MASTERS BY RESEARCH

If you feed them, they will come: How the availability of food and cover within a zoo enclosure increases the likelihood of free-living birds within.

Brass, Jake

Award date:
2021

Awarding institution:
Bangor University

PRIFYSGOL BANGOR UNIVERSITY

If you feed them, they will come:

How the availability of food and cover within a zoo enclosure increases the likelihood of free-living birds within.

Jake Matthew Brass

Supervisor: Dr.Katherine Jones

Bangor University

Keyword: native birds, conservation, zoos, point count, subsidised feed

Literature Review

Pressures upon bird communities in the UK.

The UK is home to roughly 244 breeding species of birds. In 1996, an effort was made to identify birds of conservation concern (BoCC). During this assessment, Gibbons et al., (1996b) established the ‘traffic light system’ separating species into green (least critical), amber (unfavourable status) and red (high conservation priority) (Gibbons, Avery et al. 1996). The 1996 assessment placed 36 species on the red list, raising awareness to the plight of native birds and inspiring action. Since the first assessment, the red list has continued to grow, from 36 to 40 in 2002, 40 to 52 in 2009 and from 52 to 67 in 2017, accounting for 27.5% of the total number of breeding species in the UK(Eaton, Aebischer et al. 2015). With an ever increasing numbers of species being classified as a high conservation priority, it is clear that the UK’s breeding birds are in need of conservation action. British breeding birds are widely categorised into four groups based upon habitat; Farmland, Woodland, Water and Wetland, and Sea birds. The latest report on wild birds in the UK suggests that each group has declined since surveying began (Wild Bird Populations in the, U K, 1970 to 2017).

The decline of woodland birds has been widely attributed to anthropogenic influence, with declines present across most of Europe ranging from 13-18% of the population (Gregory, Vorisek et al. 2007, Amar, Hewson et al. 2006). A widely accepted cause for declines is a change in woodland composition**,** with both a reduction in canopy cover and a reduction in forest floor vegetation being apparent (Hewson, Noble 2009). A reduction in forest floor vegetation caused strong declines of woodland species associated with dense understorey vegetation such as the bullfinch and dunnock (Fuller, R. J., Smith et al. 2007). A complex understory vegetation has been linked to increased invertebrate abundance and so declines may be due to reduced food availability. High levels of grazing on forest floor vegetation has further been shown to reduce invertebrate availability in woodlands. This is supported by the absence of grazers in a woodland being shown to increase species richness (Gill, Fuller 2007). Ultimately however, a multitude of pressures acting simultaneously upon woodland habitats are proposed to be responsible for population declines (Ford, Barrett et al. 2001). Indeed, environmental changes to breeding grounds due to climate change, a reduction in invertebrate abundance and increased land use on woodland edges have all been linked to the decline of woodland birds (Fuller, Robert J., Noble et al. 2005).

Similar to woodland birds, declines in farmland birds have been seen across Europe (Donald, Green et al. 2001). Here declines have been linked to changing agricultural practices, with losses as high as 80% in less than 20 years (Newton 2004a). Changes in farmland practices have generated a reduction in overall food abundance over winter, and so farmland birds that over winter in the UK have seen increased mortality rates (Anderson, Bradbury et al. 2001, Siriwardena, Baillie et al. 1998, Vickery, Bradbury et al. 2004). Alongside a reduction in food availability, the removal of hedgerows has been shown to be a significant factor in the decline of farmland bird (Batáry, Matthiesen et al. 2010). Indeed, hedgerows are a key habitat for farmland birds, providing protective cover while supporting invertebrate populations vital for fledgling success rates (Arnold 1983, Hinsley, Bellamy 2000). Moreover, a shift towards single crop, monocultural farming has been linked to the declines to many farmland birds including finches, buntings and sparrows (Robinson 2001, Newton 2004b). Thus, changes in farmland practices implemented to improve farm efficiency and productivity have imparted pressures upon farmland birds and caused significant declines.

While the decline of seabirds and water and wetland birds is not as severe compared to that of farmland and woodland birds in the UK, the seabird index is 24% lower than in 1986 while water and wetland birds populations are the lowest they've been in a decade (Wild Bird Populations in the, U K, 1970 to 2017). Factors negatively affecting water and wetland birds are predominantly land management changes, such as the conversion of floodplain marshes to arable land and drainage of historical wetland sites. Further, anthropogenic disturbance, such as noise, may negatively affect overall fitness of wader species (McBlain, Jones et al. 2020). The decline of seabirds has been linked to simultaneous bottom-up and top-down pressures reducing breeding success (Church, Furness et al. 2019). Indeed these pressures, such as reduced food availability, have driven declines as great as 80%

in iconic seabird species such as the Arctic skua (Perkins, Ratcliffe et al. 2018). The Atlantic Puffin has also suffered major declines, with studies showing that breeding success and overall quantities of food brought ashore have decreased significantly over a 27 year study period (Miles, Mavor et al. 2015).

A further pressure causing declines of avian communities globally is urbanisation, being the conversion of natural land to artificial so as to meet the demands of an expanding human population (Marzluff 2001). Compared to a more natural setting, urban environments have been shown to have increased anthropogenic noise, a greater predator abundance and higher levels of pollution (Francis, C. D., Ortega et al. 2009, Bonnington, Gaston et al. 2015, Dauwe, Bervoets et al. 2003, Meillère, Brischoux et al. 2015). Domestic cats thrive, often uncontested, within urban environments. It has been estimated that there are 160 individuals/km^2, and that this incredible presence exerts a negative effect on both small mammals (Baker, Bentley et al. 2005, Baker, Ansell et al. 2003) and garden bird populations (Churcher, Lawton 1987, Woods, McDonald et al. 2003) . Further to domestic cats, urban avian assemblages are negatively influenced by other predators thriving within urban ecosystems, including both corvids and grey squirrels. When accumulated these pressures lead to significantly higher local extinction rates of both flora and fauna (Peach, Mallord et al. 2018).

Not all bird species struggle within urban environments however, many predators, such as gulls, can reach greater population numbers in urban areas due to increased supplementary food (Brousseau, Lefebvre et al. 1996). Moreover, the Peregrine Falcon (*Falco pergrinus*) has a higher breeding performance and an overall positive response to urbanisation due to an increased prey abundance (Kettel, Gentle et al. 2018). The Feral Pigeon (*Columba livia*) is one such species that the Peregrine predates upon. Feral Pigeons are well known for thriving within urban environments, with population sizes increasing with increased urban cover (Tratalos, Fuller et al. 2007). Thus, it is necessary to consider the specific needs of a species and whether these can be met within urban areas.

Key influences of biodiversity within urban environments.

The urban green spaces, be it urban parks or gardens, throughout urban areas are vital for urban biodiversity (Aronson, Lepczyk et al. 2017a). The management of these green spaces however is complex and throughout urban areas the management styles differ in accordance with differing social and economic factors (Aronson, Lepczyk et al. 2017b). As such, the urban green spaces within a city are diverse in heterogenous habitat (Angold, Sadler et al. 2006). Such diversity creates a multitude of niches which have the potential to support a diverse urban species assemblage (Kühn, Brandl et al. 2004). Indeed, the vegetation used within urban areas directly influences the diversity and abundance of urban living species (Salisbury, Al-Beidh et al. 2017). Given the unusual levels of fragmentation within an urban environment, connectivity is a key driver of biodiversity within (Braaker, Moretti et al. 2014a). High levels of connectivity allow for patches of otherwise isolated habitat to be connected, allowing for the free movement between suitable habitat patches (Braaker, Moretti et al. 2014b). The planting of hedgerows and/or the construction of wildlife crossings has been shown to compensate biodiversity losses within urban areas (Tarabon, Bergès et al. 2019). By improving and increasing vegetation within urban areas, protective cover is also increased, aiding in overall species fitness (Lazarus, Symonds 1992). Moreover, connectivity provided by vegetation increases the volume and complexity of understory vegetation which has been shown to be a key driver of biodiversity (Threlfall, Williams et al. 2016).

Another key factor influencing bird assemblages is the availability of food (Crooks, Burdett et al. 2011). Within urban environments, bird assemblages have access to novel food sources created by anthropogenic activities, be it unintentional via foraging through waste or deliberate via the provision of bird feeders (Lowry, Lill et al. 2013). Indeed, bird feeding is common across many parts of the world and in the UK alone it is estimated that 60% of households with feed their garden visitors (Cannon 2000, Dunn, Tessaglia-Hymes 2001, DEFRA 2002). The act of providing food for birds within our gardens creates a connection to nature, a connection which is often scarce as urbanisation becomes ever more prominent (Cox, Gaston 2016). In urban environments, where natural food sources are often less abundant, the provision of additional protein rich food has been shown to influence bird abundance, condition and reproductive output (Fuller, Richard A., Warren et al. 2008). However the likelihood of coming into contact with avian diseases does increase when accessibility to a single food site is high. Indeed, many studies have shown feeders to be important in the spread of avian diseases(Lawson, Robinson et al. 2018). Bird feeding stations generate unique scenarios in which interspecific mixing is common and occurs over a prolonged period of time (Kirkwood 1998). Thus, when a feeder is contaminated due to poor hygiene, the risk of disease spread is high (Hawley, Jennelle et al. 2007, Strandin, Babayan et al. 2018). Further, high levels of intraspecific competition and overcrowding upon a single feeder may lead to both immunosuppression and stress, with certain species being more dominant and so benefiting more from feeding stations (Hawley, Jennelle et al. 2007).

It is clear that, if managed in a way conducive to supporting nature, urban areas have huge potential to support biodiversity. For nature to thrive within these environments however, a shift in management practices towards nature friendly styles is needed. Therefore it is necessary to consider new avenues in which to approach urban bird assemblage conservation. Thus, a new audience has to be reached that is not currently engaged in the conservation of native species. Here it is proposed that zoos could support native wildlife immensely through many relatively unexplored conservation actions.

The role of zoos in Conservation

With 700million people visiting zoos globally per year, zoos are unparalleled in the audience they attract (Gusset, Dick 2011). Recently, many zoos have shifted their focus to be more conservation orientated, be it through planned in-situ projects or educational programmes within zoo grounds (Patrick, Caplow 2018). Furthermore, there has been a shift in how zoos approach conservation. Many of the famous zoo conservation success stories involve the reintroduction of a previously extinct species to an area; see California condor (Ralls, Ballou 2004) , Mauritius kestrel (Jones, Heck et al. 1995) & Guam rail (Haig, Ballou et al. 1990). Unfortunately not all reintroductions are a success, and while the evolution of reintroduction science has allowed for successes to be more frequent, reintroduction themselves are expensive and often limited to a few zoos with breeding programmes of the select species (Seddon, Armstrong et al. 2007). More-over, many reintroductions have a high risk of mortality during the early phases of reintroduction due to an inability to adjust to wild living (Zimmermann, Hatchwell et al. 2007). As a result, zoos are focusing conservation efforts upon raising awareness through visitor education while engaging in limited in-situ conservation projects. An area of conservation relatively unexplored within zoos is the promotion of free-living species across zoo grounds.

Zoos and Native Species

There is little peer reviewed literature identifying how free-living species utilise zoo grounds. Zoos themselves do collect data, however the type of surveyed performed and the type of data collected often differs between zoos and prevents cross zoo analysis. Further, due to the nature of the data collected, it is difficult to infer more than presence/absence. Of the research conducted upon free-living species within zoo grounds, Baur et al, 2011 quantified the biodiversity of Basel Zoo but made no attempt to relate biodiversity to zoo design (Baur 2011). A multi zoo study conducted by Kover et al., 2019 identified factors influencing presence of free-living corvids across zoos, yet no attempt was made to relate presence to zoo design (Kövér, Lengyel et al. 2019). Given zoos are long-established areas of heterogeneous land, managed in a way to produce naturalistic environments, vegetation on site is often mature, complex and assumed good habitat for free-living species. Data collected by BIAZA, compiling bioblitz' performed within zoo grounds, shows over 2000 free-living species having been observed (BIAZA Personal Communication 2017). Indeed, the vast vegetation within zoo grounds makes them comparable to urban parks and gardens, being a mix of natural and non-natural components (Bjerke, Østdahl et al. 2006). Given urban parks and gardens have been shown to support large amounts of urban biodiversity, with studies showing how their design influences urban biodiversity (Cornelis, Hermy 2004), it is not difficult to assume the same can be said for zoos. By furthering the understanding of factors influencing free-living species across zoo sites, zoos can further their conservation mantras by incorporating strategies designed to conserve native bird species facing strong declines.

Conclusion

Given the continuing extensive declines of bird species native to the UK, it is critical that new initiatives are investigated so as to ensure declining populations. Here it is proposed that zoos could play multiple roles in the conservation of native bird species. Initially promoting free-living bird assemblages within zoo grounds leading to the development of biodiversity action plans targeting native species conservation. The following thesis hopes to quantify the potential role zoos could play in the conservation of free-living birds across zoo grounds.

Abstract

Recently, the role of the modern day zoo has shifted away from ‘Ark’ style conservation towards more *in-situ* conservation and education. *In-situ* conservation often focuses upon exotic species and actions taken to conserve these species require extensive funding. A relatively unexplored and less costly form of conservation for zoos to become involved in is the promotion of free-living bird species within zoo grounds. However, within the peer reviewed literature there are few studies which describe free-living species across zoos and only one study (on free-living corvids) attempts to attribute factors within a zoo to the distribution of free-living species. The current study investigates factors affecting free-living bird abundances across multiple zoos. Over a period of seven months, four zoos were studied with 10 enclosures from each zoo being randomly selected for survey. Utilising point counts at an enclosure level, interactions of free-living bird species were recorded. Results highlight both cover within enclosures and the availability of subsidised feeding as significant factors in influencing free-living bird presence within. Further, results demonstrate that factors affecting bird presence and abundance are species specific, with access to subsidised food within an enclosure being significant for both house sparrows (*Passer* domesticus) and blue tits (*Cyanistes caeruleus*), but not for the robin (*Erithacus rubecula*). While the study presented is limited by sample size and survey effort, a novel approach in relating free-living bird presence to enclosure design is described, allowing for future studies to fully explore factors influencing free-living species distributions across zoo grounds. Utilising results from this study, zoos can begin to actively promote native bird conservation and further develop biodiversity action plans designed to encourage native species conservation within zoo grounds.

Introduction

When considering areas of high importance for the promotion of conservation, it is easy to overlook the role of the modern zoo. Indeed, WAZA's (World Association of Zoos and Aquariums) vision of 'a world where zoos and aquariums maximise their conservation effort' highlights clearly that for the modern day zoo, engaging in conservation is a fundamental necessity (WAZA 2019). Analysis of zoo mission statements demonstrates that conservation through education is a prominent goal for zoos collectively (Patrick, Caplow 2018). Three of the main avenues of conservation that zoos strive to engage in are; *Ex-situ* conservation through breeding programmes, *In-situ* conservation targeting high-priority species with the aim to conserve animal and habitat, and on-site educational programmes, often in the form of signage (Redford, Jensen et al. 2012, Walters, Derrickson et al. 2010). As conservation strategies develop, zoos are beginning to shift their focus away from the 'Ark' style breeding programmes so as to support and engage in more *in-situ* conservation, while delivering succinct and meaningful educational messages on site. Indeed, conserving a species habitat through *in-situ* conservation is considered a conservation priority and in 2008 alone zoos contributed roughly $350million USD to various *in-situ* projects (Gusset, Dick 2010, Conde, Flesness et al. 2011).

In-situ conservation is limited however by funds available, meaning that choices have to be made about which species to protect, with funds and research regularly focusing upon more charismatic taxa such as mammals (Rose, Brereton et al. 2019). While most *ex-situ* and *in-situ* conservation done by zoos is to the benefit of exotic species, native species, some of which can be found living freely upon zoo sites, are also facing strong declines. Recognizing the threat to some native species, many zoos are striving to encompass native species into their conservation remit. The 2016 top 10 projects report released by BIAZA highlights ways in which zoos, in alongside various partners, are currently taking action to conserve 10 specific native species (BIAZA 2016). A broader conservation approach taken by Chester zoo is the restoration of an off-site area encompassing 60,000m^2 designated for native species to thrive and for visitors to engage in conservation learning. This falls in line with a well-established initiative set out by the UN (Convention on Biological Diversity 2010) in which the need to restore habitat back to a state in which nature can thrive is highlighted as vital to support native species populations. Zoos themselves however have a unique opportunity to, in addition to creating and managing nature reserves off site, manage their own sites in a way conducive to supporting free-living species as well as the exotics they house.

A role for zoos in the promotion of free-living species on site.

Many zoos across the UK have been established for over 50 years and as such the vegetation within the zoos is well established. It is assumed, and observationally noted, that many free-living bird species live and interact within the zoo, in a broadly similar to how they would in a nature reserve or park. Indeed, data from BIAZA Bioblitz reveals that over 2000 species across various taxa have been recorded within zoo sites, 88 of which are deemed species of high conservation concern requiring urgent action (BIAZA Personal Communication 2017). While factors influencing free-living species diversity within parks and nature reserves are well established, the factors influencing free-living bird species presence within and across zoos is relatively unexplored (Graham, Blake 2001, Bennett, Hinsley et al. 2004, Xie, Lu et al. 2016). Currently there are only a few studies in the peer reviewed literature which focus upon free-living species within zoo grounds and no attempts have been made to relate free-living species count to specific zoo designs. Indeed, a study based in Basel zoo in 2011 identified 3110 free-living species living within the zoo grounds. This study highlighted the clear importance of Basel zoo as a habitat for free-living species, but made no attempt to determine factors influencing their presence (Baur 2011). A further study in 2019 aimed to identify factors influencing corvid presence across zoos. By recording location of crows seen across four zoos, Edinburgh, Vienna, Debrecen and Sappora, across 445 surveys and 297 days, attempts were made to identify factors influencing crow presence. Seasonality was shown to affect crows differently dependent upon which zoo was being surveyed. More crows were present in winter at Edinburgh, Debrecen and Vienna Zoo, whereas at Sappora more crows were present in Summer. Moreover the time of day at which more crows were observed varied across zoos, with significantly more crows being present in the afternoon at Debrecen, Vienna and Sappora, but not at Edinburgh. Further differences were identified in foraging times, with frequency of foraging being highest across seasons in Vienna, but significant declines were seen from summer to winter at Edinburgh Zoo. Both Debrecen and Sappora showed increased foraging frequency in winter compared to summer. One of the main results reported was that there is significant variation in the distribution of foraging and feeding behaviours of crows between zoos with natural food availability being highly important in Vienna, but less so in Edinburgh. Natural vs anthropogenic food sources being a driver of crow presence within zoos is thus suggested to be related to specific habitat structures within zoos. No attempt was made however to directly link zoo design to corvid presence (Kövér, Lengyel et al. 2019). Thus, while both studies clearly highlight the potential that zoos hold in terms of supporting and promoting free-living species, and the roles zoos play in developing urban biodiversity, they do not inform as to what changes would need to be made to further promote free-living species at other zoos. Indeed, zoos wishing to incorporate free-living species conservation into their mantras currently have limited specific research to follow. Given many zoos are located within urban areas, it is suggested that the closest source of literature available is that researching urban biodiversity.

Urban garden management (UGM) is heavily influenced by social, cultural and economic factors (Lepczyk, Aronson et al. 2017, Aronson, Lepczyk et al. 2017) As such, the heterogeneity in garden management is vast (Young, Frey et al. 2019). While heterogeneity is often deemed a positive environmental characteristic in terms of promoting biodiversity (Nichols, Killingbeck et al. 1998), heterogeneity within an urban environment incorporates management practices that are exclusionary, such as artificial grass and plants, and so do not promote biodiversity (Francis, R. A. 2018). Thompson (2007) describes garden management styles as either conventional management practices or near-natural management practices. In reality, these two practices represent the extremes of a gradient of management styles (Thompson, Owen 2007). Near-natural management practices lean towards good understory vegetation, native flora and mature shrubs and trees. All of which have been shown to increase invertebrate abundance and thus positively influence overall bird assemblage (Salisbury, Armitage et al. 2015, Salisbury, Al-Beidh et al. 2020, Zivanovic, Luck 2016, White, Antos et al. 2005). Alternatively, the conventional gardening practice tends to more of a homogeneous, easy to manage garden, creating spaces less favorable for nature (Smith, Gaston et al. 2006). Thus, within an urban setting, the heterogeneity of garden management styles may be further isolating the small patches of habitat which are designed in a way that is conducive to supporting nature. It is therefore paramount that we identify new ways in which we can educate and promote wildlife friendly gardening to those currently not engaged in supporting nature within their back gardens.

Biosecurity

There is, however a trade-off between biosecurity and the health of the exotic animals that they house, and promoting free-living bird species on and within zoo grounds. To protect their housed species from disease, zoos globally have a responsibility to provide suitable biosecurity measures. Good biosecurity ensures the overall health of captive flocks while reducing the spread of disease both on and off premises. DEFRA (Department of Environment Food & Rural Affairs) have recommended biosecurity measures that are good practice and limit overall disease risk (DEFRA 2018).

In 2002, an outbreak of HPAI H5N1 in Hong Kong caused mass mortality in wild migratory avian species (Alexander 2007). HPAI (Highly pathogenic Avian Influenza) is generally characterised by both high morbidity and mortality. To date, HPAI has been found in at least 12 orders of wild birds (Philippa, Baas et al. 2007) however the main reservoir for AIV are birds within the order *Anseriformes* and *Charadriiformes* (Ellis, Barry Bousfield et al. 2004). The only passerine species known to have been infected by this disease is *Passer montanus* (Tree sparrow), however more recent studies concluded that small birds are unlikely to be bridge species in the transmission of AIV(Houston, Azeem et al. 2017).

More prevalent within passerine birds is Avian Pox (van Riper III, van Riper et al. 2002). Caused by avipoxviruses, avian pox has been reported in over 278 avian species (van Riper III, Forrester 2007) and reduces overall species fitness through, decreased predatory avoidance (Laiolo, Serrano et al. 2007), decreased nesting success (Lachish, Bonsall et al. 2012a), decreased pairing success (Wiedenfeld, Fessl et al. 2007) and increased mortality rates (Lachish, Bonsall et al. 2012b). In wild bird populations, one of the major sites for transmission is infected bird feeders (Moyers, Adelman et al. 2018).

Both Avian Pox and HPAI have the potential to be detrimental to captive animals within a zoo site. As such, most zoos have protocols in place so as to, if necessary, prevent contact between exotic and free-living species and contact between exotic enclosures (DEFRA 2018). This protocol however, may be paradoxical to many zoos efforts in the conservation field. By excluding native species from avian enclosures, we are also seeing exclusions from substantial subsidised feeding within zoo grounds. This exclusion has the potential to make the zoo less desirable, especially during harsh winter months, to free-living bird species. The strict hygiene protocols currently followed within zoo grounds, such as the daily replacement of left over foods and fresh water access, could be sufficient enough to ensure that free-living bird species within zoo grounds are not at risk of contracting avian disease. Further, allowing access for smaller passerine species does not require there be access for waterfowl, a major source of the HPAI.

The importance of enclosure design

Within the UK there are over 100 BIAZA accredited zoos and aquariums striving towards best possible care and conservation of the exotic species housed. A significant part of this care is the provision of privacy and regular upkeep of the enclosures in which the animals reside (Hosey, Melfi et al. 2013). A relatively recent shift in overall enclosure design is towards a more naturalistic look, so as to promote wild-type behaviors (Chang, Forthman et al. 1999). Indeed, the design of an enclosure is shown to directly influence the wellbeing of the animal housed (Fàbregas, Guillén-Salazar et al. 2012). In the current study it is suggested that enclosure design has secondary consequences for the free-living species interacting within zoo grounds. Indeed, similar to good practice within urban garden management, varied enclosure designs may be playing a significant role in the promotion of free-living bird assemblages across zoo sites. The influence enclosure design has upon free-living species however is unknown, yet the importance of small isolated habitat for conserving biodiversity is well established (Wintle, Kujala et al. 2019). Given enclosures within a zoo are isolated from one another and are often areas with high vegetation density these may therefore be critical for the zoo's free-living biodiversity. Within urban biodiversity literature, urban gardens are likewise isolated patches of nature key to promoting biodiversity (Goddard, Dougill et al. 2010). Gardens, like enclosures, create a patchwork of fenced spaces all managed independently of one another. Thus, by considering the factors synonymous with influencing bird diversity and abundance within urban gardens it is possible to begin formulating hypotheses as to which factors are key in promoting native species across zoo enclosures.

A hypothesized role for connectivity and food availability

The availability of food within an environment and its overall connectivity are commonly identified as key drivers of urban biodiversity (Plummer, Risely et al. 2019, Crooks, Burdett et al. 2011). Within an enclosure, sources of connectivity may be attributed to the vegetation both within the enclosure itself and the surrounding area. A source of food within an enclosure may also be attributed to the vegetation, providing habitats for invertebrates which are key parts of birds diets (Holland, Hutchison et al. 2006). Alternatively, another source of food is that provided for the exotic species zoos house. Dependent upon the exotic species dietary needs, the food provided will vary and as such it cannot be assumed to be consistently incorporated within the free-living species diet. Further, as and where necessary, biosecurity protocols set in place by zoos may require the feed provided to be indoors or in a place inaccessible to free-living species (DEFRA 2018). In enclosures where subsidized feed is suitable for and accessible to free-living species, it may be a driving factor influencing the presence of free-living species. However, identifying the most influential factors affecting free-living species presence and abundance across zoos remains a significant gap in the research literature. Such research would enable zoos to take up the conservation mantra to a higher degree. By identifying key features of enclosure design hypothesized to be drivers of free-living bird presence within, the influence of enclosure design on the presence and abundance of free-living bird species is investigated.

Aims and Objectives

The aim of the study presented was to identify key drivers of free-living bird presence within zoo grounds. The presence of free-living birds within zoos is commonly accepted, but reasons for their presence, and how zoo design influences their presence is underrepresented within peer reviewed literature. As such this study hopes to act as the stepping stone for future studies to further the research presented and thus further the conservation efforts and overall status of the UK's declining bird species.

It is hypothesized that food availability within an enclosure will be a significant influence on the presence of free-living bird species within, similar to anthropogenic bird feeding within urban environments. Moreover, enclosure design, specifically the structure of natural vegetation within and around, is hypothesized to be a significant factor in influencing the presence free-living bird species due to its ability to provide birds with protective habitat, natural feed and nesting sites. While it could be assumed that a bigger enclosure would promote greater number of free-living bird species, the complexity of vegetation within a large enclosure may not be equal to that of a smaller enclosure housing an exotic species requiring more cover. As such, enclosure size is hypothesized to be a non-significant factor in promoting free-living birds within zoos.

Methods

Study Area

Within each zoo, 10 enclosures deemed accessible to garden birds were selected to be surveyed. To ensure surveying represented a good proportion of the overall zoo, each zoo was split into quadrants. Efforts were made to survey an equivalent number of enclosures across each quadrant, thus reducing the risk of only surveying a small section of the zoo and the impact of double counting. Each enclosure within a quadrat of the zoo was given a number and a random number generator was used to select two enclosures from within each quadrat. The final two enclosures were randomly selected irrespective of the quadrat they were in. If an enclosure selected was not accessible to free-living bird species then it was discarded and a new number generated. Descriptive characteristics of each

Bird assemblages were assessed across four different zoos within the United Kingdom (Table 1).

Table 1

The four zoos surveyed differed in overall size and scale. As shown in Table 1, Chester zoo was the largest in terms of species held, animal numbers, visitors per year and overall size.

enclosure were recorded (Table 2). Enclosure size was measured using Google Earth Pro, utilizing the historic view feature to ensure boundaries were representative (figure 1).

Zoos (2018)	Size (Ha)	Number of species held	Number of animals held	Visitors per year	Year Opened
Chester Zoo	50.6	488	4225 + fish & inverts	1,866,628	1931
Blackpool Zoo)	14.5	184	1118+inverts	458,480	1972
Twycross Zoo	17	122	537 + inverts	521,323	1963
Welsh Mountain Zoo	15	115	616 + fish	144,292	1963

Bird Survey

Data was collected as close to the opening times of each zoo as possible (earliest 9:30am, latest- 10am) and was concluded following the final enclosure point count. While birds are most active at dawn, collecting data within zoo grounds limited the point at which surveys could begin. Future studies could attempt to gain entry to zoos earlier to survey at dawn, this however was not possible within the study presented as access to zoos was not allowed outside normal visiting hours. Moreover, bird behavior changes significantly depending upon the season (Golabek, Ridley et al. 2012). Given this surveys could not be performed across multiple years, data collected is insufficient to allow for seasonality to be investigated as a factor influencing bird presence. An effort was made to survey each zoo once per month for a period of seven months in the year 2019, from January to July. Due to issues with weather and completing paperwork formalities, Welsh Mountain Zoo was visited six times, Twycross and Blackpool five times and Chester Zoo four times. 16 bird species were selected to be surveyed. Birds chosen were of a size able to gain access to partially enclosed enclosures. Birds chosen were also representative of those commonly found within urban gardens. Future studies should look to conduct full bird surveys so as to be able to quantify diversity and thus factors influencing it across zoo sites. Birds chosen to be included within the survey were: blue tit, bullfinch (*Pyrrhula pyrrhula*), chaffinch (*Fringella coelebs*), coal tit (*Periparus ater*), dunnock (*Prunella modularis*), goldcrest (*Regulus regulus*), goldfinch (*Carduelis carduelis*), great tit (*Parus major*), grey wagtail (*Motacilla cinereal*), house sparrow, long tailed tit (*Aegithalos caudatus*), nuthatch (*Sitta*), pied wagtail (*Motacilla alba*), robin, treecreeper (*Certhiidae*) and wren (*Troglodytidae*).

Specified bird assemblages within enclosures were determined utilizing point counts performed for a period of five minutes (Bibby, Burgess et al. 2000) . Due to the variance in enclosure shape and size, observation points were chosen selectively based upon the position at which most of the enclosure could be seen, with the ideal position being that at which the entire enclosure was in view. Point counts were performed for a period of 5 minutes, during which any passerine species seen interacting with the enclosure was recorded. Interactions were defined as a species physically interacting with any part of the enclosure. Birds seen flying through or over the enclosure without physically interacting with the enclosure were not counted.

Enclosure Characteristics

Overall enclosure design was predicted to be a key indicator of both free-living bird diversity and abundance. A summary of the key enclosure characteristics taken into account during surveying is shown in Table 2. Given the inability to access enclosures, a species level description of flora within each enclosure was not feasible, as such the vegetation characteristics of each enclosure was simplified to; Total number of trees and total number of bushes. Where possible trees and bushes were identified to a species level. Otherwise, bushes were defined as woody plants with a height between 0.5 and 3m. Trees were defined as woody plants often considerably taller than bushes with a single stem and branches above ground level (Gschwantner, Schadauer et al. 2009).

It was hypothesized that the surrounding enclosure cover (2m from the designated enclosure boundary) would be a key predictor in garden bird abundance. As access to the surrounding cover was not always available, cover was classified as either; Vegetation on one side, Vegetation on two sides, Vegetation on three sides or Open

Enclosures deemed inaccessible to free-living garden bird species were not surveyed. Variance left within the remaining enclosures accessibility was categorized as either open top or partially open. Open top enclosures were defined as those with only a low wall or fence between the exotic species and free- living species. Partially open enclosures were defined as those surrounded by mesh of a size that allowed, either by design or otherwise, access to free-living birds.

Enclosures in which subsidized feeding was accessible to free-living species were noted. Accessible feed had to be both within the free-living birds diet and available within the enclosure. Further, any water (purposely designed) within the enclosure was noted.

As enclosures surveyed were not uniform in shape, the total coverage visible from the chosen survey point varied. As such enclosures were categorized as either; Partially Visible or Completely Visible

Enclosure size was calculated using Google Earth Pro. Google Earth Pro provides accessible satellite imagery. Utilising both the zoo map provided by the zoo and surveyor knowledge, each enclosure was found on Google Earth Pro and boundaries were drawn to determine enclosure size. If the boundaries of said enclosure were difficult to determine due to vegetation, historic view was used to find an alternate view (fig.1). If no alternate view in which boundaries of the enclosure were clear, the observer used his knowledge of the site to best determine enclosure boundaries.

Table 2

A summary of the enclosure characteristics accounted for during surveys are shown in Table 2.

Enclosure Characteristic	**Explanation**
Subsidised feeding	The presence of feed within an enclosure which is both accessible to free-living bird species and fits within their diet.
Access to water	The presence of a purposely built water feature within an enclosure accessible to free-living bird species.
Cover 2m around the enclosure	The total number of sides with vegetation within a 2m distance from the boundary of the enclosure.
Accessibility	Categorized as either open top or partially open.
Total number of trees	Number of trees within the enclosure. Only counted if the trunk of the tree is within the enclosure boundary.
Total number of bushes	Number of bushes within the enclosure. Only counted if the majority of the bush is within the enclosure boundary.
Enclosure size	Total amount of space within the enclosure boundary (see figure 1).
Visibility	Categorized as either full or partial depending upon the amount of enclosure visible from the viewing point.

Figure 1

Utilising the historic view function in Google Earth Pro to improve boundary visibility. (a) shows satellite imagery of the Red Panda enclosure at Chester Zoo (25/06/2018), in which the boundary of the enclosure is hidden by tree foliage. (b) shows satellite imagery of the same site on the 21st of April 2015, in which the boundaries are clearly visible. Here the enclosure size was determined to be 270m^2.

Statistical Analysis

Differences between zoos were explored through average number of birds per species seen between the four zoos across surveys. The influence of set enclosure variables (Table 2) over the presence of garden birds within an enclosure was explored utilizing linear models with the lme4 package in R (Bates, Mächler et al. 2015). An initial full model was created with enclosure visibility, number of trees, number of bushes, subsidized feeding, access to water, cover within 2m around the enclosure, enclosure size, enclosure accessibility as predictors. The zoo in which the enclosure was located was input as a random factor due to the focus of the study presented being on the influence of zoo design rather than the differences between zoos themselves. Future studies could look to use enclosures as a random factor and the zoo as a fixed effect to allow for direct zoo comparisons to be made. The model was fitted assuming a binomial distribution utilising odds ratios. Odds ratios were calculated using presence/absence data. If an enclosure were surveyed six times, and of those six times, house sparrows were seen interacting within three times, then the odds of it being present within would be 0.5. This 0.5 is then input to the model as the likelihood of the house sparrow interacting within the enclosure. The risk of double counting individuals was notably higher at enclosures in which supplementary feed was available, with large groups of free-living bird species feeding at any one time. Thus using odds ratios calculated using presence/absence data helps to reduce the bias introduced by potential double counting. Further, predicting the distribution of bird species has been shown to be strengthened by utilising presence/absence methods (Brotons, Thuiller et al. 2004). A follow up stepwise variable selection was performed to determine the model with the lowest Akaike Information Criterion (AIC) and so best fit (Petrov 1973). This process was repeated for 3 separate models focusing upon factors affecting house sparrows, blue tits and robins. These three species represented the most abundant and so had data viable for separate models to be run. Due to overdispersion, quasi-AICs were calculated for models created utilising the R package AICcmodavg (Marc J. Mazerolle). A further negative binomial mixed effect model was run to determine factors influencing the abundance of garden birds within an enclosure. This model was again stepped to determine a best fit model however here the model was fit to a quasipoisson distribution to account for overdispersion (Crawley 2012). The impact of overdispersion within models run however cannot be ignored. The sample size for this study is lower than desired, due to a lack of time and funding to collect data at zoos more frequently. Given this study is novel in attributing free-living bird presence to enclosure design, and the enclosures were chosen at random, the distribution of the variables within models across enclosures surveyed is not even. The only variable with a sample size less than 10 however is supplementary feeding, being present in only three of the forty enclosures surveyed. As such it is paramount that future studies aim to increase the number of enclosures surveyed, perhaps identifying specific enclosures with variable of interest highlighted by this study, so as to ascertain their true influence on free-living bird presence. Thus allowing for the results presented to be generalized beyond this study alone. Statistical analysis was performed in R Package version 2. 2-2 (Team 2019).

Results

Differences between zoos in overall garden bird abundance and diversity

Table 3

The average number of individuals seen per survey are presented in table 3 to highlight differences between garden bird assemblages across the four zoos surveyed. Data is accumulated from both point counts and transects walked.

Bird Species	**Zoo Surveyed**			
	Average number of individuals seen per species			
	Blackpool Zoo	**Chester Zoo**	**Twycross Zoo**	**Welsh Mountain Zoo**
Blue Tit	4.20	4.50	1.20	2.33
Bullfinch	0.00	0.25	0.00	0.00
Chaffinch	0.00	1.50	0.40	1.83
Coal Tit	0.60	1.25	0.00	0.50
Dunnock	0.20	0.25	1.40	2.17
Goldcrest	0.00	0.25	0.00	0.17
Goldfinch	1.00	0.50	0.00	0.50
Great Tit	2.00	0.25	0.60	0.67
Grey Wagtail	0.40	0.25	0.00	0.00
House Sparrow	0.00	30.50	34.80	44.67
Long Tailed Tit	0.00	0.50	0.20	0.50
Nuthatch	0.00	0.00	0.20	0.17
Pied Wagtail	0.20	0.50	0.00	0.00
Robin	6.20	7.00	3.80	4.67
Treecreeper	1.00	0.00	0.00	0.00
Wren	1.00	3.00	0.60	0.33
Total	**16.80**	**50.50**	**43.20**	**58.50**

Of the 16 garden bird species recorded, 858 individual birds were seen across all four zoos. Welsh mountain zoo saw the highest average number of birds seen (58.5) independent of species, while Blackpool zoo had the lowest (16.8).There were only five species of passerines found across all zoos (blue tit, dunnock, great tit, robin and wren). The house sparrow was seen across all zoos except for Blackpool. At the zoos it was present at, the house sparrow represented over fifty percent of all birds seen during surveying (Table 2) .

Identifying variance between enclosures.

Clear differences are presented in the diversity and abundance of free-living birds surveyed within the 40 enclosures studied (Figure 2). The Agouti enclosure at Welsh mountain zoo had the highest overall diversity (1.818)(Figure 2A). Whereas the Amazonia Bird enclosure had the highest overall number of free-living birds within (149). Only 19 of the 40 enclosures surveyed had an abundance of free-living birds high enough for a Shannon Weiner to be calculated, with nine enclosures having no birds seen during all point counts conducted (Figure 2).

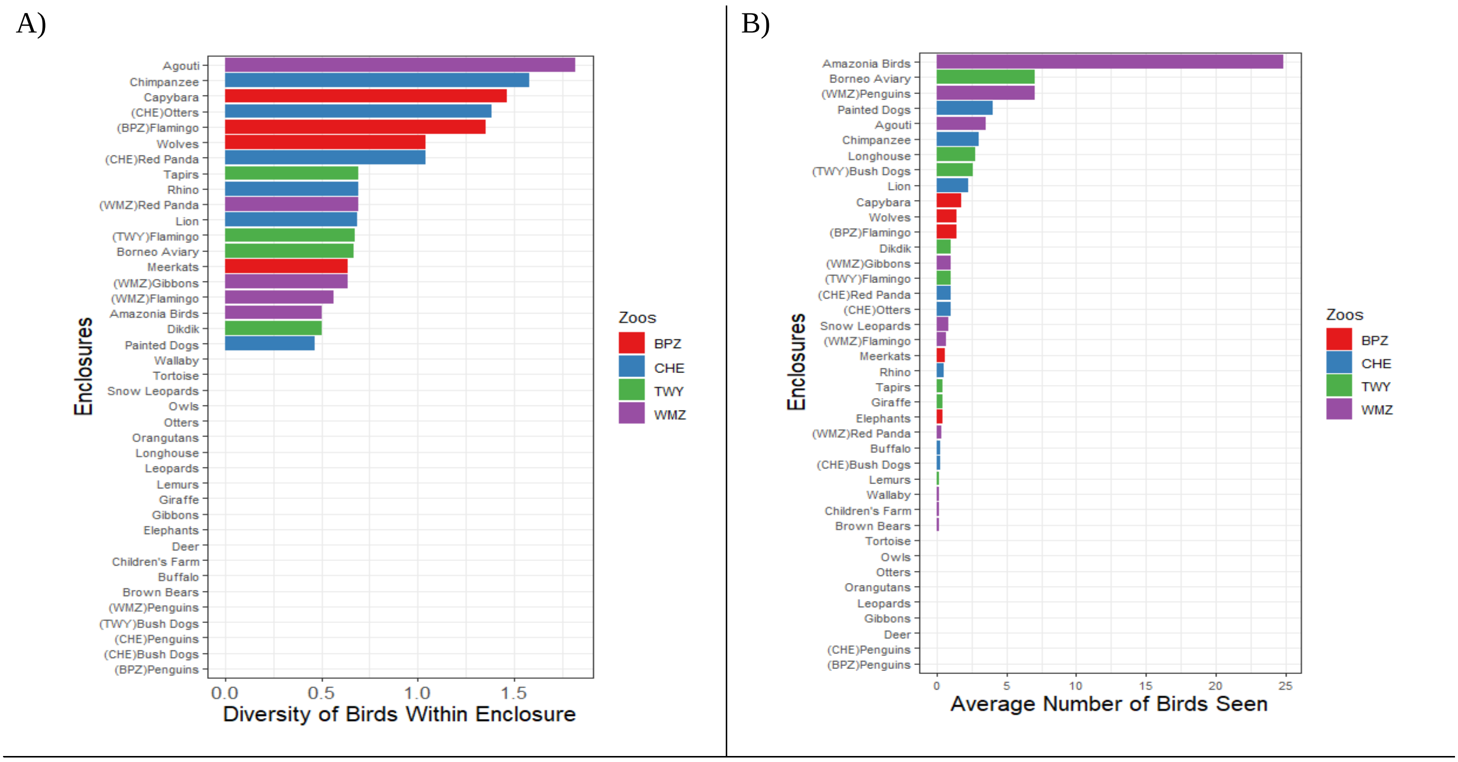


Figure 2
Variance in bird diversity (A) and total number of birds seen (B) across 40 enclosures. Diversity values are calculated using Shannon Weiner index and only include birds species surveyed for, as such diversity values are not a true representation of overall zoo diversity. Enclosures were surveyed by performing point counts for five minutes at a point at which the majority of the enclosure could be observed. BPZ = Blackpool Zoo, CHE = Chester Zoo, TWY = Twycross Zoo, WMZ = Welsh Mountain Zoo.

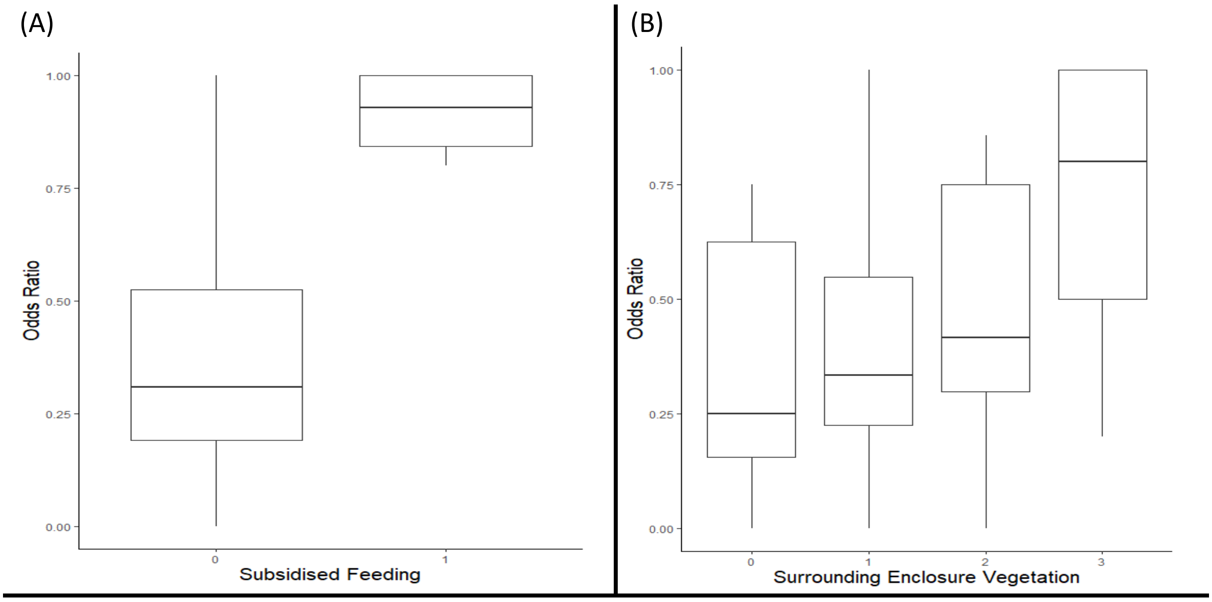
Identifying specific enclosure design factors influencing the presence of free-living bird species.

The most significant variable influencing the presence/absence of garden birds within enclosures was subsidised feeding (p = 0.0002) (Table 4). The presence of subsidised food within an enclosure is shown to significantly increase the likelihood of free-living birds within (Figure 3a). While not significant (P = 0.113) (Table 4), a clear directionality is shown that as surrounding enclosure vegetation increases so does the likelihood of presence of free-living birds within (Figure 3b).

Table 4

Results of a generalized mixed effect model. The likelihood of birds interacting within enclosures tested again fixed factors, with the zoo in which the enclosure was located as a random factor.
Family = binomial. Significant results are highlighted in bold. P values were calculated via model comparison.

Fixed Factor	**Full Model**			**Final Model**		
	Chi Sq	***Df***	***p***	***Chi Sq***	***Df***	***p***
Subsidised Feeding	**15.375**	**1**	**<0.001**	**17.836**	**1**	**<0.001**
Cover around enclosure	6.711	3	0.082	5.973	3	0.113
Number of Trees	0.085	1	0.770			
Number of Bushes	2.353	1	0.125	2.693	1	0.101
Access to Water	0.029	1	0.865			
Enclosure Visibility	1.523	1	0.217	2.309	1	0.129
Open/Enclosed enclosure	0.215	1	0.230			
Enclosure Size	0.030	1	0.864			



<u>Figure 3</u>

Factors influencing the presence of garden birds within enclosures. (A) Subsidised feeding (B) Number of sides with vegetation within a 2m of the designated boundaries of the enclosure. The availability of subsidised food within an enclosure is represented by 1, whereas no feed is 0.

<u>Identifying specific enclosure design factors influencing the abundance of free-living birds within enclosures</u>

Analysis of factors of enclosure design the availability of subsidised food to be a significant factor in influencing abundance of free-living birds ($P < 0.001$) (Table 5). The number of bushes within an enclosure is also shown to significantly influence the abundance of free-living birds within ($P = 0.016$) (Table 5). Both the availability of subsidised food and an increased number of bushes within an enclosure are shown to significantly increase the abundance of free-living bird species within (Figure 6).

Table 5

Results of a negative binomial generalized mixed effect model. Fixed effects were tested against the abundance of birds seen interacting within an enclosure, with the zoo in which enclosures were located designated as a random effect. Family = quasipoisson. Significant results are highlighted in bold and were calculated via model comparison.

Fixed Factor	**Full Model**			**Final Model**		
	Chi Sq	***Df 1***	***P***	***Chi Sq***	***Df 1***	***P***
Subsidised Feeding	**14.396**	**1**	**<0.001**	**19.263**	**1**	**<0.001**
Cover around enclosure	6.999	3	0.072			
Number of Trees	0.038	1	0.846			
Number of Bushes	**4.669**	**1**	**0.031**	**5.859**	**1**	**0.0155**
Access to Water	1.546	1	0.214			
Enclosure Visibility	3.668	2	0.160			
Open/Enclosed enclosure	**4.212**	**1**	**0.040**			

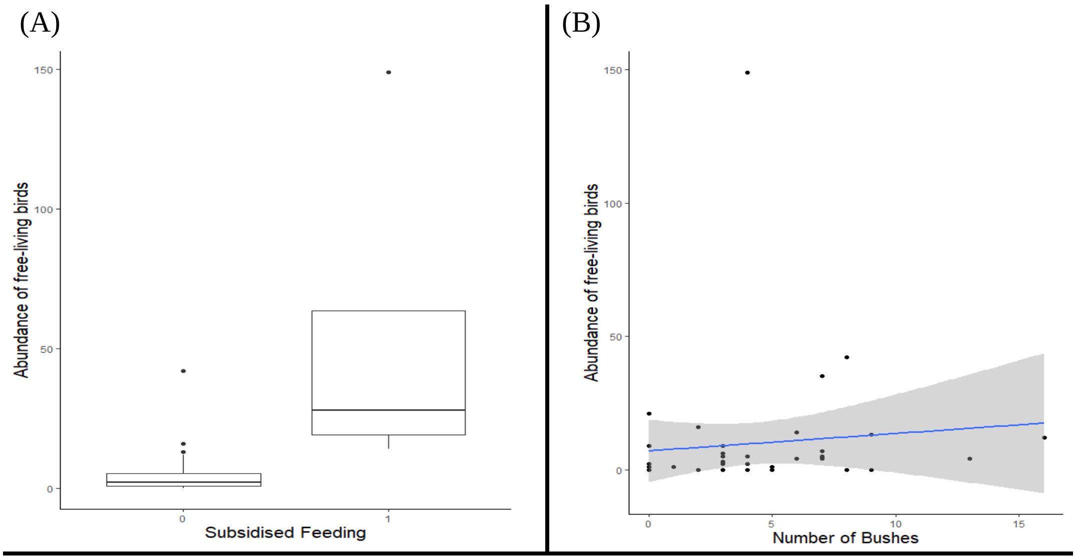


Figure 4

Significant factors in influencing the abundance of garden bids within zoo enclosures. The availability of subsidised food within an enclosure is represented by 1, whereas no feed is 0.

Species specific directionality of the significant factors influencing the presence of free-living birds within enclosures.

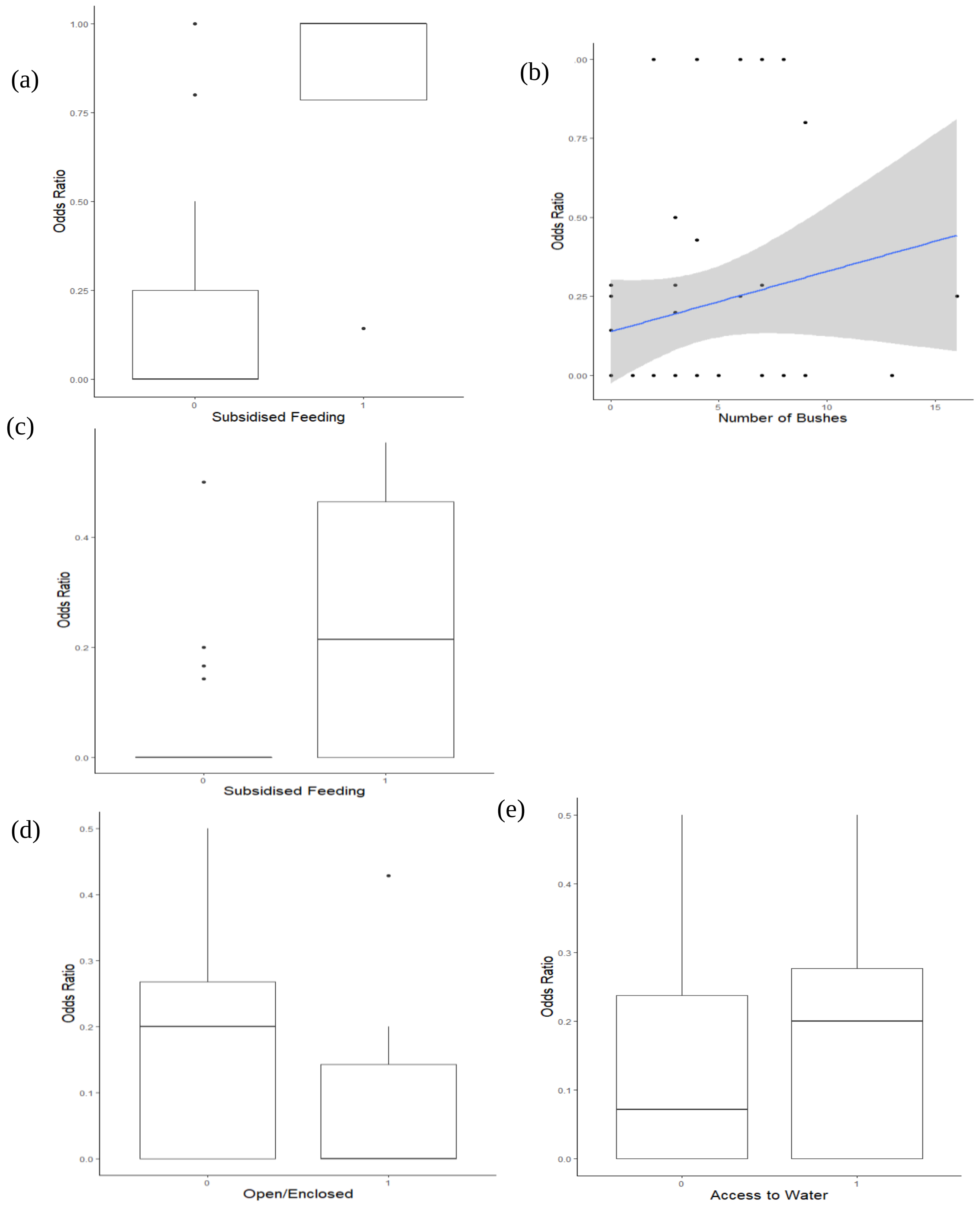


Figure 5 - Directionality of significant factors influencing the presence of the house sparrow (a and b), blue tit (c) and robin (d and e).

Significant factors in influencing the presence of individual species within enclosures. The availability of subsidised feeding within an enclosure is represented by 1, whereas no subsidised feeding is 0. An open top enclosure is represented by 0 and a partially enclosed enclosure 1 (d). An enclosure in which water is accessible is represented by 1 and 0 if not (f).

When looking at individual species, subsidised feed is shown to be a significant factor in influencing the presence of both house sparrows (a) ($P < 0.001$) and blue tits (c) ($P < 0.001$) within an enclosure, as is the number of bushes (b) ($P < 0.001$) for house sparrows.. Robins were significantly more likely to be present in open top enclosures with access to water (e and f) ($P < 0.001$ and $P < 0.001$) whereas the availability of subsidised feeding within and the number of bushes was not significant.

Results presented highlight the importance of subsidised feed in promoting both presence within enclosures and overall abundance of free-living birds. Indeed, in the five models run the presence of subsidised feeding within an enclosure was a significant factor in four (Table 4, Table 5 and Figure 5a and c). Moreover, three of the five models showed the number of bushes to have a significant positive correlation to presence and abundance of free-living birds within an enclosure (Figure 4b, Figure 6b and Figure 7b). An enclosure with an abundance of free-living birds interacting within however does not mean said enclosure is diverse in species (Figure 2). Indeed of the 16 birds species surveyed for, the house sparrow represented over 50% of all birds seen (Figure 2). Thus, the factors of enclosure design important in influencing bird assemblages are shown to be species dependent (Figure 7).

Discussion

Data presented describes a novel approach in relating surveyed free-living native birds within zoos to enclosure design. The study presented aids in the understanding of how surveyed free-living birds interact within zoo grounds.

The major finding is that the addition of subsidised feed within an enclosure clearly increases the likelihood of the surveyed free-living birds interacting within (Figure 3). This relationship was further linked with the overall abundance of birds surveyed within an enclosure (Figure 4a). The availability of feed within an environment has been shown to be a key factor in driving bird assemblages many times across varying environments (Martin 1987, Newton 1998, Lack 1954). In keeping with the current study, food availability has been shown to directly influence bird abundances in urban environments (Fuller, Warren et al. 2008). Within a zoo environment, Kövér and Lengyel demonstrate the importance of food in driving corvids to zoos, and results presented in the current study suggest this relationship is true to other free-living birds surveyed within enclosures (Kövér, Lengyel et al. 2019). Of the free-living birds surveyed within enclosures, the house sparrow, blue tit and robin were most abundant, allowing for separate models to be created exploring species specific influences (Table 3). Results show that for both the house sparrow and blue tit, the availability of food within an enclosure significantly increases the likelihood of presence within an enclosure. Previous studies on house sparrow declines have linked population changes with reduced food availability due to changes in farmland practice (Newton 2004). Indeed, reduction in winter survival in house sparrow populations has been linked to the loss of seed-rich winter stubble (MacLeod, Barnett et al. 2006). Further, chick mortality rate has shown to increase with the reduction of natural feed availability (Vincent 2005a). Thus, while the decline of the house sparrow is still debated, it is difficult to overlook food availability as a key driver. Here, support is provided for food being a key factor in influencing house sparrow populations, with the availability of food increasing the likelihood of the house sparrow interacting within an enclosure. Further studies should aim to identify whether this influence is because food is lacking within the environment, or whether the subsidised food presented is easier accessed than simply foraging for food.

Conversely, the robin was the second most abundant species seen during surveys (Table 3), yet the availability of food was not a significant factor in influencing it's presence. Rather, both open top enclosures and the access to water were significant influential factors (Figure 5, d and e). The differences seen in the significant factors across species are likely due to variance in species specific traits, such as diet and sociality, which impact how feed availability can be utilized. Given the house sparrow is a colonial nester, it is a highly social bird and so large populations are common (Summers-Smith 1963). The blue tit is also a social bird outside of the breeding season, often seen in interspecies flocks so as to avoid predation (Piper 1997). Here it is suggested that the larger range of blue tits compared to the house sparrow allows for blue tits to be less reliant upon site specific vegetation and so any form of enclosure cover is not a significant factor in influencing their presence. Instead the subsidised feed is the only significant factor accounted for within the study presented that is shown to positively influence the presence of blue tits within zoo enclosures. The robin however is highly territorial and so an enclosure with subsidised feeding may only fall into a single robin's territory (Tobias 1997). Food availability within an enclosure is therefore suggested to benefit some free-living bird species more than others, particularly more social bird species. Further, the type of feed available was not quantified, yet remains an important variable in understanding reasons for the significance determined. Future studies should look to identify types of subsidised feed and its influence upon species interactions within an enclosure.

Another important factor that appeared significant in influencing abundance of free-living birds within an enclosure and the presence of house sparrows (Figure 4b and Figure 5b) was the number of bushes present. The number of bushes and number of trees within an enclosure were factors chosen to quantify overall enclosure vegetation without needing to gain access to inside an enclosure. While the number of bushes was a significant factor in both the full model and stepped model for abundance (Table 5) ($P = 0.031$, $P = 0.016$ respectively) the number of trees was not significant in the full model ($P = 0.846$) and cut from the stepped model. It is suggested that the number of bushes is representative of understory vegetation which has been shown in urban literature to be a key driver of biodiversity (Salisbury, Armitage et al. 2015). Here it is suggested that the number of trees within an enclosure may not be as important as factors such as canopy height or canopy coverage and so did not present as significant. Future studies should aim to quantify enclosure vegetation in depth to better explain driving factors of free-living bird presence and hopefully reduce overdispersion commonly occurring in count data. Suggestions include vegetation density which could be quantified utilising point clouds with programmes such as GeoSlam & vegetation diversity within the enclosure (Wang, Sun et al. 2019). The number of bushes being a significant factor in influencing bird presence within enclosures can be explained utilising urban biodiversity literature. Fledgling success rate has been shown to increase with invertebrate abundance within urban environments and specifically linked to house sparrow fledgling success rate (Peach, Mallord et al. 2015, Vincent 2005b). Increased vegetation, specifically understory vegetation has further been linked to increased invertebrate abundance (Salisbury, Al-Beidh et al. 2017, Salisbury, Al-Beidh et al. 2020). Thus, having an increased number of bushes within an enclosure increases the availability of natural feed. Further to supporting invertebrates, bushes within an enclosure aid free-living birds through creating cover and increasing overall connectivity of the site. Cover has been shown to positively influence both species richness and diversity (Trzcinski, Fahrig et al. 1999, Radford, Bennett et al. 2005). Bushes and hedgerows have further been shown to act as protective cover for house sparrows (Lazarus, Symonds 1992). Thus, increased connectivity and number of bushes within an enclosure aids in predator avoidance and allowing the free movement of individuals throughout the zoo site. While no attempt was made made here to quantify the connectivity of an enclosure directly, the future studies could attempt to quantify overall zoo connectivity following methods set about by Spnowicz and Jaeger (2019)(Spanowicz, Jaeger 2019).

Subsidised feed and enclosure cover (represented here by the number of bushes) are therefore key factors in influencing the presence of surveyed birds within enclosures and their overall abundance. This in turn provides support for the suggestion made by Kover et al., (2019) in which it was suggested that the structure of habitats is important in influencing free-living bird presence within zoos. Differences seen between the house sparrow, blue tit and robin across significant factors however (figure 5) demonstrates that these factors are not equally influential across all species. This is further supported by the average number of birds seen per species (Table 3) which demonstrates that across the zoos it was seen at, the house sparrow was the most abundant. Given the house sparrow has faced significant declines throughout Europe, zoos surveyed are suggested to be important sites for the conservation of this species (Anderson 2006).f

A key difference noted between zoos is that no house sparrows were observed at Blackpool Zoo (Table 3). It is important to note that data from BIAZA's own bioblitz records show them being previously observed here, so the lack of presence could be as a result of survey error. For example it is possible that house sparrows are present are present at Blackpool zoo but in areas not encompassed by survey efforts. Indeed, only 10 enclosures were surveyed within each zoo so it is possible that an enclosure not encompassed within survey efforts may be supporting house sparrows. Given the zoo sample size for this study is small (four) it is difficult to ascertain significant differences between zoos so as to begin to understand why bird assemblage differed at Blackpool zoo. Observationally, there were no enclosures at Blackpool zoo at which suitable subsidised feed was available. Given the significance of feed in influencing bird presence within enclosures it is suggested that this could be a reason for lower abundances. Chester zoo however also had no access to subsidised feed for free-living species within enclosures surveyed and yet had both large numbers of house sparrows and overall the second highest bird abundance. It is worth noting that the anthropogenic bird feeding done by visitors at zoos either by purposely feeding free-living birds or accidently through discarded food was not included within this study. Indeed within a zoo the feed provided by visitors and around outdoor eating areas could be another driver of bird presence and warrants future study. It is possible that the connectivity across Blackpool zoo is lower compared to the other zoos surveyed, thus acting as a limiting factor for the abundance of birds surveyed. It is suggested that an attempt to quantify vegetation structure across multiple zoo sites is required to identify reasons for differences in bird assemblages between zoos.

Indeed if a biodiversity action plan were to be developed based on the results presented, clear goals need to be outlined to ensure target species are represented within enclosure design changes made. Alternatively, if an enclosure provides poor habitat for targeted free-living bird species, habitat elsewhere within the zoo should be developed so as to mitigate potential losses. Previous studies have shown that bird communities have lower species richness but a higher overall abundance in urban environments (Mills, Dunning Jr et al. 1989). This relationship between diversity and abundance is shown within zoo enclosures surveyed in which an enclosure with high abundance does not necessarily have high diversity (figure 2). If the aim is to increase overall abundance of free-living birds across zoos then the changes initiated may differ to an aim of increasing overall diversity within a zoo. Future studies should attempt to identify a preferred conservation approach when utilising zoos as a space for free-living bird conservation so as to ensure that the approach employed achieves the desired goals. Moreover, enclosures such as the Chimpanzee enclosure at Chester Zoo show high diversity with low overall abundance. This follows trends seen in more natural environments in which diversity and species evenness is high (Pautasso, Böhning-Gaese et al. 2011). It is suggested here that by investigating the factors of enclosure design influencing free-living bird diversity and abundance, changes could be initiated within urban environments that are designed to promote bird assemblages more representative of a natural environment. Thus aiding in the conservation of many urban living species through the promotion of greater species diversity and reducing the taxonomic homogeneity which is more typical in this environment (Ortega-Álvarez, MacGregor-Fors 2009).

Before any attempt is made to further encourage the presence of free-living bird species within enclosures, it is vital to assess the risk to exotic species this brings. By encouraging free-living bird species into enclosures the risk to housed species is assumed to increase, however that this risk can be negated through careful consideration of which enclosures free-bird species are encouraged within. One of the major diseases prevalent to passerine bird species is Avian Pox, caused by avipoxviruses (van Riper III, van Riper et al. 2002, van Riper III, Forrester 2007). Reported in over 278 avian species, Avian Pox significantly increases mortality rates and in wild populations a major site for transmission is infected bird feeders (van Riper III, van Riper et al. 2002, van Riper III, Forrester 2007, Lachish, Bonsall et al. 2012, Wiedenfeld, Fessl et al. 2007, Moyers, Adelman et al. 2018). Here it is suggested that specific zoo protocols, such as regular enclosure inspection and cleaning will help to reduce the risk of avian pox across zoo sites. Thus reducing the risk that encouraging free-living passerine species will bring to exotic species health and wellbeing.

Moreover, through identifying factors influencing bird presence within an enclosure, it is possible to begin to actively exclude free-living birds from at risk enclosures. Further, while this study focused upon the presence of free-living birds within enclosures, significant factors identified can be used to create areas outside of enclosures for the promotion of free-living species, thus limiting any risks. This is supported by the abundance of free-living birds at Chester Zoo where no enclosures surveyed had accessible subsidised food within. Demonstrating how good habitat outside of enclosures also promotes free-living bird species. It is important to also consider that through encouraging free-living bird species into enclosures, the exotic species within that draws visitor attention can be used as an umbrella species to highlight the plight of native bird species. Many of the conservation issues facing free-living bird species is as a result of urbanisation, and the lack of natural spaces. By promoting free-living bird species within a zoo site, conservation schemes can be developed targeting specific enclosures, likely to have free-living species interacting within. These enclosures can be utilized to highlight conservation issues surrounding the free-living bird species and suggest ways in which visitors can manage their own urban gardens to further support nature.

Conclusion

When we consider the role of a modern day zoo, that of preserving and protecting animals in their natural habitat, it is easy to get lost in the exotic side of the conservation. Here findings are presented which demonstrate clearly that enclosure design (subsidised feeding and enclosure cover) within a zoo directly correlates with the presence of free-living garden birds within. While the conservation of exotic species in their natural habitat remains in many ways the most important role modern day zoos play, simple strategies based upon enclosure design can bring an additional layer to the conservation remit zoos strive for.

Acknowledgements

First and foremost, I'd like to thank my supervisor Dr Katherine Jones, whose guidance throughout this project has been invaluable. I'd like to extend this thanks to multiple members of staff working throughout Bangor University, whom without, my time at University would have been less enjoyable and successful. Most notably, Katie Coburn and Wynn Johnston who have provided countless opportunities which have, and will, benefit me beyond University.

I'd further like to thank the four zoos that allowed me to conduct research at their site, Chester Zoo, Blackpool Zoo, Welsh Mountain Zoo and Twycross Zoo and the respective research officers whom I communicated with Kate Brankin; Jess Aikman, Helen Shaw and Freisha Patel.

Finally, I'd like to thank my flat mate Devlan Alkins, for distracting me to no end, my girlfriend Lucy Walker for supporting me with countless cups of tea and crumble, and my parents, for supporting me and ultimately making me a better academic.

References

AKAIKE, H., PETROV, B.N. and CSAKI, F., 1973. Second international symposium on information theory.

ALEXANDER, D.J., 2007. Summary of avian influenza activity in Europe, Asia, Africa, and Australasia, 2002–2006. *Avian Diseases,* **51**(s1), pp. 161-166.

ALVEY, A.A., 2006. Promoting and preserving biodiversity in the urban forest. *Urban Forestry & Urban Greening,* **5**(4), pp. 195-201.

AMAR, A., HEWSON, C.M., THEWLIS, R.M., SMITH, K.W., FULLER, R.J., LINDSELL, J.A., CONWAY, G., BUTLER, S. and MACDONALD, M.A., 2006. What's happening to our woodland birds. *Long-term changes in the populations of woodland birds,* .

ANDERSON, G., BRADBURY, R.B. and EVANS, A.D., 2001. *Evidence for the effects of agricultural intensification on wild bird populations in the UK.* RSPB.

ANDERSON, T.R., 2006. *Biology of the ubiquitous house sparrow: from genes to populations.* Oxford University Press.

ANGOLD, P.G., SADLER, J.P., HILL, M.O., PULLIN, A., RUSHTON, S., AUSTIN, K., SMALL, E., WOOD, B., WADSWORTH, R. and SANDERSON, R., 2006. Biodiversity in urban habitat patches. *Science of the Total environment,* **360**(1-3), pp. 196-204.

ARAÚJO, M.B., 2003. The coincidence of people and biodiversity in Europe. *Global Ecology and Biogeography,* **12**(1), pp. 5-12.

ARNOLD, G.W., 1983. The influence of ditch and hedgerow structure, length of hedgerows, and area of woodland and garden on bird numbers on farmland. *Journal of Applied Ecology,* , pp. 731-750.

ARONSON, M.F., LEPCZYK, C.A., EVANS, K.L., GODDARD, M.A., LERMAN, S.B., MACIVOR, J.S., NILON, C.H. and VARGO, T., 2017b. Biodiversity in the city: key challenges for urban green space management. *Frontiers in Ecology and the Environment,* **15**(4), pp. 189-196.

ARONSON, M.F., HANDEL, S.N., LA PUMA, I.P. and CLEMANTS, S.E., 2015. Urbanization promotes non-native woody species and diverse plant assemblages in the New York metropolitan region. *Urban Ecosystems,* **18**(1), pp. 31-45.

ARONSON, M.F., LA SORTE, F.A., NILON, C.H., KATTI, M., GODDARD, M.A., LEPCZYK, C.A., WARREN, P.S., WILLIAMS, N.S., CILLIERS, S. and CLARKSON, B., 2014. A global analysis of the impacts of urbanization on bird and plant diversity reveals key anthropogenic drivers. *Proceedings of the Royal Society B: Biological Sciences*, **281**(1780), pp. 20133330.

ALEXANDER, D.J., 2007. Summary of avian influenza activity in Europe, Asia, Africa, and Australasia, 2002–2006. *Avian Diseases,* **51**(s1), pp. 161-166.

AMAR, A., HEWSON, C.M., THEWLIS, R.M., SMITH, K.W., FULLER, R.J., LINDSELL, J.A., CONWAY, G., BUTLER, S. and MACDONALD, M.A., 2006. What's happening to our woodland birds. *Long-term changes in the populations of woodland birds,* .

ANDERSON, G., BRADBURY, R.B. and EVANS, A.D., 2001. *Evidence for the effects of agricultural intensification on wild bird populations in the UK.* RSPB.

ANGOLD, P.G., SADLER, J.P., HILL, M.O., PULLIN, A., RUSHTON, S., AUSTIN, K., SMALL, E., WOOD, B., WADSWORTH, R. and SANDERSON, R., 2006. Biodiversity in urban habitat patches. *Science of the Total environment,* **360**(1-3), pp. 196-204.

ARNOLD, G.W., 1983. The influence of ditch and hedgerow structure, length of hedgerows, and area of woodland and garden on bird numbers on farmland. *Journal of Applied Ecology,* , pp. 731-750.

ARONSON, M.F., LEPCZYK, C.A., EVANS, K.L., GODDARD, M.A., LERMAN, S.B., MACIVOR, J.S., NILON, C.H. and VARGO, T., 2017a. Biodiversity in the city: key challenges for urban green space management. *Frontiers in Ecology and the Environment,* **15**(4), pp. 189-196.

ARONSON, M.F., LEPCZYK, C.A., EVANS, K.L., GODDARD, M.A., LERMAN, S.B., MACIVOR, J.S., NILON, C.H. and VARGO, T., 2017b. Biodiversity in the city: key challenges for urban green space management. *Frontiers in Ecology and the Environment,* **15**(4), pp. 189-196.

BATÁRY, P., MATTHIESEN, T. and TSCHARNTKE, T., 2010. Landscape-moderated importance of hedges in conserving farmland bird diversity of organic vs. conventional croplands and grasslands. *Biological Conservation,* **143**(9), pp. 2020-2027.

BATES, D., MÄCHLER, M., BOLKER, B. and WALKER, S., 2015. Fitting Linear Mixed-Effects Models Using lme4. *Journal of statistical software,* **67**(1), pp. 1-48.

BAUR, B., 2011. Basel Zoo and its native biodiversity between the enclosures: a new strategy of cooperation with academic institutions. *International zoo yearbook,* **45**(1), pp. 48-54.

BENNETT, A.F., HINSLEY, S.A., BELLAMY, P.E., SWETNAM, R.D. and MAC NALLY, R., 2004. Do regional gradients in land-use influence richness, composition and turnover of bird assemblages in small woods? *Biological Conservation,* **119**(2), pp. 191-206.

BIAZA, 25th August, 2016-last update, Top Ten Native Species Benefiting From Zoos and Aquariums. Available: https://biaza.org.uk/news/detail/top-ten-native-species-benefitting-from-zoos-and-aquariums.

BIAZA PERSONAL COMMUNICATION 2017, *Unpublished Bioblitz Data.*

BJERKE, T., ØSTDAHL, T., THRANE, C. and STRUMSE, E., 2006. Vegetation density of urban parks and perceived appropriateness for recreation. *Urban Forestry & Urban Greening,* **5**(1), pp. 35-44.

BONNINGTON, C., GASTON, K.J. and EVANS, K.L., 2015. Ecological traps and behavioural adjustments of urban songbirds to fine-scale spatial variation in predator activity. *Animal Conservation,* **18**(6), pp. 529-538.

BRAAKER, S., MORETTI, M., BOESCH, R., GHAZOUL, J., OBRIST, M.K. and BONTADINA, F., 2014a. Assessing habitat connectivity for ground-dwelling animals in an urban environment. *Ecological Applications,* **24**(7), pp. 1583-1595.

BRAAKER, S., MORETTI, M., BOESCH, R., GHAZOUL, J., OBRIST, M.K. and BONTADINA, F., 2014b. Assessing habitat connectivity for ground-dwelling animals in an urban environment. *Ecological Applications,* **24**(7), pp. 1583-1595.

BROTONS, L., THUILLER, W., ARAÚJO, M.B. and HIRZEL, A.H., 2004. Presence-absence versus presence-only modelling methods for predicting bird habitat suitability. *Ecography,* **27**(4), pp. 437-448.

BROUSSEAU, P., LEFEBVRE, J. and GIROUX, J., 1996. Diet of ring-billed gull chicks in urban and non-urban colonies in Quebec. *Colonial Waterbirds,* , pp. 22-30.

CANNON, A., 2000. *Garden birdwatch handbook.* Copus.

CHANG, T.R., FORTHMAN, D.L. and MAPLE, T.L., 1999. Comparison of confined mandrill (Mandrillus sphinx) behavior in traditional and "ecologically representative" exhibits. *Zoo Biology: Published in affiliation with the American Zoo and Aquarium Association,* **18**(3), pp. 163-176.

CHURCH, G.E., FURNESS, R.W., TYLER, G., GILBERT, L. and VOTIER, S.C., 2019. Change in the North Sea ecosystem from the 1970s to the 2010s: great skua diets reflect changing forage fish, seabirds, and fisheries. *ICES Journal of Marine Science,* **76**(4), pp. 925-937.

CONVENTION ON BIOLOGICAL DIVERSITY, 2010. Strategic plan for biodiversity 2011â€"2020â€• COP 10, decision X/2, *Convention on Biological Diversity* 2010.

CORNELIS, J. and HERMY, M., 2004. Biodiversity relationships in urban and suburban parks in Flanders. *Landscape and Urban Planning,* **69**(4), pp. 385-401.

COX, D.T. and GASTON, K.J., 2016. Urban bird feeding: connecting people with nature. *PloS one,* **11**(7), pp. e0158717.

CRAWLEY, M.J., 2012. *The R book.* John Wiley & Sons.

CROOKS, K.R., BURDETT, C.L., THEOBALD, D.M., RONDININI, C. and BOITANI, L., 2011. Global patterns of fragmentation and connectivity of mammalian carnivore habitat. *Philosophical Transactions of the Royal Society B: Biological Sciences,* **366**(1578), pp. 2642-2651.

DAUWE, T., BERVOETS, L., PINXTEN, R., BLUST, R. and EENS, M., 2003. Variation of heavy metals within and among feathers of birds of prey: effects of molt and external contamination. *Environmental Pollution,* **124**(3), pp. 429-436.

DEFRA 2018, *Biosecurity and preventing welfare impacts in poultry and captive birds.*

DEFRA, A., 2002. Department for Environment, Food and Rural Affairs CSG 15, *The Workshop on Climate Change Research (" Farming and Climate Change: adapting to the challenges") was a one-day event held at the University of Gloucestershire on* 2002.

DONALD, P.F., GREEN, R.E. and HEATH, M.F., 2001. Agricultural intensification and the collapse of Europe's farmland bird populations. *Proceedings of the Royal Society of London.Series B: Biological Sciences,* **268**(1462), pp. 25-29.

DUNN, E.H. and TESSAGLIA-HYMES, D.L., 2001. *Birds at your feeder: a guide to feeding habits behavior distribution and abundance.* WW Norton & Company.

EATON, M., AEBISCHER, N., BROWN, A., HEARN, R., LOCK, L., MUSGROVE, A., NOBLE, D., STROUD, D. and GREGORY, R., 2015. Birds of Conservation Concern 4: the population status of birds in the UK, Channel Islands and Isle of Man. *British Birds,* **108**, pp. 708-746.

ELLIS, T.M., BARRY BOUSFIELD, R., BISSETT, L.A., DYRTING, K.C., LUK, G.S., TSIM, S.T., STURM-RAMIREZ, K., WEBSTER, R.G., GUAN, Y. and PEIRIS, J.M., 2004. Investigation of outbreaks of highly pathogenic H5N1 avian influenza in waterfowl and wild birds in Hong Kong in late 2002. *Avian Pathology,* **33**(5), pp. 492-505.

FÀBREGAS, M.C., GUILLÉN-SALAZAR, F. and GARCÉS-NARRO, C., 2012. Do naturalistic enclosures provide suitable environments for zoo animals? *Zoo biology,* **31**(3), pp. 362-373.

FORD, H.A., BARRETT, G.W., SAUNDERS, D.A. and RECHER, H.F., 2001. Why have birds in the woodlands of southern Australia declined? *Biological Conservation,* **97**(1), pp. 71-88.

FRANCIS, C.D., ORTEGA, C.P. and CRUZ, A., 2009. Noise pollution changes avian communities and species interactions. *Current biology,* **19**(16), pp. 1415-1419.

FRANCIS, R.A., 2018. Artificial lawns: Environmental and societal considerations of an ecological simulacrum. *Urban Forestry & Urban Greening,* **30**, pp. 152-156.

FULLER, R.J., SMITH, K.W., GRICE, P.V., CURRIE, F.A. and QUINE, C.P., 2007. Habitat change and woodland birds in Britain: implications for management and future research. *Ibis,* **149**, pp. 261-268.

FULLER, R.A., WARREN, P.H., ARMSWORTH, P.R., BARBOSA, O. and GASTON, K.J., 2008. Garden bird feeding predicts the structure of urban avian assemblages. *Diversity and Distributions,* **14**(1), pp. 131-137.

FULLER, R.J., NOBLE, D.G., SMITH, K.W. and VANHINSBERGH, D., 2005. Recent declines in populations of woodland birds in Britain. *British Birds,* **98**, pp. 116-143.

GIBBONS, D.W., AVERY, M., BAILLIE, S., GREGORY, R.D., KIRBY, J., PORTER, R., TUCKER, G. and WILLIAMS, G., 1996. Bird species of conservation concern in the United Kingdom, Channel Islands and Isle of Man: revising the Red Data List. *RSPB Conservation Review,* **10**, pp. 7-18.

GILL, R.M. and FULLER, R.J., 2007. The effects of deer browsing on woodland structure and songbirds in lowland Britain. *Ibis,* **149**, pp. 119-127.

GODDARD, M.A., DOUGILL, A.J. and BENTON, T.G., 2010. Scaling up from gardens: biodiversity conservation in urban environments. *Trends in ecology & evolution,* **25**(2), pp. 90-98.

GOLABEK, K.A., RIDLEY, A.R. and RADFORD, A.N., 2012. Food availability affects strength of seasonal territorial behaviour in a cooperatively breeding bird. *Animal Behaviour,* **83**(3), pp. 613-619.

GRAHAM, C.H. and BLAKE, J.G., 2001. Influence of patch-and landscape-level factors on bird assemblages in a fragmented tropical landscape. *Ecological Applications,* **11**(6), pp. 1709-1721.

GREGORY, R.D., VORISEK, P., VAN STRIEN, A., GMELIG MEYLING, A.W., JIGUET, F., FORNASARI, L., REIF, J., CHYLARECKI, P. and BURFIELD, I.J., 2007. Population trends of widespread woodland birds in Europe. *Ibis,* **149**, pp. 78-97.

GSCHWANTNER, T., SCHADAUER, K., VIDAL, C., LANZ, A., TOMPPO, E., DI COSMO, L., ROBERT, N., ENGLERT DUURSMA, D. and LAWRENCE, M., 2009. Common tree definitions for national forest inventories in Europe.

GUSSET, M. and DICK, G., 2011. The global reach of zoos and aquariums in visitor numbers and conservation expenditures. *Zoo biology,* **30**(5), pp. 566-569.

HAIG, S.M., BALLOU, J.D. and DERRICKSON, S.R., 1990. Management options for preserving genetic diversity: reintroduction of Guam rails to the wild. *Conservation Biology,* **4**(3), pp. 290-300.

HAWLEY, D.M., JENNELLE, C.S., SYDENSTRICKER, K.V. and DHONDT, A.A., 2007. Pathogen resistance and immunocompetence covary with social status in house finches (Carpodacus mexicanus). *Functional Ecology,* **21**(3), pp. 520-527.

HEWSON, C.M. and NOBLE, D.G., 2009. Population trends of breeding birds in British woodlands over a 32-year period: relationships with food, habitat use and migratory behaviour. *Ibis,* **151**(3), pp. 464-486.

HINSLEY, S.A. and BELLAMY, P.E., 2000. The influence of hedge structure, management and landscape context on the value of hedgerows to birds: a review. *Journal of environmental management,* **60**(1), pp. 33-49.

HOLLAND, J.M., HUTCHISON, M., SMITH, B. and AEBISCHER, N.J., 2006. A review of invertebrates and seed-bearing plants as food for farmland birds in Europe. *Annals of Applied Biology,* **148**(1), pp. 49-71.

HOSEY, G., MELFI, V. and PANKHURST, S., 2013. *Zoo animals: behaviour, management, and welfare.* Oxford University Press.

HOUSTON, D.D., AZEEM, S., LUNDY, C.W., SATO, Y., GUO, B., BLANCHONG, J.A., GAUGER, P.C., MARKS, D.R., YOON, K. and ADELMAN, J.S., 2017. Evaluating the role of wild songbirds or rodents in spreading avian influenza virus across an agricultural landscape. *PeerJ,* **5**, pp. e4060.

JONES, C.G., HECK, W., LEWIS, R.E., MUNGROO, Y., SLADE, G. and CADE, T., 1995. The restoration of the Mauritius kestrel Falco punctatus population. *Ibis,* **137**, pp. S173-S180.

KETTEL, E.F., GENTLE, L.K., QUINN, J.L. and YARNELL, R.W., 2018. The breeding performance of raptors in urban landscapes: a review and meta-analysis. *Journal of Ornithology,* **159**(1), pp. 1-18.

KIRKWOOD, J.T., 1998. Population density and infectious disease at bird tables [Correspondence]. *Veterinary Record (United Kingdom),* .

KÖVÉR, L., LENGYEL, S., TAKENAKA, M., KIRCHMEIR, A., UHL, F., MILLER, R. and SCHWAB, C., 2019. Why do zoos attract crows? A comparative study from Europe and Asia. *Ecology and Evolution,* **9**(24), pp. 14465-14475.

KÜHN, I., BRANDL, R. and KLOTZ, S., 2004. The flora of German cities is naturally species rich. *Evolutionary Ecology Research,* **6**(5), pp. 749-764.

LACHISH, S., BONSALL, M.B., LAWSON, B., CUNNINGHAM, A.A. and SHELDON, B.C., 2012a. Individual and population-level impacts of an emerging poxvirus disease in a wild population of great tits. *PLoS One,* **7**(11), pp. e48545.

LACHISH, S., BONSALL, M.B., LAWSON, B., CUNNINGHAM, A.A. and SHELDON, B.C., 2012b. Individual and population-level impacts of an emerging poxvirus disease in a wild population of great tits. *PLoS One,* **7**(11), pp. e48545.

LACK, D., 1954. The natural regulation of animal numbers. *The Natural Regulation of Animal Numbers.,* .

LAIOLO, P., SERRANO, D., TELLA, J.L., CARRETE, M., LOPEZ, G. and NAVARRO, C., 2007. Distress calls reflect poxvirus infection in lesser short-toed lark Calandrella rufescens. *Behavioral Ecology,* **18**(3), pp. 507-512.

LAWSON, B., ROBINSON, R.A., TOMS, M.P., RISELY, K., MACDONALD, S. and CUNNINGHAM, A.A., 2018. Health hazards to wild birds and risk factors associated with anthropogenic food provisioning. *Philosophical Transactions of the Royal Society B: Biological Sciences,* **373**(1745), pp. 20170091.

LAZARUS, J. and SYMONDS, M., 1992. Contrasting effects of protective and obstructive cover on avian vigilance. *Animal Behaviour,* .

LEPCZYK, C.A., ARONSON, M.F., EVANS, K.L., GODDARD, M.A., LERMAN, S.B. and MACIVOR, J.S., 2017. Biodiversity in the city: fundamental questions for understanding the ecology of urban green spaces for biodiversity conservation. *Bioscience,* **67**(9), pp. 799-807.

LOWRY, H., LILL, A. and WONG, B.B., 2013. Behavioural responses of wildlife to urban environments. *Biological reviews,* **88**(3), pp. 537-549.

MACLEOD, R., BARNETT, P., CLARK, J. and CRESSWELL, W., 2006. Mass-dependent predation risk as a mechanism for house sparrow declines? *Biology Letters,* **2**(1), pp. 43-46.

MARC J. MAZEROLLE, AICcmodavg: Model selection and multimodel inference based on (Q)AIC(c). .

MARTIN, T.E., 1987. Food as a limit on breeding birds: a life-history perspective. *Annual Review of Ecology and Systematics,* **18**(1), pp. 453-487.

MARZLUFF, J.M., 2001. Worldwide urbanization and its effects on birds. *Avian ecology and conservation in an urbanizing world.* Springer, pp. 19-47.

MCBLAIN, M., JONES, K.A. and SHANNON, G., 2020. Sleeping Eurasian oystercatchers adjust their vigilance in response to the behaviour of neighbours, human disturbance and environmental conditions. *Journal of zoology,* **312**(2), pp. 75-84.

MEILLÈRE, A., BRISCHOUX, F. and ANGELIER, F., 2015. Impact of chronic noise exposure on antipredator behavior: an experiment in breeding house sparrows. *Behavioral Ecology,* **26**(2), pp. 569-577.

MILES, W.T., MAVOR, R., RIDDIFORD, N.J., HARVEY, P.V., RIDDINGTON, R., SHAW, D.N., PARNABY, D. and REID, J.M., 2015. Decline in an Atlantic puffin population: evaluation of magnitude and mechanisms. *Plos one,* **10**(7), pp. e0131527.

MILLS, G.S., DUNNING JR, J.B. and BATES, J.M., 1989. Effects of urbanization on breeding bird community structure in southwestern desert habitats. *The Condor,* **91**(2), pp. 416-428.

MOYERS, S.C., ADELMAN, J.S., FARINE, D.R., THOMASON, C.A. and HAWLEY, D.M., 2018. Feeder density enhances house finch disease transmission in experimental epidemics. *Philosophical Transactions of the Royal Society B: Biological Sciences,* **373**(1745), pp. 20170090.

NEWTON, I., 2004a. The recent declines of farmland bird populations in Britain: an appraisal of causal factors and conservation actions. *Ibis,* **146**(4), pp. 579-600.

NEWTON, I., 2004b. The recent declines of farmland bird populations in Britain: an appraisal of causal factors and conservation actions. *Ibis,* **146**(4), pp. 579-600.

NEWTON, I., 1998. *Population limitation in birds.* Academic press.

NICHOLS, W.F., KILLINGBECK, K.T. and AUGUST, P.V., 1998. The influence of geomorphological heterogeneity on biodiversity II. A landscape perspective. *Conservation Biology,* **12**(2), pp. 371-379.

ORTEGA-ÁLVAREZ, R. and MACGREGOR-FORS, I., 2009. Living in the big city: Effects of urban land-use on bird community structure, diversity, and composition. *Landscape and Urban Planning,* **90**(3-4), pp. 189-195.

PATRICK, P.G. and CAPLOW, S., 2018. Identifying the foci of mission statements of the zoo and aquarium community. *Museum Management and Curatorship,* **33**(2), pp. 120-135.

PAUTASSO, M., BÖHNING-GAESE, K., CLERGEAU, P., CUETO, V.R., DINETTI, M., FERNÁNDEZ-JURICIC, E., KAISANLAHTI-JOKIMÄKI, M., JOKIMÄKI, J., MCKINNEY, M.L. and SODHI, N.S., 2011. Global macroecology of bird assemblages in urbanized and semi-natural ecosystems. *Global Ecology and Biogeography,* **20**(3), pp. 426-436.

PEACH, W.J., MALLORD, J.W., OCKENDON, N., ORSMAN, C.J. and HAINES, W.G., 2018. Depleted suburban house sparrow Passer domesticus population not limited by food availability. *Urban ecosystems,* **21**(6), pp. 1053-1065.

PEACH, W.J., MALLORD, J.W., OCKENDON, N., ORSMAN, C.J. and HAINES, W.G., 2015. Invertebrate prey availability limits reproductive success but not breeding population size in suburban House Sparrows Passer domesticus. *Ibis,* **157**(3), pp. 601-613.

PERKINS, A., RATCLIFFE, N., SUDDABY, D., RIBBANDS, B., SMITH, C., ELLIS, P., MEEK, E. and BOLTON, M., 2018. Combined bottom-up and top-down pressures drive catastrophic population declines of Arctic skuas in Scotland. *Journal of Animal Ecology,* **87**(6), pp. 1573-1586.

PETROV, B.N., 1973. Second International Symposium on Information Theory, 1973, Akad.Kiadó.

PHILIPPA, J., BAAS, C., BEYER, W., BESTEBROER, T., FOUCHIER, R., SMITH, D., SCHAFTENAAR, W. and OSTERHAUS, A., 2007. Vaccination against highly pathogenic avian influenza H5N1 virus in zoos using an adjuvanted inactivated H5N2 vaccine. *Vaccine,* **25**(19), pp. 3800-3808.

PIPER, W.H., 1997. Social dominance in birds. *Current ornithology.* Springer, pp. 125-187.

PLUMMER, K.E., RISELY, K., TOMS, M.P. and SIRIWARDENA, G.M., 2019. The composition of British bird communities is associated with long-term garden bird feeding. *Nature communications,* **10**(1), pp. 1-8.

RADFORD, J.Q., BENNETT, A.F. and CHEERS, G.J., 2005. Landscape-level thresholds of habitat cover for woodland-dependent birds. *Biological Conservation,* **124**(3), pp. 317-337.

RALLS, K. and BALLOU, J.D., 2004. Genetic status and management of California condors. *The Condor,* **106**(2), pp. 215-228.

ROBINSON, R.A., 2001. Feeding ecology of skylarks Alauda arvensis in winter–a possible mechanism for population decline. *The ecology and conservation of skylarks Alauda arvensis,* , pp. 129-138.

ROSE, P.E., BRERETON, J.E., ROWDEN, L.J., DE FIGUEIREDO, R.L. and RILEY, L.M., 2019. What's new from the zoo? An analysis of ten years of zoo-themed research output. *Palgrave Communications,* **5**(1), pp. 1-10.

SALISBURY, A., AL-BEIDH, S., ARMITAGE, J., BIRD, S., BOSTOCK, H., PLATONI, A., TATCHELL, M., THOMPSON, K. and PERRY, J., 2020. Enhancing gardens as habitats for soil-surface-active invertebrates: should we plant native or exotic species? *Biodiversity and Conservation,* **29**(1), pp. 129-151.

SALISBURY, A., AL-BEIDH, S., ARMITAGE, J., BIRD, S., BOSTOCK, H., PLATONI, A., TATCHELL, M., THOMPSON, K. and PERRY, J., 2017. Enhancing gardens as habitats for plant-associated invertebrates: should we plant native or exotic species? *Biodiversity and Conservation,* **26**(11), pp. 2657-2673.

SALISBURY, A., ARMITAGE, J., BOSTOCK, H., PERRY, J., TATCHELL, M. and THOMPSON, K., 2015. EDITOR'S CHOICE: Enhancing gardens as habitats for flowerâ€• visiting aerial insects (pollinators): should we plant native or exotic species? *Journal of Applied Ecology,* **52**(5), pp. 1156-1164.

SEDDON, P.J., ARMSTRONG, D.P. and MALONEY, R.F., 2007. Developing the science of reintroduction biology. *Conservation Biology,* **21**(2), pp. 303-312.

SIRIWARDENA, G.M., BAILLIE, S.R. and WILSON, J.D., 1998. Variation in the survival rates of some British passerines with respect to their population trends on farmland. *Bird Study,* **45**(3), pp. 276-292.

SMITH, R.M., GASTON, K.J., WARREN, P.H. and THOMPSON, K., 2006. Urban domestic gardens (VIII): environmental correlates of invertebrate abundance. *Biodiversity & Conservation,* **15**(8), pp. 2515-2545.

SPANOWICZ, A.G. and JAEGER, J.A., 2019. Measuring landscape connectivity: On the importance of within-patch connectivity. *Landscape Ecology,* **34**(10), pp. 2261-2278.

STRANDIN, T., BABAYAN, S.A. and FORBES, K.M., 2018. Reviewing the effects of food provisioning on wildlife immunity. *Philosophical Transactions of the Royal Society B: Biological Sciences,* **373**(1745), pp. 20170088.

SUMMERS-SMITH, J.D., 1963. The house sparrow. W. Collins & sons, London. *Kamil Kryński,* .

TARABON, S., BERGÈS, L., DUTOIT, T. and ISSELIN-NONDEDEU, F., 2019. Maximizing habitat connectivity in the mitigation hierarchy. A case study on three terrestrial mammals in an urban environment. *Journal of environmental management,* **243**, pp. 340-349.

TEAM, R.C., 2019. R: A language and environment for statistical computing.

THRELFALL, C.G., WILLIAMS, N.S., HAHS, A.K. and LIVESLEY, S.J., 2016. Approaches to urban vegetation management and the impacts on urban bird and bat assemblages. *Landscape and Urban Planning,* **153**, pp. 28-39.

TOBIAS, J., 1997. Food availability as a determinant of pairing behaviour in the European robin. *Journal of Animal Ecology,* , pp. 629-639.

TRATALOS, J., FULLER, R.A., EVANS, K.L., DAVIES, R.G., NEWSON, S.E., GREENWOOD, J.J. and GASTON, K.J., 2007. Bird densities are associated with household densities. *Global Change Biology,* **13**(8), pp. 1685-1695.

TRZCINSKI, M.K., FAHRIG, L. and MERRIAM, G., 1999. Independent effects of forest cover and fragmentation on the distribution of forest breeding birds. *Ecological Applications,* **9**(2), pp. 586-593.

VAN RIPER III, C. and FORRESTER, D.J., 2007. Avian pox. *Infectious diseases of wild birds,* , pp. 131-176.

VAN RIPER III, C., VAN RIPER, S.G. and HANSEN, W.R., 2002. Epizootiology and effect of avian pox on Hawaiian forest birds. *The Auk,* **119**(4), pp. 929-942.

VICKERY, J.A., BRADBURY, R.B., HENDERSON, I.G., EATON, M.A. and GRICE, P.V., 2004. The role of agri-environment schemes and farm management practices in reversing the decline of farmland birds in England. *Biological Conservation,* **119**(1), pp. 19-39.

VINCENT, K.E., 2005a. Investigating the causes of the decline of the urban house sparrow Passer domesticus population in Britain.

VINCENT, K.E., 2005b. Investigating the causes of the decline of the urban house sparrow Passer domesticus population in Britain.

WANG, Y., SUN, Y., LIU, Z., SARMA, S.E., BRONSTEIN, M.M. and SOLOMON, J.M., 2019. Dynamic graph cnn for learning on point clouds. *Acm Transactions On Graphics (tog),* **38**(5), pp. 1-12.

WAZA 2019, , Commiting to Conservation. Available: https://www.waza.org/wp-content/uploads/2019/03/WAZA-Conservation-Strategy-2015_Portrait.pdf.

WIEDENFELD, D.A., FESSL, B., KLEINDORFER, S. and VALAREZO, J.C., 2007. Distribution of the introduced parasitic fly Philornis downsi (Diptera, Muscidae) in the Galápagos Islands. *Pacific Conservation Biology,* **13**(1), pp. 14-19.

WILD BIRD POPULATIONS IN THE, U K and 1970 TO 2017, *Wild Bird Populations in the UK, 1970 to 2017.*

WINTLE, B.A., KUJALA, H., WHITEHEAD, A., CAMERON, A., VELOZ, S., KUKKALA, A., MOILANEN, A., GORDON, A., LENTINI, P.E. and CADENHEAD, N.C., 2019. Global synthesis of conservation studies reveals the importance of small habitat patches for biodiversity. *Proceedings of the National Academy of Sciences,* **116**(3), pp. 909-914.

XIE, S., LU, F., CAO, L., ZHOU, W. and OUYANG, Z., 2016. Multi-scale factors influencing the characteristics of avian communities in urban parks across Beijing during the breeding season. *Scientific reports,* **6**(1), pp. 1-9.

YOUNG, C., FREY, D., MORETTI, M. and BAUER, N., 2019. Research Note: Garden-owner reported habitat heterogeneity predicts plant species richness in urban gardens. *Landscape and Urban Planning,* **185**, pp. 222-227.

ZIMMERMANN, A., HATCHWELL, M., DICKIE, L.A. and WEST, C., 2007. *Zoos in the 21st century: catalysts for conservation?* Cambridge University Press Cambridge, UK.