space and resource availability to growing trees, allowing them to optimize growth and density for a given site. Although historical disturbance regimes (high-impact, infrequent or low-impact, frequent) are essential to natural forest dynamics, long-term shifts in the mortality-growth/survival balance could be a consequence of novel disturbances or changes to natural disturbance regimes (Bergeron et al., 1999; Teale and Castello, 2011). Imbalance can be evaluated from landscape-level census data of individual tree species or mixed-species forests using baseline mortality analysis (BMA). Using tree size (diameter) classes and density in each class, BMA compares observed mortality levels to a conditional reference level derived from and calibrated to the distribution of living trees (i.e., baseline mortality) (Manion and Griffin, 2001; Teale and Castello, 2011). The statistical significance of differences between observed and baseline mortality is tested for each diameter class to identify portions of the size distribution potentially experiencing overcrowding (i.e., observed mortality less than baseline) or heavy killing-agent activity (i.e., observed mortality greater than baseline) (Castello et al., 2011).

While BMA is useful in identifying diameter classes with mortality-growth/survival imbalances, it does not objectively determine the structural sustainability of the distribution as a whole (i.e, all size classes together). Similarly, BMA results alone cannot be used to evaluate the relative structural sustainability among species nor over time or among regions for a single species. A structural sustainability index (SSI) was recently developed by Cale et al. (2014) to address these limitations by parameterizing several aspects of the BMA results, such as how clustered or numerous diameter classes with significant differences between observed and baseline mortality are in these results. This index does not predict a static estimate of structural sustainability but instead an estimate for a given point in time, which could be recalculated over time given new data. Further, SSI can discriminate between structurally sustainable and unsustainable forests. The conditionally-calibrated baseline mortality foundation (Teale and Castello, 2011) of SSI may make this index ideally suited for monitoring disturbance-associated changes in forest conditions (both biotic and abiotic) over time as climate change undermines the continued utility of historical references of healthy tree structures to monitoring programs. Similarly, SSI may be an equally valuable tool to prioritizing the allocation of forest management resources (i.e., ecosystem triage) by allowing managers to compare relative disturbance impacts among sites. However, the utility of SSI for these purposes has not been investigated.

Mountain pine beetle is a tree-killing insect native to western North America whose primary host is lodgepole pine (*Pinus contorta* Douglas ex P. Lawson & C. Lawson) (Amman and Cole, 1983; Amman, 1977; Safranyik and Carroll, 2006). Although MPB-associated mortality is minimal when beetle populations are at endemic levels, landscape-level mortality occurs when beetle populations reach outbreak densities and overcome defense mechanisms of healthy trees by attacking *en masse* (Raffa et al., 2008; Safranyik and Carroll, 2006). While considered invasive in parts of Canada (Cullingham et al., 2011; Erbilgin et al., 2014; Safranyik et al., 2010) where it has killed millions of hectares of pine forest, the beetle is a natural mortality agent in its native range where beetle outbreaks, as well as fire, are an essential part of the ecosystem that help regulate forest structure through stand-initiating mortality levels (Amman, 1977; Roe and Amman, 1970).

Here, we retrospectively analyzed the structural sustainability of a lodgepole pine-dominated landscape in Colorado during a MPB outbreak to investigate two questions: is SSI a viable tool for monitoring forest-change, and is SSI a useful platform upon which to base forest triage decisions? Further, we had two specific objectives: evaluate the utility of SSI for monitoring forest-change by assessing index score response to MPB-induced mortality over time, and for aiding triage decisions by comparing this mortality to SSI among several watersheds for a given year. The MPB-lodgepole pine system affords some advantages over other disturbance systems for this type of retrospective analysis: (1) the year in which each tree was killed by MPB can be estimated with high relative accuracy (Keen, 1955; Klutsch et al., 2009), (2) during an outbreak MPB is the dominant mortality agent allowing us to attribute forest structural changes solely to beetle activity, and (3) annual beetle-induced mortality follows a predictable exponential increase before declining after nearly all susceptible hosts have been killed (Raffa et al., 2008; Safranyik and Carroll, 2006). Further, the ability to accurately estimate mortality year allows us to recreate pre-outbreak structures by including MPB-killed trees among counts of living trees.

2. Materials and methods

2.1. Study site and data collection

We used data collected from lodgepole pine-dominated forests in the Sulphur Range District, Arapaho-Roosevelt National Forest in eastern Grand County (approximately 40°4′N, 106°0′W), Colorado. A MPB outbreak affected at least 90% of these lodgepole pine forests between 2000 and 2007 (USDA Forest Service, 2007). Lodgepole pine-dominated forest covers approximately 54% (158,000 ha) of the eastern Grand County study area, occurring at 2500–3500 m above sea level.

In 2006 and 2007, 221 plots (0.02 ha each) were established on US Forest Service land using a geographic information system (GIS) to select plot locations in areas with and without MPB infestation in lodgepole-dominated forests (Klutsch et al., 2009). Plots were a minimum of 400 m from each other and from roads. A total of 51 plots were classified non-infested, while the remaining 170 had varying amounts of infested lodgepole pine and infestation periods in 2000–2006. Although this dataset included other species, we extracted and analyzed records only for lodgepole pine trees with a diameter at breast height (DBH) $\geq$ 5.0 cm, which represented 69% of all species of all sizes and included DBH measurements, health status (live, dead from unknown cause, killed by MPB), and year of MPB infestation.