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Livestock Grazing in Afromontane Grasslands in the Northern Bale Mountains, Ethiopia: Implications for Bird Conservation

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Abstract	Article Information
The aim of this study was to examine the effects of livestock grazing on Afromontane grassland bird assemblages in the Bale Mountains, Ethiopia. Birds were counted along 28 (14 in ungrazed site and 14 in grazed site) 1-km transects both during wet season (in June 2014) and dry season (November 2014). In addition, height and cover of shrubs, grasses and herbs were recorded within 10m ×10m quadrats established along each transect at a distance of 250m. These data were used to compare bird species richness, evenness and abundances of overall assemblages and guilds (habitat, feeding and/or conservation priority guilds) between the ungrazed site and grazed site, and to examine how these patterns are related to grazing-induced vegetation changes. Although the ungrazed site showed relatively greater species evenness compared to the grazed site, both observed and rarefied species richness estimators showed non-significant difference in overall assemblage richness between the two sites. Bird assemblage abundance was significantly greater in the grazed site than the ungrazed site, especially during wet season and when seasons were pooled. At guild level however, species richness and/or abundances of grassland habitat specialist, insectivore dietary, and high conservation priority guilds were significantly greater in the ungrazed site compared to the grazed site. Bird assemblages significantly differed between sites and showed significant positive relationships with shrub and grass height. These findings suggest that the effect of grazing on birds of the area is changing assemblage composition rather than resulting into declined assemblage species richness. Thus allowing livestock grazing in the ungrazed site in the future will lead to loss of several grassland specialist and high conservation priority species.	

INTRODUCTION

The degradation and destruction of natural habitats due to human-induced actions are major causes of global biodiversity decline (Brooks *et al.,* 2006; Chown, 2010). This is expected to be increasing in future in developing Tropical countries like Ethiopia, coupled to the ever-increasing rate of population growth, where there is a high demand for extensive arable land and grazing areas to meet their basic life needs (Mckee, 2005; Chown, 2010). Understanding the associated threats these actions cause and the way in which plants and animals respond to them is thus important for efficient and effective conservation decision making and management actions (Bleher *et al*., 2006; Brooks *et al*., 2006).

Grazing by livestock has been considered to be one of such major factors that lead to habitat alteration in different ecosystems worldwide (e.g. Martin and Possingham, 2005, in Australian woodlands; Whittingham and Devereux, 2008, in grasslands in UK; Yosef Mamo *et*

al., 2014, in Ethiopian Afromotane grasslands). It causes changes in the vertical and horizontal structural composition of vegetation through a combination of trampling, grazing, changes in nutrient fluxes and loss of recruitment (Jensen, 1985; McIntyre *et al*., 2003), and facilitates encroachments of non-native species (Kimball and Schiffman, 2003). Further, as a result of the differential responses of different plant species (some respond positively while others respond negatively) to grazing, it can also alter species composition (Rasran *et al*., 2007; Whitehorn *et al*., 2011; Yosef Mamo *et al*., 2014). Such grazing-induced changes in vegetation structure and composition can in turn impact animal diversity, including bird communities. Covering more than 40% of Earth's land surface, grasslands are the most converted biome due to grazing (Hoekstra et al., 2005), resulting to a more rapid decline in diversity and population of grassland birds than birds of any other habitat type worldwide (Hoekstra *et al*., 2005; Rahmig *et*

al., 2009). These studies however have found that different species showed widely varying responses and the extent and form of the responses often depend on species-specific ecological traits, including dietary and habitat specialization, and season (Fuller and Gough, 1999; Martin and Possingham, 2005; Evans *et al.*, 2006; Whitehorn *et al.*, 2011). Thus knowledge regarding the ecology and conservation of grassland birds at local level, a scale at which most conservations management decisions are often made, is of important to devise effective management plans.

In this paper we examine the effects on Afromontane grassland bird assemblages of grazing in the Bale Mountains, Ethiopia. The Bale Mountains region is recognized as the centre of endemicity and evolution for several biological taxa (Williams *et al.,* 2004; Addisu Asefa, 2011). It is also one of an Important Bird Area of Ethiopia (EWNHS, 1996). However, this biologically and ecologically important area has been highly altered and degraded due to agricultural expansion, settlement and livestock overgrazing (OBARD, 2007). The northern montane grassland represents a key habitat for several ungulate species, including the endangered endemic mountain nyala (*Tragelaphus buxtoni*), and of (near) endemic and/or globally threatened birds species such as the Abyssinian long-claw and Rouget's rail (Yosef Mamo *et al.*, 2014). Consequently, most conservation efforts in the Bale Mountains National Park (BMNP) have been concentrated to this area. Despite its immense ecological importance and regular ranger-based law-enforcement activities being carried out in the area, this grassland ecosystem and its associated biodiversity have been threatened mainly by livestock grazing (Stephens *et al.*, 2001; OARDB, 2007; Yosef Mamo *et al.*, 2014).

We have recently shown that livestock grazing in the area has led to reduced cover and height of shrubs and height of grass, but increased covers of grass, grazing-weedy herbs and bare ground (Yosef Mamo *et al.*, 2014). It is likely therefore that this grazing-induced vegetation change could affect bird assemblages of the area due to differential responses of species to such habitat change depending on their specific ecological traits, including habitat and dietary requirements. Focusing on two sets of different guilds: (i) habitat guild (i.e., shrub and/or tall grass specialists *vs.* generalists) and dietary guilds (insectivore, granivore, omnivore and nectarivore), we examined the impact of livestock grazing on avifaunal diversity (i.e. species richness and evenness), population abundance and assemblage composition. We also assessed separately the effects of grazing on species groups considered to be relatively with high conservation priority (i.e. endemic, globally threatened and/or biome-restricted). We predicted that the diversity and abundance of overall avian assemblage and of guilds of species that primarily prefer shrub and/or tall grass habitat and/or are insectivore would be higher in the ungrazed site compared to the grazed site, and *vice-versa* for habitat generalist.

MATERIALS AND METHODS
Study Area
The Bale Mountains region is located in the south-eastern highlands of Ethiopia (Figure 1). It is part of the Eastern Afromontane Hotspot Biodiversity area designated by Conservation International (Williams *et al.*, 2004). At the heart of these mountains is the Bale

Mountains National Park (BMNP), which is located at about 400 Km southeast of the capital, Addis Ababa (OBARD, 2007). The national park covers an area of 2200 km^2 and ranges in altitude from 1500 – 4377m a.s.l. (OBARD, 2007). To date about 78 species of mammals and 278 bird species have been recorded from the Bale Mountains area; of which 17 mammals and 6 bird species are endemic to Ethiopia (Addisu Asefa, 2007, 2011). The Bale Mountains area is characterized by eight months (March-October) of rainy season and four months (November-February) of dry season (OARDB, 2007). The present study was carried out in the northern montane grassland area which occurs as a central broad flat valley (between altitudes of 3000 - 3150 m a.s.l.) between two mountainous ranges (Figure 1). This grassland has an area of 37 km^2 of which ~ 15 km^2 is included in the Park (hereafter referred to as ungrazed site) and illegal livestock grazing takes place seldom. The remaining area that falls outside the park is being used as a communal livestock grazing land by the surrounding local community (hereafter referred to as grazed site) (Figure 1; see also OABRD, 2007). On the average, (mean ± S.D.) 1528 ± 86 heads of livestock (cattle and horses) are reported to use the grazed site, respectively, every day (Yosef Mamo *et al.*, 2014).
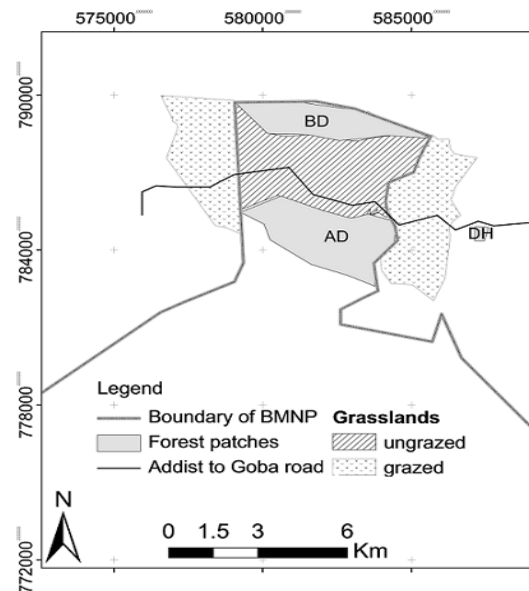

Figure 1: Map of the Bale Mountains National Park (only partly shown) and the ungrazed and grazed—those shown on the east and west of the ungrazed site—grassland sites, and the three forest patches found around the grasslands. Abbreviations of the forest patches: DH = Dinsho Hill (BMNP HQs); AD = Adellay; & BD = Boditti.

The vegetation of this grassland is broadly classified in to three types: open grassland (areas covered by short grasses), marsh grassland (characterized by swamp grasses and sedges of *Cyperus* and *Scirpus* genera), and shrubland (covered by bushes of *Artemesia afra* and *Helichrysum splendidum*) (OARDB, 2007). The extent of the open grasslands is 9 and 15 km^2 and of marsh grassland is 10 and 15 km^2, respectively, in the ungrazed and grazed sites. Shrublands are completely destroyed in the grazing site and currently occur only in the ungrazed site (Yosef Mamo *et al.,* 2014). Based on 14 altitude measurements taken during this study period using

Garmin GPS unit at mid-points of each transect in each land use type, elevation ranges between 3042m – 3070m a.s.l. (mean = 3056m a.s.l.) in the ungrazed site and between 3049 – 3112m a.s.l. (mean = 3091m a.s.l.) in grazed site. This montane grassland is considered as a key habitat for several ungulate species, including the endangered endemic mountain nyala (*Tragelaphus buxtoni*) and other ungulates (OBARD, 2007).

Data Collection

Bird data were collected in June (wet season) and November (dry season) 2014 along systematically established 28 transects (14 in each land use type and at a minimum distance of 300m apart) of each 1-km long, covering ~ 9% and 6% of each vegetation type in the ungrazed and grazed sites, respectively. The start and end geographical coordinates of each transect were saved in Garmin GPS unit to ensure same transects were repeated during the dry season. Birds were counted within 50 m width on both sides of each transect. All counts (recording of species identity and number of individuals) were made by the same expert observer, early in the morning (between 07:00-10:00) when birds are thought to be more active, while slowly walking at speed of ~ 2 km hr^{-1}. Aerial feeders (raptors, swallows and swifts) and wetland birds were not recorded as the primary objective of the study was on terrestrial birds.

To determine the extent to which the two land use types (ungrazed *vs*. gazed) might differ in their vegetation structure, which in turn might affect avian diversity, vegetation data (heights and percentage covers of plant life-forms [shrub, herb and grass, and percent of bare ground]) were recorded within four 10 m × 10 m quadrates established along each transects at 200 m distance intervals. Separate data was recorded for two giant herb species (*Ferula communis* and *Kniphofia foliosa*) that grow up to 2m high as we thought that their ecological role as bird habitat and food may differ from that provided by small herbs that measure up to 0.5cm high. To determine heights of each plant growth forms, four different measurements were taken at each quadrat (totalling to 16 measurements per transect) using a labelled measuring stick and cover was visually estimated (Newton, 2007).

Data Analysis
Effects of Grazing on Species Diversity and Population Abundance

We used species richness and evenness measures to compare species diversity between the two sites (ungrazed and grazed sites) across the two seasons. Sample-based accumulation curves were compiled for each land use types to determine sampling adequacy by comparing the observed curves with that estimated by appropriate richness estimator. The observed curves were computed using the moment-based interpolation method (i.e. Mao Tau method) in EstimateS package (EstimateS v. 8.2 (http://viceroy.eeb.uconn.edu/estimates) by allowing 500 runs of randomizations without sample replacement, following Colwell (2009). Samples are considered to be adequate if the observed accumulation curves (based on Mao Tau) flatten off at the maximum point the accumulated sample, or if they closely converge with a richness estimator. Estimated richness were computed using all the 10 estimators' available in EsimateS package by allowing 500 runs of randomizations with sample replacement (Colwell, 2009). We selected two of the

estimators that provided the lowest (in this case, Chao 1 was chosen) and the highest (Jacknife 2) value of estimated number of species to assess sampling representativeness by comparing them with the observed richness.

To compare total species richness between and across treatments (two sites × two seasons), the observed sample-based accumulation curves (based on Mao Tau) were rescaled to the number of individuals (Gotelli and Colwell, 2001; Colwell, 2009). Species evenness (a measure of diversity how similar species of an assemblage are in their abundance; higher evenness means more diverse) was also computed in same application using the reciprocal of Simpson diversity index (i.e. as 1/D) (Colwell, 2009). Simpson index was use as it is suggested to be an appropriate measure of diversity for small sample size (Colwell, 2009). However, computation of this diversity index combines information both from number of species and evenness of species abundances of a given assemblage (Magurran, 2004). Thus the reciprocal values Simpson index (1/D) obtained from the EstimateS output for each treatment were divided to the total number of species recorded for each treatment to compute their respective evenness indices (Magurran, 2004). Comparisons of richness, and evenness were made at lower number of individuals recorded in a given treatment category.

Generalized linear model with Poisson probability distribution and log-link function, appropriate for counting data (Quinn and Gough, 2002), was used to assess the effects of grazing, season and their interaction on bird abundance. Finally, one-way ANOVA was used to test the difference in mean abundance between dry and wet seasons within each site and between ungrazed and grazed sites within each season. Both analyses were undertaken in SPSS version 20 statistical package.

Effects of Grazing on Assemblage Composition

We followed two approaches to examine the effects of grazing on bird assemblage composition: guild composition and multivariate similarity analyses. Prior to analysis we first classified bird species in to two sets of functional guilds based on their habitat and dietary requirements. BirdLife International classifies bird habitats as: artificial terrestrial/aquatic and introduced/exotic vegetation, forest, savannah, shrubland, grassland, rocky areas, caves, desert and several different types of wetlands. The importance of each habitat inhabited by a species was classified as major or suitable (BirdLife International, 2014, http://www.birdlife.org/datazone/info/spchabalt). For the purposes of this study, species were considered as shrubland and/or grassland specialists, if shrubland and/or grassland were/was recorded as being of major importance to the species; otherwise, were treated as generalists. Species dietary requirement was classified in to four categories, following Addisu Asefa Mitiku (2013) and Gove *et al.* (2013) as: insectivore (species whose food predominantly constitutes invertebrates), granivore (seed eaters), ominivore (species feeding on insects and plant materials), and nectarivore (species feeding on flower nectar). Detailed information on species-specific habitat, dietary and conservation categories are provided in the Table 1.

Furthermore, although habitat specialists in general are supposed to be more vulnerable to grazing than

generalists, some of these specialist species may need relatively greater due attention for conservation actions than others by virtue of their higher local, regional and/or global conservation significance. Thus using three main criteria (which have also been used to select Important Bird Areas globally)—endemicity, biome-restrictedness and global threat status—we assessed if the effects of grazing observed on overall assemblage might also be reflected on conservation concern species (i.e. habitat specialist species that are of national and/or global conservation concern). Accordingly, habitat specialist species that are endemic to Ethiopia, [highland] biome-restricted and/or globally threatened (critically endangered, endangered, vulnerable or near-threatened; BrdLife International, 2014) were considered as of high conservation significance; otherwise, considered as of low significance.

Table 1: Species recorded during the study period from the Afromontane grasslands in the northern Bale Mountains, southeast Ethiopia and their habitat, dietary and conservation priority categories. (Abbreviations for guild categories are defined as follows; habitat: spc = specialist, gen = generalist; dietary: insec: insectivore, omn = omnivore, gran = granivore, nect = nectarivore; For site/season: UG DS = ungrazed dry season; UG WS = ungrazed wet season; GR DS = grazed dry season; GR WS = grazed wet season.)

common name	scientific name	habitat	dietary	conservation priority	UG DS	UG WS	UG SUM	GR DS	GRWS	GR SUM
Abyssinian longclaw	*Macronyx flavicollis*	spe	insec	high	34	30	64		7	7
Baglafecht weaver	*Ploceus baglafecht*	gen	insec	low		11	11		7	7
Cape canary	*Serinus canicollis*	spe	gran	high	36	23	59	6	234	240
Cape crow	*Corvus capensis*	gen	omn	low				12	32	44
chestnut-naped francolin	*Francolinus castaneicollis*	spe	omn	high		10	10			
cinnamon-bracken warbler	*Bradypterus cinnamomeus*	spe	insec	low	9	8	17			
common fiscal	*Lanius collaris*	gen	insec	low		2	2		1	1
common quail	*Coturnix coturnix*	spe	omn	low		2	2			
common stone-chat	*Saxicola torquatus*	spe	insec	low	29	45	74	1	2	3
common waxbill	*Estrilda astrild*	spe	gran	low	11	43	54		4	4
Ethiopian Siskin	*Serinus nigriceps*	spe	gran	high	294	80	374	134	164	298
grassland pipit	*Anthus cinnamomeus*	gen	insec	low				13	11	24
grey wagtail	*motacilla cinerea*	gen	insec	low				15		15
ground scraper thrush	*Turdus litisitsirup*	gen	insec	low	11	6	17	69	86	155
Issabelline wheatear	*oenathe isabellina*	gen	insec	low	1		1			
malachite sunbird	*Nectaniria famosa*	spe	nect	low		9	9		6	6
moorland chat	*Cercomela sordida*	spe	insec	high	54	69	123	17	41	58
moorland francolin	*Francolinus psilolaemus*	spe	omn	high	2	3	5			
red-billed oxpecker	*Buphagus erthrorhynchus*	gen	insec	low				29	8	37
red-breasted wheatear	*Oenathe bottae*	spe	insec	low	4		4	12	7	19
red-throated pipit	*Anthus cervinus*	gen	insec	low				31		31
Rouget's rail	*Rougetius rougetii*	spe	insec	high	4	19	23			
sacred ibis	*Threkiornis aethiopicus*	gen	insec	low				95	19	114
slender-billed starling	*Oegnathus tenuirostris*	gen	omn	high		2	2			
streaky seedeater	*Serinus striolatus*	spe	gran	high	25	51	76	2	5	7
tacazze sunbird	*Nectaniria tacazze*	gen	nect	low		13	13		49	49
tawny-flanked prinia	*Prinia subflava*	spe	insec	low		5	5			
Thekla lark	*Galerida theklae*	gen	insec	low	13	24	37	40	98	138
thick-billed raven	*corvus crassirostis*	gen	omn	low				3		3
wattled ibis	*Bostrychia carunculata*	gen	insec	low				245	620	865
winding cisticola	*Cisticola galactotes*	spe	insec	low	32	115	147	10	44	54
yellow bishop	*Euplectes capensis*	spe	gran	low	80	47	127	3	66	69
yellow wagtail	*Motacilla flava*	gen	insec	low				62		62

We then computed the sample-based species accumulation curves discussed above for each guild type and used rarefaction curves (rarefied to lowest number of individuals recorded in a site) to compare species richness of each functional guild and conservation significance groups between the grazed and ungrazed sites. We also calculated the proportion each guild contributed to the total number of species and abundances of the bird assemblages in each site. Chi-square was used to test the presence of any significant deviation from expected guild contribution (assuming that each guild would contribute equal proportions to the two assemblages if grazing was absent). All these analyses were conducted by pooling data from the two seasons as number of species and/or individuals for most of the guilds were found to be small to treat seasons separately. Finally, we used two-way crossed ANOSIM (Analysis of similarity) routine in PRIMER software (Clarke and Gorley, 2006) to assess variations in bird assemblage composition between the two sites across seasons (i.e. grazing and season as factors), and one-way ANOSIM to compare between seasons for each site and between sites within each season. A Bray-Curtis similarity index was used to calculate similarities in composition among assemblages; data for each species in each site were first standardized by multiplying the raw values by species' dispersion weight (to minimized the effects of species occurring in large flocks) (Clarke and Gorley, 2006). These standardized values then were square-root transformed beforehand to down-weight common species relative to those that are rare (Clarke and Gorley, 2006). Global R values were used to determine the degree of similarity among treatments. This is a non-parametric permutation procedure applied to rank similarity matrices underlying sample ordinations (Clarke and Gorley, 2006). The closer the value of R value is to 1, the more dissimilar species assemblages are.

Effects of Grazing-Induced Vegetation Change on Bird Assemblage

Data on vegetation height and cover were log $(x + 1)$ and arc-sin transformed, respectively, and then were normalized prior to analysis (Clarke and Gorley, 2006). Two-way crossed (using grazing and season as factors) ANOSIM was used to assess the effects of grazing and season on vegetation structure. The results of this analysis showed that only the effect of grazing was significant (Global R = 0.873, $P<0.05$), but of season was not both within and across grazing treatments. Thus mean values of data for each vegetation variable was calculated from the two seasons and used to assess if the grazing-associated changes in vegetation characteristics affected bird assemblages. The BEST procedure was used to examine this relationship between the multivariate community analysis (e.g. similarity among samples) and that from the habitat variables associated with those samples; the extent to which these two patterns match reflects the degree to which the chosen habitat variables explain the diversity pattern (Clarke and Gorley, 2006). Variable selection was made using a Bio-Env algorithm, which searches all possible combinations from the primary datasheet. Spearman rank correlation (P_s) was used to measure the strength of the relationship between the bird diversity and habitat variables (Clarke and Gorley, 2006). All these analyses were undertaken in Primer software (Clarke and Gorley, 2006).

RESULTS
Effects of Grazing on Species Diversity and Population Abundance

Sample-based rarefaction curves did almost reach an asymptote, and they did completely overlap with Chao 1 estimator in all cases (indicating 100% sampling completeness) and did converge closely with the observed Jacknife 2 richness estimator curves (indicating sampling representation of 87% and 88% in the grazed site, and 90% and 92% for the ungrazed site, respectively, during dry and wet seasons (Table 2). Comparisons between land use types and/or seasons were therefore made based on rarefied (to the lowest number of individuals recorded in a given treatment) number of observed richness (i.e. Sobs based on Mao Tau estimated richness values).

Overall, 1 236 observations (611 in the ungrazed site and 625 in the grazed site) were recorded. In total, 33 species were recorded across both land use types during both seasons, of which 24 (16 during the dry season and 22 during the wet season) and 25 species (19 during the dry season and 21 during the wet season) were recorded in the ungrazed and grazed sites, respectively (Table 2; see also the Appendix). Both observed (based on Mao Tau) and estimated (Jacknife 2) species richness values were almost similar between ungrazed and grazed sites either when seasons were pooled or separately compared (Table 2). Independent of site, these values were also similar between seasons (Table 2). Similar results were found when rarefied species richness of sites was compared both within and across season (Table 2. Figure 2 a-d). The 95% confidence intervals of rarefied species richness of sites did overlap in all cases (Figure 2a-d), indicating the lack of statistically significant difference between sites, as well between season. Nonetheless, the ungrazed site showed relatively greater evenness compared to the grazed site when seasons were treated separately as well when pooled (Table 2).

Both grazing and season, and their interaction, had significant effects on the abundance of bird assemblage (grazing Wald χ^2=265.622; season, χ^2=83.168; interaction, χ^2=73.518, in all cases, df=1, $P<0.05$). Independent of season, bird abundance was significantly greater in the grazed site than the ungrazed site, and wet season was significantly greater compared to dry season (Table 2). When each season were separately compared between the two sites, only wet season abundance data for the grazed site was significantly greater than the ungrazed site (Table 2).

Effects of Grazing on Assemblage Composition

Species richness of grassland habitat specialist guild, insectivore dietary guild, and high conservation priority guild were significantly greater in the ungrazed site compared to the grazed site (in all cases, $P<0.05$ at α = 0.05; Table 3). The percentage contribution of the different habitat guilds and conservation significance guilds to the over total richness and abundances of the ungrazed and grazed assemblages were significantly different (richness, habitat guild: χ^2 =12.93, df =3, $P<0.05$; conservation priority: χ^2=13.075, df=3, $P<0.05$; abundance, habitat guild = χ^2=81.016, $P<0.05$; conservation priority: χ^2 = 31.631, $P<0.05$). Both habitat specialist guild and high conservation priority guild relatively contributed greater proportion to the total richness and abundance of bird assemblage in the ungrazed site compared to the grazed site (Figure 3a-f).

Table 2: Mean (±S.E) abundance, species richness [based on observed (S_{ob}), Jacknife 2 estimator (S_{jk}) and rarefied (S_{rar})] and species evenness of the grazed and ungrazed sites across seasons

Groups	Site/season	Abundance		Species richness			Evenness
		n	Mean (± S.E.)	S_{ob}	S_{jk}	S^*_{rar}	
dry	ungrazed	639	43.21(± 1.76)[a]	16	19	16	0.23
	grazed	799	57.07 (±1.77)[a]	19	22	18	0.18
wet	ungrazed	617	44.07 (± 1.77)[a]	22	24	22	0.45
	grazed	1511	107.93 (± 2.02)[b]	21	24	18	0.18
Pooled season	ungrazed	1256	43.64 (± 1.25)[a]	24	26	24	0.30
	grazed	2310	78.48 (± 1.72)[b]	25	28	23	0.23
Pooled site	dry	1438	49.66 (± 1.33)[a]	23	26	23	0.38
	wet	2128	68.97 (± 1.65)[b]	26	27	25	0.28

Significant differences in mean abundance of birds between the ungrazed and grazed sites within each season and pooled season, and between seasons (based on pooled sites) are indicated by different superscript letters. * Rarefied species richness represents the number of species at the lowest number of individuals recorded in either of the treatments compared (e.g. 639 individuals for comparison between ungrazed and grazed sites during dry season).
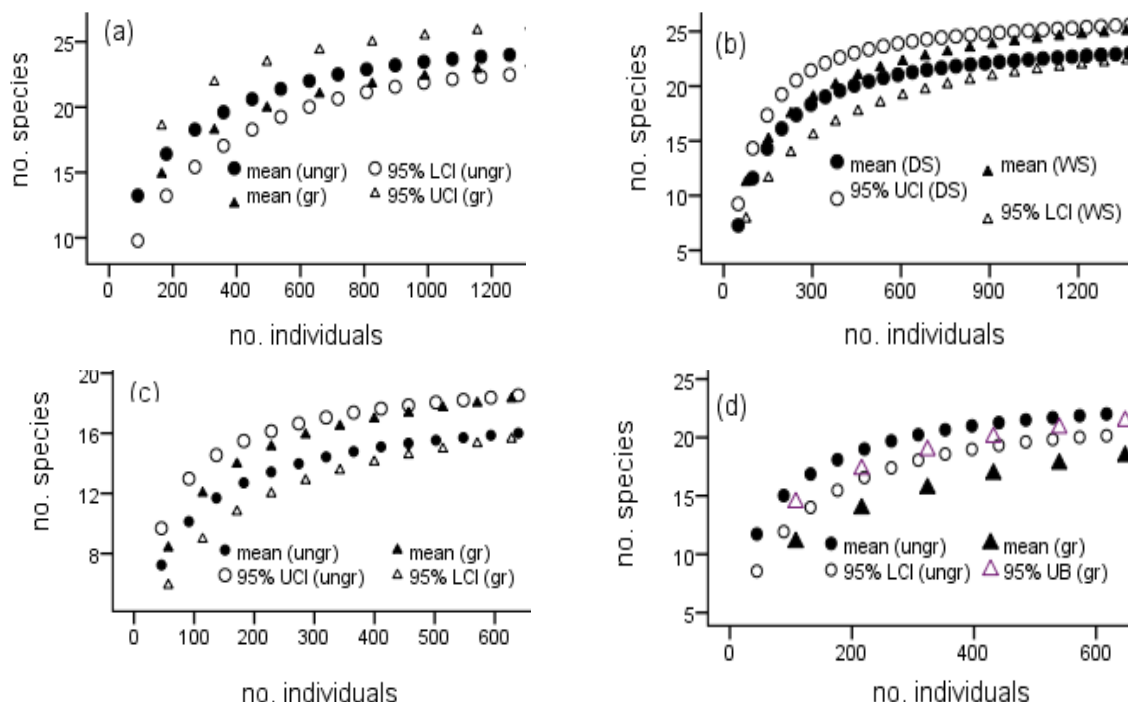

Figure 2: Rarefied species richness (by rescaling the observed Moa Tau sample-based rarefaction curves to number of individuals) of the ungrazed (ungr) and grazed (gr) sites (a), between seasons (b), between sites during dry season (c) and wet season (d). Given are mean of 500 randomizations and 95% lower confidence intervals (LCI) for the species rich site/season and the upper confidence intervals (UCI) for the species poor site/season. Differences were considered to be significant if the 95% LCI of the species rich site did not overlap with the UCI of the poor site

Table 3: Observed (S_{obs}) and rarefied [S_{rare} (95% CI)] species richness of the three sets of guilds in the grazed and ungrazed sites

Guild type	Guild category	site	No. individuals	S_{obs}	S_{rare} (95% CI)*
habitat	specialist	ungrazed	1173	17	16.8 (16.7-16.9)[a]
		grazed	765	11	11.0 (9.7-12.3)[b]
	generalists	ungrazed	83	7	7.0 (4.6-9.4)[a]
		grazed	1545	14	8.4 (5.6-11.1)[a]
diet	insectivore	ungrazed	525	13	13 (11.7-14.3)[a]
		grazed	765	11	10.6 (9.5-11.6)[b]
	others	ungrazed	731	11	11.0 (10.1-11.9)[a]
		grazed	1545	14	12.6 (9.8-14.4)[a]
conservation priority	High	ungrazed	736	9	8.8 (7.5-10.1)[a]
		grazed	710	5	5.0 (5.0-5.0)[b]
	low	ungrazed	520	15	15.0 (14.1-15.9)[a]
		grazed	1700	20	16.7 (13.4-20.0)[a]

* Significant differences in rarefied species richness between sites for each guild category are indicated by different superscript letters. Differences were assumed to be significant if the 95% lower confidence interval of the species rich site did not overlap with the upper interval of the poor site.
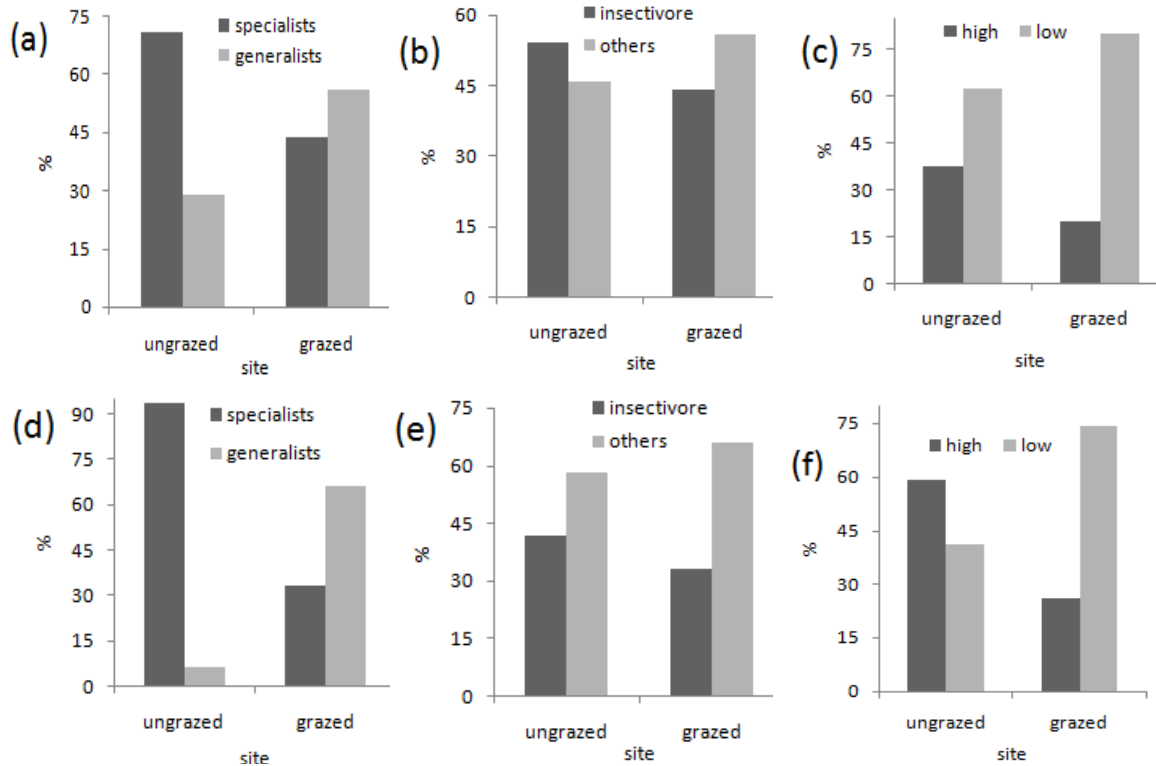
Figure 3: The relative contribution (measured as a percentage) each guild makes to the species richness (a-c) and abundance (d-f) of the bird assemblages in the ungrazed and grazed sites [(a) and (d) = habitat guilds, (b) and (e) = feeding guilds, and (c) and (f) = conservation importance guilds)].

Analysis of similarity (ANOSIM) revealed significant differences in bird assemblages composition between sites both across seasons (Global R = 0.890, $P<0.05$) and when seasons were considered separately (dry season, R =, 0.881; wet season, R = 0.900, in both cases, $P<0.05$; Figure 4). Furthermore, assemblages were also significantly different between seasons both across sites (R = 0.506, $P<0.05$) and within site (ungrazed site, R = 0.356; grazed site, R = 0.656. in both cases, $P<0.05$; Figure 4).

Effects of Grazing-induced Vegetation Change on Bird Assemblage

The result of BEST analysis revealed that the assemblages of both the ungrazed and grazed sites showed significant relationships with the vegetation variables considered. The ungrazed site assemblage was best correlated with heights of shrub and grass (R = 0.405, $P<0.05$) and that of the grazed site with heights of herb and grass and cover of herb and bare ground (R = 0.324, $P<0.05$).
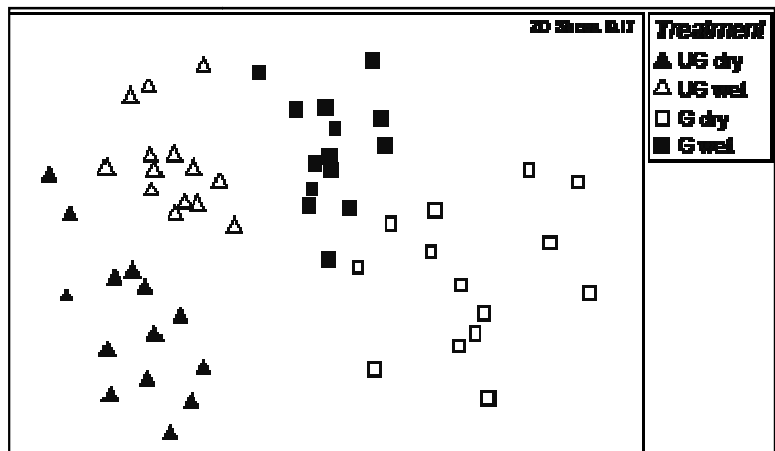

Figure 4: Non-metric multi-dimensional scaling (nMDS) ordination of bird assemblages from each sampling point in the grazed and ungrazed sites across wet and dry seasons in the Afromontane grasslands of the Bale mountains (UG dry = ungrazed site dry season; UG wet = ungrazed wet season; G dry = grazed site dry season; and, G wet = grazed site wet season).

Addisu Asefa *et al.,*

Sci. Technol. Arts Res. J., April-June 2015, 4(2): 112-121

DISCUSSION

Our study demonstrated that, depending on season, livestock grazing in Ethiopian Afromontane grasslands to have neutral, positive or negative effects on overall species richness, abundance and composition of bird assemblages. Non-significant difference was found between grazed and ungrazed sites in species richness both across seasons and in dry season. In contrast, bird assemblage abundance during dry season and across seasons was significantly greater in the grazed site than the ungrazed site. These nearly equal richness and greater abundance in the grazed site could be as a result of the persistence of some typical grassland bird species, together with the invasion/colonization of non-grassland (generalist, see discussion below) avian species from the surrounding cultivation fields and woodlands into the grazed grassland site.

The adversity of livestock grazing on birds of the area was more revealed to be altering assemblage composition (species evenness and guild composition) than decreasing overall richness and abundance. As predicted, our study found a higher richness of species considered to be grassland/shrub-land habitat specialist, insectivore feeding and high conservation priority guilds in ungrazed site. These findings were further supported by the relative percentage contributions they made to the overall richness and abundance of bird assemblage in each respective site. Consequently, bird assemblages of the two sites were found to be significantly distinct. There are several reasons why this might be the case. Grazing affects birds in several ways, including change in vegetation structure and food availability and abundance (Muchai *et al.*, 2002; Barbaro and van Halder, 2009, Rahmig *et al.*, 2009). These effects, depending on season, grazing intensity and species-specific ecological traits (e.g. preferred habitat/dietary requirements), can be either positive or negative (Whittingham and Devereux, 2008; see also discussion below). The assumption is that grazing reduces vegetation height and cover and the abundance and availability of food in tall grass/shrubby vegetation (Jensen, 1985; McIntyre *et al.*, 2003; Tallowin *et al.*, 2005). Thus, while bird species that prefer tall vegetation habitat for feeding and reproduction respond negatively to grazing, those species preferring open habitat and short grass respond positively (Martin and Possingham, 2005; Evans *et al.,* 2006; Whitehorn *et al.*, 2011). Thus the dissimilarity in assemblage composition found between the two sites can generally be explained to be as a result of varying responses of different species to grazing-induced habitat changes.

Seasonal difference in habitat variables was non-significant, which may be expected as grazing in the area takes place throughout the year, perhaps indicating that the effect of grazing on the vegetation variables considered in this study to be constant. This finding seems to be unexpected, coupled to the significant difference found in bird assemblage composition between seasons, given our argument that any difference found between sites or seasons to be attributed to difference in vegetation structure. This lack of consistency in variations of bird assemblage composition and habitat variables between seasons may suggest that there might be other important habitat/environmental variables that might vary with season and affects bird assemblage composition, which we should have to measure. Habitat variables that potentially influence the occurrence and abundance of bird species, but we failed to measure, include plant phenology such as availability and abundance of flowers and seeds. This supposition could hold true as both specialist (e.g. sunbirds) and opportunistic [e.g. baglafecht weaver (*Ploceus baglafecht*) and slender-billed starling (*Oegnathus tenuirostris*)] nectar-feeding bird species were not encountered during dry season survey when flowers were not available (see Appendix). These species were however abundant in both sites during wet season survey when *Knophofia foliosa* plants bear flowers.

In addition to its influence on vegetation phenology, season, independently or interactively with grazing, affects the presence and abundances of other birds' feeding resources (e.g. invertebrates; see also, Gibson *et al.*, 1992; Buckingham *et al.*, 2004). For example, the abundance of wattled ibis (*Bostrychia carunculata)* the most abundant insectivore bird species in the grazed site, was markedly high during wet season compared to dry season (620 *vs*. 245; see Appendix). However, we do not have any empirical evidence for such seasonal variations in invertebrate richness and abundance in our study sites to attribute our results to such variations, and further research is required to test if this is one of the possible causes for the observed assemblage differences. Nonetheless, regardless of season, bird assemblages of the two sites showed positive correlations with shrub height and/or grass height. Given that grassland specialist and/or high conservation priority species were found to be negatively impacted by grazing, coupled to the findings that these vegetation features are those most adversely affected by grazing in the area (Yosef Mamo *et al.*, 2014), the observed bird-habitat relationships have significant conservation implications. Allowing livestock grazing in the protected (i.e. currently ungrazed site) site on the future will lead to loss of several grassland specialist and/or high conservation significant species.

CONCLUSIONS

This study showed that livestock grazing is neither equally beneficial nor harmful to all bird species, and the responses of birds to grazing depend on species-specific habitat and/or food preferences. Although overall assemblage species richness and/or population abundance was nearly equal between sites, grassland habitat specialist, insectivore and/or high conservation priority species were negatively affected due to grazing while the other guilds showing the opposite response. Thus the effect of grazing on birds of the area is changing assemblage composition rather than resulting into declined assemblage species richness. Given that the effects of grazing could be positive or negative, future studies focusing on such impact assessment should be undertaken and evaluated in terms of the conservation goal of the habitat in question (e.g. increasing or maintaining of overall assemblage richness and abundance, or of groups of species typical to that habitat; or of species with high conservation priority). This approach (comparison of the responses of different guilds) has enabled us uncover the effect of grazing which was obscured while the analyses/comparisons were made at assemblage level.

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Conflict of Interest

All the authors don't have any conflict of interest

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