Forest Ecology and Management 369 (2016) 66–73

Contents lists available at ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

Retention tree characteristics have major influence on the post-harvest tree mortality and availability of coarse woody debris in clear-cut areas

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ARTICLE INFO

Article history:
Received 13 January 2016
Received in revised form 14 March 2016
Accepted 14 March 2016
Available online 21 March 2016

Keywords:
Retention forestry
Prescribed burning
Tree dynamics
Dead wood
Biodiversity
Survival analysis

ABSTRACT

Retention forestry is widely used for reducing the negative biodiversity impacts of forest management. Its effectiveness depends, however, on the post-harvest dynamics of the retention trees, which in turn are likely affected by both tree- and stand-level factors. Understanding the effects of these factors on tree dynamics is thus essential for evaluating the biodiversity effects of retention forestry. We studied the impact of tree-level factors on the mortality and falling patterns of retention trees in pine-dominated boreal forests of eastern Finland. In total, 2738 retention trees were followed individually for 10 years after harvest on 12 study sites with an initial retention level of either 10 or 50 m^3 ha^{-1}. Prescribed burning was applied to half of the sites after harvest. Effects of tree species (pine, spruce or deciduous), diameter and location (grouped or scattered trees) on tree mortality and tree fall after death were assessed using survival analysis. During the 10-year survey period, 55% of the trees died and 81% of the dead trees fell down. The risk of mortality decreased with increasing diameter in all tree species on burned sites. On unburned sites, this applied only to pines and deciduous trees whereas for spruce the largest trees had the highest mortality. On all sites, dispersed trees had higher mortality than grouped trees. Pines had a lower risk to fall down after dying than spruces or deciduous trees. We conclude that tree species, diameter and location can be used to predict the post-harvest mortality and falling of retention trees and to select trees that best meet the aims of retention. However, considering the biodiversity effects, the predicted tree dynamics should not be the only basis for tree selection, but the habitat value of the retained trees must be accounted for as well.

1. Introduction

In the past decades, the negative biodiversity effects of forest management, especially clear-cutting, in boreal forests have received increasing attention. Consequently, various management methods aiming to decrease the loss of biodiversity have been developed. One widely applied method is retention forestry, in which some of the trees or dead wood objects are retained on the harvested clear-cut sites, aiming to create structural legacies similar to those occurring after natural disturbances (Gustafsson et al., 2012). The retained trees can benefit forest-dwelling species by supporting their populations on the cut site through the young successional phases (Baker et al., 2015), and later function as dispersal sources to the developing stand (Hedenås and Hedström, 2007), while standing dead trees and downed dead wood increase the structural variability of the harvested stand and provide habitats for species associated with dead wood (Gustafsson et al., 2010). In addition, to further mimic the effects of natural disturbances, tree retention can be combined with prescribed burning after harvest, to create specific fire-related habitats such as charred wood which host several specialist species. This is particularly important in regions such as Fennoscandia, where naturally occurring forest fires are currently rare (Zackrisson, 1977; Granström, 2001) and, consequently, species requiring fire-related habitats have become threatened (Rassi et al., 2010).

While both living and dead retention trees are substrates or resources for several forest-dwelling species, the species assemblages occurring on them are different (Martikainen, 2001; Hämäläinen et al., 2014). The assemblage composition differs also between standing and fallen dead trees and trees of different decay stages (Runnel et al., 2013; Seibold et al., 2015). Thus, in practice the impacts of retention forestry depend on the post-harvest mortality of the retained trees and the following dead wood dynamics,

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http://dx.doi.org/10.1016/j.foreco.2016.03.037
0378-1127/

as these determine which kinds of habitats the trees will provide for the forest-dwelling species. Knowledge on the dynamics of the retention trees after harvest is therefore necessary for evaluating the effects of retention forestry. Generally, tree mortality increases after harvest in comparison to uncut stands (e.g. Hautala and Vanha-Majamaa, 2006; Bladon et al., 2008; Lavoie et al., 2012). Mortality tends to be higher on small retention patches (Jönsson et al., 2007; Steventon, 2011) or when the overall retention level is low (Scott and Mitchell, 2005; Busby et al., 2006); also prescribed burning after harvest increases mortality at the stand level (Heikkala et al., 2014).

However, post-harvest tree mortality is affected also by tree-level factors such as tree species or size. Understanding these factors could help in selecting the trees to be retained in such a way that the desired targets are reached; for example, by retaining trees with the highest survival probability in order to maintain a continuity of living trees. Both the survival probability (Rosenvald et al., 2008; Lavoie et al., 2012) and mode of death, i.e. whether the trees die standing or fall due to windthrow, vary between tree species (Bladon et al., 2008). Tree diameter and height can also influence the survival, though the direction and extent of this impact depend on tree species (Rosenvald et al., 2008; Lavoie et al., 2012). In addition, the shape of the tree can be significant, with the risk of windthrow generally increasing for trees with larger height-diameter ratio (Scott and Mitchell, 2005). Finally, the location of the tree is often important: for example, trees close to the edge of the surrounding forest or close to older openings such as roads have lower mortality (Rosenvald et al., 2008), and trees retained in groups tend to survive better than dispersed trees (Scott and Mitchell, 2005).

The effects of tree-level factors on retention tree mortality have been assessed in some studies, but these have been mostly conducted in North America where the tree species composition is different from Europe. Application of the results from one geographical region to another can thus be difficult given the differences in species composition, climatic conditions and the applied harvesting methods (Lavoie et al., 2012). Furthermore, to date the mortality of individual retention trees has not been studied on sites where prescribed burning has been applied after harvest. On a stand scale, prescribed burning increases the mortality notably (Heikkala et al., 2014), but the significance of tree-level factors for the mortality is not known. In mature closed-canopy, unharvested forests, tree species (Hely et al., 2003), diameter (Linder et al., 1998; Sidoroff et al., 2007) and height (Hely et al., 2003; Kobziar et al., 2006) have all been found to affect mortality after fire; similar impacts could be expected on burned, harvested sites as well.

In this study, we assess the mortality of retention trees in Scots pine -dominated boreal forest stands in eastern Finland. Individually marked trees were followed for 10 years after harvest both in unburned sites and in sites where prescribed burning had been applied after harvesting. The stand-level mortality, with the impacts of retention tree volume and prescribed burning, during this 10-year period was assessed in a previous study (Heikkala et al., 2014), whereas in the current study we evaluate the dynamics of individual trees. More specifically, we examine which tree-level factors influence the survival of retained trees and the mode of death, i.e. whether the trees fall down or remain standing after dying. Moreover, we assess whether the impacts of the tree-level factors differ between the burned and unburned sites.

2. Materials and methods

2.1. Experimental design and field inventory

The field inventories were carried out in eastern Finland, in the municipality of Lieksa. The area is situated on the border of southern and middle boreal zones (Ahti et al., 1968). In this area a total of 12 sites were studied. The sites were 3–5 ha in size, of *Empetrum-Vaccinium* site type and situated on a mixture of sandy and moraine soils. Before treatments, all sites were approximately 150 years old forest stands dominated by Scots pine (*Pinus sylvestris*). The sites had not been clear-cut prior to the treatments, but some selective cuttings where scattered individual trees were removed have taken place during early 20th century. In addition to Scots pine, Norway spruce (*Picea abies*) and birches (*Betula pendula* and *B. pubescens*) were common on the sites, and aspen (*Populus tremula*), grey alder (*Alnus incana*), goat willow (*Salix caprea*) and rowan (*Sorbus aucuparia*) occurred in smaller numbers. Before treatments (see below), the treatment categories did not differ in the volume of living trees or dead wood (Hyvärinen et al., 2005). The total pre-treatment volume of living trees was on average 288 $m^3\ ha^{-1}$ (S.D. = 67.8) and of dead wood 40 $m^3\ ha^{-1}$ (S.D. = 16.9).

Experimental treatments were allocated randomly to the 12 study sites (for a map of the sites, see Heikkala et al., 2014). Treatments were combinations of two factors: green-tree retention level and prescribed burning. The sites were logged leaving either 10 or 50 $m^3\ ha^{-1}$ of living retention trees (averaging 3.5% and 17.4% of the pre-treatment tree volumes, respectively); both of these retention levels were replicated on six sites. After harvest, prescribed burning was applied to three of the sites on both retention levels, resulting in four different treatment combinations with three replicates of each. The sites were harvested in winter 2000–2001. Prescribed burning was applied to the sites during two days in June 2001 (see Hyvärinen et al., 2005 for a more detailed description of the burning).

The trees were mainly retained in small groups (approximately 200–500 m^2 in size), but also individual, scattered trees were left on each site. Prior to the treatments in 2000 all retained trees were alive. The trees were numbered individually, and tree species, diameter ($dbh_{1.3m}$), height (with a tachymeter or hypsometer) and location (mapped GPS coordinates) of each tree were measured. The trees were re-inventoried in summer 2001, one month after the treatments, and again in years 2005, 2008 and 2011. At each of these inventories, the condition of each tree was assessed: the trees were marked either as living, standing dead or fallen.

2.2. Statistical analyses

The impact of the tree characteristics on their survival was modelled using Cox proportional hazards model. The model is defined as

$$h(t|X) = h(t)\exp(X\beta)$$

where $h(t)$ is an underlying baseline hazard function representing the risk of death (or other event of interest) at time t (Harrell, 2001). The covariates X are assumed to have a multiplicative effect on this baseline hazard. The baseline hazard function itself is not estimated and the data are therefore not required to follow any specific distribution (Harrell, 2001).

The models were fit with the package *survival* of the software R (Therneau, 2015). Two separate models were run: the first one modelling the survival of the trees, i.e. the time from the harvest until the death of the tree, and the second modelling the fall of the dead trees, i.e. the time from tree death until tree fall. As the trees were not followed annually, the tree deaths and falls could not be exactly positioned to a specific year, but instead at four time intervals that lapsed between the temporal measurements on the sites: 2000–2001, 2001–2005, 2005–2008 and 2008–2011. The middle points of these intervals were used as event times, i.e. as the times of tree death or fall, in the models. Both models were first run with a same set of potential covariates: tree species (pine, spruce or deciduous), diameter, height-diameter ratio, location