Simulation models predict that habitat loss may reach a threshold level below which ecological processes change abruptly (Fahrig 1998; With and King 1999; Flather and Bevers 2002). Effects of fragmentation sensu stricto are then detected, whereby increasing distances among patches and matrix resistance reduce the likelihood of recolonization after local extinctions (Lande 1987; With and King 1999). By definition, models simplify reality; therefore, they do not always represent appropriate metaphors for real landscapes because they tend to assume sharply contrasted habitat mosaics. None the less, different species perceive the same landscape differently and those specializing on habitat types dramatically altered by management may exhibit actual fragmentation effects and species-specific thresholds.

Surprisingly few studies have examined the landscape-scale effects of forest management on animals and plants. Previous landscape-scale studies have mainly examined species response to different landscape contexts or landscape structures in forests fragmented by agriculture (see Chapter 8, this volume, for a review). Managed forest landscapes are thought to be more permeable to the movements of vertebrate forest animals because contrasts among forest stands are generally softer than between forests and cropfields or pastures, for example. Indeed, fragmentation effects have mainly been detected in forests fragmented by agriculture or in island archipelagoes

A second potential stumbling block to detection of landscape thresholds in forest mosaics is the coarseness of the predictor variables generally available (e.g. “forest”, “developed land”). It is important to accurately define the distribution of habitat for species under consideration to ensure that landscape metrics are relevant to the species at hand (Vos et al. 2001; Betts et al. 2007). On the other hand, it is not always possible to develop species-specific models for all species of concern. Ideally, conservation targets should be defined based on the requirements of the focal region’s most sensitive species (e.g. Guenette and Villard 2005).

A few studies have tested for threshold (non-linear) relationships in species occurrence as a function of landscape structure (e.g. Homan et al. 2004; Betts et al. 2007; Denoël and Ficetola 2007; Chapter 7, this volume). These studies tend to recommend conservation targets that are on or above threshold values. However, if landscape-scale thresholds are to be applied beyond the boundaries of a given study area, it is critical for conservation planners to know about the robustness of such conservation targets (Chapter 7, this volume). The generality of habitat studies to broader spatial extents and longer time periods is often assumed in the development of conservation policy, but rarely tested (Wallington et al. 2005).

In this chapter, we test for thresholds in the occurrence of forest songbirds in relation to landscape structure by using two different statistical techniques: receiver-operating characteristic (ROC) analysis and segmented logistic regression (Muggée 2003). We examine the strengths and weaknesses of both of these approaches to species distribution modeling and conservation planning. Using data from three geographically discrete study areas each located within a 250 km radius, we determine to what extent the thresholds obtained can be generalized (a) across space; (b) over time; (c) across spatial extents; and (d) across species. Each of these comparisons also provides an opportunity to compare the behavior of threshold detection methods. Species-specific thresholds are examined using the proportion of mature forest in the landscape as a common currency, and more complex landscape metrics derived from spatially explicit, species-specific distribution models (the “organism-based” or “species-centered” approach) (Betts et al. 2006b).

METHODS

We used data from three different study areas located in the province of New Brunswick, Canada. Two of these (Black Brook, 47°23’N, 67°40’W; Riley Brook, 47°11’N, 67°13’W) are located in north-western New Brunswick.
areas the third is located c. 250 km away, in south-eastern New Brunswick (Riley Brook, Fundy) or 100 m radius (Black Brook) (see énette and Villard 2005; Betts et al. 2006a). Differences in point count methodology among studies should result in higher overall probability of detection of the same species in Black Brook than in the other two study areas. However, all other things being equal, the shape of the response of individual species to landscape structure should remain the same. Nevertheless, we acknowledge that methodological differences undoubtedly decreased the likelihood of finding similarities in threshold values among study areas. Hence, our results should be considered conservative.

Protocols also varied slightly with respect to habitat sampling and characterization of landscape structure. Therefore, we conducted pairwise comparisons between these study areas depending on the compatibility of the habitat and landscape data sets.

Among the bird species surveyed, we selected five wood warbler species (family Parulidae) that are known to be associated with mature (>60 years) forest (Blackburnian warbler Dendroica fusca; black-throated blue warbler D. caerulescens; black-throated green warbler D. virens; northern parula Parula americana; and ovenbird Seiurus aurocapilla). In New Brunswick, nature forests are the stand age classes declining at the fastest rate (Betts et al. 2003, 2007a). The five species represent a variety of ecological niches: the ovenbird nests and forages on the ground; the black-throated blue warbler nests and forages in the shrub layer; the three other species nest and forage in the subcanopy and canopy (Poole 2005). In a separate study we developed spatially explicit models for the distribution of 21 bird species based on local scale variables derived from a geographic information system GIS) as predictor variables (e.g. age class, stand composition, canopy cover, elevation; Betts et al. 2006a). The GIS land-cover data originated from the New Brunswick Forest Inventory (NBDNR 1993), which are based on interpreted and digitized aerial photographs taken in 1993 (1:12 500 scale, color) and updated to 2000 with satellite imagery (30 m² resolution; Betts et al. 2003). None of these initial models relied on landscape-scale data. We developed habitat-suitability maps (30 m² resolution) for each species by mapping the fitted values of GIS habitat models.
Vegetation plot data were used to account for local habitat suitability (see Guenette and Villard 2005; Betts et al. 2007a). We modeled bird–habitat relationships by using logistic regression. Thresholds in either (a) mature forest or (b) amount of habitat (defined above) were detected either by using ROC analysis (Guenette and Villard 2005) or segmented logistic regression (Muggeo 2003; Betts et al. 2007b). These statistical threshold-detection methods are explained in Box 9.1. We used locally weighted regression splines (loess) plots to show the smoothed distribution of raw presences and absences in relation to amount of mature forest and habitat amount (e.g. Fig. 9.3a).

First, we compared thresholds detected for a given species in two different study areas. Depending on the need for detailed stand-level or landscape-level data, we compared the Fundy study area with either the Black Brook or the Riley Brook (north-western New Brunswick) study area. Then, we compared thresholds detected for a given species over three successive years. A third comparison pertained to variations in thresholds detected for the same species across spatial extents (150 m – 2 km surrounding each sample point). We also compared threshold responses (or lack thereof) among forest bird species at a given location and year. Finally, we examined whether landscape metrics defined from an "organism-based perspective" (i.e. spatially explicit habitat models developed for individual species) improve the consistency in thresholds (a) among species, (b) between study areas (Riley Brook and Fundy), and (c) between statistical techniques. This final test could only be conducted for three species because GIS models did not have adequate prediction success for the other species in both regions. In all cases of threshold detection, we controlled for the effects of both spatial autocorrelation and variation introduced by differences in local variation (Box 9.2; see Guenette and Villard 2005; Betts et al. 2007b).

RESULTS

We found statistical support for segmented thresholds in amount of mature forest in at least one study area for two of five species (black-throated blue warbler, Blackburnian warbler; Table 9.1). Thresholds were supported for all three species in at least one study area as a function of habitat amount (Table 9.2). However, in both cases segmented thresholds tended to have less support when we statistically controlled for local habitat variation (Tables 9.1 and 9.2; see Box 9.2). We estimated ROC thresholds for all species in all study areas. Both threshold methods showed varying...
For the Blackburnian warbler (Table 9.1; Fig. 9.3). For this species, the segmented threshold value we found was nearly identical between the Fundy and Black Brook study areas. However, this non-linear relationship did not receive consistent statistical support in both study areas. A threshold in Blackburnian warbler occurrence was strongly supported in Black Brook, but received only moderate support in Fundy (Δ AIC 2.63 from the linear model). The segmented model that controlled for local variation did not even converge in Fundy (Table 9.1). Similarly, thresholds for black-throated blue warbler were roughly similar (c. 11% difference); 95% confidence intervals (1.96 x s.e.) of the estimated thresholds in each study area overlapped. However, the strength of the Fundy model was equivocal (segmented model Δ AIC 0.04 from linear model). Ovenbird and black-throated green warbler showed greater variation in threshold values between study areas (c. 20%), irrespective of the threshold detection method used. However, for these species, thresholds in species occurrence as a function of mature forest did not receive much support (Table 9.1).

Our comparison of thresholds in species occurrence as a function of (species-specific) habitat amount did not show any greater consistency between study areas (Table 9.2; Fig. 9.4). Thresholds were strongly
Table 9.1. Thresholds (ROC and segmented: see Box 9.1) in probability of occurrence of five species of wood warbler as a function of the percent of mature forest within a 1 km radius landscape in two study areas of New Brunswick, Canada.

Note that negative Δ AIC values indicate low or no support for segmented threshold models. Rows in bold indicate statistical support for segmented thresholds in either models including only landscape terms or models controlling for local variation (full models).

<table>
<thead>
<tr>
<th>Location</th>
<th>AUC (AUC full model)</th>
<th>ROC threshold</th>
<th>Segmented threshold (SE)</th>
<th>Segmented threshold full model (SE)</th>
<th>Δ AIC to linear model (full model)</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Blackburnian</em></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fundy</td>
<td>0.60 (0.69)</td>
<td>83.26</td>
<td>89.67 (5.56)</td>
<td>92.96 (1.18)</td>
<td>-2.63</td>
</tr>
<tr>
<td>Black Brook</td>
<td>0.70 (0.86)</td>
<td>68.12</td>
<td>92.57 (2.32)</td>
<td>92.96 (1.18)</td>
<td>8.09 (3.13)</td>
</tr>
<tr>
<td><em>Ovenbird</em></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fundy</td>
<td>0.52 (0.75)</td>
<td>85.15</td>
<td>76.09 (15.46)</td>
<td>71.2 (15.95)</td>
<td>-2.45</td>
</tr>
<tr>
<td>Black Brook</td>
<td>0.61 (0.88)</td>
<td>64.29</td>
<td>51.30 (14.74)</td>
<td>68.67 (11.25)</td>
<td>1.50 (0.04)</td>
</tr>
<tr>
<td><em>Black-throated green</em></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fundy</td>
<td>0.54 (0.58)</td>
<td>86.21</td>
<td>88.33 (12.81)</td>
<td>92.96 (1.18)</td>
<td>-2.45</td>
</tr>
<tr>
<td>Black Brook</td>
<td>0.62 (0.89)</td>
<td>62.80</td>
<td>NA</td>
<td>95.27 (8.52)</td>
<td>-3.00 (0.24)</td>
</tr>
<tr>
<td><em>Black-throated blue</em></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fundy</td>
<td>0.51 (0.72)</td>
<td>85.01</td>
<td>69.91 (7.00)</td>
<td>70.45 (9.50)</td>
<td>1.50 (0.04)</td>
</tr>
<tr>
<td>Black Brook</td>
<td>0.50 (0.87)</td>
<td>67.15</td>
<td>45.01 (6.43)</td>
<td>81.67 (11.52)</td>
<td>3.02 (0.81)</td>
</tr>
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<td><em>Northern parula</em></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fundy</td>
<td>0.50 (0.70)</td>
<td>84.72</td>
<td>81.10 (10.65)</td>
<td>77.76 (22.44)</td>
<td>-1.83 (3.19)</td>
</tr>
<tr>
<td>Black Brook</td>
<td>0.60 (0.82)</td>
<td>64.03</td>
<td>76.64 (9.11)</td>
<td>73.34 (12.76)</td>
<td>0.80 (2.05)</td>
</tr>
</tbody>
</table>

* Full model includes autocovariates, local, and landscape variables.

b No model convergence.

c AIC values in brackets correspond to models that include landscape and local variables as well as spatial autocovariates.

Table 9.2. Thresholds (ROC and segmented: see Box 9.1) in probability of occurrence of forest birds as a function of the percent of habitat in a 1 km (Blackburnian warbler) or 2 km radius landscape (black-throated blue warbler, ovenbird) in two study areas of New Brunswick (Greater Fundy Ecosystem, Riley Brook).

Note that negative Δ AIC values indicate low or no support for segmented threshold models. Rows in bold indicate statistical support for segmented thresholds in either models including only landscape terms or models controlling for local variation (full models).

<table>
<thead>
<tr>
<th>Location</th>
<th>AUC (AUC full model)</th>
<th>ROC threshold</th>
<th>Segmented threshold (SE)</th>
<th>Segmented threshold full model (SE)</th>
<th>Δ AIC to linear model (full model)</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Blackburnian</em></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fundy</td>
<td>0.59 (0.66)</td>
<td>29.67</td>
<td>16.74 (1.18)</td>
<td>14.73 (2.03)</td>
<td>0.1 (1.34)</td>
</tr>
<tr>
<td>Riley Brook</td>
<td>0.61 (0.81)</td>
<td>25.90</td>
<td>33.76 (8.04)</td>
<td>15.74 (2.07)</td>
<td>-3.42 (2.05)</td>
</tr>
<tr>
<td><em>Ovenbird</em></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fundy</td>
<td>0.56 (0.74)</td>
<td>38.71</td>
<td>29.88 (14.48)</td>
<td>28.20 (5.08)</td>
<td>8.29 (3.19)</td>
</tr>
<tr>
<td>Riley Brook</td>
<td>0.72 (0.78)</td>
<td>20.11</td>
<td>NA</td>
<td>19.79 (6.87)</td>
<td>NA (2.34)</td>
</tr>
<tr>
<td><em>Black-throated blue</em></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fundy</td>
<td>0.61 (0.75)</td>
<td>15.60</td>
<td>8.83 (0.62)</td>
<td>8.63 (0.70)</td>
<td>13.77 (5.0)</td>
</tr>
<tr>
<td>Riley Brook</td>
<td>0.68 (0.72)</td>
<td>19.65</td>
<td>NA</td>
<td>NA</td>
<td>NA</td>
</tr>
</tbody>
</table>

* Full model includes autocovariates, local, and landscape variables.

b No model convergence.

c AIC values in brackets are from models that include landscape and local variables as well as spatial autocovariates.
Temporal consistency in thresholds

Using both segmented and ROC methods, there appeared to be some consistency in threshold values obtained for individual species in the Fundy study area over three successive years (Fig. 9.5). It should be noted that although surveys took place in the same study area, birds were sampled at different study plots in 2001 than in 2002 and 2003. The occurrence of one species, the black-throated blue warbler, exhibited segmented thresholds that were largely robust to changes in time and space, at least within a local study region. The ROC approach identified thresholds that were remarkably consistent over the three years of the study (Fig. 9.5). The main difference in threshold values observed in Fig. 9.5 pertains to the threshold detection method we used rather than to the study year (see Discussion).

Consistency in thresholds across spatial extents

Thresholds in habitat amount associated with particular species decreased as spatial extent increased in two of the three species considered (Fig. 9.6). For the black-throated blue warbler, threshold values were

Figure 9.4. Occurrence of three species in relation to the amount of habitat in a landscape in Fundy (F) and Riley Brook (RB) (Blackburnian warbler, 1 km radius; other species, 2 km radius). Dashed lines show ROC thresholds. Segmented thresholds are shown as solid black vertical lines (gray shading, 95% CI).

Figure 9.5. Occurrence of three species in relation to the amount of habitat within a 1 km radius (Blackburnian warbler) or a 2 km radius (black-throated blue warbler, ovenbird) over three years. Samples were collected in different spatial locations in 2001 than in 2002 and 2003. Dashed lines indicate ROC threshold. Black solid line indicates segmented threshold (gray shading, 95% CI). The presence of segmented thresholds in the figure does not indicate that a threshold model was the most parsimonious (see Tables 9.1 and 9.2).
Figure 9.6. Thresholds in habitat amount for three species of forest songbirds in relation to spatial extent considered. Empty symbols indicate no statistical support for a threshold at a particular extent. Thresholds were not calculated if habitat amount data at multiple spatial extents were too highly correlated to allow for comparison.

Variation in thresholds across species
As reported elsewhere (Chapter 7, this volume), there were species-specific variations in threshold values in response to both forest cover and habitat amount in the landscape. Different species appear to decline at different points along the forest or habitat loss gradient (With and Crist 1995). In Black Brook, the Blackburnian warbler exhibited the highest threshold in relation to mature forest (93%). This model was strongly supported. The threshold for the black-throated blue warbler was lower in the same study area (82%). Interestingly, thresholds in mature forest as detected by ROC were surprisingly consistent across species. In Fundy, thresholds ranged from 83% (Blackburnian warbler) to 86% (ovenbird). In Black Brook the range was from 62% (black-throated green warbler) to 68% (Blackburnian warbler). Thresholds in habitat amount varied from 9% (Fundy, black-throated blue warbler) to 28% (Fundy, ovenbird). It is important to note that this substantial range is partly an artifact of the way in which “habitat” was quantified for each species. By definition, rare species have lower probabilities of occurrence. When species-specific habitat suitability is summed at landscape scales (as we have done), the range in habitat amount is lower for rare species. When correcting for this issue by recalculating the proportion of habitat in fully contiguous landscapes to sum to 100%, the black-throated blue warbler and ovenbird models were more similar (23.07% and 30.65%, respectively). Confidence intervals for both adjusted segmented thresholds overlapped (see Betts et al. 2007a).

Influence of landscape variable(s) considered
Another source of variation in the threshold values obtained is the actual variable or set of variables used to characterize habitat at the landscape scale. By definition, habitats are species-specific. Hence, one may potentially improve the quality of models predicting the presence of a species by using spatial models incorporating the habitat variables influencing the distribution of a given species (Fig. 9.1; Betts et al. 2006b, 2007a). The thresholds we detected by using this species-centered approach tended to be more marked: segmented threshold models for both ovenbird and black-throated blue warbler were strongly supported. However, because we only considered three species with the species-centered approach, it is difficult to draw firm conclusions. The thresholds we detected by using this approach were fairly consistent: across study areas and among species, both segmented and ROC thresholds tended to range from 9% to 30%. Not surprisingly, species-centered thresholds were substantially lower than mature forest thresholds (Tables 9.1 and 9.2).

DISCUSSION
Do threshold responses to landscape structure provide robust conservation targets? Answering this question is critical if we are to use “landscape thresholds” to guide forest management and conservation planning. We expected threshold responses to be species-specific, but we did not know what to expect with regards to their consistency across space, time, or the spatial extents considered. Finally, we wanted to determine to what extent thresholds might vary as a function of the statistical method or analytical approach used.
G. Betts and M.-A. Villard

Geographic consistency in landscape thresholds

The first insight from our analyses is that thresholds do exist for some species in managed forest landscapes, even though boundaries among land-use elements are fuzzier than in many other managed landscape types (farmland, urbanized landscapes). The threshold values obtained also revealed a high degree of consistency through time for a given study area. However, there was fairly high variability in threshold values among study areas. This feature of the thresholds we detected implies that managers should exhibit considerable caution when extrapolating thresholds from region to another.

Given the relatively short distances between the study areas we examined, within-region variation in segmented thresholds as a function of both mature forest and habitat amount was somewhat surprising. There are several possible explanations for such variation. First, thresholds in response to forest age may have varied between Black Brook and Fundy simply because the composition of mature forest varied; mature forest in each case consisted of different proportions of habitat for each species. For this to be the main cause of variation, we would expect the species-centered approach to exhibit more geographical consistency. Based on the ROC threshold approach, this was true. However, when we used the segmented approach, organism-based thresholds did not appear to be much more robust to geographic location.

A second explanation for lack of geographic consistency relates to differences in sampling design among study areas. This may have influenced statistical power to detect effects. For instance, in Fundy most sampling points were located in mature forest (n = 505) whereas in Riley Brook, sampling points were systematically distributed, resulting in <1/3 of the samples in mature forest age class (n = 90). In support of this explanation, locally fitted regression splines (loess) plots showed surprising consistency between study areas, even though segmented thresholds were not consistently detected in Riley Brook (Fig. 9.4). A second aspect of this explanation relates to the range of sampling. For instance, it would have been possible to detect the same organism-based threshold for ovenbird in Black Brook and Fundy because very few landscapes containing more than 70% habitat (the Fundy threshold) were actually sampled in Riley Brook. Interestingly, the slope of the sub-threshold relationship in both areas was remarkably similar (Fig. 9.4). A third potential reason for differences in threshold values between study areas is the distinct radii used in Black Brook vs. Fundy (see Methods). Because the detection radius used in Black Brook was greater, we might have expected systematically lower thresholds. We did not find this to be the case. Finally, differences in the thresholds observed could be due to within-region variation in species sensitivity to landscape structure. This seems highly unlikely as only ca. 250 km separates north-western and south-eastern study areas.

Threshold variation among species

Not surprisingly, thresholds in mature forest amount tended to vary considerably among species. Such variations may be due to differences in each species' particular life-history characteristics. None the less, differences such as the ones observed here indicate that conservation planners and forest managers should consider the requirements of the most demanding species in a region, e.g. the Blackburnian warbler. Guenette and Villard (2005) also found that this species was among the most demanding with respect to structures associated with older forest stands.

Controlling for local habitat variation, organism-based thresholds ranged from 9% to 28%. Although this is substantial variation, as noted in the Results, part of this probably reflects variation in species prevalence. Interestingly, organism-based thresholds fell within the range (10%-30%) predicted by previous quantitative (Andrén 1994) and modeling studies (Fahrig 1998).

Thresholds in mature forest vs. habitat amount

We considered thresholds as a function of both mature forest and species-specific "habitat". Thresholds according to the organism-based approach tended to be much lower. This is likely because habitat for each species is a subset of the total amount of mature forest. There is some evidence that organism-based thresholds were more consistent over time and geographic space. However, as we only considered a small number of species, this question requires further research. It is important to note that, for some species, using an organism-based approach is important for detecting effects of landscape structure. For instance, we did not detect a strong influence of mature forest at the landscape scale on either ovenbird or black-throated blue warbler (Fig. 9.3a). In contrast, in Fundy at least, both species were strongly influenced by habitat amount in the landscape and exhibited strongly-supported thresholds (Table 9.2). If managers had considered only the single variable "mature forest", these sensitivities would have been missed.

On the other hand, using a "generic" land cover variable such as percentage of mature forest makes it possible to compare the responses of different species to the same landscapes. This latter approach may be more appropriate for use in some instances by conservation planners, particularly if the
habitat amount values below a rapid increase in probability of occurrence; that is, thresholds exhibit a “hockey stick” pattern rather than an asymptote (e.g. Fig. 9.3b). Although this may be of interest from the standpoint of understanding ecological processes, it is less useful from a management perspective. Managers should set targets for the amount of habitat above which there is a minimal increase in probability of species occurrence (asymptotes). However, a precautionary approach must be applied when using empirical thresholds as guides to define management targets. Thresholds in species richness as a function of habitat amount must also be interpreted with caution because species start declining in abundance at habitat amounts far above the actual threshold value (Chapter 8, this volume).

A key difference between the two threshold detection techniques is that the segmented approach will only calculate a threshold in the instance of a non-linear relationship between the predictor and response variables. It is important to note that we did not detect thresholds for several species in each study area. However, lack of a non-linear relationship does not imply lack of an effect. In the absence of a strongly non-linear response, managers must still rely on an objective approach to establish a conservation target. In such cases, the ROC approach seems highly appropriate. The simulation presented by Villard (Chapter 3, this volume) clearly illustrates this type of situation.

Nevertheless, it is critical to test for non-linear relationships even when using the ROC approach. In several instances, strong relationships between landscape structure and species occurrence could be masked by such non-linearities (Betts et al. 2007); very sharp thresholds are poorly modeled by linear models to the extent that important variables can be missed as a result of considering only linear relationships. Incorporating non-linear relationships also improves model calibration (Vaughan and Ormerod 2005); predicted and observed values are more closely matched.

In the case of both threshold detection methods, it is important to note that the thresholds correspond to the detection of a species at sampling points. If species detection is imperfect, threshold estimates may be prone to error or bias. In the current study, all species were characterized by reasonably high detection probability (75%–90%) (Farnsworth et al. 2004). The risk of bias obviously becomes greater with decreases in detectability (MacKenzie et al. 2003). Further research needs to be undertaken on accounting for detectability in statistical threshold tests.

The thresholds we report should not be considered as amounts of habitat below which a population will not persist. Such thresholds should be


Habitat thresholds and effects of forest landscape change on species occurrence. Hence, thresholds in the probability of occurrence of individual species should be viewed as what they are: minimum requirements for a species’ presence at a given location. Management for establishing “conservation blocks” within managed forest landscapes should err on the side of caution until monitoring data confirm that requirements of species being managed are actually met.

ACKNOWLEDGEMENTS

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References


INTRODUCTION

Habitats for species vary substantially in their stability, whether temporally or spatially. Some habitats, such as rocks and lakes, may be present for centuries and only subject to changes in climate, whereas others, such as dung patches or carcasses, may be available only for a single season or even shorter. This variation in predictability constitutes an important selective force for associated species. It has been suggested that the habitat is the template for ecological strategies (Southwood 1977). In this sense, individuals of a species are confronted with the challenge of determining whether reproduction is best achieved “here” or “somewhere else” as well as “now” or “sometime in the future”. By addressing these tradeoffs, the successful reproductive strategy will represent the life history of the species. Species adapted to long-lasting, predictable habitats are expected to generally be more sedentary; for them, the spatial distribution of the habitat is particularly important. This is a basic ecological starting point for appreciating the need to include time and space to a greater extent in forest conservation management. If only total habitat amount is considered, but the temporal and spatial distribution is ignored, important factors influencing the long-term viability of focal species will be missed and thus, at least for some species, the risk of decline and extinction will be severely underestimated.