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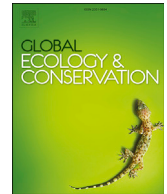


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Decline in forest bird species and guilds due to land use change in the Western Himalaya

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ABSTRACT

Land use change is the most widespread driver of biodiversity loss in densely populated tropical countries. Biodiversity loss, in turn, results in changes in functional guilds responsible for various forest ecosystem services. It is thus necessary to understand the extent and types of biodiversity loss and functional guild alteration caused by land use change in order to facilitate sustainable land use policies. Here we study the effects of land use change on forest bird species and guilds in a human-dominated landscape in the Western Himalaya, India. We carried out systematic breeding-season surveys in six land use types within moist temperate forest: natural (protected) oak forest, degraded (lightly used) oak forest, lopped (heavily used) oak forest, pine forest, cultivation and built-up sites, in two adjoining landscapes, over two consecutive years. Our study shows moderate to drastic species loss in all modified land use types in comparison to natural oak forest. Species composition in modified land use types diverged significantly from natural oak; this effect was highest in cultivation and built-up sites and least in degraded forests. Compositional change in modified land uses occurred due to partial replacement of forest specialists with commensals and open country species, whereas abundance of forest generalists was relatively constant across the gradient. We also find a steep decline in the abundance of functional guilds such as pollinators, and insectivorous pest controllers in all modified land uses in comparison to natural oak forest. Our results have important implications for conservation in biodiverse mountain landscapes with significant human imprint. In particular, (a) low faunal diversity in monocultures and urban sites (b) drastic (50% loss or more) loss of forest specialists, pollinators and insectivores in degraded forests, monocultures and urbanised sites; and (c) the potential for degraded forest as refugia for forest species, are findings that can be globally applied to land use and conservation planning in human-dominated landscapes.

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1. Introduction

Land use change is considered to be one of the leading drivers of global biodiversity loss (Young et al., 2016; Maxwell et al., 2016). Such species losses, which were earlier expressed over long periods of time, have drastically increased in speed and extent in the Anthropocene primarily due to expanding populations and national economies. An increasing demand for food has transformed a third of the world's land surface to agro-pastoral production; of this, half has occurred at the cost of tropical forests (Díaz et al., 2020). By 2030, 1.2 million km² of global land area will be urbanised to accommodate five billion people (Seto et al., 2012).

Land use change comes at a huge cost to natural habitats such forests and wetlands (Hansen et al., 2013) and therefore, to biodiversity (Maxwell et al., 2016). Birds as a taxon have been relatively better studied: numerous studies show loss of bird species due to land use change across many biomes. For instance, the impacts of land use change on bird diversity have been studied in various geographic contexts such as Amazon basin (Barlow et al., 2007; Srinivas and Koh, 2016), Eastern Himalaya (Mandal and Shankar Raman, 2016), Western Ghats in India and Sri Lanka (Sreekar et al., 2015) and Mexican highlands (Carillo-Rubio et al., 2014). Declines in species abundance can be related largely to reduction in structural complexity and resources of forest habitats as well as alterations in predator-prey and competitive relations (Sol et al., 2020; Menon et al., 2019; Shochat et al., 2006).

Land use change additionally leads to divergence in species composition caused by selective loss and decline in species and functional guilds that are more vulnerable than others (Menon et al., 2019; Sreekar et al., 2015; Srinivas and Koh, 2016). Recent global scale meta-analysis (Newbold et al., 2015; Sol et al., 2020) and predictive modelling exercises (Hinz et al., 2020; Seto et al., 2012) have revealed the patterns of avian loss in terms of vulnerable species and guilds. Differential vulnerability of species and guilds depends on degree of resource specialisation, primary foraging substrate, population resilience as well as dispersal ability, among other factors (De Lima et al., 2013; Menon et al., 2019; Srinivas and Koh, 2016; Newbold et al., 2013). For example, forest specialists and insectivores tend to decrease (Menon et al., 2019) while commensal species and granivores tend to increase in abundance in intensively used landscapes (De Lima et al., 2013; Shochat et al., 2006).

Avian community change, in turn, can have significant repercussions on forest ecosystem function (Gaston et al., 2018). For instance, decline in population of certain bird species has been linked to decline in critical ecosystem services such as pollination, biological pest control and seed dispersal (Whelan et al., 2015; Maas et al., 2013). Recent meta-analyses also indicate significant alteration in functional guild abundance and composition due to land use change (Newbold et al., 2020; Sol et al., 2020). Empirical studies quantifying abundance and composition of such functional guilds in managed forests and alternative land uses in specific biomes, are urgently required because avian population declines can significantly affect ecosystem processes (Whelan et al., 2015; Maas et al., 2013; Gaston et al., 2018). It is for this reason that scholars emphasize the importance of documenting not just species losses, but also abundance declines in species, a process termed as 'defaunation' (Young et al., 2016).

The Himalayas and their terrestrial ecosystems, have been recognised as a global biodiversity hotspot owing to high species richness and endemism, as well as high levels of threat due to land use changes (Mittermeier et al., 2004). Himalayan forests and their biodiversity are threatened by expanding forest modification, degradation and conversion to other land uses as well as climate change (Pandit et al., 2007, 2014). Furthermore, there are few strictly protected areas except in the highest reaches of the alpine zone. While agricultural land uses have been studied to some extent in the Himalayas (Elsen et al., 2017), there is still little information on the diverse array of other land uses such as new types of monocultures, expanding urban centres and intensively extracted forests (Pandit et al., 2014). In this paper, we examine the effects of land use change on bird assemblages in forests of Western Himalaya, as well as the relative vulnerabilities of various habitat and functional guilds.

We hypothesised that both bird species richness as well as the degree of community similarity to natural forest stands, would decrease along the land use intensification gradient. We then examine the effect of land use change on various habitat guilds of birds in the study landscape, expecting a decrease in forest specialists, little or no change in forest generalists and an increase in commensals and open country species along the land use intensification gradient. Finally, we assess the effect of land use change on the abundance of three avian functional guilds - pollinators, ecosystem engineers and insectivorous pest controllers. All three groups of ecosystem service providers are significant for agricultural activity. Pollinators help in fertilizing crops and fruit-trees; ecosystem engineers (primarily barbets and woodpeckers) expand the niches available for insectivorous birds, while insectivorous birds provide biological control on insect pests in field and orchards (Sekercioglu 2006).

2. Materials and methods

2.1. Study area

Our study was focussed on the natural oak-dominated forests occurring between 1500 and 2400 m asl in the Western Himalaya, dominated by *Quercus leucotrichophora*, *Q. floribunda* and other oak species. Such forests form part of the Western Himalayan moist temperate forest biome as well as an Endemic Bird Area (Birdlife International, 2016). Himalayan oak forests host high bird diversity due to their complex vegetation structure, evergreen nature, species-rich flora, dense leaf litter and moist micro-climate (Shahabuddin et al., 2017).

A 1285 km² study area was selected for study within the oak-dominated forests, bounded by the latitudes 29.48 and 29.37 N, and longitudes 79.35 to 79.67 E between the altitudes of 1700 m–2400 m (Fig. 1). In the oak forests, *Q.floribunda* and *Q.leucotrichophora* occur as the dominant species in the late-successional oak forest, along with associates such as *Rhododendron arboreum*, *Lyonia ovalifolia*, *Myrica esculenta* and *Aesculus indica*. Two separate landscapes separated by 5 km, were chosen for detailed study: Landscape A was centred around the town of Mukteshwar and Landscape B was around Pangot village (see Fig. 1). Both landscapes had adequate representation of the six different land use types in this region as well as substantial natural oak forest area.

2.2. Selection of study sites

Six major land use types were identified in the study area based on degree and type of anthropogenic use and are described below in increasing order of use intensity and modification of natural habitat (see also Fig. 2a,b,c and Fig. 3a,b,c for photographs of each land use category).

Natural or protected oak forest (hereafter ‘natural oak’) represents the natural forest of that landscape with no recorded history of felling or management (Fig. 2a). Degraded oak forest (hereafter ‘degraded oak’; Fig. 2b) represents lightly used forest from where fire-wood, leaves and leaf litter are collected. Such forests show poor understorey and leaf litter layer, fewer large/tall trees but a moderately intact canopy cover. Heavily degraded or lopped oak forest (hereafter ‘lopped oak’; Fig. 2c), represents forest with highest intensity of use, primarily, leaves harvested for cattle feed. Such forests have poor canopy, heavily lopped trees, no leaf litter and high densities of stumps and coppices. Our delineated land use classes were based on both visual observation, but also on the well-studied disturbance regimes present in the study area (see Thadani and Ashton 1995). However, in order to confirm that our visual classification on the basis of presumed use intensity was useful, we quantified two variables related to disturbance in each of the study sites (see below): extent of leaf litter collection and extent of tree-lopping. The graphed data can be seen in Appendix A, and confirm that the three oak forest classes were quantifiably different from each other in intensity of use.

Pine forest comprises of primarily chir pine (*Pinus roxburghii*), an early successional native species, occurring in largely mono-dominant stands. Pine forest shows low tree density and poor canopy and grows on poor soils (Fig. 3a). There is evidence that pine forests are expanding in area at the cost of the oak forests due to chronic use, frequent fire and warming (Naudiyal and Schmerbeck 2017; Shahabuddin and Thadani 2018). Pine forests are often burnt for fodder, and used for extraction of pine needles and pine resin. Cultivation sites represent a mix of subsistence and cash-cropping where fruit trees

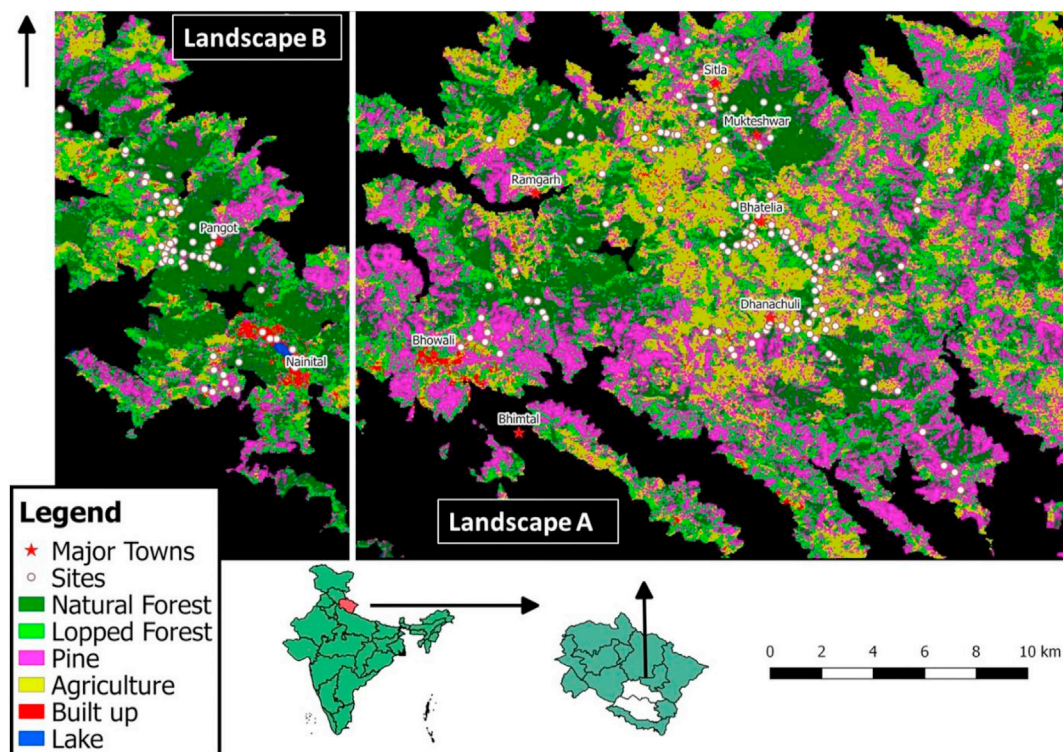


Fig. 1. Map of the 198 study sites in the two neighbouring landscapes in the Western Himalaya, where breeding bird surveys were done using similar protocols in 2016 (Landscape A) and 2017 (Landscape B).



Fig. 2. Images of (a) natural oak forest, (b) degraded oak forest (lightly used), and (c) lopped oak forest (intensively used) that were studied as three distinct land use types in the Western Himalaya, India.



Fig. 3. Images of the three non-oak land use types studied for birds in Western Himalaya: (a) pine forest, (b) cultivation and (c) built-up sites. Detailed descriptions are in the text.

coupled with seasonal vegetables and grains are cultivated along terraces (Fig. 3b). Built-up sites are semi-urbanised landscapes, comprising clusters of village houses, tourist resorts and summer homes often separated by fallow areas, fruit trees and private gardens (Fig. 3c).

In Landscape A, 138 study sites were selected, with 23 in each of the six land use categories. In Landscape B, 60 study sites were selected with 10 in each land use category. Thus, there were 198 study sites in all, distributed equally over the six delineated land use categories. Each study site had a homogeneous land use buffer of at least 50 m and was located at least 200 m from the next nearest site. The study sites in each land use category were spatially well-distributed without any clustering of sites belonging to one category. Metadata on each of the 198 sites are given in [Appendix B](#) and [Appendix C](#), corresponding respectively to Landscape A and Landscape B.

2.3. Data collection

We surveyed bird communities twice in each study site during the breeding season of 2016 in Landscape A (April 2 to June 7, 2016). Given resource constraints, our surveys were limited to the breeding season because birds are far more specific in their resource needs during this period rather than during winter (eg. [Kumar et al., 2011](#)). In Landscape B, we surveyed bird communities twice in 2017 (April 17 to May 5, 2017), using point counts. At any given site, the two replicate counts were done with a gap of 10–14 days during the field season, in order to account for any weather variation during the season. Both landscapes could not be covered during the same year due to the brief breeding period for birds (March to May) in Western Himalaya. During a bird count, birds seen or heard within a 30 m radius of the central point were identified and recorded over a period of 15 min, after a 5-minute rest period. Species were visually identified using a well-known field guide for birds of the Indian subcontinent ([Grimmett et al., 2011](#)). Unfamiliar bird calls were recorded in the field and later identified using the online resource *Xenocanto* ([Xeno-canto, 2020](#)). Bird observations started at sunrise (0700 h) and went on until 1000 h, to match periods of maximal bird activity. Bird nomenclature was revised based on a peer-reviewed list that incorporates all recent world-wide nomenclatural changes for Indian species ([Praveen et al., 2016](#)). Bird surveys were carried out by the same observer (RG) throughout the study, thus minimising observer-caused biases. It was assumed that detectability of birds did not vary across different habitat categories, as birds were actively vocalising during the breeding period, even if they were not easily visible.

2.4. Data analysis

Bird species records were totalled over the two temporal replicates for each site and a species-site matrix was created, summarizing the abundance of each species in each of the 198 study sites. A separate species-site matrix was created for each of Landscapes A and B. Species data from the two landscapes were analysed separately as they were from two different years and were surveyed using variable number of spatial replicates (23 in 2016 and 10 in 2017). Rarefaction analysis was carried out to find out if there had been adequate sampling in each of the two study landscapes following Colwell et al. (1994).

Species richness in each land use type was estimated using rarefaction analysis (using the bootstrapping estimator), with sites as replicates, following Colwell et al. (1994). This method allows for the fact that the species-effort curve for a given site may not have levelled off completely during the study; species may remain to be discovered. Species estimates using rarefaction are generally higher than the observed species richness due to this reason. Since Landscape B appeared to be inadequately sampled based on the rarefaction curves (see Results), mean species richness per site (cumulative number of species sighted during the two replicate counts) was also compared across land use categories using an estimate of variation (standard error). Since the time and effort per count was standardized, species richness per site can be considered robust estimates of species richness, in the absence of incomplete sampling of bird species.

In order to assess differences in species composition among land use types, we used non-metric multidimensional scaling (NMDS) following Clarke and Green (1988). The stress value for each NMDS was calculated in order to assess its usefulness, with low values being preferred. In NMDS graphs, each ellipse (with its centroid) corresponds to a given land use category and the distance between any two centroids signify degree of dissimilarity in species composition between them. The size of each ellipse shows the degree of dissimilarity among sites within a given land use category. We then quantified compositional divergence using Permutational Analysis of Variance or PERMANOVA, using both abundance-based Bray-Curtis and incidence-based Jaccard indices (Anderson et al., 2006). The two indices allowed us to discern whether patterns in species richness were being driven by species relative abundances or just presence-absence of species. Statistical significance of among-group similarity was assessed by permuting the matrices 999 times and generating the P-value for significance. An alpha of 0.05 (p-value) was used as the threshold to assess significance of the PERMANOVA tests for species composition. We conducted all community analyses for diversity and species composition using packages 'hierDiversity' (Marion et al., 2015) and 'vegan' in R Software version 3.3.0 (Oksanen et al., 2011).

In order to understand the relative extinction vulnerability of forest bird species based on habitat guilds, we assigned a category of habitat preference to each species based on two widely used field guides (Ali, 2012; Grimmer et al., 1998). A species was assigned to one of the following guilds: oak forest specialist (seen primarily in dense hardwood forest), forest generalist (species using secondary forest and horticulture in addition to dense hardwood forest), open country species (species seen primarily in cultivation, grasslands and other open areas) and commensal species (largely found in and around human settlements). Appendix D shows the assignment of guilds to each of the detected bird species. It should be noted that habitat preferences of bird species in our field site, may be different from those that seen at other altitudes or vegetation zones within the Himalayas, particularly during non-breeding season. We then used histograms to visually compare the total abundance of species belonging to each habitat guild, combined over the two landscapes.

We assigned ecosystem service categories to bird species based on their documented feeding habits and behaviour, corroborated by our field observations (Ali, 2012; Grimmer et al., 1998). The pollinator category comprised of species that had been observed either feeding on flower nectar, or browsing insects around flowers, thus making them likely to pollinate flowers. The pest controller category comprised of all insectivorous birds, irrespective of whether they opportunistically fed on other items as well (such as fruit or nectar), since such birds would be active in pest control. The category of ecosystem engineer comprised of all species known to construct cavities for nesting and roosting, such as barbets and woodpeckers. Functional guild assignments are also given in Appendix D. Abundances of each functional guild were then compared across land use categories using histograms. Abundance of both habitat guilds and functional guilds were analysed separately for Landscapes A and B.

3. Results

During the study period, we recorded a total of 8549 bird observations belonging to 124 species, over the 198 study sites in the two landscapes. In Landscape A, we recorded a total of 110 species and in Landscape B, 98 species were recorded during the bird surveys. Species abundances are given in Appendix E and Appendix F, corresponding to the two landscapes.

Rarefaction analysis for the two sites showed that the species richness for Landscape A had levelled off with 23 replicates in each land use, but that sampling effort had not been sufficient for all land use types in Landscape B (Fig. 4a and b).

The species estimates drawn from the bootstrapping analysis showed that in Landscape A, natural oak had the highest estimated number of bird species amongst all the land use types (78.8 ± 2.51 ; Fig. 4a). Degraded oak, cultivation and lopped oak showed estimated species richness lower than that seen in natural oak (Degraded oak: 72.9 ± 2.81 ; Cultivation: 67.4 ± 2.59 ; and lopped oak: 68.4 ± 3.32 species) representing between 85 and 92% of the species in natural oak. Pine forest and built-up sites had lowest species richness (pine forest: 56.6 ± 2.8 and 57.4 ± 2.93 respectively), which was approximately 72% of the species richness in natural oak.

In Landscape B, degraded forests showed similar species estimate as the natural oak, while lopped oak and cultivation recorded 66–79% of species richness in natural oak. (Natural Oak: 61.3 ± 2.53 ; Degraded oak: 67.2 ± 6.14 ; Cultivation:

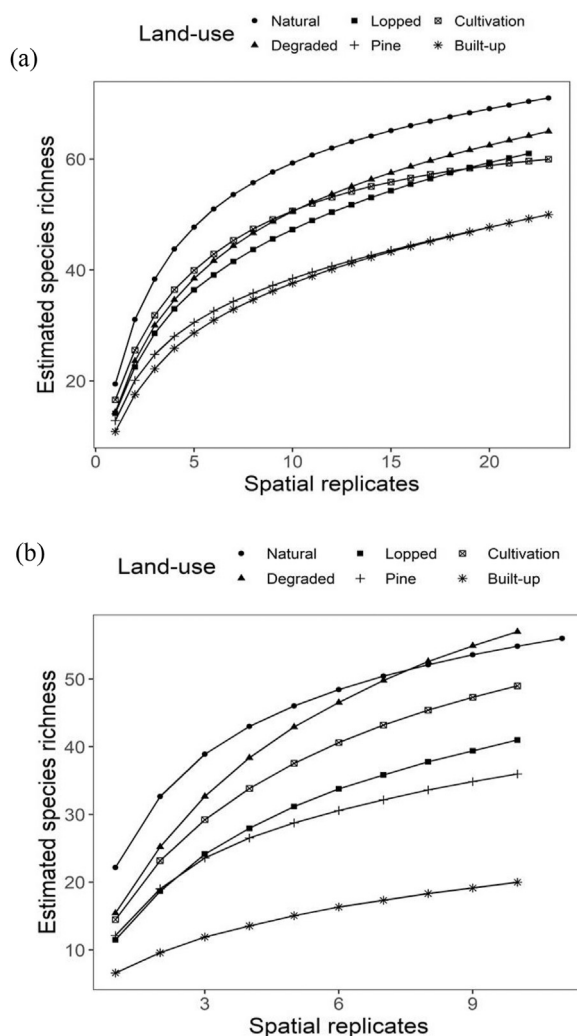


Fig. 4. Rarefaction analysis to see the extent of sampling of bird richness in (a) Landscape A, based on 23 spatial replicates per land use and (b) Landscape B based on 10 spatial replicates per land use. (Natural: Natural Oak Forest; Degraded: Degraded Oak Forest; Lopped: Lopped Oak Forest; Pine: Pine Forest; Built-up: Built-up Sites).

56.8 ± 3.8 ; and lopped oak: 47.7 ± 4.14 species). Pine forest recorded 66% of natural oak richness and built up sites recorded only 36% of the species richness of natural oak (Pine forest: 40.5 ± 2.94 ; Built-up Sites: 23.6 ± 2.96). In both Landscapes A and B, pine forest and built-up sites had the lowest numbers of bird species among all land uses.

Fig. 5 shows the average species number per site in the two landscapes. In Landscape A, natural oak showed the highest value, followed by the modified forest categories and cultivation sites which recorded 70–75% of the species found in natural oak. Built-up sites recorded only 48% of the richness recorded in natural oak. In Landscape B, too, natural oak showed the highest value, but the modified forest categories and cultivation recorded only 50% of the species richness of natural oak (Figs. 4 and 5). Cultivation sites were on par with modified forest categories in Landscape B while built-up sites recorded a meagre 25% of natural oak species richness. In both landscapes, however, the rank order of species-per-site across land use categories was similar.

Rank order of species-per-site values was similar to that found in the bootstrapped species estimates (Fig. 4a and b), with the highest values in natural oak, moderate values in the modified forest categories and the least in built-up sites. Cultivation showed relatively higher richness compared to the modified forest categories in both cases, but still less than natural oak.

The NMDS for both landscapes, shows that bird species composition diverged considerably among forest and non-forest land uses as shown by the inter-centroid distances (Fig. 6a and b). The stress levels for NMDS in both landscapes were low, showing the value of the analysis (Landscape A: 0.17; Landscape B: 0.13). PERMANOVA analysis confirmed that all land uses were significantly different from each other in terms of species composition. This was shown by both the Bray-Curtis index (Landscape A: $F = 14.58$, $p < 0.001$; Landscape B: $F = 10.34$, $p < 0.01$) and the Jaccard Index (Landscape A: $F = 8.53$, $p < 0.001$;

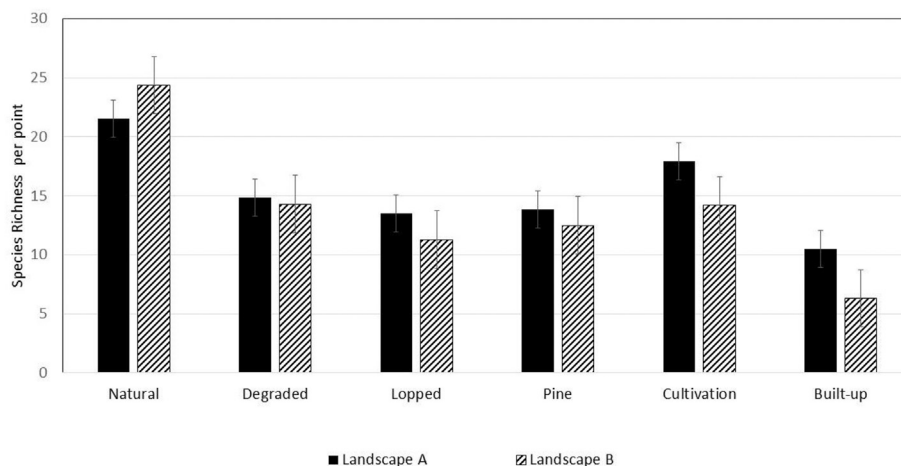


Fig. 5. Average species richness-per-site for each land use type for Landscape A (based on 23 study sites per land use) and for Landscape B (based on 10 study sites per land use). Species richness per site is cumulative over two temporal replicates at each site. (Natural: Natural Oak Forest; Degraded: Degraded Oak Forest; Lopped: Lopped Oak Forest; Pine: Pine Forest; Built-up: Built-up Sites).

Landscape B: $F = 6.22$, $p < 0.01$). Our results thus show that both species number and relative abundances are driving the observed significant changes in species composition across all the land use categories.

Within the forested land uses, natural oak was most similar to degraded oak and least similar to pine forest, as seen by the degree of overlap of the ellipses corresponding to each land use and the inter-centroid distances (Fig. 6a and b). This was true for both landscapes. The most dissimilar sites were natural oak and built-up sites. Cultivated sites showed highest similarity with built-up sites in Landscape A but were closer to pine forest in Landscape B.

The assignment of habitat preference category, i.e., oak forest specialists, forest generalists, open country species and commensal species in the bird community is given in Appendix D. The abundance of oak forest specialists declined sharply (by about 75%) from natural oak forest to modified land uses as seen in data from both Landscapes A and B (Fig. 7a and b). Abundance of forest generalist species was much more constant across the land use categories in both landscapes, but showed a steep drop in built-up sites (Fig. 7a and b). Further, open country species abundances increased along the anthropogenic intensification gradient, with the maximum being seen in cultivation sites. There were no commensal species in any of the four forest categories, but were reported in cultivation and built-up sites (Fig. 7a and b).

Fig. 8 shows the abundance of the three functional guilds that were compared across the six land use categories. Insectivores and pollinators were most abundant in natural oak forest and sharply declined in all other land use categories, by almost 50%. Ecosystem engineers do not show a clear pattern, except that they were absent in built-up sites and showed extremely low numbers in cultivation. In this case, abundance in pine forest was seen to be as high as in natural oak forest, but cultivation and built-up sites showed severe depletion.

4. Discussion

Our study corroborates patterns seen in global meta-analyses: that land use change, caused by conversion of natural forests to urban centres, monocultures, cultivation and even managed forests, adversely affects species richness, and significantly alters species composition. Further, both habitat preference guilds and functional guilds are adversely affected, with the exception of ecosystem engineers, which may have been too low in numbers for a valid analysis (Gardner et al., 2009; Sol et al., 2020). Our results thus suggest that it is as important to look at abundances of functional guilds, rather than only species richness of avian communities in altered ecosystems, because ecosystem services depend on healthy populations of service providers (Young et al., 2016; Maas et al., 2013; Whelan et al., 2015). Our results have significant implications for conservation planning in the increasingly human-dominated forest landscapes of developing countries (Sol et al., 2020; Newbold et al., 2020).

Our NMDS results suggest significant divergence of forested and non-forested land uses in species composition as well as relative abundances of species. Low stress values underline the significance of our findings. Our results indicate that the divergence in species composition was caused primarily by the changes in proportion of species belonging to different habitat preference guilds and functional guilds, as well as their abundances (Menon et al., 2019). Many of the species dropping out of modified land uses are recognised oak forest specialists such as rufous-bellied woodpecker (*Dendrocopos hyperythrus*), greater yellowthroat (*Chrysophlegma flavinucha*), rufous sibia (*Heterophasia capistrata*), white-throated laughing thrush (*Garrulax albogularis*) and black-faced warbler (*Abroscopus schisticeps*) (Fig. 5; Appendix D). Forest specialists are recognised to show greater vulnerability to land use change which likely occurs due to alterations in food and nesting resources, and competitive relationships among species comprising the regional species pool (Shochat et al., 2006; Kumar et al., 2011;

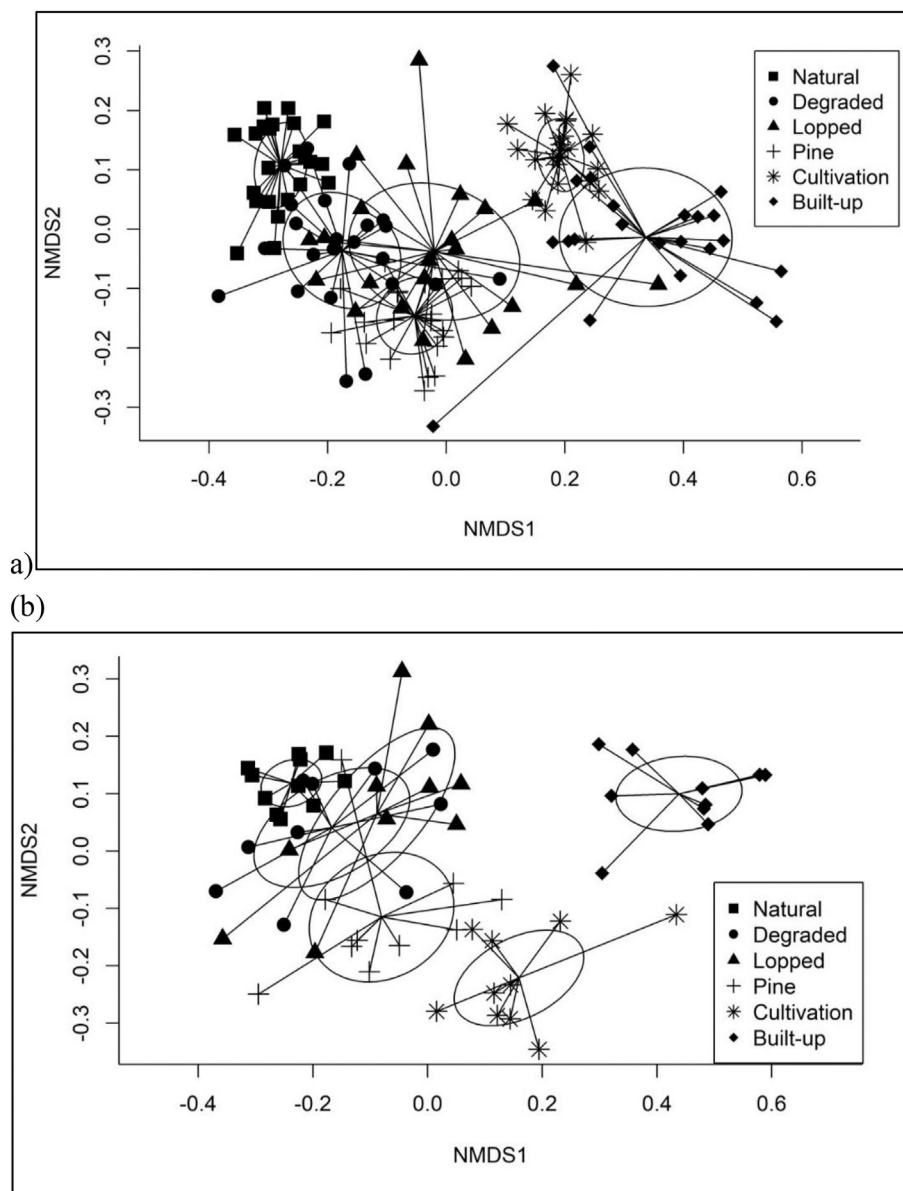


Fig. 6. Non-metric Dimensional Scaling (NMDS) plots showing degree of divergence in bird species composition among land use categories in (a) Landscape A and (b) Landscape B (below). The size of each ellipse is proportional to within-group similarity of the given land use and the degree of overlap between any two ellipses shows the degree of community similarity between the respective land uses.

Menon et al., 2019). Forest generalists are less affected by changing land use, because they are opportunistic in using resources from a wider range of habitat types. The much more uniform pattern of abundance in forest generalists across land use categories (apart from built-up sites) in our own study is striking (Fig. 7a and b), and corroborates the above inference. Alteration in species composition is also likely due to the accompanying drastic declines in abundance of pollinators and insectivores, pointing to uneven effects of land use change on different functional guilds, a pattern that has been seen globally as well (see Newbold et al., 2020 for a recent review).

Globally, the replacement of natural forest with monocultures of tree species has been an important cause of biodiversity loss in different parts of the world (e.g. Srinivas and Koh, 2016; Calvino-Cancela, 2013; Mandal and Shankar-Raman, 2016). We find very similar patterns with respect to the chir pine forest stands in our study site, so that stands that are managed as monocultures show poor ability to host bird species. This was found to be particularly true for forest specialists, pollinators and insectivores. It is important to note that this finding may not be completely generalizable to all types of monocultures. The poor species richness of birds in Himalayan chir pine stands may be ascribed to the relatively open canopy and low tree density, as well as poor plant diversity in our study landscape. However, such features may not be reflective of all other types

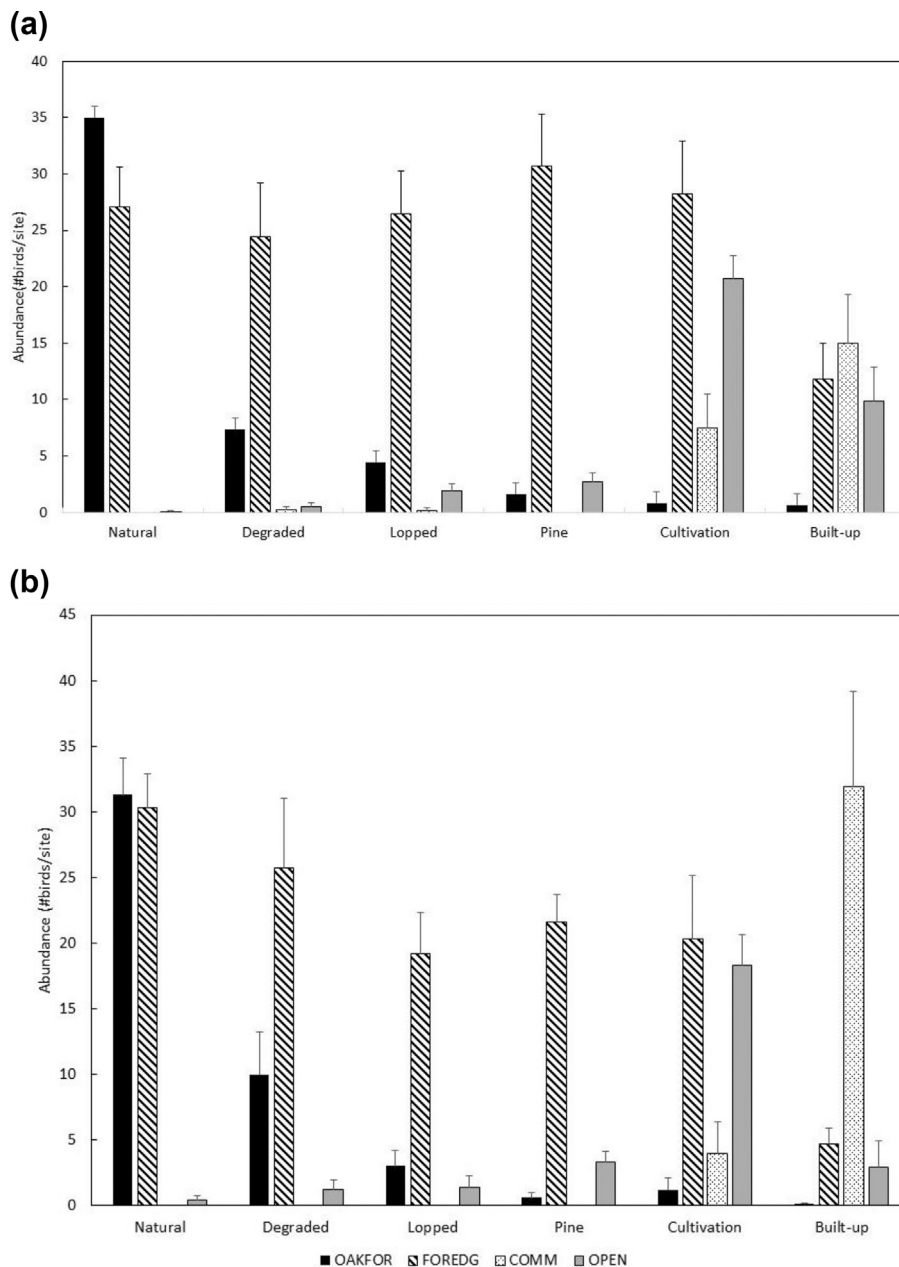


Fig. 7. Abundance of the four habitat guilds of birds in each of the six land use categories in (a) Landscape A and (b) Landscape B. Natural: Natural Oak Forest; Degraded: Degraded Oak Forest; Lopped: Lopped Oak Forest; Pine: Pine Forest; Built-up: Built-up (Urban) Sites. (OAKFOR: Oak Forest Specialists; FOREDG: Forest generalists; COMM: Commensals; OPEN: Open Area Generalists).

of monocultures (see also Kumar et al., 2011; Menon et al., 2019). Our findings have serious implications for Western Himalayan biodiversity as both scientists and locals believe that pine stands are expanding in area at the cost of diverse hardwood stands at middle elevations (Shahabuddin and Thadani, 2018; Naudiyal and Schmerbeck, 2017).

Cultivation sites differed substantially in composition from all the forest land uses, being most similar to built-up sites. We found that cultivation sites provide breeding habitats for open country species such as grey bushchat, black francolin, russet sparrow, streaked laughing thrush (*Trochaloxyron lineatus*) and Himalayan bulbul (*Pycnonotus leucogenis*), which are not seen in any forest land use. In addition, a large number of forest generalists such as blue whistling thrush and red-billed blue magpie were also recorded in cultivation sites. However, cultivation sites are not able to support forest specialists, even whilst providing considerable space for forest generalists, commensals and open country species (see Fig. 5a and b). Our results thus suggest that forest species are largely not able to survive in cultivation areas, as has been seen elsewhere (Phalan et al., 2011;

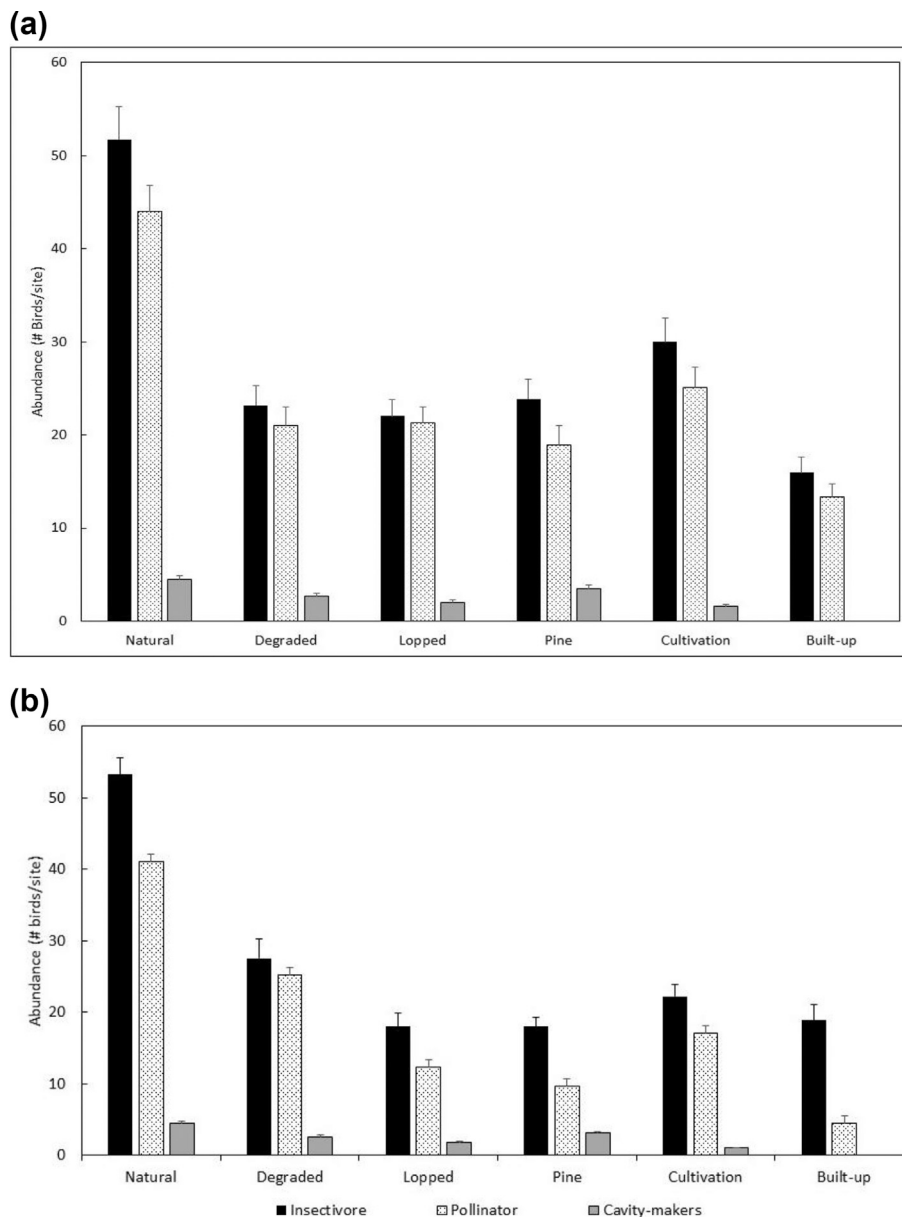


Fig. 8. Abundance of functional guilds of birds compared across the six land use categories in (a) Landscape A and (b) Landscape B. Natural: Natural Oak Forest; Degraded: Degraded Oak Forest; Lopped: Lopped Oak Forest; Pine: Pine Forest; Built-up: Built-up or Urban Sites.

De Lima et al., 2013). At the same time, cultivation sites show higher species richness, as well as similar abundances of pollinators and insectivores in comparison to the modified oak forests and pine, indicating that cultivation may help sustain significant ecosystem providers (although the species involved may be different). Elsen et al. (2017) found that cultivation sites harboured a rich diversity of high-altitude species in the Western Himalaya in the winter season. Our study, however, shows that it is essential to explore species composition as much as species richness, as our results suggest a significantly different avian community in cultivation in comparison to modified forests and monocultures.

Urbanization has been discussed as one of the long-term causes of species declines globally, a factor that is likely to become more important with time (Sol et al., 2020; Carvajal-Castro et al., 2019; Shochat et al., 2006; Tiwari and Urfi, 2016). In accordance with such predictions, our built-up sites were found to be the most depauperate in bird species and abundance among all the six land uses in our study area. In the case of Landscape B, species richness was lower than that in natural oak by almost 75%. This was true for both of our study landscapes. Further, all three functional groups were least abundant (or absent) in this land use category, and the avifauna was dominated by open country and commensal species with very few forest species (either specialists or generalists). Keeping in mind possible landscape effects on bird distribution that have been

noted in the same landscape (see Menon et al., 2019), it is clear that growing expansion of urban and suburban land use into natural habitats, as is happening globally, will likely lead to highly impoverished and homogeneous fauna over the long-term. It is necessary to come up with plans to improve biodiversity values of urban habitats and reduce their impacts on adjacent wild habitats (for example see Sol et al., 2020).

Our study also speaks to the effect of biomass extraction on avifauna, showing that lightly used forest (degraded category; Fig. 2b) can still harbour a large proportion (50–75%) of forest species (see also Menon et al., 2019). However, if avian forest specialists are considered, the degraded forest category harboured only 25% or less of bird abundance in comparison to natural oak, although it had comparable overall species richness (Fig. 7a and b). Similarly, degraded forests harboured 50% or less of pollinators and insectivores in terms of abundance (Fig. 8a and b). The above patterns in guild abundance were seen in both Landscape A and Landscape B, suggesting that degraded forests represent only suboptimal habitat for forest specialists, pollinators and insectivores. Yet, since degraded forests appear to provide opportunistic foraging areas for a high proportion of forest bird species, they could possibly be used as avifaunal refugia, as long as remnants of natural forest are retained in the landscapes to provide source populations (see also Menon et al., 2019 and Shahabuddin and Kumar, 2006). This prediction should be tested in future studies.

Due to lack of sufficient quantitative data on forest structure, we were unable to relate the abundance of species richness or guild abundance to specific structural or compositional changes in the used forest sites. However, the differences in extractive activities such as leaf litter collection, and tree-lopping in the three forest categories (Appendix A) are likely to cause significant vegetational changes which in turn, may affect avian communities. Specifically, the forest specialist, pollinator and insectivore guilds are likely to respond strongly to forest changes brought about by various use intensities (see Menon et al., 2019). Our study clearly shows that degraded forests may be superior to lopped forests in their richness as well as ability to host forest specialists, so the use intensity needs to be controlled if bird diversity is to be maintained (Shahabuddin and Kumar, 2006). Further studies are required to better understand bird-vegetation relationships in multiple use forests which can also lead to more focussed management recommendations.

Our study also finds that ecosystem service-providers among birds are significantly depleted in all other land use types in comparison to natural oak forest. Such patterns concur with those detected in global reviews of field studies by Newbold et al. (2020) and Sol et al. (2020) who found significant restructuring of functional groups at large scales. It is possible that our data on ecosystem engineers is not sufficient for discerning clear patterns as woodpeckers generally occur at low densities in forests (see Fig. 6). Yet, our data indicates that cultivation sites and built-up sites are depleted in terms of ecosystem engineers. Depletion of pollinators and insectivores can have significant repercussions for provision of ecosystem functions in production landscapes such as agriculture and monocultures (Sekercioglu 2006). It is necessary to carry out focussed studies on the functioning of particular groups of ecosystem providers to study the impact on ecosystem functions in our study landscapes (eg. Maas et al., 2013).

5. Conclusion

Our study shows that there is significant avifaunal and guild impoverishment due to land use change from natural forests to other forms of land use. Lightly used forests retain a large proportion of forest avifauna but urbanization, intensive use and pine monocultures show drastic adverse effects on bird diversity and guild composition. The impact of cultivation is not so severe in terms of species richness but it creates highly altered bird assemblages during breeding season. Species composition is significantly impacted by land use change as well, with modified oak forest categories showing much more similarity with natural oak, than do pine, urban or cultivation sites. In terms of abundance, forest specialists, pollinators and insectivores are affected more by land use change than other habitat guilds such as commensals, open country species and forest generalists, which are better able to survive in modified landscapes. Such observed declines in certain functional guilds could affect bird-provided ecosystem services both in forests and in adjoining production landscapes. Thus our results show that it may be as important to take abundances into account, as much as species loss itself when studying patterns of defaunation (Young et al., 2016).

A number of scholars have predicted biodiversity declines in the Himalayas due to recent developmental pressures (e.g. Pandit et al., 2007, 2014) but our study is among the few that quantify these changes based on systematic field studies in a diverse range of land uses (but see Srinivasan et al., 2015; Elsen et al., 2017). We recommend safeguarding of Himalayan oak forests and their avifaunal diversity by limiting extraction of forest produce to sustainable levels, curtailing expansion of pine monocultures, and retaining well-protected natural forest stands in the mosaic of natural and modified land uses.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.gecco.2020.e01447>.

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