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RESEARCH ARTICLE

Impacts of mountaintop mining on terrestrial ecosystem integrity: identifying landscape thresholds for avian species in the central Appalachians, United States

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Abstract
Context Mountaintop removal/valley fill (MTR/VF) mining in the central Appalachians is a major driver of landscape change within terrestrial ecosystems.
Objective We quantified avian community and individual taxa thresholds in response to changing landscapes from MTR/VF using a Threshold Indicator Taxa Analysis approach.
Methods We conducted 50-m fixed radius avian surveys (n = 707) within forest adjacent to mine lands in 2012–2013 and obtained data for additional surveys (n = 905) sampled using comparable methods during 2008–2013. We quantified positive and negative community, habitat guild, and species thresholds in abundance and occurrence for each of five landscape metrics within a 1-km radius of each survey point.
Results Reclaimed mine-dominated landscapes (less forest and more grassland/shrubland cover) elicited more negative (57 %) than positive (39 %) species responses. Negative thresholds for each landscape metric generally occurred at lower values than positive thresholds, thus negatively responding species were detrimentally affected before positively responding species benefitted. Forest interior birds generally responded negatively to landscape metric thresholds, interior edge species responses were mixed, and early successional birds responded positively. The forest interior guild declined most at 4 % forest loss, while the shrubland guild increased greatest after 52 % loss. Based on random forest importance ranks, total

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amount of landscape grassland/shrubland had the most influence, although this varied by guild.
Conclusions Because of little overlap in habitat requirements, managing landscapes simultaneously to maximally benefit both guilds may not be possible. Our avian thresholds identify single community management targets accounting for scarce species. Guild or individual species thresholds allow for species-specific management.

Keywords Forest fragmentation and management · TITAN · Avian landscape response · Surface mining · Community thresholds · Species-specific change points

Introduction

The Central Appalachian landscape is naturally dominated by interior forest with nationally and globally recognized regional biodiversity (Ricketts et al. 1999). Since the 1990s, however, the expansion of mountaintop removal/valley fill (MTR/VF) mining in the region has been a major driver of landscape change (Townsend et al. 2009) converting over 1.1 million ha of forests to surface mines and burying more than 2,000 km of stream channels beneath overburden (Bernhardt and Palmer 2011). The conversion of this intact, forested ecosystem to barren, grassland, or other land-cover types may shift the native faunal assemblage to that of an edge-dwelling and grassland fauna (USEPA (Environmental Protection Agency) 2011).

Current assessments of mountaintop mining do not account for potential long-term effects of landscape-scale changes associated with MTR/VF mining (USEPA (Environmental Protection Agency) 2011). The effects of broad scale land-cover changes on regional biodiversity can persist for decades despite preservation of forested areas and reforestation of riparian zones (USEPA (Environmental Protection Agency) 2011). To date, environmental assessments of mountaintop mining have not been comprehensive because they have focused mainly on the Clean Water Act regulatory endpoints of aquatic impacts (USEPA (Environmental Protection Agency) 2011) and have overlooked terrestrial impacts (Wickham et al. 2013). Research has clearly documented that decreasing forest cover negatively impacts water quantity, water quality, and other factors of aquatic habitat (Brabec 2002; Ernst et al. 2004; USEPA (Environmental Protection Agency) 2011; Bernhardt and Palmer 2011).

MTR/VF mining can profoundly change terrestrial ecosystems by altering soil, microclimate, and elevation; thereby reducing overall topological complexity. This in turn disrupts the complex processes that influence regional biodiversity (Maxwell and Strager 2013; Wickham et al. 2013). Fragmentation and loss of interior forest is 1.5-5 times greater than actual forest loss in mountaintop mining landscapes (Wickham et al. 2007; Wickham et al. 2013) and introduces edge effects (Harper et al. 2005) which have been little studied with respect to mountaintop mining (Weakland and Wood 2005; Wood et al. 2006; Wood and Williams 2013). Forests on reclaimed mines have reduced carbon sequestration (Amichev et al. 2008; Campbell et al. 2012) and productivity (Groninger et al. 2006). Soil loss and compaction of reclaimed mining sites limits successful forest reestablishment and reduces the likelihood that succession will recreate the original forest community (Acton et al. 2011; Zipper et al. 2011). These changes between forest and reclaimed habitats ultimately result in shifts in the community composition of songbirds, raptors, and herpetofauna (Wood et al. 2001; USEPA (Environmental Protection Agency) 2003).

The biodiversity of breeding birds is an effective indicator of biotic integrity (Glennon and Porter 2005; O'Connell et al. 2007) and can be used to quantify the impacts of mountaintop mining on terrestrial ecosystems. Most avian research conducted in MTR/VF mining areas has focused on characterizing bird communities' responses to creation of post-mining early successional grasslands and scrub-shrub habitats. These studies emphasize successful occupation by grassland birds (Monroe and Ritchison 2005; Stauffer et al. 2011), golden-winged warblers (Bulluck and Buehler 2008), and upland game birds (Gregg et al. 2001; Beckerle 2004). Studies of the effects of MTR/VF land-cover conversion from forests to early successional habitats on forest-dwelling songbirds, other than cerulean warblers (Weakland and Wood 2005; Buehler et al. 2006; Wood et al. 2006), are lacking, especially with respect to quality of remaining forests near reclaimed mines. There is also a dearth of research on landscape-level effects and cumulative impacts of multiple mining sites (Buehler and Percy 2012).

Our objective was to quantify avian community, habitat guild, and individual taxa thresholds (or change points) within remaining forests to evaluate response to a changing landscape resulting from mountaintop mining. We used Threshold Indicator Taxa ANalysis (TITAN) (Baker and King 2010), which identifies abrupt changes in occurrence and relative abundance of taxa along an environmental gradient, and has been rarely applied to bird community analyses (Suarez-Rubio et al. 2013). Our goal was to quantify landscape-level influences on bird community composition to inform management decisions in regards to mountaintop mining and terrestrial ecosystems.

Methods

Study area

The MTR/VF region (Fig. 1) primarily encompasses southern West Virginia and eastern Kentucky with smaller portions in Tennessee and western Virginia. It lies within the broader mixed mesophytic forest biogeographic region (Dyer 2006) with a mild climate and precipitation throughout all seasons. The MTR/VF region in West Virginia is characterized by elevation ranges between 171 and 1,080 m with an average elevation of 487 m. This area contains a high degree of elevation complexity and ruggedness measured by a 3.57 roughness index (Blaszczynski 1997; Riley et al. 1999) and a surface relief ratio (Pike and Wilson 1971) of 0.492. Metrics were computed using the Geomorphometry and Gradient Metrics Toolbox (Evans et al. 2014).

Land-cover classes and extent of each within the MTR/VF region of West Virginia and Kentucky include approximately 85 % forest, 7 % grassland/shrubland/pasture/hay, 2 % developed, and 5 % mineland. The mineland land-cover class includes reclaimed grasslands/shrublands, mine barren, valley fills, slurry impoundments, and mine facilities. Forest communities include primarily mixed mesophytic forests with some cove hardwoods, Appalachian oaks, northern hardwoods, and minimal amounts of floodplain forests, oak-pine forests, and pine forests (USEPA (Environmental Protection Agency) 2003).

Land-cover data and deriving landscape metrics

We used land-cover data for West Virginia and Kentucky developed using the methods of Maxwell et al. (2014a, b). Aerial photographs were classified into nine groups (forest, mining barren, mining grass/shrub, mining facilities, other barren, other grass/shrub, valley fill, slurry pond, and open water) a priori. The classification approach focused on producing specific classes using high resolution, temporally appropriate imagery with object-based image analysis and GIS overlay. Integral to the classification approach was the creation of training data sites that represented the cover classes of interest and their variability. Due to access constraints at the permitted mine sites it was not possible to collect training data on the ground. Instead, a manual aerial photograph interpretation process was performed with the orthophotography as reference. Within the study area a total of 752 examples of forest, 408 examples of grass, and 309 examples of barren were collected for the final classification. Land cover data were extracted from the true and color infrared National Agriculture Imagery Program imagery as forested, grasslands, or barren using the manually interpreted training data and the object-based image analysis program of Feature Analyst 5 by Overwatch (VLS, 2004). Feature Analyst uses machine-learning algorithms and techniques as object recognition to automate feature-recognition from imagery (VLS, 2004). Two iterations between training data and classification were performed in order to examine the results of the first attempt and to identify troublesome areas and enhance the results. The overall accuracy of the land-cover map was 92 %.

We calculated metrics within a 1 km radius plot around each count location using FRAGSTATS (McGarigal et al. 2002) including the percent cover by land-cover class, the percent core forest (>100 m from a non-forest edge; the 1-km radius boundary edge is treated as background, not edge, and should minimally bias this metric as the boundary is primarily forest), and the edge density between forest and non-forest habitat. A 1 km radius (314.2 ha) was chosen as it surpasses the average mine permit size in both West Virginia (66 ha) and Kentucky (147.6 ha) and approaches the size of some watersheds. Although the 1-km radius plots overlap for some points, the avian survey data are independent and the landscape metrics relate to avian data at each point. We derived a

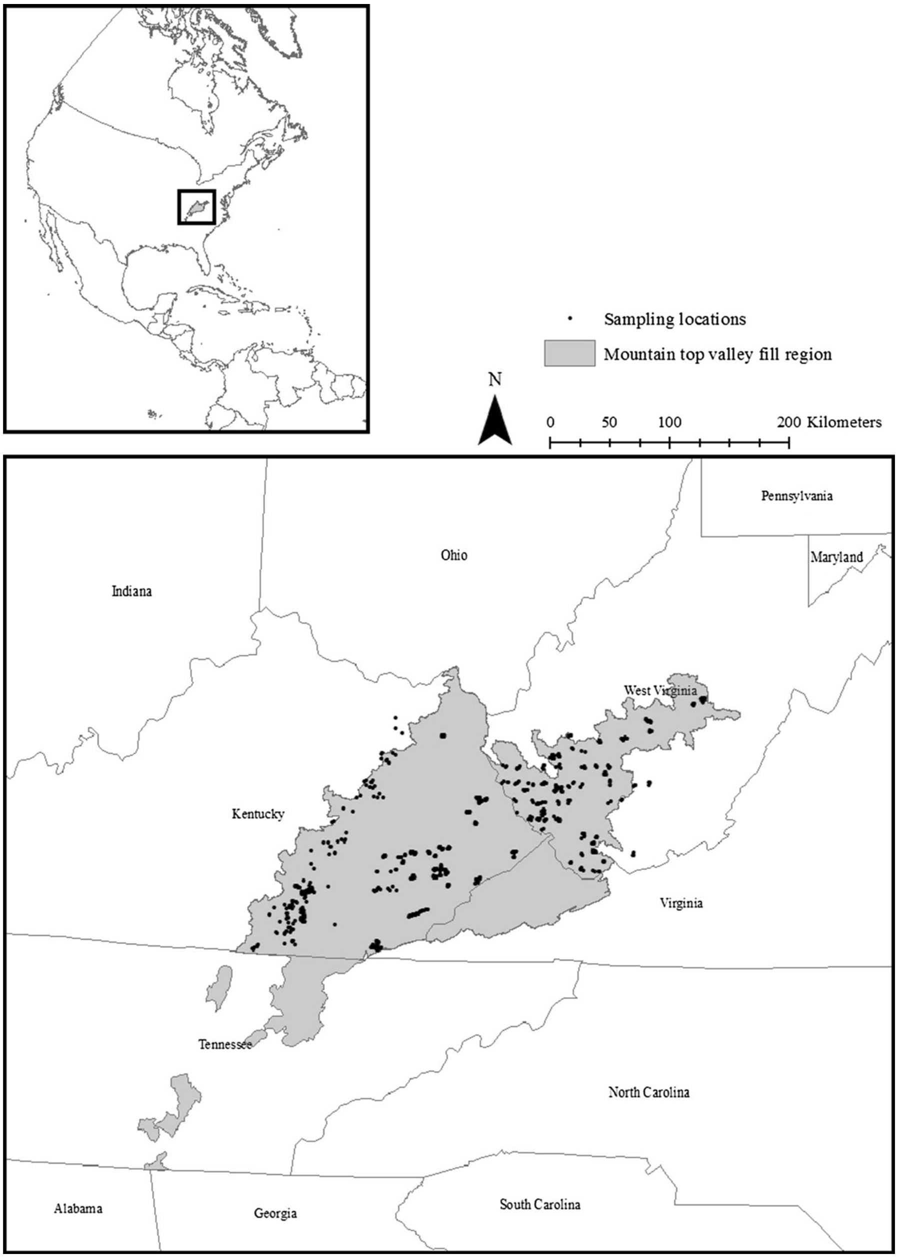

Fig. 1 Geographic distribution of all 1,612 bird survey sampling locations within MTR/VF region sampled 2008–2013

subset of five non-mutually exclusive metrics (Table 2) that we thought might be important to avian species. In our heterogeneous landscape, an increase in percent cover of one land-cover type potentially means a decrease in one or more other land-cover types, creating some dependence in the relationship between land-cover types. We subtracted percent forest and percent core forest from 100 % to determine forest and core forest loss, so that an increase in values for all five metrics corresponded to a shift away from a forest-dominated landscape to a more mine-dominated one.

Avian surveys

We used four sets of survey data for our analysis (Table 1). In 2012–2013, we conducted avian surveys (n = 707) adjacent to minelands across a representative portion of the MTR/VF region in southern West Virginia (353 sampling points in 2012) and eastern

Table 1 Summary of point count sources included in analyses

Dataset name	Source	# Points	Years	Land-cover layer
WVDNR	West Virginia Division of Natural Resources	436	2008–10[a]	2010
WVDNR	West Virginia University: Mizel	75	2010	2010
WVDNR	West Virginia University: Cerulean Project	83	2010	2010
WV Atlas	West Virginia Breeding Bird Atlas	139	2012	2012
WVU	West Virginia University: Becker, WV	353	2012	2012
WVU	West Virginia University: Becker, KY	354	2013	2013
KY	USFS Daniel Boone National Forest	90	2013	2013
KY	Grayson Cerulean Project	2	2013	2013
KY	Kentucky Department of Fish and Wildlife	80	2012–13[a]	2013

[a] For points sampled in multiple years, only the count closest in time to the available land-cover layer was used in analyses

Kentucky (354 points in 2013), referred to hereafter as WVU data. Surveying locations were selected to encompass the broadest geographic coverage of the MTR/VF region given limitations in access and to ensure the overall forest type remained largely hardwood, the predominant forest type, to minimize bias in bird community composition due to changing forest type. Site selection (Fig. S1 in electronic supplementary material) within minelands prioritized sites with interior patches of forest followed by sites with peninsulas of forest extending within the minelands and then forest patches adjacent to minelands along ridges, slopes, and valleys (Fig. S2) to ensure sampling of bird response to habitats surrounded by the greatest percentage of mineland cover. Sample points were placed in forest at least 50 m from the forest edge and were at least 250 m apart to ensure statistical independence (Ralph et al. 1993).

We obtained data for additional survey points (n = 905) that were sampled during 2008–2013 in the same geographic region. We aggregated points into three additional datasets: WVDNR data, WV Atlas data, and KY data. We reviewed all points using GIS and aerial photographs to ensure that they were located in forest, and where multiple years of data were available, used only the count closest in time to the land-cover layer (Table 1). Points again were >250 m apart to ensure independence of sampling points.

Point count survey methods

For the WVU data, we counted breeding birds using 50-m fixed-radius plots (Ralph et al. 1993; Bibby et al. 2000) during 16 May–28 June 2012 in WV and during 21 May–30 June 2013 in KY. Each sample point was surveyed once in either 2012 or 2013 by one of two observers proficient in bird identification and distance estimation. Counts began at 0600 h (EST) and ended no later than 1000 h (EST) on mornings with suitable weather conditions (i.e., no rain, little wind). We recorded by species all individuals heard or observed within a 10-minute time span, the type of detection (song, call, visual, or fly-over), and sex if possible. Each individual was also categorized by the time of first detection (0–2, >2–3, >3–4, >4–5, >5–6, >6–7, >7–10 min) and distance from observer (0–25, >25–50, >50–100, and >100 m). Recently fledged young and flyovers were noted but excluded from analyses as were species not well detected by point counts. If a bird could not be identified to species, the observer attempted to locate and identify the individual after completing the count.

All additionally obtained survey data (WVDNR, WV Atlas, KY data) followed a similar protocol as the WVU surveys and used multiple observers who were experienced in conducting point count surveys. Distance classes within these datasets were variable but all had the 0–50 m class in common, so analyses were completed on data only from this class. Time intervals were not available for all of the WVDNR data surveys while WV Atlas and KY data were collapsed into the three time intervals consistent across all data sets (0–3, >3–5, and >5–10 min).

Assessing detection probability

We performed three removal model analyses (Farnsworth et al. 2002) as a diagnostic tool to examine differences in detectability among observers and to ensure consistency of the data from different sources. We evaluated differences between 1) the two WVU observers, 2) the WVU most experienced observer (EO) and WV Atlas data combined observers, and 3) WVU EO and Kentucky data combined observers. Because time intervals were not available for the WVDNR data, we could not complete removal models for this dataset.

We used birds detected aurally within the 50-m radius using seven time intervals (0–2, >2–3, >3–4, >4–5, >5–6, >6–7, and >7–10 min) for the first analysis and three time intervals (0–3, >3–5, and >5–10 min) for the second and third. For each of the three analyses, we tested two different models, each with and without observer differences, for each species that had at least 10 detections. Based on past experience, 10 detections is the minimum number required to attempt removal models. The basic model assumed an equal detectability for all individual birds throughout the count, while the heterogeneity model assumed that certain subsets of individuals were always detected during the first time interval. We selected the best model based on the lowest AIC value and Chi square likelihood ratio tests using program SURVIV (White 1992).

Analyzing thresholds

We analyzed avian count data with TITAN (Baker and King 2010) in R (R Development Core Team 2013) using the TITAN package. TITAN combines change point (King and Richardson 2003; Qian et al. 2003) and indicator species analyses (Dufrene and Legendre 1997) to calculate community thresholds based on relative abundance and frequency of occurrence, with the advantage that it incorporates taxa detected at low frequencies ($n \geq 5$ detections). TITAN identifies midpoints between unique values of environmental metrics as candidate change points and tests them using change-point analysis to maximize a deviance reduction statistic comparing with-in and between group dissimilarity based on a user defined ecological distance metric. TITAN replaces the aggregate, community dissimilarity response, typical of change-point analysis, with a taxon-specific indicator value based on indicator species analysis. Indicator values measure strength of association between a taxon and an environmental gradient measured as the product of cross-group relative abundance (proportion of abundance among all sample units belonging to a group) and within-group occurrence frequency (proportion of sample units in that group with a positive abundance value). Frequency of occurrence within each group is used to weight a taxon's relative abundance by how consistently it is observed in each group such that a large abundance within one sample group results in a greater score only if the taxon also occurs with great regularity in that same group. Indicator value scores are calculated for each taxon above and below each candidate change point such that the relative magnitude between each group defines classification as a positive or negative response and the greater the difference in association created by a split, the greater the individual value score. TITAN uses 250 random permutations to calculate p (frequency of obtaining random indicator value > than observe indicator values) and the mean and standard deviation of random indicator scores. Indicator score by taxon are then standardized into Z scores using the permutated mean and standard deviations and then summed by response-group for each candidate change point such that the maxima for each response-group is the community level change point. Finally, bootstraps with replacement estimate change point uncertainty.

A positive response threshold (Z+) occurs for change-points after which the community or species abundance/occurrence increases as the environmental gradient increases. A negative response threshold (Z−) occurs when the community or species abundance/occurrence declines after the change point. We calculated thresholds separately for each of our individual landscape metrics for the overall avian community, for the three habitat guilds (FI = forest interior, IE = interior edge, and ES = early successional based on Whitcomb et al. (1981), Ehrlich et al. (1988), and observations from previous research), and for individual species. Because we purposely placed sample points within forest habitats, the early successional guild is represented primarily by shrubland rather than grassland species.

We excluded birds detected outside of the 50-m radius to minimize variation in detectability among observers, data sources, and site differences in topography and habitat. This distance also ensured that birds included in the analysis were associated with forest at the sampling points. Species were excluded from the threshold analysis if they did not have a minimum of five detections (Baker and King 2010). We used 250 bootstraps, sampling with replacement, to estimate empirical quantiles of the change point distribution derived from the bootstraps for community and taxon-specific change points. These quantiles should not be strictly interpreted as confidence limits due to the inclusion of taxa with low frequencies (Manly 1997). Narrow intervals between upper and lower change-point quantiles represent sharp, nonlinear responses in taxon abundances, while broad quantile intervals represent linear or more gradual responses (Baker and King 2010).

We then identified which species had significant responses and the direction of the response (positive or negative). Similar to Baker and King (2010), we considered individual species responses to be significant if visual inspection of presence/absence spatial locations were not highly aggregated and the following three criteria were met: 1) the p value was <0.05 based on its Z score derived from its standardized indicator value scores, 2) the purity (percent of bootstraps with the same positive or negative response classification) was greater than 95 %, and 3) 95 % of the bootstraps had a p value <0.05. We calculated overall community and guild thresholds using all species with >5 detections, while we calculated individual species' thresholds using only significant species.

Ranking metric importance

We used a random forest modeling technique (Breiman 2001) to test the relative importance of each landscape metric in the avian community relative abundance responses. Random forests are a learning ensemble consisting of multiple un-pruned decision (classification and regression) trees with a randomized selection of features at each split. The error is calculated before and after the permutation of all trees with variables producing larger differences being ranked as more important. We created random forests using 5,000 decision trees generated with the five landscape metrics for each avian species that was determined to be significant in TITAN analyses. The importance of each metric was then ranked (five being the most important and one being the least important) for each species based on mean squared error (MSE). We then determined the average importance of each landscape metric by species, for all species combined, by habitat guild, and by positive or negative response.

Results

Landscape metrics

The landscape metric distributions (Fig. 2) indicate that we were able to sample a large range of conditions within our study region. The range of values for each landscape metric was: forest (0–99 %), core forest (9.7–100 %), total mineland (0–83.2 %), total grassland/shrubland (0–62.9 %), and contrast-weighted edge density (0–444.9 m/ha). In general, the MTR/VF landscape includes high amounts of forest cover with lesser amounts of mineland and shrubland. The more extreme values (high mineland and grassland/shrubland and low amounts of forest) are driven by mountaintop mining activities. These distributions are important in interpreting the avian thresholds as the threshold value is constrained within the maximum and minimum values available on the landscape for any given metric.

Removal models

We detected observer differences within the WVU data for only two of 20 species that could be tested (10 %), however, the difference in detectability was negligible for scarlet tanager (SCTA, Table 3) with ΔAIC <2 and a non-significant Chi square likelihood ratio test. The three species with detection probability differences between the WVU data and the WV Atlas data (17.6 % of species, 3 of 17) had similar final detection probabilities, ΔAIC values <2 and non-significant Chi square likelihood ratio tests. The largest differences occurred between the WVU and KY datasets (31.8 %, 7 of 22) with detectability typically being lower for the KY data but differences again were minimal based on ΔAIC for 3 of 7 species. In general, removal models identified some differences among observers and datasets but differences were neither systematic nor large for most species (Table 3). Thus, we retained all bird species and combined all datasets for the TITAN threshold analyses.

Threshold analysis

We detected 79 species (8,057 detections) within 50 m of our 1,612 sample points (Table S1). These species were reliably sampled with point counts and met our criteria for inclusion in threshold analyses.

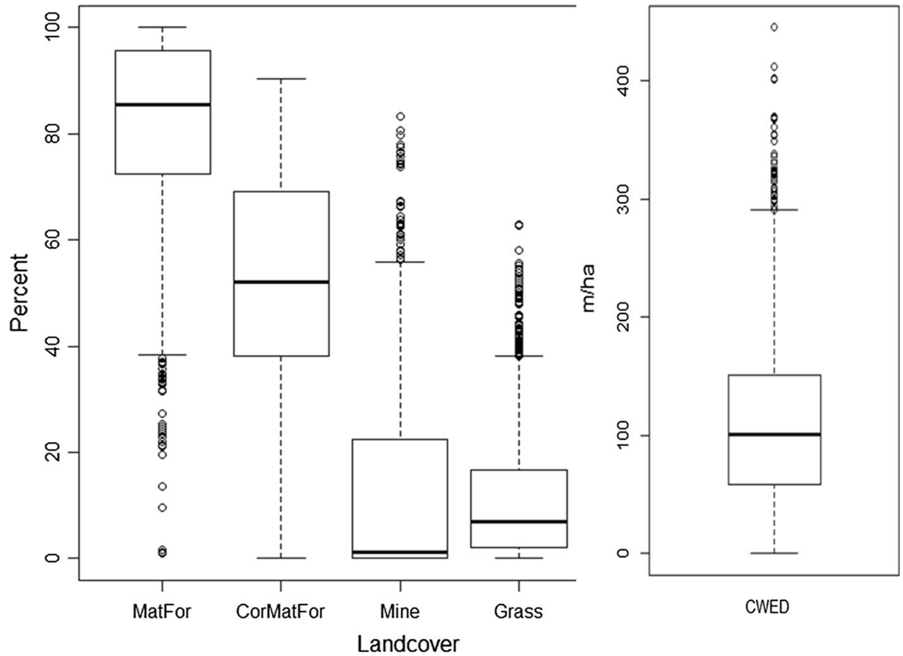

Fig. 2 Boxplots of landscape metrics within a 1 km radius of the survey centers based on the combined data sets (n = 1,612 points) from 2008 to 2013. Mineland and total grassland/shrubland cover both include cover categorized as mine grassland/shrublands so should not be added together as an estimate of early successional habitats. The boxes for forest and core forest represent the distribution of forest present (not forest loss) and are not subtracted from 100 %. The *box* represents the 1st and 3rd quartiles, the *bold line* is the median, the *tails* represent 1.5 times the interquartile range, and the *circles* represent values greater than 1.5 times the interquartile range. Metric abbreviations are defined in Table 2

The overall avian community (all species combined) had a negative response at a very low threshold for mineland and total grassland/shrubland land-cover classes (<2 %; Table 4). The negative response threshold to forest loss occurred at 11 % and to core forest loss at 35 %. The magnitude of decline in relative abundance after surpassing thresholds was >42 % for all metrics except total grassland/shrubland (14 %; Table 5). In contrast, thresholds for positive responses were reached after greater amounts of landscape change had occurred (Table 4). For example, the positive response threshold for forest loss was 51 % and for grassland/shrubland was 48 %. Percent increase in relative abundance after surpassing thresholds was >65 % (maximum 292 % for mineland; Table 5). Generally for each landscape metric, the negative response occurred at much lower threshold values than the positive response, thus negatively responding species were detrimentally affected before positively responding species benefitted.

The overall avian community thresholds incorporate varying responses of individual species with many different habitat requirements, thus the change point quantile intervals are very broad (Table 4). For example, negative responses to forest loss could occur at 4–83 % of the landscape. After we partitioned the avian community into habitat guilds, the change point quantile intervals often became narrower, indicating confidence in the existence of a threshold. For example, we found much narrower Z- intervals for response of interior edge species to forest loss and core forest loss and for forest interior species response to core forest loss and edge density. Many species that had significant responses had even tighter intervals (Fig. S3), suggesting that the broad community intervals result from inclusion of the non-significant species into the community threshold as well as variability in the thresholds for significant individual species.

Thresholds at which negative responses occurred varied among habitat guilds in a consistent pattern. For each landscape metric, the forest interior guild had the lowest threshold (Table 4), suggesting that this guild was most sensitive to small amounts of landscape

Table 3 Detection probabilities (P-hat) based on time removal models for species that had observer differences in the best fitting of the four models tested. Species codes are defined in Table S1

Comparison (# species tested)	Species	P-hat	SE
WVU Observer 1 versus 2 (N = 20)	DOWO	0.65	0.55
		0.99	0.12
	SCTA	0.98	0.01
		1.00	0.00
WVU versus WV Atlas (N = 17)	OVEN	0.99	0.10
		1.00	0.04
	REVI	0.98	0.10
		1.00	0.04
	SCTA	0.99	0.11
		1.00	0.04
WVU versus KY (N = 22)	ACFL	0.99	0.08
		0.91	0.26
	BAWW	0.96	0.15
		0.75	0.35
	BGGN	0.96	0.16
		0.80	0.34
	HOWA	0.90	0.10
		0.79	0.14
	INBU	1.00	0.05
		0.91	0.22
	REVI	0.32	1.33
		0.68	0.26
	SCTA	0.98	0.11
		0.86	0.27

change. The interior edge guild had intermediate threshold values, while the early successional guild had highest thresholds. Forest interior birds did have greater threshold uncertainty in response to forest loss (Table 4) and broader bootstrap quantiles for significant individual species (Fig. S3) indicating a weaker non-linear response than interior edge birds. Forest interior birds declined in abundance by between 46 and 57 % after surpassing thresholds depending on the metric, while the interior edge guild declined more moderately (36–46 %; Table 5). In comparing bird response to changes in forest structure relative to avoidance of mining habitats, forest interior and interior edge birds declined in equal or greater percentages for forest loss relative to mineland and grassland/shrubland, indicating that response to forest change is at least equivalent to the avoidance of the early successional habitats created by mining (Table 5).

For thresholds with positive responses from habitat guilds, there was no consistent pattern across all landscape metrics and the change point quantile distributions were generally broad (Table 4), suggesting that positive responses were more gradual. For forest loss and core forest loss, the negative threshold occurred for smaller amounts of landscape change than positive thresholds for each guild. Percent increase in relative abundance relative to thresholds varied greatly in the interior edge guild, from 13 % for forest loss to 344 % for total grassland/shrubland, but generally was greater for the early successional guild (>129 % up to 614%; Table 5).

Of all individual species that had significant responses, more responded negatively (28 species, 57 %) to landscape change than positively (19 species, 39 %; Table 6). The forest interior guild had almost exclusively negative threshold responses (Negative: 12 species; Positive: 1 species) with only the Kentucky warbler (KEWA) responding positively to two metrics (Table 6, Fig. S3). The interior edge guild had varied responses, although more species had negative responses (Negative: 11 species; Positive: 8 species). The early successional guild had primarily positive responses (Negative: 5 species; Positive: 10 species; Table 6) particularly to increased edge density and forest or core forest loss. The negative thresholds of the early successional guild are primarily driven by four species [eastern towhee (EATO), indigo bunting (INBU), mourning dove (MODO), and prairie warbler (PRAW)], providing support that some forest habitat could be beneficial for at least certain members of this guild.

Random forest

Based on the MSE ranked importance values, total grassland/shrubland ranked highest for the overall bird community and interior edge guild, core forest loss ranked highest for the forest interior guild, and non-forest mineland ranked highest for the early successional guild (Fig. 3). Total grassland/shrubland ranked a close second in importance when not ranked most important. Non-forest mineland was ranked lowest except for the early successional guild. Core forest was typically ranked slightly higher than forest. The

Table 4 Negative response (Z−) and positive response (Z+) TITAN thresholds (CP = change point) and 5–95 % change point quantiles based on the bootstrap change point distribution for all species combined and by habitat guild

Metric		Combined		Forest interior		Interior edge		Early succession	
		CP	5–95 % Quantiles	CP	5–95 % Quantiles	CP	5–95 % Quantiles	CP	5–95 % Quantiles
For	Z−	10.9	4.1–83.3	4.4	2.0–76.8	11.1	4.4–12.0	18.4	11.0–79.6
	Z+	51.3	0.0–83.3	49.9	0.1–79.6	16.1	7.7–83.3	52.0	0.0–79.6
CorFor	Z−	35.1	23.2–97.9	23.0	19.1–36.1	43.3	32.4–44.9	54.0	40.0–98.6
	Z+	87.6	9.7–97.9	93.6	10.7–97.3	89.7	51.7–98.6	88.1	9.7–97.5
CWED	Z−	50.8	24.8–369.2	22.9	21.2–54.1	50.9	29.1–86.7	367.9	68.2–369.2
	Z+	280.5	0.0–310.2	3.7	2.9–369.2	139.5	115.8–304.1	279.8	0.0–333.2
Mine	Z−	1.6	0.5–77.0	0.0	0.0–4.4	6.6	0.3–7.9	6.7	1.7–77.0
	Z+	77.0	0.0–77.0	49.7	0.0–58.7	2.6	1.5–77.0	77.0	0.0–77.0
Grass	Z−	0.2	0.1–54.8	0.2	0.0–0.7	9.1	0.3–9.6	14.0	10.6–54.8
	Z+	47.9	0.0–51.6	0.0	0.0–45.6	0.3	0.1–49.4	46.9	0.0–51.6

The change point indicates the threshold value of the metric for which the maximum combined indicator value score was determined based on all species detected including both significant and non-significant species responses. Narrow intervals between upper and lower change-point quantiles represent sharp, nonlinear responses in taxon abundances, while broad quantile intervals represent linear or more gradual responses (Baker and King 2010)

Table 5 Average mean, standard deviation, and percent change of bird relative abundance before and after surpassing metric thresholds of all species combined and by habitat guild

Metric		Combined					Forest interior					Interior edge					Early successional				
		Before		After			Before		After			Before		After			Before		After		
		$\bar{X}$	SD	$\bar{X}$	SD	%	$\bar{X}$	SD	$\bar{X}$	SD	%	$\bar{X}$	SD	$\bar{X}$	SD	%	$\bar{X}$	SD	$\bar{X}$	SD	%
For	Z−	4.7	3.7	2.3	1.8	−51	2.6	2.1	1.2	1.3	−54	1.9	2.0	1.0	1.0	−45	0.5	1.1	0.2	0.6	−61
	Z+	1.6	1.9	3.4	4.4	112	0.1	0.3	0.1	0.3	50	0.9	1.2	1.0	1.1	14	0.6	1.4	2.0	3.5	215
CorFor	Z−	4.6	3.6	2.7	2.4	−42	2.6	2.1	1.2	1.4	−55	2.0	2.0	1.2	1.2	−37	3.9	3.4	2.2	1.7	−45
	Z+	1.7	2.1	4.1	5.0	143	0.2	0.6	0.4	0.6	50	0.8	1.1	1.5	1.7	104	1.7	2.1	4.4	5.1	165
CWED	Z−	3.7	3.1	1.9	2.0	−47	2.6	2.2	1.1	1.4	−57	1.2	1.6	0.8	1.0	−36	0.2	0.6	0.0	0.0	−100
	Z+	2.6	2.5	4.4	5.0	65	0.0	0.0	0.4	0.7	n/a	1.4	1.5	1.6	1.4	18	0.8	1.6	2.1	4.2	152
Mine	Z−	5.2	3.9	2.7	2.1	−49	2.0	1.9	0.9	1.1	−54	2.3	2.3	1.5	1.2	−36	1.0	1.9	0.3	0.7	−72
	Z+	1.0	1.3	4.0	3.7	292	0.3	0.5	0.3	0.5	12	0.3	0.7	0.4	0.7	45	0.4	0.8	3.0	2.9	614
Grass	Z−	3.6	1.9	3.1	2.8	−15	2.6	1.4	1.4	1.6	−46	1.5	1.5	0.9	1.0	−39	0.5	1.0	0.2	0.5	−63
	Z+	1.8	2.2	3.1	2.9	70	0.0	0.0	0.1	0.3	n/a	0.3	0.7	1.1	1.3	344	0.7	1.6	1.6	2.3	129

highest ranked metric by guild was the same whether considering the average rank (Fig. 3) or the percent of species within the guild for which the metric was ranked highest (Table 7) except for forest interior species. For this guild, average rank identified core forest loss as highest rank whereas percent of species ranked grassland/shrubland as highest. However, some metrics of secondary overall importance for guilds were the top metric for individual species (e.g. non-forest mineland for all birds combined and edge density for interior edge), suggesting the metrics are of high importance to specific species but less so for the entire guild.

Ranked importance varied when the avian community was grouped based on the direction of the species threshold response, positive or negative. Total grassland/shrubland followed by core forest loss and edge density were the top three ranked metrics for

Table 6 TITAN thresholds for individual species with significant responses by landscape metric for each guild analyzed separately

Response	Species	For	CorFor	CWED	Mine	Grass
Forest interior						
Negative	BHVI	−3.1	−33.6	−48.6	−0.0	−2.5
	BLBW	−2.6	−28.5	−30.8	−2.5	−0.0
	BTBW	−9.5	−21.2	−22.9	−2.2	−0.3
	BTNW	−4.6	−23.3	−44.6	−0.0	−0.2
	CERW	−11.6	-	-	-	-
	EAWP	−3.0	−30.9	−50.3	−0.0	-
	LOWA	−16.5	−52.1	-	−0.0	−4.2
	OVEN	−10.8	−32.9	−56.9	−3.4	−3.4
	SCTA	−11.1	−33.8	−51.0	−1.6	−5.5
	SWTH	−0.6	−10.7	−5.1	-	−0.2
	WBNU	−27.0	−63.0	-	−0.3	−19.9
	WEWA	−4.5	−43.9	−51.0	−1.0	−3.8
Positive	KEWA	-	-	60.4	-	49.2
Interior edge						
Negative	AMRE	-	-	-	-	−9.4
	AMRO	-	-	-	−5.3	−0.0
	GCFL	-	-	-	−0.0	−2.3
	NOPA	−9.1	−42.4	-	−0.0	−7.8
	RBGR	−4.3	−30.8	−55.8	−0.0	−2.3
	RBWO	−8.9	−35.1	−71.8	−0.8	−3.7
	REVI	-	-	-	−6.0	-
	VEER	-	−13.8	-	−0.0	-
	YBCU	−5.1	−36.6	−86.5	−0.2	−6.7
	YTWA	−12.8	-	−87.4	-	−5.6
Positive	BGGN	-	-	-	-	7.2
	BHCO	64.8	96.2	141.5	66.8	2.1
	CACH	-	-	122.2	-	0.3
	CEDW	78.8	82.8	-	-	51.4
	DOWO	-	-	191.7	-	-
	SUTA	23.9	53.7	-	-	6.1
	TUTI	-	-	-	-	0.4
	YTVI	10.5	59.2	283.2	2.6	9.6
Mixed	HOWA	−10.4	-	-	-	0.1
Early succession						
Negative	EATO	−11.2	−54.3	-	−3.8	−12.2
	FISP	-	-	-	−9.9	-
	INBU	−18.5	-	-	−8.5	-
	MODO	-	-	-	−0.0	-
	PRAW	−7.1	−41.0	−71.8	−0.0	−7.1
Positive	AMGO	-	-	229.2	-	-
	BRTH	27.3	52.6	-	-	50.4
	CARW	38.4	69.7	15.6	24.0	13.6

Table 6 continued

Response	Species	For	CorFor	CWED	Mine	Grass
	COGR	-	84.3	256.9	-	-
	COYE	-	97.8	-	-	-
	CSWA	-	-	155.6	-	-
	NOCA	2.0	97.3	43.1	-	0.9
	RWBL	-	81.6	284.6	-	-
	SOSP	51.6	82.1	248.6	-	-
	YWAR	78.8	88.9	280.5	-	46.8
Mixed	EABL	-	-	102.5	−0.0	-
	GRCA	-	-	129.1	−3.2	-

Thresholds indicate the biggest before/after change in relative abundance. Plus sign indicates a positive response, minus sign a negative response. For graphical representation, see TITAN graphs in Fig. S3. Species codes are in Table S1

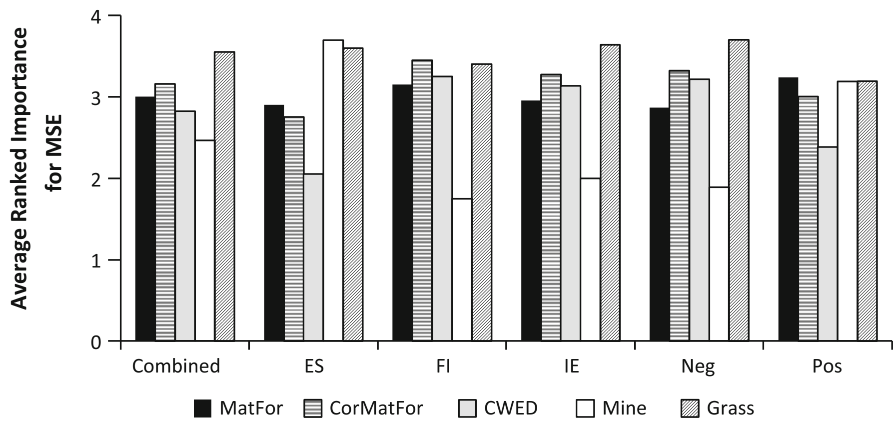

Fig. 3 Average random forest (n = 5,000) mean squared error (MSE) ranked importance values by landscape metric for all significant TITAN avian species for all species combined, by habitat guild (FI = forest interior, IE = interior edge, and ES = early successional), and by positive or negative response. Metrics were ranked from 5 (most important) to 1 (least important) for each species and then averaged

Table 7 Number of species (% of total) for which each landscape metric was identified as being the most important based on random forest (n = 5,000) MSE importance values for all species combined, by guild, and direction of response

	Combined		Early succession		Forest interior		Interior edge		Negative		Positive	
	#	%	#	%	#	%	#	%	#	%	#	%
For	6	12.5	1	6	2	15	3	16	2	8	4	21
CorFor	6	12.5	–	–	4	31	2	10.5	5	19	1	5
CWED	8	17	1	6	1	8	6	31.5	6	23	2	11
Mine	12	25	10	59	1	8	1	5	2	8	7	37
Grass	16	33	5	29	5	38	7	37	11	42	5	26

negatively responding species (Fig. 3), while forest loss followed closely by a tie between total mineland and grassland/shrubland were the top three for positively responding species. Grassland/shrubland was the only metric consistently ranked among the top three ranked metrics. Only considering the top metric by species, total mineland was most important for species responding positively while total grassland/shrub was most important for species responding negatively (Table 7).

Discussion

Avian response

Our results identified nonlinear avian responses to habitat loss and fragmentation and variation in sensitivity among guilds and species to alteration of landscape structure from MTR/VF. In general, the transition away from forest land-cover in our 1-km radius landscapes elicited more negative responses in relative abundance than positive ones within the avian community thresholds. This suggests that even when forest habitat remains post-mining, it becomes degraded and supports fewer forest interior birds. The introduction of mineland and grassland/shrubland land-cover types to a landscape produced immediate negative responses at thresholds of 1–2 % for the overall avian community and <10 % for most species. Negative responses of species to core forest loss, forest loss, and increased edge density, however, were more delayed occurring at 10 % loss of forest cover and 25 % loss of core forest, indicating some resilience to forest loss and introduced edges within the forest songbird community but within limits.

Landscape conversion to MTR/VF mining was not detrimental to all species. We identified a positive response to thresholds across all metrics for 19 species (39 %). However, a majority of these positive relative abundance responses by individual species occurred at greater amounts of land-cover change than the negative responses, meaning that negative responses occur before any beneficial responses and the gap between negative and positive responses was often large. All overall community thresholds had large gaps between the negative and positive thresholds, up to 75 % for total mineland.

The broad range in the TITAN bootstrap quantiles for the overall community thresholds indicate a more gradual response within the bird community (Baker and King 2010) most likely driven by the diversity of responses by individual species that have quite different habitat requirements ranging from early successional shrublands to core forest. Partitioning the overall community into habitat guilds identified narrower thresholds, more useful for management decisions directed at forest interior, interior edge, and early successional habitat guilds. Thresholds for individual species allow for evaluation of landscape change on a species by species basis and may be particularly useful for species of conservation concern.

Forest interior birds prefer interior forest habitats with minimal edge (Whitcomb et al. 1981) and our study supports that landscape change away from a forest-dominated landscape was detrimental to relative abundance of these species. This guild also was the most sensitive to landscape change though the greater uncertainty in forest loss thresholds as compared to interior edge species could suggest the need to reconsider the classification of some species between these two guilds in relation to forest change. It had the lowest thresholds for negative responses of the three guilds and all but one species KEWA within the guild had negative responses to all thresholds. Although Kentucky warblers are considered to be forest interior species, they also use shrubby woodland borders (DeGraaf and Rappole 1995) and were observed in forest with thick understory growth at our sample points, which explains their positive response to increased edge density and grassland/shrubland. The worm-eating warbler (WEWA) is an area sensitive (Robbins et al. 1989) forest interior species of conservation concern that prefers steep wooded hillsides with dense patches of shrub cover in deciduous and mixed deciduous-coniferous forests (Hanners and Patton 1998). Our thresholds suggest declines would occur in this species after 5 % forest loss (i.e. at <95 % forest within 1 km), 44 % core forest loss, and almost any amount of increase in mineland or grassland/shrubland. The cerulean warbler (CERW), another forest interior species of conservation concern that requires large blocks of forest (Hamel et al. 2004; Wood et al. 2006), had a negative response threshold of 11.6 % forest loss (Fig. 4a). Previous research suggests some tolerance to forest edges (Weakland and Wood 2005; Wood et al. 2006), which is supported by the absence of thresholds for mineland and core forest. Detrimental edge effects on forest interior species are more limited in heavily forested landscapes (Rudnicky and Hunter 1993) supported by the forest interior guild thresholds that indicated benefits from small increases in edge density (3.7–22.9 m/ha) after which the negative threshold was surpassed.

The majority of responses for the interior edge guild were negative but as expected, the guild seems resilient to some forest loss. This guild had higher negative response thresholds to forest loss (11 %) than the forest interior guild (4 %; Table 4), but individual species had thresholds of 4–13 % (Table 6) suggesting that 89 % forest in the 1-km landscape is a useful target for forest interior as well as interior edge species. A subset of species within this guild (e.g. brown-headed cowbird [BHCO], cedar waxwing [CEDW], summer tanager [SUTA], and yellow-throated vireo [YTVI]) responded positively to forest and core forest loss, which resulted in increased amounts of edge. However, only small amounts of mineland (7 %) and grassland/shrubland (9 %) within our 1-km landscapes were beneficial for this guild as these values are the threshold at which negative responses occurred. The magnitude of change in relative abundance was also reduced after surpassing thresholds relative to the other guilds (Table 5), indicating more gradual shifts in abundance.

The early successional guild, mostly shrubland species due to our sampling scheme, was the primary beneficiary of the observed landscape change; 10 species had significant positive responses to thresholds while only five had negative ones. Previous research has found successful occupation of reclaimed mines by grassland (Monroe and Ritchison 2005; Stauffer et al. 2011) and shrubland (Bulluck and Buehler 2008) birds. Although we sampled only within forest habitat, it would be expected that more grassland and shrubland habitats nearby would also increase detections of shrubland species within adjacent forest habitats; shrubland species often use forest edges. The positive response of species within the guild was due more to the presence of total grassland/shrubland rather than strictly mineland, which was the combination of reclaimed grassland and barren. Barren habitats seemed to have lesser utility for birds than grassland/shrubland as only Carolina wren [CARW] had a positive response to mineland, while four species responded positively to total grassland/shrubland (Table 6). However, negative response to forest thresholds (7–18.5 % forest loss) for EATO, INBU, and PRAW, Fig. 4c) indicates a benefit to maintaining forest within a landscape. These species may be using smaller patches from canopy gaps within the forested landscape, the transitional forest edge as habitat, or benefiting from the presence of tall trees as song perches. Overall, the guild benefitted from edge densities between 280–368 m/ha.

Management implications

To facilitate management decisions, we evaluated the diversity of community thresholds across different metrics with random forest importance ranks to identify key metrics. The identification of key metrics

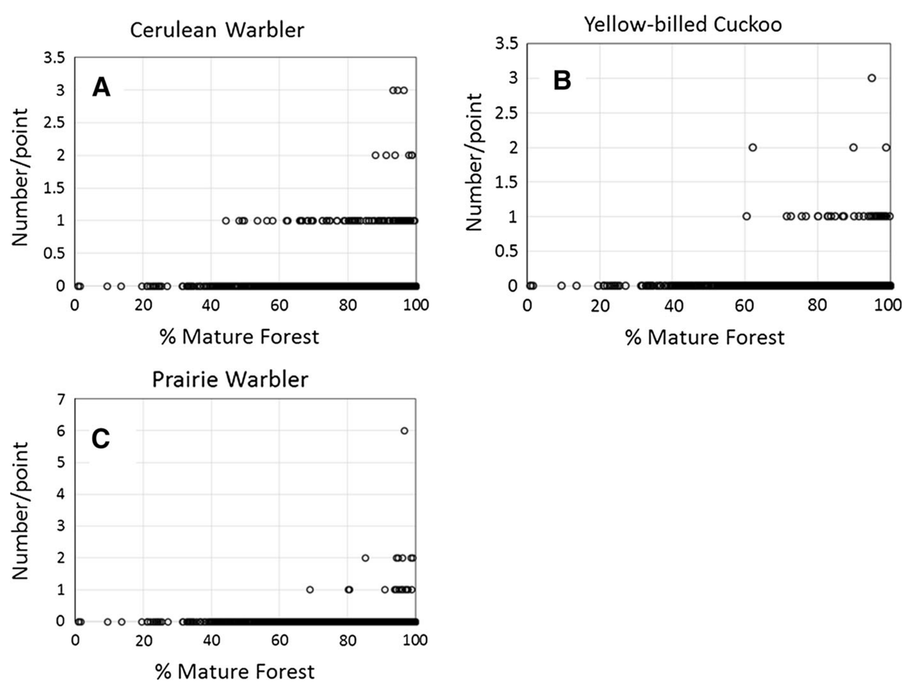

Fig. 4 Relative abundance (number/point) for a species within each guild with the greatest conservation need as identified in wildlife action plans for states within the central Appalachians at various levels of forest land-cover within 1-km radius landscapes

Table 2 Landscape metrics used for threshold analyses and calculated within 1 km radius around each sample point

Abbreviation	Description
For	100 minus percent of cover within a 1 km radius classified as forest
CorFor	100 minus percent of cover within a 1 km radius classified as forest and greater than 100 m from a non-forest edge
Grass	Percent of cover within a 1 km radius classified as mine grassland/shrubland or other grassland/shrubland. The majority is from mining when mining activity is present
Mine	Sum of % mine barren, % mine shrub or grass, % valley fill, % mining facilities, and % slurry impoundment within a 1 km radius
CWED	Meters/hectare for all edges occurring between forest and any other non-forest land-cover class. Edges between non-forest classes were given a weight of 0

is also important to facilitate interpretation among metrics thresholds, as each threshold analysis was completed individually. For example the percent mineland thresholds seem very important for forest interior species based on Table 4 but not important based on Fig. 3. This apparent contradiction exists because percent mineland, by itself, does identify specific thresholds; however, in the presence of other variables as part of the random forest analysis, the explanatory power of this metric is superseded by other metrics such as forest loss and grassland/shrubland.

Grassland/shrubland land-cover had the highest overall importance rank followed by the loss of both core forest and forest. We hypothesize that grassland/shrubland was important, even for negatively responding species because beyond estimating forest loss, it is also important to determine the composition of the habitat replacing the forest, with grassland/shrubland being better than barren habitat. In this context grassland/shrubland change can incorporate not only an estimate of forest loss but the type of habitat replacing forest possibly influencing its importance as a metric. Forest loss and core forest loss ranked closely among guilds with a slight edge to core forest, suggesting the need to maintain some intact interior forest.

We suggest management be focused on the amount of land-cover that is in forest, core forest, and grassland/shrubland as the management decision often is whether to emphasize forest or early successional habitat. Interior forest is important given the national importance of biodiversity within the region (Ricketts et al. 1999), the declines in many forest songbirds (Peterjohn et al. 1995; Rittenhouse et al. 2010), and the large extent of forest relative to other regions in the eastern United States. At the same time, early successional habitat is limited in the region (Trani et al. 2001) and birds dependent on grasslands, shrublands, and disturbed forest habitats have declined steeply (Hunter et al. 2001; Dettmers 2003). In choosing which goal to emphasize, little middle ground exists because any landscape change to promote benefits to early successional species will surpass all negative thresholds for forest interior species. Negative response thresholds to loss of forest (forest interior neg. 4 %, interior edge neg. 11 %) occur before reaching positive response thresholds (interior edge pos. 16 %, early successional pos. 52 %). Negative thresholds also occur before positive thresholds for forest interior and early successional species for the grassland/shrubland land-cover, although providing small amounts of grassland/shrubland could benefit interior edge species ($<$9 % total grassland/shrubland).

Many methods have been proposed to detect ecological thresholds (e.g. Brenden et al. 2008; Sonderegger et al. 2009; Dodds et al. 2010) with the goal to establish criteria for more informed management decisions. However, these approaches are limited because they fail to use multivariate species abundances (Brenden et al. 2008; Andersen et al. 2009). The advantage of our application of the TITAN approach is the derivation of quantitative community thresholds across multiple species as opposed to the need to derive multiple single-species thresholds. Our thresholds can simplify management decisions by condensing the variability of managing for multiple taxa into single quantified community management targets that include rarely detected species, often the focus of conservation concern, while maintaining the ability to derive and use individual species' thresholds as needed for management.

However, the overall community thresholds are harder to interpret given the different responses combined within one value compared to the more habitat specific avian guilds. The guild organization is more feasible within a management framework as the decision often is whether to manage for forest birds or to create habitat for early successional species. Thus, specifically defining management units or targets is important in maximizing the utility of the thresholds.

We caution against interpreting our results at broader landscape scales than the 1 km radius in which the analysis was conducted. Robbins et al. (1989) found a non-linear relationship between forest area and bird presence and Mitchell et al. (2001) found no single scale appropriate for all species and guilds in assessing landscape associations. Thompson et al. (2012) did find that forest cover within a 10 km radius was of the greatest importance for cerulean warbler abundance, creating the possibly this could also be true for our study. Overall, a need exists to expand this research across multiple and larger landscape scales as determined most vital for management. Additionally, these thresholds are only in response to changes in relative abundance and do not consider whether increased abundance also leads to increased reproductive success or survival.

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