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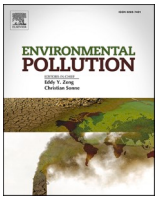

Interspecific differences in microplastic accumulation and polymer profiles from regurgitated pellets of coexisting raptors

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ABSTRACT

Microplastics have emerged as pervasive environmental contaminants, raising concerns over their impacts on wildlife. Birds of prey occupy high trophic positions and hence are particularly vulnerable to accumulating high contaminant levels due to potential biomagnification. We used regurgitated pellets as a non-invasive proxy for dietary microplastic exposure in two apex predators that share overlapping habitats in the United Kingdom (UK): barn owls (*Tyto alba*) and common kestrels (*Falco tinnunculus*). We collected regurgitated pellets (n = 24 per species), which were chemically digested and microplastics were enumerated and identified using Fourier-transform infrared spectroscopy (FTIR), and sorted into five size classes ranging from 25 to 200+ μm. Overall, 92% of kestrel pellets and 83% of barn owl pellets contained microplastics. Kestrel pellets had 18.46 microplastic particles g^{-1}, which was significantly higher than in barn owl pellets, which had 0.42 particles g^{-1} ($p = 0.03$). Of a possible 20 polymers, 13 were detected with significant variations in occurrence between the two species. Overall, the dominant polymers were polypropylene (8.2%), polyurethane (9.2%), polyethylene (36.8%), ethylene vinyl alcohol (3.1%) and ethylene vinyl acetate (37.3%). Additionally, there were significant differences in number of microplastic particles between repeat sampling of the same nests. The largest number of particles were found in the 25-50 μm range, with 25 μm being the detection limit of our FTIR instrument. These findings demonstrate species-specific microplastic exposure, which may be driven by the variable microplastic load of different prey. To our knowledge, this study is the first to investigate microplastic ingestion by birds of prey in the UK.

1. Introduction

Plastics entered widespread commercial use in the 1940s (Thompson et al., 2009). Since then, plastic production for commercial and industrial use has risen exponentially and, from 1950 to 2017, it is estimated that humans produced 9.2 billion metric tons of virgin plastics globally, of which 7 billion tons ended up as waste (Geyer et al., 2017; Geyer, 2020). This plastic enters the environment and is fragmented and broken down into microplastics (plastics ≤5 mm), which are then able to enter food chains via inhalation or ingestion by organisms (De Pascalis et al., 2022; Horton and Dixon, 2018; Huerta-Lwanga et al., 2017; Thompson et al., 2024; Wang et al., 2021). Microplastics enter the environment as either primary or secondary microplastics. Primary microplastics are plastic particles that are produced as virgin particles ≤5 mm in size, whereas secondary microplastics are particles produced by the breakdown of larger non-virgin macroplastics (>5 mm) (Barnes et al., 2009; Thompson et al., 2009).

By the 1960s, the first evidence of birds ingesting plastic was reported (Kenyon and Kridler, 1969). Since then, numerous studies have documented ingestion of both macroplastics and microplastics (Elias and Corbin, 2024; Mansfield et al., 2024; Nicastro et al., 2018; Susanti et al., 2020; Wang et al., 2021), and increasing research into the incidence and effects of plastic consumption by birds has raised concerns about the potential animal welfare and conservation implications (Roman et al., 2021; Charlton-Howard et al., 2023). Microplastics are classed as an "emerging pollutant" and have been detected in a large

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number of avian species inhabiting both terrestrial and marine environments (Elias and Corbin, 2024; Flemming et al., 2022; Nessi et al., 2022; Wang et al., 2021). While macroplastic ingestion has been associated with physical harm in some cases, the prevalence, exposure pathways and potential implications of microplastic ingestion by birds remain far less clear. There are potential indications that microplastic ingestion by birds can cause harmful effects, such as the disruption of physicochemical processes (Luo et al., 2022; Schrank et al., 2019; Zhang et al., 2025), introduction of pathogens and other pollutants associated with microplastics (Borges-Ramírez et al., 2021), and physiological damage caused by the physical presence of microplastics in organs (Hamed et al., 2021; Rivers-Auty et al., 2023). This uncertainty is particularly pronounced for terrestrial birds of prey, as most research to date has focused on marine and freshwater species (Elias and Corbin, 2024; Mansfield et al., 2024). Studies of other contaminants, such as trace elements and pesticides, show that raptors can accumulate high contaminant levels due to their high trophic positions (Burger, 1995; Leblanc, 1995; Walker, 1990); however, whether similar processes apply to microplastics or their associated chemicals is still largely unknown.

This study focused on microplastic ingestion by two species of terrestrial raptors, barn owls (*Tyto alba*) and kestrels (*Falco tinnunculus*) in southern England, United Kingdom (UK), via an analysis of regurgitated pellets. Pellets offer several advantages: they can be collected without disturbing the birds, they allow repeated sampling from the same nests, and they enable larger datasets that capture within-nest variation (Seyfe et al., 2025; Tınaz et al., 2025). Moreover, multiple pellets can be collected from the same birds, facilitating larger data sets and the ability to account for variation within nests. However, pellets represent egested material and hence cannot be used to quantify plastic remaining within the body (Provencher et al., 2019). The objective of this study was to determine the current levels of microplastic ingestion by these two species and examine interspecific differences. Given that both species frequently occupy the same habitat and have similar feeding ecologies, we hypothesise that they will show similar levels of microplastic contamination. Indeed, the diets of these species overlap to some extent in England, with both species primarily consuming small mammals; although both species also supplement their diet with small birds, lizards, worms and insects, with kestrels showing a higher propensity to do so (Jones et al., 2024; Marti et al., 2024; Taylor, 2003). On average, the short-tailed field vole (*Microtus agrestis*) makes up the largest part of the diet for both species in the UK, including ~60% of the barn owl diet (Bond et al., 2005) and a large percentage of the kestrel diet (Village, 1990; Jones et al., 2024). However, relative contributions of different prey items to the diets of both species in the UK are known to vary geographically based on availability (Bunn et al., 1992; Village, 1990). Additionally, both species often share the same habitat, which consists of open grassland and farmland, often nesting in human-made structures (BTO, 2023; 2009; Orta et al., 2020a; Taylor, 2003). While both species are resident breeders in the UK, they also occur widely across Europe, Asia and Africa, where their diets differ from populations in the UK (Van Zyl, 1994; Taylor and Olsen, 2016).

Previous research on birds has quantified microplastic ingestion either through necropsy of the digestive tracts of dead individuals (Carlin et al., 2020; Wayman et al., 2024) or through analysis of regurgitated pellets (Álvarez et al., 2018; Borges-Ramírez et al., 2021; Nessi et al., 2022; Winkler et al., 2020). Most pellet-based studies of birds of prey have relied primarily on visual identification, with Raman or Fourier-transform infrared spectroscopy (FT-IR) used only to confirm a subset of visually identified particles. However, visual identification alone has an error rate of 20–70%, particularly for smaller particles (Löder and Gerdts, 2015), making it unsuitable for accurate quantification of fine-fraction microplastics. FT-IR spectroscopy, while not error-free, enables reliable polymer-level identification and avoids the size-dependent misclassification inherent to visual sorting. Determining both the polymer types and size distributions of ingested microplastics is essential for inferring exposure pathways because polymers differ in their applications, additives, and environmental behaviour (Geyer et al., 2017), and particle size influences the likelihood of ingestion, retention, and translocation within the body.

Despite growing interest in avian microplastic exposure, major knowledge gaps remain regarding interspecific variation in ingestion risk and the ecological traits that drive it. Early evidence suggests that foraging strategy (Mylius et al., 2023) and diet composition (Zeng et al., 2025) may influence exposure, yet empirical data remains sparse, particularly for raptors. Very few studies have examined microplastic ingestion in UK birds, and none have focused on birds of prey (Mansfield et al., 2024). This limits our ability to assess whether apex predators experience elevated microplastic burdens through trophic transfer and whether pellet analysis can serve as a reliable non-invasive indicator of environmental contamination. By applying μFTIR spectroscopy to regurgitated pellets from two sympatric raptor species occupying overlapping landscapes, this study aims to improve understanding of how feeding ecology and habitat use shape microplastic exposure at higher trophic levels.

2. Materials and methods

2.1. Study site and sample collection

Fieldwork for this study was undertaken in August 2024 in Hampshire, UK. Regurgitated pellets were collected from either inside barn owl and kestrel nest boxes, or from the ground directly underneath, once the breeding season had finished. This sampling protocol was designed to limit disturbance, as the nests used in our study are part of a larger conservation project. This, however, means we are unable to observe seasonal variation in microplastics in our samples. As we avoided sampling while young were still being tended by the parents it is impossible to determine what sex or age the individuals that produced the pellets were. All pellets were surface rinsed to ensure that microplastics in our samples were coming from within the pellets. Sampled nests were separated into eight pairs with one nest for each species in the pair, for a total of 16 nests. These paired nests were >10 km apart from other pairs and <1 km apart from the other nest in the pair, to avoid pseudoreplication. Pellets were collected by hand from each nest (n = 3 per nest) and wrapped in aluminium foil to prevent contamination and then placed in sealed paper envelopes for transport to the lab. Samples were stored at −20 °C prior to processing to prevent the growth of mould and the degradation of the organic material that makes up the bulk of pellet material.

2.2. Microplastic extraction

Prior to chemical digestion, each pellet was placed in a clean beaker covered with aluminium foil which was placed in an oven to dry for 48 h at 40^0 C. After drying, the dry weight (dw) of each pellet was recorded. Next, 10% potassium hydroxide (KOH) was added to the beaker, with 40 ml added to beakers with kestrel pellets and 100 ml to those with barn owl pellets. Each beaker was placed in a shaking incubator (40 °C, 40 rpm) for 48 h. Care was taken to ensure that samples were never subjected to temperatures >40 °C, as it has been shown that exposure to high temperatures causes degradation of plastic particles (Munno et al., 2018). Karami et al. (2017) demonstrated that incubating samples at 40 °C for 48 h in 10% KOH produced high levels of recovery for all polymer types and proved to be the method least destructive to microplastic particles. KOH digests organic material, such as fur and feathers in the samples, without degrading microplastic particles. Following KOH digestion, samples were vacuum filtered through a series of 1 mm and 5 μm filters to remove KOH from the sample matrix. All the sample material left after the KOH digestion step was collected on the 5 μm steel filter. Following digestion with KOH, a Fenton's reaction was produced by adding 30 ml Fenton's reagent (10 ml 0.05M aqueous $FeSO_4$ and 20

ml 30% H_2O_2) to digest any remaining organic material in the sample. Sample beakers were kept in a foil lined water bath to prevent temperatures rising above 40 °C. After an hour another 20 ml of H_2O_2 was added, followed by a final 20 ml of H_2O_2 after another hour has passed. The Fenton's reaction was allowed to run for 48 h after which samples were acidified with H_2SO_4. Prior to the start of the pellet digestions a pilot study was conducted where different digestion methods were tested. The combination of KOH digestion followed by a Fenton's reaction was found to produce better results than either method on their own, with the KOH digestion making the remaining organic material available to be digested by the Fenton's reaction. The sample was then vacuum filtered through the same 5 μm steel filter in preparation for density separation. Density separation was performed using $ZnCl_2$ at 1.6-1.7 g/mL, which allowed us to further separate plastic particles from surrounding material due to the lower density of plastic polymers. The resulting sample was then deposited onto a 5 μm silver membrane filter using a glass pipette. A silicone washer with internal diameter of 14 mm was used to define the deposition area.

2.3. FT-IR analysis and characterisation of microplastic particles

Microplastics analysis to determine total number of microplastic particles, polymer type and particle size, was conducted using micro-FT-IR (μFT-IR, a PerkinElmer Spotlight 400). Microplastic particles on the silver filter were identified and quantified with a pixel size of 25 μm (which set our lowest detection limit). The instrument was set to collect spectra between 4000 and 700 cm^{-1} (Table S1). Once the scan by the FT-IR is complete the spectral data produced from the scan is analysed by Purency Microplastic Finder (Hufnagl et al., 2022). Purency performs automated spectral matching using machine learning, allowing us to eliminate operator bias. The software is able to identify polymer types, quantifying particles in terms of polymer type and dimensions (Hufnagl et al., 2022).

2.4. Quantified polymers

We quantified the following polymers; acrylonitrile butadiene styrene (ABS), cellulose acetate (CA), ethylene vinyl acetate (EVAc), ethylene vinyl alcohol (EVOH), polyamide (PA), polyacrylonitrile (PAN), polycarbonate (PC), poly-ether-ether-ketone (PEEK), polyethylene (PE), polyoxymethylene (POM), polyphenylene sulfone (PPSU), polysulfone (PSU), polyethylene terephthalate (PET), polylactic acid (PLA), poly(methyl methacrylate) (PMMA), polypropylene (PP), polystyrene (PS), polyurethane (PU), polyvinyl chloride (PVC) and silicone. These were selected based on their widespread use and documented occurrence in the environment (Adediran et al., 2026, 2024; Rendell-Bhatti et al., 2023)

2.5. Cross contamination prevention measures

Measures were taken to prevent contamination at every stage of sample processing. Between processing steps, and during oven drying, beakers containing the samples were covered with clean aluminium foil. Each reagent used in sample processing was filtered through glass fibre filters with 0.7 μm pore size (Whatman GF/F), and stored in glass bottles with polytetrafluoroethylene (PTFE) caps and pouring rings. De-ionised (DI) water used to dilute reagents and agitate the sample material was GF/F filtered prior to use. Before processing began, all pellets were rinsed with GF/F filtered water to remove any atmospheric microplastic deposition on the outer surface of the pellets. All glassware and metal tools used in sample processing were triple washed with GF/F filtered DI water. Equipment and glassware were exclusively washed with natural fibre scouring brushes. Tools used to agitate or handle samples were made of either glass or stainless steel. When the use of plastic was required, the equipment was made of uncommon polymers that are not present in Purency's spectral library. All foil was washed with GF/F filtered water prior to use. Once samples had been processed and deposited onto a silver filter membrane, the slide hosting the silver filter was stored in paper slide holders and placed in cardboard boxes to await analysis. Boxes were stored in a specialised plastic free clean room.

Steps requiring the exposure of samples were conducted in a Microbiological Safety Cabinet (MSC) equipped with a high-efficiency particulate air (HEPA) filter, which are the most efficient clean air systems at eliminating atmospheric microplastic contamination and provide the cleanest working environment (Wesch et al., 2017). The HEPA filter is capable of removing 99.999% of particles >0.3 μm. The MSC was wiped down before and after lab work using GF/F filtered ethanol. Lab users wore 100% cotton lab coats at all times to prevent contamination from synthetic fibres and effort was taken to wear clothing of 100% natural fibres when conducting lab and field work.

2.6. Spike recovery

The recovery rate of the sample processing methodology was tested by performing spike recovery tests. Spike recovery tests were conducted on extra pellets obtained from both species as well as a corresponding series of blanks. Particles of a known polymer (PVC) and size (90-150 μm), that are brightly coloured, were counted then added to the sample beakers. After the spiked samples were processed and analysed using FT-IR, the number of remaining spike particles in each sample was counted. The spike recovery rate was 72% for blanks, 68% for kestrel pellets and 67% for barn owl pellets. Similar recovery rates have been observed in other studies that estimated spike recovery (Adediran et al., 2026, 2024; Rendell-Bhatti et al., 2023). However, the absence of appropriate standards for different polymer types across various size ranges limits the effectiveness of spike recovery, rendering positive controls merely indicative.

2.7. Blank correction

All data used in the study were blank corrected. Samples were processed in numbered batches and each batch had a corresponding blank (s). Procedural blanks comprised empty beakers processed through the same sample processing steps as sample beakers, thus capturing only microplastic contamination present during the processing steps. Blank correction was performed by sorting particles from blanks into the size classes used in analysis (25-50 μm, 50-100 μm, 100-150 μm, 150-200 μm and 200+ μm). The average number of particles across 18 blanks was quantified for each size class (25-50 μm: 18.55, 50-100 μm: 9.16, 100-150 μm: 4.22, 150-200 μm: 1.11, 200+ μm: 1.66) but the total number of particles varied significantly over batches and individual blanks (S.2). We subtracted the number of microplastics obtained from the blanks from the samples that corresponded to that blank, similar to the approach taken by Adediran et al. (2026). Number of microplastics were subtracted for each polymer type, where a polymer was absent in a corresponding blank no value was subtracted. When examining polymer specific microplastic profiles in our blanks it was determined that PA be excluded from the results of our study due to high levels of PA in many of our blanks. Field blanks were not included in this study.

2.8. Statistical analysis

The number of microplastics were adjusted according to the mass of pellets, since those of kestrels were far smaller than of barn owls (ranges: 0.4-1.6 g and 5-20 g, respectively). Results were therefore expressed as the number per g of pellet (particles g^{-1}). Interspecific differences in the distributions of particles g^{-1} were investigated by conducting a quantile regression to generate species-specific medians and corresponding confidence intervals (CIs). Initial data exploration indicated the data was non-normally distributed with a pronounced right skew. We trialled log-normal and gamma as candidate distributions. The model assumptions were tested using DHARMa and Akaike Information Criterion

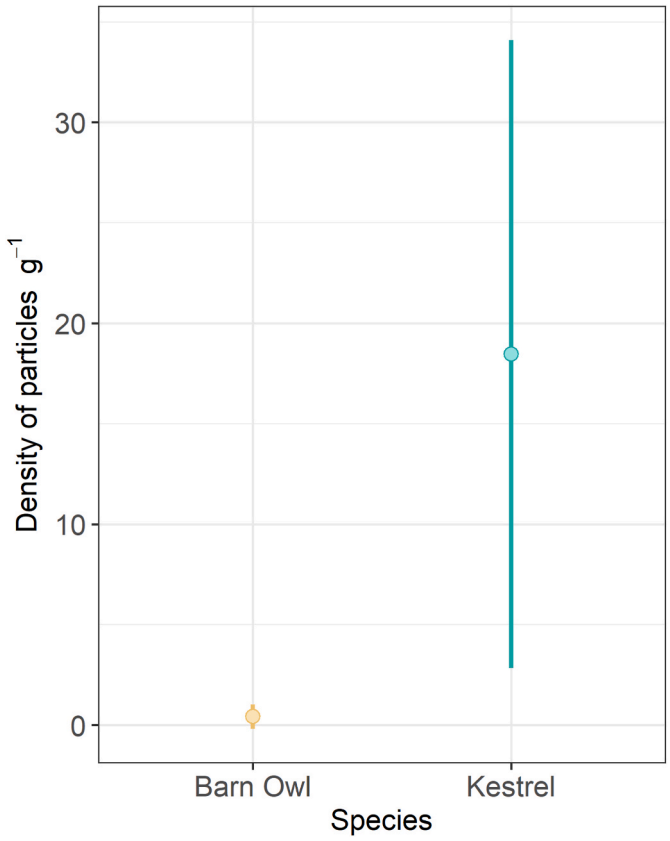

Fig. 1. Median microplastic concentrations (particles g^{-1} dry weight) in barn owl (*Tyto alba*) and kestrel (*Falco tinnunculus*) pellets collected in Hampshire, southern England.

(AIC) was used to select the final model. The model we selected contained nest ID as a random effect (to account for multiple pellets being sampled from the same nest). To test for differences in polymer types between the two species we used generalised linear mixed effect models (GLMMs). The model with the best fit was a negative binomial regression model with log link. The model contained the nest ID as a random effect and was offset by the log of the pellet masses. The model assumptions were tested using DHARMa and AIC values were used to select the final model. Means and standard deviations were reported from the selected model for each polymer for both species. A pairwise comparison was then conducted on the means to determine significance.

We report the median total number of microplastic particles g^{-1} over all polymers, and the mean microplastic particles g^{-1} for the polymer specific analysis. The median was selected as a measure of average for the all polymers analysis as it is robust to outliers in the data set, which initial exploration indicated was skewed. The mean was selected for as the measure of average for the polymer specific analysis due to the high number of non-detects, with many polymers only occurring in a small number of pellets. All statistical analyses were performed with R version 4.4.3 (R Core Team, 2025), using the packages lme4 (Bates et al., 2015), glmmTMB (Brooks et al., 2017; McGillycuddy et al., 2025), quantreg (Koenker, 2025), lqmm (Geraci, 2022; 2014), MASS (Ripley and Venables, 2025), emmeans (Lenth, 2025) and DHARMa (Hartig, 2024). Figures were generated using the R packages ggplot2 (Wickham et al., 2025), ssdtools (Thorley et al., 2025), ssd4mosaic (Siberchicot et al., 2025), ggstance (Henry et al., 2024) and interactions (Long, 2024).

3. Results

3.1. Microplastic occurrence and concentrations in pellets

All nests surveyed had at least one pellet that contained detectable microplastics. Overall, 83% of barn owl pellets (n = 20/24) and 92% of kestrel pellets (n = 22/24) contained microplastic particles. Kestrel pellets had a significantly higher (quantile regression, p = 0.03) median density of microplastic particles g^{-1} than barn owl pellets (18.46 particles g^{-1} and 0.42 particles g^{-1}, respectively) (Fig. 1). The lowest density of particles detected in a single sample was 0 particles g^{-1}, while the highest was 97.82 particles g^{-1}. Based on the average mass of pellets for each species and median microplastic particles g^{-1}, the average barn owl pellet would contain 3.82 total microplastic particles while the average kestrel pellet would contain 16.98 total microplastic particles.

3.2. Polymer distributions

Of the twenty polymer types investigated, thirteen were positively identified in the samples. (Fig. 2). All the thirteen polymers occurred in the pellets of both species except PAN, which occurred only in kestrel

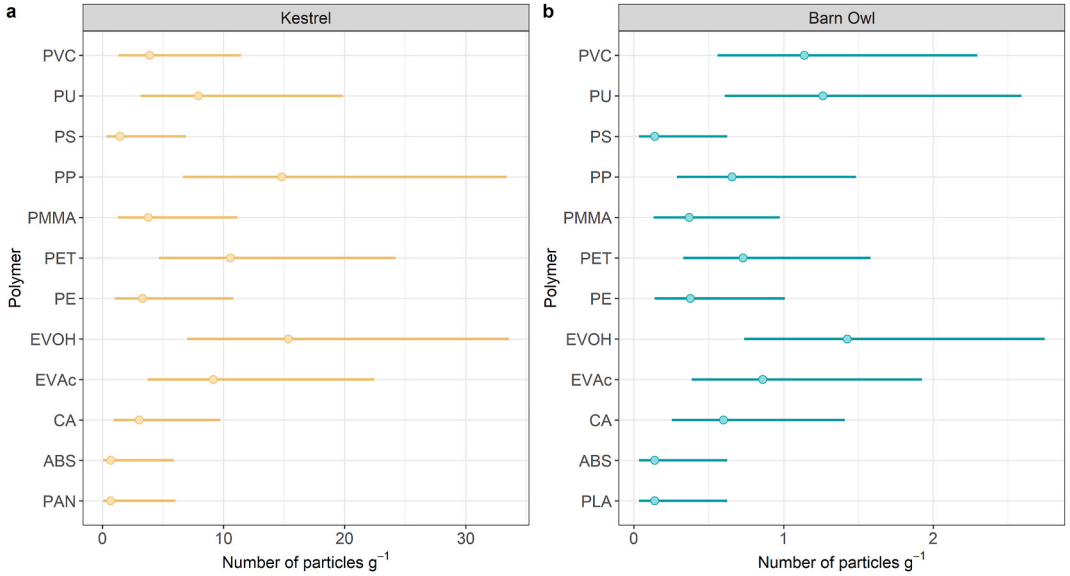

Fig. 2. Mean microplastic concentrations (particles g^{-1} dry weight) by polymer type for kestrel and barn owl pellets from Hampshire, southern England.

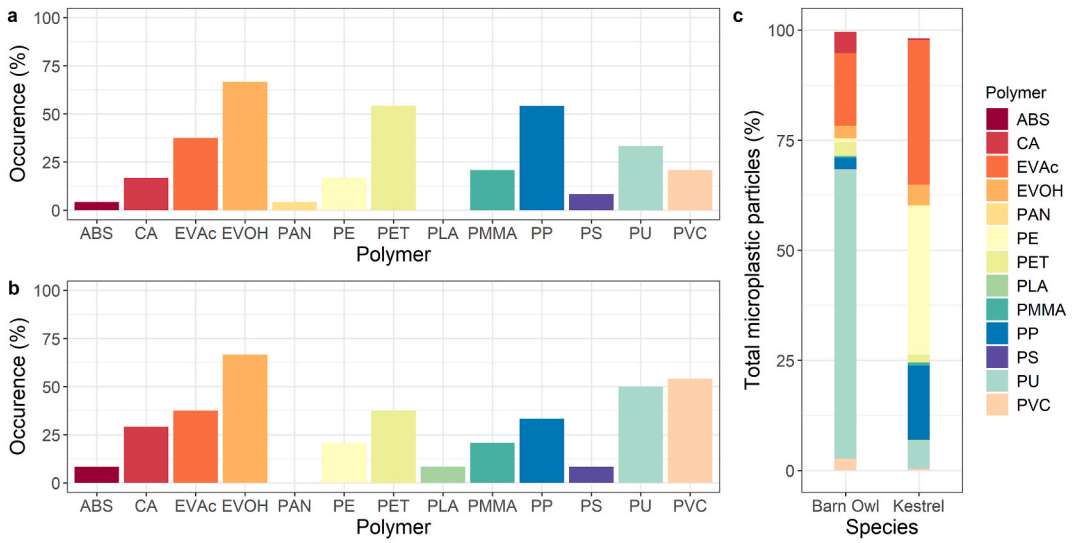

Fig. 3. Percentage occurrence of each polymer across all pellets in the study, separated by (a) kestrels (*Falco tinnunculus*) and (b) barn owls (*Tyto alba*) in Hampshire, southern England. Figure shows how often a polymer occurred over n = 24 pellets for each species. (c) Percentage of total particles corresponding to each polymer type separated by species.

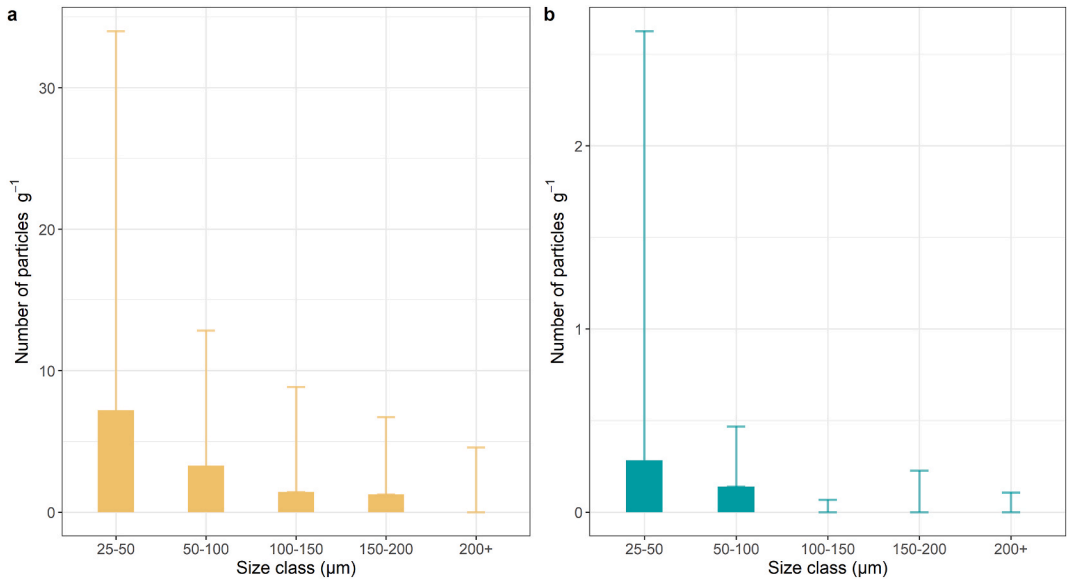

Fig. 4. Median numbers of microplastic particles per g (g^{-1}) of pellet and 25th and 75th centiles for (a) kestrels (*Falco tinnunculus*) and (b) barn owls (*Tyto alba*) in southern England grouped by particle size (μm).

pellets, and PLA, which occurred only in barn owl pellets (Fig. 3). Overall, kestrel pellets had higher mean number of polymers that co-occurred in both species. The dominant polymers in kestrel pellets were EVOH, PP, PU, PET, and EVAc (Fig. 2). Barn owl pellets were dominated by EVOH, PU, EVAc, and CA (Fig. 2). Differences between the means of polymers for the two species were significant for EVAc, EVOH, PET, PP, and PU (all $p < 0.05$) (Table S5). Over 50% of pellets for both species contained EVOH. Other polymers that were frequently detected in samples were PET, PP, EVAc, PU, and PVC. The dominant polymers in terms of mean particle count also corresponded to higher percentage occurrence, however did not consistently translate to higher percentage of total particles (Fig. 2; Fig. 3).

3.3. Particle size distribution

The pellets of both species had higher concentrations of smaller particles, with the highest particle counts in the 25-50 μm range (Fig. 4). 52.07% of particles in kestrel pellets were 25-50 μm in size and 67.18% of particles in barn owl pellets were between 25-50 μm.

4. Discussion

4.1. Interspecific differences in microplastic ingestion

To our knowledge, this study is the first to investigate plastic ingestion by raptors in the UK (Mansfield et al., 2024). Microplastic concentrations in kestrel pellets were significantly higher than those in barn owl pellets by ~26-fold (Fig. 1), a finding that contradicts our initial hypothesis that the two species would exhibit similar levels of contamination. Additionally, there were interspecific differences in the dominant polymers in pellets. In terms of occurrence and number of particles g^{-1}, EVOH and EVAc were the dominant polymer types for both species. PP was dominant in kestrel samples and for barn owl samples PU and PVC were dominant. However, in terms of the

percentage of the total particles identified, PU was dominant in barn owl samples and PE and PP in kestrel samples (Fig. 3). Ingestion of microplastics via prey is the main route of exposure for both species, since they do not drink water (Bunn et al., 1992; Village, 1990). Other than ingestion of prey, the only other potential route of exposure is through preening, where atmospheric plastic may be present on their feathers (Jeong et al., 2023). It should be noted that while our study uses pellets as a proxy for microplastic ingestion, pellets represent egested material and therefore cannot be used to estimate plastics retained in the body (Provencher et al., 2019).

The foraging ecology of barn owls and kestrels shows a high degree of similarity, with both species primarily feeding on small mammals and small birds (Marti et al., 2024; Orta et al., 2020b), and small mammals form the largest percentage of prey taken (Bond et al., 2005). Moreover, both species produce one to two pellets every 24 h, with pellets often consisting of multiple food items consumed during the foraging period (Davis, 1975; Marti et al., 2024; Orta et al., 2020b; Yalden and Yalden, 1985). Within the UK, both species are considered to be vole specialists, with the short-tailed field vole making up a significant percentage of prey taken (Bunn et al., 1992; Village, 1990). Previous research conducted in the UK, albeit with a limited sample size, has shown that 33% of short-tailed field voles ingest microplastics (Thrift et al., 2022). Both species are known to only supplement their diet with other prey species when short-tailed field vole abundance is poor (Bunn et al., 1992; Village, 1990), though we did not conduct a diet analysis of pellets, the sampling year we conducted our study in was a year of good abundance for short-tailed field voles (M. Stevens, Personal Communication). Hence, it is unlikely that birds of either species in our study were heavily supplementing their diets.

One possible explanation for the interspecific differences in microplastic ingestion and polymer composition is variation in the microplastic load of individual prey. Both bird species are mobile and able to forage over large areas (Bunn et al., 1992; Village, 1990), meaning they encounter prey feeding in different locations, which may be exposed to different sources and levels of pollution. These results indicate that relying on pre-existing knowledge of foraging ecology may not be enough to determine risk of microplastic exposure. Diet analysis utilising stable isotope analysis or prey identification in pellets may further aid future studies in determining relationships between what species are eating and their microplastic exposure. This variation in exposure also makes it harder to identify 'hotspots' of exposure for such mobile species. Further studies or monitoring attempts utilising GPS tracking to determine exact foraging locations may be needed to provide more accurate geographic information on exposure sources (De Pascalis et al., 2022).

These high levels of pellet-to-pellet variation also demonstrate that much larger sample sizes are needed in future to obtain an accurate picture of microplastic exposure to birds of prey with large foraging ranges. This result, coupled with the high levels of variance between the highest and lowest number of particles recorded in individual pellets over all samples, indicates that exposure is highly variable, potentially changing from meal-to-meal. There were also individual pellets present in our study that contained very high numbers of particles of a single polymer compared to other pellets taken from the same nest. This can be seen when comparing the results from Figs. 2 and 3, in which the dominance of polymers in these figures differs. For example, while PE has a relatively medium frequency of occurrence and mean number of microplastic particles for kestrels, it makes up a significant portion of percentage of total particles. This trend is then reversed for EVOH, which has high frequency of occurrence and mean number of particles but forms a much lower percentage of total particles. This indicates meal-to-meal exposure also differs in terms of the polymers a bird is exposed to as well and that it is likely that they are not consistently exposed to one single source of contamination.

Another factor that may affect microplastic ingestion rates is where species forage. There is evidence that subtle differences in where in the same environment species with highly overlapping diets take prey from can significantly influence how much microplastics they consume (Mylius et al., 2023). Additionally, seasonal and temporal variation can influence the microplastic load of prey species which in turn effects how much microplastics predators ingest (Carrillo et al., 2025). There are also physiological differences between the two species that may contribute to the differences in microplastic concentrations in their pellets. Barn owls lack a crop, whereas kestrels possess a functional crop that temporarily stores food prior to digestion (Bunn et al., 1992; Village, 1990). Differences in proximal gut residence time may determine whether microplastics are incorporated into the forming pellet or pass distally into the intestine. Increasing our understanding is especially important as habitat destruction and fragmentation forces predatory bird species to supplement their diets or forage in less favourable areas (Hindmarch et al., 2017; Wells et al., 2014), potentially increasing their exposure to anthropogenic pollution.

A critical methodological consideration when comparing mass-normalised microplastic concentrations (particles g^{-1}) between species is the substantial disparity in pellet mass. In our dataset, barn owl pellets weighed 5–20 g whereas kestrel pellets weighed only 0.4–1.6 g. Even if both species ingested similar absolute numbers of microplastics, the barn owl's larger pellet matrix would yield lower mass-normalised concentrations through simple dilution. This dilution effect represents a major caveat for interspecific comparisons based on pellet analysis. We attempt to compensate for this by estimating the number of microplastics present in an "average pellet", using the average weight of pellets for each species and the particles g^{-1} results from our study. Based on this we estimate that the average kestrel pellet would still contain a much higher concentration of microplastic particles than the average barn owl pellet.

4.2. Sources of most commonly detected polymers

The divergence between species extended to polymer composition, providing additional evidence that foraging microhabitat shapes the qualitative character of microplastic exposure. Across both species, EVOH and EVAc were the most frequently detected polymer types in terms of both occurrence and concentration. Both polymers are widely used in multilayer food packaging and agricultural films (Briassoulis and Dejean, 2010; Lagaron et al., 2004; Schettini et al., 2014) and their prevalence across both species is consistent with the degradation and fragmentation of such materials across agricultural landscapes. EVAc is also commonly utilised as a co-polymer to PE. EVAc/PE blends show improvements over PE alone and therefore EVAc use has increased in recent years as EVAc/PE blends become more common (Alothman, 2012). It is possible that the prey of both species are consuming EVAc when it occurs in co-polymer form with PE. Thrift et al. (2022) examined microplastics in small mammal faeces and found EVAc in 27% of samples, which supports the hypothesis that kestrels and barn owls are exposed to EVAc through their small mammal prey.

EVOH is particularly valued for its oxygen barrier properties in food packaging, while EVAc is commonly used in stretch films, greenhouse covers, and silage wraps—materials subject to UV-induced and mechanical fragmentation in field environments (Briassoulis and Dejean, 2010; Schettini et al., 2014). EVOH often co-occurs with other polymers due to its use as a permanent gas barrier (Mokwena and Tang, 2012). It is therefore possible that the high occurrence of EVOH in pellets is the result polymer co-occurrence, similar to EVAc. The co-occurrence of both polymers across species suggests a shared landscape-level source that transcends any interspecific differences in foraging strategy.

Despite this commonality, clear divergences in polymer composition were apparent. PP was the predominant polymer in kestrel samples, consistent with its widespread use in agricultural twine, mulch films, and food-contact packaging, whose fragmentation products are concentrated in the soil surface layer accessible to invertebrate prey

(Chamas et al., 2020; Kumar et al., 2020). By contrast, PU and PVC were predominant in barn owl samples, with PU also accounting for the largest proportion of total particles identified in this species when expressed as a percentage of all particles. PU is extensively used in construction insulation, coatings, and foam products, and its occurrence in barn owl pellets may reflect the species' strong association with agricultural buildings and barns, where weathering and abrasion of PU-based construction materials could contribute to indoor or peri-structural MP loading (Hamilton et al., 2025). PVC is similarly ubiquitous in agricultural and built environments, used in piping, cable sheathing, and flooring, and its detection in barn owl samples may reflect exposure through prey inhabiting such structures. When polymer occurrence was expressed as a percentage of total identified particles, PE and PP dominated in kestrels, while PU dominated in barn owls. PE and PP are among the most abundantly produced and environmentally dispersed plastics globally (Geyer et al., 2017; Hahladakis et al., 2018) and their dominance in kestrel samples reinforces the inference that kestrels are exposed primarily through soil and agricultural pathways where these polymers are well-represented. Despite a limited sample size, the presence of microplastics has been detected in agricultural soil in Hampshire (Radford et al., 2022). However, the lack of large-scale environmental monitoring data of atmospheric and soil microplastics in Hampshire presents limitations to linking soil microplastic contamination and ingestion of microplastics by our study species. These compositional differences between species highlight the utility of polymer profiling as a tool for tracing exposure pathways in wildlife, complementing abundance-based metrics.

4.3. Particle size distribution

For both species, microplastics concentration expressed as particles g^{-1} increased consistently with decreasing particle size, down to the 25 μm detection limit of our instrument. This inverse relationship between particle size and abundance is consistent with findings across a wide range of environmental matrices and biological samples, reflecting the progressive fragmentation of larger plastic items into exponentially greater numbers of smaller particles over time (Adediran et al., 2026; Andrady, 2011; Koelmans et al., 2015). The prevalence of smaller particle sizes is also tied to the effect of digestion on plastic particles. The process of digestion can cause plastic particles to become brittle (Babkiewicz et al., 2025; Prabhu et al., 2024), which can lead to fracturing of the particles in the digestive tract (Babkiewicz et al., 2025; Meng et al., 2023; Prabhu et al., 2024). The dominance of smaller particles in our samples could indicate that the birds in our study ingested plastic indirectly via their prey, where particles have been fractured by the digestive process of the prey prior to the prey being ingested by the birds. The prevalence of smaller size fractions has important implications for biological uptake and toxicity. Smaller MPs present a greater collective surface area per unit mass, enhancing their capacity for adsorption of hydrophobic contaminants, heavy metals, and persistent organic pollutants (Menéndez-Pedriza and Jaumot, 2020; Rochman et al., 2013; Uber et al., 2019).

The predominance of small particles in both species therefore suggests that the toxicological significance of microplastics exposure in these raptors may be substantially underestimated by studies that focus on visible particle counts alone without accounting for sub-micron size distribution. It is important to note that the observed size distribution is necessarily constrained by our instrument's lower detection limit of 25 μm. Particles below this threshold, which are likely present in considerable numbers given the observed trend, were not captured in our analysis, representing a systematic underestimation of total MP load. Additionally, because pellet formation involves compaction and mechanical processing of ingested material within the proventriculus and gizzard (Bunn et al., 1992; Village, 1990), it is plausible that some fragmentation of larger particles occurs during digestion, potentially generating smaller particles *in situ* and contributing to the observed size distribution independently of environmental inputs.

5. Conclusion

This study provides the first characterisation of microplastic exposure in birds of prey in the United Kingdom, using regurgitated pellets as a non-invasive sampling matrix to assess dietary microplastic burden in barn owls and common kestrels sharing overlapping habitat in Hampshire. Microplastics were detected in the majority of pellets from both species, confirming that dietary exposure to microplastics is widespread among UK raptors and that pellet-based approaches represent a viable and ethically sound method for biomonitoring in these protected species. Importantly, our study in rural Hampshire presented the opportunity to quantify microplastic exposure to birds that forage in non-urban habitats, far from significant local sources of pollution, such as cities and landfills. High microplastic detection rates and significant variation in the polymers detected in our study indicate that microplastic exposure in these species may be a widespread problem.

Polymer profiles further revealed that the two raptors are exposed to distinct suites of landscape-derived plastics. Shared dominance of EVOH and EVAc indicates a common source, while species-specific enrichment of PP in kestrels and PU/PVC in barn owls reflects contrasting trophic pathways and microhabitat use. Particle-size distributions were similarly consistent across species, with strong enrichment of the smallest detectable fractions, highlighting the ecological relevance of fine-scale microplastic exposure and the limitations imposed by current analytical detection thresholds.

Together, these findings show that pellet analysis offers a powerful but inherently filtered view of raptor microplastic exposure. Integrating abundance, polymer composition, and particle-size data with species-specific ecological traits and diet analysis could provide a more accurate framework for interpreting pellet-derived contamination metrics. Such an approach will be essential for future work advancing the use of raptors as bioindicators of terrestrial plastic pollution and for tracing the movement of plastics through agricultural landscapes. Future work will expand to additional raptor species, incorporate sub-25 μm particles, and link pellet-based metrics with in-tissue burdens to refine exposure and risk assessments.

CRediT authorship contribution statement

Rana Ozturk: Writing – review & editing, Writing – original draft, Visualization, Validation, Methodology, Investigation, Formal analysis, Conceptualization. **Gbotemi A. Adediran:** Writing – review & editing, Supervision, Methodology, Conceptualization. **Dominic P. Brass:** Writing – review & editing, Formal analysis. **William F. Mills:** Writing – review & editing, Supervision. **Matt Stevens:** Writing – review & editing, Resources. **Alexander Robinson:** Writing – review & editing, Supervision, Methodology. **Stuart Black:** Writing – review & editing, Supervision, Project administration.

Declaration of competing interest

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

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envpol.2026.128483.

Data availability

Data will be made available on request.

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