



Microplastic ingestion and diet analysis in migratory shorebirds at Bahía Lomas Ramsar site, Chile

Ludovica Giovinazzi^{a,*}, Herald V. Norambuena^b, Carmen Espoz Larraín^b, Erik M. Sandvig^{b,c}, Michael Di Gioacchino^a, Armida Sodo^a, Massimiliano Scalici^{a,d}, Juan A. Amat^e, Andy J. Green^{f,g}, Cristina Coccia^{a,b,d,**}

^a Department of Science, University of Rome Tre, Viale Guglielmo Marconi 446, Rome, 00146, Italy

^b Centro Bahía Lomas, Facultad de Ciencias Universidad Santo Tomás, Av. Ejército 146, Santiago, Chile

^c Red de Observadores de Aves y Vida Silvestre de Chile, Chile

^d National Biodiversity Future Center (NBFC), Università di Palermo, Piazza Marina 61, Palermo, 90133, Italy

^e Department of Ecology and Evolution, Estación Biológica de Doñana (EBD), CSIC, C/ Américo Vespucio 26, Sevilla, 41092, Spain

^f Department of Conservation Biology and Global Change, Estación Biológica de Doñana (EBD), CSIC, C/ Américo Vespucio 26, Sevilla, 41092, Spain

^g Department of Natural Sciences, Manchester Metropolitan University, Manchester, M1 5GD, UK

ARTICLE INFO

Keywords:

Wetlands
Emergent contaminants
Trophic transfer
Stable isotopes

ABSTRACT

Microplastic (MP) pollution is an emerging threat to marine and coastal ecosystems and their fauna, even in remote and protected areas. Shorebirds depend on wetlands and feed mainly on benthic invertebrates, making them vulnerable to contaminants that accumulate in sediments and their prey. Stable isotope analysis provides a useful framework to interpret MP ingestion in relation to trophic ecology. We applied stable isotopes of carbon ($\delta^{13}\text{C}$) and nitrogen ($\delta^{15}\text{N}$) to investigate the diet of three migratory shorebirds: the Two-banded Plover (*Anarhynchus falklandicus*), the Magellanic Oystercatcher (*Haematopus leucopodus*), and the Hudsonian Godwit (*Limosa haemastica*) at Bahía Lomas (Tierra del Fuego, Chile), and we quantified MP ingestion.

Stable isotope analyses revealed that the Two-banded Plover exhibited a larger and differentiated isotopic niche respect to the other two species, whose niche widths were similar and showed substantial overlap. Mixing models indicated a greater dietary contribution of biofilm for Two-banded Plover, suggesting feeding at lower trophic levels and a stronger association with surface sediments. MPs were detected in sediments, macro-invertebrate prey, and fecal samples of all species, but MP contamination was higher in Two-banded Plover. This difference may be associated with its greater proportional reliance on biofilm. However, this exposure pathway remains hypothetical, as MPs were not directly quantified in biofilm. All detected MPs were fibres, predominantly blue, with polyethylene the most frequent polymer. We provide the first evidence of MP contamination in shorebirds and their trophic resources at Bahía Lomas and anywhere in Chile, illustrating how plastic pollution impacts remote and protected wetlands.

1. Introduction

Plastic production has increased exponentially since the 1950s, reaching 8.3 billion tons globally (Rhodes, 2018; Williams and Rangel-Buitrago, 2022). This rapid growth in production has resulted in a parallel accumulation in the environment. In aquatic ecosystems, including wetlands, plastic waste is expected to increase threefold by 2060 (Ramsar The Convention on Wetlands, 2023). The extensive use of

plastics, combined with inefficient waste management, has further contributed to its widespread dispersal across all environments, even in remote regions (Williams and Rangel-Buitrago, 2022; Ziani et al., 2023).

As a consequence of its ubiquity and persistence, plastic pollution is considered one of the major global environmental challenges with impacts on biodiversity, ecosystem functioning, and human health (de Souza Machado et al., 2018; Wang et al., 2021; Horton, 2022).

Once released into the environment, plastics fragment into smaller

* Corresponding author.

** Corresponding author. Department of Science, University of Rome Tre, Viale Guglielmo Marconi 446, 00146, Rome, Italy.

E-mail addresses: ludovicagiovinazzibio@gmail.com (L. Giovinazzi), cristina.coccia@uniroma3.it (C. Coccia).

<https://doi.org/10.1016/j.envres.2026.125336>

Received 15 April 2026; Received in revised form 14 July 2026; Accepted 26 July 2026

Available online 27 July 2026

0013-9351/© 2026 Elsevier Inc. All rights are reserved, including those for text and data mining, AI training, and similar technologies.

particles known as mesoplastics (5–25 mm) and microplastics (<5 mm; MPs), through physical, chemical, and biological degradation (GESAMP, 2019; Susanti et al., 2020; Issac and Kandasubramanian, 2021). Due to their small size, persistence, and mobility, MPs are easily transported by wind and water and become widely available to aquatic organisms, entering aquatic food webs at lower trophic levels such as plankton and benthic invertebrates and being transferred to higher consumers including fish, birds, and mammals (Desforges et al., 2015; Issac and Kandasubramanian, 2021; Gallitelli et al., 2024).

In addition to their physical effects, MPs can adsorb persistent organic pollutants, heavy metals, and pharmaceutical compounds, acting as vectors of chemical contaminants along trophic chains (Bakir et al., 2012; Holmes et al., 2012). Once ingested, MPs can cause tissue damage, metabolic alterations, oxidative stress, immune disruption, and reproductive impairment (Zolotova et al., 2022; Wang et al., 2021; Liu et al., 2023).

Shorebirds (or “waders”, order *Charadriiformes*) rely on wetlands for foraging and resting, feeding on benthic invertebrates –such as polychaetes, oligochaetes, crustaceans, bivalves and aquatic insect larvae– by probing or pecking into soft sediments. This makes them vulnerable to contaminants that accumulate in sedimentary environments or are transferred through contaminated prey (Thomas et al., 2004; Ng et al., 2024).

Many shorebird species are long-distance migrants that depend on wintering and stopover sites to accumulate the energetic reserves necessary for migration and reproduction (Johnson and Sherry, 2001; Piersma and Lindström, 2004). MP ingestion may reduce body mass, cause nutritional depletion, gut dysbiosis, and exposure to associated chemical contaminants, potentially affecting migratory performance and reproductive success (Jin et al., 2019; de Souza et al., 2022; Liu et al., 2025). Documented declines in shorebird populations are often not explained by food availability alone (Morrison et al., 2004; Dey et al., 2011), highlighting the need to investigate MP contamination in migratory species that can be exposed to pollution hotspots anywhere along flyways (Flemming et al., 2022; Athira et al., 2024).

In remote southern Chilean coasts, plastics are the most abundant form of anthropogenic debris (Bravo et al., 2009). Despite the implementation of national strategies to reduce plastic waste, including bans on single-use plastics, plastic contamination remains a major issue (Brooks et al., 2020). It has already been documented in the Magellanic region in Chile (Andrade and Ovando, 2017), an area that hosts critical wintering habitats for shorebirds, such as the Bahía Lomas wetland that supports large populations of many migratory and resident shorebird species (Morrison and Ross, 1989; Espoz et al., 2008; Norambuena et al., 2022a). Bahía Lomas is exposed to multiple anthropogenic pressures, including oil and gas extraction, livestock grazing, maritime traffic, mining activities, and the expansion of wind farms, all of which may contribute to pollution, including MPs (Espoz et al., 2016; Espoz et al., 2022, Norambuena et al. 2022a,b).

This study aimed to investigate the risk of exposure and ingestion of MPs in three shorebird species: the Nearctic migrant Hudsonian Godwit (*Limosa haemastica*) and two Neotropical migratory species, the Two-banded Plover (*Anarhynchus falklandicus*), and the Magellanic Oystercatcher (*Haematopus leucopus*), which breed or overwinter at Bahía Lomas. These species differ in body size and bill length, but have overlapping diets based on benthic prey, including polychaetes and bivalves (Atkinson et al., 2005; Espoz et al., 2008; Senner and Coddington, 2011; Leiva and Ferrari, 2014; Kuwae et al., 2012; Musmeci et al., 2013).

By applying stable isotopes of carbon ($\delta^{13}\text{C}$) and nitrogen ($\delta^{15}\text{N}$) to fecal samples and their potential prey, as well as quantifying MPs, we aimed to relate shorebird diet to potential pathways of MP exposure and contamination.

Our specific objectives were:

1. To characterize shorebird diet by applying stable isotope in both consumers and their potential prey.

2. To quantify MPs in shorebird prey (benthic macroinvertebrates), sediments and in fecal samples, and characterize physical and chemical properties of these MPs, to identify the main exposure pathways of MPs within the food web.

Our initial hypotheses were:

H1) The Godwit and Oystercatcher would have similar $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ and overlapping diets due to their similar bill length (6–8 cm) that allow them to exploit prey at similar depths (Nebel et al., 2005). In contrast, we expected the smaller Plover with a shorter bill (1–2 cm), to feed on different prey in the upper sediment layers and so have different isotopic signatures.

H2) For the same reasons, we hypothesized that the Godwit and Oystercatcher would exhibit similar levels of MP contamination and MP morphological and polymeric composition. In contrast, the Plover was expected to have different contamination levels and distinct MP characteristics.

2. Materials and methods

2.1. Study area and species

Bahía Lomas is a large bay located in the southern Strait of Magellan, northeastern Tierra del Fuego, Chile (Fig. 1). It is a Ramsar Site and a Site of Hemispheric Importance by the Western Hemisphere Shorebird Reserve Network (Espoz et al., 2016), extending from Punta Catalina (52°32′05″ S, 68°49′4″ W) to Punta Anegada (52°27′3″ S, 69°26′14″ W), covering ~69 km (Espoz et al., 2011). Bahía Lomas is a coastal–marine ecosystem, characterized by a tidal range of 8–9 m (Espoz et al., 2008) exposing vast mudflats and sandy areas during low tide that can extend >7 km offshore (Morrison and Ross, 1989). The climate is cold-temperate, with austral summer (December–March) temperatures of 6–12 °C and winds of 80–90 km/h, and winter (June–August) temperatures < −1 °C, with winds of 60–70 km/h (Espoz et al., 2011). Low precipitation results in a Patagonian steppe in upper intertidal zones dominated by *Festuca* grasses, with brackish areas covered by *Salicornia* (https://whsrn.org/whsrn_sites/bahia-lomas/).

The Godwit breeds in Alaska and overwinters in South America, including at Bahía Lomas. It is IUCN Vulnerable due to widespread population declines. The Plover breeds in southern Chile, including Tierra del Fuego, with a partial north migration along the coast up to northern Chile and southeastern Brazil (Wiersma et al., 2023). It forages on small benthic invertebrates, and occasionally on terrestrial invertebrates within *Salicornia* near the nesting area. The Oystercatcher breeds in southern Patagonia and Tierra del Fuego, with non-breeding flocks present throughout the year in Bahía Lomas.

2.2. Selected sampling stations and field data collection

Three main sampling areas were selected within Bahía Lomas (Fig. 1), corresponding to sites where the target species roost (Godwit and Oystercatcher), feed (all species) or nest (Plover). The roosting and foraging areas were characterized by sandy and muddy substrates, typical of the extensive tidal flats. The nesting area was further inland, where plovers nest in small ground depressions or among grass tussocks. Field work was carried out during the austral summer (November and December 2024), i.e. the breeding season of Plover and Oystercatcher and the non-breeding season for Godwit.

Fecal samples were collected following two different protocols. Plover samples were obtained from individuals at nesting sites captured using a spring trap under permits issued by the Servicio Agrícola y Ganadero (SAG; permit codes 6610_2024 and 7069_2024). Plovers were ringed, measured and temporarily (up to 10 min) placed in a sampling box lined with a clean aluminium foil and a metal mesh (40 × 30 cm) to allow fecal collection (Fig. S1), following Knutie and Gotanda (2018). Feces were collected from the foil using a stainless-steel spatula and

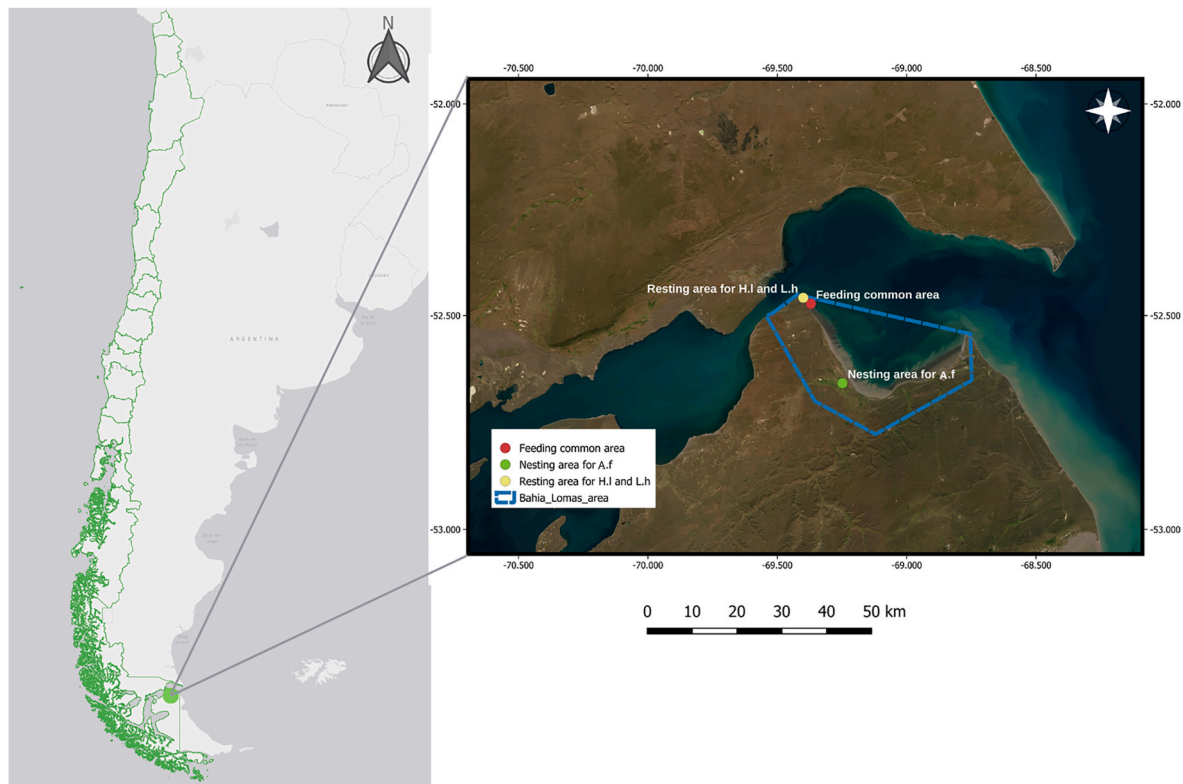


Fig. 1. Location of the Bahía Lomas Ramsar Site in northeastern Tierra del Fuego. The three main sampling zones were the nesting area for Two-banded Plover (green); the common feeding area (red); and the resting area for Magellanic Oystercatcher and Hudsonian Godwit (yellow).

transferred into glass vials. The foil was replaced between individuals, while equipment was cleaned with 90% ethanol to avoid cross-contamination.

For Oystercatcher and Godwit, fresh feces were collected from the ground at the roosting site after locating flocks of each species. Sampling began 10-min after individuals arrived and droppings were collected at least 3 m apart to minimize resampling of individuals (Lu et al., 2022). Only fresh and intact droppings were collected, and the bottom layer in contact with the substrate was discarded. The upper portion of each dropping was collected with a stainless-steel spatula to minimize the incorporation of adhered sediment particles. The spatula was sterilized with 90% ethanol after each use. Vials containing feces were stored at -18°C until analysis (Silberberger et al., 2020).

2.2.1. Sediment and food resource sampling

Sediment samples and potential food sources were collected within the intertidal foraging area during low tide. Sampling was conducted along four transects of 250 m in length perpendicular to the shoreline and spaced 250 m apart. Along each transect, five sampling points spaced 50 m apart were established, resulting in 20 sampling points within an area of 1 km^2 .

At each point, sediments and macroinvertebrates were collected using a metal shovel to a depth of 8 cm, to represent the depth accessible to all shorebirds. Samples were stored in aluminium trays and covered with lids. Surface biofilm samples were collected at the beginning of each transect by brushing the sediment surface with a sterile toothbrush (Kuwaie et al., 2008). Biofilm samples ($n = 4$) were stored into 30 ml Falcon tubes at -18°C .

2.3. Laboratory processing

At the field station, macroinvertebrates were sorted from sediments and processed for both isotope and MP analyses (see Supplementary

Materials; Section 1.1. for details).

Individuals were identified to the lowest possible taxonomic level, using available keys (Domínguez and Fernández, 2009; Hanson et al., 2015) and the most abundant taxa (bivalves, annelids, crustaceans, and dipterans) were selected for analyses (Table S1).

For stable isotope analyses, when sample size per taxon was insufficient, individuals were pooled across transects (Table S1), whereas widespread taxa, such as Bivalvia and Isopoda were analyzed from each sampling point to capture spatial variability. In this latter case, samples consisted of two individuals per replicate. Three replicates were analyzed for pooled samples.

2.3.1. Stable isotope analysis

Fecal and frozen macroinvertebrate samples were oven-dried at 45°C for 48 h, weighed with a precision balance ($\pm 0.001\text{ g}$), and packed in tin capsules ($\pm 0.3\text{ mg}$ for fecal samples and $0.9\text{--}1\text{ mg}$ for macroinvertebrates).

Each biofilm sample was vacuum filtered using glass fiber filters ($1\text{ }\mu\text{m}$ pore) to retain the biofilm on the filter surface. After filtration, the filters were oven-dried at 45°C for 48 h to ensure complete dehydration of biological material. The upper portion of the filter containing the biofilm was then carefully scraped off using sterile tweezers and a spatula. The biofilm was homogenized, weighed to obtain approximately 0.3 mg of dry material, and packed into a tin capsule.

Finally, samples were sent to the Estación Biológica de Doñana (EBD-CSIC), Spain, where carbon and nitrogen isotopic ratios ($\delta^{13}\text{C}$ and $\delta^{15}\text{N}$) were determined using Isotope Ratio Mass Spectrometry (IRMS).

2.3.2. Microplastic extraction

Quantification and characterization of MPs were carried out on remaining fecal (individual sample) and macroinvertebrate (minimum of three individuals per taxon) samples using the same standardized digestion protocol. All samples were weighed ($\pm 0.001\text{ g}$), prior to

processing. Samples were digested with pre-filtered hydrogen peroxide 35% (H_2O_2) and incubating for 48 h, until complete dissolution of organic matter (Gallitelli et al., 2024). Digested samples were vacuum filtered using 1 μm glass fiber filters, and stored in clean, closed Petri dishes until analysis.

Sediment samples were sieved (1 mm) to remove macroinvertebrates and oven-dried at 45 °C for 48 h. A 50 g of dry subsample was treated with zinc chloride (ZnCl_2 at a 1:3 V/L), homogenized and left to settle for 48 h to separate lighter MPs from denser particles (Ekstrand, 2018; Duong et al., 2022). The supernatant was vacuum filtered, and glass fiber filters were then stored in Petri dishes. Filters from each matrix (feces, macroinvertebrates, and sediments) were examined under a stereomicroscope to identify suspected MPs, which were then classified according to shape (e.g. fragments and fibers) and color (i.e., red, blue, and black) following Lusher et al. (2020) and (Cowger et al., 2020).

2.3.3. Analysis of identified items using Raman spectroscopy

When necessary, subsamples of suspected MPs, selected based on shape and color were analyzed through Raman spectroscopy to determine polymeric composition (Käppler et al., 2016; Gallitelli et al., 2024, see supplementary Material; Section 1.2 for instrumental details).

Spectral processing and polymer identification were performed using OpenSpecy software, comparing sample spectra to those contained in the program's reference database. Only matches with a similarity score ≥ 0.6 were accepted (Morgado et al., 2021). However, to assess the potential influence of the identification threshold on polymer-specific patterns, Raman spectra were additionally re-evaluated using a more conservative similarity threshold (≥ 0.7). When Raman spectra were dominated by pigment-related bands and the underlying material could not be reliably identified, particles were classified as "pigment". These particles were considered as anthropogenic contaminants, rather than MPs because the presence of colourant signatures is indicative of anthropogenic processing (Lusher et al., 2020; Munno et al., 2020). This distinction was made to avoid potential MP overestimation, while still accounting for both the anthropogenic origin and potential environmental implications of these particles. Overall, all fibers detected in macroinvertebrates, 73% of those detected in sediments, and 84% of those detected in fecal samples were analyzed spectroscopically.

2.3.4. Quality control

Quality control was implemented during both field and laboratory work. To assess field contamination, procedural blanks consisting of 1 μm glass fiber filters inside a sterile Petri dish were exposed during sample collection. For Plover, field blanks were associated with individual capture sessions and processed alongside the corresponding fecal samples. For Godwit and Oystercatcher, one field blank per species was exposed during the entire fecal collection period (approximately 30 min). In addition, one field blank was exposed during sediment and macroinvertebrate sampling, which were conducted simultaneously at the same sampling sites.

In the laboratory, blanks were used during all critical steps including weighing, filtration, taxonomic identification and separation. All filters and laboratory tools (tweezers, spatulas and pipettes) were pre-screened under a stereomicroscope and cleaned with filtered distilled water and 95% ethanol. Hydrogen peroxide (35% H_2O_2) and distilled water were prefiltered before use. Filtration steps and sediment processing were performed under a sterile laminar flow hood switched-off in order to minimize potential turbulence and airborne contamination associated with active airflow. The work was not conducted under sterile laboratory conditions; however, the hood was used as a physical barrier to reduce exposure of samples to ambient laboratory air. Procedural blanks were included during homogenisation and handling steps to monitor potential contamination.

All control filters were examined under a stereomicroscope, and any suspected MPs were analyzed by Raman spectroscopy. A total of three final control filters showed MP contamination: two field blanks exposed

during the collection of plover samples in the field were identified as polyester, and one laboratory blank associated with amphipod filtration was identified as polypropylene. However, no MPs were detected in the corresponding biological samples. Therefore, contaminated blanks were not used for quantitative blank correction; instead, they were evaluated through polymer-mismatch verification to assess whether external contamination could have influenced polymer composition patterns.

Taken together, these observations indicate that sporadic airborne contamination occurred during field sampling and laboratory processing. However, the absence of MP in the corresponding biological samples suggests a limited impact of external contamination on the study results.

2.4. Statistical analyses

Data were analyzed in R software (version 4.1.1) using the packages *vegan*, *glmmTMB*, *emmeans*, *DHARMA*, *SIBER* (version 2.1.9) and *simmr* (version 0.5.1.217) and results were visualized with *ggplot2*.

2.4.1. Stable isotope analyses

Differences in $\delta^{13}\text{C}$ and in $\delta^{15}\text{N}$ between taxa were tested using a generalized linear model (GLM) with a Gaussian distribution. Model assumptions (normality and homogeneity of residuals) were tested using the *DHARMA* package (Hartig, 2017; Fig. S2). When significant effects were detected, pairwise comparisons across species were performed using the *emmeans* package (Lenth, 2018) with Tukey's HSD for multiple testing. We visualized isotopic niche overlap among shorebird species using Standard Ellipse Areas corrected for sample size (SEAc expressed in ‰²), and used Bayesian standard ellipse area (SEAb) implemented in *SIBER* package in R (Jackson et al., 2011). SEAc ellipses contained approximately 40% of the data and were derived from the variance-covariance structure of $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values, representing the core isotopic niche of each species and serving as a proxy for richness and evenness of resource use (Bearhop et al., 2004). SEAb was used to evaluate probabilistic differences in isotopic niche width among species based on posterior distributions. The extent of niche overlap (%) among species was quantified using maximum likelihood estimates of 95% standard ellipse areas ($\text{SEA}_{95\text{ ML}}$), and proportional overlap was calculated as the ratio between the overlapping area and the total combined ellipse area.

Dietary contribution of macroinvertebrates and biofilm to the diet of each shorebird species were estimated, using a Bayesian mixing model in the *simmr* package (Parnell et al., 2013). Trophic enrichment factors (TEFs) of 0 for Carbon and -0.6 ± 0.5 SD for Nitrogen were applied following Kuwae et al. (2022). Prey taxa with similar δ -values were grouped to form single sources (see Table S2). The *SIMMR* model was run using the default MCMC settings (10,000 iterations, burn-in of 1,000, thinning of 10, and 4 chains). Model convergence and fit were assessed using the Gelman–Rubin diagnostic ($\hat{R} \approx 1.00$ for all parameters), and plotting posterior predictive distribution. Effective sample sizes (n.eff) ranged from approximately 1700 to 3600 across parameters and species, indicating adequate independent sampling of posterior distributions. A random seed (set to 100) was used to ensure full reproducibility of the analyses.

2.4.2. Microplastic analyses

A Spearman correlation test was conducted to assess the relationship between MP concentrations in macroinvertebrates and in sediment (MP/g) along the same transects. For fecal samples, a Spearman correlation test was performed to assess the relationship between MP abundance and fecal sample weight (g). Since a significant correlation was detected (Spearman correlation: $\rho = 0.39$, $p < 0.05$), subsequent analyses used MP abundance per gram (MP/g) as the response variable.

Differences in MP abundance among the shorebird species were tested using a GLM, with MP abundance per gram (MP/g) as the

response variable and species as explanatory variable. The model was applied using the *glmmTMB* function (Bolker, 2019) with a Tweedie distribution. Model validation was performed using the DHARMA package (Hartig, 2017; Fig. S3). When significant effects were detected, pairwise species comparisons were performed using Tukey's HSD post hoc tests implemented in the emmeans package to identify significant differences among species. Results were represented using the ggplot2 package (Wickham, 2011).

To evaluate whether MP contamination was associated with individual Plover body condition, we calculated the Scaled Mass Index (SMI) following Peig and Green (2009).

$$1) SMI_i = M_i(L_0/L_i)^{b_{SMA}}$$

where M_i is the body mass of individual i , L_i is its wing length, and L_0 corresponds to the mean wing length of the sampled population. The scaling exponent (b_{SMA}) was the slope estimated from the regression between log-transformed body mass and log-transformed wing length.

SMI values were then correlated with MP abundances (MP/g) using Spearman correlation.

3. Results

3.1. Diet determination of the three species through stable isotope analysis

In this study, $\delta^{13}C$ values ranged from a minimum of -23.71 in Plover to a maximum of -16.02 in Godwit, while $\delta^{15}N$ values ranged from 10.41 in Plover to 12.67 in Godwit. The GLM showed significant differences in both $\delta^{13}C$ and $\delta^{15}N$ across the three species ($p < 0.001$). Specifically, the post-hoc Tukey test showed that Plover was more depleted in $\delta^{13}C$ respect to the other species ($p < 0.0001$; Table 1), whereas $\delta^{13}C$ values in Oystercatcher and Godwit did not differ significantly from each other ($p = 0.4782$). For $\delta^{15}N$, the post-hoc Tukey test indicated a lower $\delta^{15}N$ in Plover respect to Godwit ($p = 0.0496$), whereas no differences were found between Plover and Oystercatcher, nor between Oystercatcher and Godwit.

The standard ellipse area corrected for small sample size (SEAc) provides a point estimate of isotopic niche width, whereas the Bayesian standard ellipse area (SEAb) represents a probabilistic estimate that accounts for uncertainty in niche width derived from the posterior distribution. SEAb values showed no strong posterior support for isotopic niche width differences between Godwit and Oystercatcher (posterior probability = 0.63 , Table S3), which also exhibited the highest niche overlap observed (32%) based on maximum likelihood 95% standard ellipse areas (SEA_{95 ML}, Table S3). In contrast there was a strong posterior support for the Two-banded Plover exhibiting larger isotopic niche width than both Oystercatcher and Godwit (posterior probability = 0.92 and 0.99 , respectively, Table S3). In term of isotopic niche space, the SEAc showed a clear separation between the Two-banded Plover and the Godwit and Oystercatcher (Table 1; Fig. 2).

The Bayesian SIMMR showed that all macroinvertebrates sampled were incorporated in the diet of the three shorebird species, but in different proportions. The diet of Plover showed a higher proportional contribution of biofilm compared to other resources (Fig. 3). In Oystercatcher, the diet was mainly based on biofilm, the group amphipods-chironomids-polychaetes-oligochaetes, and bivalves.

Table 1

Mean ($\pm$ SD) values of $\delta^{13}C$, $\delta^{15}N$, corrected standard ellipse area (SEAc) and Bayesian standard ellipse area (SEAb) for the three shorebirds species.

Species	N	$\delta^{13}C$ ($\pm$ SD)	$\delta^{15}N$ ($\pm$ SD)	SEAc ($\% ^2$)	SEAb ($\% ^2$)
<i>A. falklandicus</i>	13	-23.71 ± 3.38	10.41 ± 3.19	27.16	26.77
<i>H. leucopodus</i>	22	-17.46 ± 2.20	12.42 ± 2.11	11.97	11.93
<i>L. haemastica</i>	8	-16.02 ± 4.08	12.67 ± 1.13	14.82	13.72

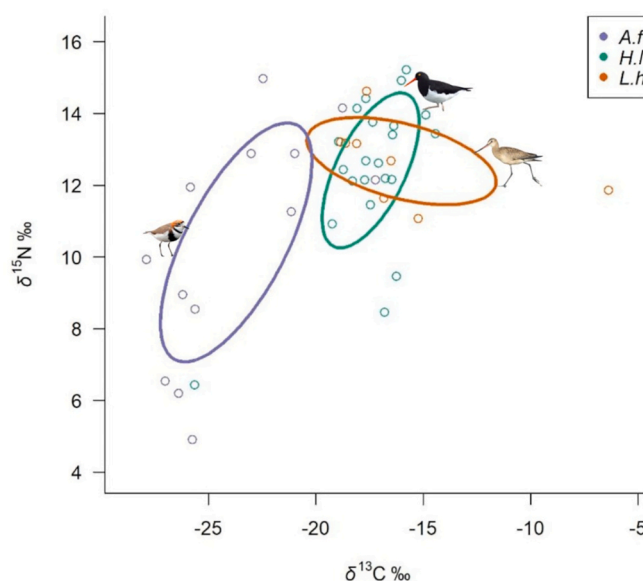


Fig. 2. Isotopic ellipses (SEAc) based on $\delta^{13}C$ and $\delta^{15}N$ values of the three shorebird species. Each ellipse represents the trophic niche width of each species, estimated from the carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) isotopic signatures obtained from the isotopic analyses. A.f = *Anarhynchus falklandicus*; H.l = *Haematopus leucopodus*; L.h = *Limosa haemastica*.

Finally, for Godwit our model suggested a similar median contribution of all sampled sources to its diet. However, the substantial overlap in posterior distributions, limited our ability to clearly distinguish the individual contribution of each source (Table S4).

3.2. Microplastic contamination across networks and bird species

A total of 416 macroinvertebrate individuals and 20 sediment samples were analyzed. In addition, 48 shorebird fecal samples were processed. MPs were detected in 21% of macroinvertebrate samples, 70% of sediment samples and 44% of fecal samples. Overall, 8 suspected MPs were recorded in macroinvertebrates, 66 in sediments and 31 in shorebirds.

MP abundance (MP/g) ranged between 0.06 MP/g in sediments to 0.81 MP/g in individual shorebird fecal samples (Fig. S4). Mean MP abundance ($\pm$ SD) per individual bird was 0.92 ± 0.76 MPs in Plover ($n = 13$), 0.28 ± 0.61 MPs in Oystercatcher ($n = 25$), and 1.20 ± 1.40 MPs in Godwit ($n = 10$). MP abundance in macroinvertebrate (either as individual or pooled) varied across taxa, ranging from 0 MP/g in Isopoda and Sipuncula to 32.05 ± 55.51 MP/g in Chironomidae. No association was detected between the MP/g found in sediments and in macroinvertebrates from the same transect ($n = 4$; $\rho = -0.04$, $p = 0.47$). However, the small number of transects substantially limited statistical power, and moderate relationships may have remained undetected.

The MP abundance was significantly different among the three shorebird species (GLM, $p < 0.0001$; Fig. S5). The Tukey post-hoc test indicated that Plover had significantly higher MP concentration than Oystercatcher ($p < 0.0001$) and Godwit ($p < 0.0001$), while no significant difference was found between Oystercatcher and Godwit ($p = 0.06$).

All the 105 suspected MPs were fibers (Fig. S6). Blue was the most frequent colour across the three matrices (macroinvertebrates, sediment, and fecal samples), followed by transparent fibers and red fibers. Black fibers were present only in macroinvertebrate samples.

A total of six polymer types were identified across the three matrices: Polyesters/ethers/diglycidyl ethers/terephthalates (PET); Polyethylene (PE); Polyester (PES); Polyvinyl alcohols (PVA); Poly-propylene (PP);

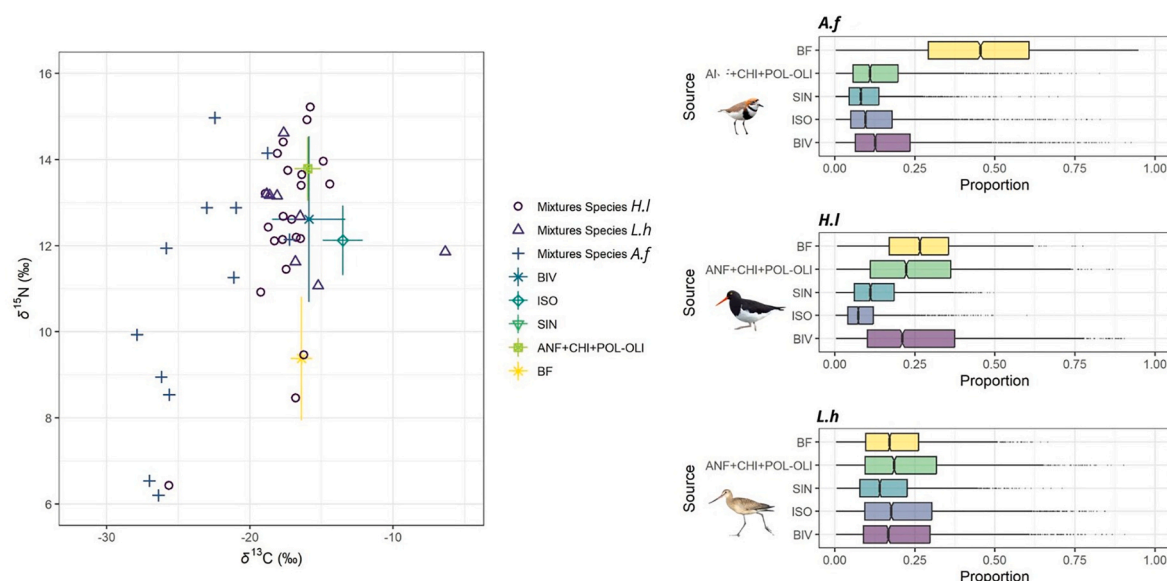


Fig. 3. Shorebird dietary proportions estimated by the SIMMR mixing model based on $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values. The left panel shows the isotopic space of consumers and sources, while the right panels display posterior distributions of dietary contributions for each species. Prey groups are indicated by colors (see legend): BIV = bivalves, ISO = isopods, SIN = sipunculids, ANF-CHI-POL-OLI = amphipods, chironomids, polychaetes and oligochaetes, and BF = biofilm. *A.f* = *Anarhynchus falklandicus*; *H.I* = *Haematopus leucopus*; *L.h* = *Limosa haemastica*.

and Polystyrenes/polyphenylethylenes/methylstyrene (PS) (Fig. S7, Table S5). Among these, only PET, PE, PES, and PVA were found in shorebird feces, although not all polymers were present in every species (Fig. 4; Table S6). Pigments were common, but their relative proportions varied between networks and among bird species (Fig. 4 and S5; Table S6). PE was the only polymer found across all three matrices, although its relative abundance varied among networks and bird species. PP was detected exclusively in sediments and PS only in macroinvertebrates. PES and PVA were restricted to shorebirds, but not in all three species (Fig. 4 and S7; Tables S5 and 6). PET occurred in both macroinvertebrates (specifically, in chironomids and bivalves) and shorebirds (in Plover and Oystercatcher) (Fig. 4 and S7; Tables S5 and 6). The sensitivity analysis using a similarity threshold ≥ 0.7 reduced the number of accepted polymer identifications (Table S7). Except for PVA in shorebirds and PP in sediments, the main polymer assignments remained consistent with those obtained using a similarity threshold ≥ 0.6 (Fig. S8 and Table S8).

For Plover, SMI values ranged from 64.0 to 82.7 ($n = 13$), indicating

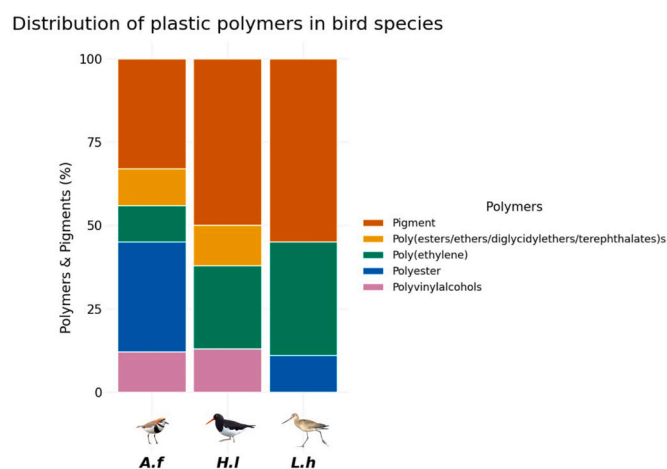


Fig. 4. Percentage distribution (%) of pigments and polymers in fecal samples of shorebirds. *A.f* = *Anarhynchus falklandicus*; *H.I* = *Haematopus leucopus*; *L.h* = *Limosa haemastica*.

variability in body condition among individuals. However, no significant association was found between SMI and MP abundance ($p = 0.06$, $p = 0.86$), although the power to detect moderate effects was limited.

4. Discussion

MPs were found across all networks, confirming their widespread occurrence in this remote ecosystem. Stable isotopes revealed that Plover had a different trophic niche and higher MP load values compared to Godwit and Oystercatcher. This pattern may be associated with its greater reliance on biofilm, which has been proposed as a potential pathway of MP exposure in coastal ecosystems. However, because MPs were not quantified directly in biofilm in the present study, this mechanism remains hypothetical. The polymeric and morphological characteristics of MPs detected across shorebird species indicated a combination of shared environmental exposure and species-specific pathways, potentially influenced by differences in their diet and habitat use. The major differences in breeding ranges and migratory behaviour are unlikely to explain the observed patterns, as MPs are not expected to have long gut retention times and are likely to be ingested within our study area.

4.1. Diet determination of the three species through stable isotope analysis

We found significant differences in carbon ($\delta^{13}\text{C}$) and nitrogen ($\delta^{15}\text{N}$) values across the studied species. Specifically, Plover was significantly $\delta^{13}\text{C}$ depleted compared to Godwit and Oystercatcher, and had lower $\delta^{15}\text{N}$ than Godwit suggesting a greater contribution of marine pelagic or depleted carbon terrestrial and marsh C3 vegetation-derived sources to its diet and a tendency to feed at a lower trophic level (Bearhop et al., 2002; Post, 2002; Bocher et al., 2014). This is also supported by the niche width analyses (Fig. 2), showing a clear separation in the isotopic space of Plover from those of Godwit and Oystercatcher, whose niches overlapped. Further, the larger isotopic niche width (as indicated by SEAb) of Plover suggests that it adopts a more variable foraging strategies and potentially utilizes additional prey or basal resources not sampled in this study (Post, 2002; Bocher et al., 2014).

The depleted $\delta^{13}\text{C}$ signature of Plover can be interpreted in two, not mutually exclusive, pathways each supported by the available evidence.

On the one hand the SIMMR model showed that Plover exhibited the highest estimated proportional contribution of biofilm among the food sources included in the model, accompanied by a more balanced intake of the other trophic resources, although posterior distribution indicated considerable uncertainty and overlap among species.

This relative high contribution of biofilm is only partially consistent with Musmeci et al. (2013), who reported for Plover a diet composed mainly of polychaetes, molluscs, and isopods. This may reflect the limited detection of biofilm in traditional gut content studies, as it is not easily identifiable through sieving and visual inspection, and highlights the value of stable isotope analysis, which provides a more comprehensive assessment of trophic contributions. Our results are also consistent with other studies showing the inclusion of biofilm in the diet of closely-related plover species (Kuwaie et al., 2022; Canham et al., 2023). According to Kuwaie et al. (2012), shorebird species with lower body mass, such as Plover, make greater use of biofilm than larger species. This can be partly explained by their more specialized tongues and oral structures, which allow them to scrape the biofilm. Additionally, larger species have higher energy requirements and therefore may prefer richer food sources, such as benthic invertebrates.

On the other hand, the contribution of terrestrial marsh-derived carbon sources cannot be excluded. C3 vegetation (e.g., *Salicornia* spp.) or associated sediment organic matters are characterized by similarly depleted $\delta^{13}\text{C}$ values (Cloern et al., 2002; Duarte et al., 2018), yet these sources were not included in the mixing model. In addition, Plover individuals were observed feeding within the shared foraging area during sampling, nesting sites where samples were collected were located up to ~50 km away in habitats dominated by *Salicornia*, providing a plausible opportunity for the incorporation of marsh-derived carbon into Plover diet. Because these sources were not explicitly included in the mixing model, their relative contribution cannot be quantified, and their influence cannot be distinguished from that of biofilm.

Importantly, some Plover individuals exhibited $\delta^{15}\text{N}$ values lower than those of biofilm, representing an isotopic mass-balance constraint that suggests the contribution of additional nitrogen sources not captured by the sampled resources. Consequently, the available isotopic evidence does not allow to separate the relative contributions of biofilm and unsampled marsh-derived carbon sources. Future studies incorporating terrestrial primary producers and marsh-derived organic matter into isotopic mixing models will be necessary to resolve the relative importance of these pathways to further clarify the dietary and energetic role of biofilm in Plover.

Although Godwit and Oystercatcher exhibited similar isotopic niche width and partial overlap, indicating the use of similar trophic resources and feeding at comparable trophic levels, the SIMMR mixing model revealed differences in the relative contribution of these resources between the two species. In particular, Oystercatcher included all sampled resources, but with a slightly higher contribution of bivalves and biofilm, which partially agree with Leiva and Ferrari (2014), who reported that Oystercatcher feeds primarily on molluscs (including bivalves) and polychaetes. This may reflect its ability to exploit different intertidal substrates, as this species forages both on rocky substrates, where bivalves are abundant, and on soft sediments, which hold a wide range of benthic prey. In contrast, Godwit exhibited a more balanced diet, reflecting a more generalized exploitation of the intertidal food web. However, posterior distributions showed substantial overlap, indicating that these differences are not fully resolved and reflect only partial differentiation in resource use within a shared trophic space.

This study provides the first isotopic evidence supporting a contribution of biofilm to the diet of Oystercatcher and Godwit, and highlights the need for further research on its ecological role within intertidal food webs.

All together, these results supported our initial hypothesis (H1), suggesting that Godwit and Oystercatcher exhibit partially overlapping diets due to their similar bill lengths, whereas the short-billed Plover

shows a different trophic niche.

4.2. Microplastic contamination

MPs were detected in all network levels (macroinvertebrates, sediments, and feces), indicating that shorebirds may be exposed to MP contamination through both sediment and their prey. Nevertheless, shorebirds may also be exposed to MPs through pathways other than sediments or their prey, including biofilms, which have been proposed as important reservoirs of MPs in coastal ecosystems. However, we did not directly quantify MPs within biofilm, so this pathway remains a plausible component of exposure network. Future studies should explicitly assess MP abundance in biofilm to clarify its role as a potential exposure vector.

Shorebird feces contained the highest MP concentration, consistent with other studies from coastal systems (Liu et al., 2023; Mylius et al., 2023; Athira et al., 2024, 2025). However, the MP abundance found at Bahía Lomas (0.81 MP/g) was lower than that reported in shorebirds from more urbanised and anthropogenically impacted coastal systems, including tidal flats of the Yellow Sea (3.77 MP/g), and estuarine and coastal habitats along the west coast of India (41.0 MP/g) (Liu et al., 2023; Athira et al., 2025), reflecting its relatively remote and less urbanised nature.

We also found interspecific differences in MP abundance. Plover exhibited higher MP concentration than Godwit and Oystercatcher when normalized by fecal dry mass, in line with our second hypothesis (H2), which predicted comparable levels of MP contamination in Godwit and Oystercatcher due to their similar diet, and differences with Plover. Indeed, MP abundance normalized by fecal mass and MP abundance expressed per individual may provide different patterns of interspecific variation, as they reflect different aspects of MP occurrence in fecal samples. Therefore, the higher MP concentration in Plover result should be interpreted cautiously, as it does not necessarily reflect a higher absolute number of MPs ingested per individual. Differences in fecal production, digestion, and excretion patterns among species may influence MP concentration value.

Despite this limitation, the higher MP concentration in Plover may reflect its diet more reliant on biofilm, which can accumulate high concentrations of MPs into the polysaccharide matrix present at the water surface (Kuwaie et al., 2012; Flemming et al., 2022), as well as its broader trophic niche. However, because MPs were not quantified directly in biofilm in the present study, this explanation should be considered a plausible hypothesis rather than a demonstrated mechanism. In contrast, similar MP concentrations in Godwit and Oystercatcher together with their niche overlap, suggest comparable MP exposure linked to similar feeding habits. At least in the case of Plover, the body condition was not affected by MP abundance, indicating that in the sampled individuals it did not appear to be influenced by the ingestion of MPs.

Regarding MP characteristics, all MPs detected across matrices and species were fibres, predominantly blue, a pattern consistent with other studies in coastal environments in other parts of the world (Lusher et al., 2020; Cincinelli et al., 2017; Peng et al., 2017; Aves et al., 2022) and in marine environments across Latin America (Terrazas-López et al., 2024). The prevalence of blue fibres observed in this study may therefore indicate a marine origin of the MPs ingested by shorebirds. Blue plastics are among the most produced and widely distributed plastic materials, particularly in fishing gear (e.g. ropes, nets, and lines), packaging materials, and synthetic textiles, increasing their likelihood of entering marine and coastal food webs.

Most particles were pigments, characterised by Raman signatures that masked the underlying material and hindered reliable identification of the polymer or fibre composition. Although some of these particles may not be synthetic polymers, pigment- or dye-related compounds associated with them may contribute to potential toxicological effects through the presence of chemical additives, dyes, or trace metals

potentially posing additional toxicological risks to organisms (Gallo et al., 2018; Munno et al., 2020). PE was the most widespread polymer and the only one shared between sediments, macroinvertebrates and all shorebird species, suggesting that shorebird exposure to this polymer may occur both directly, through contact with contaminated sediments during foraging, and indirectly, through the ingestion of prey items that have already incorporated MPs. PE is the most frequent polymer found in aquatic ecosystems across the Americas and elsewhere (Kalogerakis et al., 2017; Cera et al., 2020) and is also the most common polymer found in shorebirds (Mylius et al., 2023; Athira et al., 2025). The dominance of PE may be explained by its high global production, widespread use, and positive buoyancy in seawater, promoting dispersion and accumulation in coastal environments (Mylius et al., 2023).

Since the stricter criterion reduced the number of accepted identifications, interpretations of individual polymer occurrences should be considered cautiously. In particular, PP and PVA assignments were not retained under the more stringent similarity threshold (≥ 0.7), highlighting the need for caution when interpreting these specific polymer occurrences.

PP was identified exclusively in sediments; since the density of this polymer ($0.83\text{--}0.85\text{ g/cm}^3$) is lower than seawater (1.025 g/cm^3), it would be expected to remain buoyant (Andrady, 2011). Nonetheless, PP can accumulate in sediments due to biofouling, i.e., the colonisation of plastic surfaces by microorganisms, algae, and other biological material, which increases particle density and promotes sinking (Kooi et al., 2017), reducing its potential to adhere to biofilm and be ingested by shorebirds. Its absence in shorebirds in this study contrasts with Mylius et al. (2023), who reported PP as the dominant polymer in both shorebirds and the sediments on which the birds foraged along Tasmanian coasts. This difference may reflect site-specific environmental conditions or lower exposure of PP to shorebirds, e.g. due to different trophic webs.

The presence of PET and PS in macroinvertebrates and shorebirds is consistent with the possibility that these polymers can be ingested by shorebirds through their prey. In particular, PET was found in chironomids and bivalves, which constitute prey for all three shorebirds. Although its occurrence in these taxa was limited, this finding suggests a common trophic transfer pathway for this polymer, as reported in other shorebird systems (Athira et al., 2025). PET is typically associated with synthetic textiles and packaging materials, and its presence may reflect inputs from household wastewater, textile-derived microfibres transported by marine currents, or the breakdown of consumer plastic debris reaching the Magellanic region. PS is linked to food packaging and insulation materials, and marine litter may derive from ship-based waste, fishing gear degradation, or long-distance transport of buoyant polystyrene fragments (Andrady, 2011; Recabarren-Villalón et al., 2023), consistent with the intense marine traffic in the Strait of Magellan.

PET and PVA were detected in Plover and Oystercatcher, while PES was found in Godwit and Plover. To our knowledge these polymers have not been previously reported in shorebirds (Mylius et al., 2023; Athira et al., 2025). PES is commonly associated with synthetic textiles and fishing-related materials, whereas PVA is widely used in industrial, biomedical and pharmaceutical applications, including fibers and films for packaging and contact lenses (De Merlis and Schoneker, 2003).

Several factors can explain the apparent absence of certain polymers in some species, including sampling stochasticity the limited sample size and the high proportion of pigment-dominated spectra, which prevent polymer identification. However, the detection of certain polymers in shorebirds but not in the other environmental matrices may suggest the existence of different exposure pathways not addressed in this study. Further investigation of potential pathways, including biofilm and other dietary resources, would be needed to evaluate this hypothesis. Although preliminary, these observations further support the potential of shorebirds as sentinels capable of revealing different sources of MP contamination. Although the polymers detected in this study are

classified as low to moderate hazards according to Lithner et al. (2011), their environmental persistence and ability to transport associated contaminants still result in toxicological risks, particularly for species exposed to a greater diversity or higher number of polymers. Increasing the number of fecal samples in future work could assess interspecific variability in polymer occurrence and the influence of species-specific dietary proportions on the ingestion of these polymers.

5. Limitations of this study

This study presents several methodological limitations that should be considered when interpreting the results.

First, differences in sampling areas between Plover from on the one hand and Godwits and Oystercatchers on the other may limit the ability to fully disentangle species identity from spatial differences (i.e. habitat use).

Second, this study is subject to several differences in sampling and analytical procedures across matrices. These include different fecal collection protocols among species, as well as differences in MP extraction and separation methods applied to sediments versus biological samples. Such methodological heterogeneity may have influenced the comparability of MP counts and polymer composition across matrices and species. In addition, sediment processing involved sieving through a 1 mm mesh and may have introduced a size-selective bias by excluding the coarsest sediment fraction. This may have influenced the recovery of sediment-associated MPs and should be considered when interpreting the polymer composition across matrices. Furthermore, the practical lower detection limit of approximately $20\text{ }\mu\text{m}$ during stereomicroscopic examination may have resulted in the underestimation of smaller MP particles and fibers, which could have been systematically overlooked.

Third, the relatively limited sample size, together with the low number of spectrally confirmed particles and the high proportion of pigment-dominated spectra, reduced the statistical power to detect ecological patterns and limited the robustness of inferences regarding polymer-specific differences among species and matrices. The difficulty of obtaining reliable polymer identifications from pigment-dominated Raman spectra is however a well-recognised limitation in environmental MP studies (Azari et al., 2024; Randhawa, 2025).

Fourth, we applied the trophic enrichment factor derived from other shorebird taxa (Kuwaie et al., 2022), which may not fully reflect isotopic discrimination in the species examined here. In particular, the use of a single TEF across multiple shorebird families (Charadriidae, Haematopodidae, and Scolopacidae) represents a simplification that may introduce additional uncertainty, which is propagated through the SIMMR model.

Furthermore, not all potential food sources available in the study area, particularly within the nesting area (e.g., *Salicornia* and sediment organic matter), were sampled and isotopically characterized, and therefore these sources were not included in the SIMMR model. In addition, the fact that several plover individuals showed $\delta^{15}\text{N}$ values below those of biofilm suggests that additional low- $\delta^{15}\text{N}$ food sources, not included in our study, may contribute to their diet. Alternatively, this may reflect spatial heterogeneity in biofilm isotopic composition that was not captured by the limited number of samples collected, or uncertainty associated with the trophic enrichment factor applied to fecal material, as discussed above. Therefore, while our results suggest that biofilm may represent an important dietary component for Plover, its contribution should be interpreted with caution.

6. Conservation implication and conclusions

This study represents the first effort to assess the presence of MPs in shorebirds, their prey, and sediments at Bahía Lomas, of the most important wetlands in the Southern Hemisphere and a site of international relevance for migratory bird conservation (Espoz et al., 2011) and

the first in Chile to link MP ingestion with diet using stable isotope analysis. Our results provide the first evidence of widespread, but still low MP contamination in this ecosystem, confirming their presence even in remote areas.

MPs were detected in shorebird feces, benthic macroinvertebrates, and sediments, across all shorebird species studied, indicating that these contaminants have become an integral component of coastal trophic chains, with shared exposure pathways.

Among the three shorebird species, Plover showed the highest MP abundance, potentially linked to its feeding ecology. Biofilm represents an important component of Plover diet within the range of resources exploited. However, isotopic values were also consistent with the use of terrestrial C3 vegetation sources or sediment organic matter. Biofilm can retain MPs in their polysaccharide matrix, and may therefore represent an important MP exposure pathway. Furthermore, the frequent use of inland vegetated areas dominated by *Sarcocornia* spp, where biofilm and insects are abundant, particularly during high tide may have further contributed to the higher MP contamination found in this species. However, because MPs were not quantified directly in biofilm or *Sarcocornia* spp. in the present study, these pathways remain speculative and require further investigation.

Importantly, the Godwit, currently classified as Vulnerable (VU) on the IUCN Red List (BirdLife International, 2024), may be particularly at risk, due to its long migratory journey, as MP contamination can reduce energetic capacities in shorebirds (Lourenço et al., 2017; de Souza et al., 2022; Athira et al., 2024). In this species, both pigments and PE were detected. Pigments may carry chemical additives or trace metals with potential toxicological and endocrine-disrupting effects (Gallo et al., 2018), while PE may act as a vector of contaminant due to its widespread occurrence and persistence. Although PE is generally considered to have moderate toxicity compared to other polymers (Lithner et al., 2011), its occurrence in a long distance migratory species facing high energetic demands highlights the need to assess its potential sublethal effects and cumulative physiological stress. Given the increasing anthropogenic pressures and the high conservation values of Bahía Lomas, continued monitoring of contamination, and expansion to other species present are highly recommended.

We showed that shorebirds can serve as sentinels of MP pollution in these ecosystems. Moreover, migratory shorebirds may act as vectors of MPs across distant ecosystems, potentially contributing to their redistribution at large spatial scale. This highlights their dual role as bio-indicators of environmental contamination and as agents of MP transport across ecosystems (Carrasco et al., 2025; Carrillo et al., 2025).

Future studies should include additional feeding areas, sources and exposure pathways (i.e. water). to improve understanding of MP dynamics in Magellanic coastal ecosystems and support integrated, conservation and mitigation strategies.

CRedit authorship contribution statement

Ludovica Giovanazzi: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Writing – original draft, Writing – review & editing. **Heraldo V. Norambuena:** Methodology, Writing – review & editing. **Carmen Espoz Larraín:** Methodology, Resources, Writing – review & editing. **Erik M. Sandvig:** Methodology, Writing – review & editing. **Michael Di Gioacchino:** Formal analysis, Writing – review & editing. **Armida Sodo:** Formal analysis, Writing – review & editing. **Massimiliano Scalici:** Resources, Writing – review & editing. **Juan A. Amat:** Formal analysis, Writing – review & editing. **Andy J. Green:** Formal analysis, Resources, Writing – original draft, Writing – review & editing. **Cristina Coccia:** Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Supervision, Validation, Visualization, Writing – original draft, Writing – review & editing.

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.

Acknowledgements

CC acknowledges support from the NBFC at the University of Roma Tre, funded by the National Recovery and Resilience Plan (NRRP), Mission 4 Component 2 Investment 1.4—Call for Tender No. 3138 of 16 December 2021, rectified by Decree No. 3175 of 18 December 2021 of the Italian Ministry of University and Research, funded by the European Union—NextGenerationEU. Project Code CN00000033, Concession Decree No. 1034 of 17 June 2022 adopted by the Italian Ministry of University and Research, CUP F83C22000730006, Project title “National Biodiversity Future Center—NBFC”. ES was supported by the Chilean National Agency for Research and Development (ANID) under grant [SIA85220018].

We also thank Sofia Brunetti for her assistance with microplastic identification through Raman spectroscopy. We acknowledge the International Mobility Grant for thesis research provided by the University of Roma Tre. We thank Instituto Santo Tomás at Punta Arenas, Centro Bahía Lomas and Gabriella Riquelme for logistical support and coordination during fieldwork, travel, and laboratory activities. We also thank Javiera Consuelo for her assistance with QGIS. We also acknowledge the permits issued by the Servicio Agrícola y Ganadero (SAG; permit codes 6610_2024 and 7069_2024) for field activities. We thank the anonymous reviewers for their constructive feedback, which contributed to improving the scientific robustness of our work.

Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.envres.2026.125336>.

Data availability

Data will be made available on request.

References

- Andrade, C., Ovando, F., 2017. First record of microplastics in stomach content of the southern king crab *Lithodes santolla* (Anomura: lithodidae), Nassau bay, Cape Horn, Chile. *An. Inst. Patagonia* 45 (3), 59–65. <https://doi.org/10.4067/S0718-686X2017000300059>.
- Andrady, A.L., 2011. Microplastics in the marine environment. *Mar. Pollut. Bull.* 62 (8), 1596–1605.
- Athira, T.R., Aarif, K.M., Rubeena, K.A., Musilova, Z., Musil, P., Angarita-Báez, J.A., et al., 2025. Tracing microplastics in environmental sources and migratory shorebirds along the Central Asian Flyway. *Mar. Pollut. Bull.* 212, 117571.
- Athira, T.R., Aarif, K.M., Thomas, J.A., Alatawi, A.S., Muzaffar, S.B., Nefla, A., et al., 2024. The threat of microplastics: exploring pollution in coastal ecosystems and migratory shorebirds along the west coast of India. *Mar. Pollut. Bull.* 198, 115912.
- Atkinson, P.W., Baker, A.J., Bevan, R.M., Clark, N.A., Cole, K.B., Gonzalez, P.M., et al., 2005. Unravelling the migration and moult strategies of a long-distance migrant using stable isotopes: red Knot *Calidris canutus* movements in the Americas. *Ibis* 147 (4), 738–749.
- Aves, A.R., Revell, L.E., Gaw, S., Ruffell, H., Schuddeboom, A., Wotherspoon, N.E., LaRue, M., McDonald, A.J., 2022. First evidence of microplastics in Antarctic snow. *Cryosphere* 16 (6), 2127–2145. <https://doi.org/10.5194/tc-16-2127-2022>.
- Azari, A., Ronsmans, S., Vanoirbeek, J.A., Hoet, P.H., Ghosh, M., 2024. Challenges in Raman spectroscopy of (micro)plastics. *Environ. Pollut.* 367, 125224.
- Bakir, A., Rowland, S.J., Thompson, R.C., 2012. Competitive sorption of persistent organic pollutants onto microplastics in the marine environment. *Mar. Pollut. Bull.* 64 (12), 2782–2789. <https://doi.org/10.1016/j.marpolbul.2012.09.019>.
- Bearhop, S., Hilton, G.M., Votier, S.C., Waldron, S., 2004. Stable isotope ratios indicate that body condition in migrating passerines is influenced by winter habitat. *Proc. R. Soc. Lond. B Biol. Sci.* 271 (suppl.4). <https://doi.org/10.1098/rsbl.2003.0129>.
- Bearhop, S., Waldron, S., Votier, S.C., Furness, R.W., 2002. Factors that influence assimilation rates and fractionation of nitrogen and carbon stable isotopes in avian blood and feathers. *Physiol. Biochem. Zool.* 75 (5), 451–458. <https://doi.org/10.1086/342800>.

- BirdLife International, 2024. *Limosa haemastica*. The IUCN red list of threatened Species e. T22693154A255829868. <https://dx.doi.org/10.2305/IUCN.UK.20242.RLTS.T22693154A255829868.en>.
- Bocher, P., Robin, F., Chiaradia, A., Le Dréan-Quénéché, S., 2014. Trophic resource partitioning within a shorebird community revealed by stable isotopes. *J. Sea Res.* 92, 115–122. <https://doi.org/10.1016/j.seares.2014.03.005>.
- Bolker, B., 2019. Getting started with the glmmTMB package. *9. Cran. R-project vignette*.
- Bravo, M., de Los Angeles Gallardo, M., Luna-Jorquera, G., Núñez, P., Vásquez, N., Thiel, M., 2009. Anthropogenic debris on beaches in the SE Pacific (Chile): results from a national survey supported by volunteers. *Mar. Pollut. Bull.* 58 (11), 1718–1726. <https://doi.org/10.1016/j.marpolbul.2009.06.017>.
- Brooks, A., Jambeck, J., Mozo-Reyes, E., 2020. Plastic Waste Management and Leakage in Latin America and the Caribbean. <https://doi.org/10.18235/0002873>.
- Canham, R., Rourke, J., Ydenberg, R.C., 2023. The exploitation of biofilm by migrant western sandpipers (*Calidris mauri*). *Heliyon* 9 (6), e17268.
- Carrasco, L., Jiménez-Mora, E., Utrilla, M.J., Pizarro, I.T., Reglero, M.M., Rico-San Román, L., Martín-Maldonado, B., 2025. Birds as bioindicators: revealing the widespread impact of microplastics. *Birds* 6 (1), 10.
- Carrillo, M.S., Archuby, D.L., Lunardelli, M., Castresana, G., Montalti, D., Ibáñez, A.E., 2025. Dynamics of microplastic transfer through the food web in a migratory seabird. *Environ. Pollut.* 383, 126784.
- Cera, A., Cesarini, G., Scalici, M., 2020. Microplastics in freshwater: what is the news from the world? *Diversity* 12 (7), 276. <https://doi.org/10.3390/d12070276>.
- Cincinelli, A., Scopetani, C., Chelazzi, D., Lombardini, E., Martellini, T., Katsoyiannis, A., Fossi, M.C., Corsolini, S., 2017. Microplastic in the surface waters of the Ross Sea (Antarctica). *Chemosphere* 175, 391–400.
- Cloern, J.E., Canuel, E.A., Harris, D., 2002. Stable carbon and nitrogen isotope composition of aquatic and terrestrial plants of the San Francisco Bay estuarine system. *Limnol. Oceanogr.* 47 (3), 713–729.
- Cowger, W., Gray, A., Christiansen, S.H., DeFrond, H., Deshpande, A.D., Hemabessiere, L., Lee, E., Mill, L., Munno, K., Ossmann, B.E., Pittroff, M., Rochman, C., Sarau, G., Tarby, S., Primpke, S., 2020. Critical review of processing and classification techniques for images and spectra in microplastic research. *Appl. Spectro.* 74 (9), 989–1010. <https://doi.org/10.1177/0003702820929064>.
- de Souza, S.S., Freitas, I.N., Gonçalves, S.O., Luz, T.M.D., Araújo, A.P.D.C., Rajagopal, R., et al., 2022. Toxicity induced via ingestion of naturally-aged polystyrene microplastics by a small-sized terrestrial bird. *J. Hazard Mater.* 434, 128814.
- de Souza Machado, A.A., Kloas, W., Zarfl, C., Hempel, S., Rillig, M.C., 2018. Microplastics as an emerging threat to terrestrial ecosystems. *Glob. Change Biol.* 24 (4), 1405–1416.
- DeMerlis, C.C., Schoneker, D.R., 2003. Review of the oral toxicity of polyvinyl alcohol (PVA). *Food Chem. Toxicol.* 41 (3), 319–326.
- Desforges, J.P.W., Galbraith, M., Ross, P.S., 2015. Ingestion of microplastics by zooplankton in the Northeast Pacific Ocean. *Arch. Environ. Contam. Toxicol.* 69, 320–330.
- Dey, A.D., Niles, L.J., Sitters, H.P., Kalasz, K.E.V.I.N., Morrison, R.G., 2011. Update to the Status of the Red Knot (*Calidris Canutus*) in the Western Hemisphere. Western Hemisphere Shorebird Reserve Network.
- Domínguez, Fernández, H., 2009. Macroinvertebrados acuáticos de Sudamérica. *Sistemática y Biología. Tucumán*, (1), 3–44.
- Duarte, C.M., Delgado-Huertas, A., Anton, A., Carrillo-de-Albornoz, P., López-Sandoval, D.C., Agustí, S., et al., 2018. Stable isotope ($\delta^{13}\text{C}$, $\delta^{15}\text{N}$, $\delta^{18}\text{O}$, δD) composition and nutrient concentration of Red Sea primary producers. *Front. Mar. Sci.* 5, 298.
- Duong, T.T., Le, P.T., Nguyen, T.N.H., Hoang, T.Q., Ngo, H.M., Doan, T.O., Le, T.P.Q., Bui, H.T., Bui, M.H., Trinh, V.T., Nguyen, T.L., Da Le, N., Vu, T.M., Tran, T.K.C., Ho, T.C., Phuong, N.N., Strady, E., 2022. Selection of a density separation solution to study microplastics in tropical riverine sediment. *Environ. Monitor. Assess.* 194 (2), 65. <https://doi.org/10.1007/s10661-021-09664-0>.
- Ekstrand, E., 2018. Optimization of Microplastic Sediment Separation and Application to a City-to-Fjord Gradient Study.
- Espoz, C., Labra, F., Matus, R., Ponce, A., Barría, I., Saavedra, B., Figueroa, A., Rondanelli, M., 2011. Plan de manejo para el sitio Ramsar Bahía Lomas. Ministerio del Medio Ambiente/Universidad Santo Tomás/Wildlife Conservation Society, Santiago, Chile, p. 131.
- Espoz, C., Matus, R., Haro, D., Luna, D., Norambuena, H.V., 2022. Effective Conservation and Good Governance at the Ramsar Site Bahía Lomas.
- Espoz, C., Matus, R., Luna-Quevedo, D., 2016. Bahía lomas, Ramsar site (Chile). In: *The Wetland Book*. Springer.
- Espoz, C., Ponce, A., Matus, R., Blank, O., Rozbaczyl, N., Sitters, H.P., et al., 2008. Trophic ecology of the Red Knot *Calidris canutus rufa* at Bahía Lomas, Tierra del Fuego, Chile. *Wader Study Group Bull.* 115 (2), 69–76.
- Flemming, S.A., Lanctot, R.B., Price, C., Mallory, M.L., Kühn, S., Drever, M.C., et al., 2022. Shorebirds ingest plastics too. *Environ. Res.* 30 (4), 537–551.
- Gallitelli, L., Cera, A., Scalici, M., Sodo, A., Di Gioacchino, M., Luzzi, B., et al., 2024. Plastic ingestion in aquatic insects. *Sci. Total Environ.* 924, 176707.
- Gallo, F., Fossi, C., Weber, R., Santillo, D., Sousa, J., Ingram, I., Nadal, A., Romano, D., 2018. Marine litter plastics and microplastics and their toxic chemicals components: the need for urgent preventive measures. *Environ. Sci. Eur.* 30 (1), 13. <https://doi.org/10.1186/s12302-018-0139-z>.
- GESAMP, 2019. Guidelines for the Monitoring and Assessment of Plastic Litter in the Ocean. GESAMP Reports and Studies No. 99.
- Hanson, P., Springer, S., Ramirez, A., 2015. Introducción a los grupos de macroinvertebrados acuáticos. *Revista de Biología Tropical* 3–37. <https://doi.org/10.15517/rbt.v58i4.20080>.
- Hartig, F., 2017. Package 'dharma'. *R package* 531.532, 435.
- Holmes, L.A., Turner, A., Thompson, R.C., 2012. Adsorption of trace metals to plastic resin pellets. *Environ. Pollut.* 160, 42–48.
- Horton, A.A., 2022. Plastic pollution: when do we know enough? *J. Hazard Mater.* 422, 126885.
- Issac, M.N., Kandasubramanian, B., 2021. Effect of microplastics in water and aquatic systems. *Environ. Sci. Pollut. Control Ser.* 28, 19544–19562.
- Jackson, A.L., Inger, R., Parnell, A.C., Bearhop, S., 2011. Comparing isotopic niche widths. *J. Anim. Ecol.* 80 (3), 595–602.
- Jin, Y., Lu, L., Tu, W., Luo, T., Fu, Z., 2019. Impacts of polystyrene microplastic on the gut barrier. *Sci. Total Environ.* 649, 308–317.
- Johnson, M.D., Sherry, T.W., 2001. Effects of food availability on migratory warblers. *J. Anim. Ecol.* 546–560.
- Kalogerakis, N., Karkanorachaki, K., Kalogerakis, G.C., Triantafyllidis, E.I., Gotsis, A.D., Partsinevelos, P., Fava, F., 2017. Microplastics generation: Onset of fragmentation of polyethylene films in marine environment mesocosms. *Front. Marine Sci.* 4. <https://doi.org/10.3389/fmars.2017.00084>.
- Knutie, S.A., Gotanda, K.M., 2018. A Non-invasive method to collect fecal samples from wild birds for microbiome studies. *Microb. Ecol.* 76 (4), 851–855. <https://doi.org/10.1007/s00248-018-1182-4>.
- Kooi, M., van Nes, E.H., Scheffer, M., Koelmans, A.A., 2017. Effects of biofouling on vertical transport of microplastics. *Environ. Sci. Technol.* 51 (14), 7963–7971.
- Käppler, A., Fischer, D., Oberbeckmann, S., Schernegg, G., Labrenz, M., Eichhorn, K.-J., Voit, B., 2016. Analysis of environmental microplastics. *Anal. Bioanal. Chem.* 408, 8377–8391.
- Kuwaie, T., Beninger, P.G., Decottignies, P., Mathot, K.J., Lund, D.R., Elner, R.W., 2008. Biofilm grazing in a higher vertebrate: the western sandpiper, *Calidris mauri*. *Ecology* 89 (3), 599–606.
- Kuwaie, T., Hosoya, J., Ichimi, K., Watanabe, K., Drever, M.C., Moriya, T., et al., 2022. Using stable isotope ($\delta^{13}\text{C}$, $\delta^{15}\text{N}$) values from feces and breath to infer shorebird diets. *Oecologia* 200 (1), 23–35.
- Kuwaie, T., Miyoshi, E., Hosokawa, S., Ichimi, K., Hosoya, J., Amano, T., et al., 2012. Variable and complex food web structures revealed by exploring missing trophic links between birds and biofilm. *Ecol. Lett.* 15 (4), 347–356.
- Leiva, E.J.P., Ferrari, S.N., 2014. Ecología trófica de una especie endémica de Patagonia Sur: el Ostrero Austral (*Haematopus Leucopus*) en el estuario del Río Gallegos (Santa Cruz). *Informes Científicos Técnicos - UNPA* 3 (2), 23–40. <https://doi.org/10.22305/ict-unpa.v3i2.33>.
- Lenth, R., 2018. Package 'lsmeans'. *Am. Statistician* 34 (4), 216–221.
- Lithner, D., Larsson, A., Dave, G., 2011. Environmental and health hazard ranking and assessment of plastic polymers based on chemical composition. *Sci. Total Environ.* 409 (18), 3309–3324. <https://doi.org/10.1016/j.scitotenv.2011.04.038>.
- Liu, W., Chen, X., Liang, T., et al., 2023. Varying abundance of microplastics in tissues associates with foraging strategies of shorebirds. *Sci. Total Environ.* 866, 161417.
- Liu, W., Zhao, C., Zhang, P., et al., 2025. Effects of microplastics on energy accumulation in a migratory shorebird. *J. Hazard Mater.* 488, 137386.
- Lourenço, P.M., Serra-González, C., Ferreira, J.L., Catry, T., Granadeiro, J.P., 2017. Plastic and other microfibers in sediments, macroinvertebrates and shorebirds. *Environ. Pollut.* 231, 123–133.
- Lu, Z., Li, S., Wang, M., Wang, C., Meng, D., Liu, J., 2022. Comparative analysis of the gut microbiota of three sympatric terrestrial wild bird species overwintering in farmland habitats. *Front. Microbiol.* 13, 905668. <https://doi.org/10.3389/fmicb.2022.905668>.
- Lusher, A.L., Bråte, L.L.N., Munno, K., Hurley, R.R., Welden, N.A., 2020. Is it or isn't it: visual classification of microplastics. *Appl. Spectrosc.* 74 (9), 1139–1153.
- Morgado, V., Palma, C., Bettencourt da Silva, R.J.N., 2021. Microplastics identification by infrared spectroscopy – Evaluation of identification criteria and uncertainty by the Bootstrap method. *Talanta* 224, 121814. <https://doi.org/10.1016/j.talanta.2020.121814>.
- Morrison, R.I.G., Ross, R.K., Niles, L.J., et al., 2004. Declines in wintering populations of Red Knots. *Condor* 106 (1), 60–70.
- Morrison, R.I.G., Ross, R.K., 1989. Atlas of Nearctic Shorebirds on the Coast of South America. Canadian Wildlife Service.
- Munno, K., De Frond, H., O'Donnell, B., Rochman, C.M., 2020. Increasing the accessibility for characterizing microplastics: Introducing new application-based and spectral libraries of plastic particles (SLoPP and SLoPP-E). *Anal. Chem.* 92 (3), 2443–2451. <https://doi.org/10.1021/acs.analchem.9b03626>.
- Musmeci, L., Bala, L.O., Hernández, M.D.L.A., 2013. Dieta del Chorlito Doble Collar (*Charadrius falklandicus*). *El Hornero* 28 (1), 15–21.
- Mylius, K.A., Lavers, J.L., Woehler, E.J., et al., 2023. Foraging strategy influences microplastic ingestion in shorebirds. *Environ. Pollut.* 319, 120844.
- Nebel, S., Jackson, D.L., Elner, R.W., 2005. Functional association of bill morphology and foraging behaviour. *Anim. Biol.* 55 (3), 235–243.
- Ng, J.Y., Ting, S.H., Rahman, M.A., Lee, P.L., 2024. Foraging behaviour of three shorebird species. *Malaysian Applied Biology Journal* 53 (2), 115–123.
- Norambuena, H.V., Labra, F.A., Matus, R., Gómez, H., Luna-Quevedo, D., Espoz, C., 2022b. Green energy threatens Chile's Magallanes region. *Science* 376 (6591), 361–362.
- Norambuena, H.V., Matus, R., Larrea, A., Espoz, C., 2022a. Censos aéreos de aves playeras en el santuario de la naturaleza Bahía Lomas, Enero 2022. Punta Arenas, Region de Magallanes, Chile: Informe Técnico-Centro Bahía Lomas, Universidad Santo Tomás 20.
- Parnell, A.C., Phillips, D.L., Bearhop, S., et al., 2013. Bayesian stable isotope mixing models. *Environmetrics* 24 (6), 387–399.
- Peig, J., Green, A.J., 2009. New perspectives for estimating body condition from mass/length data: the scaled mass index as an alternative method. *Oikos* 118 (12), 1883–1891.

- Peng, G., Zhu, B., Yang, D., et al., 2017. Microplastics in sediments of the Changjiang Estuary. *Environ. Pollut.* 225, 283–290.
- Piersma, T., Lindström, Å., 2004. Migrating shorebirds as sentinels of global change. *Ibis* 146, 61–69.
- Post, D.M., 2002. Using stable isotopes to estimate trophic position. *Ecology* 83, 703–718.
- Ramsar The Convention on Wetlands, 2023. The fight against plastic pollution is crucial for wetlands and for the services they provide us. https://www.ramsar.org/sites/default/files/2023-11/Wetlands_Plastic_Pollution_e.pdf.
- Randhawa, K.S., 2025. Polymer composites with advanced pigments: enhancing properties and applications. *Pigment Resin Technol.* 54 (4), 620–636.
- Recabarren-Villalón, T., Ronda, A.C., La Sala, L., Sanhueza, C., Díaz, L., Pirani, L.S.R., et al., 2023. First assessment of debris pollution in the gastrointestinal content of juvenile magellanic penguins (*Spheniscus magellanicus*) stranded on the west south Atlantic coasts. *Mar. Pollut. Bull.* 188, 114628.
- Rhodes, C.J., 2018. Plastic pollution and potential solutions. *Sci. Prog.* 101 (3), 207–260.
- Senner, N.R., Coddington, K.S., 2011. Habitat use and foraging ecology of Hudsonian godwits. *Wader Study Group Bull.* 118 (2).
- Silberberger, M.J., Kozirowska-Makuch, K., Kuliński, K., Kędra, M., 2020. Sample preservation and pre-treatment in stable isotope analysis: Implications for the study of aquatic food webs. *Biogeosci. Discuss.* <https://doi.org/10.5194/bg-2020-199>.
- Susanti, N.K., Mardiatuti, A., Wardiatno, Y., 2020. Microplastics and wildlife: a review. *Environ. Earth Sci.* 528, 012013.
- Terrazas-López, R., Guadarrama-Guzman, P., Sujitha, S.B., et al., 2024. Microplastics in the marine food web in Latin America. *Sustainability* 16 (14), 5905.
- Thomas, G.H., Wills, M.A., Székely, T., 2004. A supertree approach to shorebird phylogeny. *BMC Evol. Biol.* 4, 28.
- Wang, L., Nabi, G., Yin, L., et al., 2021. Birds and plastic pollution. *Avian Res.* 12, 59.
- Wickham, H., 2011. ggplot2. *WIREs Comput. Stat.* 3 (2), 180–185. <https://doi.org/10.1002/wics.147>.
- Wiersma, P., Kirwan, G.M., Boesman, P.F.D., 2023. Two-banded Plover (*Anarhynchus falklandicus*), version 1.1. In: del Hoyo, J., Elliott, A., Sargatal, J., Christie, D.A., de Juana, E. (Eds.), *Birds of the World*. Cornell Lab of Ornithology, Ithaca, NY, USA. <https://doi.org/10.2173/bow.twbplo1.01.1>.
- Williams, A.T., Rangel-Buitrago, N., 2022. The past, present, and future of plastic pollution. *Mar. Pollut. Bull.* 176, 113429.
- Ziani, K., Ioniță-Mindrican, C.B., Mititelu, M., et al., 2023. Microplastics: a global threat. *Nutrients* 15 (3), 617.
- Zolotova, N., Kosyreva, A., Dzhililova, D., Fokichev, N., Makarova, O., 2022. Harmful effects of microplastic pollution on animal health. *PeerJ* 10, e13503.