esa

ECOSPHERE

ECOSPHERE NATURALIST

If a tree falls in a forest: implications of forest structure persistence for the Pacific marten (*Martes caurina*)

Matthew S. Delheimer[1,†] Katie M. Moriarty,[1,3] Mark A. Linnell,[1] and Bryce V. Woodruff[2]

[1]*Corvallis Forestry Sciences Laboratory, USDA Forest Service, Pacific Northwest Research Station, Corvallis, Oregon 97331 USA*
[2]*Oak Ridge Institute for Science and Education, 100 ORAU Way, Oak Ridge, Tennessee 37830 USA*
[3]*National Council for Air and Stream Improvement, 720 SW 4th Street, Corvallis, Oregon 97333 USA*

Citation: Delheimer, M. S., K. M. Moriarty, M. A. Linnell, and B. V. Woodruff. 2019. If a tree falls in a forest: implications of forest structure persistence for the Pacific marten (*Martes caurina*). Ecosphere 10(8):e02819. 10.1002/ecs2.2819

Abstract. A global decline in large trees, largely driven by anthropogenic influences, has potentially dire consequences for cavity-dependent wildlife. For several species of conservation concern in western North America, including Pacific martens (*Martes caurina*), fishers (*Pekania pennanti*), and spotted owls (*Strix occidentalis*), large live and dead trees (snags) with cavities are forest structural elements that are requisite to fulfilling life history requirements. More than a century may be necessary for trees to develop cavities of a suitable size for use by martens, fishers, and owls, yet the relative amount of time that such trees persist as usable prior to senescence is largely unknown, despite the implications for how forests are managed as habitat for these wildlife species. To address this uncertainty, in 2016, we re-located 44 live trees and snags identified as marten rest structures between 2009 and 2012 in the Lassen National Forest, California. Upon re-location, we assessed structure persistence by determining whether structures had changed type (e.g., a live tree that died and transitioned to a snag) or usability (any structure that was degraded to a point where it was no longer usable by a marten). After 5–7 yr, 25% (n = 11) of marten rest structures changed type and 16% (n = 7) appeared to be unusable—a stark contrast compared to a similar study investigating persistence of fisher rest structures, which documented one-third of the change in structure type and no change in usability over roughly twice the return interval (8–12 yr). Our results emphasize the vast disparity between the lengthy period required for trees to become suitable for marten use and the potentially short period for which they may remain usable. We suggest that forest managers consider the importance of these fine-scale forest structural elements when developing broad-scale strategies to reduce fire risk and improve resilience in western North American forests.

Key words: cavity; fisher; forest management; marten; recruitment; structure persistence; tree loss.

Received 14 May 2019; accepted 17 May 2019; final version received 21 June 2019. Corresponding Editor: Debra P. C. Peters.

† **E-mail:** mdelheimer@fs.fed.us

In the absence of catastrophic disturbances such as wildfire, cycles of tree growth, decadence, and senescence form the base of rich ecological communities (Franklin et al. 1987). Over sufficiently long time periods and broad spatial scales, natural processes of attrition and recruitment should be roughly in equilibrium—as trees die, fall to the ground, and rot into the earth, other trees grow in size and eventually take their place. However, populations of large, old trees are in global decline (Lindenmayer et al. 2012*a*, Lindenmayer and Laurance 2017), driven by loss from timber harvest, disrupted fire regimes, drought, and conversion of forest to agriculture or urban development (Lindenmayer et al. 2012*b*, Jones et al. 2016). Tree loss can be

complete and almost instantaneous, such as in a forest stand that has been clearcut or burned at high intensity; conversely, growth and recruitment of a new cohort of trees to replace those lost can take multiple decades to centuries (Franklin et al. 2002, Hennon and Mulvey 2014). Recruitment of large, old trees worldwide appears to be far outpaced by loss, as a result of asymmetry in the amount of time over which these processes, respectively, occur (Lindenmayer et al. 2012*b*, Easterday et al. 2018).

Loss of large trees presents potentially dire consequences for cavity-dependent wildlife. In western North America, large, old, live trees and dead trees (snags) with cavities are critical forest structural elements for several species of conservation concern, including Pacific martens (*Martes caurina*), fishers (*Pekania pennanti*), and spotted owls (*Strix occidentalis*). Cavity-bearing live trees and snags provide martens and fishers with buffering from temperature extremes and refuge from larger-bodied predators (Taylor and Buskirk 1994, Green et al. 2018) and are particularly important for housing altricial kits (Ruggiero et al. 1998, Weir et al. 2012). In the Pacific Northwest, live trees and snags used by martens and fishers for resting (hereafter, "rest structures") are frequently in the largest diameter size classes available (Raphael and Jones 1997, Zielinski et al. 2004), may be hundreds of years old (Zielinski et al. 2004, Slauson and Zielinski 2009), and are, almost by definition, "big, sick, and rotting" (Weir et al. 2012).

Cavity creation is a lengthy process, as trees must first grow to sufficient diameter, and then undergo processes of natural decay or activity of primary cavity excavators such as woodpeckers (Manion 1981) to develop cavities of a suitable size for martens and fishers to use. Despite this lengthy development period, the amount of time for which live trees or snags persist in a state of usability before senescence generally remains unknown. To begin to address this uncertainty, Zielinski and Schlexer (2015) re-located 49 live trees and 21 snags identified as fisher rest structures approximately 8–12 yr prior in northwestern California (Zielinski et al. 2004). Upon re-location, the authors determined rest structure persistence by assessing changes relative to type (e.g., a live tree that died and transitioned to a snag, or a snag that fell and transitioned to a log) and usability (whether or not a structure was still usable by a fisher). While a small percentage of fisher rest structures (8%) changed type, including three live trees that transitioned to snags, two snags that transitioned to logs, and one live tree that transitioned to a log, no structures appeared to be decomposed or degraded to a point where a fisher could no longer use it (Zielinski and Schlexer 2015).

Our objective was similar: to determine the persistence of marten rest structures by assessing changes in type and usability. Between 2009 and 2012, we identified 44 marten rest structures (Fig. 1) in the Lassen National Forest, California, including 31 snags and 13 live trees; in November 2016, we re-located all 44 structures, approximately 5–7 yr after initial identification. Of 44 marten rest structures, 25% ($n = 11$) changed type, including two live trees that transitioned to snags (15% of all live trees) and nine snags that transitioned to logs (29% of all snags). Of the 11 structures that changed type, 78% also appeared to be unusable by a marten, consisting of seven snags that transitioned to logs.

Both marten rest and fisher rest structures were large in diameter ($\bar{x} > 95$ cm and $\bar{x} > 100$ cm, respectively) and likely old in age. However, we documented almost triple the change in type of marten rest structures, over roughly one-half the amount of time, compared with fisher rest structures. Further, while live trees that transitioned to snags appeared to retain their usability in both studies, snags that transitioned to logs in our study (representing one-sixth of all structures) largely became unusable, as they were characterized by extensive heart rot and typically disintegrated upon falling (Fig. 2a, b). Given that live trees with cavities generally persist longer than snags with cavities (Lindenmayer and Wood 2010, Edworthy et al. 2012, Edworthy and Martin 2014), the discrepancies between our study and Zielinski and Schlexer (2015) could simply be a representation of marten and fisher structure type preferences, as martens selected for snags (70%) over live trees (30%), while fishers selected for live trees (70%) over snags (30%). In other words, it may be more likely for a marten rest structure to change type or become unusable than a fisher rest structure, because martens prefer snags, and snags are in more advanced stages of decay than live trees.

Fig. 1. Marten M01 peers out of a cavity in 2010. This red fir (*Abies magnifica*) snag is well-representative of marten rest structures in the Lassen National Forest—large in size, old, and decadent.

Regardless, our results emphasize the vast disparity between the amount of time it takes for structures to become usable by martens and to the amount of time for which they remain usable and are subsequently lost to senescence. This time disparity may be of particular consequence for martens and other cavity-dependent wildlife whenever recruitment is insufficient to keep pace with loss—for example, spotted owls may already be in an "extinction debt" (Kuussaari et al. 2009) in California, where owl populations are in continued decline following widespread removal of large trees and a subsequent reduction in high canopy cover during timber harvest, even though harvest occurred more than 25 yr ago (Jones et al. 2018). Marten populations in California appear to be in decline as well (Zielinski et al. 2005); at the Sagehen Experimental Forest, marten detections decreased by >60% in less than 30 yr (Moriarty et al. 2011), with decreased detections coinciding with increased timber harvest. While numerous studies have indicated that martens are sensitive to broad-scale disturbances from timber harvest (e.g., reduced canopy cover and forest fragmentation; Bissonette et al. 1997, Hargis et al. 1999, Potvin et al. 2000), the effects of finer-scale disturbances, such as reduced availability of individual large-diameter live trees and snags, are poorly understood (Purcell et al. 2012).

Over the last 150 yr, timber harvest has dramatically reduced numbers of large-diameter trees and snags in California (Laudenslayer and Darr 1990, McIntyre et al. 2015) and, in conjunction with a regime of fire suppression, has resulted in increased densities of small-diameter trees. Densification of forests and accumulations of fine fuels, along with a warming climate, has escalated the likelihood of large-scale, stand-replacing fires in California (Collins et al. 2015, Jones et al. 2016); this includes higher-elevation

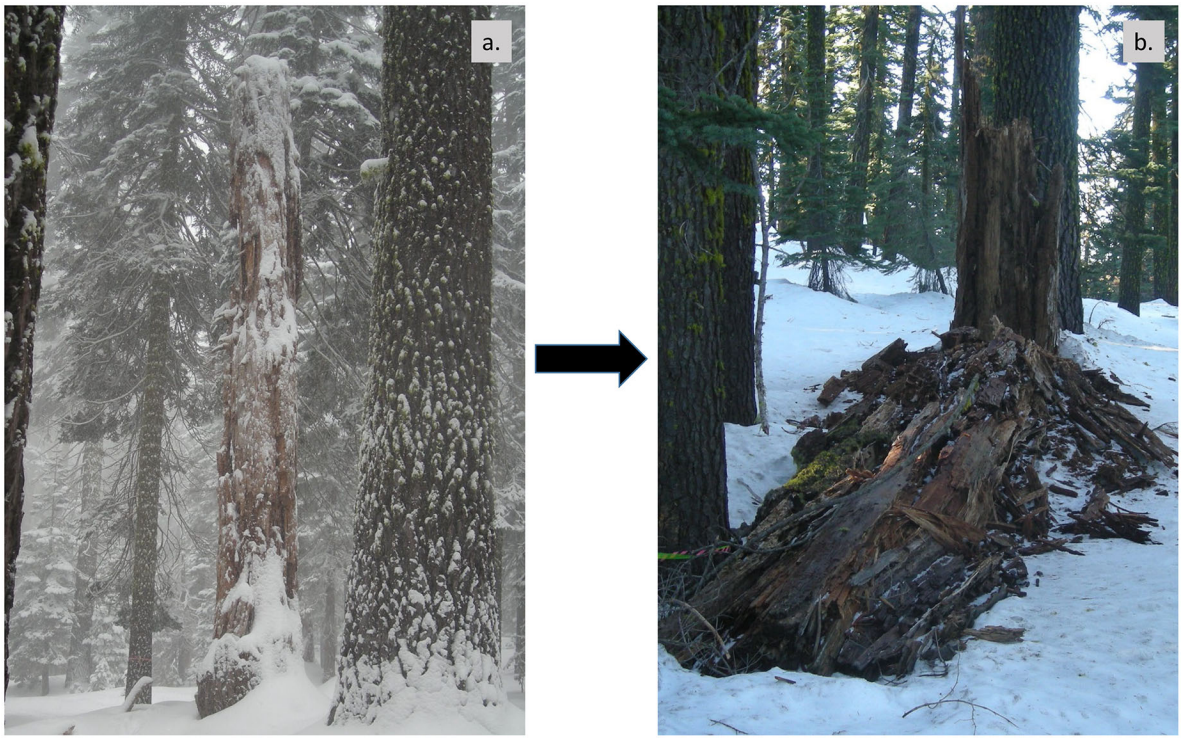

Fig. 2. (a) Condition of a heavily decayed snag upon re-location in 2016. (b) The same snag, which fell shortly after re-location. We deemed the fallen structure to be no longer usable by a marten.

areas inhabited by martens that have historically been characterized by longer fire-return intervals (50–200 yr; Van de Water and Safford 2011, Fryer and Luensmann 2012). Restorative forest management practices, such as mechanical thinning and prescribed fire, are increasingly used to mitigate wildfire risk by reducing fuel loads (Stephens et al. 2012, North et al. 2015). Over longer periods of time, fuel reduction treatments such as thinning could reduce competition for dominant and co-dominant trees (Stephens et al. 2012), thereby supporting growth and recruitment of the large-diameter structures that martens require. However, in the short-term, fuel reduction treatments often remove downed woody debris and understory cover, resulting in forest stands with simplified structure that martens may avoid (Moriarty et al. 2016).

Providing for habitat requirements of cavity-dependent wildlife, such as martens, fishers, and spotted owls, while addressing concerns of forest resiliency and fire risk, presents a complex challenge with an unclear path to resolution. For example, recent research has suggested that fishers are tolerant to low levels of disturbance (i.e., where treatments affected <3% of high-use fisher areas) from restorative treatments; yet, the authors also indicated that it was unclear whether treating such small areas would effectively reduce high-severity fire risk (Zielinski et al. 2013). The impacts of fuel reduction treatments on spotted owl populations are similarly uncertain, with impacts potentially differing by type, size, spatial configuration, and timing of treatments (Stephens et al. 2014). However, both studies advocated the importance of retaining large, decadent trees during treatments, as rare but ecologically important forest structural elements.

Given the increasing need for and application of restorative treatments, we encourage managers to consider the importance of fine-scale forest composition to martens and other cavity-dependent wildlife when developing broad-scale strategies to reduce fire risk and improve forest resilience. We suggest that additional research may be necessary to better describe the landscape processes that maintain a suitable availability of large trees with cavities, inform

whether those processes will be supported or hindered by forest management practices, and determine the appropriate timescales that should be accounted for. Further, we encourage future investigations into the effects of structure availability on martens—for example, whether or not limited structure availability influences individual marten fitness or, conversely, whether or not martens exhibit some degree of plasticity in rest structure selection.

Acknowledgments

Funding was provided by the USDA Forest Service, including the Lassen National Forest and Pacific Northwest Research Station, as well as Oregon State University and Oak Ridge Institute for Science and Education. Logistical support was provided by staff at the Lassen National Forest, Almanor Ranger District. Field data collection was assisted by numerous hard-working technicians, including L. Kreiensieck, K. Mansfield, M. Martin, A. Roddy, and P. Tweedy. This publication represents the views of the authors, and any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. government.

Literature Cited

Bissonette, J. A., D. J. Harrison, C. D. Hargis, and T. G. Chapin. 1997. The influence of spatial scale and scale-sensitive properties on habitat selection by American marten. Pages 365–385 *in* J. A. Bissonette, editor. Wildlife and landscape ecology: effects of pattern and scale. Springer-Verlag, New York, New York, USA.

Collins, B. M., J. M. Lydersen, R. G. Everett, D. L. Fry, and S. L. Stephens. 2015. Novel characterization of landscape-level variability in historical vegetation structure. Ecological Applications 25:1167–1174.

Easterday, K., P. McIntyre, and M. Kelly. 2018. Land ownership and 20th century changes to forest structure in California. Forest Ecology and Management 422:137–146.

Edworthy, A. B., and K. Martin. 2014. Long-term dynamics of the characteristics of tree cavities used for nesting by vertebrates. Forest Ecology and Management 334:122–128.

Edworthy, A. B., K. L. Wiebe, and K. Martin. 2012. Survival analysis of a critical resource for cavity-nesting communities: patterns of tree cavity longevity. Ecological Applications 22:1733–1742.

Franklin, J. F., H. H. Shugart, and M. E. Harmon. 1987. Tree death as an ecological process. BioScience 37:550–556.

Franklin, J. F., et al. 2002. Disturbances and structural development of natural forest ecosystems with silvicultural implications, using Douglas-fir forests as an example. Forest Ecology and Management 155:399–423.

Fryer, J. L., and P. S. Luensmann. 2012. Fire regimes of the conterminous United States. Fire Effects Information System (FEIS). USDA Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory, Missoula, Montana, USA.

Green, R. E., K. L. Purcell, C. M. Thompson, D. A. Kelt, and H. U. Wittmer. 2018. Reproductive parameters of the fisher (*Pekania pennanti*) in the southern Sierra Nevada, California. Journal of Mammalogy 99:537–553.

Hargis, C. D., J. A. Bissonette, and D. L. Turner. 1999. The influence of forest fragmentation and landscape pattern on American martens. Journal of Applied Ecology 36:157–172.

Hennon, P. E., and R. L. Mulvey. 2014. Managing heart rot in live trees for wildlife habitat in young-growth forests of coastal Alaska. GTR-PNW-890. USDA Forest Service, Pacific Northwest Research Station, Anchorage Forestry Sciences Laboratory, Anchorage, Alaska, USA.

Jones, G. M., R. J. Gutiérrez, D. J. Tempel, S. A. Whitmore, W. J. Berigan, and M. Z. Peery. 2016. Megafires: an emerging threat to old-forest species. Frontiers in Ecology and the Environment 14:300–306.

Jones, G. M., J. J. Keane, R. J. Gutierrez, and M. Z. Peery. 2018. Declining old-forest species as a legacy of large trees lost. Diversity and Distributions 24:341–351.

Kuussaari, M., et al. 2009. Extinction debt: a challenge for biodiversity conservation. Trends in Ecology and Evolution 24:564–571.

Laudenslayer, W. F., and H. H. Darr. 1990. Historical effects of logging on forests of the Cascade and Sierra Nevada Ranges of California. Transactions of the Western Section of the Wildlife Society 26:12–23.

Lindenmayer, D. B., W. F. Laurance, and J. F. Franklin. 2012*a*. Global decline in large old trees. Science 338:1305–1306.

Lindenmayer, D. B., W. Blanchard, L. McBurney, D. Blair, S. Banks, G. E. Likens, J. F. Franklin, W. F. Laurance, J. A. Stein, and P. Gibbons. 2012*b*. Interacting factors driving a major loss of large trees with cavities in a forest ecosystem. PLOS ONE 7:e41864.

Lindenmayer, D. B., and W. F. Laurance. 2017. The ecology, distribution, conservation and management of large old trees. Biological Reviews 92:1434–1458.

Lindenmayer, D. B., and J. T. Wood. 2010. Long-term patterns in the decay, collapse, and abundance of trees with hollows in the mountain ash (*Eucalyptus*

regnans) forests of Victoria, southeastern Australia. Canadian Journal of Forest Research 40:48–54.

Manion, P. D. 1981. Tree disease concepts. Prentice Hall, Upper Saddle River, New Jersey, USA.

McIntyre, P. J., J. H. Thorne, C. R. Dolanc, A. L. Flint, L. E. Flint, M. Kelly, and D. D. Ackerly. 2015. Twentieth-century shifts in forest structure in California: denser forests, smaller trees, and increased dominance of oaks. Proceedings of the National Academy of Sciences USA 112:1458–1463.

Moriarty, K. M., C. W. Epps, and W. J. Zielinski. 2016. Forest thinning changes movement patterns and habitat use by Pacific marten. Journal of Wildlife Management 80:621–633.

Moriarty, K. M., W. J. Zielinski, and E. D. Forsman. 2011. Decline in American marten occupancy rates at Sagehen Experimental Forest, California. Journal of Wildlife Management 75:1774–1787.

North, M. P., S. L. Stephens, B. M. Collins, J. K. Agee, G. Aplet, J. F. Franklin, and P. Z. Fulé. 2015. Reform forest fire management. Science 349:1280–1281.

Potvin, F., L. Bélanger, and K. Lowell. 2000. Marten habitat selection in a clearcut boreal landscape. Conservation Biology 14:844–857.

Purcell, K. L., C. M. Thompson, and W. J. Zielinski. 2012. Fishers and American martens. Pages 47–60 *in* M. North, editor. Managing Sierra Nevada Forests. GTR-PSW-237. USDA Forest Service, Pacific Southwest Research Station, Albany, California, USA.

Raphael, M. G., and L. L. Jones. 1997. Characteristics of resting and denning sites of American martens in central Oregon and western Washington. Pages 146–165 *in* G. Proulx, H. N. Bryant, and P. M. Woodard, editors. Martes: taxonomy, ecology, techniques, and management. The Provincial Museum of Alberta, Edmonton, Alberta, Canada.

Ruggiero, L. F., E. Pearson, and S. E. Henry. 1998. Characteristics of American marten den sites in Wyoming. Journal of Wildlife Management 62: 663–673.

Slauson, K. M., and W. J. Zielinski. 2009. Characteristics of summer and fall diurnal resting habitat used by American martens in coastal northwestern California. Northwest Science 83:35–45.

Stephens, S. L., J. D. McIver, R. E. Boerner, C. J. Fettig, J. B. Fontaine, B. R. Hartsough, P. L. Kennedy, and D. W. Schwilk. 2012. The effects of forest fuel-reduction treatments in the United States. BioScience 62:549–560.

Stephens, S. L., et al. 2014. California spotted owl, songbird, and small mammal responses to landscape fuel treatments. BioScience 64:893–906.

Taylor, S. L., and S. W. Buskirk. 1994. Forest microenvironments and resting energetics of the American marten *Martes americana*. Ecography 17:249–256.

Van de Water, K. M., and H. D. Safford. 2011. A summary of fire frequency estimates for California vegetation before Euro-American settlement. Fire Ecology 7:26–58.

Weir, R. D., M. Phinney, and E. C. Lofroth. 2012. Big, sick, and rotting: Why tree size, damage, and decay are important to fisher reproductive habitat. Forest Ecology and Management 265:230–240.

Zielinski, W. J., and R. V. Schlexer. 2015. Changes in the structural and functional characteristics of fisher (*Pekania pennanti*) rest structures over time. Northwest Science 89:148–153.

Zielinski, W. J., C. M. Thompson, K. L. Purcell, and J. D. Garner. 2013. An assessment of fisher (*Pekania pennanti*) tolerance to forest management intensity on the landscape. Forest Ecology and Management 310:821–826.

Zielinski, W. J., R. L. Truex, F. V. Schlexer, L. A. Campbell, and C. Carroll. 2005. Historical and contemporary distributions of carnivores in forests of the Sierra Nevada, California, USA. Journal of Biogeography 32:1385–1407.

Zielinski, W. J., R. L. Truex, G. A. Schmidt, F. V. Schlexer, K. N. Schmidt, and R. H. Barrett. 2004. Resting habitat selection by fishers in California. Journal of Wildlife Management 68:475–492.