How does surface mining impact surrounding miombo woodland bird communities?

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Abstract

Africa is estimated to contain almost one quarter of global biodiversity, but also a third of global non-fuel mineral resources. Together with relatively low past exploitation, this has led to a surge in mining investment and development over the past few decades. Mining negatively impacts biodiversity in numerous ways. One hotspot of mining activity in Africa is the Central African Copperbelt, which contains vast copper reserves and half of global minable cobalt. The Central African Copperbelt is located roughly in the centre of the miombo ecoregion, a biodiverse area that is listed as a global conservation priority, which provides ecosystem services that support over 100 million livelihoods. Despite its ecological and social importance, research on the impacts of mining and possible mitigation measures in the ecoregion is limited. This study was conducted in miombo woodlands inside and outside an ecological protection zone surrounding an open pit copper mine in north-western Zambia. It tested effect of distance from the mine, habitat structure and woodland protection on total avian species richness and the species richness of frugivores, granivores, insectivores, woodland and open habitat specialists, breeding and non-breeding species and Zambezian endemics. The results showed that distance from the mine and habitat structure did not significantly explain variation in total species richness or in the species richness of any of the ecological guilds tested. The most likely explanation for this is that mine effects on avian species richness are not detectable at the scale used in this study, implying that by the closest sampling sites (500m from the mines edge) the woodland had mitigated the mine’s effects. This suggests that using woodlands as a barrier to mine pollution may be an effective form of mitigation. The results also showed that species richness of frugivores, insectivores, woodland habitat specialists, Zambezian endemic species and breeding species was not significantly explained by protection. However, total species richness and the species richness of granivores, open habitat specialists and non-breeding species all increased significantly outside of the ecological protection zone, which corresponded to woodland surrounded by cropland. These findings highlight the importance of unprotected woodland patches in the conservation of miombo bird species, since these may persist (likely in lower numbers) even in such fragmented patches within transformed habitats. Cropland expansion threatens woodland patches throughout the miombo ecoregion, and while protected areas are undeniably important in biodiversity conservation, matrix habitats may also potentially contribute to the maintenance of miombo species.

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Plagiarism Declaration

1. I know that plagiarism is wrong. Plagiarism is to use another's work and pretend that it is one's own.
2. I have used the journal Conservation Biology as the convention for citation and referencing. Each contribution to, and quotation in, this project from the work(s) of other people has been attributed and has been cited and referenced.
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4. I have not allowed and will not allow anyone to copy my work with the intention of passing it off as their own work.

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Introduction

Mining development in Africa and its associated impacts

Africa contains almost one third of the global non-fuel mineral resources, but by 2009 under 5% of global exploitation had occurred on the continent, and large areas remained geologically unexplored (Taylor et al. 2009). These large, unexplored deposits mean that there is huge growth potential for the extractive industry in Africa, which has resulted in it becoming one of the global centres of extractive industry investment (Ahmed et al. 2021). One example of this is Chinese investment into sub-Saharan African mining sectors, which was estimated at US$58 billion between 2005 and 2017, resulting in China having a 7% control of the value of total mine production in the region (Mudronova 2018; Ericsson et al. 2020). Other countries are also investing large amounts into the sector, such as Australian mining companies, which by 2011 had financed over US$45 billion of mining developments (USGS 2011).

This surge of investment has led to mining booms in many African countries (Busia & Akong 2017). A recent study across sub-Saharan Africa, found that 58% of the 469 company-owned and community-managed mines identified had become operational since 2000, and during this period the areal extent of mines in the region expanded by 1,892 km^2 (Ahmed et al. 2021). As the extractive industry in Africa grows, so too will its direct or indirect socio-economic and environmental impacts. (Edwards et al. 2014). This is concerning as approximately one quarter of global biodiversity is found in Africa (UNEP-WCMC 2016).

The positive socio-economic impacts of mining are significant and are often used as part of the justification for new mining developments. These include employment creation, infrastructure development, and provision of raw materials required to produce the modern amenities that much of society requires for their day to day lives (Fleming & Measham 2014; von der Goltz & Barnwal 2019). Thus, mining can play an important role in contributing to local and regional economies. For example, the copper and cobalt mining industry in Zambia provides approximately 12% of Zambia's gross domestic product (Kolala & Dokowe 2021).

The negative socio-economic impacts of mining can be substantial and may include the displacement and resettlement of local communities, in some cases against their will (Downing 2002; Owen & Kemp 2015). Mining often leads to a loss in ecosystem services, which may have been important for the well-being of local communities (Roy et al. 2018; Boldy et al. 2021). Pollution produced by mining can also have severe negative impacts to human health, including the loss of clean drinking water, and illness caused by direct exposure to harmful pollutants such as heavy metals (Gautam et al. 2012; von der Goltz & Barnwal 2019; Mwaanga et al. 2019).

The impact of mining on biodiversity

Globally, biodiversity is under threat from a wide range of anthropogenic activities, one of which is mining (Sonter et al. 2018). The negative impacts of mining are often most detrimental in areas containing the highest biodiversity (Roy et al. 2018). This is especially concerning considering that in 2016, 86% of mines globally were in areas of high or intermediate plant diversity (Murguía et al. 2016).

A major impact is the physical process of mining, which involves the clearing and transformation of large areas, resulting in the loss, fragmentation, and degradation of natural habitats

(Siqueira-Gay et al. 2020). Another impact is that extractive industries are also often responsible, at least in part, for the downgrading, downsizing, and degazetting of protected areas. One example is Zambia, which in 1998 downgraded 63,585 km^2 of protected areas, in large part to allow mining development (Mascia et al. 2014). This prioritization of mining over protected areas is concerning considering that 14% of protected areas globally hold mines within their bounds or are less than 10 km away from one, and in sub-Saharan Africa the number of mines less than 10 km from a protected area doubled between 2000 and 2020 (Durán et al. 2013). The geological regions containing valuable mineral deposits may be the same areas that contain endemic species (Mayani-Parás et al. 2019; Salles et al. 2019). Mining activity in these areas can directly negatively impact these endemic species as well as exacerbating other threats they face (Mayani-Parás et al. 2019; Salles et al. 2019).

Mining operations are often partly responsible for infrastructure developments, such as the expansion or creation of transport networks. These developments can have a large environmental impact, through their construction which causes habitat loss and fragmentation, and through wildlife collisions with vehicles causing population declines (Laurance et al. 2009; Grilo et al. 2020). These infrastructure developments and the prospect of extractive industry related employment encourage human migration into previously difficult to access, sparsely populated areas leading to increased demand for bushmeat as a protein source for the growing population (Rodrigues et al. 2021). The new transport networks also make areas more accessible for other forms of wildlife exploitation; for example, proximity to expanding infrastructure was found to be an important predictor of poaching of African forest elephants (van Vliet et al. 2012; Lindsey et al. 2013).

Pollution produced by mining operations can also have a major impact on biodiversity. Air, water, and soil pollution can occur, particularly in operations with poor environmental standards or countries with weak enforcement (Razo et al. 2004; Pourret et al. 2016; Fugiel et al. 2017). These forms of pollution can have a large impact on biodiversity in affected areas, negatively impacting fitness and lowering species richness and abundance in a wide range of taxa (Saha & Padhy 2011; Palacios-Torres et al. 2018; Giam et al. 2018). Mining activity can also generate a large amount of noise pollution, either constantly through machinery operation, or in short intense bursts when activities such as blasting occur (Duarte et al. 2015; Lokhande et al. 2018). This noise pollution is known to affect the behaviour and fitness of a wide range of animal species, which can lead to reduced species richness and diversity in affected areas, with specialist species often being most affected (Patricelli & Blickley 2006; Francis et al. 2009; Kight & Swaddle 2011; Ditmer et al. 2021). Mining infrastructure also creates light pollution, which is known to negatively impact animals, by altering light-dependent biological processes causing reduced fitness and declines in some species (Dominoni et al. 2013; Schoeman 2016; Owens et al. 2020).

Another pollutant generated by mining is greenhouse gases, mainly through the large amounts of energy required for extraction and purification, as well as through the clearing of land for mining-related development (Farjana et al. 2019). It was estimated that in 2018 approximately 10% of global energy-related greenhouse gas emissions were from primary mineral and metal production (Azadi et al. 2020). Climate change as the result of greenhouse gas emissions poses a major threat to global biodiversity (Dawson et al. 2011; Bellard et al. 2012).

Mitigating the impacts of mining on biodiversity

Responsible mining companies seek to avoid and minimise negative environmental impacts. When these impacts are unavoidable, mining companies seek to mitigate and offset them, particularly when required through legislation (Ranängen & Zobel 2014). However, weak governance and enforcement capacity along with corruption in some countries can result in poor environmental protections (Edwards et al. 2014).

A key focus is reducing waste wherever possible and ensuring its proper disposal. This is particularly important in the case of pollutants that can have widespread negative environmental impacts. For example, tailings produced during purification must be stored underwater, in secure dams or they risk contaminating nearby water bodies (Kossoff et al. 2014). Improving efficiency can also help to reduce the environmental impact of mining operations. For example, new technologies can reduce the energy required for grinding ore, thus reducing greenhouse gas emissions (Norgate & Haque 2010). Another key mitigation measure is undertaken after mining in the area is over, when the mine can be decommissioned, and waste and structures removed, and the land reclaimed. Land reclamation is often a challenging process, and its success varied, with habitat taking decades to return to its original state, if it does at all (Sheoran et al. 2010; Feng et al. 2019). While mitigation measures may be complicated and expensive, some mines have incorporated them successfully (Allan 1995).

The Central African Copperbelt

One of the hotspots of mining activity in Africa is the Central African Copperbelt, located in north-western Zambia and southern Democratic Republic of Congo (Ahmed et al. 2021). This geological region contains one of the largest sediment-hosted stratiform Copper–Cobalt provinces on earth and is estimated to contain at least 140 Mt copper and half of global minable cobalt (Cailteux et al. 2005). The size and quality of deposits have meant that most of the copper production in Africa comes from this region (Taylor et al. 2009). The Central African Copperbelt is located roughly in the centre of the miombo ecoregion.

The miombo ecoregion

The miombo ecoregion spans approximately 10% of Africa's landmass, occurring in most central and southern African countries, and covering the majority of Angola, Malawi, Mozambique, Tanzania, Zambia and Zimbabwe (Malmer 2007). A defining feature of this ecoregion is its deciduous woodlands which are dominated by species from three closely related genera*, Brachystegia, Julbernardia* and *Isoberlinia* (White 1983; Campbell et al. 1996). The ecoregion occurs on geologically old, nutrient-poor soils, and falls within a unimodal rainfall zone with a long dry-season (Campbell et al. 1996). Its woodlands are divided into wet and dry miombo; wet miombo (> 1000 mm of annual precipitation) is floristically richer and has a greater canopy height than dry miombo (Chidumayo 1987; Jinga & Palagi 2020).

The miombo ecoregion is a biodiverse area that has been listed as a global conservation priority (Mittermeier et al. 2003). It contains approximately 8,500 plant species, of which more than 54% are endemic, with Zambia being the centre of floral endemism (Campbell et al. 1996; Moura et al. 2017). The miombo ecoregion is home to large populations of a variety of large herbivorous mammals and the carnivores that rely on them, and contains many endemic or near-endemic animals, including 51 bird species, 83 reptile species and 36 amphibian species (Timberlake & Chidumayo 2001).

The miombo ecoregion is not only important to biodiversity but also to humans, providing livestock fodder, building materials, wild foods and fruits, traditional medicines, wood or charcoal as a source of energy, and arable land (Gumbo et al. 2018). It is estimated that the ecosystem services it provides directly support more than 100 million people's livelihoods and reduce the chances of households falling further into poverty when faced with economic or environmental stress (Syampungani et al. 2009; Dewees et al. 2010).

Unfortunately, the miombo ecoregion is facing a range of anthropogenic threats. Unsustainable harvesting and agricultural practices cause degradation and loss of woodlands, negatively impacting the species that rely on them (Chidumayo 2013; Kalaba et al. 2013; Jew et al. 2015). Fire that are too frequent or of high intensity, degrade woodlands and negatively impact succession (Ryan & Williams 2011). The ecoregion is also under threat from climate change, which is predicted to cause up to a 41.6% contraction of wet miombo and a 22.7% expansion of dry miombo by 2070 (Jinga & Ashley 2019; Jinga & Palagi 2020). A continued decline in populations of large herbivores, some of which are keystone species, is also likely to negatively impact fauna and flora within the ecoregion (Craigie et al. 2010; Chase et al. 2016). The valuable geological deposits within the ecoregion mean extractive industries and their associated impacts also pose a threat (Edwards et al. 2014; Peša 2021). A growing population within the ecoregion is also likely to increase resource demand and further exacerbate these threats (Syampungani et al. 2009).

Project rationale and research aim

There is a need for more research to better understand the impacts of mining on biodiversity and the ways in which these impacts can be mitigated or offset (Edwards et al. 2014; Sonter et al. 2018). This type of research within the miombo ecoregion has been very limited, despite the large geological deposits and important biodiversity the region contains. At the centre of this ecoregion lies Zambia, a country with a long history of mining and its associated negative environmental impacts (Mwaanga et al. 2019; Peša 2021). Mining effects have been shown to negatively impact avian species richness in surrounding habitats (Saha & Padhy 2011; Deikumah et al. 2014; Alvarez-Berríos et al. 2016). The distributions of many African bird species centre on Zambia, especially the distributions of Zambezian endemics, making it an ideal study location (Leonard 2001).

While there is a lack of research on African avian indicator species, the role of birds as imperfect but important indicators of species richness and diversity of other taxa (as well as overall biodiversity) has been well documented (Fraixedas et al. 2020). Previous studies have found a correlation between avian species richness and the species richness of woody plants, aquatic herpetofauna and lepidoptera (Blair 1999; Kati et al. 2004; Pearman & Weber 2007). In areas where birds are speciose, they can make good surrogates of overall biodiversity to improve biodiversity conservation decisions (Lewandowski et al. 2010; Larsen et al. 2012). Therefore, studying the effect of mining and protection on miombo woodland bird richness could also provide insights into the richness and health of other taxa.

The key mine effects that are expected to impact surrounding bird communities in this study are noise pollution and air pollution. Prior research has found avian species richness does decline with increasing noise pollution, with breeding species being most affected (Francis et al. 2009; Perillo et al. 2017). Diet has also been shown to influence how noise pollution affects birds, with omnivores and carnivores being more sensitive than species with plant-based diets (Francis 2015). Air pollution has been shown to negatively impact avian fitness in numerous ways, leading to reduced species richness (Sanderfoot & Holloway 2017). Research around copper processing plants have found that avian species richness decreases as air pollution increases (Eeva et al. 2012; Belskii & Belskaya 2013).

In this thesis, I studied miombo bird communities around an open pit copper mine in north-western Zambia. Based on evidence from previous studies, I expected that the mine's effects would result in decreased species richness at sites closer to the mine and that insectivores and breeding species would be more affected than frugivores or granivores. I also expected that the species richness of birds detected vocalising during the dawn chorus would also decrease at sites closer to the mine.

Moreover, I expected that the mine's effects should decrease with distance from mine, and that the large, ecological protection zone that surrounds the mine should help to further decrease these effects, especially noise and air pollution. This is because trees have been shown to trap various forms of particulate air pollution, which are produced by mining operations (Beckett et al. 2000; Nowak et al. 2018). Trees have also been shown to act as effective barriers to noise pollution (Maleki & Hosseini 2011; van Renterghem et al. 2012). The ecological protection zone surrounding the mine provided me with the opportunity to study its effectiveness as a pollution trap. It also provides the opportunity to test how species richness varies inside and outside the ecological protection zone. If habitat quality is better and disturbance is lower inside the ecological protection zone, then we should expect total species richness and the species richness of all the ecological guilds tested (except for granivores and open habitat specialists which prefer habitats not contained in the ecological protection zone) to increase with protection.

My main research aims were as follows:

- To test whether the mine affects avian species richness in surrounding woodland, and if so at what spatial scale.
- To test whether species with certain shared traits such as diet, habitat preference, endemism or breeding status are more or less susceptible to mine effects.
- To test whether woodland habitat structure varies with distance from the mine and inside and outside the protected area, and whether this explains variation in avian species richness.
- To test how avian species richness varies inside and outside of the ecological protection zone.

Methods

Study Area

The study was conducted from 14 September to 8 November 2020, around Sentinel Copper Mine (12°15’S, 25°18’E) in the North-Western Province of Zambia, south-central Africa (Figure 1). The region has a humid subtropical climate with a unimodal rainfall pattern resulting in two main seasons, the dry season (May to October) and the rainy season (November to April). The northern half of Zambia is wetter than the south, and its annual precipitation ranges between 1100 mm and 1400 mm (Waldman et al. 2019). According to 2010 census data, North-Western Province had the lowest population density of the ten provinces, and over three quarters of the population were based in rural areas (Central Statistical Office 2012). The 2010 census data also showed that the largest employers in the province are the agricultural industry, followed by the mining industry.

The study area falls within the Miombo Woodlands ecoregion with a wet miombo vegetation type (Byers 2001). The study area is located around Sentinel Copper Mine, a large open pit mine in the centre of a 51,000-hectare mining concession owned by First Quantum Minerals. Of this concession, just over 11,000 hectares of land surrounding the active mining and processing areas has been designated as an ecological protection zone, forming Kalumbila Wildlife Sanctuary. I visited study plots both inside the ecological protection zone, and outside in tribal lands. The woodlands within the sanctuary are mature and largely intact. Seasonal burning is not carried out and a wide range of large, native herbivore species have been reintroduced. These include buffalo (*Syncerus caffer*), wildebeest (*Connochaetes taurinus*), sable (*Hippotragus niger*), eland (*Taurotragus oryx*), zebra (*Equus quagga*), hartebeest (*Alcelaphus buselaphus*), waterbuck (*Kobus ellipsiprymnus*), kudu (*Tragelaphus strepsiceros*), puku (*Kobus vardonii*), bushbuck (*Tragelaphus scriptus*), lechwe (*Kobus leche*), and impala (*Aepyceros melampus*). Outside the sanctuary, woodlands are also mature but are much more fragmented, and sections have been cleared for subsistence agriculture and charcoal production. Seasonal burning occurs towards the end of the dry season and there is little to no presence of domestic or wild herbivores.

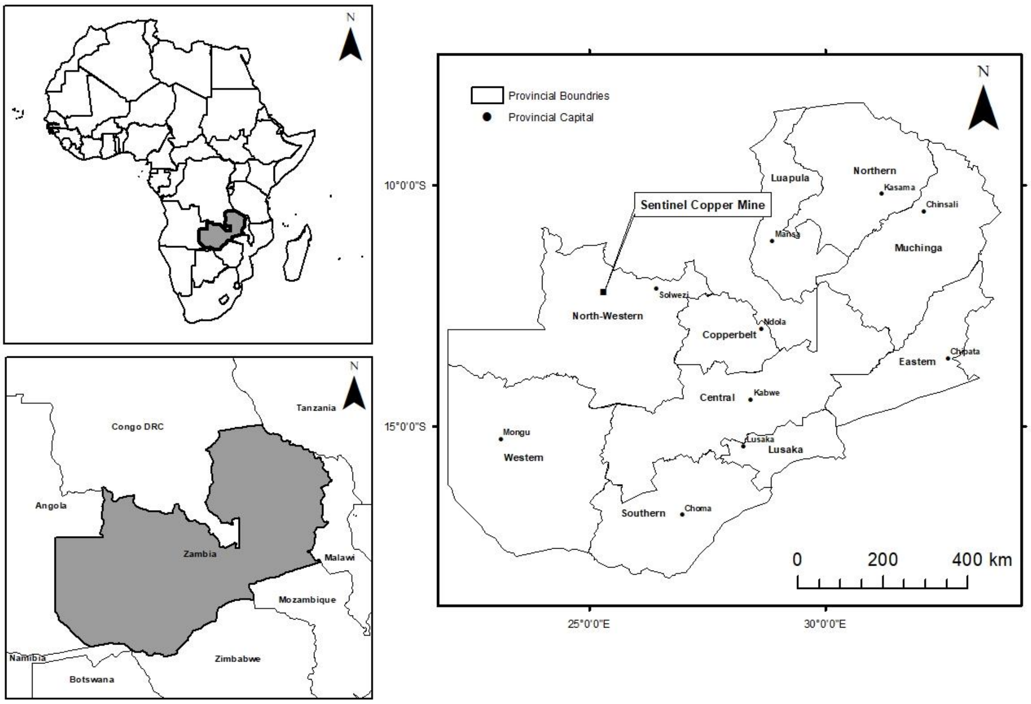


Figure 1. The location of Sentinel Copper Mine in the North-Western Province of Zambia, south central Africa

Bird Surveys

I carried out dawn chorus recordings at 14 sites, and point counts at 41 sites, in miombo woodland around Sentinel Copper Mine (Figure 2.) Sites were not selected in the part of Kalumbila Wildlife Sanctuary to the southwest of the mine, as a lack of road networks makes this area of the ecological protection zone virtually inaccessible by vehicle. Sites were also not selected in the area outside the ecological protection zone north of the mine, due to the presence of a large, circular mine tailings dam and associated safety risks. Sites were also selected outside of the roughly circular mine safety exclusion zone, which includes active mining and processing areas.

Within the suitable sampling area, 14 sites were randomly selected for dawn chorus monitoring, half of which were selected as close as safely possible to the edge of the active mining area (less than 1 km), and the other half between three and four kilometres away from the first set.

Within the suitable sampling area, point count sites were randomly selected at seven different distance bands, each 1 km apart, starting 500m from the edge of the active mining area, which was as close as safely possible. Sites were selected inside and outside of Kalumbila Wildlife Sanctuary at the last three distance bands, which were 4-6 km away from the first band.

Random selection was achieved by assigning all potential sites within a distance band a number and using a random number generator to select sample sites. All sites were a minimum of 500 m apart from each other to ensure distinct populations were sampled. Sites were also a minimum of 200 m from roads, clearings, or waterbodies, to ensure sites were as similar as possible and edge effects and disturbance did not influence avian detection and presence.

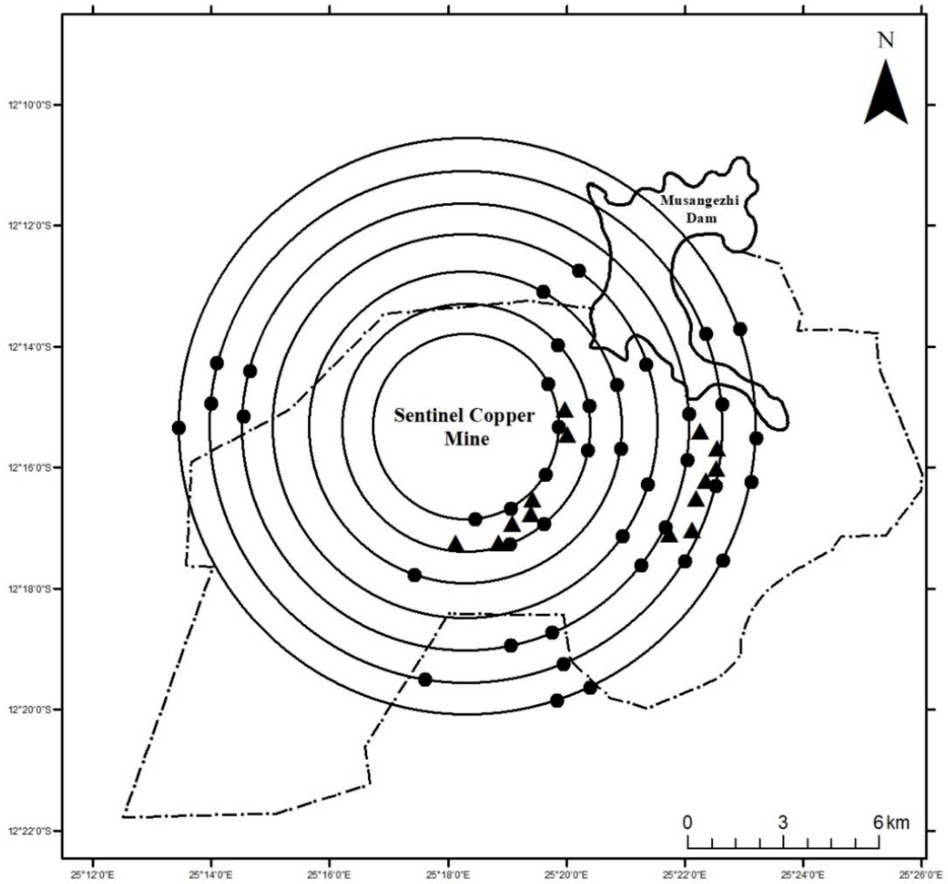


Figure 2. Map showing the locations of study sites around Sentinel Copper Mine. The circles represent sites selected for point counts, and the triangles represent sites selected for dawn chorus monitoring. The boundary fence of Kalumbila Wildlife Sanctuary and the mine concession is represented by the dashed and dotted line.

All study sites were assigned in advance in Google Earth and then located in the field using handheld GPS devices. On arrival at a site selected for a point count, the observers (myself and Akeelah Mwale) allowed 5 minutes for birds to settle following any initial disturbance caused by our presence. The point count method was then used to record every bird sighted within a 50 m radius from the observers, over a 20 minute period (Gregory et al. 2004). Wherever we could not identify a species, the observer attempted to photograph the individual and confirm identification afterwards.

During all point counts, a sound recorder (Tascam DR-05X, manufactured in China) was suspended 1.5-2 m above the ground to capture avian vocalisations during the count using its built-in microphone. Counts were conducted twice at each site, in the morning and evening, with at least two weeks between visits. They were not carried out between 10:30 and 14:00 as the high temperatures (>30°C at 12:00) would decrease avian activity and therefore detection (Robbins 1981). All counts were conducted by the same pair of observers (R. Hickman and A. Mwale) to avoid observer bias affecting comparisons between sites.

Sites selected for dawn chorus monitoring were reached at least 30 minutes before sunrise, which was around 05:59 during the 3-week sampling period (28th September to 15th October). Sound recorders were suspended, after which the observers moved at least 200m away. The recorders were not collected before 30 minutes after sunrise.

Vegetation and Habitat Measures

At the end of each visit to a study site, vegetation surveys were conducted. A 20-metre quadrat was marked out around the point used for dawn chorus recordings and point counts. The percentage of ground within quadrats covered by leaves, grasses, termite mounds, shrubs, fallen wood, bare soil, and rock was estimated. The percentage of canopy cover was estimated and recorded. All the trees within quadrats were classed as small (0.5–3 m high), medium (3–6 m high) or large (over 6 m high) and the number of trees in each class recorded. The number of fruiting and flowering trees within each quadrat were counted. If there were signs of a seasonal fire in the last five to six months (since the start of that year's dry season) this was also recorded.

Audio Analysis

Point count recordings were 20 minutes in length, and dawn chorus recordings were 40 minutes in length. Dawn chorus recordings were trimmed to 20 minutes before and after dawn to ensure the same time sample was used for all sites. For each recording, all vocalisations were identified and the time at which a species was first detected was noted. This was done together with Collins Moya who has extensive experience of Zambian bird vocalisations. Wherever the observers were uncertain about a vocalisation, the clip was shared with Gabriel Jamie, Claire Spottiswoode and Frank Willems for identification.

Avian Ecological Attributes

The ecological attributes of recorded bird species were identified. These comprised whether the species was migratory (scored as resident, intra-African migrant, or Palearctic migrant); whether its, breeding season fell during the study period; whether it was, a Zambezian biome endemic, and its habitat preference (scored as woodland or open habitat) and primary feeding guild (nectarivores, granivores, insectivores, carnivores, frugivores, piscivores, or omnivores). Information on migration, habitat preference and diet were sourced from '*The Birds of Africa*' Volumes 1–7 (Brown et al. 1982; Urban et al. 1997). Species were scored as Zambezian biome endemics following *'Important Bird Areas in Africa and associated islands – Zambia'* (Leonard 2001). The *Birds of Zambia* phone application was used to score which species were breeding or not breeding during the study period (Solomon et al. 2019).

Data Analysis

Data analyses were performed in R v4.0.2 (R Core Team 2020) and figures were produced using the ggplot2 package (Wickham 2016). The vegan package (Oksanen et al. 2019) was used to conduct diversity analysis and community ordination. Results were considered significant at $p < 0.05$.

Dawn Chorus Recording Data

The total number of species identified vocalising at each of the near sites (less than 1 km from mine) and each of the far sites (3-4 km from near sites) was calculated and termed total vocalising species richness. Unpaired two-sided t-tests were then used to test whether there was a significant difference in the mean vocalising species richness of near and far sites for all species, and for each of the different ecological guilds. Generalised linear models (with a Poisson error distribution) were then used to test the effect of the number of large trees and canopy cover (as proxies for habitat structure) on vocalising species richness.

Point Count Data

Data on bird species detected on morning and afternoon visits to a site were pooled to produce a total species richness for each site. This consisted of both species detected from sound recordings, and those detected visually on point counts. Species were classified by selected ecological attributes, and species richness then calculated for each site based on these attributes. A total of nine response variables were included in the analysis: total species richness, species richness of three primary feeding guilds (frugivores, granivores and insectivores), species richness within habitat preference group (open habitat and woodland), Zambezian endemic species richness, and species richness for breeding and non-breeding groups. Carnivorous, piscivorous, omnivorous or nectarivorous species were not used as a response variable as they had few observations for each site (<2).

The explanatory variables used were distance from the mine, number of large trees, canopy cover, and protection status (i.e., inside or outside the ecological protection zone). Habitat type was not modelled as mature wet miombo woodland was the only habitat sampled. Similarly, weather was not modelled as the weather varied little during the survey period, remaining dry and hot (20-30 °C), and on the rare occasions severe weather (high winds, rain) did occur, sampling was not carried out. Generalized linear models were used to separately test the effect of distance from the mine, protection status, number of large trees and canopy cover on each measure of species richness. To account for sites outside the ecological protection zone only occurring in the furthest three distance bands, generalized linear models were also used to test the effect of distance and protection in the same model on the various measures of species richness. Generalised linear models were also used to test the effect of distance from the mine and protection status on the number of large tree and canopy cover, both of which are measures of habitat structure. The function ‘summary’ was used to assess each model’s coefficients. The function ‘anova’ was then used to compute analysis of deviance for all the fitted model objects and explore how much residual deviance, if any, was explained by the variable. To correct for multiple testing in different ecological guilds, a Bonferroni correction was applied to the P-value of 0.05 (i.e. the significance value was devided by nine) to correct for multiple testing of the data in different ecological guilds (Bland & Altman 1995). After Bonferoni correction the significance level was 0.0056. A correlation matrix was also produced to test the whether the response variables were correlated and so to assess the extent to which models with different response variables were independent.

Results

1) Does avian species richness vary with distance from the mine?

Vocalising species richness in dawn chorus recordings

Across all the sites sampled at dawn, sixty bird species were identified through their vocalisations. At sites near to the mine (within 1 km), 43 species were detected and at sites further from the mine (3-4 km away), 41 species were detected. Of the 14 sites sampled (7 close, 7 far), the most frequently detected species were Dark-capped Bulbul (*Pycnonotus tricolor*), Fork-tailed Drongo (*Dicrurus adsimilis*), Kurrichane Thrush (*Turdus libonyana*) and Black-backed Puffback (*Dryoscopus cubla*) which were detected at 13, 11, 9 and 9 sites respectively. One globally threatened species was detected, the Southern Ground-hornbill (*Bucorvus leadbeateri*) (IUCN 2021).

There was no significant difference in the mean vocalising species richness between near and far sites, both when overall species richness was considered, and when it was subdivided into different ecological guilds (Table 1). The mean vocalising species richness of granivores, open habitat, and non-breeding species was very low owing to few detections of these species in recordings.

Table 1. Comparison of mean species richness from dawn chorus recordings at sites near (within 1 km) and far (3-4 km away) from the mine using unpaired, two-sided t-tests.

Vocalising Species Richness Group	Mean Vocalising Species Richness Near	Mean Vocalising Species Richness Far	t statistic	Degrees of freedom	p-value
All Species	12.6	14.9	0.852	11.9	0.411
Frugivores	2.43	3.43	1.65	11.2	0.127
Granivores	1.29	1.14	-0.217	10.3	0.833
Insectivores	7.86	9.43	0.794	11.3	0.444
Open Habitat	1.29	2.29	1.97	12.0	0.073
Woodland Habitat	11.3	12.6	0.539	11.9	0.600
Breeding	11.0	13.1	0.841	11.9	0.417
Non-Breeding	1.57	1.71	0.346	8.82	0.737

Species richness detected on point counts

A total of 131 species were detected at the 41 point count sites. The most detected species were Fork-tailed Drongo (*Dicrurus adsimilis*) which was detected at 29 sites, Yellow-fronted Tinkerbird (*Pogoniulus chrysoconus*) which was detected at 25 sites, and Dark-capped Bulbul (*Pycnonotus tricolor*) which was detected at 24 sites. Two globally threatened species were detected, both within the ecological protection zone; these were Southern Ground-hornbill (*Bucorvus leadbeateri*) and Bateleur (*Terathopius ecaudatus*) (IUCN 2021).

Prior to correcting for multiple testing, distance significantly explained variation in granivore and open habitat species richness (p <0.05), both of which increased with distance (Table 2). However, when a Bonferroni correction was applied, distance only significantly explained variation in open habitat specialists (p<0.005), which increased with distance from the mine. Variation in the species richness of frugivores, insectivores, woodland specialists, Zambezian endemics and species breeding or not breeding during the survey period was not explained by distance from mine.

Table 2. Generalised linear models testing relationship between species richness and distance from the mine.

Species Richness Group	AIC	Estimate	Std. Error	z value	Pr(>|z|)
All Species	273	0.027	0.019	1.41	0.158
Frugivores	151	0.054	0.047	1.16	0.244
Granivores	123	0.169	0.077	2.19	0.029
Insectivores	248	0.004	0.024	0.19	0.852
Open Habitat	189	0.139	0.048	2.91	0.004 *
Woodland Habitat	242	0.001	0.021	0.063	0.950
Zambezian Endemic	128	0.070	0.061	1.14	0.255
Breeding	256	0.023	0.021	1.09	0.275
Non-breeding	162	0.049	0.048	1.02	0.307

Another set of generalised linear models was run with both distance from mine and site protection as explanatory variables (Table 3). When site protection was also included, none of the variation in the species richness of any of the groups was explained by distance (p > 0.005).

Table 3. Generalised linear models testing the effect of distance from the mine and site protection on species richness, only the model outputs for distance are included.

Species Richness Group	AIC	Estimate	Std. Error	z value	Pr(>|z|)
All Species	269	0.002	0.022	0.084	0.933
Frugivores	153	0.057	0.052	1.09	0.271
Granivores	111	0.011	0.096	0.116	0.907
Insectivores	244	-0.026	0.027	-0.958	0.338
Open Habitat	180	0.058	0.056	1.03	0.301
Woodland Habitat	242	-0.012	0.024	-0.487	0.626
Zambezian Endemic	130	0.073	0.068	1.08	0.277
Breeding	257	0.011	0.023	0.475	0.635
Non-breeding	151	-0.056	0.059	-0.945	0.345

2) How does habitat structure vary, and does this variation explain differences in avian species richness?

Variation in habitat structure with distance from mine and site protection

The effect of distance from the mine and area protection on habitat structure was tested (Table 4). Distance from the mine did not significantly explain variation in the number of large trees or in the percentage cover ($p>0.05$). However, the protection status of the site did significantly explain variation in the number of large trees ($p<0.05$) and the percentage of canopy cover ($p<0.001$), both of which increased inside the ecological protection zone.

Table 4. Generalised linear models testing the effect of distance from the mine and protection status on the number of large trees and the percentage of canopy cover.

Independent Variable	Dependent Variable	AIC	Estimate	Std. Error	z value	Pr(>|z|)
Distance from the mine	Number of large trees	207	-0.035	0.029	-1.29	0.223
	Percentage Canopy Cover	458	0.023	0.013	1.77	0.076
Site protection	Number of large trees	204	0.313	0.142	2.21	0.027 *
	Percentage Canopy Cover	431	0.334	0.063	5.28	<0.001 ***

Vocalising species richness in dawn chorus recordings

I tested for an effect of the number of large trees (>6 m) and percentage canopy cover on species richness. Data visualisation showed no obvious relationship between vocalising species richness and the number of large trees or the percentage of canopy cover (Figures 3a and 3b). This was confirmed by linear models showing that variation in vocalising species richness was significantly explained by neither the number of large trees (Table 5) nor the percentage of canopy cover (Table 6) for any of the groups.

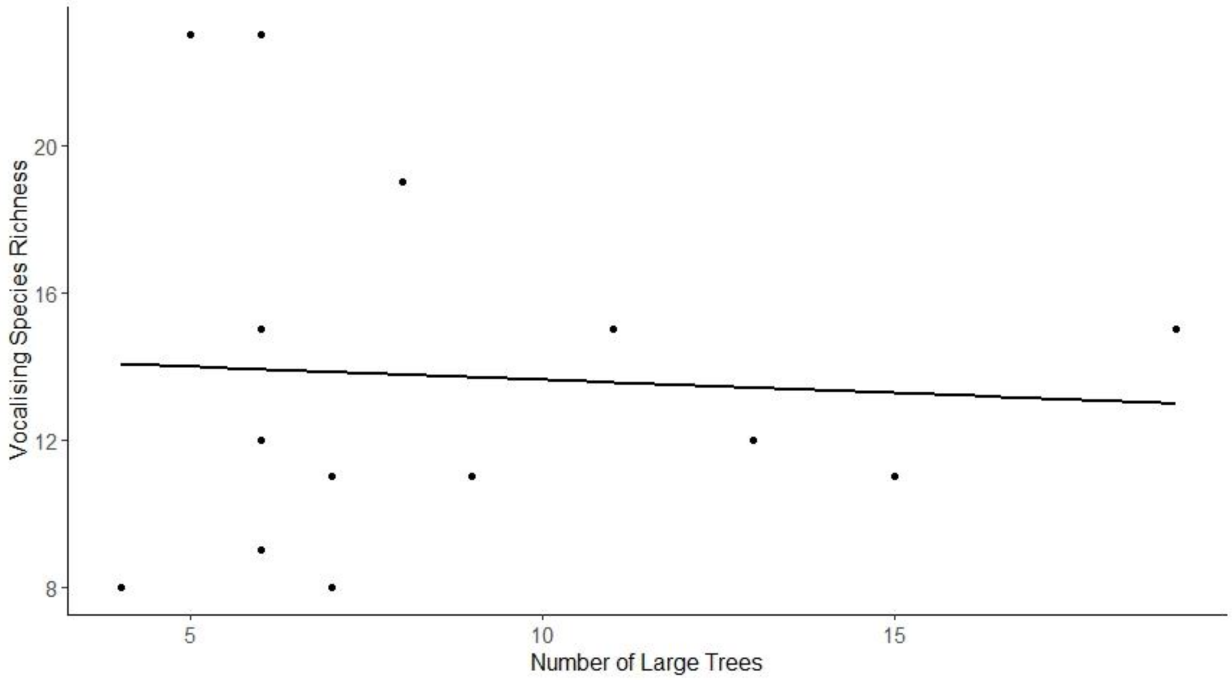


Figure 3a. Plot showing relationship between species richness and the number of large trees, with a straight regression line fitted.

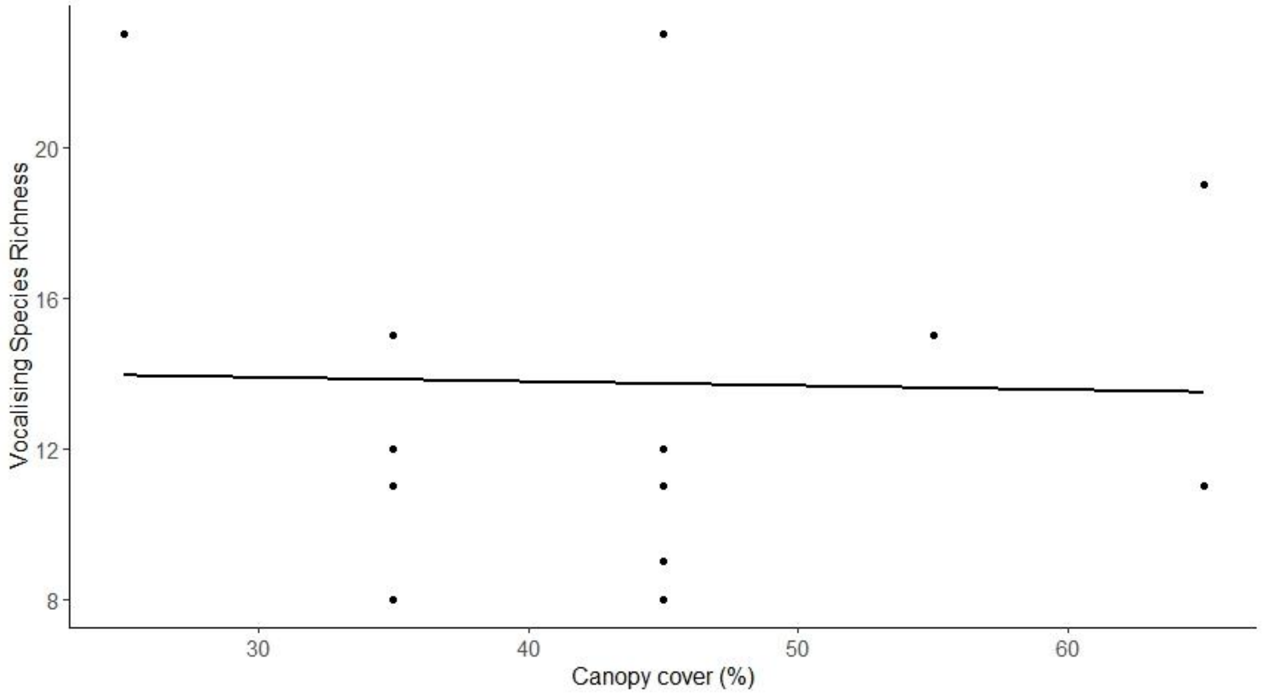


Figure 3b. Plot showing relationship between species richness and the percentage of canopy cover, with a straight regression line fitted.

Table 5. Generalised linear models testing the effect of the number of large trees on vocalising species richness.

Species Richness Group	AIC	Estimate	Std. Error	z value	Pr(>|z|)
All Species	87.9	-0.005	0.018	-0.298	0.766
Frugivores	50.3	-0.034	0.041	-0.837	0.403
Granivores	42.7	-0.080	0.073	-1.10	0.269
Insectivores	78.6	0.006	0.022	0.297	0.766
Open Habitat	44.1	-0.062	0.057	-1.09	0.274
Woodland Habitat	83.5	0.002	0.019	0.107	0.915
Breeding	86.4	-0.010	0.019	-0.551	0.582
Non-breeding	41.5	0.029	0.047	0.631	0.528

Table 6. Generalised linear models testing the effect of the percentage of canopy cover on vocalising species richness.

Species Richness Group	AIC	Estimate	Std. Error	z value	Pr(>|z|)
All Species	87.9	0.001	0.006	-0.127	0.899
Frugivores	50.8	0.006	0.014	0.413	0.680
Granivores	42.4	-0.029	0.029	-1.27	0.202
Insectivores	78.7	-0.001	0.008	-0.160	0.873
Open Habitat	43.9	-0.023	0.019	-1.23	0.219
Woodland Habitat	83.4	0.002	0.007	0.341	0.733
Zambezian Endemic	43.0	0.016	0.018	0.898	0.369
Breeding	86.7	-0.001	0.007	-0.068	0.946
Non-breeding	41.8	-0.003	0.018	-0.184	0.854

Species richness detected on point counts

On point counts, none of the variation in any of the species richness groups was significantly explained by the number of large trees both before and after Bonferroni correction (Table 7). After Bonferroni correction none of the variation in any of the species richness groups was significantly explained by the percentage of canopy cover (Table 8).

Table 7. Generalised linear models testing the relationship between species richness and the number of large trees.

Species Richness Group	AIC	Estimate	Std. Error	z value	Pr(>|z|)
All Species	273	-0.021	0.013	-1.64	0.101
Frugivores	152	-0.023	0.031	-0.757	0.449
Granivores	127	-0.072	0.052	-1.37	0.169
Insectivores	247	-0.018	0.016	-1.14	0.253
Open Habitat	195	-0.057	0.032	-1.75	0.079
Woodland Habitat	241	-0.013	0.014	-0.948	0.343
Zambezian Endemic	129	-0.005	0.039	-0.135	0.893
Breeding	255	-0.020	0.014	-1.46	0.144
Non-breeding	163	-0.024	0.032	-0.749	0.454

Table 8. Generalised linear models testing the relationship between species richness and the percentage of canopy cover.

Species Richness Group	AIC	Estimate	Std. Error	z value	Pr(>|z|)
All Species	275	-0.002	0.003	-0.666	0.505
Frugivores	152	0.005	0.006	0.860	0.390
Granivores	125	-0.019	0.010	-1.79	0.073
Insectivores	248	-0.002	0.003	-0.715	0.475
Open Habitat	193	-0.015	0.007	-2.26	0.024
Woodland Habitat	241	0.001	0.003	0.362	0.717
Zambezian Endemic	129	0.004	0.008	0.536	0.592
Breeding	257	-0.001	0.003	-0.224	0.823
Non-breeding	162	-0.008	0.007	-1.14	0.254

3) How does avian species richness vary inside and outside the ecological protection zone?

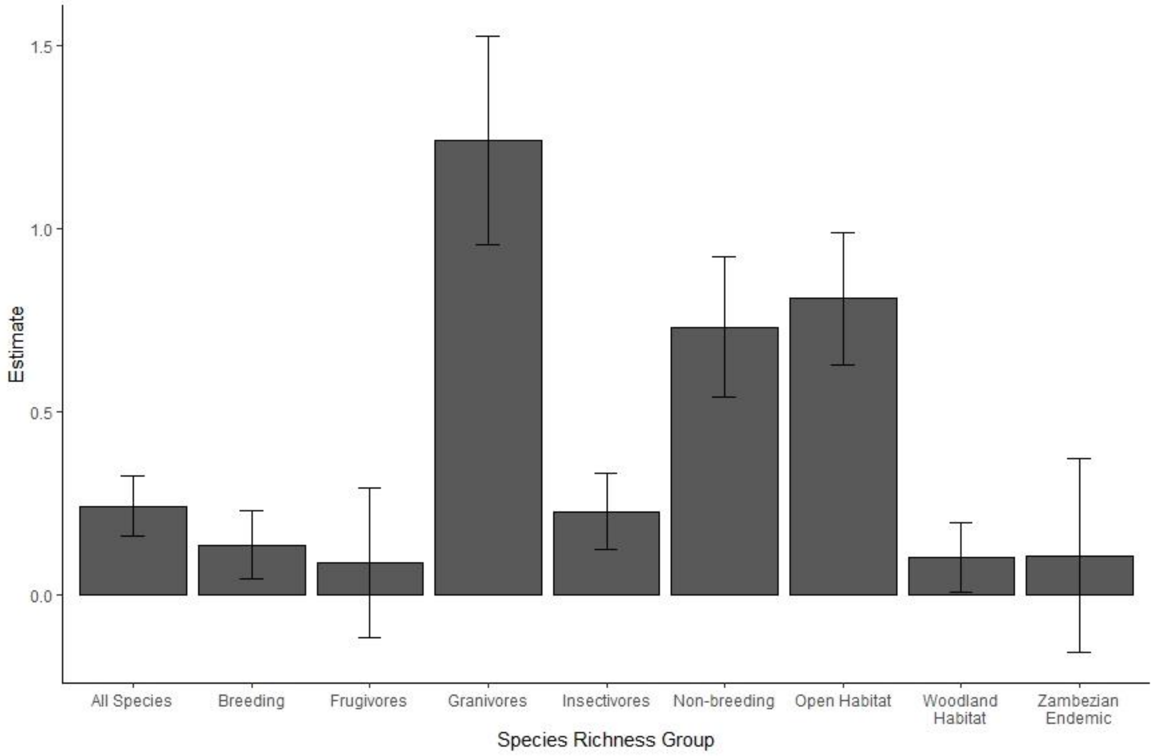
Species richness detected on point counts

Variation in the species richness of frugivores, woodland specialists, Zambezian endemics and species breeding during the survey period was not explained by whether the survey site was situated within the ecological protection zone (Table 9). After correcting for multiple testing, the species richness of insectivores was also not explained by whether the survey site was situated within the ecological protection zone. Variation in total species richness was significantly explained by whether the site was located within the ecological protection zone ($p < 0.005$), with total species richness being higher outside of the ecological protection zone. The protection of the survey site also explained significant amounts of variation in the species richness of granivores ($p<0.005$), open habitat specialists ($p<0.005$) and non-breeding species ($p<0.005$). The species richness of granivores, open habitat specialists and non-breeding species all increased outside of the ecological protection zone.

Table 9. The results of generalised linear models testing the relationship between species richness and the protection of sites.

Species Richness Group	AIC	Estimate	Std. Error	z value	Pr(>|z|)
All Species	267	-0.241	0.082	-2.92	0.003 *
Frugivores	153	-0.087	0.203	-0.429	0.668
Granivores	109	-1.24	0.285	-4.37	1.25E-05 ***
Insectivores	243	-0.227	0.103	-2.22	0.027
Open Habitat	179	-0.809	0.181	-4.48	7.33E-06 ***
Woodland Habitat	240	-0.101	0.094	-1.07	0.283
Zambezian Endemic	129	-0.107	0.265	-0.404	0.686
Breeding	255	-0.136	0.092	-1.48	0.140
Non-breeding	150	-0.731	0.192	-3.82	0.0001 **

Figure 4. Plot showing estimates and standard error from generalised linear models showing the change in species richness outside the protected area.



Tests of correlation between response variables found a strong correlation between the total species richness and the species richness of insectivores (0.919), woodland specialists (0.900), and breeding species (0.959) (Table 10). A strong correlation was also found between the species richness of insectivores and the species richness of woodland specialists (0.887) and breeding species (0.867). There was also a correlation between the species richness of open habitat specialists and non-breeding species (0.753), and between the species richness of woodland habitat specialists and breeding species (0.931).

Table 10. The correlation matrix (Pearson's r) produced from testing the correlation between response variables.

	All Species	**Frugivores**	**Granivores**	**Insectivores**	**Open Habitat**	**Woodland Habitat**	**Zambezian Endemic**	**Breeding**	**Non-breeding**
All Species	1.000	0.462	0.559	0.919	0.714	0.9	0.532	0.959	0.634
Frugivores	0.462	1.000	0.076	0.182	0.244	0.467	0.129	0.534	0.047
Granivores	0.559	0.078	1.000	0.412	0.656	0.345	0.233	0.467	0.543
Insectivores	0.919	0.182	0.412	1.000	0.568	0.888	0.566	0.877	0.594
Open Habitat	0.714	0.244	0.656	0.568	1.000	0.344	0.184	0.574	0.754
Woodland Habitat	0.9	0.467	0.345	0.887	0.344	1.000	0.599	0.931	0.386
Zambezian Endemic	0.532	0.129	0.233	0.565	0.184	0.599	1.000	0.628	0.014
Breeding	0.959	0.534	0.466	0.867	0.574	0.931	0.628	1.000	0.389
Non-breeding	0.634	0.047	0.543	0.594	0.753	0.386	0.015	0.389	1.000

Discussion

The mine's effect on avian species richness

One of the main aims of this study was to explore how avian species richness changes with distance from a large open pit copper mine, with a view to understanding how mine effects alter woodland bird communities in the miombo ecoregion. I expected that total species richness and vocalising species richness would increase as distance from the mine increased. I also expected that the species richness of breeding species and insectivores would increase as distance from the mine increased, as previous studies have found these species are more vulnerable than granivores, frugivores and non-breeding species to mine pollution effects (Francis et al. 2009; Francis 2015). However, my results did not support these expectations. Initially it appeared that open habitat specialist species richness increased with distance from the mine. However, when the effect of site protection was controlled for, none of the variation in the total species richness or in the species richness of any of the ecological guilds tested was significantly explained by distance from the mine. Vocalising species richness also did not increase significantly with distance from the mine.

The findings suggest that this mine's effects on avian species richness are not detectable at the scale used in this study. Sampling started in woodland 500 m from the edge of the mine and continued in intervals of one kilometre. It is likely that the 500 m of woodland has reduced mine effects, as even in the sites closest to the mine there were no significant changes in avian species richness. This reduction in mine pollution by the woodland is highly likely, as previous studies have found significant air and noise pollution reduction by trees over much smaller distances, (Cavanagh et al. 2009; Maleki & Hosseini 2011; van Renterghem et al. 2012). Furthermore, it has been shown that as the size of a woodland patch increases, so does its effect on pollution reduction (Santos et al. 2017). If the study was conducted closer to the edge of the mine and with smaller distance bands, a mine effect on avian species richness is more likely to have been detected but at the scale of this study it appears to be negligible. Alternatively, the mine's effects were still detectable at 500 m and remained equal in severity over the entire scale of the study area, this seems unlikely given the size of the area. Repeated studies at different opencast mines in the miombo ecoregion would help to corroborate these results.

The lack of difference in vocalising species richness between sites near and far from the mine is unsurprising, since previous studies using acoustic monitoring to test the effects of noise pollution have found that total vocalising avian species richness did not vary significantly with the distance from other mines (Duarte et al. 2015; Alvarez-Berríos et al. 2016). This is probably because in tropical or subtropical climates, such as Zambia, species richness detected in using acoustic monitoring has been shown to be a poor indicator of true species richness (Eldridge et al. 2018). While acoustic monitoring was useful when used in conjunction with point counts, it is unlikely to provide a true picture of avian species richness in a miombo woodland when used alone.

While mine pollution can affect bird species directly, it can also impact them indirectly by degrading the habitat they rely on. While pollution tolerance varies between tree species, prolonged exposure to high levels of pollution will decrease tree fitness and alter woodland structure (Gheorghe & Ion 2011; Kumar Prajapati 2012; Chaudhary & Rathore 2019). I found no significant relationship between distances from the mine and habitat structure (the percentage of canopy cover and number of large trees), suggesting that pollution effects had already been reduced by the woodland. Furthermore, to ensure that habitat structure was not influencing species richness, the relationship between species richness and the number of large trees and the percentage of canopy cover was tested. I found that none of the variation in vocalising species richness, total species richness or the species richness of any of the ecological guilds tested was explained by the number of large trees or the percentage of canopy cover.

The effect of protection on avian species richness

Another key aim of this study was to investigate how species richness varies inside and outside of the ecological protection zone that surrounds the mine, with a view to understanding the effectiveness of ecological protection zonesin conserving miombo woodland bird communities. I expected that total species richness and the species richness of all the ecological guilds tested (except for granivores, and open habitat specialists), would increase with protection.

Surprisingly, the data showed no significant relationship between protection status and the species richness of frugivores, insectivores, woodland habitat specialists, Zambezian endemic species or breeding species. The number of large trees and the percentage of canopy cover both significantly increased inside the ecological protection zone, suggesting that habitat structure was not the most important factor in determining species richness. The species richness of granivores and open habitat specialists increased significantly outside the ecological protection zone, as expected, and the total species richness and the species richness of non-breeding species also increased significantly outside of the ecological protection zone. Correlation tests showed a strong correlation ($r = 0.754$) between the species richness of open habitat specialist and non-breeding species. This suggests that my findings of a significant increase in the species richness of non-breeding species and open habitat specialists outside the ecological protection zone may not be fully independent.

Previous studies which have compared avian species richness in large unfragmented protected woodlands with woodland patches surrounded by cropland appear to support my findings. They found that woodland species were uniformly distributed in fragmented woodland patches and large protected woodlands, and that total species richness was higher in fragmented woodlands surrounded by croplands than in large unfragmented woodlands (McIntyre 1995; Pino et al. 2000; Rayner et al. 2014). These studies also found that the species richness of granivores and open habitat species is significantly higher in woodland patches outside of protected areas (Pino et al. 2000; Greve et al. 2011). The detection of granivores and open habitat species inside woodland patches is interesting, as it may suggest that woodlands still hold some value to these species.

However, while my results indicate higher species richness at sites outside the ecological protection zone, the abundance and diversity of species was not known. Previous studies have found that while protected areas contain lower avian species richness, they do contain a higher abundance and diversity (McIntyre 1995). This should be tested for in future studies.

A clear positive effect of the ecological protection zone was detected on habitat structure, since canopy cover and number of large trees were both higher within the ecological protection zone. This was likely due to the harvesting of wood for fuel and charcoal production and annual fires outside the ecological protection zone, both of which negatively impact habitat structure (Ryan & Williams 2011; Chidumayo 2013).

Conservation Implications

There is an urgent need to address biodiversity loss in Zambia and throughout the miombo woodlands ecoregion. The main threats facing miombo woodlands in Zambia and the ecoregion are mining, charcoal production, unsustainable agriculture practices, and climate change (Chomba et al. 2012; Chidumayo 2013; Kalaba et al. 2013; Jinga & Ashley 2019). Population growth throughout the ecoregion will likely exacerbate these threats and lead to increased biodiversity loss (McKee et al. 2004). The rapid rate of deforestation in Zambia is concerning, since it is estimated that an average of 250,000 ha of forests are lost yearly, and between 1972 and 2016, primary woodland decreased by 32% and woodland connectivity declined by 22% (MacDicken 2015; Phiri et al. 2019). At least 54% of plant species in the miombo woodlands ecoregion are endemic, with Zambia being the centre of this endemism (Campbell et al. 1996). The distributions of many bird species, particularly Zambezian endemics, are focused on Zambia, making its habitats and the threats they face of importance in African avian conservation(Leonard 2001). To conserve miombo woodlands and the species that rely on them, an understanding of the threats they face is essential.

One such threat is the mining industry in sub-Saharan Africa which is growing and expanding, and so too will its negative impacts on the environment (Edwards et al. 2014; Ahmed et al. 2021). It is therefore essential that environmental impact mitigation measures be tested and developed (Edwards et al. 2014). Environmental impact mitigation has largely focused on two areas, the reduction of pollutants at their source through technological advancements and improved efficiency, and the reclamation of mining areas once extraction is complete (Ranängen & Zobel 2014).

A second threat is the destruction and fragmentation of miombo woodlands, particularly by agriculture. It has been estimated that 60% of the annual deforestation in Zambia is due to cropland expansion by smallholders (Ngoma et al. 2021). It is therefore vital to understand how birds respond to fragmentation from agriculture, and to protection, in order to implement appropriate conservation measures. While my findings suggest that expansion of cropland will benefit granivores and open habitat specialists, it will almost certainly impact woodland dependent species.

My findings highlight that even woodland patches surrounded by cropland remain important to mitigate such impacts, since they still appeared to contain miombo bird species (such as Böhm's Flycatcher, Miombo Scrub Robin and Pale-billed Hornbill). There is a growing consensus on the importance of conserving matrix habitats such as forest patches in croplands, to ensure maximum biodiversity preservation, and complement biodiversity preservation in protected areas (Pino et al. 2000; Evans et al. 2006; Franklin & Lindenmayer 2009). Protected areas are also important to sensitive or threatened species; in my study, the two threatened species recorded (Southern Ground-hornbill and Bateleur) were both detected only inside the ecological protection zone.

In summary, this is the first study on the effects of active mining on avian species richness in surrounding miombo woodland, and the scale at which these effects are observed. My finding that no effect on species richness was detectable at the scale of my study is encouraging. While this is a case study of a single mine, the findings suggest that using woodlands as a barrier to mine pollution may be a viable addition to current mitigation measures, though studies of additional mines are needed to confirm generality. Reduction in pollution will not only benefit biodiversity in surrounding areas but also the human population that inhabits them. The woodland may also play a key role when post-mining reclamation occurs by acting as a source for seeds and their dispersers, particularly in mined areas adjacent to the woodland.

References

Ahmed AI, Bryant RG, Edwards DP. 2021. Where are mines located in sub-Saharan Africa and how have they expanded overtime? Land Degradation and Development **32**:112–122.

Allan RJ. 1995. Impact of mining activities on the terrestrial and aquatic environment with emphasis on mitigation and remedial measures. Pages 119–140 Heavy Metals. Springer, Berlin, Heidelberg.

Alvarez-Berríos N, Campos-Cerqueira M, Hernández-Serna A, Amanda Delgado JC, Román-Dañobeytia F, Mitchell Aide T. 2016. Impacts of small-scale gold mining on birds and anurans near the Tambopata Natural Reserve, Peru, assessed using passive acoustic monitoring. Tropical Conservation Science **9**:832–851.

Azadi M, Northey SA, Ali SH, Edraki M. 2020. Transparency on greenhouse gas emissions from mining to enable climate change mitigation. Nature Geoscience **13**:100–104.

Beckett KP, Freer-Smith PH, Taylor G. 2000. Particulate pollution capture by urban trees: effect of species and windspeed. Global Change Biology **6**:995–1003.

Bellard C, Bertelsmeier C, Leadley P, Thuiller W, Courchamp F. 2012. Impacts of climate change on the future of biodiversity. Ecology Letters **15**:365–377.

Belskii EA, Belskaya EA. 2013. Bird population in birch forests of the southern Urals affected by industrial pollution: Report 1. Reactions of species and the community. Contemporary Problems of Ecology **6**:315–322.

Blair RB. 1999. Birds and butterflies along an urban gradient: surrogate taxa for assessing biodiversity? Ecological Applications **9**:164–170.

Boldy R, Santini T, Annandale M, Erskine PD, Sonter LJ. 2021. Understanding the impacts of mining on ecosystem services through a systematic review. The Extractive Industries and Society **8**:457–466.

Brown LH, Urban EK, Newman KB. 1982. The Birds of Africa: Volume I. Bloomsbury Publishing, London.

Busia K, Akong C. 2017. The African mining vision: perspectives on mineral resource development in Africa. Journal of Sustainable Development Law and Policy **8**:145–192.

Byers B. 2001. Conserving the Miombo Ecoregion. Final Reconnaissance Summary Report, Harare.

Cailteux JLH, Kampunzu AB, Lerouge C, Kaputo AK, Milesi JP. 2005. Genesis of sediment-hosted stratiform copper–cobalt deposits, central African Copperbelt. Journal of African Earth Sciences **42**:134–158.

Campbell B, Frost P, Byron N. 1996. Miombo woodlands and their use: Overview and key issues. Pages 1–10 in B. Campbell, editor. The Miombo in Transition: Woodlands and Welfare in Africa. CIFOR, Bogor.

Cavanagh J-AE, Zawar-Reza P, Wilson JG. 2009. Spatial attenuation of ambient particulate matter air pollution within an urbanised native forest patch. Urban Forestry & Urban Greening **8**:21–30.

Central Statistical Office. 2012. Zambia: 2010 Census of Population and Housing: Population Summary Report. Central Statistical Office.

Chase MJ et al. 2016. Continent-wide survey reveals massive decline in African savannah elephants. PeerJ **4**:e2354.

Chaudhary IJ, Rathore D. 2019. Dust pollution: Its removal and effect on foliage physiology of urban trees. Sustainable Cities and Society **51**:101696. Elsevier.

Chidumayo EN. 1987. Species structure in Zambian miombo woodland. Journal of Tropical Ecology **3**:109–118.

Chidumayo EN. 2013. Forest degradation and recovery in a miombo woodland landscape in Zambia: 22 years of observations on permanent sample plots. Forest Ecology and Management **291**:154–161.

Chomba BM, Tembo O, Mutandi K, Mtongo SC, Makano A. 2012. Drivers of deforestation, identification of threatened forests and forest cobenefits other than carbon from REDD+ implementation in Zambia. A consultancy report prepared for the Forestry Department and the Food and Agriculture Organization of the United Nations under the national UN-REDD Programme. Lusaka, Zambia.

Craigie ID, Baillie JEM, Balmford A, Carbone C, Collen B, Green RE, Hutton JM. 2010. Large mammal population declines in Africa's protected areas. Biological Conservation **143**:2221–2228.

Dawson TP, Jackson ST, House JI, Prentice IC, Mace GM. 2011. Beyond predictions: biodiversity conservation in a changing climate. Science **332**:53–58.

Deikumah JP, McAlpine CA, Maron M. 2014. Mining matrix effects on West African rainforest birds. Biological Conservation **169**:334–343.

Dewees PA, Campbell BM, Katerere Y, Sitoe A, Cunningham AB, Angelsen A, Wunder S, Bank W. 2010. Managing the miombo woodlands of southern Africa: policies, incentives, and options for the rural poor. Journal of Natural Resources Policy Research **2**:57–73.

Ditmer MA, Francis CD, Barber JR, Stoner DC, Seymoure BM, Fristrup KM, Carter NH. 2021. Assessing the vulnerabilities of vertebrate species to light and noise pollution: expert surveys illuminate the impacts on specialist species. Integrative and Comparative Biology: icab091.

Dominoni DM, Quetting M, Partecke J. 2013. Long-term effects of chronic light pollution on seasonal functions of European Blackbirds (*Turdus merula*). PLoS ONE **8**: e85069

Downing TE. 2002. Avoiding New Poverty: Mining-Induced Displacement and Resettlement. London.

Duarte MHL, Sousa-Lima RS, Young RJ, Farina A, Vasconcelos M, Rodrigues M, Pieretti N. 2015. The impact of noise from open-cast mining on Atlantic forest biophony. Biological Conservation **191**:623–631.

Durán AP, Rauch J, Gaston KJ. 2013. Global spatial coincidence between protected areas and metal mining activities. Biological Conservation **160**:272–278.

Edwards DP, Sloan S, Weng L, Dirks P, Sayer J, Laurance WF. 2014. Mining and the African Environment. Conservation Letters **7**:302–311.

Eeva T, Belskii E, Gilyazov AS, Kozlov M v. 2012. Pollution impacts on bird population density and species diversity at four non-ferrous smelter sites. Biological Conservation **150**:33–41.

Eldridge A, Guyot P, Moscoso P, Johnston A, Eyre-Walker Y, Peck M. 2018. Sounding out ecoacoustic metrics: Avian species richness is predicted by acoustic indices in temperate but not tropical habitats. Ecological Indicators **95**:939–952.

Ericsson M, Löf O, Löf A. 2020. Chinese control over African and global mining—past, present and future. Mineral Economics **33**:153–181.

Evans KL, Rodrigues ASL, Chown SL, Gaston KJ. 2006. Protected areas and regional avian species richness in South Africa. Biology Letters **2**:188.

Farjana SH, Huda N, Parvez Mahmud MA, Saidur R. 2019. A review on the impact of mining and mineral processing industries through life cycle assessment. Journal of Cleaner Production **231**:1200–1217.

Feng Y, Wang J, Bai Z, Reading L. 2019. Effects of surface coal mining and land reclamation on soil properties: A review. Earth-Science Reviews **191**:12–25.

Fleming DA, Measham TG. 2014. Local job multipliers of mining. Resources Policy **41**:9–15.

Fraixedas S, Lindén A, Piha M, Cabeza M, Gregory R, Lehikoinen A. 2020. A state-of-the-art review on birds as indicators of biodiversity: Advances, challenges, and future directions. Ecological Indicators **118**:106728.

Francis CD. 2015. Vocal traits and diet explain avian sensitivities to anthropogenic noise. Global Change Biology **21**:1809–1820.

Francis CD, Ortega CP, Cruz A. 2009. Noise pollution changes avian communities and species interactions. Current Biology **19**:1415–1419.

Franklin JF, Lindenmayer DB. 2009. Importance of matrix habitats in maintaining biological diversity. Proceedings of the National Academy of Sciences **106**:349–350.

Fugiel A, Burchart-Korol D, Czaplicka-Kolarz K, Smoliński A. 2017. Environmental impact and damage categories caused by air pollution emissions from mining and quarrying sectors of European countries. Journal of Cleaner Production **143**:159–168.

Gautam S, Patra A, Prusty BK. 2012. Opencast mines: A subject to major concern for human health. International Research Journal of Geology and Mining **2**:25–31.

Gheorghe IF, Ion B. 2011. The Effects of Air Pollutants on Vegetation and the Role of Vegetation in Reducing Atmospheric Pollution. Pages 241–280 in M. K. Khallaf, editor. The Impact of Air Pollution on Health, Economy, Environment and Agricultural Sources, 1st edition. IntechOpen.

Giam X, Olden JD, Simberloff D. 2018. Impact of coal mining on stream biodiversity in the US and its regulatory implications. Nature Sustainability **1**:176–183.

Gregory RD, Gibbons DW, Donald PF. 2004. Bird census and survey techniques. Pages 17–56 Bird ecology and conservation.

Greve M, Chown SL, Rensburg BJ van, Dallimer M, Gaston KJ. 2011. The ecological effectiveness of protected areas: a case study for South African birds. Animal Conservation **14**:295–305.

Grilo C, Koroleva E, Andrášik R, Bíl M, González-Suárez M. 2020. Roadkill risk and population vulnerability in European birds and mammals. Frontiers in Ecology and the Environment **18**:323–328.

Gumbo DJ, Dumas-Johansen M, Muir G, Boerstler F, Xia Z. 2018. Sustainable management of Miombo woodlands- Food security, nutrition and wood energy. FAO, Rome.

IUCN. 2021. The IUCN Red List of Threatened Species. Version 2021-3. https://www.iucnredlist.org.

Jew EKK, Loos J, Dougill AJ, Sallu SM, Benton TG. 2015. Butterfly communities in miombo woodland: Biodiversity declines with increasing woodland utilisation. Biological Conservation **192**:436–444.

Jinga P, Ashley M v. 2019. Climate change threatens some miombo tree species of sub-Saharan Africa. Flora **257**:151421.

Jinga P, Palagi J. 2020. Dry and wet miombo woodlands of south-central Africa respond differently to climate change. Environmental Monitoring and Assessment 2020 **192**:6 192:1–16.

Kalaba FK, Quinn CH, Dougill AJ, Vinya R. 2013. Floristic composition, species diversity and carbon storage in charcoal and agriculture fallows and management implications in Miombo woodlands of Zambia. Forest Ecology and Management **304**:99–109.

Kati V, Devillers P, Dufrêne M, Legakis A, Vokou D, Lebrun P. 2004. Testing the value of six taxonomic groups as biodiversity indicators at a local scale. Conservation Biology **18**:667–675.

Kight CR, Swaddle JP. 2011. How and why environmental noise impacts animals: An integrative, mechanistic review. Ecology Letters **14**:1052–1061.

Kolala C, Dokowe A. 2021. Economic potential of industrial minerals in Zambia - A review. Resources Policy **72**:101997.

Kossoff D, Dubbin WE, Alfredsson M, Edwards SJ, Macklin MG, Hudson-Edwards KA. 2014. Mine tailings dams: characteristics, failure, environmental impacts, and remediation. Applied Geochemistry **51**:229–245.

Kumar Prajapati S. 2012. Ecological effect of airborne particulate matter on plants. Environmental Skeptics and Critics **1**:12–22.

Larsen FW, Bladt J, Balmford A, Rahbek C. 2012. Birds as biodiversity surrogates: will supplementing birds with other taxa improve effectiveness? Journal of Applied Ecology **49**:349–356.

Laurance WF, Goosem M, Laurance SGW. 2009. Impacts of roads and linear clearings on tropical forests. Trends in Ecology & Evolution **24**:659–669.

Leonard PM. 2001. Zambia. Pages 1005–1024 in L. D. C. Fishpool and M. I. Evans, editors. Important Bird Areas in Africa and Associated Islands: Priority Sites for Conservation. Birdlife International, Cambridge.

Lewandowski AS, Noss RF, Parsons DR. 2010. The effectiveness of surrogate taxa for the representation of biodiversity. Conservation Biology **24**:1367–1377.

Lindsey PA et al. 2013. The bushmeat trade in African savannas: impacts, drivers, and possible solutions. Biological Conservation **160**:80–96.

Lokhande SK, Jain MC, Dhawale SA, Gautam R, Bodhe GL. 2018. Realizing modelling and mapping tools to study the upsurge of noise pollution as a result of open-cast mining and transportation activities. Noise & Health **20**:60.

MacDicken KG. 2015. Global Forest Resources Assessment 2015: What, why and how? Forest Ecology and Management **352**:3–8.

Maleki K, Hosseini SM. 2011. Investigation of the effects of leaves, branches and canopies of trees on noise pollution reduction. Annals of Environmental Science **5**:13–21.

Malmer A. 2007. General ecological features of miombo woodlands and considerations for utilization and management. Working Papers of the Finnish Forest Research Institute **50**:34–42.

Mascia MB, Pailler S, Krithivasan R, Roshchanka V, Burns D, Mlotha MJ, Murray DR, Peng N. 2014. Protected area downgrading, downsizing, and degazettement (PADDD) in Africa, Asia, and Latin America and the Caribbean, 1900–2010. Biological Conservation **169**:355–361.

Mayani-Parás F, Botello F, Castañeda S, Sánchez-Cordero V. 2019. Impact of habitat loss and mining on the distribution of endemic species of amphibians and reptiles in Mexico. Diversity **11**.

McIntyre NE. 1995. Effects of forest patch size on avian diversity. Landscape Ecology **10**:85–99.

McKee JK, Sciulli PW, David Fooce C, Waite TA. 2004. Forecasting global biodiversity threats associated with human population growth. Biological Conservation **115**:161–164.

Mittermeier RA, Mittermeier CG, Brooks TM, Pilgrim JD, Konstant WR, Fonseca GAB da, Kormos C. 2003. Wilderness and biodiversity conservation. Proceedings of the National Academy of Sciences **100**:10309–10313.

Moura I, Maquia I, Rija AA, Ribeiro N, Ribeiro-Barros AI. 2017. biodiversity studies in key species from the african mopane and miombo woodlands. Pages 91–109 in L. Bitz, editor. Genetic Diversity. IntechOpen, London.

Mudronova J. 2018. How can Africa Maximise its Mineral Wealth? Development Stratagies and Chinese Investment in Africa's Mining Sector. Johannesburg.

Murguía DI, Bringezu S, Schaldach R. 2016. Global direct pressures on biodiversity by large-scale metal mining: Spatial distribution and implications for conservation. Journal of Environmental Management **180**:409–420.

Mwaanga P, Silondwa M, Kasali G, Banda PM. 2019. Preliminary review of mine air pollution in Zambia. Heliyon **5**:e02485.

Ngoma H, Pelletier J, Mulenga BP, Subakanya M. 2021. Climate-smart agriculture, cropland expansion and deforestation in Zambia: linkages, processes and drivers. Land Use Policy **107**:105482.

Norgate T, Haque N. 2010. Energy and greenhouse gas impacts of mining and mineral processing operations. Journal of Cleaner Production **18**:266–274.

Nowak DJ, Hirabayashi S, Doyle M, McGovern M, Pasher J. 2018. Air pollution removal by urban forests in Canada and its effect on air quality and human health. Urban Forestry & Urban Greening **29**:40–48.

Oksanen J et al. 2019. vegan: Community Ecology Package.

Owen JR, Kemp D. 2015. Mining-induced displacement and resettlement: a critical appraisal. Journal of Cleaner Production **87**:478–488.

Owens ACS, Cochard P, Durrant J, Farnworth B, Perkin EK, Seymoure B. 2020. Light pollution is a driver of insect declines. Biological Conservation **241**:108259.

Palacios-Torres Y, Caballero-Gallardo K, Olivero-Verbel J. 2018. Mercury pollution by gold mining in a global biodiversity hotspot, the Choco biogeographic region, Colombia. Chemosphere **193**:421–430.

Patricelli GL, Blickley JL. 2006. Avian communication in urban noise: causes and consequences of vocal adjustment. The Auk **123**:639–649.

Pearman PB, Weber D. 2007. Common species determine richness patterns in biodiversity indicator taxa. Biological Conservation **138**:109–119.

Perillo A, Mazzoni LG, Figueiredo Passos L, Goulart VDLR, Duca C, Young RJ. 2017. Anthropogenic noise reduces bird species richness and diversity in urban parks. Ibis **159**:638–646.

Peša I. 2021. Mining, Waste and Environmental Thought on the Central African Copperbelt, 1950–2000. Environment and History **27**. White Horse Press.

Phiri D, Morgenroth J, Xu C. 2019. Four decades of land cover and forest connectivity study in Zambia—An object-based image analysis approach. International Journal of Applied Earth Observation and Geoinformation **79**:97–109.

Pino J, Rodà F, Ribas J, Pons X. 2000. Landscape structure and bird species richness: implications for conservation in rural areas between natural parks. Landscape and Urban Planning **49**:35–48.

Pourret O, Lange B, Bonhoure J, Colinet G, Decrée S, Mahy G, Séleck M, Shutcha M, Faucon MP. 2016. Assessment of soil metal distribution and environmental impact of mining in Katanga (Democratic Republic of Congo). Applied Geochemistry **64**:43–55.

R Core Team. 2020. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Ranängen H, Zobel T. 2014. Revisiting the ‘how’ of corporate social responsibility in extractive industries and forestry. Journal of Cleaner Production **84**:299–312.

Rayner L, Lindenmayer DB, Wood JT, Gibbons P, Manning AD. 2014. Are protected areas maintaining bird diversity? Ecography 37:43–53.

Razo I, Carrizales L, Castro J, Díaz-Barriga F, Monroy M. 2004. Arsenic and heavy metal pollution of soil, water, and sediments in a semi-arid climate mining area in Mexico. Water, Air, and Soil Pollution **152**:129–152.

Robbins CS. 1981. Effect of time of day on bird activity. Studies in Avian Biology **6**:275–286.

Rodrigues CU, Mususa P, Büscher K, Cuvelier J. 2021. Boomtown urbanization and rural-urban transformation in mining and conflict regions in Angola, the DRC and Zambia. Sustainability **13**:2285.

Roy BA, Zorrilla M, Endara L, Thomas DC, Vandegrift R, Rubenstein JM, Policha T, Ríos-Touma B, Read M. 2018. New mining concessions could severely decrease biodiversity and ecosystem services in Ecuador. Tropical Conservation Science **11**:1–20.

Ryan CM, Williams M. 2011. How does fire intensity and frequency affect miombo woodland tree populations and biomass? Ecological Applications **21**:48–60. John Wiley & Sons, Ltd.

Saha DC, Padhy PK. 2011. Effect of air and noise pollution on species diversity and population density of forest birds at Lalpahari, West Bengal, India. Science of The Total Environment **409**:5328–5336.

Salles DM, do Carmo FF, Jacobi CM. 2019. Habitat Loss Challenges the Conservation of Endemic Plants in Mining-Targeted Brazilian Mountains. Environmental Conservation **46**:140–146.

Sanderfoot O v, Holloway T. 2017. Air pollution impacts on avian species via inhalation exposure and associated outcomes. Environmental Research Letters **12**:083002.

Santos A, Pinho P, Munzi S, Botelho MJ, Palma-Oliveira JM, Branquinho C. 2017. The role of forest in mitigating the impact of atmospheric dust pollution in a mixed landscape. Environmental Science and Pollution Research **24**:12038–12048.

Schoeman MC. 2016. Light pollution at stadiums favours urban exploiter bats. Animal Conservation **19**:120–130.

Sheoran V, Sheoran AS, Poonia P. 2010. Soil reclamation of abandoned mine land by revegetation: a review. International Journal of Soil, Sediment and Water **3**:13.

Siqueira-Gay J, Sonter LJ, Sánchez LE. 2020. Exploring potential impacts of mining on forest loss and fragmentation within a biodiverse region of Brazil's north-eastern Amazon. Resources Policy **67**:101662.

Solomon D, Willems F, McDougall R. 2019. The Birds of Zambia. Android app: mydigitalearth.

Sonter LJ, Ali SH, Watson JEM. 2018. Mining and biodiversity: key issues and research needs in conservation science. Proceedings of the Royal Society **285**.

Syampungani S, Chirwa PW, Akinnifesi FK, Sileshi G, Ajayi OC. 2009. The miombo woodlands at the cross roads: Potential threats, sustainable livelihoods, policy gaps and challenges. Natural Resources Forum **33**:150–159.

Taylor CD, Schulz KJ, Doebrich JL, Orris GJ, Denning PD, Kirschbaum MJ. 2009. Geology and nonfuel mineral deposits of Africa and the Middle East. California.

Timberlake J, Chidumayo E. 2001. Miombo Ecorgeion Vision Report. Bulawayo.

UNEP-WCMC. 2016. The State of Biodiversity in Africa: A mid-term review of progress towards the Aichi Biodiversity Targets. Cambridge.

Urban EK, Fry CH, Keith S. 1997. The Birds of Africa: Volume V. Bloomsbury Publishing, London.

USGS. 2011. Minerals Yearbook-2011. Washington D.C.

van Renterghem T, Botteldooren D, Verheyen K. 2012. Road traffic noise shielding by vegetation belts of limited depth. Journal of Sound and Vibration **331**:2404–2425.

van Vliet N, Nebesse C, Gambalemoke S, Akaibe D, Nasi R. 2012. The bushmeat market in Kisangani, Democratic Republic of Congo: implications for conservation and food security. Oryx **46**:196–203.

von der Goltz J, Barnwal P. 2019. Mines: The local wealth and health effects of mineral mining in developing countries. Journal of Development Economics **139**:1–16.

Waldman KB, Vergopolan N, Attari SZ, Sheffield J, Estes LD, Caylor KK, Evans TP. 2019. Cognitive biases about climate variability in smallholder farming systems in Zambia. Weather, Climate, and Society **11**:369–383.

White F. 1983. The vegetation of Africa. Natural Resources Research, UNESCO, Paris.

Wickham H. 2016. ggplot2: Elegant Graphics for Data Analysis. Springer-Verlag, New York.

Appendices

Table A. Ecological attributes for all bird species detected in the study. The migration types are resident (R), intra-African migrant (IA) and Afro-palearctic migrant (P). The primary feeding guilds comprise of nectarivores(N), granivores(G), insectivores(I), carnivores (C), frugivores(F), piscivores(P) and omnivores(O) (Brown et al. 1982).

Species	Zambezian Miombo endemic	Breeding during study	Migratory	Primary Feeding Guild	Habitat Preference
Great Egret (*Ardea alba*)	0	1	R	P	wetland
Hamerkop (*Scopus umbrette*)	0	1	R	C	open habitats
African Hawk Eagle (*Aquila spilogaster*)	0	1	R	C	woodland
African Marsh Harrier (*Circus ranivorus*)	0	1	R	C	wetland
Bateleur (*Terathopius ecaudatus*)	0	0	R	C	open habitats
Common Buzzard (*Buteo buteo*)	0	0	P	C	open habitats
Gabar Goshawk (*Micronisus gabar*)	0	1	R	C	woodland
Lizard Buzzard (*Kaupifalco monogrammicus*)	0	1	R	C	woodland
Western Banded Snake Eagle (*Circaetus cinerascens*)	0	1	R	C	woodland
Yellow-billed Kite (*Milvus aegyptius*)	0	1	IA	C	open habitats
African Fish Eagle (*Haliaeetus vocifer*)	0	1	R	P	open habitats
Red-necked Spurfowl (*Pternistis afer*)	0	1	R	O	open habitats
Coqui Francolin (*Peliperdix coqui*)	0	1	R	G	woodland
Helmeted Guineafowl (*Numida meleagris*)	0	1	R	O	open habitats
African Wattled Lapwing (*Vanellus senegallus*)	0	1	IA	I	wetland
African Green Pigeon (*Treron calvus*)	0	1	R	F	woodland
Emerald Spotted Wood Dove (*Turtur chalcospilos*)	0	1	R	G	woodland
Red-eyed Dove (*Streptopelia semitorquata*)	0	1	R	G	open habitats
Ring-necked Dove (*Streptopelia capicola)*	0	1	R	G	open habitats
Meyer’s Parrot (*Poicephalus meyeri*)	0	1	R	F	woodland

Species	Zambezian Miombo endemic	Breeding during study	Migratory	Primary Feeding Guild	Habitat Preference
Ross's Turaco (*Turaco rossae*)	0	1	R	F	woodland
Schalow's Turaco (*Tauraco schalowi*)	0	1	R	F	woodland
African Cuckoo (*Cuculus gularis*)	0	1	IA	I	open habitats
Black Cuckoo (*Cuculus clamosus*)	0	1	IA	I	woodland
Red-chested Cuckoo (*Cuculus solitarius*)	0	1	IA	I	woodland
Diederik Cuckoo (*Chrysococcyx caprius*)	0	1	IA	I	open habitats
Jacobin Cuckoo (*Clamator jacobinus*)	0	1	IA	I	woodland
Thick-billed Cuckoo (*Pachoycoccyx audeberti*)	0	1	R	I	woodland
Fiery-necked Nightjar (*Caprimulgus pectoralis*)	0	1	R	I	woodland
Pennant Winged Nightjar (*Caprimulgus vexillarius*)	0	1	IA	I	woodland
Speckled Mousebird (*Colius striatus*)	0	1	R	F	woodland
Pied Kingfisher (*Ceryle rudis*)	0	1	R	P	open habitats
Grey-headed Kingfisher (*Halcyon leucocephala*)	0	1	R	I	woodland
Woodland Kingfisher (*Halcyon senegalensis*)	0	0	R	I	woodland
Pygmy Kingfisher (*Ispidina picta*)	0	1	R	I	woodland
European Bee-eater (*Merops apiaster*)	0	0	P	I	open habitats
Little Bee-Eater (*Merops pusillus*)	0	1	R	I	woodland
Swallow-Tailed Bee-eater (*Merops hirundineus*)	0	1	IA	I	woodland
Broad-billed Roller (*Eurystomus glaucurus*)	0	1	IA	I	woodland
Racket-tailed Roller (*Coracias spatulatus*)	1	1	R	I	woodland
Lilac-breasted Roller (*Coracias caudatus*)	0	1	R	I	open habitats
Common Scimitarbill (*Rhinopomastus cyanomelas*)	0	1	R	I	woodland
Green Wood Hoopoe (*Phoeniculus purpureus*)	0	1	R	I	woodland
African Hoopoe (*Upupa africana*)	0	1	R	I	woodland

Species	Zambezian Miombo endemic	Breeding during study	Migratory	Primary Feeding Guild	Habitat Preference
Crowned Hornbill (*Lophoceros alboterminatus*)	0	1	R	C	woodland
African Grey Hornbill (*Lophoceros nasutus*)	0	1	R	I	woodland
Pale-billed Hornbill (*Lophoceros pallidirostris*)	1	1	R	I	woodland
Trumpeter Hornbill (*Bycanistes buccinator*)	0	1	R	F	woodland
Southern Ground Hornbill (*Bucorvus leadbeateri*)	0	1	R	O	woodland
Yellow-fronted Tinkerbird (*Pogoniulus chrysoconus*)	0	1	R	I	woodland
Black-collared Barbet (*Lybius torquatus*)	0	1	R	F	woodland
Miombo Pied Barbet (*Tricholaema frontata*)	1	1	R	F	woodland
Bearded Woodpecker (*Chloropicus namaquus*)	0	1	R	I	woodland
Golden-tailed Woodpecker (*Campethera abingoni*)	0	1	R	I	woodland
Green-backed Woodpecker (*Campethera cailliautii*)	0	1	R	I	woodland
Cardinal Woodpecker (*Dendropicos fuscescens*)	0	1	R	I	woodland
Olive Woodpecker (*Dendropicos griseocephalus*)	0	1	R	I	woodland
Flappet Lark (*Mirafra rufocinnamomea*)	0	1	R	I	open habitats
European Barn Swallow (*Hirundo rustica*)	0	0	P	I	open habitats
Mosque Swallow (*Cecropis senegalensis*)	0	1	IA	I	woodland
Tree Pipit (*Anthus trivialis*)	0	0	P	I	woodland
Wood Pipit (*Anthus nyassae*)	0	1	R	I	woodland
Black Cuckooshrike (*Campephaga flava*)	0	1	R	I	woodland
White-breasted Cuckooshrike (*Ceblepyris pectoralis*)	0	1	R	I	woodland
Dark-capped Bulbul (*Pycnonotus tricolor*)	0	1	R	F	woodland
Yellow-bellied Greenbul (*Chlorocichla flaviventris*)	0	1	R	F	woodland
Arrow-marked Babbler (*Turdoides jardineii*)	0	1	R	I	woodland
Groundscraper Thrush (*Psophocichla litsitsirupa*)	0	1	R	I	woodland

Species	**Zambezian Miombo endemic**	**Breeding during study**	**Migratory**	**Primary Feeding Guild**	**Habitat Preference**
Kurrichane Thrush (*Turdus libonyana*)	1	1	R	I	woodland
Miombo Scrub Robin (*Cercotrichas barbata*)	1	1	R	I	woodland
White-Browed Scrub Robin (*Cercotrichas leucophrys*)	0	1	R	I	woodland
White-browed Robin-chat (*Cossypha heuglini*)	0	0	R	I	open habitats
Collared Flycatcher (*Ficedula albicollis*)	0	0	P	I	woodland
Ashy Flycatcher (*Muscicapa caerulescens*)	0	1	R	I	woodland
Böhm’s Flycatcher (*Muscicapa boehmi*)	1	1	R	I	woodland
Spotted Flycatcher (*Muscicapa striata*)	0	0	P	I	woodland
Grey Tit-Flycatcher (*Myioparus plumbeus*)	0	1	R	I	woodland
Pale Flycatcher (*Melaenornis pallidus*)	0	1	R	I	woodland
Southern Black Flycatcher (*Melaenornis pammelaina*)	0	1	R	I	woodland
Arnott's Chat (*Myrmecocichla arnotti*)	1	1	R	I	woodland
Miombo Rock Thrush (*Monticola angolensis*)	1	1	R	I	woodland
African Paradise Flycatcher (*Terpsiphone viridis*)	0	1	R	I	woodland
Willow Warbler (*Phylloscopus trochilus*)	0	0	P	I	woodland
Grey-backed Camaroptera (*Camaroptera brevicaudata*)	0	1	R	I	woodland
Miombo Wren-Warbler (*Calamonastes undosus*)	1	1	R	I	woodland
Red-faced Cisticola (*Cisticola erythrops*)	0	0	R	I	open habitats
Neddicky (*Cisticola fulvicapilla*)	0	1	R	I	woodland
Trilling Cisticola (*Cisticola woosnami*)	0	0	R	I	woodland
Black-necked Eremomela (*Eremomela atricollis*)	1	1	R	I	woodland
Yellow-bellied Eremomela (*Eremomela icteropygialis*)	0	1	R	I	woodland
Green-capped Eremomela (*Eremomela scotops*)	0	1	R	I	woodland

Species	**Zambezian Miombo endemic**	**Breeding during study**	**Migratory**	**Primary Feeding Guild**	**Habitat Preference**
Tawny-flanked Prinia (*Prinia subflava*)	0	1	R	I	woodland
Brown-headed Apalis (*Apalis alticola*)	0	1	R	I	woodland
Grey Tit-Flycatcher (*Myioparus plumbeus*)	0	1	R	I	woodland
Southern Hyliota (*Hyliota australis*)	0	1	R	I	woodland
Yellow Bellied Hyliota (*Hyliota flavigaster*)	0	1	R	I	woodland
Chinspot Batis (*Batis molitor*)	0	1	R	I	woodland
Miombo Tit (*Melaniparus griseiventris*)	1	1	R	I	woodland
Grey Penduline Tit (*Anthoscopus caroli*)	0	1	R	I	woodland
African Spotted Creeper (*Salpornis salvadori*)	0	1	R	I	woodland
Amethyst Sunbird (*Chalcomitra amethystine*)	0	1	R	N	woodland
Scarlet-chested Sunbird (*Chalcomitra senegalensis*)	0	1	R	N	open habitats
Purple-banded Sunbird (*Cinnyris bifasciatus*)	0	1	R	N	woodland
Collared Sunbird (*Hedydipna collaris*)	0	1	R	N	woodland
African Yellow White-Eye (*Zosterops senegalensis*)	0	1	R	I	woodland
Lesser Grey Shrike (*Lanius minor*)	0	0	P	I	open habitats
Northern Fiscal Shrike (*Lanius humeralis*)	0	1	R	C	open habitats
Red-backed Shrike (*Lanius collurio*)	0	0	P	I	woodland
Black-crowned Tcharga (*Tchagra senegalus*)	0	1	R	I	open habitats
Orange-breasted Bushshrike (*Chlorophoneus sulfureopectus*)	0	1	R	I	woodland
Black-backed Puffback (*Dryoscopus cubla*)	0	1	R	I	woodland
Tropical Boubou (*Laniarius major*)	0	1	R	C	woodland
Brubru (*Nilaus afer*)	0	1	R	I	woodland
Retz's Helmetshrike (*Prionops retzii*)	0	1	R	I	woodland
White-crested Helmetshrike (*Prionops plumatus*)	0	1	R	I	woodland

Species	**Zambezian Miombo endemic**	**Breeding during study**	**Migratory**	**Primary Feeding Guild**	**Habitat Preference**
African Golden Oriole (*Oriolus auratus*)	0	1	R	I	woodland
Black-headed Oriole (*Oriolus larvatus*)	0	0	R	F	woodland
Eurasian Golden Oriole (*Oriolus oriolus*)	0	0	P	F	woodland
Fork-tailed Drongo (*Dicrurus adsimilis*)	0	1	R	I	woodland
Square-tailed Drongo (*Dicrurus ludwigii*)	0	1	R	I	woodland
Pied Crow (*Corvus albus*)	0	1	R	O	open habitats
Violet-backed Starling (*Cinnyricinclus leucogaster*)	0	1	IA	F	woodland
Miombo Blue-Eared Starling (*Lamprotornis Elisabeth*)	1	1	R	F	woodland
Greater Blue-eared Starling (*Lamprotornis chalybaeus*)	0	1	R	F	woodland
Yellow-Throated Petronia (*Gymnoris superciliaris*)	0	1	R	G	woodland
House Sparrow (*Passer domesticus*)	0	1	R	G	open habitats
Red-headed Weaver (*Anaplectes rubriceps*)	0	1	R	I	woodland
African Firefinch (*Lagonosticta rubricata*)	0	0	R	G	open habitats
Brown Firefinch (*Lagonosticta nitidula*)	1	1	R	G	wetland
Bronze Mannikin (*Spermestes cucullate*)	0	1	R	G	open habitats
Fawn-breasted Waxbill (*Estrilda paludicola*)	0	0	R	G	open habitats
Blue Waxbill (*Uraeginthus angolensis*)	0	0	R	G	open habitats
Orange-winged Pytilia (*Pytilia afra*)	0	0	R	G	woodland
Red-throated Twinspot (*Hypargos niveoguttatus*)	0	0	R	G	woodland
Yellow-fronted Canary (*Crithagra mozambica*)	0	1	R	G	open habitats
Black-eared Seedeater (*Crithagra mennelli*)	1	0	R	G	woodland
Cabanis's Bunting (*Emberiza cabanisi*)	0	1	R	I	open habitats
Golden-breasted Bunting (*Emberiza flaviventris*)	0	1	R	G	woodland