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An indicator system to assess ecological integrity of managed forests

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A B S T R A C T

Ecological integrity of managed forests includes the ability of an ecosystem to support a community of organisms with a similar species composition and functional organization as found in nearby natural systems. We developed an indicator system for ecological integrity based on simulated natural disturbance and indicator species to test if forest condition and habitat in managed forests are similar to that found or expected in natural systems. We then applied the method in an area of the boreal forest (Ontario, Canada) where the objective of Ontario's strategic forest management planning approach is, in part, to conserve ecological integrity through the emulation of the natural disturbance process. Forest condition controls the supply of habitat to support the diversity of native organisms, and historically in boreal forests the natural disturbance process drove forest condition. We selected indicators of forest condition (landscape pattern and compositional mosaic) and habitat function (occupancy rates for a broad range of forest birds), and applied our assessment system to test whether indicators of forest condition and habitat function reflect outcomes expected if natural disturbance processes were successfully emulated. We collected occupancy data in natural and managed forest disturbance types using autonomous acoustic recorders, applied occupancy/detection modeling to estimate corrected occupancy rates (ψ), and then tested for differences in ψ between disturbance types. Some indicators of forest condition were within the range expected under natural disturbance, but we found relatively less old conifer, more young deciduous and greater edge density in managed forests relative to forests of natural disturbance origin. Most species (11 of 14) occurred with equal ψ in habitat originating from the two disturbance types. Brown creeper (*Certhia americana*), bay-breasted warbler (*Mniotilta varia*) and red-eyed vireo (*Vireo olivaceus)* differed between disturbance types. Brown creeper uses older conifer and occurred at lower rates in managed forest, while red-eyed vireo uses a range of deciduous forest ages, and occurred at higher rates in managed forest. Differences in quantity and/or quality of specific habitat types likely explain the responses. The results suggest what directional changes in the forest pattern and compositional mosaic would improve ecological similarity with natural systems, but also indicate what further research is required. We believe this approach to assessing ecological integrity can be adapted to study the effectiveness of conservation management strategies in other systems, and will contribute to adaptive management approaches and evidence-based policy development.

1. Introduction

In many European and North American jurisdictions boreal forest management and conservation planning have evolved from a focus on the production of a range of socio-economic products (including habitat for wildlife of significant socio-economic value) to a broader focus on ecological integrity, where the goal is sustainable provision of a range of ecosystem goods and services while conserving biodiversity and ecological processes (Hunter, 1999). Ecological integrity includes the ability of an ecosystem to support a community of organisms with a similar species composition and functional organization as found in nearby natural systems (Parrish

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et al., 2003), and contributes to ecosystem resilience, the capacity of ecosystems to absorb disturbances without undergoing fundamental change (Drever et al., 2006). For ecological systems where integrity has been conserved, the critical structural, functional, and process components of the system (e.g., forest condition, diversity and quality of habitat, and disturbance process) occur within the natural range of variation.

Forest management strategies for conserving ecological integrity remain largely untested (Drever et al., 2006; Klenk et al., 2009; Perera et al., 2007; Simberloff, 2001). An evaluation mechanism is required to objectively assess whether selected management approaches are indeed conserving ecological integrity, and to provide critical evaluation and feed-back for adjustment or abandonment of the approach. Without a mechanism to assess success, the management strategy becomes untestable and largely unscientific (Carignan and Villard, 2002), depending upon anecdotal ad hoc assessment of how well natural forest conditions have been conserved. Rather, it is useful to view the policy or management strategy as a hypothesis with an appropriate reference condition or null model (Thompson, 2006; Thompson and Harestad, 2004), and design a monitoring system that provides information to ultimately improve or reject the management strategy.

In practice, ecosystem processes are too complex and the number of species is too large to assess ecological integrity directly (Wiens et al., 2008). Simplified systems using indicators have been used to plan for and assess elements of integrity (Angelstam, 1998; Angelstam et al., 2003; Lambeck, 1997; Parrish et al., 2003; Villard et al., 2007; Wiens et al., 2008). From an adaptive management context the model system is most useful if the indicators relate directly to the management system, as this provides a feedback mechanism to adjust management based on monitoring outcomes.

In this study, we develop an assessment system for ecological integrity using forest bird species as indicators for forest condition and habitat function, and explore the application of this system in a managed boreal forest region of northwestern Ontario, Canada. We illustrate how a properly defined system can be linked to conservation and management policy objectives, and provide critical feedback for review and adjustment. We focus on emulation of natural disturbance because this is the strategic management approach adopted in the case study, but the approach could be applied to other conservation and management approaches.

2. Assessment system for ecological integrity

Our assessment system is designed to address the question, "has forest management emulated the conditions found in a natural forest, resulting in the diversity and quality of habitat necessary to support native biodiversity?" This question is fundamentally related to ecological integrity, which reflects both process and pattern. An assessment system of ecological integrity should relate key ecological processes (e.g., forest disturbance) to measurable patterns of ecosystem structure (e.g., forest condition) and function (e.g., diversity and quality of habitat) (Fig. 1). Forest condition drives the creation and supply of habitat, and some forest certification systems rely on forest condition as their indicator of how well biodiversity has been conserved. Although evaluation of forest condition is a necessary component of assessment, it is insufficient for evaluating ecological integrity because it does not directly evaluate the diversity and quality of habitat created. If habitat functions similarly between forest disturbance types, then the response of a broad range of representative wildlife should also be similar between forest disturbance types, reflecting similarity in diversity and quality of habitat.

2.1. Disturbance dynamics and forest condition indicators

Fire is a significant driver of natural disturbance in much of the boreal forest (Angelstam, 1998; Hunter, 1993; Rowe and Scotter, 1973), and affects three principal measurable characteristics of forest condition: pattern, composition, and structure (Fig. 1a,b). Forest pattern, including the interspersion of young and older forest, the size class distribution of young forest patches, and the contiguous nature of the mature forest matrix are all shaped by disturbance processes (Angelstam, 1998; Bergeron and Harvey, 1997; Hunter, 1993; Johnson, 1996; Perera and Buse, 2014; Rowe and Scotter, 1973). The forest composition (mosaic of deciduous and conifer species) is influenced by the interaction of soil moisture, nutrient availability and disturbance dynamics, while stand structure is largely driven by stand age. Stand age affects tree height and volume, accumulation of carbon stores and vertical and horizontal complexity.

Disturbance processes are largely stochastic, affecting the extent, intensity, and timing of disturbance events, successional pathways and post disturbance transitions. Consequently the expected natural forest condition cannot be measured directly, and the pre-industrial forest condition is only a single instance of how these factors combined for a particular outcome. Instead, we simulated natural disturbance on the landscape to estimate the natural forest condition and associated range of natural variation. We used a process-based landscape-level simulation model that integrates the Canadian Forest Fire Behavior Prediction System and Fire Weather Index system (Wotton et al., 2009) with locally calibrated, empirical forest succession rules driven by a time-dependent Markov model (Perera et al., 2008).

2.2. Forest condition and habitat indicators

Forest condition (e.g., landscape interspersion pattern, tree composition and stand structure) contributes to the diversity and quality of habitat that supports native biodiversity (Fig. 1c). If habitat functions equivalently between naturally disturbed areas and those created through forest harvesting, then there should be no difference in the diversity and quality of habitat between disturbance types. Diversity of habitat can be revealed by those species with the most extreme environmental requirements, as they are representative of the range of variability in habitat function (Carignan and Villard, 2002; Lambeck, 1997). Habitat quality refers to all aspects of habitat related to individual fitness, including forest conditions necessary for attracting mates, breeding, rearing, food sources, predator avoidance, etc., and consequently is difficult or impossible to measure directly. Instead we use the response (occupancy rate) of representative wildlife as a surrogate for habitat quality, where occupancy rate is expected to be similar if habitat function (diversity and quality) is similar between forests of natural and managed disturbance origin.

We used forest birds as indicators because they occupy a broad range of forest habitat types and food sources, are responsive to the types of changes in forest condition caused by forest management, are an unexploited species, can be cost-effectively and unobtrusively monitored, vocally defend breeding territories, and are a high conservation priority and responsibility for resource managers in the boreal forest. Collectively the food resources required by forest birds are diverse, with birds feeding on a host of insects, ground invertebrates and seeds (Canterbury et al., 2000). Conceptually, species are selected from the corners of the habitat niche-space box, where collectively these species occupy a broad range in habitat conditions representative of the natural range of variation (Fig. 1c) in critical landscape condition variables. If the box "shrinks" or "shifts" because forest management is not sufficiently emulating the suite of natural forest conditions

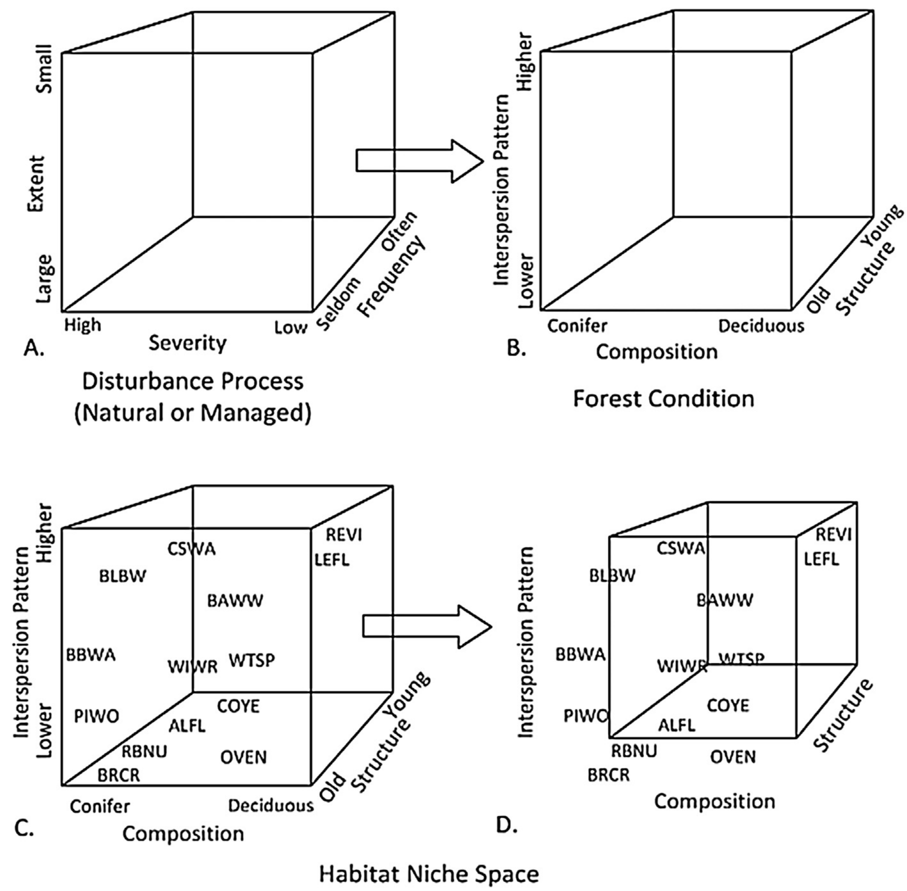

Fig. 1. Disturbance process and habitat model, illustrating (A) the 3 axes defining the disturbance process (extent, frequency and severity of disturbance), (B) the 3 indicators of measurable forest condition (landscape interspersion pattern, forest composition and stand structure), and (C) the relationship between forest condition and diversity of habitat. "Shrinking the box" (D) represents a reduction or shift in the diversity and quality of habitat. Bird species codes defined in Table 1.

(Fig. 1d), then species associated with habitat condition extremes will respond first. A "shift" in the direction of a critical landscape condition implies an increase in the availability of that condition (e.g., deciduous forest), whereas a "shrink" implies a decrease.

In this study we selected 14 forest birds that represent the natural range of variability in forest condition and habitat types (Table 1). Previous analysis of habitat associations using community ordination (CCA) and resource selection functions revealed that these species demonstrated statistically significant associations with the most extreme forest conditions (Rempel, 2007). Collectively, these species revealed strong habitat associations ranging from high to low edge density relating to stand age, fragmented to contiguous forest matrix, deciduous to conifer dominated overstory, young to old stands, and open to closed canopy (Rempel, 2007). For example, brown creeper (*Certhia americana*) requires older conifer stands set in a landscape context of a contiguous mature forest matrix (Poulin and Villard, 2011; Rempel, 2007), while red-eyed vireo (*Vireo olivaceus*) requires younger deciduous dominated stands set in a context of high edge-density (Lawrence, 1953; Rempel, 2007).

3. Case study – methods

3.1. Study area

As a case study, we applied the assessment system in a study area located in ecoregion 3W, which surrounds Lake Nipigon, north of Lake Superior (Fig. 2) (Crins et al., 2009). Vegetation is typical of the boreal forest, with conifer tree species including black spruce (*Picea mariana*), white spruce (*P. glauca*), balsam fir (*Abies balsamea*), and jack pine (*Pinus banksiana*), with pockets of white pine (*P. strobus*) and red pine (*P. resinosa*) in southern portions (Crins et al., 2009). Deciduous tree species are dominated by trembling aspen (*Populus tremuloides*) and white birch (*Betula papyrifera*).

We stratified sampling using broad forest categories of young (presapling, sapling, and immature seral stages) versus old (mature and late seral stages), conifer (≥50% conifer) versus deciduous (>50% deciduous), and high versus low age class interspersion of stands in the surrounding landscape (using contrast weighted edge density). We selected 292 survey sites comprised of 254 upland stands and 38 wetlands. Wetlands were sampled in both managed and natural forests, included fens, bogs, swamps and marshes, and all wetlands has a least one treed fringe. To account for differences in forest structure, sites were dispersed across broad seral stage groups, with 196 and 96 sites in older and younger forest, respectively. Sites were spatially dispersed across the study area, and were placed within 90 landscapes, where we delineated landscapes as multiple, contiguous forest stands associated with a common disturbance event, e.g., wildfire or forest harvest spanning a period of <5 years (Fig. 2). Satellite imagery, digital forest inventory maps, and on-site inspection were used to delineate landscapes and verify disturbance history. Within this stratification we sampled 133 sites in naturally disturbed landscapes and 159 in managed landscapes. We placed 3–5 survey sites within each landscape, each survey site was >100 m from a road or significant edge, and sites were spaced >1000 m from each other. Where available a forest wetland site was associated with a landscape.

Table 1
Selection of occupancy (ψ) models based on AIC for 14 focal boreal bird species.

Common name	Code	Scientific name	Model specification[a]	Evidence for disturbance type[b]	Weight for disturbance type model[c]
Alder flycatcher	ALFL	*Empidonax alnorum*	$\psi(\cdot)$, $p(\cdot)$	0.00	0.27
Black and white warbler	BAWW	*Mniotilta varia*	$\psi(\cdot)$, p(DSS)	0.00	0.36
Bay-breasted warbler	BBWA	*Setophaga castanea*	ψ(DT), p(RQ,RH)	**2.14**	**0.74**
Blackburnian warbler	BLBW	*Setophaga fusca*	$\psi(\cdot)$, p(DSS,pHWV)	0.00	0.36
Brown creeper	BRCR	*Certhia americana*	ψ(DT) p(RQ)	**6.51**	**0.96**
Common yellowthroat	COYE	*Geothlypis trichas*	$\psi(\cdot)$, p(RQ,RH,pHWV)	0.00	0.39
Chestnut-sided warbler	CSWA	*Setophaga pensylvanica*	$\psi(\cdot)$, p(DSS)	0.00	0.30
Least flycatcher	LEFL	*Empidonax minimus*	ψ(DT), p(RAIN, pHWV)	0.95	0.62
Ovenbird	OVEN	*Seiurus aurocapilla*	$\psi(\cdot)$, p(RAIN,RQ, pHWV)	0.00	0.40
Pileated woodpecker	PIWO	*Dryocopus pileatusis*	$\psi(\cdot)$, $p(\cdot)$	0.00	0.27
Red-breasted nuthatch	RBNU	*ta canadensis*	$\psi(\cdot)$, p(RQ)	0.00	0.00
Red-eyed vireo	REVI	*Vireo olivaceus*	$\psi(\cdot)$, p(DSS,RQ,RH)	**14.21**	**0.99**
Winter wren	WIWR	*Troglodytes hiemalis*	$\psi(\cdot)$, p(DSS, RQ)	0.00	0.43
White-throated sparrow	WTSP	*Zonotrichia albicollis*	$\psi(\cdot)$, p(RQ,RH)	0.00	0.22

[a] Selected occupancy/detection model, where $\psi(x)$ indicates which variable was included in the occupancy model, and $p(x)$ indicates which detection variables were included. DSS = days since spring; DT = disturbance type; RQ = recording quality; RH = relative humidity; RAIN = rainfall in mm; pHWV = proportion hardwood (deciduous) volume in the stand.

[b] Evidence for disturbance type is the difference in AIC between best occupancy model and model that also includes disturbance type. If value = 0 then the best model did not include disturbance type; otherwise value is amount AIC decreased when disturbance type was added. Bolded entries have ΔAIC values > 2.

[c] Weight for model that includes disturbance type relative to model without disturbance type. Maximum weight = 1.

3.2. Forest bird occupancy rate

We used automated recording units (Wildlife Acoustics Inc.) to collect recordings of forest bird vocalizations (Rempel et al., 2013) while simultaneously collecting information on conditions that could influence their detectability. Units were programmed to record for three 10-minute periods each morning (at sunrise and 30 min before and after sunrise). At each survey site we made recordings over a contiguous period of 2–5 days, and selected up to 6 recordings at each site as repeat samples (of which one was an evening recording at wetland sites), for a total of 1168 10-minute recordings. Detailed technical specifications and performance of the recording units are found in Rempel et al. (2013).

Detection covariates (temperature, humidity, rain, recording quality, Julian date, and time-since-sunrise) were recorded for each site observation to model their effect on detection probability and improve the estimate of occupancy rate (ψ) (MacKenzie et al., 2006). Detection models correct for imperfect detection (i.e., when the bird occupies a site but is not detected by the observation method for one reason or another). For all but one species (pileated woodpecker) we studied male birds that defended discreet breeding territories of <5 ha in size, where the majority of the territory would likely be within detection distance of the audio recorder. We assumed no new territories were established or abandoned during the 3–5 day sample period. We used differences in Akaike's Information Criterion (AIC) to select the best detection model, and then assessed goodness-of-fit to ensure the detection model was plausible (MacKenzie and Bailey, 2004).

We used estimated occupancy rate (ψ), based on presence/absence, rather than population density as the appropriate response variable because most species were observed at spatial scale of individual territories, and most observations were of 0 or 1 individual of a species at a site. In our study, we could not estimate distance from the recording unit, which is necessary for estimating corrected density. By occupancy rate we explicitly mean the probability that a male bird used a survey site as a possible breeding territory at least once during the 2–5 day contiguous recording period (and within the 6 week breeding season). Area of a site is defined by the species-specific audio detection distance (1.8–4.9 ha). Because birds are detected through audio cues given while defending breeding territories and attracting mates, we interpreted ψ as an indicator of habitat quality, with lower ψ expected in habitat types of lower quality that do not sufficiently provide for life history needs during the breeding seasons. Collectively, ψ estimates the proportion of sites in the overall survey area that were used at least once during the breeding season.

After accounting for effects on detection we assessed whether disturbance type (natural disturbance versus forest management) influenced ψ by adding disturbance type (DT) as a site habitat variable in the occupancy model. If AIC decreased by >2 we concluded there was sufficient evidence to select the model inferring a difference in ψ between natural and managed sites; ψ and associated standard errors were plotted to visualize the direction and magnitude of differences (effect size) between disturbance types.

The overall percent similarity of occupancy rates (Buckland et al., 2005; Lamb et al., 2009) among of all focal species was calculated to summarize and compare the similarity, or intactness, of the focal species community between landscape origins. The occurrence similarity index (OSI) evaluates the proportion of sites occupied by the focal species in the natural versus managed landscapes, and is calculated as:

$$\mathrm{OSI} = \frac{1}{s}\sum_{t}\frac{\min(f,h)}{\max(f,h)} \times 100,$$

where s = number of species, and f and h are corrected occupancy rates (ψ) for each of i species in natural and forest management landscapes, respectively. This metric provides an aggregate index of species intactness that can supplement assessment of individual species responses, and that is appropriate when the estimated species response is occupancy rate.

3.3. Forest condition (landscape simulation)

We simulated natural disturbance patterns on the landscape using BFOLDS (Perera et al., 2008) to produce 80 maps of disturbance for the study area, and these were used to estimate the range of expected natural forest composition and landscape pattern associated with the forest condition indicators. To characterize the landscape compositional mosaic we used broad forest cover groupings comprised of 7 classes representing composition and seral stage (Fig. 3), and for each landscape class we calculated the median and expected range of variability (25th to 75th percentiles, termed inter-quartile range). To estimate departure from expected natural conditions we compared these simulated values to values observed from 2011 digital forest resource inventory (FRI) maps.

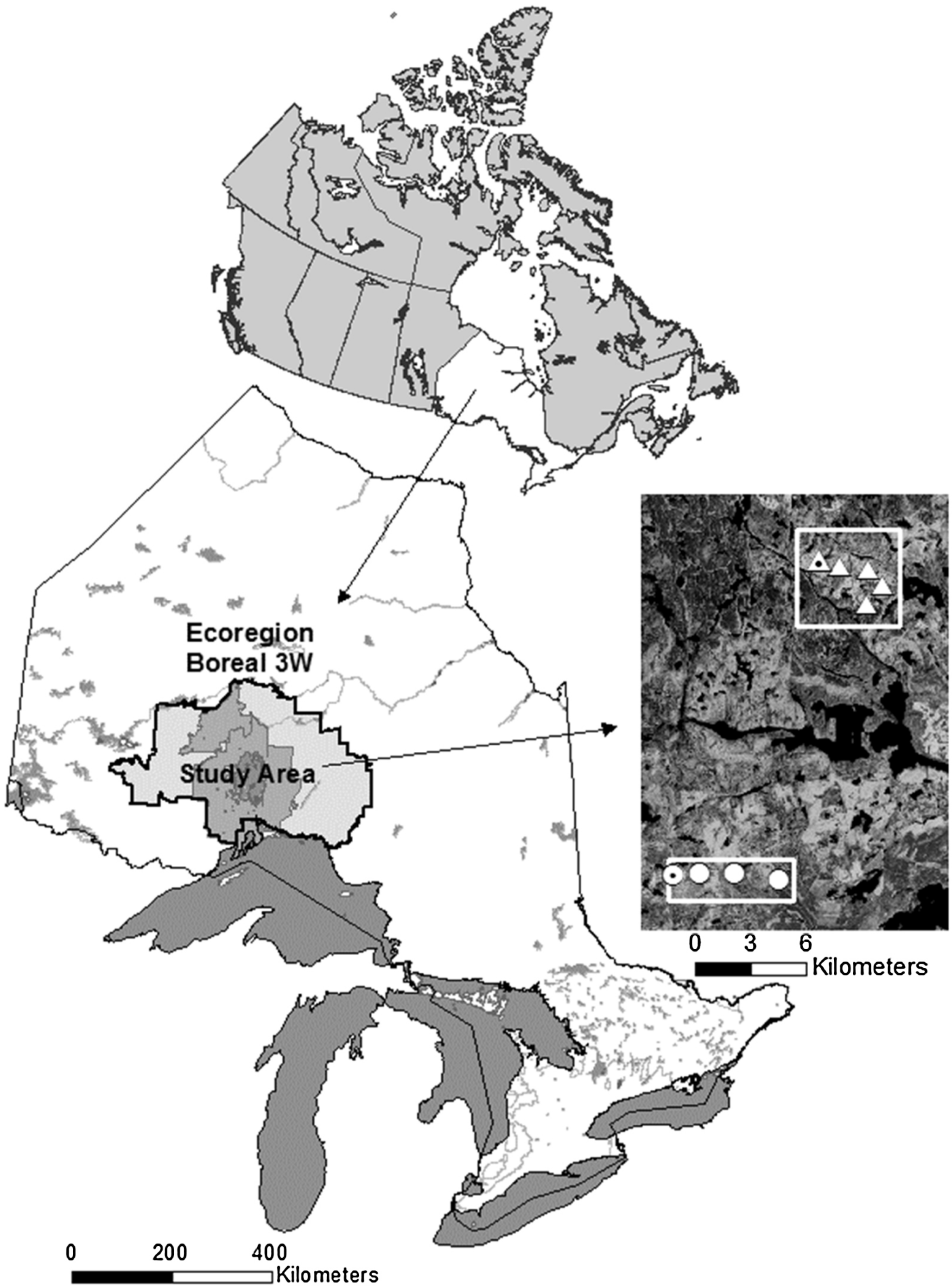

Fig. 2. Location of the study area (darker shading) within ecoregion 3W in boreal Ontario, Canada. 292 sample sites were positioned within 90 burn or harvest landscapes spread across the study area. Inset shows 5 and 4 survey sites in a natural disturbance and managed forest landscape, respectively. Symbols with dots are wetland sites associated with the landscape. All survey sites separated >1000 m.

To characterize and compare landscape pattern of the mature forest matrix we used a histogram to depict the frequency distribution of 500 ha cells ranging from filled (100%), partially filled (50%), to devoid (0%) of older forest (Fig. 4). A u-shaped frequency distribution indicates a coarse-grained pattern with many cells (large patches) of either disturbed or undisturbed forest, while a bell-shaped distribution indicates a fine-grained, highly fragmented matrix, with most cells partially disturbed, and very few undisturbed. To characterize and compare landscape pattern created by disturbance patches we used the size class-frequency distribution of young forest patches.

3.4. Forest condition (observed at survey sites)

We used 2011 FRI maps to compare forest condition across multiple spatial scales in natural disturbance versus forest management survey sites. We generated hexagon-based spatial summaries using spatial analysis units (SAUs) of 2, 50, and 5000 ha, and used hexagon offsets to spatially average estimates (Table 2), and assigned these values to each of the 292 survey sites. At the local stand level (2 ha SAU) we calculated average tree height (HT), proportion of young forest (pYng), mean stand age (AGE), mean crown closure (CC) and proportion of hardwood (deciduous)

Table 2
Forest condition variables associated with survey sites, characterized at the stand (2 ha), ecosite (500 ha), and landscape (5000 ha) levels.

Code	SAU area (ha)	Representative scale	Unit	Managed (*N* = 159)	Natural disturbance (*N* = 133)	Description[a]
Age	2	Stand	Years	61.43 (2.79)	45.86 (3.67)	Age of dominant trees in the stand, interpolated from tree height/site class relationships. Represents time since disturbance (structure).
CC	2	Stand	Percent	35.95 (1.24)	27.20 (1.84)	% canopy closure in the stand based on density of trees (structure).
HT	2	Stand	m	11.04 (0.50)	8.24 (0.64)	Photo-interpreted mean height of dominant trees in the stand (structure)
pHWV	2	Stand	Proportion	0.29 (0.03)	0.17 (0.02)	Proportion hardwood (deciduous) volume, based on Plonski yield table[b] using age and site class (composition).
pYng	2	Stand	Proportion	0.24 (0.03)	0.16 (0.029)	Proportion young forest, pre-sapling and sapling seral stages (structure).
pWet	50	Ecosite	Proportion	0.02 (0.00)	0.48 (0.01)	Proportion wetland, including shrub wetlands, marshes, bogs and fens.
pMat	5000	Landscape	Proportion	0.54 (0.01)	0.38 (0.03)	Proportion mature forest (mature and late seral stages). Represents intact forest matrix at landscape level (pattern).
EdgeDen	5000	Landscape	m/ha	18.98 (0.43)	13.54 (0.78)	Contrast weighted edge density, where weight of immature forest juxtaposed with young or mature forest has a reduced weight of 0.5. Represents age-class interspersion (pattern).

[a] All variables based on photo-interpreted, digital forest resource inventory (FRI) attributes.
[b] Payandeh (1991).

volume (pHWV) within an SAU. Deciduous volume was estimated using a modified version of Plonski's yield tables (Payandeh, 1991) that relate tree height and site productivity to yield for each species. At the ecosite level (50 ha SAU) we calculated proportion of wetland (pWet), including shrub wetlands, open marsh, and treed bog and fen. Landscape context of individual sites was evaluated at the landscape level (5000 ha SAU), where we calculated contrast-weighted edge density (EdgeDen) and proportion of mature and late seral stage forest (pMat).

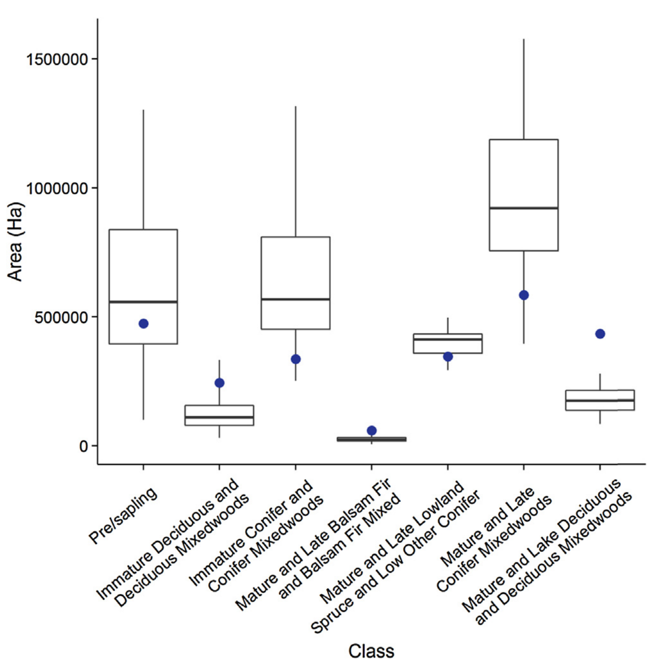

Fig. 3. Area of each landscape class observed in forest resource inventories for the current managed forest (2011) (solid dots) relative to expected area (simulated natural disturbance regime). Vertical lines are the full range, open boxes the interquartile range (IQR), and horizontal lines the median value from the 80 simulation runs.

4. Results

4.1. Forest condition indicators

At the landscape level, we found that the forest has a compositional mosaic with higher levels of immature deciduous and deciduous mixedwoods, and mature and late deciduous and deciduous mixedwoods than expected under a natural disturbance regime (above the interquartile range; Fig. 3). In addition, the current managed forest has lower levels of immature conifer and conifer mixedwoods and mature and late conifer mixedwoods than that expected (Fig. 3). At the survey site level we also found a greater proportion of deciduous forest and less wetland area associated with forest management sites (Table 2; Fig. 5).

Landscape pattern analysis suggests a higher level of age-class interspersion and fragmentation of the mature forest matrix than that expected under a natural disturbance regime (Fig. 4). For example, partial disturbance in the 0.41–0.60 SAU range occurs across 21% of the landscape rather than the expected 11% (Fig. 4). At the survey site level, forest management sites were associated with higher edge density (Table 2; Fig. 5). We found fewer small disturbance patches (<100 ha) but more medium to large patches (100–2500 ha) relative to expected (Fig. 6). However, there are also fewer of the very large patches (>2500 ha) relative to what is expected in a natural forest. Relative to expected natural conditions, there is less overall variability in patch size distribution, with a predominance of more moderate sized patches in the current managed forest.

4.2. Habitat indicators

Three of the 14 habitat indicator species, bay-breasted warbler, brown creeper, and red-eyed vireo showed strong evidence that disturbance type affected ψ, with ΔAIC > 2 and with AIC model weights ranging from 0.74 to 0.99 (Table 1). This is evidence that habitat differs between disturbance types. Least flycatcher showed very weak evidence, with ΔAIC of 0.95 and model weight of 0.62, so did not meet our ΔAIC threshold of 2.0 to conclude that ψ differed

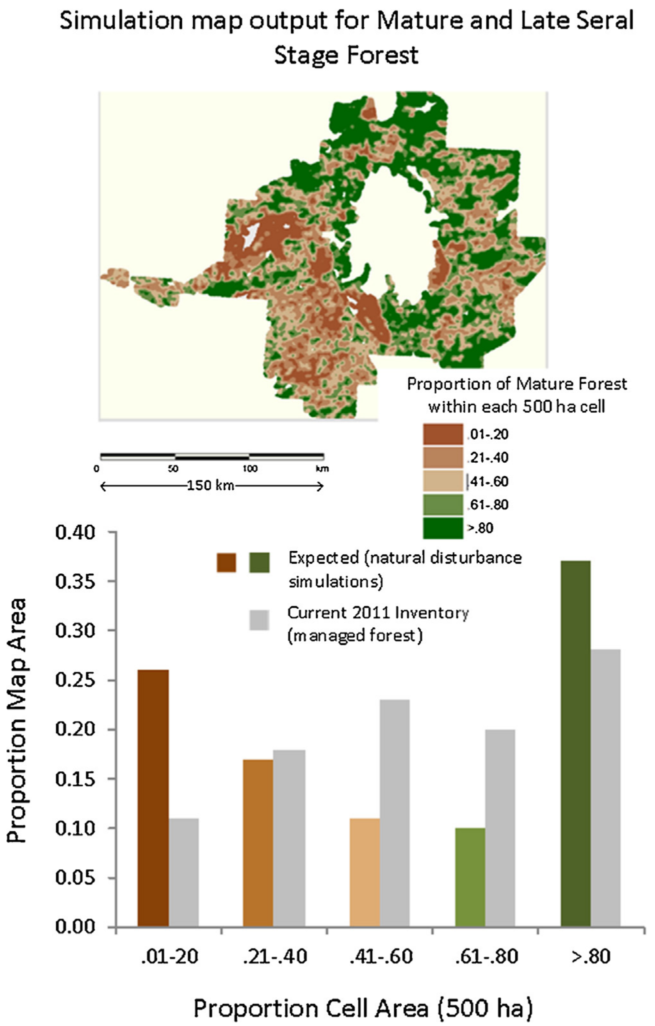

Fig. 4. Landscape pattern (texture) analysis. Bars represent the proportional area of 500 ha spatial analysis cells found within the simulated maps, and are grouped by the proportion of older (mature and late seral stage) forest found within individual analysis cells. Light gray bars represent comparison to current (observed) forest resource inventory. The simulated forest has a u-shaped distribution, indicating relatively little matrix fragmentation, with most cells either relatively full or empty of mature forest.

between disturbance types. The magnitude of the differences and variance within estimates of ψ are revealed using variables standardized to unit variance, and for brown creeper and red-eyed vireo the standard errors of ψ are separated, while for bay-breasted warbler the error bars slightly overlap (Fig. 7). The overall similarity of ψ between disturbance types (OSI) for the 14 focal species was 79.4 (where 100 is the maximum value).

5. Discussion

Ecological integrity is the outcome of complex ecological interactions, and is likely impossible to measure directly. Our simple indicator-based assessment system addresses two important elements of forest ecological integrity, disturbance process and habitat function, using forest condition and forest birds as surrogates. The process to develop the assessment system involves having a clear understanding of the dominant ecological processes that structure forest condition and habitat, and knowledge of which species are associated with the extremes in habitat condition. This requires more background investigation than using indices based on traditional feeding and nesting guilds, as the linkages between

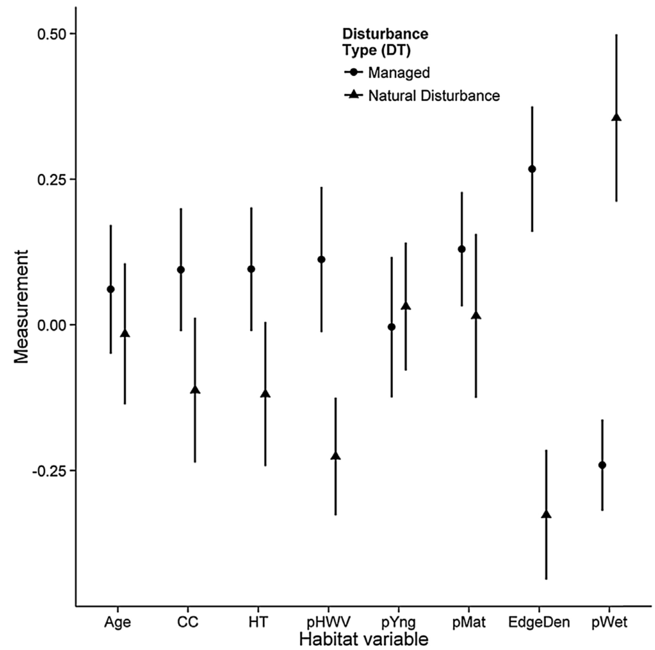

Fig. 5. Comparison of standardized (z-deviate) forest condition variables (and 95% confidence limits) between managed (N = 159) and natural disturbance (N = 133) survey sites (DT) for tree age (Age), canopy closure (CC), tree height (HT), edge density (EdgeDen), proportion hardwood (deciduous) volume (pHWV), young forest (pYng), mature forest matrix (pMat), and wetland (pWet). Variables and scales described in Table 2.

ecological process and habitat in the area of study must be first identified, relationships established, and representative species selected based on an understanding of community ecology and resource selection. The benefits, however, are stronger and more clearly defined linkages between management actions and their influence on ecological integrity than can be revealed through traditional feeding and nesting guilds. For example, where management has the objective to create specific conditions (e.g., emulate

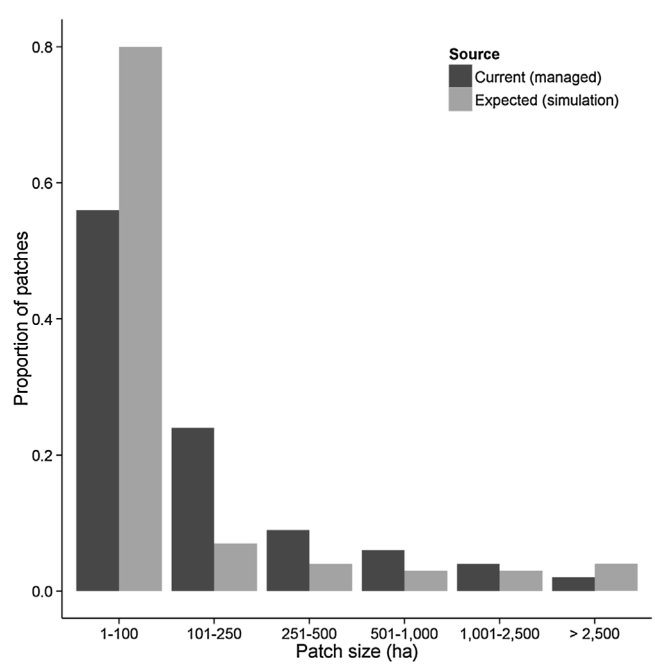

Fig. 6. Comparison of patch size class-frequency distribution of young forest (age < 36 years) between 2011 inventory for the current managed forest and expected (natural disturbance simulations).

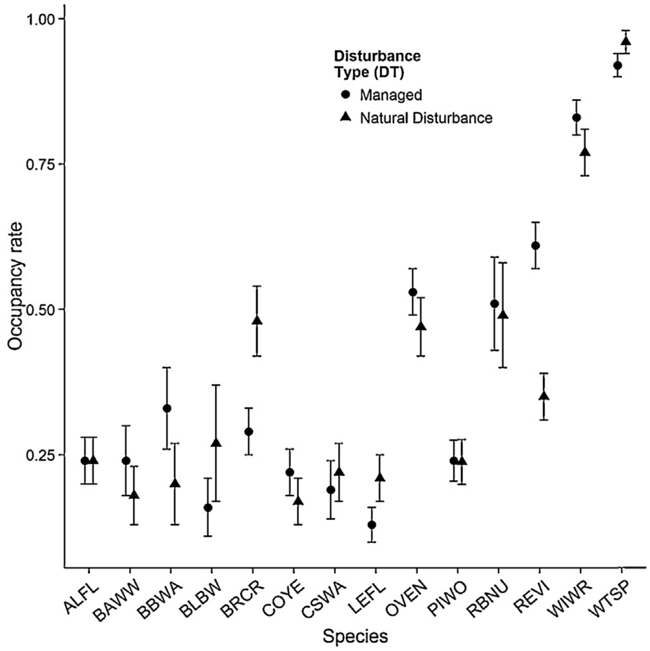

Fig. 7. Comparison of occupancy rate (ψ) between managed and natural disturbance forest, with standard errors of ψ indicated by vertical bars. Occurrence Similarity Index (OSI) comparing overall similarity of ψ between disturbance types = 79.4. Species codes defined in Table 1.

natural forest patterns), then biota that respond directly to those conditions will have a higher correlation with management actions (Canterbury et al., 2000; Szaro, 1986). If edge-density at the landscape level is changed, then biota that responds to edge-density at that level should be included as indicators. Likewise, if the disturbance gradient of young to old forest is managed, then species reflecting this gradient should be selected (Croonquist and Brooks, 1991).

We applied the indicator system to explore its performance and assess its value for evaluating ecological integrity in a case study in northern Ontario. The indicator system did not provide a simple pass/fail evaluation, but rather provided insights into where forest management needs improvement to maintain ecological integrity, and identified areas where more specific research is required. For example, we found that the forest condition indicators revealed the managed landscapes had relatively more deciduous, less conifer, a more fragmented mature forest matrix and greater edge density relative to natural disturbance landscapes. Increased amount of deciduous forest in the managed landscapes may reflect how the regeneration strategies of different tree species facilitate differential adaptation to fire and harvest (e.g., Carleton and Maclennan, 1994). This may also be related to inadequate regeneration of conifer or control of deciduous without fire (i.e., inadequate tending resulting in conifer plantation failure and deciduous pioneer species dominance). Greater edge density is likely partly due to previous guidance that encouraged the creation of edge to support moose populations, i.e., small dispersed clearcuts with internal residual (Rempel et al., 1997).

There was significantly less wetland area in survey sites located in managed forest landscapes, and this may be the result of previous guidance that discouraged harvest of riparian forest, and this may have inadvertently caused a decline in American beaver (*Castor candensis*) and the creation of beaver-controlled wetlands (Naylor et al., 2012). Alternatively, wetland sites were generally selected close to roads, and a road-building bias against wet areas may have also contributed to lower wetland area in the managed forest survey sites.

The forest birds we selected were shown in previous work to be responsive to forest condition, yet overall had a 79% similarity in occupancy rates between disturbance types. However, the response of three species suggests that past forest management practices in Ontario have resulted in a forest condition that differs from that expected in a natural forest. These differences in forest condition appear to have changed the quality and supply of habitat, as indicated by higher or lower ψ for 3 of the 14 species. Brown creeper occurred at lower rates in managed landscapes relative to natural disturbance landscapes, while the red-eyed vireo and bay-breasted warbler showed the opposite pattern. Red-eyed vireos show an affinity for deciduous or mixed forest (Cimprich et al., 2000) while brown creepers show an affinity for conifer or mixed forest (Poulin et al., 2013). Thus, trends for these species were not surprising considering the increase in deciduous and decrease in conifer in managed landscapes. Given that similar responses did not occur for other species, it is possible that some qualitative nature of the deciduous and conifer forest differed with respect to specific habitat needs for these species.

Model selection revealed ψ for bay-breasted warbler was influenced by disturbance type, but the effect size was small, with no separation of standard errors. The response of bay-breasted warbler increasing in managed forest landscapes was surprising since it is considered a species of conifer and mixed forest (Venier et al., 2011). Bay-breasted warbler is a spruce budworm (*Choristoneura fumiferana* (Clemens)) specialist (Venier et al., 2009), a lepidopteron that prefers balsam fir and white spruce, so this warbler may be responding to other insect prey (e.g., forest tent caterpillar (*Malacosoma disstria* Hubner)) associated with the increase in poplar forest in managed landscapes. Alternatively, bay-breasted warbler may be responding to forest regeneration practices at the site level that favor immediate creation of pure conifer stands over natural forest succession that more gradually moves to a conifer state (Bergeron and Harvey, 1997). These explanations are speculative, and further study is required to better understand the response of this species, but regardless, results suggest habitat function differs in some qualitative manner between the two disturbance types.

In our study we used a simple probability of occupancy model to focus on the general indicator approach, and chose to include only disturbance type as a habitat covariate. In preliminary analysis we found no substantive effect of additional habitat covariates on the interpretation of how disturbance type affects overall occupancy. Alternative statistical approaches could include Bayesian modeling to incorporate site specific habitat characteristics, or analysis of covariance methods. By accounting for amount of habitat in more complex statistical models, further insights could be revealed concerning differences in the nature or quality of specific habitat types.

We were not able to estimate distance from the audio recordings, a prerequisite for accurately modeling density, but this may indeed be possible given further study on sound transmission among different species. Density is especially important for estimating population viability, as low densities put populations at extinction risk. Occupancy rate is suitable for comparing habitat use between disturbance types, and for 13 or 14 species we were confident in the accuracy of the occupancy estimates. We had less confidence in estimates for pileated woodpecker, which was included here because of its keystone importance in nest webs. It is an exception among the other indicator species as its territory is >100 ha, and for this species detection probability is related not only to site level factors, but also movement within its very large territory. For pileated woodpecker the comparison of ψ between disturbance types is still valid, but the magnitude of ψ should be interpreted with caution.

Changes in landscape pattern and the responses of some of our indicator species suggests that historical forest management in Ontario has "shrunk the box", causing variability in forest condition to be less or different than what would have occurred under natural conditions. With the insights provided by forest condition and habitat indicators, direction can be provided to ensure modern forest management practices better emulate the patterns and processes created by natural disturbance at a variety of scales and help restore the natural range of variability. For example, Ontario's recently implemented forest management guides provide direction to better reflect the natural forest compositional and age mosaic expected under a natural disturbance regime (Ontario Ministry of Natural Resources, 2010, 2014). As noted above, more research on why the 3 species differed in occupancy rates could provide additional insights into how forest management could be changed to better reflect natural conditions.

We also calculated a summary statistic (OSI) that describes the overall "community" similarity of indicator species-occurrence within the two disturbance types. Although this biotic index of species intactness (79.4 in this study) provides a simple metric for trend-over-time monitoring, and perhaps could be used as a scorecard of how well forest management is emulating natural forest, the index itself it does not easily provide clear insight that can be used to direct change to forest management. Similar issues exist with summary metrics of species diversity and evenness, and in addition, the reference starting values will become less meaningful as species are added or removed as indicators. It is not that aggregate indices are without value. For example Canterbury et al. (2000) and O'Connell et al. (2000) used a bird community index based on cumulative distribution of selected forest birds to assess forest condition, and community biotic indices were originally used in aquatic systems (Karr, 1981) to assess integrity. These approaches can provide a useful summary of system response and may increase understanding of functional change; but as noted by Canterbury et al. (2000), may also need to be supplemented with species specific information to interpret index values and understand what they mean to forest management.

There are several key assumptions and limitations of the approach, however, which need to be identified. Our approach uses occupancy as an indicator of the state of habitat quality, and there are limitations to occupancy rate as a state variable. For example, there is concern that higher occupancy rates may reflect habitat sinks, and that for some species habitat preference may be associated with non-ideal habitat (Arlt and Pärt, 2007). Some studies have shown that the timing or order of settlement may better reflect site preference and habitat quality, with older males settling in higher quality habitat first, or shifting their occupancy to select better quality habitat as site conditions become more revealing (Betts et al., 2008). Further research into relationships between timing of settlement, habitat quality and individual finesses, including use of dynamic occupancy models may improve estimates of habitat quality based on occupancy.

The approach directly tests whether designed landscapes can support the focal forest songbird species, but does not test whether the landscapes can support all other non-focal species, including those whose habitat falls with the bounds of the "habitat box". In this study we followed Lambeck (1997) by defining critical landscape parameters and then selecting focal bird species that required "extreme" levels of these parameters. Our study differed from Lambeck (1997) in that we based focal species selection on process limited factors, but did not rank highly whether or not the population was secure, in part because our focus was on maintaining quality habitat conditions rather than reconstruction of degraded habitat. Indeed, species with sufficient data to demonstrate strong habitat associations with forest condition were among the more abundant populations. While selecting focal species with demonstrated strong association with ecological process (i.e., habitat dynamics) was useful in developing the model system, future implementations should consider including species at risk whose population is less secure. A stressor and risk-based vulnerability assessment could be used to select appropriate species at risk. Future studies could evaluate whether other groups of species, including invertebrates, mammals, and amphibians are also limited by these same parameters. If not, then those species with different landscape and stand level requirements should be included to broaden the scope of the assessment model. Finally, monitoring programs should test the assumption that landscape viability is being maintained for non-focal species by monitoring for some time a broad range of non-focal species.

Selection of landscapes (and associated wetlands) was not random, and was designed to ensure balanced coverage of habitat types, spatial separation of both landscapes and sample points within landscapes, and be relatively efficient in terms of cost and distance from roads. Great care was taken to reduce bias, and it is generally believed that the effect of forest roads diminishes after 100 m, but nonetheless it is assumed that results are not affected by the sample selection strategy. Future studies could attempt a random sampling approach, and compare similarity of results and overall cost with the more stratified approach.

The concept of the "shrinking" and "shifting" habitat box is useful, but in some cases fails to exactly capture the nature of species' response. For example, the increase in occupancy rate for red-eyed vireo only partially reflects the concept of a shifting box, in that the box shifts toward the condition of more deciduous forest, outside of the bounds of natural variation; it is important to not over extend the utility of the conceptual model. In addition, species located within the "centre" of the box may require more general habitat conditions (e.g., mixedwood forest), and so should not be dropped too quickly from consideration as focal species.

Forest management can never exactly duplicate the complex and extensive range of conditions created by natural disturbance processes; hence forest managers need to decide "how close is close enough" in their assessments. This decision will be partly based on what is socially acceptable and practically feasible. Useful indicators should assess if management has already deviated from the natural range of variability in forest condition to the extent that current condition risks degrading ecological integrity and ecosystem resilience. A logical next step is to more precisely map the threshold response of indicators; this would allow managers to better understand where risk to integrity changes precipitously.

6. Conclusions

A system for assessing ecological integrity should integrate indicators of both ecological process and pattern, be informative, and be linked explicitly to management objectives and actions. Properly selected indicators should provide insight into how management actions can be improved relative to the principal ecological objectives. Our indicators of forest condition (landscape pattern and compositional mosaic) and habitat (ψ for a broad range of forest birds) reveal what directional changes in the forest pattern and compositional mosaic are required to improve ecological similarity with natural systems. The assessment approach described here can contribute to both evidence-based policy analysis and adaptive management for these more complex ecological objectives, provide insights into why undesirable effects are occurring, and ultimately lead to faster improvements in forest management policy that strives to protect ecological integrity.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.ecolind.2015.08.033. These data include Google maps of the most important areas described in this article.

References

Angelstam, P.K., 1998. Maintaining and restoring biodiversity in European boreal forests by developing natural disturbance regimes. J. Veg. Sci. 9, 593–602.

Angelstam, P.K., Butler, R., Lazdinis, M., Mikusinski, G., Roberge, J.-M., Buetler, R., 2003. Habitat thresholds for focal species at multiple scales and forest biodiversity conservation: dead wood as an example. Ann. Zool. Fenn. 40, 473–482.

Arlt, D., Pärt, T., 2007. Nonideal breeding habitat selection: a mismatch between preference and fitness. Ecology 88, 792–801.

Bergeron, Y., Harvey, B., 1997. Basing silviculture on natural ecosystem dynamics: an approach applied to the southern boreal mixed wood forest of Quebec. For. Ecol. Manag. 92, 235–242.

Betts, M.G., Rodenhouse, N.L., Scott Sillett, T., Doran, P.J., Holmes, R.T., 2008. Dynamic occupancy models reveal within-breeding season movement up a habitat quality gradient by a migratory songbird. Ecography 31, 592–600.

Buckland, S.T., Magurran, A.E., Green, R.E., Fewster, R.M., 2005. Monitoring change in biodiversity through composite indices. Philos. Trans. R. Soc. B: Biol. Sci. 360, 243–254.

Canterbury, G.E., Martin, T.E., Petit, D.R., Petit, L.J., Bradford, D.F., 2000. Bird communities and habitat as ecological indicators of forest condition in regional monitoring. Conserv. Biol. 14, 544–558.

Carignan, V., Villard, M.-A., 2002. Selecting indicator species to monitor ecological integrity: a review. Environ. Monit. Assess. 78, 45–61.

Carleton, T.J., Maclennan, P., 1994. Woody vegetation responses to fire versus clearcutting logging: a comparative survey in the central Canadian boreal forest. Ecoscience 1, 141–152.

Cimprich, D.A., Moore, F.R., Guilfoyle, M.P., 2000. Red-eyed Vireo (*Vireo olivaceus*). In: Poole, A. (Ed.), The Birds of North America Online. Cornell Lab of Ornithology, Ithaca, Retrieved from the Birds of North America Online.

Crins, W.J., Gray, P.A., Uhlig, P.W., Wester, M.C., 2009. The Ecosystems of Ontario, Part 1: Ecozones and Ecoregions.

Croonquist, M., Brooks, R., 1991. Use of avian and mammalian guilds as indicators of cumulative impacts in riparian-wetland areas. Environ. Manag. 15, 701–714.

Drever, C.R., Peterson, G., Messier, C., Bergeron, Y., Flannigan, M., 2006. Can forest management based on natural disturbances maintain ecological resilience? Can. J. For. Res. 36, 2285–2299.

Hunter, M.L., 1993. Natural fire regimes as spatial models for managing boreal forests. Biol. Conserv. 65, 115–120.

Hunter, M.L., 1999. Maintaining Biodiversity in Forest Ecosystems. Cambridge University Press.

Johnson, E.A., 1996. Fire and Vegetation Dynamics: Studies from the North American Boreal Forest. Cambridge University Press.

Karr, J.R., 1981. Assessment of biotic integrity using fish communities. Fisheries 6, 21–27.

Klenk, N.L., Bull, G.Q., MacLellan, J.I., 2009. The emulation of natural disturbance (END) management approach in Canadian forestry: a critical evaluation. For. Chron. 85, 440–445.

Lamb, E.G., Bayne, E., Holloway, G., Schieck, J., Boutin, S., Herbers, J., Haughland, D.L., 2009. Indices for monitoring biodiversity change: are some more effective than others? Ecol. Indic. 9, 432–444.

Lambeck, R.J., 1997. Focal species: a multi-species umbrella for nature conservation. Conserv. Biol. 11, 849–856.

Lawrence, L.K., 1953. Nesting life and behavior of the Red-eyed Vireo. Can. Field-Nat. 67, 47–87.

MacKenzie, D.I., Bailey, L.L., 2004. Assessing the fit of site-occupancy models. J. Agric. Biol. Environ. Stat. 9, 300–318.

MacKenzie, D.I., Nichols, J.D., Royle, J.A., Pollock, K.H., Bailey, L.L., Hines, J.E., 2006. Occupancy Estimation and Modeling: Inferring Patterns and Dynamics of Species Occurrence. Academic Press.

Naylor, B.J., Mackereth, R.W., Kreutzweiser, D.P., Sibley, P.K., 2012. Merging END concepts with protection of fish habitat and water quality in new direction for riparian forests in Ontario: a case study of science guiding policy and practice. Freshw. Sci. 31, 248–257.

O'Connell, T.J., Jackson, L.E., Brooks, R.P., 2000. Bird guilds as indicators of ecological condition in the central Appalachians. Ecol. Appl. 10, 1706–1721.

Ontario Ministry of Natural Resources, 2010. Forest Management Guide for Conserving Biodiversity at the Stand and Site Scales. Queen's Printer for Ontario. Toronto. Ontario Ministry of Natural Resources, Toronto.

Ontario Ministry of Natural Resources, 2014. Forest Management Guide for Boreal Landscapes. Queen's Printer for Ontario. Toronto. Ontario Ministry of Natural Resources, Toronto.

Parrish, J.D., Braun, D.P., Unnasch, R.S., 2003. Are we conserving what we say we are? Measuring ecological integrity within protected areas. Bioscience 53, 851–860.

Payandeh, B., 1991. Plonski's (metric) yield tables formulated. For. Chron. 67, 545–546.

Perera, A.H., Buse, L.J., 2014. Ecology of Wildfire Residuals in Boreal Forests. Wiley-Blackwel.

Perera, A.H., Buse, L.J., Weber, M.G., 2007. Emulating Natural Forest Landscape Disturbances: Concepts and Applications. Columbia University Press.

Perera, A.H., Ouellette, M., WenBin, C., Drescher, M., Boychuk, D., 2008. BFOLDS 1.0: a spatial simulation model for exploring large scale fire regimes and succession in boreal forest landscapes. Ontario Forest Research Institute.

Poulin, J.-F., D'Astous, E., Villard, M.-A., Hejl, S.J., Newlon, K.R., Mcfadzen, M.E., Young, J.S., Ghalambor, C.K., 2013. Brown Creeper (*Certhia americana*). In: Poole, A. (Ed.), The Birds of North America Online. Cornell Lab of Ornithology, Ithaca, Retrieved from the Birds of North America Online.

Poulin, J.F., Villard, M.A., 2011. Edge effect and matrix influence on the nest survival of an old forest specialist, the Brown Creeper (*Certhia americana*). Landsc. Ecol. 26, 911–922.

Rempel, R.S., 2007. Selecting focal songbird species for biodiversity conservation assessment: response to forest cover amount and configuration. Avian Conserv. Ecol. 2, AN Article 6.

Rempel, R.S., Elkie, P.C., Rodgers, A.R., Gluck, M.J., 1997. Timber-management and natural-disturbance effects on moose habitat: landscape evaluation. J. Wildl. Manag. 61, 517–524.

Rempel, R.S., Francis, C.M., Robinson, J.N., Campbell, M., 2013. Comparison of audio recording system performance for detecting and monitoring songbirds. J. Field Ornithol. 84, 86–97.

Rowe, J.S., Scotter, G.W., 1973. Fire in the boreal forest. Quat. Res. 3, 444–464.

Simberloff, D., 2001. Management of boreal forest biodiversity – a view from the outside. Scand. J. For. Res. 16, 105–118.

Szaro, R.C., 1986. Guild management: an evaluation of avian guilds as a predictive tool. Environ. Manag. 10, 681–688.

Thompson, I., 2006. Monitoring of biodiversity indicators in boreal forests: a need for improved focus. Environ. Monit. Assess. 121, 263–273.

Thompson, I., Harestad, A., 2004. The ecological and genetic basis for emulating natural disturbance in forest management: Theory guiding practice.

Venier, L., Holmes, S., Williams, J.M., 2011. Bay-breasted Warbler (*Setophaga castanea*). In: Poole, A. (Ed.), The Birds of North America Online. Cornell Lab of Ornithology, Ithaca, Retrieved from the Birds of North America Online.

Venier, L.A., Pearce, J.L., Fillman, D.R., McNicol, D.K., Welsh, D.A., 2009. Effects of spruce budworm (*Christoneura fumiferana*) (Clem.)) outbreaks on Boreal mixed-wood bird communities. Avian Conserv. Ecol. 4, AN article 3.

Villard, M.A., Schmiegelow, F.K.A., Trzcinski, M.K., 2007. Short-term response of forest birds to experimental clearcut edges. Auk 124, 828–840.

Wiens, J.A., Hayward, G.D., Holthausen, R.S., Wisdom, M.J., 2008. Using surrogate species and groups for conservation planning and management. Bioscience 58, 241–252.

Wotton, B.M., Alexander, M.E., Taylor, S.W., 2009. Updates and revisions to the 1992 Canadian forest fire behavior prediction system. Great Lakes Forestry Centre.