Prioritize diversity or declining species? Trade-offs and synergies in spatial planning for the conservation of migratory birds

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Abstract

Stemming biodiversity loss requires strategic conservation guided by well-articulated and achievable targets, whether they be proactive (e.g., protect diverse places) or reactive (e.g., protect threatened or declining species). Both types of targets can be effective, but there are trade-offs, especially for broadly-distributed ecosystems or taxa, such as migratory species, a group for which conservation has been challenged by limited knowledge of distributions throughout the annual cycle. We combined fine scale abundance-based distribution models using crowd-sourced data with population trend analyses to evaluate how conservation targets emphasizing proactive (high species diversity) versus reactive (declining species richness and average severity of decline) approaches influenced spatial priorities for the conservation of Neotropical migratory birds (n=112 species), during the non-breeding period in the Western Hemisphere. We also examined how priority landscapes for each target were affected by the magnitude and changes in human footprint, forest cover, and land protection. Spatial priorities were strikingly different when targets emphasized areas of high species diversity (southern Mexico, northern Central America) versus areas with more severe average declines across species (Andean cordilleras of South America). Targeting richness of declining species was most congruent with high diversity. Only a fraction of non-breeding landscapes (0.03%), mostly in southern Central America and northern Colombia, were selected across all targets. Priority areas for the most severe average declines across the community were subject to rapid increases in human footprint, unlike areas selected for high diversity or declining species richness. Our results highlight how even different reactive conservation approaches may emphasize different regions with varying threats and opportunities for conservation. Overlap between areas prioritized for the conservation of high species diversity and declining species richness in the

northern Neotropics indicate a potential for conservation to support declining species without compromising conservation of diverse communities of Neotropical migrants. However, our study provides new evidence that rapid land conversion in the northern Andes may drive declines of Neotropical migrants overwintering in the region; conserving these communities of Neotropical migrants will require an emphasis on protecting and restoring habitat in northwest South America.

Keywords: citizen science, conservation prioritization, eBird, human footprint, Neotropical migrant, population decline, protected area

Introduction

Stemming the unprecedented rates of current biodiversity loss (Pimm et al., 2014; Ceballos, Ehrlich, & Dirzo, 2017) will require strategic and sustained investment in conservation but socio-economic limits on such investments make difficult choices inevitable (Wilson, McBride, Bode & Possingham, 2006; Carwardine et al., 2012; Martin et al., 2018). Although sophisticated decision-support tools can help inform such choices, the selection of a conservation target is a key initial step as different targets will often lead to different outcomes (e.g. Orme et al., 2005; Grenyer et al., 2006; Klein et al., 2009; Somveille, Manica, Butchart, & Rodrigues, 2013). Potential targets include areas of high species diversity (Buchanan, Donald & Butchart, 2011), critical habitat for threatened species (ESA, 1973; SARA, 2002), endemism (Myers, Mittermeier, Mittermeier, da Fonseca, & Kent, 2000) and areas of intact wilderness (Watson et al., 2009). Targets may also reflect contrasting conservation paradigms. For example, prioritizing high species diversity may be viewed as a proactive approach to conserving areas of high biodiversity value, whereas targeting areas with a smaller number of species in rapid decline, or a large number of species in modest decline, could be viewed as a more reactive response to species in

need of conservation effort (*cf.* Myers et al., 2000; Buchanan et al., 2011; Martin et al., 2018). In this paper, we illustrate the consequences of proactive versus reactive approaches for area-based conservation strategies of > 100 Neotropical migratory birds during the non-breeding season in the Western Hemisphere.

Migratory species are declining globally (Wilcove & Wikelski, 2008; Harris, Thurgood, Hopcraft, Cromsight, & Berger et al., 2009) and present unique challenges in conservation planning (Martin et al., 2007; Runge, Martin, Possingham, Willis, & Fuller, 2014). Because migrants traverse vast distances throughout the annual cycle we often do not know what locations and periods of the annual cycle most limit population growth. While threats on both the breeding and non-breeding grounds likely combine to influence abundance in most cases, the relative influence of threats at different stages of the annual cycle may vary among species within a community and among different populations (Rushing, Ryder, & Marra, 2016; Morrison, Robinson, Butler, Clark, & Gill, 2016; Taylor & Stutchbury, 2016; Wilson et al. 2018a; Wilson, Smith, & Naujokaitis-Lewis, 2018b). This issue creates uncertainty for multi-species conservation efforts because it is unclear which species would benefit from management actions within a particular region. A proactive strategy could still emphasize areas with high species diversity to protect against future threats in the region while simultaneously benefiting declining species in the community. However, this may be an inefficient use of scarce conservation resources if the recovery of threatened species is the priority. In that case our approach could focus on areas showing the strongest average declines across species. A region with steep community-wide declines provides evidence that threats within those regions are negatively impacting populations.

Neotropical migrant songbirds have recently received much attention (e.g. NABCI, 2012; NABCI, 2016) in large part due to long-term population declines of several species (Environment Canada, 2017; Sauer et al., 2017). The majority of conservation investment for this group is focused on the breeding period of the annual cycle in North America and primarily through reactive single-species conservation measures for those species undergoing the steepest declines (ESA, 1973; SARA, 2002). Yet, there is a growing concern and recognition that rapid land-use change on the non-breeding grounds is a key driver of declines for some threatened Neotropical migrants (Taylor & Stutchbury, 2016; Kramer et al., 2018; Wilson et al., 2018a). Thus, we may be able to more effectively conserve these species if we focus conservation resources on this period of the annual cycle. Historically, there has been far less information on the non-breeding season distribution and ecology of migrants in Neotropical regions, which may have in part contributed to the greater conservation focus on the breeding grounds. However, advances in crowd-sourced data and their application to species distribution models now allow us to generate weekly estimates of the abundance and distribution for most Neotropical migratory passerines over their entire annual cycles (Fink et al., 2010; Sullivan et al., 2014; La Sorte et al., 2017; Fink et al., 2018). Precise estimates of abundance are critical for predicting fine-scale species habitat use and response to landscape change that will allow for conservation planning at local scales (Possingham & Wilson, 2005).

Our first objective was to evaluate how the geographic regions and ecosystems targeted for conservation would differ depending on a proactive approach towards areas of high species diversity versus a reactive approach towards areas with the strongest average declines across species and the largest number of declining species. To accomplish this, we used distribution models based on crowd-sourced data for 112 Neotropical migrants during their stationary non-

breeding period (hereafter ‘non-breeding period’). The abundance-based 8.34 × 8.34km spatial grain we used in this analysis is one of the finest spatial resolutions to date for assessing the location of conservation targets at a hemispheric scale. We were also interested in identifying complementarity where conservation efforts might enhance protection of diversity hotspots and species in need of conservation attention. Our second objective was to examine the extent of recent land-use change and protected area status for priority areas based on proactive versus reactive conservation targets. We had two goals within this objective. First, by examining human land-use change and degree of protection within priority areas, we provide key information for the urgency of conservation and types of conservation actions that might be most suitable. Second, we identified the extent to which hotspots of community-wide decline and declining species richness coincided with rapid and recent land-use change within those regions. The identification of these regions would provide evidence for the location of threats during the non-breeding season that are contributing to population declines across the community.

Methods

Species selection

We initially identified 117 species of Neotropical migratory passerines for which weekly distributions were available based on eBird spatio-temporal exploratory models (Fink et al., 2010; Fink et al., 2018; Supporting Information (SI) Table S1). We set the non-breeding period as November 14 – March 14 to exclude spring and fall migration and retained all species with weekly distribution maps throughout that period (n = 112 species). Species fell into two broad groups based on their breeding and stationary non-breeding ranges: 1) species that are solely migratory, wherein all individuals breed in North America north of the US-Mexico border and migrate south of the Tropic of Cancer during the non-breeding period (n = 95 species); 2) species

with migratory and resident populations in the Neotropics, wherein individuals from migratory populations north of the US-Mexico border migrate south of the Tropic of Cancer during the non-breeding period (n = 17 species). STEM maps were not available for all Neotropical migrants passerines and thus a small number were not included in our analysis.

Spatiotemporal exploratory models

We used the eBird citizen-science database (Sullivan et al., 2014) to estimate the weekly abundance of all species in each eBird cell (hereafter 'landscape', Fig. 1) during the non-breeding season. Estimates of abundance were derived previously using spatiotemporal exploratory models (STEM) and detailed descriptions of the development and application of STEM are provided elsewhere (Fink et al., 2010; Fink et al., 2018). Briefly, the eBird data used in STEM includes species counts from complete checklists collected under the "traveling", "stationary", and "areal" protocols from January 1, 2004 to December 31, 2016 within the spatial extent bounded by 180° to 30° W Longitude (as well as Alaska between 150° E and 180° E). This resulted in a dataset consisting of 14 million checklists collected at 1.7 million unique locations. The STEM developed from this database are an ensemble of local regression models generated from a spatiotemporal block subsampling design. Abundance of each species and in each week are assumed to be stationary. Zero-inflated boosted regression trees are used to predict the observed counts (abundance) of species based on spatial covariates (e.g. relationships with land cover types), temporal covariates to account for trends, and predictors that describe the observation/detection process to incorporate variation in detectability. Estimates of relative abundance are rendered at weekly temporal resolution and at an 8.34 × 8.34 km spatial resolution. The quantity estimated per eBird landscape was the expected number of birds of a given species by a typical eBird participant on a search starting from the center of the pixel from

7:00 to 8:00 AM while traveling 1 km. The final set of models contained estimates of abundance for all weeks throughout the annual cycle; for this analysis, we restricted the time window to the non-breeding period as described above.

Human Footprint

We used the global human footprint index (Sanderson et al., 2002; Venter et al., 2016) to identify the extent and trends in human pressures for all eBird landscapes. The human footprint index has been shown to be a strong predictor of ecological and conservation metrics (Yackulic, Sanderson, & Uriarte, 2011; Hand, Cushman, Landguth, & Lucotch, 2014). The index is a composite measure of human impact derived from eight separate measures: 1) built environments, 2) crop land, 3) pasture land, 4) human population density, 5) night-time lights, 6) railways, 7) roads, and 8) navigable waterways. These eight measures are individually weighted based on their relative levels of human pressure and summed to create a single standardized estimate. The index varies from 0 (no footprint) to 50 (very high footprint) and is estimated at a 1km resolution across all global terrestrial lands except Antarctica. The index was first measured in 1993 and again in 2009 and therefore, for all eBird landscapes in our analysis we obtained an average footprint in these two years as well as the change over the 16-yr period. With the $1km^2$ resolution there were approximately 70 estimates of footprint in each landscape and these were averaged to create a single measure.

Land Cover

We used the 2015 data set of the global land cover map (ESA Climate Change Initiative, 2017) at a 300m resolution to examine the extent of forest cover and diversity of land covers among the priority landscapes for each target as well as for the areas where priority landscapes of different targets overlap. The land cover data layer was clipped to the study region with 32 land cover

types that comprised at least one pixel. With the 300m resolution there were approximately 773 land cover pixels in each eBird landscape and for each landscape, we calculated the proportion of pixels that comprised forest cover and a Shannon index of land cover diversity using the Vegan package (Oksanen et al., 2017) in R Version 3.4.2 (http://www.r-project.org/).

Protected area coverage

We used the World Database on Protected Areas (WDPA, UNEP-WCMC, 2018) to estimate the extent of protected area coverage for the three conservation targets. The WDPA includes seven categories of protected area coverage: (Ia) strict nature reserve, (Ib) wilderness area, (II) national park, (III) national monument, (IV) habitat/species management, (V) protected landscape/seascape, (VI) managed resource protected area. Following La Sorte et al. (2017) we defined these seven classes into three representing high (Ia, Ib), medium (II, III) and low (IV, V, VI) protection status. We then intersected the WDPA layer with the eBird landscape layer. For all targets we calculated the proportion of landscapes that contained each of the 7 protected area types and the high, medium and low protected area categories.

Analyses

We used the eBird abundance information to calculate Shannon diversity, median long-term BBS trends (hereafter referred to as “average severity of decline”) and number of declining species for all eBird landscapes as defined above. A few species in our analysis also over-winter in regions well outside of the Neotropics (e.g. Yellow-rumped Warbler). Therefore, we defined the non-breeding region based on all landscapes containing five or more species. This definition allowed us to include the distributions from all species while excluding large, peripheral areas that are well outside of the main non-breeding range for the vast majority of Neotropical migrants. Long-term estimates of population trend in Neotropical migratory birds do not exist for

the non-breeding period; thus, we used continent-wide trends estimated for each species from 1966-2015 via the North American Breeding Bird Survey (BBS, Environment Canada, 2017; Sauer et al., 2017). In this case, for each species present in each landscape we used the long-term continental trend to estimate the median trend across all species detected in that landscape. To estimate the richness of declining species in each landscape we summed the number of species present with a long-term continental trend below 0. Population trends were unavailable for three species (Table S1).

For each target, we defined the top 10% and 5% (see Appendix 1) of landscapes as a priority (i.e. based on highest diversity, average severity of decline and declining species richness). This approach of selecting a top percentage of values in each target category is similar to that used in earlier studies (e.g. Orme et al. 2005; Hof, Araújo, Jetz, & Rahbek, 2011). We then identified the geographic regions of these priority landscapes for each target individually as well as the regions where the priority landscapes of different targets overlapped. To quantify the spatial overlap among targets, we used a Bray-Curtis measure of dissimilarity between the priority landscapes. Dissimilarity calculations were performed using the Vegan package (Oksanen et al. 2017) in R Version 3.4.4.

Results

Geographic variation in priority landscapes

Our results highlighted a strong north-south contrast between regions containing the highest diversity and those with the most severe average declines across the community (Fig. 2). This contrast resulted in very different regions where landscapes were prioritized for the two targets. Priority landscapes for high species diversity were concentrated along the Pacific slope of Mexico, the Yucatan peninsula, western Cuba and southern Mexico through Nicaragua (Fig. 2,

Supporting Information (SI) Fig. S1). In contrast, priority landscapes based on average severity of decline were mostly in Andean regions of Colombia through Bolivia (Fig. 2, S1). The regional concentration of priority landscapes for declining species richness was also in the northern Neotropics, similar to diversity, although with fewer priority landscapes on Mexico's Pacific slope (Fig. 2).

Variation in focal regions for the proactive versus reactive conservation targets were even more apparent when we examined the extent to which the targets overlapped. Of 96,079 landscapes in the non-breeding region, only 26 were in the top 10% for scenarios based on all three targets (Table 1). Overlapping landscapes were mainly in two regions; the Cordillera de Talamanca of Costa Rica and west Panama, and the Cordillera Occidental of Colombia (Fig. S2). The overlap was also low between landscapes for average severity of decline and high diversity targets (n=35 landscapes, Table 1, Fig. S3) and between average severity of decline and number of declining species (n=155 landscapes, Table 1, Fig. S4). In contrast, there was a strong, positive correlation (r=0.79) in the target areas for high diversity and number of declining species, particularly in southern Mexico, the Yucatan peninsula and Honduras (Table 1, Fig. S5).

Human footprint and land cover among target landscapes

The magnitude and trend in the human footprint index differed among the three targets. Priority landscapes for diversity and average severity of decline had a smaller average human footprint than the non-breeding region overall, whereas sites prioritized based on the number of declining species had footprints slightly above the region average (Table 2). From 1993 to 2009, the human footprint index declined by 6.4% in priority landscapes based on high diversity and number of declining species, but increased by 11% for priority landscapes for average severity of decline. This pattern was also reflected in a higher proportion of landscapes where the footprint

increased between 1993 and 2009 for priority average severity of decline landscapes (67.2%) than for priority diversity (43.0%) or declining species richness (42.5%) landscapes. The three targets had similar land cover patterns with relatively greater forest cover and lower land cover diversity compared to the regional average (Table 2).

Differences in protected area coverage among target landscapes

Across the entire non-breeding region, 15.9% of all landscapes had some form of protected area status (Table 3). Priority landscapes for high diversity and number of declining species targets were similar in their protected area coverage, with 19.44% and 18.86% respectively having some form of protection and the majority of those with low protection status (69% and 66% respectively), mostly in the form of managed resource protected areas (Table 3, A1). Priority landscapes for average severity of decline had a similar percentage of landscapes with some form of protection (18.12%) but in contrast to the other two targets, the majority of these (64%) had medium protection status and were primarily National Parks (Table S2). Thirteen of the 26 landscapes (50%) that met all three targets had protected area status with medium status in 7 (all National Parks) and low status in 7 (all managed resource protection areas) with one landscape containing both types (Table 3, S2).

Discussion

To our knowledge this is the first study combining fine-scale abundance-based distribution data with population trend analyses to examine the complementarity of proactive versus reactive conservation approaches for migratory species. Our results highlight the importance of target selection when prioritizing landscapes for conservation (Possingham & Wilson 2005; Klein et al., 2009). We observed marked differences in landscapes prioritized for conservation using a proactive target that emphasized high species diversity versus a reactive target focusing on areas

with more severe average declines across the community. When the target was high species diversity priority landscapes were distributed mainly in the northern portion of the non-breeding region in southern Mexico, the Yucatan Peninsula and northern Central America, whereas targeting areas with more severe average declines across the community emphasized the Andean cordilleras of South America. Interestingly, the priority landscapes for an alternative reactive approach that emphasized the total number of declining species, irrespective of the extent of that decline, coincided to a much greater extent with those selected for high diversity than those selected for average severity of decline.

Global analyses of resident taxa offer mixed results on the extent to which areas containing threatened species are congruent with areas of high species diversity. In amphibians, strong spatial overlap exists between the total number of species and the number at risk of extinction (Hof et al., 2011), but in birds and mammals there is less congruence at global scales (Orme et al., 2005; Schipper et al., 2008; Jenkins, Pimm, & Jopa, 2013; Pimm et al., 2014). To our knowledge, only one previous study has addressed this topic for migratory species and found little congruence between areas of high richness, endemism and threatened status at global scales (Somveille et al., 2013). Whereas some regional studies show high spatial overlap in species diversity and endangerment (e.g. Chilean mammals, Cofré & Marquet, 1999), it is not evident in other cases (e.g. British fauna, Prendergast, Quinn, Lawton, Eversham, & Gibbons, 1993). The extent of congruence between these two targets often depends on the degree to which threatening anthropogenic processes overlap areas rich in resident taxa (Orme et al., 2005; Hof et al., 2011) as well as the timeframe examined (Bennett & Arcese, 2013). For example, habitat conversion by humans might initially lead to spatial correlations between diversity, population decline and human influence at regional scales, but eventually leads to negative correlations between rare

species occurrence and human influence as declining species are extirpated from human-dominated landscapes (Bennett & Arcese, 2013).

Our results on the extent of congruence among landscapes supporting diversity versus declining species were complex and depended on how we used the information on population trend to define a species in need of conservation attention. When our reactive approach emphasized areas with species showing more severe average declines, the results were consistent with those described above with little congruence between diversity and species in need of conservation attention when threatening processes and diversity do not overlap. Landscapes in more southern regions where average species declines were greater showed a recent increase in the human footprint of about 1% annually and even this rapid change understates the longer-term extent of habitat loss in the northern Andean region where less than 10% of the original forest cover remains (Henderson, Churchill, & Luteyn, 1991). In contrast, there has been a slight reduction in the human footprint in landscapes prioritized for the highest Neotropical migrant diversity.

When we changed the definition of our reactive conservation target to the number of declining species there was a strong congruence with total diversity but little congruence with recent anthropogenic change or areas prioritized for average severity of decline. This finding stresses that even different conservation targets with a similar theme (i.e. declining species) can lead to different outcomes for spatial conservation planning. We suspect that the lack of overlap in regions with more severe average declines across the community and those with the largest number of declining species is the combined result of two processes 1) the decrease in Neotropical migrant diversity towards the southern portion of the non-breeding range and 2) the influence of multiple stages of the annual cycle on population abundance and trends. We would

expect a high congruence between average severity of decline and declining species richness if threats only occurred during the stationary non-breeding period but this is very unlikely to be the case. Approximately 55% of the species in the northern portion of the non-breeding range still had negative trends even if the majority of those were slight compared to species overwintering in South America (see below). There is less evidence that conditions on the non-breeding grounds have negatively impacted species overwintering in Mexico, Central America and the Caribbean (see also NABCI 2012) and it is possible that species overwintering in these regions are more evenly influenced by threats throughout the annual cycle.

Our study provides new evidence that recent land conversion in the South American Andes has contributed to population declines of Neotropical migrants overwintering in the region (Jones, Barg, Sillett, Veit, & Robertson, 2004; González-Prieto, Bayly, Colorado, & Hobson, 2017; Kramer et al. 2018, Wilson et al. 2018a). Indeed, 18 of 20 species in our analysis whose winter range primarily includes northwest South America had declining trends with an average decline for the community of approximately 51% between 1966 and 2015. Yet, because the stationary portion of the non-breeding period represents only 50-60% of the annual cycle in most species, other drivers may be involved. The group of declining Neotropical migrants in South America occupy broad and often different regions on the breeding grounds and thus, a common breeding ground threat for community wide declines in this group is unlikely. Yet, many of these species use similar routes for at least part of the spring and/or fall migration and further study is needed on the extent to which threats along shared migratory pathways are contributing to the stronger declines for more southern overwintering Neotropical migrants. Steeper population declines have previously been reported for long-distance compared to short-distance migrants (Sanderson, Donald, Pain, Burfield, & van Bommel, 2006). This finding is consistent with

observations that high energy demands and unfamiliar landscapes make migration a particularly risky stage of the annual cycle (Sillett & Holmes 2002, Klassen et al. 2014), especially where migratory pathways intersect with human development (Berger, Young, & Berger, 2008; Hewson, Thorup, Pearce-Higgins, Atkinson, 2016).

Conclusions

Considerable overlap between areas prioritized for the conservation of high species diversity and declining species richness indicate a potential for conservation investment in landscapes likely to support declining species without compromising conservation of diverse communities of Neotropical migrant birds. Because contemporary declines often predict future declines (Keith et al., 2015), area prioritizations based on the above targets are consistent with strategies aimed at 'keeping common species common' and stemming declines, including for species currently of low conservation concern (Ceballos et al., 2017). However, as we have shown, an emphasis on areas of high diversity or higher numbers of declining species will not incorporate areas where the Neotropical migrant community is declining most strongly. Many of the species in these communities are currently the focus of species-specific conservation efforts on the breeding grounds (ESA 1973, SARA 2002) but our results point to the importance of protection and restoration of habitat in the northern Andes for effective conservation of these species. Although we found limited potential to meet all targets, we did identify ~2400km^2 of habitat in southern Central America and northwest Colombia that met all three; about 30% of this area already is in a National Park, another ~30% is managed with lower conservation emphasis. This type of information also shows how the method we used in this study could inform planning at a local scale. As an example, if the priority is to protect areas that meet proactive and reactive targets, then the types of information provided in the approach we used could be combined with socio-

economic information (e.g. Naidoo et al. 2006; Wilson et al. 2006) to identify the costs and benefits of allocating conservation status to sites within other landscapes that currently do not have protected area status.

Authors' Contributions

SW, RS, ADR, JRB and PA designed the study; RS, SW and ACS analyzed data; SW wrote the manuscript with feedback and contributions from all co-authors.

Data Accessibility

eBird STEM maps are in the process of being made publicly available (Fink et al. 2018, https://doi.org/10.1101/251868; see also https://ebird.org/science/modeling). All other databases used in this analysis are publicly available: Breeding Bird Survey trend data (Environment Canada, 2017, https://www.canada.ca/en/environment-climate-change/services/bird-surveys/landbird/north-american-breeding/overview.html; Sauer et al., 2017, https://www.pwrc.usgs.gov/bbs/), Human Footprint Index (Venter et al., 2016, https://doi.org/10.1038/ncomms12558), Land Cover (ESA Climate Change Initiative, 2017, http://maps.elie.ucl.ac.be/CCI/viewer/download.php), Protected Areas (UNEP-WCMC, 2018, https://www.protectedplanet.net/c/world-database-on-protected-areas).

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Table 1. Comparison of landscape overlap for conservation targets based on diversity, average severity of decline and number of declining species. Target landscapes are based on the upper 10% of values for each case. Landscapes = number of overlapping landscapes in each comparison, Percent of total = percent of all landscapes in the study region that overlap, dissimilarity = Bray-Curtis dissimilarity index for the landscape pairs in each comparison.

Comparison	Landscapes	Percent of total	Dissimilarity
Diversity/severity of decline	35	0.04	0.996
Diversity/number declining	6622	6.89	0.355
Severity of decline/number declining	155	0.16	0.985
All targets	26	0.03	-

Table 2. Comparison of trends in the human footprint index, land cover diversity and proportion of forest cover for priority landscapes selected for each target (means with 95% CI in brackets underneath). HF = average human footprint index, ΔHF = percent change in HF between 1993 and 2009, LC diversity = Shannon index of land cover diversity, proportion forest = average proportion of landscapes covered by forest land cover types.

Response	HF 1993	HF 2009	ΔHF	LC diversity	Proportion forest
Diversity	7.63	7.14	-6.38	0.636	0.763
	(7.53, 7.73)	(7.04, 7.25)		(0.627, 0.646)	(0.757, 0.768)
Severity of decline	5.72	6.35	11.04	0.701	0.663
	(5.60, 5.85)	(6.23, 6.48)		(0.688, 713)	(0.656, 0.670)
Number declining	10.62	9.94	-6.37	0.738	0.627
	(10.45, 10.78)	(9.78, 10.10)		(0.728, 0.748)	(0.620, 0.634)
Diversity/Severity of decline	7.20	8.41	16.79	0.269	0.890
	(4.87, 9.53)	(5.97, 10.85)		(0.127, 0.411)	(0.820, 0.960)
Diversity/number declining	7.94	7.37	-7.17	0.616	0.627
	(7.81, 8.07)	(7.25, 7.50)		(0.604, 0.628)	(0.618, 0.635)

Response	HF 1993	HF 2009	ΔHF	LC diversity	Proportion forest
Severity of decline/number declining	8.00	8.84	10.50	0.633	0.753
	(6.89, 9.12)	(7.69, 9.99)		(0.550, 0.716)	(0.709, 0.797)
All targets	8.15	9.48	16.31	0.339	0.856
	(5.12, 11.19)	(6.31, 12.64)		(0.156, 0.522)	(0.765, 0.947)
Region	9.09	9.01	-0.88	0.778	0.465
	(9.04, 9.14)	(8.96, 9.06)		(0.774, 0.781)	(0.463, 0.468)

Table 3. Comparison of protected area coverage for priority landscapes selected by each target. Values show the percent of selected landscapes for each target that contained high (strict nature reserve, wilderness area), medium (national park, national monument, habitat/species management) and low (protected landscape/seascape, managed resource protected area) categories. The number of selected landscapes for each target are shown after the target name. See Appendix A for the proportion of target landscapes for all 7 individual protected area classes.

Response	High	Medium	Low	Any
Diversity (9609)	1.63	6.68	13.34	19.44
Severity of decline (9726)	0.56	11.56	6.89	18.12
Number declining (10922)	1.64	6.92	12.42	18.86
Diversity/severity of decline (35)	0.00	28.57	28.57	51.43
Diversity/number declining (6622)	2.01	8.17	14.97	22.38
Severity of decline/number declining (155)	0.00	9.68	9.68	18.71
All targets (26)	0.00	26.92	26.92	50.00
Non-breeding region (96079)	2.52	5.53	9.78	15.93

Figure 1. An example eBird landscape approximately 8.3 × 8.3 km in southwest Costa Rica. Using STEM distribution maps for 112 Neotropical migrants we predicted that this landscape would have a Shannon diversity = 3.04, declining species richness = 26 and a median population decline across species = 34.2% between 1966 and 2015. Forest cover in this unprotected landscape was measured at 67% in 2016 and the human footprint index increased by 15.7% between 1993 and 2009. We estimated these metrics for 96,079 landscapes across the non-breeding range of Neotropical migrants in the western hemisphere.

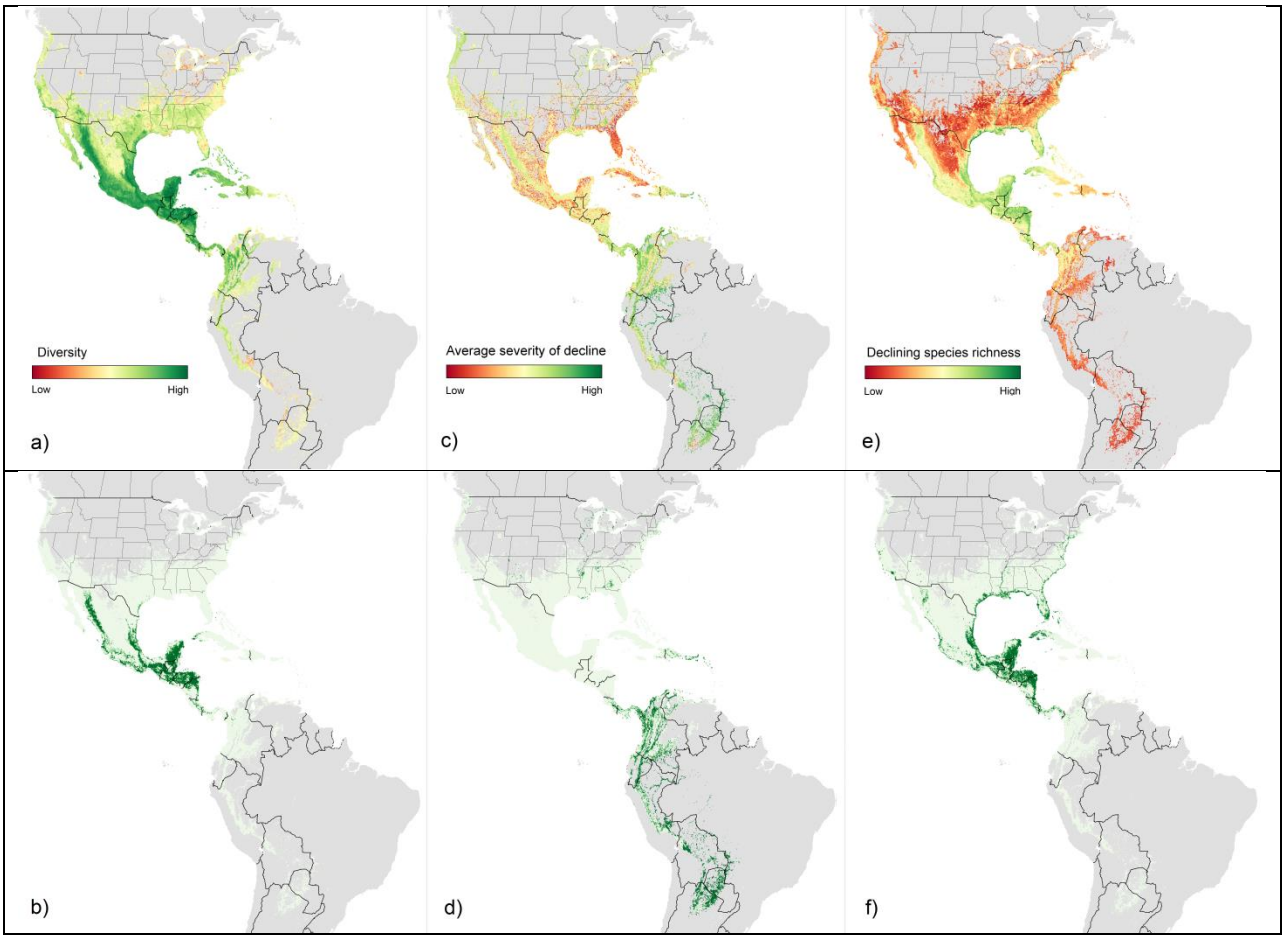

Figure 2. Fine-scale patterns (top row) and priority landscapes (bottom row) for Neotropical migrants based on high diversity (a,b), average severity of population decline (c,d) and declining species richness (e,f). Priority landscapes in the bottom row are based on the top 10% of values for each target. The stationary non-breeding region was defined by landscapes with at least 5 species present between Nov 14 and Mar 14 and is shown in light green in the bottom panel figures.