

# Effects of extractive use on forest birds in Western Himalayas: Role of local and landscape factors

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## ABSTRACT

Rural populations in developing countries depend on forests for fuelwood, fodder and other non-timber forest products. Such dependence has significant effects on faunal diversity and composition, primarily due to alteration of vegetation structure. While earlier studies have assessed the consequences of extractive activities for forest fauna, very few simultaneously acknowledge the role of landscape context, which includes agricultural land-uses and managed forests. This study was carried out in oak-dominated forests located in the middle elevations of the Western Himalayas to investigate effects of extractive activities on avifaunal communities, taking into account the role of landscape composition. Birds were sampled at 74 sites, six times each, over two seasons using fixed-radius point counts. We analysed avian species richness, abundance, composition and guild abundances (based on foraging and habitat preference) as a function of vegetation structure and proportion of dense oak forest around each site. Our results suggest that extractive activities in forests can result in reduced species richness, abundance and altered composition of forest avifauna, brought about by changes in vegetation structure. We also find that these effects may be intensified by the loss of dense oak forests in the landscape. Insectivorous birds and forest specialists were most sensitive to extractive activities and landscape composition. Decreasing canopy cover and proportion of oak forest in the landscape had strong negative effects on insectivorous and forest specialist species. Other local factors such as vertical stratification and understory density also had effects on these avian guilds. Our study indicates that in order to conserve bird species in an increasingly human-dominated landscape of the Western Himalayas, it may be necessary to control extractive activities that affect forest structure as well as retain protected forest stands within a continuum of harvested forest and agricultural landscapes.

## 1. Introduction

Biomass extraction from forests, in the form of grazing, firewood extraction, fodder collection and other Non-Timber Forest Produce (NTFP) extraction, puts considerable and widespread pressure on forests in developing countries (Shahabuddin and Kumar, 2007; Borghesio, 2008). Such extractive pressures can cause changes in forest vegetation structure and composition, with concomitant effects on forest flora and fauna including birds, mammals and arthropods (Hansen et al., 1995; Chazdon, 2003). Yet there are still very few studies specifically exploring the impacts of low-intensity extractive activities on forests in developing countries.

In addition to local or site level factors related to habitat structure,

landscape factors may be required to adequately predict species responses (Lichstein et al., 2002; Warren et al., 2005; Dahal et al., 2015). For instance, landscape-scale topographic and climatic variables have been shown to significantly influence plant, invertebrate and bird community structure (Svenning, 1999; Harms et al., 2001). In Canadian boreal forests, landscape factors were found to be as important as local habitat conditions in determining bird community composition (Drapeau et al., 2000). On the other hand, some studies suggest that in certain cases local environmental variables may be of greater importance than landscape-scale variables (Schmiegelow et al., 1997; Dahal et al., 2015; Touihri et al., 2017). It is thus important to simultaneously assess the relative importance of landscape factors and local habitat factors in influencing the diversity of various faunal taxa

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in harvested forests. Such an understanding may also have important implications for land-use planning and forest management that aim to maximise biodiversity conservation in multiple-use landscapes (Mazerolle and Villard, 1999; Cleary et al., 2005). Our study aims to understand the influences of both local and landscape level features in determining bird-habitat relationships in a human-dominated landscape consisting of both protected and harvested forests.

Bird communities are known to be especially sensitive to changes in vegetation structure and composition caused by biomass extraction, which is evident from the vast literature on bird-habitat relationships (Hansen et al., 1995; Imbeau and Desrochers, 2002; Chettri et al., 2005; Jayapal et al., 2009; Fuller, 2012). Most of these studies find that forest degradation significantly reduces bird species richness and abundance while also altering species composition as a whole.

As it is known that certain guilds tend to be more vulnerable than others (O'Connell et al., 2000), grouping species with shared traits into guilds can facilitate generalisations regarding the effects of landscape composition and vegetation structure on bird communities. Considering foraging guilds, we expect insectivores to be more vulnerable than other guilds due to their dependence on insects, which in turn, require specific regimes of micro-climate and specific sets of vegetation structural attributes (Thiollay, 1997; Sekercioglu et al., 2002; Silveira et al., 2010). Frugivores on the other hand are expected to be less affected by local forest structure since they are more wide-ranging, having the ability to locate resource-rich patches in a heterogeneous landscape (Canaday, 1996; Moran and Catterall, 2014; Morante-Filho et al., 2015). Granivores and omnivores may not be affected significantly by anthropogenic disturbance as they adapt well to resources present around cultivation and human settlements (Thiollay, 1992; Watson et al., 2004; Schulze and Riedl, 2008). When looking at guilds based on habitat specialisation, forest specialists tend to be more affected by biomass extraction than forest generalists. This is because extractive activities create edge-like open habitat within dense forests, thus favouring generalist species which tend to out compete forest-specialists under such conditions (Datta, 1998; Borghesio, 2008). Both open area/agricultural species and human commensals are largely unaffected by forest loss as they are adapted to living around agricultural fields and human settlements (Elsen et al., 2017). Thus, of all guilds, insectivorous guilds and forest specialists, are expected to be more vulnerable to extractive activities within harvested forests in comparison to frugivores, granivores, omnivores, open cultivation species, generalist forest species and commensals.

The Western Himalayan temperate forest biome is faunally diverse and an Endemic Bird Area due to a concentration of restricted-range avifauna (Birdlife International, 2016a; 2016b). Occurring in the elevational range of 1500–2200 m, old growth broadleaved forests dominated by *Quercus leucotrichophora* (or locally known as banj oak) are known to support high bird diversity, including a large proportion of hardwood specialists (Shahabuddin et al., 2017). These forests are known to be critical for hydrological balance of the region while also acting as a livelihood support system for local inhabitants who are primarily dependent on agriculture and animal husbandry (Singh and Singh, 1987; Naudiyal and Schmerbeck, 2017).

Oak forests in the middle Himalayas are managed as state-owned Reserved Forests, private forests or community/village forests. Due to varying land tenure in the region, forest stands experience varying intensities of biomass extraction. Entry into Reserved Forests is restricted under the Indian Forest Act 1927, and therefore they tend to be better protected. Most private forests in the area are part of government institutions which have similar restrictions as Reserved Forests. Community-managed forests in this region (also known as Van Panchayat forests) is a form of forest governance unique to Uttarakhand state in India, where government and local village committees co-manage forests under agreements for sustainable use (Shahabuddin and Thadani, 2018). Community-managed oak forests form the mainstay of rural livelihoods and therefore support a range of human activities such

as livestock grazing, fuelwood extraction, lopping of branches for fodder, collection of leaf litter for compost and several other commercial forest products. Thus, such community-managed forests experience a range of extraction intensities (Makino, 2011; Singh et al., 2014). An increasing population (of both tourists and locals) has put an increased demand on these forest resources during the last few decades. Apart from extractive use, infrastructural development, tourism activities and range expansion of native chir pine (*Pinus roxburghii*) also threaten the structure and function of the oak forests in this region (Singh and Singh, 1986; Shahabuddin and Thadani, 2018). The expansion of chir pine is thought to be caused by anthropogenic factors like intense lopping in oak forests, warming events and forest fires, which create conducive conditions for pine regeneration in parallel with the suppression of oak saplings (Singh and Singh, 1986; Naudiyal and Schmerbeck, 2017). Currently, the landscape is composed of oak forest patches (both protected and harvested) separated by cultivation, horticulture, chir pine stands, water bodies and built-up areas.

The broad goal of this study is to quantify the effects of changes in forest structure, caused by rural extractive activities, as well as landscape-level factors on avian communities in these forests. The specific objectives of this study were (1) to investigate effects of extractive activities on forest vegetation structure (2) to quantify relative effects of altered vegetation structure and landscape composition on bird species richness, abundance and community composition, and (3) to examine how different avian guilds (based on foraging and habitat preferences) respond to changes in vegetation structure and landscape composition.

## 2. Methodology

### 2.1. Study area

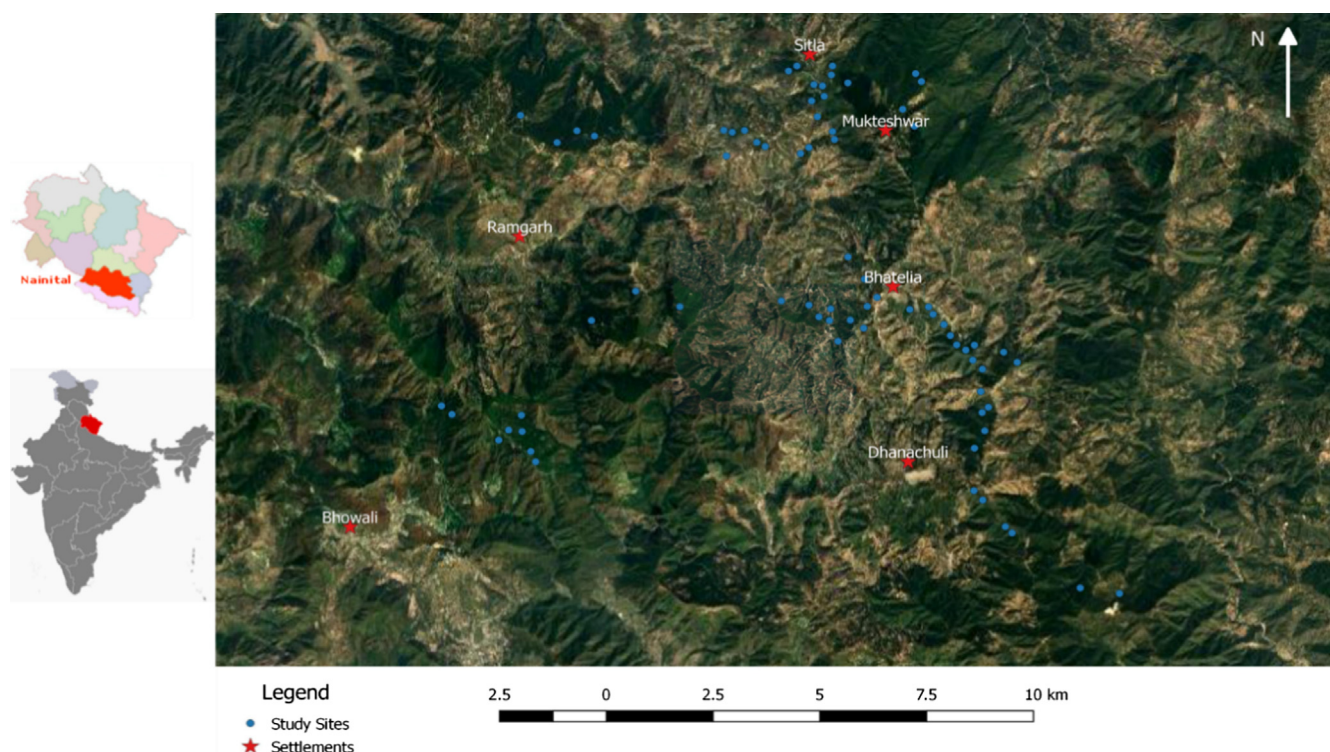
The study was carried out in the north-western state of Uttarakhand (straddling Nainital and Almora districts of Kumaon) in the middle elevation zone (1700–2200 m) of the Western Himalayas (Fig. 1). Classified as Himalayan wet/moist temperate forests (Champion and Seth, 1968), they are dominated by banj oak and chir pine (*Quercus leucotrichophora* – *Pinus roxburghii* association) mixed with rhododendron (*Rhododendron arboreum*), chestnut (*Aesculus indica*), alder (*Alnus nepalensis*) and other deciduous species. Monodominant oak forests form the climax community in the Western Himalayas at this elevational range (Singh and Singh, 1986). Banj oak occurs as the dominant species in hardwood forest stands at this altitude, with tilonj oak (*Quercus floribunda*) replacing it at 2200–2400 m (Singh and Singh, 1986).

### 2.2. Selection of sites

Using a combination of local knowledge and high-resolution satellite imagery (Google Earth), a study area of 400 km<sup>2</sup> was identified to include adequate representation of banj oak forest experiencing varying levels of protection, chir pine forest and adjoining human use landscapes (Fig. 1). Within the study area, 74 study sites were selected in forest stands subject to varying degrees of extraction: from well-protected old-growth to intensively harvested forests. Harvested forests tended to have more open canopy, smaller and shorter trees, and sparser understory. Protected forest stands had taller and larger trees along with a dense canopy and understory. Care was taken to maintain at least 200 m between sites to avoid double counting of individual birds. Sites were located at least 50 m from edges of the forest stands to reduce magnitude of edge effects.

### 2.3. Bird counts

Fixed-radius point counts of 30 m radius (Hutto et al., 1986; Bibby et al., 2000) were used to survey birds at each study site during two



**Fig. 1.** Map of study area in the Nainital district of Uttarakhand, India showing locations of study sites (blue circles). Study sites were distributed across oak forests in the region to encompass a gradient of anthropogenic disturbances and proportions of dense oak forest in the surrounding landscape. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

seasons- winter (27th November 2017 to 20th February 2018) and spring-summer (1st March 2018 to 26th April 2018). Every site was sampled three times during each season at intervals of 18–20 days. Visibility may not have been the same across all sites and hence may have created biases of imperfect detection in denser forest. In order to test for uneven visibility, pilot studies were carried out using point counts at nine sites in both dense and relatively open forests. The results showed that 79% of detections were aural. At sites in dense forests, where detectability is likely to be poorest, most visual detections were under 30 m and aural detections started tapering off between 30 and 40 m. Therefore, we used a 30 m fixed radius for our point counts. Point counts were carried out jointly by two observers for 15 min on clear days during the first four hours post sunrise when bird activity was maximum. At each point count there was one primary observer who identified and counted all birds observed and one secondary observer who did the same in addition to writing down observations. These two observers as well as their roles remained constant throughout the study, thus controlling for any observer-caused effects that may have occurred otherwise. All birds seen or heard in the 15-minute period were counted. Birds seen flying above the canopy were not included in the count. Bird species were classified in two types of guilds : (1) based on primary diet (insectivore, frugivore, omnivore, nectarivore, granivore) and (2) habitat preference (forest specialists, forest generalists, open-area/agricultural species and commensal species) using secondary data sources (Grimmett et al., 1998; Rasmussen and Anderton, 2005; Ali and Ripley, 1987; Shahabuddin, Goswami, and Gupta, 2017) (refer Appendix A: Table A1).

#### 2.4. Habitat and vegetation structure

To characterise vegetation structure, three 100 m<sup>2</sup> circular plots (a radius of 5.64 m) were established within a 30 m radius of each study site. Each of the three 100 m<sup>2</sup> plots was established 15 m from the centre of the point count and at an angle of 120° from each other. All

vegetation variables were measured at the level of the 100 m<sup>2</sup> plot and then averaged across the three plots to obtain a single value for each site, thus accounting for variability in vegetation structure within each site. Within each plot of 100 m<sup>2</sup>, two kinds of vegetation measurements were made: (1) signs of extractive use (signs of lopping and leaf litter removal); (2) vegetation structural variables (tree density, tree GBH (Girth at Breast Height), tree height, understory density (stems < 1.5 m height), understory density (stems > 1.5 m in height), canopy cover and vertical stratification).

##### 2.4.1. Signs of extractive use

Two variables related to extractive use were measured within each plot: tree-logging intensity and leaf litter removal. To measure lopping intensity, each tree within a plot was scored as 1 (no lopping), 2 (moderate lopping) and 3 (extreme lopping); a median lopping score was derived for each of the three plots. The median of these measurements in each plot was used to get a lopping score for each study site. Degree of leaf litter removal was measured at the centre of each plot on a scale between 1 and 3, where 1 = presence of dry leaves along with a well-developed and deep duff or humus layer; 2 = presence of a thin layer of dry leaves indicating one year's litter; and 3 = bare ground with little or no leaf litter, and a median leaf litter removal score was derived for each site.

##### 2.4.2. Vegetation structure

Within each plot, all trees with a GBH greater than 31.4 cm (i.e. Diameter at Breast Height > 10 cm) were identified, counted and their GBH recorded. Basal area, tree density and average GBH (mean tree girth) for each plot were calculated from this data. Canopy cover was measured at the centre of each plot, using a canopy densitometer. One reading was taken in each of the four cardinal directions and averaged to get a reading for canopy density per plot. The height of each tree was estimated by calibrating the height of one of the observers, who would stand next to the base of the tree, while the other observer estimated the



tree height, approximating it as a multiple of the observer's height. It was not possible to use a clinometer for measuring tree heights due to the steep terrain and dense vegetation in most sites. The height of the tallest tree in each plot was averaged across the three plots to get a single value of maximum canopy height per site. Understory density was defined as total number of stems of saplings, shrubs and bushes that had a minimum height of 0.5 m and GBH  $\leq$  31.4 cm. Understory density was estimated within a circular sub-plot of 2.55 m radius (20 m<sup>2</sup> in area) which was nested within each of the three plots at a site. Stems in the understory were further divided into two height categories (0.5–1.5 m and  $>$  1.5 m) and numbers in each category were counted. Vertical stratification was calculated using Shannon's diversity index based on presence or absence of vegetation at different height intervals (0–1, 1–2, 2–3, 3–4, 4–6, 6–8, 8–12, 12–16,  $>$  16 m) directly above and within a 50 cm radius of a given point, as indicated by position of a 6 m pole (Daniels et al., 1992; Shahabuddin and Kumar, 2007). Measurements for vertical stratification were made in 12 locations per site (4 per plot).

## 2.5. Landscape composition

Multispectral Landsat 8 satellite imagery was used to classify the study area into six land cover classes that represent the major land uses in the study area. These land uses included dense (protected) oak forests, lopped (heavily degraded) forest, pine forest, agricultural land, built up area and water bodies. Supervised classification using the Spectral-Angle Mapper algorithm was carried out using the Semi-Automatic Classification Plugin (SCP) in Quantum GIS version 2.18.1.

A total of 295 ground control points (GCP) were collected from across the study area using a Garmin Etrex 20 GPS. Since the imagery had a minimum resolution (pixel size) of 30 m, care was taken to select GCPs within land cover types that were homogenous within an area covered by 30 m radius at the minimum. The 295 GCPs were then split into a 70:30 ratio, with 70% (206) being used to train the image and 30% (89) being used to test accuracy of the image. The training of the image was done by a combination of both GCPs and by visual assessment using various multispectral band combinations that highlight different land cover classes. Land cover classification resulted in an overall accuracy of 82% with a kappa hat value of 0.77 which can be considered satisfactory for the purposes of this study (Foody, 2008). Classification of dense oak forests had a producer accuracy of 95%, a user accuracy of 86.36% and kappa hat of 0.79. This suggests that the spectral signatures of dense oak forests were sufficiently different from other landscape classes (Foody, 2008). The classified image (Appendix A: Fig. A1) was then used to calculate proportion of area covered by dense oak forest within 500 m and 200 m radius of each point count location (hereafter referred to as “forest proportion”) using the LECOS (Landscape Ecology Statistics) plugin in QGIS version 2.18.1.

## 3. Analysis

### 3.1. Effects of extractive activities

A composite disturbance index was created by rescaling the litter and lopping score per site between 0 & 1 and then summing them. This index thus ranged from 0 to 2 with 0 representing least disturbance and 2 representing most disturbance. To understand how extractive activities affect vegetation structure, we correlated each vegetation structure variable with the composite disturbance index using Spearman's rank correlation coefficient.

### 3.2. Role of vegetation and landscape factors

To assess relative roles of local and landscape factors on bird community composition, we calculated dissimilarity values for all possible pairs of 74 sites with respect to (1) bird species composition using the

Bray–Curtis distance (with abundances log transformed to prevent dominant species from influencing the analysis), (2) vegetation structure and (3) forest proportion using Euclidean distance. The three dissimilarity matrices were then analysed for any significant associations using Mantel's tests (Legendre and Legendre, 2012). Mantel's test calculates degree of association between the two variables (converted to distance matrices) using simple correlation. The statistical significance of correlations computed using Mantel's tests were computed by performing Monte Carlo simulations with 10,000 randomised runs using the “vegan” package in R 3.5.0 (Oksanen et al., 2011; R Development Core Team, 2014).

Generalised Linear Mixed Models (GLMM) were used to evaluate the influence of vegetation covariates and forest proportion on bird species richness, overall bird abundances and guild-wise abundances. Prior to modelling, we selected a subset of explanatory variables deemed most likely to affect overall bird abundances as well as that of individual guilds. We used Pearson's correlation coefficient to test for collinearity amongst explanatory variables (Appendix A: Table A2). If the correlation coefficient between a pair of covariates was greater than |0.6|, only one (the one we deemed to be ecologically more relevant) was included in the model. Appendix A: Table A3 provides a list of explanatory variables finally selected for use in the models and the rationale behind their consideration. All explanatory variables were standardised (mean = 0, standard deviation = 1) to allow comparison of model parameter estimates. Six apriori models (per response group) were fitted using GLMMs with Poisson errors, using each round of sampling as a random effect (6 temporal replicates) (refer Appendix A: Table A4). AICc (Akaike Information Criterion corrected for small sample sizes) was used to rank the most parsimonious model. Since in all our model sets, no single model was most likely (Appendix A: Table A4), inferences were made using top models with a cumulative AICc weight  $>$  0.95 (Burnham and Anderson, 2002). To evaluate goodness-of-fit of the models to the data, we calculated marginal R<sup>2</sup> (variance explained by the fixed effects) and conditional R<sup>2</sup> (variance explained by the entire model) values for the most complex models using the method described by Nakagawa & Schielzeth (2013). All modelling was carried out in R 3.5.0 (R Development Core Team, 2014), using the packages “lme4” (Bates et al., 2014), “MuMIn” (Barton, 2018) and “aiccmodavg” (Mazerolle, 2017).

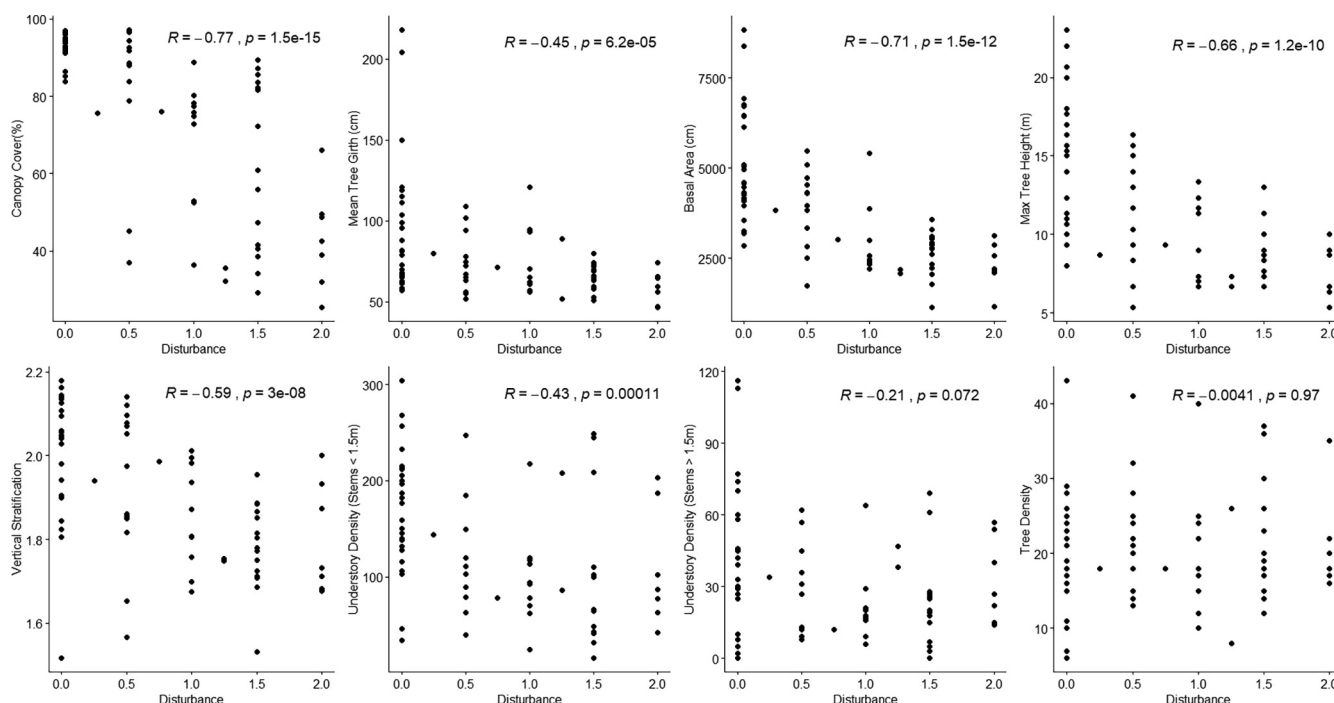
Although our sites were at least 200 m apart to minimise biases from spatial autocorrelation, we additionally checked for spatial autocorrelation by calculating Moran's I on model residuals of the most complex models for all response variables to confirm independence of sites. Moran's I was calculated using the package “ape” in R 3.5.0 (Paradis et al., 2004; R Development Core Team, 2014).

## 4. Results

During the study, a total of 10 115 individual bird detections were made, belonging to 92 species and 33 families. The insectivore guild was the most species-rich foraging guild with 72 species. Frugivores, granivores, nectarivores and omnivores were represented by 9, 4, 2 and 5 species respectively. In terms of habitat guilds, 41 species were forest specialists, 47 were forest generalists, 3 were open area/agricultural species and 1 was a commensal (house sparrow) (Appendix A: Table A1).

### 4.1. Effects of extractive activities on vegetation structure

Percentage canopy cover, maximum canopy height, mean tree girth, basal area, understory density (stems  $<$  1.5 m) and vertical stratification were all significantly and negatively correlated with the disturbance index (Fig. 2). Understory density (stems  $>$  1.5) and tree density were also negatively correlated with the disturbance index but not significantly.



**Fig. 2.** Correlation plots of vegetation variables against the calculated disturbance index along with Spearman's correlation coefficient ( $R$ ) and the associated  $p$  value. Each data point (black circle) represents one of the 74 sites sampled.

#### 4.2. Influence of vegetation structure and forest proportion on bird communities

##### 4.2.1. Community composition

Bivariate Mantel's tests showed substantial effects of vegetation structure ( $r = 0.34$ ,  $p < 0.01$ ) and forest proportion ( $r = 0.38$ ,  $p < 0.01$ ) on bird species composition with both predictor variables showing significant positive correlations.

##### 4.2.2. Overall richness and abundance

Although the top model set contained all covariates examined, standardized  $\beta$ -coefficients for individual covariates from these four models indicated that canopy cover and understory density (stems < 1.5 m) had the strongest (positive) effects on bird species richness and bird abundance relative to other covariates (Table 1). All other covariates, which included understory density (stems > 1.5 m), vertical stratification, tree density, mean tree girth and forest proportion, had far smaller effects. Tree density and mean tree girth had weak negative effects on both bird species richness and abundance (Table 1). Goodness-of-fit test using marginal and conditional  $R^2$  showed that the bird abundance model had good fit (marginal  $R^2 = 0.37$ , conditional  $R^2 = 0.62$ ) while bird species richness showed a weaker predictive

value (marginal  $R^2 = 0.15$ , conditional  $R^2 = 0.15$ ).

##### 4.2.3. Abundance of dietary guilds

Abundance of insectivorous birds was best explained by models containing canopy cover, understory density, vertical stratification, mean tree girth and forest proportion (Table 2). Examination of the standardized  $\beta$ -coefficients for individual covariates from the model set indicated that insectivore abundances were strongly dependent on canopy cover, forest proportion and to a lesser extent on vertical stratification, while understory density (both variables) did not have any substantial influence (Table 2). Models of insectivore abundance had good fit (marginal  $R^2 = 0.44$  and conditional  $R^2 = 0.72$ ). In the case of frugivores, the top models contained canopy cover, understory density, vertical stratification and forest proportion. Comparison of the standardised  $\beta$ -coefficients showed frugivore abundance depended strongly on understory density (< 1.5 m) but on no other vegetation variable. Frugivore abundance also showed a substantial negative dependence on forest proportion (Table 3). Models of frugivore abundance had average fit (marginal  $R^2$  of 0.14 and conditional  $R^2$  of 0.44). Omnivore and granivore abundances could not be sufficiently predicted by any of our covariates (Table 2) and model fit was very poor (Omnivore: Marginal  $R^2 = 0.01$ , Conditional  $R^2 = 0.02$ , Granivore: Marginal  $R^2 = 0.05$ ,

**Table 1**

Predictors of bird abundance and richness in the mid-elevation oak forests of Kumaon. Table values are beta co-efficients of top models with associated standard errors (values in bold represent values within the 95% CI).

Response	Model (Explanatory Variables)	Intercept	CC	LS	TD	USL	VS	USH	AG
Abundance	CC + VS + USL + USH + LS	3.24 (0.02)	<b>0.08 (0.03)</b>	0.05 (0.03)	−0.03 (0.02)	<b>0.07 (0.03)</b>	0.05 (0.03)	NA	NA
	CC + VS + USL + LS + TD	3.24 (0.02)	<b>0.09 (0.03)</b>	0.03 (0.03)	NA	<b>0.06 (0.03)</b>	0.05 (0.03)	0.04 (0.03)	NA
	CC + VS + USL + USH + LS + TD + AG	3.24 (0.02)	<b>0.11 (0.03)</b>	0.04 (0.03)	−0.06 (0.03)	<b>0.07 (0.02)</b>	0.04 (0.03)	0.04 (0.03)	−0.04 (0.03)
	CC + USL + LS + TD	3.24 (0.02)	<b>0.11 (0.03)</b>	0.05 (0.03)	−0.05 (0.02)	<b>0.08 (0.03)</b>	NA	NA	NA
Richness	CC + USL + LS + TD	2.41 (0.01)	<b>0.08 (0.02)</b>	0.04 (0.02)	−0.03 (0.02)	<b>0.05 (0.02)</b>	NA	NA	NA
	CC + VS + USL + USH + LS	2.41 (0.01)	<b>0.08 (0.02)</b>	0.03 (0.02)	−0.03 (0.02)	<b>0.05 (0.02)</b>	0 (0.02)	NA	NA
	CC + VS + USL + USH + LS + TD + AG	2.41 (0.01)	<b>0.09 (0.02)</b>	0.03 (0.02)	−0.05 (0.02)	<b>0.06 (0.02)</b>	−0.01 (0.02)	0.02 (0.02)	−0.03 (0.02)

Legend: CC: canopy cover, VS: vertical stratification, USL: understory density (stems < 1.5 m), USH: understory density (stems > 1.5 m), LS: forest proportion, TD: tree density, AG: mean tree girth.

**Table 2**

Predictors of bird dietary guild abundances in the mid-elevational oak forests of Kumaon. Table values are beta co-efficients of top models with associated standard errors (values in bold represent values within the 95% CI).

Dietary Guild	Model (Explanatory Variables)	Intercept	CC	LS	USL	VS	USH	TD	AG
Insectivore	CC + VS + USL + LS	2.85 (0.03)	<b>0.15 (0.04)</b>	<b>0.11 (0.04)</b>	0.06 (0.04)	0.07 (0.04)	NA	NA	NA
	CC + VS + LS	2.85 (0.03)	<b>0.12 (0.04)</b>	<b>0.14 (0.04)</b>	NA	<b>0.09 (0.04)</b>	NA	NA	NA
	CC + VS + USL + USH + LS + AG	2.85 (0.03)	<b>0.16 (0.04)</b>	<b>0.09 (0.04)</b>	0.04 (0.04)	0.05 (0.04)	0.05 (0.04)	NA	0.02 (0.03)
Frugivore	USL + VS + LS	0.75 (0.07)	NA	<b>-0.18 (0.08)</b>	<b>0.33 (0.08)</b>	0.14 (0.08)	NA	NA	NA
	USL + USH + VS + LS	0.75 (0.07)	NA	<b>-0.17 (0.08)</b>	<b>0.35 (0.08)</b>	0.15 (0.08)	-0.05 (0.08)	NA	NA
	USL + USH + VS + LS + CC	0.75 (0.07)	0.07 (0.1)	<b>-0.2 (0.09)</b>	<b>0.36 (0.08)</b>	0.11 (0.09)	-0.04 (0.08)	NA	NA
Omnivore	CC + TD + LS	-0.13 (0.09)	-0.17 (0.1)	-0.11 (0.1)	NA	NA	NA	-0.1 (0.09)	NA
	CC + VS + LS	-0.13 (0.09)	-0.15 (0.11)	-0.11 (0.1)	NA	-0.04 (0.11)	NA	NA	NA
	USL + TD + LS	-0.13 (0.09)	NA	<b>-0.2 (0.1)</b>	-0.01 (0.19)	NA	NA	-0.1 (0.09)	NA
	CC + VS + TD + LS	-0.13 (0.09)	-0.16 (0.12)	-0.1 (0.12)	-0.02 (0.10)	-0.03 (0.11)	NA	NA	NA
Granivore	CC + TD + LS	-0.81 (0.12)	-0.19 (0.13)	-0.23 (0.14)	NA	NA	NA	-0.23 (0.12)	NA
	USL + TD + LS	-0.81 (0.12)	NA	<b>-0.34 (0.13)</b>	0.01 (0.13)	NA	NA	-0.23 (0.12)	NA
	CC + VS + USH + LS + TD	-0.81 (0.12)	-0.09 (0.16)	-0.20 (0.16)	NA	-0.19 (0.15)	NA	-0.27 (0.12)	NA

Legend: CC: canopy cover, VS: vertical stratification, USL: understory density (stems < 1.5 m), USH: understory density (stems > 1.5 m), LS: forest proportion, TD: tree density, AG: mean tree girth.

Conditional  $R^2 = 0.2$ ). We did not run models for avian nectarivore abundances since this guild was represented by only two species with insufficient detections (7 sites) (Appendix A: Table A1).

#### 4.2.4. Abundance of habitat preference guilds

Abundance of forest specialist species was explained best by models that included canopy cover, forest proportion, vertical stratification and understory density (stems < 1.5 m) (Table 3). Comparison of standardised  $\beta$ -coefficients suggests that canopy cover and forest proportion have a very strong and positive effect on the abundances of forest specialist species, and vertical stratification has a weaker yet substantial effect (Table 3). Our models of forest specialist species abundance had a strong fit (Marginal  $R^2 = 0.69$ , Conditional  $R^2 = 0.84$ ). Forest generalist species abundance was explained best by models that included canopy cover, understory density, vertical stratification, maximum canopy height and forest proportion. Comparison of the standardised  $\beta$ -coefficients suggests that understory density (stems < 1.5 m) is the only variable that had a substantial effect on forest generalist species (Table 3). Models of forest generalist species abundance had poor fit (Marginal  $R^2 = 0.07$ , Conditional  $R^2 = 0.45$ ). We did not run models for open-area/agricultural species abundances since this guild was represented by only three species with not enough detections (9 sites) (Appendix A: Table A1).

Results of the Moran's I test on model residuals found no autocorrelation in any of the response variables. Moran's I coefficient was very low with  $p > 0.05$  for all response groups.

## 5. Discussion

Researchers have devoted considerable attention to the effects of human activities on biodiversity in relatively natural ecosystems (Tilman et al., 1994; Vitousek et al., 1997; Hill and Hamer, 2004).

**Table 3**

Predictors of bird habitat guild abundances in the mid-elevational oak forests of Kumaon. Table values are beta co-efficients of top models with associated standard errors (values in bold represent values within the 95% CI).

Habitat Guild	Models (Explanatory Variables)	Intercept	CC	HT	LS	VS	USL	USH
Forest Specialist	CC + VS + LS	1.64 (0.05)	<b>0.34 (0.07)</b>		<b>0.34 (0.06)</b>	<b>0.27 (0.06)</b>	NA	NA
	CC + VS + USL + LS	1.64 (0.05)	<b>0.36 (0.07)</b>		<b>0.31 (0.06)</b>	<b>0.25 (0.06)</b>	0.06 (0.06)	NA
Forest Generalist	HT + LS + USL	2.7 (0.03)	NA	0.04 (0.03)	-0.03 (0.04)	NA	<b>0.08 (0.03)</b>	NA
	CC + USL + USH + LS	2.7 (0.03)	0.06 (0.03)	NA	-0.04 (0.04)	NA	<b>0.08 (0.03)</b>	0.03 (0.03)
	CC + VS + USL + LS	2.7 (0.03)	0.06 (0.03)	NA	-0.03 (0.04)	0	<b>0.09 (0.03)</b>	NA

Legend: CC: canopy cover, VS: vertical stratification, USL: understory density (stems < 1.5 m), USH: understory density (stems > 1.5 m), LS: forest proportion, HT: maximum canopy height.

However low-intensity extractive activities have been far less studied in this regard than more drastic forms of disturbance, such as selective logging or conversion to plantations (Thiollay, 1997; Dunn, 2004; Aratrakorn et al., 2006; Wilcove et al., 2013). Our study fills a significant lacuna in exploring not only the impacts of long-term extractive use but also the underlying causative mechanisms. Our results suggest that extractive activities have significant impacts on bird communities and therefore, are as important to consider in conservation planning as more drastic forms of disturbance. Forest extraction had negative effects on vegetation structure, including opening up of the canopy, reduction in tree girth, canopy height and vertical stratification, suggesting overall simplification of forest structure. Thus, we find causal mechanisms similar to those identified in the few other studies that have addressed this topic (Bhat et al., 1995; Kumar and Shahabuddin, 2005).

#### 5.1. Influence of vegetation structure and forest proportion on bird communities

##### 5.1.1. Influence on avian richness and abundance

Our prediction that both avian richness and abundance would be significantly affected in harvested forests in comparison to protected forests was borne out by our results. The two most important factors related to vegetation structure that we found to affect avian species richness and abundance were canopy cover and understory density: these are structural variables found to be important in several other studies as well (Shankar Raman et al., 1998; Martin and McIntyre, 2007; Shahabuddin and Kumar, 2007; Dahal et al., 2015). In our study area, the forest understory is an important refuge for several bird groups like pheasants and laughingthrushes, while a healthy canopy maintains the microclimatic factors required for these groups to feed and nest in (Sekercioğlu et al., 2002).

### 5.1.2. Influence on community composition

While abundance and species richness responded more to local vegetation structure, avian composition was strongly influenced by both local and landscape factors. We find that both local vegetation structure and forest proportion are important in preserving the structure of bird species composition in the mid-elevation oak forests of Kumaon. With extractive activities increasingly altering forest vegetation structure, our results imply that extractive activities significantly influence the composition of forest bird communities. Our results reflect the general trends observed by Mazerolle and Villard (1999) who showed that landscape characteristics were as significant as local patch characteristics, in predicting species responses of a number of taxa (especially birds) across landscape types; thus, protection of dense forests within multiple-use landscapes can enhance bird conservation.

### 5.1.3. Influence on dietary guilds

Our predictions for the vulnerability of insectivorous birds were borne out by the results; insectivores were found to respond to both local habitat and landscape factors. Globally, insectivorous birds have been identified as one of the dietary guilds most vulnerable to forest modification and conversion in other landscape contexts as well (Canaday, 1996; Thiollay, 1997; Sekercioglu et al., 2002; Shahabuddin and Kumar, 2007). Due to their high ecological specialization, many forest insectivores are highly sensitive to habitat degradation and fragmentation (Sekercioglu et al., 2002; Sekercioglu et al., 2004). Insectivores such as tits, warblers, woodpeckers and thrushes, which were among the most abundant species in our study area, were also the most vulnerable to extractive activities. Further, insectivore abundances were strongly associated with both canopy cover and forest proportion indicating that they require larger forest stands where canopies are mostly undisturbed from logging. Opening up of the canopy may adversely affect forest invertebrate densities as a result of an increase in day temperatures and desiccation, thus impacting most midstorey and understory insectivores (Barlow and Peres, 2004; Richards and Windsor, 2007; Silveira et al., 2010).

Adverse effects of forest use on insectivores have significant implications for ecosystem services. Since many insectivorous birds are highly specialised, it is unlikely that other taxa could replace them. Insectivorous birds play an important role in controlling insect populations, the loss of which could cause pest outbreaks and consequent tree damage, resulting in economic losses (Nyffeler et al., 2018; Sekercioglu et al., 2004).

The reverse trend was observed for omnivore and granivore abundances: these guilds showed weak negative effects of canopy cover and forest proportion. Omnivores and granivores tend to be less affected by forest disturbance in most cases as has been seen in several other studies (Canaday, 1996; Waltert et al., 2005; Watson et al., 2004; Aratrakorn et al., 2006). Species in these guilds (such as Black-headed Jay and Oriental Turtle Dove) are commonly observed near human settlements, showing their adaptability.

In many studies, frugivores do not appear to be as vulnerable to forest extraction as insectivores, a pattern borne out by our study (Thiollay, 1992; Canaday, 1996; Gomes et al., 2008). This can be attributed to their wide-ranging behaviour which allows them to locate widely-dispersed fruiting trees (Thiollay, 1997; Moran and Catterall, 2014). Further, there were no large frugivores such as hornbills in our study area, which tend to be far more specialised in their nesting and feeding requirements and hence more vulnerable to habitat degradation (Gomes et al., 2008; Velho et al., 2012). In our study area, frugivore abundance was negatively associated with forest proportion, suggesting a preference for forest edge, cultivation and orchards, likely due to greater fruit availability in these habitats. Yet due to the small number of frugivores observed in our study, this finding should be treated as tentative and needs confirmation with additional observations.

### 5.1.4. Influence on habitat guilds

In studies of land use change, forest specialists have been found to be more affected by habitat alteration in comparison to open area species and forest generalists (Pardini et al., 2009; Sodhi et al., 2010; Morante-Filho et al., 2015). Our study provides unequivocal evidence that forest specialists are most strongly affected by changes in vegetation structure (caused by forest use and extraction) and landscape composition in comparison to other guilds. The abundances of forest specialists, which includes Rufous-bellied Woodpecker (*Dendrocopos hyperythrus*) and Black-faced Warbler (*Abroscopus schisticeps*), were found to have a strong positive association with canopy cover, vertical stratification and forest proportion. As reduced canopy cover and vertical stratification are two key effects of logging, it is likely that excessive logging may have drastic negative effects on this guild. Forest generalists (eg. Verditer Flycatcher, Blue-whistling Thrush), were largely unaffected by extractive activities and related changes in vegetation structure, which can be attributed to their adaptable nature (see also Imbeau et al., 2003). It is clear from our results that progression of forest loss and degradation in future will inevitably result in loss of oak forest specialists in the Western Himalayas.

## 6. Conclusions

This study shows that extractive use of forests can have adverse effects on vegetation structure and, in turn, on avifaunal communities. While a number of studies have explored the role of local vegetation factors in determining avian composition and diversity, few have simultaneously addressed the role of landscape composition. Our study shows that proportion of protected oak forest in the surrounding landscape may be as important as local stand-level vegetation structure in determining avian community composition, richness and abundance. In particular, insectivores and forest specialists have been identified as vulnerable guilds in harvested forests, a result that concurs with studies carried out globally. Canopy cover was most strongly associated with higher abundances and richness of all species as well as more vulnerable guilds like forest specialists and insectivores. This is a matter of concern because loss of canopy cover was found to be the most significant effect of logging. Our findings also represent a significant addition to knowledge about the biodiversity value of Himalayan oak forests, a biome that has remained largely understudied despite its considerable conservation value and imminent threats from over-exploitation and land-use change.

Our results also have important implications for community-based conservation in the forests of the Himalayan region. Local village-based extraction of NTFP is often considered to be sustainable as it is believed that subsistence users ensure resource sustainability via strong village institutions (e.g., Gadgil, 1992; Wood, 1995). Yet, our results indicate that extractive activities can significantly modify vegetation structure which, in turn, results in local extinction of certain species within vulnerable avifaunal guilds. In particular, a number of insectivores and forest specialists, are missing from harvested forests or are highly reduced in abundance. The findings of our study suggest that community-based conservation may not always result in long-term protection of local biodiversity (see also review in Shahabuddin and Rao, 2010). The reason for this being that despite having regulations on forest extraction, community-controlled forests are subject to high subsistence pressures in which the needs of livelihood take precedence (Shahabuddin and Thadani, 2018). In our study area, it may be necessary to control extraction in ways designed to preserve canopy cover, vertical stratification and understory density (Leal et al., 2013). For instance, forest stands can be harvested in rotational sequence so that harvested portions can be regenerated over time. Further, it may be necessary to develop alternatives to widely used plant products that are required by households, especially fuelwood and fodder. Finally, since the ability of harvested forests to protect native avifauna depends significantly on the proportion of dense undisturbed forest in the



landscape, retention of old growth forest stands within the harvested forest matrix could help better conserve avian forest fauna.

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## Declaration of Competing Interest

This is to state that we do not have any actual or potential conflict of interest including any financial, personal or other relationships with other people or organizations within one year of beginning the research work leading to the submitted manuscript entitled “Effects of Extractive Use on Forest Birds in Western Himalayas: Role of Local and Landscape Factors” that could inappropriately influence or be perceived to influence the research work.

## Appendix A. Supplementary material

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.foreco.2019.06.033>.

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