

# Individual-level analysis of damage to residual trees after single-tree selection harvesting in northern Japanese mixedwood stands

Shinichi Tatsumi · Toshiaki Owari ·  
Hisatomi Kasahara · Yuji Nakagawa

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**Abstract** Quantifying tree damage and mortality caused by single-tree selection harvesting is critical to understanding postharvest forest dynamics and management. In this study, we quantified the effects of tree size and species and the distance from residual trees to felled trees and skid trails on damage to residual trees and mortality in mixed coniferous–broadleaved stands of Hokkaido, northern Japan. Among the 4,961 trees that we studied, 373 (7.5 %) were damaged, and 148 of these trees (3.0 %) died during or immediately after logging. Hierarchical Bayesian modeling showed that the risk of damage to residual trees increased with increasing size of the felled trees and with increasing proximity to felled trees and skid trails. Smaller residual trees had the greatest risk of damage. Species differed in their susceptibility to damage; *Abies sachalinensis* (Fr. Schm.) Masters and *Picea jezoensis* (Sieb. et Zucc.) Carr. were the most susceptible species in our sample plots. Smaller damaged trees had the highest risk of mortality. The damaged trees that did not die at the time of logging had a higher risk of postharvest mortality than undamaged trees. Our results indicate that, to minimize logging-induced damage and mortality: (1) the spatial arrangement of skid trails should be optimized and fixed, (2) the risk of skidding damage should be considered before marking the trees to be harvested and choosing the felling direction, and (3) logging operators should be

instructed to avoid damaging small trees, especially those of the species that are most susceptible to damage.

**Keywords** Hierarchical Bayesian analysis · Interspecific variation · Logging impacts · Selection harvesting · Tree mortality

## Introduction

In recent decades, single-tree selection harvesting has been implemented as a major alternative to clearcutting in forest regions around the world (Caspersen 2006; Yoshida et al. 2006; O'Hara et al. 2007). In the mixed coniferous–broadleaved forests of Hokkaido, spanning parts of the cool-temperate and subboreal zones in northern Japan, single-tree selection has been widely conducted since the early twentieth century (Yoshida et al. 2006). Single-tree selection is a silvicultural system in which individual trees are removed throughout a harvesting site at regular time intervals (Nyland 2002). Single-tree selection is considered to be a more benign method of harvesting because it promotes regeneration of the site by the surviving residual trees (Zingg 1999) and helps to stabilize the stand structure (Caspersen 2006). However, in Hokkaido, decreases in tree density (Nigi and Koshika 1997; The University of Tokyo Hokkaido Forest 2012) and changes in species composition (Yoshida et al. 2006; Tatsumi et al. 2010) have been observed in many forests, even when the harvesting volume was kept below the annual growth rate.

It is widely acknowledged that such changes in stand structure result, at least in part, from the damage to residual trees (including tree mortality) caused by logging (Fukushi et al. 1997; Sasaki et al. 2005, 2006, 2007, 2008; Kuramoto et al. 2010; Takahashi et al. 2011).

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S. Tatsumi (✉) · T. Owari · H. Kasahara · Y. Nakagawa  
The University of Tokyo Hokkaido Forest, The University  
of Tokyo, 9-61 Yamabe-higashimachi, Furano,  
Hokkaido 079-1563, Japan  
e-mail: jeyms23@gmail.com

Numerous studies have quantified the levels of residual tree damage in selection-harvested stands caused by the felled trees and by extraction equipment such as skidders (e.g., Lamson et al. 1985; Granhus and Fjeld 2001; Caspersen 2006; Surakka et al. 2011; Modig et al. 2012). Among these studies, ones based on individual-tree analysis found clear evidence that the damage level depended on the tree size and the distance from residual trees to felled trees and skid trails (Granhus and Fjeld 2001; Surakka et al. 2011). However, such analysis has only been conducted in single-species stands of Norway spruce (*Picea abies* Karst.) in northern Europe (Granhus and Fjeld 2001; Surakka et al. 2011). An individual-level analysis in the mixed coniferous–broadleaved forests of Hokkaido would improve our knowledge about the risk of logging damage caused by selection harvesting and its influence on stand structure.

Because the previous individual-level analyses of logging damage (Granhus and Fjeld 2001; Surakka et al. 2011) were limited to single-species stands, the interspecific variation in susceptibility to damage has not been considered. This is an important omission, because understanding interspecific variation is critical for quantifying disturbance-induced tree mortality in mixed-species forests (Canham et al. 2001). For example, studies of wind damage have found clear variations among species in their vulnerability to wind, and these differences can greatly affect postdisturbance forest dynamics (Canham et al. 2001; Peterson 2007; Yoshida and Noguchi 2009). We hypothesize that such interspecific variation also exists in the context of the mortality caused by single-tree selection harvesting, which is an important form of anthropogenic disturbance. If this hypothesis is true, then the species that are most vulnerable to logging damage should be given priority when planning to prevent the damage.

The objectives of this study are to quantify the effects of individual-level factors (tree size and species, and distance from residual trees to felled trees and skid trails) on damage to residual trees and mortality caused by single-tree selection harvesting in the mixed coniferous–broadleaved stands of Hokkaido. To explicitly examine the influence of the individual-level factors, we used a hierarchical Bayesian approach. Such an approach provides a flexible modeling framework in which interspecific variations can be efficiently estimated (Zipkin et al. 2009; Comita et al. 2010; Tatsumi et al. 2013); this framework is essential for addressing our study objectives. Our results provide new insights into the harvesting-induced changes in stand structure and represent a critical step toward developing management plans that account for the tree damage and mortality caused by logging.

## Materials and methods

### Study site

We conducted this study in the first working section of the University of Tokyo Hokkaido Forest (43°10–20'N, 142°18–36'E, 190 to 850 m elevation, 12,234 ha) in central Hokkaido, northern Japan. The mean annual temperature and annual precipitation at the study site from 2001 to 2008 were 6.3 °C and 1,210 mm, respectively (The University of Tokyo Hokkaido Forest 2012). Mean monthly temperatures in 2012 ranged from a minimum of –8.4 °C in January to a maximum of 21.2 °C in August (The University of Tokyo Hokkaido Forest 2013). The forest floor is covered by snow from November to April, reaching a depth of ca. 1 m (Owari et al. 2011). The typical substrate and soil type are welded tuff underlying dark-brown forest soil. The forest mainly consists of mixed coniferous–broadleaved stands with evergreen conifers and deciduous broadleaved species. The dominant canopy species are *Abies sachalinensis* (Fr. Schm.) Masters, *Picea jezoensis* (Sieb. et Zucc.) Carr., *Tilia japonica* (Miq.) Simonkai, and *Acer mono* Maxim. (Owari et al. 2011).

We collected data from 13 permanent sample plots (rectangular plots ranging in size from 0.24 to 0.53 ha, with a mean of 0.38 ha) located throughout the study area. Single-tree selection has been conducted in these study plots since 1968, with a harvesting cycle of 5 to 26 years. Each plot has undergone from two to four harvests. The most recent logging operations (conducted from 2002 to 2009) were done by means of motor-manual felling with chainsaws and skidding of the felled stems by a winch-equipped crawler tractor (12-t Komatsu Ltd. D40, with 2.8-m blade width). The operations were conducted using a tree-length method, in which the trees were felled, delimited at the stump, dragged by winch to the skid trails, and skidded to the landing areas. This is currently the most widely used method in Hokkaido (Sasaki et al. 2005, 2006, 2007, 2008; Kuramoto et al. 2010). In the most recent operations, skidding was done mainly by reusing the skid trails that had been established in previous entries, with several new trails being made as well. The operations were conducted in winter, with snow cover on the forest floor. The mean tree density, mean diameter at breast height (DBH), and stand basal area (BA) for all trees with DBH  $\geq 5$  cm in the 13 plots before the most recent harvesting were 1,034.6 trees ha<sup>-1</sup> ( $\pm 301.4$  SD), 17.8 cm ( $\pm 2.7$  SD), and 38.3 m<sup>2</sup> ha<sup>-1</sup> ( $\pm 7.6$  SD), respectively. The mean rate of removal (the harvesting intensity) was 13.8 % of stand BA ( $\pm 2.5$  SD) during each logging entry. The mean slope gradient was 20.5° ( $\pm 5.9$  SD).

Data collection

In the study plots, the DBH of all trees with  $DBH \geq 5$  cm has been measured by the staff of the university forest since 1968 at 5-year intervals, with 0.1-cm precision. These trees, including those that had been newly recruited at each measurement date, were numbered with metal tags. Separately from the DBH measurements, the damage to residual trees caused by logging was surveyed in the spring after each logging entry by the university forest staff. The damaged trees were classified into two categories: “damaged and died” and “damaged only.” The former category included trees that lost their live leaves due to stem breakage or uprooting (i.e., knocked to the ground). The latter category included trees that had live leaves, even if their stems were broken, bent, or debarked; trees with broken branches; trees that had been knocked down but that remained alive; and trees with combinations of these forms of damage. For trees in the latter category, the types and extent of the damage were not recorded by the university forest staff. For the felled trees, the felling direction was also not recorded.

We mapped the locations ( $x$ - $y$  coordinates) of the trees with 0.1-m precision from 2008 to 2012. The positions of the trees that had died or had been harvested before this period were identified using the numbered tags for the remaining trees and the positions of the stumps of the harvested trees. Most of the trees (>97 %) that were alive after 2000 could be mapped, whereas some of the trees that had died and had been harvested before 2000 were lost. We therefore included only the data collected after 2000 in the analysis. In total, 261 trees were harvested from 2000 to 2012, and we observed 4,961 residual trees from 38 species (Table S1).

In August and September 2012, we mapped the locations of the skid trails in and around the sample plots. This mapping was done by surveying the position of the center of the skid trails with 0.1-m precision using a TruPulse 200 laser rangefinder with a Mapstar Electronic Compass

Module II attachment (Laser Technology Inc.) and measuring the trail width around that center with 0.5-m precision using a surveying pole. We mapped not only the skid trails inside the plots but also those located within 30 m of the plot edges to account for the influence of skidding in adjacent areas. Although 4 to 10 years had passed since the most recent logging entries, we were able to clearly determine the locations of the skid trails from the remaining disturbance of the forest floor, such as exposure of the subsoil (B horizon) or the absence of understory vegetation.

Statistical modeling

We used hierarchical Bayesian models to analyze the effects of tree size and species and of felling and skidding proximity on the mortality of residual trees. We used three models that expressed different processes within the occurrence of logging-induced damage and mortality: (1) the probability of an individual tree being damaged by logging, (2) the probability of the damaged tree dying at the time of logging or immediately after, and (3) the probability of damaged and undamaged trees dying within 10 years after logging (Fig. 1). By using these three models, we could separately analyze (1) the factors that affect the occurrence of damage, (2) the factors that affect the probability of death at or immediately after logging, and (3) the effect of harvesting damage on postharvest tree mortality. The damage status was simply expressed as a binary value (undamaged = 0, damaged = 1) because the types and extent of the damage were not recorded by the university forest staff. In the third model, we could only examine the mortality up to 10 years after logging because there have been only two 5-year measurement intervals since the oldest harvesting entries.

In the first model, we examined the effects of tree size (of both residual and felled trees) and of felling and skidding proximity on the damage risk using a logit-link function. The damage risk was defined as the probability that a residual tree would be damaged. We assumed that

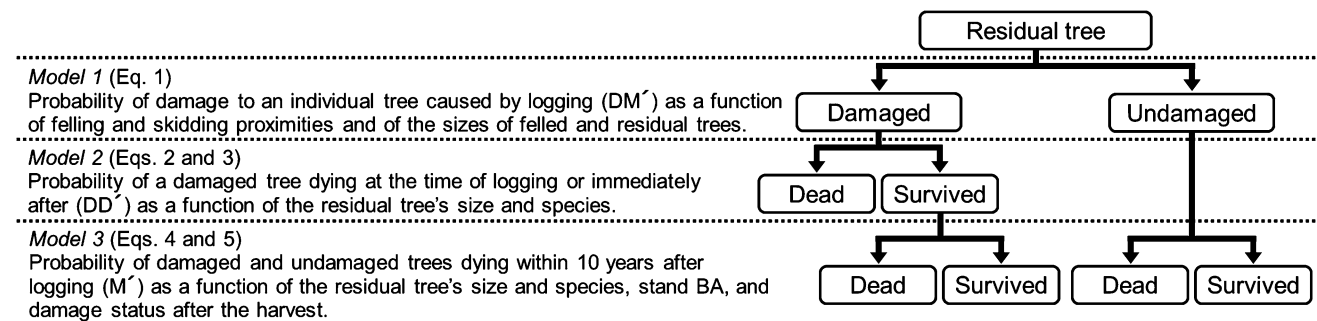


Fig. 1 Schematic representation of the three models used to express different processes within the occurrence of logging-induced damage and mortality of individual trees

the observed status of residual tree  $i$  in plot  $j$  (where  $DM_{ij} = 0$  and  $DM_{ij} = 1$  corresponded to undamaged and damaged, respectively) followed a Bernoulli distribution:

$$DM_{ij} \sim \text{Bernoulli}(DM'_{ij}),$$

$$\text{logit}(DM'_{ij}) = \alpha_0 + \alpha_1 DB_{ij} + \alpha_2 DS_{ij} + \alpha_3 FL_{ij} + \alpha_4 SK_{ij} + \psi_{1j}, \quad (1)$$

where  $DM'_{ij}$  is the risk of damage to residual tree  $i$  in plot  $j$ ,  $DB_{ij}$  is the DBH (cm) of residual tree  $i$  in plot  $j$ ,  $DS_{ij}$  is the distance (m) from residual tree  $i$  in plot  $j$  to the nearest felled tree,  $FL_{ij}$  is the DBH (cm) of the felled tree nearest to the residual tree  $i$  in plot  $j$ ,  $SK_{ij}$  is the distance (m) from residual tree  $i$  in plot  $j$  to the nearest skid trail,  $\alpha_0$  to  $\alpha_5$  are regression parameters, and  $\psi_{1j}$  is a random parameter for plot  $j$ . Here, we used the explanatory variables  $DB_{ij}$ ,  $DS_{ij}$ ,  $FL_{ij}$ , and  $SK_{ij}$  because similar variables were found to strongly influence the risk of damage in previous individual-level analyses (Granhus and Fjeld 2001; Surakka et al. 2011). We included the random effect  $\psi_{1j}$  to account for stand-level variability in factors such as tree density and slope gradient. The priors of  $\alpha_0$  to  $\alpha_5$  were defined as noninformative normal distributions,  $\mathcal{N}(0, 10,000)$ . We defined the prior of the random parameter  $\psi_1$  as  $\mathcal{N}(0, \sigma_1^2)$  and that of the hyperparameter  $\sigma_1$  as a noninformative uniform distribution,  $\text{Unif}(0, 10,000)$ .

In the second model, we examined the effects of residual tree size and species on the probability of damaged residual trees (i.e.,  $DM_{ij} = 1$ ) dying at the time of logging or immediately after. We assumed that the observed status of residual tree  $i$  in plot  $j$  (where  $DD_{ij} = 0$  and  $DD_{ij} = 1$  correspond to “damaged only” and “damaged and dead,” respectively) followed a Bernoulli distribution:

$$DD_{ij} \sim \text{Bernoulli}(DD'_{ij}),$$

$$\text{logit}(DD'_{ij}) = \beta_{0,SP_{ij}} + \beta_{1,SP_{ij}} DB_{ij} + \psi_{2j}, \quad (2)$$

where  $DD'_{ij}$  is the probability of damaged tree  $i$  in plot  $j$  being dead,  $SP_{ij}$  is an identification number for the species of damaged tree  $i$  in plot  $j$ ,  $\beta_0$  and  $\beta_1$  are regression parameters, and  $\psi_{2j}$  is a random parameter for plot  $j$ . Here, we used the variable  $DB_{ij}$  (the residual tree's DBH) because previous research showed that it strongly influenced disturbance-induced mortality (Canham et al. 2001; Peterson 2007; Yoshida and Noguchi 2009). The priors of  $\psi_2$  and  $\sigma_2$  were defined as  $\mathcal{N}(0, \sigma_2^2)$  and  $\text{Unif}(0, 10,000)$ , respectively. The regression parameters  $\beta_0$  and  $\beta_1$  were defined to have hierarchical structures:

$$\beta_{x,SP} \sim \mathcal{N}(\beta'_x, \tau_{1x}^2) \quad (x = 0, 1; SP = 1, 2, \dots, s), \quad (3)$$

where  $\tau_{10}$  and  $\tau_{11}$  are hyperparameters, and  $s$  is the number of species analyzed (defined later in this section in detail).

Here,  $\beta_{0,SP}$  and  $\beta_{1,SP}$  were defined to take different values for different species ( $SP = 1, 2, \dots, s$ ) but to share common mean values  $\beta'_0$  and  $\beta'_1$ , respectively. Such an approach for expressing the parameters (regression coefficients) as random parameters is widely used in ecological data analysis (Bolker et al. 2009), especially when the analysis deals with interspecific variations (Suzuki 2011; Tatsumi et al. 2013). The priors of  $\tau_{10}$  and  $\tau_{11}$  were defined as  $\text{Unif}(0, 10,000)$ .

In the third model, we examined the effects of the residual tree's size and species, stand BA, and the tree's damage status on the mortality of residual trees within 10 years after logging. We assumed that the observed status of residual tree  $i$  in plot  $j$  (where  $M_{ij} = 0$  and  $M_{ij} = 1$  correspond to live and dead, respectively) followed a Bernoulli distribution:

$$M_{ij} \sim \text{Bernoulli}(M'_{ij}),$$

$$\text{logit}(M'_{ij}) = \gamma_{0,SP_{ij}} + \gamma_{1,SP_{ij}} DB_{ij} + \gamma_{2,SP_{ij}} DB_{ij}^2 + \gamma_{3,SP_{ij}} BA_{ij} + \gamma_{4,SP_{ij}} DM_{ij} + \psi_{3j}, \quad (4)$$

where  $M'_{ij}$  is the mortality of residual tree  $i$  in plot  $j$  within 10 years after logging,  $BA_{ij}$  is the stand BA ( $\text{m}^2 \text{ha}^{-1}$ ) in plot  $j$  without including the BA of tree  $i$ ,  $\gamma_0$  to  $\gamma_4$  are regression parameters, and  $\psi_{3j}$  is a random parameter for plot  $j$ . Here, we included the variable  $DB_{ij}^2$  to express a U-shaped response of mortality to  $DB_{ij}$  (Fortin et al. 2008); that is, both the smallest and the largest trees had the highest mortality, with lower mortality rates for intermediate trees. The variable  $BA_{ij}$  was included as a surrogate for tree competition (Fortin et al. 2008). The variable  $DM_{ij}$  was included because we hypothesized that the damaged trees would have higher rates of postharvest mortality. The priors of  $\psi_3$  and  $\sigma_3$  were defined as  $\mathcal{N}(0, \sigma_3^2)$  and  $\text{Unif}(0, 10,000)$ , respectively. The regression parameters  $\gamma_0$  to  $\gamma_4$  were defined to have hierarchical structures:

$$\gamma_{y,SP} \sim \mathcal{N}(\gamma'_y, \tau_{2y}^2) \quad (y = 0, 1, \dots, 4; SP = 1, 2, \dots, s), \quad (5)$$

where  $\tau_{20}$  to  $\tau_{24}$  are hyperparameters. Here,  $\gamma_{0,SP}$  to  $\gamma_{4,SP}$  were defined to take different values for different species but to share common mean values  $\gamma'_0$  to  $\gamma'_4$ , respectively. The priors of  $\tau_{20}$  to  $\tau_{24}$  were defined as  $\text{Unif}(0, 10,000)$ . The definitions and summary statistics of the explanatory variables are listed in Table 1.

#### Parameter estimation and model selection

We fit the three models to the data by using the Markov chain Monte Carlo (MCMC) method implemented by the WinBUGS 1.4.3 software (Lunn et al. 2000) via the R2WinBUGS package (Sturtz et al. 2005) for the R 2.15.0

**Table 1** Definitions and summary statistics of the explanatory variables used in analysis

Variable	Definition	Mean	Median	SD	Min.	Max.
DB	DBH (cm) of the residual tree	16.2	11.1	12.7	5.0	84.0
DS	Distance (m) from a residual tree to the nearest felled tree	6.9	6.2	3.9	0.0	31.2
FL	DBH (cm) of the felled tree nearest to the residual tree	35.4	34.2	12.0	13.3	80.1
SK	Distance (m) from a residual tree to the nearest skid trail	15.4	12.6	12.0	0.0	65.4
BA	Stand basal area (m <sup>2</sup> ha <sup>-1</sup> ) without including the basal area of the focal residual tree	32.5	33.5	5.8	19.6	40.7
DM	Damage status (undamaged = 0, damaged = 1) of a residual tree	0.04	0	0.20	0	1

software (R Development Core Team 2012). For each model, we obtained posterior samples using three independent MCMC samplings, in each of which 20,000 values were sampled with a 5-step interval after a burn-in period of 1,000 MCMC steps. We confirmed the convergence of the MCMC calculations by using the Gelman and Rubin  $\hat{R}$  parameter (Gelman et al. 2004).

The process of model selection was divided into two steps. First, for all three models, we conducted backward stepwise selections to eliminate the explanatory variables that did not contribute to improving the goodness of fit of the model. In this first step, the interspecific variation in susceptibility to damage was not considered; that is, SP was held constant irrespective of the tree species in Eqs. 3 and 5. We used the deviance information criterion (DIC; Spiegelhalter et al. 2002) for model selection; the model with the lowest value of DIC was retained. The priors of the parameters  $\alpha_0$  to  $\alpha_5$ ,  $\beta_0$  to  $\beta_1$ , and  $\gamma_0$  to  $\gamma_4$  were defined as noninformative normal distributions,  $\mathcal{N}(0, 10,000)$ , in this first step.

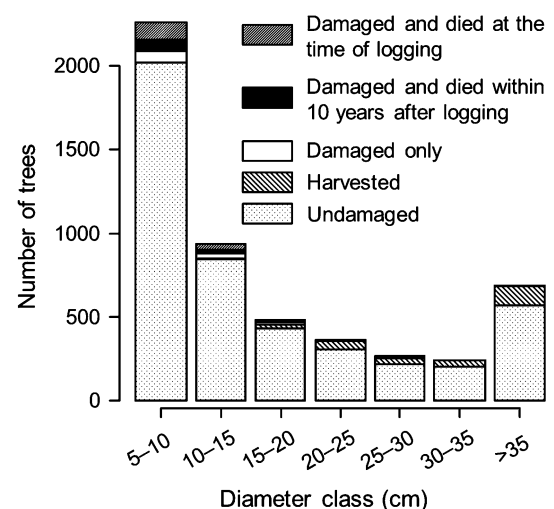
In the second step, we tested the hypothesis that species differed in their susceptibility to damage. We conducted this test by modifying the parameters of the best models selected by the backward stepwise procedure for the second model (Eq. 2) and the third model (Eq. 4) to have hierarchical structures (Eqs. 3 and 5). The significance of adopting the hierarchical structure for each parameter was tested by means of forward stepwise selection based on DIC. Among the 38 species observed in our sample plots, any species that met one or more of the following five conditions was integrated in the category “other species”: (1) all of the individuals were undamaged (DM = 0), (2) all of the damaged trees died at or immediately after the time of logging (DD = 1), (3) all of the damaged trees survived after logging (DD = 0), (4) all of the individuals died within 10 years after logging (M = 1), and (5) all of the individuals survived at least 10 years after logging (M = 0). These species were integrated into a single category because their parameter values ( $\beta_{0,SP}$  to  $\beta_{1,SP}$  and  $\gamma_{0,SP}$  to  $\gamma_{4,SP}$ ) diverged to  $\infty$  or  $-\infty$  during the estimation process. After integrating these species, we were able to examine the interspecific variation among 21 species

(including the category “other species”); that is,  $s = 21$  (Table S1).

**Results**

The density of skid trails within the 13 plots ranged from 0 to 396.8 m ha<sup>-1</sup>, with a mean of 190.7 m ha<sup>-1</sup>. The density was 0 m ha<sup>-1</sup> in one plot in which all the harvested trees were dragged by winch to skid trails located outside the plot. Among the 4,961 trees we observed, 373 trees (7.5 %) were damaged, and 148 of these trees (3.0 %) died at or immediately after the time of logging (Fig. 2). Among the 225 damaged trees that survived at the time of logging, 99 trees (44.0 %) died within 10 years. In contrast, among the 4,588 undamaged trees, 321 trees (7.0 %) died within 10 years.

All of the associated parameters of the models (Tables S2 and S3) converged adequately ( $\hat{R} < 1.1$ ; Gelman et al. 2004). For the first model (Eq. 1), the full model had the lowest DIC and was selected (Table 2). According to this model and the associated parameter estimates, residual



**Fig. 2** Frequency distributions for the DBH of the harvested and residual trees



**Table 2** Results of model selection for the first model (Eq. 1)

Backward stepwise selection from the full model	DIC
Full <sup>a</sup>	<b>2,398.3</b>
Full – $\alpha_4$ DB	2,521.3
Full – $\alpha_1$ DS	2,423.5
Full – $\alpha_2$ FL	2,398.4
Full – $\alpha_3$ SK	2,411.7

The models selected by backward stepwise selection with the best fit (i.e., with the lowest DIC) are shown in bold

See Table 1 for definitions of the explanatory variables

See Table S2 for the parameter estimates for the selected model

<sup>a</sup> The selected model:  $\alpha_0 + \alpha_1 DB_{ij} + \alpha_2 DS_{ij} + \alpha_3 FL_{ij} + \alpha_4 SK_{ij} + \psi_{ij}$

trees with the smallest DBH were at the highest risk of damage (Fig. 3a; note that in this figure, the mean values of the posterior samples for each parameter were used for prediction, and variables other than DB were fixed at the medians of their observed values). The predicted risk of damage increased with increasing proximity to a felled tree and increased with increasing DBH of the felled tree (Fig. 3b). Using a felled tree with 50-cm DBH as an example, we can see that the risk of damage to residual trees at 0 and 31.2 m from a felled tree (the latter representing the maximum observed value; Table 1) was 12.8 and 1.0 %, respectively. The predicted risk of damage to a residual tree also increased with increasing proximity to the skid trail (Fig. 3c). The predicted risk of damage to trees at 0 and 65.4 m from a skid trail was 9.8 and 2.3 %, respectively.

For the second model (Eqs. 2 and 3), the full model with a hierarchical parameter structure had the lowest DIC and was selected (Table 3). The probability of a damaged tree dying at or immediately after logging decreased with

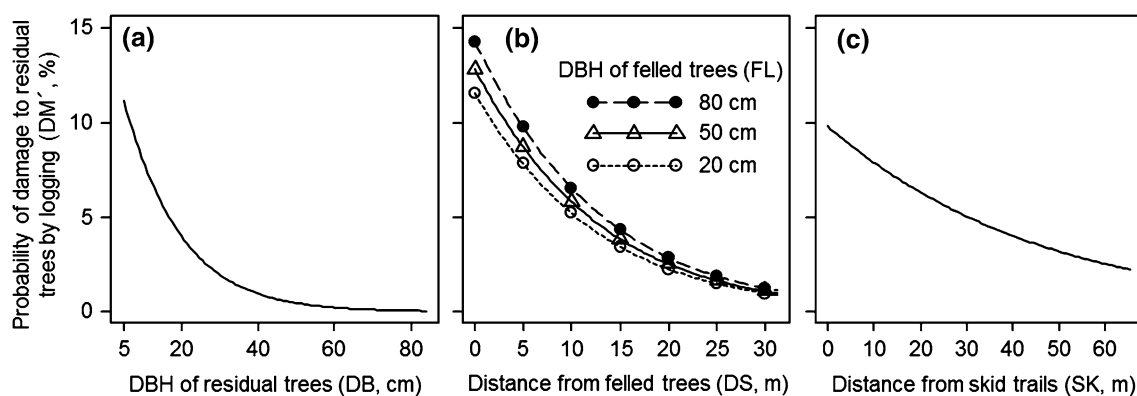
increasing DBH of the residual tree (Figs. 4 and S1; note that for each species in these figures, the lines are drawn across the range of observed DBH values. The maximum DBH values are listed in Table S1). This figure also reveals interspecific variation in the susceptibility to damage; *A. sachalinensis* and *P. jezoensis* were the most susceptible species, whereas *Taxus cuspidata* Sieb. et Zucc. and *A. mono* were the least susceptible species. Taking a tree with 20-cm DBH as an example, damaged trees of *A. sachalinensis* and *T. cuspidata* had a 36.0 and 3.5 % risk of dying, respectively.

The mortality after logging, which we examined using the third model (Eqs. 4 and 5), was significantly affected by residual tree DBH, stand BA, and postharvesting damage status (Table 4). The residual tree mortality increased with increasing stand BA (Fig. 5), and damaged trees had a higher risk of postharvest mortality than the undamaged trees in the same DBH class.

## Discussion

Effects of tree DBH, felling, and skidding on damage risk

The DBH of the felled and residual trees and the proximity of the residual tree to felled trees and skid trails significantly affected the risk of damage to the residual tree (Table 2; Fig. 3). Residual trees with smaller DBH clearly had higher risk of damage (Figs. 2, 3a). This result is consistent with previous reports (Lamson et al. 1985; Sasaki et al. 2005, 2006, 2007). Felling and skidding workers typically try to avoid damaging large residual trees, but pay less attention to protecting small trees (Sasaki et al. 2005; Surakka et al. 2011; Takahashi et al. 2011). At our study



**Fig. 3** Predicted probability of damage to residual trees caused by logging as a function of **a** residual tree DBH, **b** distance from and DBH of the nearest felled tree, and **c** distance from the nearest skid trail. The selected model (Table 2) and the mean values of the

posterior samples for the parameters (Table S2) were used for prediction. In each graph, the values of all variables except the variable on the x-axis were fixed at the medians of their observed values (Table 1)

**Table 3** Results of model selection for the second model (Eqs. 2 and 3)

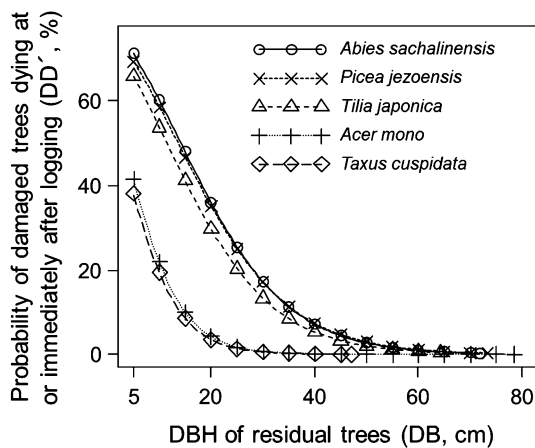
Backward stepwise selection from the full model	DIC
Full	<b>430.9</b>
Full – $\beta_1$ DB	446.5
Forward stepwise selection for the model selected by backward stepwise selection	DIC
<b>+<math>\beta_{1,SP}DB^a</math></b>	<b>416.7</b>

The models selected by backward and forward stepwise selection with the best fit (i.e., with the lowest DIC) are shown in bold

See Table 1 for definition of the explanatory variable

See Tables S2 and S3 for the parameter estimates for the selected model

<sup>a</sup> The selected model:  $\beta_{0,SP_j} + \beta_{1,SP_j}DB_{ij} + \psi_{2j}$



**Fig. 4** Predicted probability of damaged trees dying at or immediately after the time of logging as a function of residual tree DBH and species. For each species, the lines are drawn across the range of observed DBH values (Table S1). The selected model (Table 3) and the mean values of the posterior samples for the parameters (Tables S2 and S3) were used for prediction. Curves for all species are presented in Fig. S1

site, harvesting operations are currently conducted by contractors who must pay fines if they damage trees with DBH larger than ca. 11 cm. Such contract provisions may have further enhanced the likelihood of smaller trees being damaged. Our results suggest that the workers’ choices of felling direction, the skid trail to which each felled tree is dragged, and the locations of skid trails can substantially affect the diameter-class distribution after single-tree selection harvesting.

The risk of damage decreased with increasing distance between residual trees and the felled trees (Fig. 3b) and the skid trails (Fig. 3c). The slope of the decrease was gentler for the distance from skid trails than for the distance from felled trees. This difference likely arose because the

**Table 4** Results of model selection for the third model (Eqs. 4 and 5)

Backward stepwise selection from the full model	DIC
Full	<b>2,577.8</b>
Full – $\gamma_1$ DB	2,605.1
Full – $\gamma_2$ DB <sup>2</sup>	2,585.2
Full – $\gamma_3$ BA	2,579.1
Full – $\gamma_4$ DM	2,800.1
Forward stepwise selection for the model selected by backward stepwise selection	DIC
<b>+<math>\gamma_{1,SP}DB</math></b>	<b>2,517.1</b>
<b>+<math>\gamma_{2,SP}DB^2</math></b>	<b>2,514.1</b>
<b>+<math>\gamma_{3,SP}BA</math></b>	<b>2,521.1</b>
<b>+<math>\gamma_{4,SP}DM</math></b>	<b>2,524.6</b>
<b>+<math>\gamma_{1,SP}DB + \gamma_{2,SP}DB^2</math></b>	<b>2,514.9</b>
<b>+<math>\gamma_{2,SP}DB^2 + \gamma_{3,SP}BA^a</math></b>	<b>2,510.8</b>
<b>+<math>\gamma_{2,SP}DB^2 + \gamma_{4,SP}DM</math></b>	<b>2,515.8</b>
<b>+<math>\gamma_{1,SP}DB + \gamma_{2,SP}DB^2 + \gamma_{3,SP}BA</math></b>	<b>2,511.1</b>
<b>+<math>\gamma_{2,SP}DB^2 + \gamma_{3,SP}BA + \gamma_{4,SP}DM</math></b>	<b>2,513.1</b>

The models selected by backward and forward stepwise selection with the best fit (i.e., with the lowest DIC) are shown in bold

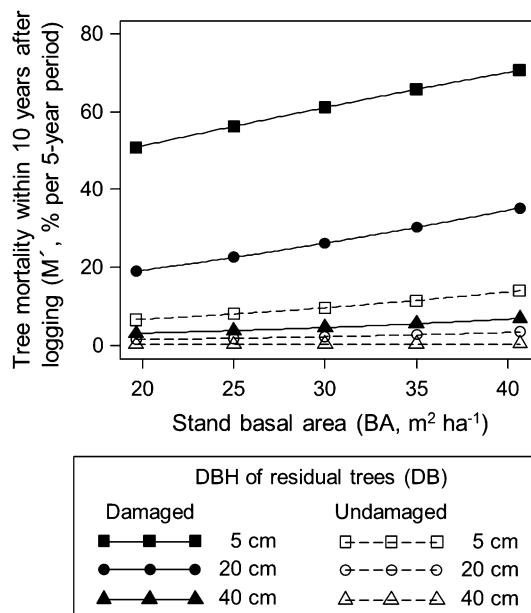
See Table 1 for definitions of the explanatory variables

See Tables S2 and S3 for the parameter estimates for the selected model

<sup>a</sup> The selected model:  $\gamma_{0,SP_j} + \gamma_1DB_{ij} + \gamma_{2,SP_j}DB_{ij}^2 + \gamma_{3,SP_j}BA_{ij} + \gamma_4DM_{ij} + \psi_{3j}$

damage caused by felling mainly occurs only within a circle whose radius equals the height of the felled tree, whereas the damage caused by skidding can occur anywhere along the path joining a skid trail with a felled tree, as the felled trees were dragged by winch to the skid trail at our study site.

Although our model successfully quantified the spatial extent of the damage caused by felling and skidding, there is clearly potential for improvement. Because of the limitations in our data, we could not analyze the effects of the height of the individual harvested trees or of the felling direction on the spatial distribution of damaged trees. Also, although we used SK (the distance from a residual tree to the nearest skid trail) as a proxy for the intensity of the skidding disturbance (Granhus and Fjeld 2001; Surakka et al. 2011), the felled trees could not always be dragged to the nearest trail because of complications created by the felling direction and the local stand structure. To account for such cases, we should determine the skid trail to which each felled tree was dragged. A model that explicitly incorporates such factors would likely provide a more detailed understanding of the spatial distribution of the damage intensity.



**Fig. 5** Predicted tree mortality within 10 years after logging as a function of stand basal area, and residual tree DBH and damage status (damaged or undamaged). The selected model (Table 4) and the mean values of the posterior samples for the parameters (Tables S2 and S3) were used for prediction

#### Interspecific variation in susceptibility to damage

The model selection results supported our hypothesis that there is interspecific variation in the susceptibility to logging damage (Table 3; Figs. 4 and S1). This variation could have mainly resulted from differences in the wood strength of the species, which is strongly related to the mechanical breakage rate of standing trees after disturbance (Peterson 2007; Yoshida and Noguchi 2009). Yoshida and Noguchi (2009) analyzed the interspecific variation in susceptibility to wind damage in a mixed coniferous–broadleaved forest in northern Hokkaido, where wind disturbance is a major type of natural disturbance (Ishikawa and Ito 1989). They found that *A. sachalinensis* and *P. jezoensis*, the two coniferous species in their study area, were the two most susceptible species among the 12 species they studied. They also found a significant negative relationship between the risk of damage to a species and its wood strength, using “absorbed energy in impact bending ( $\text{kg m}^2$ )” as a proxy for strength. They found that species with weak wood, such as the two coniferous species, have a higher risk of dying because their stems break more easily than those of species with stronger wood. In our study, the two dominant coniferous species (*A. sachalinensis* and *P. jezoensis*) were also estimated to be the two species most susceptible to logging damage (Figs. 4 and S1). Furthermore, *T. cuspidata*, a species with very high wood density (Miyabe and Kudo

1986), was the least susceptible species in our study. (Unfortunately, Yoshida and Noguchi did not study this species, so we cannot compare this finding with their results.) For the broadleaved species, *T. japonica*, which has relatively low wood density (Miyabe and Kudo 1986), was the most susceptible species in our study. In contrast, *A. mono*, which has relatively high wood strength (high Young’s modulus; Yazawa et al. 1965), was the least susceptible in our study. These results suggest that the wood strength of a species could be an important factor that determines the species-specific susceptibility to logging damage, as is the case for wind damage.

On the other hand, logging damage had the opposite impact on the size distribution of the dead trees compared with wind damage. Whereas large trees are generally more susceptible to wind damage because of their greater cross-sectional area (Canham et al. 2001; Peterson 2007; Yoshida and Noguchi 2009), our results showed that small trees were more susceptible to logging damage (Fig. 4). A previous study of changes in stand structure at our study site (Tatsumi et al. 2010) reported a long-term decrease in the density of small trees since 1964, especially for *A. sachalinensis* and *P. jezoensis*. We hypothesize that the high mortality of small *A. sachalinensis* and *P. jezoensis* caused by the anthropogenic disturbance of logging (Figs. 4 and S1), which was rarely observed in naturally disturbed forests (Yoshida and Noguchi 2009), could be an important driver of the recent changes in stand structure observed in our study area.

#### Postharvest mortality

The damaged trees had remarkably higher risks of postharvest mortality than the undamaged trees in the same diameter class (Fig. 5). This suggests that focusing only on the immediate tree mortality caused by harvesting, as was done in most previous studies (e.g., Lamson et al. 1985; Sasaki et al. 2005, 2006, 2007, 2008; Modig et al. 2012; but see Kuramoto et al. 2010 for an exception), could underestimate the effects of logging damage on postharvest forest dynamics. On the other hand, postharvest mortality is affected both by direct physical damage and by increased physiological and mechanical stresses on trees driven by the felling and skidding activities (Caspersen 2006; Yoshida and Noguchi 2010). Although the influences of such stresses are beyond the scope of our study, incorporating them into the current model could further improve its accuracy and applicability.

#### Proportion of damaged trees

Among the 4,961 trees we observed, 7.5 % (373 trees) were damaged (Fig. 2). This value is smaller than those



reported in the series of studies by Sasaki et al. (2005, 2006, 2007, 2008), who reported that 13.0 to 27.6 % of the trees were damaged in four stands located across Hokkaido, where harvesting operations were conducted for the first time over the previous few decades. The value was smaller at our study sites likely because most of the skid trails had been previously established and were reused in the recent logging entries, whereas those studied by Sasaki et al. were newly established at each entry by destroying all the trees on the trails. Other than this difference, similar operating procedures were used at both sets of study sites. Stand-level factors, such as tree density, slope gradient, harvesting intensity, and skid trail density, were also roughly the same; the values were 526.0–1,024.0 trees ha<sup>-1</sup>, 0–20°, 12.5–21.4 %, and ca. 260 m ha<sup>-1</sup>, respectively, at the study sites of Sasaki et al. Although the proportion of damaged trees might be influenced by other factors as well, such as operator skill level (Sirén 1999), our results imply that the proportion of damaged trees in the second and subsequent harvesting entries could be reduced by fixing and reusing the skid trails.

#### Management implications

Our results indicated that the recent changes in the stand structure in our study region could be partly attributed to (1) a lack of consideration of the impacts of logging damage on small trees when harvesting contractors choose the felling direction, select the skid trail to which each felled tree is dragged, and plan the spatial arrangement of the skid trails, and to (2) a bias in logging-induced mortality toward small trees and species with higher susceptibility. These findings lead to three major management recommendations: (1) the spatial arrangement of the skid trails should be optimized and fixed to minimize damage to the residual trees, (2) the potential for skidding damage should be considered before beginning a harvesting operation and should be minimized when marking trees for harvesting and choosing the felling direction, and (3) operators should be instructed to avoid damaging any trees, not just the largest ones, as much as possible, especially for the species that are most susceptible to damage (e.g., *A. sachalinensis* and *P. jezoensis*).

Our results show that the damage to residual trees and their subsequent mortality depended on tree DBH and species, as well as on the proximity of the residual tree to felling and skidding activities. To examine the effects of such individual-level factors on the overall forest dynamics and future yield, the use of spatially explicit individual-based forest dynamics models would be an effective approach (Thorpe et al. 2010; Tatsumi et al. 2012). Future

research could incorporate the results of the present study into such models to help foresters develop management plans that explicitly account for logging-induced tree mortality.

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#### References

- Bolker BM, Brooks ME, Clark CJ, Geange SW, Poulsen JR, Stevens HH, White JS (2009) Generalized linear mixed models: a practical guide for ecology and evolution. *Trends Ecol Evol* 24:127–135
- Canham CD, Papaik MJ, Latty EF (2001) Interspecific variation in susceptibility to windthrow as a function of tree size and storm severity for northern temperate tree species. *Can J For Res* 31:1–10
- Caspersen JP (2006) Elevated mortality of residual trees following single-tree felling in northern hardwood forests. *Can J For Res* 36:1255–1265
- Comita LS, Muller-Landau HC, Aguilar S, Hubbell SP (2010) Asymmetric density dependence shapes species abundances in a tropical tree community. *Science* 329:330–332
- Fortin M, Bédard S, DeBlois J, Meunier S (2008) Predicting individual tree mortality in northern hardwood stands under uneven-aged management in southern Québec, Canada. *Ann For Sci* 65:art205
- Fukushi K, Iwamoto S, Kasahara H, Sakaguchi T, Ihara S, Nitami T (1997) Development of timber harvesting system with a grapple skidder for selective cutting operation in natural forest management: forest disturbance and operational efficiency through skidding operation. *Trans Meet Hokkaido Branch Jap For Soc* 45:146–148 (in Japanese)
- Gelman A, Carlin JB, Stern HS, Rubin DB (2004) Bayesian data analysis, 2nd edn. CRC Press, London
- Granhuis A, Fjeld D (2001) Spatial distribution of injuries to Norway spruce advance growth after selection harvesting. *Can J For Res* 31:1903–1913
- Ishikawa Y, Ito K (1989) The regeneration process in a mixed forest in central Hokkaido, Japan. *Vegetatio* 79:75–84
- Kuramoto S, Sasaki S, Abe S, Ishibashi S (2010) Post-harvest damage and subsequent survival following selection harvesting of small understory trees in a mixed conifer–hardwood forest in Hokkaido Island, northern Japan. In: Proceedings of the 43rd FORMEC Symposium. Padova, pp 1–4
- Lamson NI, Smith HC, Miller GW (1985) Logging damage using an individual-tree selection practice in Appalachian hardwood stands. *North J Appl For* 2:117–120
- Lunn DJ, Thomas A, Best N, Spiegelhalter D (2000) WinBUGS—a Bayesian modelling framework: concepts, structure, and extensibility. *Stat Comput* 10:325–337
- Miyabe K, Kudo Y (1986) Icones of the essential forest trees of Hokkaido. Hokkaido University Press, Hokkaido (in Japanese)
- Modig E, Magnusson B, Valinger E, Cedergren J, Lundqvist L (2012) Damage to residual stand caused by mechanized selection harvest in uneven-aged *Picea abies* dominated stands. *Silva Fenn* 46:267–274

- Nigi T, Koshika K (1997) Management of natural mixed forests in northern Hokkaido using the control method: 30-year report. *J For Plann* 3:107–112
- Nyland RD (2002) *Silviculture: concepts and applications*, 2nd edn. Waveland, Illinois
- O'Hara KL, Hasenauer H, Kindermann G (2007) Sustainability in multi-aged stands: an analysis of long-term plenter systems. *Forestry* 80:163–181
- Owari T, Matsui M, Inukai H, Kaji M (2011) Stand structure and geographic conditions of natural selection forests in central Hokkaido, northern Japan. *J For Plann* 16:207–214
- Peterson CJ (2007) Consistent influence of tree diameter and species on damage in nine eastern North America tornado blowdowns. *For Ecol Manage* 250:96–108
- R Development Core Team (2012) *R: a language and environment for statistical computing*. R Foundation for Statistical Computing, Vienna
- Sasaki S, Ishibashi S, Takao G, Takahashi M, Abe S, Sakai Y, Yamaguchi T (2005) Damages to a natural forest caused by a selective cutting operation in Ikutora, Hokkaido. *J Jpn For Eng Soc* 19:301–304 (in Japanese)
- Sasaki S, Ishibashi S, Takao G, Takahashi M, Sasaki T (2006) Damages to a natural forest caused by a selective cutting operation in Soranuma, Hokkaido. *Trans Meet Hokkaido Branch Jap For Soc* 54:112–114 (in Japanese)
- Sasaki S, Ishibashi S, Takahashi M (2007) Damages to a natural forest caused by selective cutting operations in Hokkaido: a comparison of damage between Ikutora, Soranuma, and Kushiro natural forests. *Trans Meet Hokkaido Branch Jap For Soc* 55:95–97 (in Japanese)
- Sasaki S, Ishibashi S, Kuramoto S, Takahashi M (2008) Damages to a natural forest caused by selective cutting operations in Hokkaido: the damage in Hidaka natural forest. *Trans Meet Hokkaido Branch Jap For Soc* 56:129–131 (in Japanese)
- Sirén M (1999) One-grip harvester operations, silvicultural results and possibilities to predict tree damage. In: Keane MG, Kofman PD (eds) *Proceedings of a IUFRO conference on harvesting and economics of thinnings*. Ennis, pp 152–167
- Spiegelhalter DJ, Best NG, Carlin BP, van der Linde A (2002) Bayesian measures of model complexity and fit. *J R Stat Soc B* 64:583–616
- Sturtz S, Ligges U, Gelman A (2005) R2WinBUGS: a package for running WinBUGS from R. *J Stat Softw* 12:1–16
- Surakka H, Sirén M, Heikkinen J, Valkonen S (2011) Damage to saplings in mechanized selection cutting in uneven-aged Norway spruce stands. *Scand J For Res* 26:232–244
- Suzuki M (2011) Effects of the topographic niche differentiation on the coexistence of major and minor species in a species-rich temperate forest. *Ecol Res* 26:317–326
- Takahashi K, Fukushi K, Koike Y, Inukai S, Sanyoushi A, Owari T (2011) Residual tree damage, seedling loss and regeneration after selection logging in a natural forest: a case study within the compartment no. 51 at the Tokyo University Forest in Hokkaido. *Trans Meet Hokkaido Branch Jap For Soc* 59:87–90 (in Japanese)
- Tatsumi S, Owari T, Yamamoto H, Shiraishi N (2010) Forty-two years of stand structure development in a natural sub-boreal forest under selection system in central Hokkaido, Japan. *Int For Rev* 12:87
- Tatsumi S, Owari T, Toyama K, Shiraishi N (2012) Adaptation of a spatially-explicit individual-based forest dynamics model SORTIE-ND to conifer–broadleaved mixed stands in the University of Tokyo Hokkaido Forest. *Formath* 11:1–26
- Tatsumi S, Owari T, Ohkawa A, Nakagawa Y (2013) Bayesian modeling of neighborhood competition in uneven-aged mixed-species stands. *Formath* 12:191–209
- The University of Tokyo Hokkaido Forest (2012) The 13th education and research plan of the University of Tokyo Hokkaido Forest (2011–2020). *Misc Inform Tokyo Univ For* 51:67–176 (in Japanese)
- The University of Tokyo Hokkaido Forest (2013) Annual report of meteorological observations in the University forests, the University of Tokyo (Jan. 2011–Dec. 2011). *Misc Inform Tokyo Univ For* 53:195–220 (in Japanese)
- Thorpe HC, Vanderwel MC, Fuller MM, Thomas SC, Caspersen JP (2010) Modelling stand development after partial harvests: an empirically based, spatially explicit analysis for lowland black spruce. *Ecol Model* 221:256–267
- Yazawa K, Miyajima H, Bito K, Hayashi R, Ito K (1965) Studies on the laminated woods made from various hardwoods grown in Hokkaido (report 2): straight laminated wooden beams made from Ezo-itaya (*Acer* sp.), Buna (*Fagus* sp.), Harunire (*Ulmus* sp.) and coniferous woods. *Res Bull Coll Exp For Hokkaido Univ* 24:235–274 (in Japanese with English abstract)
- Yoshida T, Noguchi M (2009) Vulnerability to strong winds for major tree species in a northern Japanese mixed forest: analyses of historical data. *Ecol Res* 24:909–919
- Yoshida T, Noguchi M (2010) Growth and survival of *Abies sachalinensis* seedlings for three years after selection harvesting in northern Hokkaido, Japan. *Landsc Ecol Eng* 6:37–42
- Yoshida T, Noguchi M, Akibayashi Y, Noda M, Kadomatsu M, Sasa K (2006) Twenty years of community dynamics in a mixed conifer–broad-leaved forest under a selection system in northern Japan. *Can J For Res* 36:1363–1375
- Zingg A (1999) English and German terminologies in forestry research on growth and yield: a few examples. *For Snow Landsc Res* 74:179–187
- Zipkin EF, DeWan A, Royle JA (2009) Impacts of forest fragmentation on species richness: a hierarchical approach to community modeling. *J Appl Ecol* 46:815–822