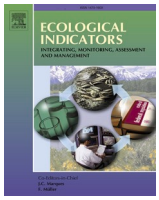
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Pollination under attack: First insights from Türkiye Plain reveal microplastics in bees from both urban and rural areas

Abdullah Altunışık [a,*], Damià Barcelò [b,c], Luca Gallitelli [d,*]

[a] Recep Tayyip Erdoğan University, Faculty of Arts and Sciences, Biology Department, Rize, Türkiye
[b] University of Almeria, Chemistry and Physics Department, Almería, Spain
[c] Sustainability Cluster, School of Engineering University of Petroleum and Energy Studies Dehradun, Uttarakhand, India
[d] Roma Tre University, Department of Sciences, Roma, Italy

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ABSTRACT

The increasing production and environmental release of plastics have led to the widespread presence of microplastics (MPs) in diverse ecosystems. Pollinators, such as honeybees (*Apis mellifera*), are particularly vulnerable to MP exposure due to their foraging behavior, which brings them into contact with contaminated environmental matrices, including air, water, and floral resources. This study aims to investigate the presence, characterization, and potential ecological implications of MPs in honeybees from two distinct regions in Türkiye: a polluted urban area and a pristine rural area. Honeybee samples were collected, digested, and analyzed for MP presence using ATR-FTIR spectroscopy. Our findings confirm the presence of MPs in honeybees from both study sites, with a significantly higher concentration in the urban area ($p < 0.01$). The majority of detected MPs were fibers (76 %), followed by fragments (19 %) and films (5 %). Five polymer types were identified, with polyethylene terephthalate (PET) being the most prevalent. Concerning the health index, 90.5 % of bees were found dead, while 9.5 % were alive. MPs in honeybees suggest potential implications for pollination services and pollinator health. This study highlights the need for further research on the ecological consequences of MP contamination in pollinators and its potential cascading effects on biodiversity and agricultural productivity.

1. Introduction

The global production of plastics has led to enormous widespread quantities, reaching more than 400 million tons of plastic annually worldwide (PlasticsEurope, 2024). When mismanaged or leaked into the environment, plastics provide many environmental contaminants, with microplastics (MPs) becoming a significant concern (Roy et al., 2022; Bexeitova et al., 2024; Zhao et al., 2024). MPs, defined as items measuring between 1 μm and 5 mm, were pervasive in ecosystems, persisting in aquatic and terrestrial habitats ranging from remote forests to urban centers (Chanda et al., 2024). Rivers act as major transport pathways for macro and MPs, carrying these contaminants from terrestrial sources to aquatic ecosystems (Schmidt et al., 2017; Gallitelli & Scalici, 2022; Mai et al., 2023). In aquatic and terrestrial environments alike, MPs have been shown to accumulate in the food web, posing a serious threat to biota, ecosystem services, and human health (Altunışık, 2023; Olmo-Gilabert et al., 2024; Swain et al., 2025). Among these threats, literature highlighted that MPs may be ingested by organisms, altering important ecosystem functioning, and threatening their health (Altunışık et al., 2024a,b; Deng et al., 2023; Prata et al., 2021). Notably, the last studies showed that MPs may interact with pollinators, which are directly exposed to these contaminants through their foraging activities (Al Naggar et al., 2024). Pollinators encounter MPs in various environmental matrices, including water sources, soil, and floral nectar, potentially compromising their physiological health and foraging behavior (Bashir et al., 2024). Such disruptions could have cascading effects on pollination efficiency and, consequently, on the stability and resilience of ecosystem services (Christmann, 2019). Among ecosystem services, the impact of MPs on pollination—a critical ecosystem service—is of growing concern. As of global knowledge, pollinators, such as bees, play an essential role in the fertilization of plants, enabling the production of fruit, seeds, and plants that sustain both wildlife and human food systems (Shah et al., 2023). In detail, it is estimated that approximately 75 % of flowering plants and 35 % of global food crops depend on animal-mediated pollination, with an economic value of over 500 billion USD annually (IPBES, 2016; Jordan

* Corresponding authors.
E-mail addresses: abdullahaltunisik@gmail.com (A. Altunışık), luca.gallitelli@uniroma3.it (L. Gallitelli).

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et al., 2021). Given the direct dependence of global food production on pollinator health, even subtle disruptions in bee populations due to emerging contaminants such as MPs could have far-reaching ecological and socioeconomic consequences for global biodiversity and food security (Brunet et al., 2024). It is estimated that over 70 % of the world's most important food crops, including fruits, vegetables, nuts, and coffee, rely on pollination. The decline of pollinators could lead to reduced agricultural yields, increased food prices, and nutritional deficiencies in human diets (Li et al., 2025). Additionally, crops such as almonds, apples, and blueberries, which are heavily dependent on bee pollination, would be significantly affected, threatening global food supply chains (Klein et al., 2007).

Recently, growing attention has been focused on the potential interaction between plastics and pollinators, especially bees (Papa et al. 2022). These insects are crucial for pollination and, being increasingly exposed to environmental pollutants, including MPs, the assessment of bees' health might be of pivotal attention to guarantee the ecosystem service of pollination (Al Naggar et al., 2024). Precisely, the literature showed that MPs can impact bees' health and behavior (Balzani et al., 2022; Pasquini et al., 2024). MPs, often ingested or absorbed by bees during foraging, may lead to a range of adverse effects, from physical harm to disruption of their natural behaviors and colony functioning. Several studies observed the presence of MPs in bees and their products (i.e., honey, beeswax, etc) (Alma et al., 2023; Bashir et al., 2024), however detailed research on the specific implications for pollination and ecosystem services remains limited. In this way, a study conducted in the Sabana de Bogotá, Colombia, characterized MPs in bees from both urban and rural areas, revealing significant contamination levels (Gómez-Méndez et al., 2024). However, knowledge gaps persist regarding the variation in plastic types and their concentration across different ecosystems, as well as the subsequent impact on bee health and pollination efficiency.

Given those gaps, our research seeks to expand upon these findings by investigating the presence and characterization of MPs in honeybees across a broader range of rural and urban habitats. Specifically, we aimed to (1) explore the types, sizes, and abundance of MPs found in bees from two different ecological zones and (2) assess the potential impacts of these MPs on the pollination services provided by honeybees. Given that our study area is located near the Yeşilırmak River, it is essential to consider the role of rivers as potential sources of MP exposure for pollinators. We hypothesize that the concentration and diversity of MPs would be higher in urban urban areas and that this contamination will negatively correlate with the efficiency of honeybees in pollination activities. By addressing these knowledge gaps, our study might contribute valuable insights into the broader environmental and ecological implications of plastic pollution concerning the ecosystem service of pollination and, consequently, food production and security.

2. Materials and methods

2.1. Study area

Çarşamba Plain (Samsun), which is one of the most important delta plains of Turkey where the study was conducted, stands out in terms of crop production. The total area of the plain is 77,756 ha. Of this area, 76 % is used for agriculture. Of the total agricultural area of the plain, 50.4 % is used for orchards and 21.4 % for vegetable gardens. 73 % of all agricultural land in Çarşamba district is considered irrigable land (Çarşamba Agriculture Sector Report, 2018). Two regions, designated as "urban" and "rural," were selected as the study area. To classify the sampling sites as urban or rural, we used a combination of land-use data obtained from publicly available GIS platforms (e.g., Corine Land Cover, https://osmlanduse.org/) and satellite imagery. Sites were categorized as "urban" if they were located near settlements, roads, or industrial/agricultural activity zones, and as "rural" if they were surrounded by natural vegetation with minimal or no visible anthropogenic disturbance. Additionally, previously published environmental studies from the same regions were consulted to support the classification of each site as either pristine or potentially polluted.

The study was conducted in the Çarşamba Plain, an agriculturally important area intersected by the Yeşilırmak River. The urban study sites were located within 0.5–3 km of the river, which is a major hydrological feature of the region. Given that rivers can act as carriers of MPs, it is plausible that some of the contamination observed in honeybees originates from riverine sources, either through atmospheric deposition or direct contact with water and vegetation. The first region comprised six stations (1–6) in the urban zone (Fig. 1), was situated in close proximity to the city center and human settlements (Tatlı et al., 2025a). These sites were 0.5 km from the Yeşilırmak River, which runs through the centre of the city, at the closest, and about 3 km at the furthest. Additionally, these locations were located at a distance of 0.5–1 km from the Çarşamba Solid Waste Landfill (Samsun, Türkiye), the site where the city's garbage is collected and utilized. There is also an organised industrial zone and a sugar factory near this area. This area was selected as a 'urban zone' due to its proximity to the city centre and the waste landfill, and the fact that there is also an industrial zone in the area. Kiwi and hazelnut are produced as agricultural products in this region (Canan et al., 2017). According to local producers in the region, honey is mainly produced from *Castanea sativa*, but also from other plants such as *Tilia cordata*, *Medicago sativa* and *Sambucus nigra* (SBBTarım, 2024).

Conversely, the second region (i.e., rural zone) was characterized by its rural and agricultural setting, and therefore it was selected to serve as a control area. Its distance from industrial enterprises and factories ensured minimal exposure to urban pollutants. This area reflects a natural and largely undisturbed ecosystem, allowing for a comparison to determine the extent to which human activities in the urban zone influence the variables of interest. This region is comprised of locations situated approximately 10 km from the city center. These areas are devoid of industrial enterprises and factories, exhibiting instead a predominantly rural character. The region is also notable for its agricultural activities and beekeeping operations (SBBTarım, 2024).

2.2. Collection of samples and microplastic extraction

Honey bee *Apis mellifera* (Linnaeus, 1758) was used as a model in this study. Honey bees were preferred because of their economic importance and their potential to be used as bioindicators due to their potential to transport MP pollutants (Edo et al., 2021; Rodrigues et al., 2024). Samples were taken between June and September 2024. The locations of the hives and their surroundings are shown in Fig. 1. Zone A (Urban, locations 1–6) is close to the town and the landfill, while zone B (Rural, locations 7–12) is far from the town. The sampling period (June–September) corresponds to the peak flowering and bee foraging season in the region, thereby maximizing environmental exposure. At leeast 20 bees per location were targeted based on previous literature (e.g., Rodrigues et al., 2024), ensuring comparability and statistical robustness. The samples (n = 262) consisted of dead and alive bees from around the hives. The bees were taken shortly after their death into 100 ml glass jars containing 70 % alcohol and kept in this container until analysis, labelled with the location of the bee found.

Samples were removed from the alcohol and weighed to the nearest 0.01 g using an analytical balance (Shimadzu AW 320AW, Japan) after washing with distilled water (Fig. 2).

Samples from each location were transferred to separate flasks and 50 ml H_2O_2 (30 %) was added (Tatlı et al., 2022). The flasks were sealed with aluminum foil and maintained at 65 °C for three days under continuous shaking at 80 rpm. To ensure contamination control, hydrogen peroxide (30 %) was introduced into two flasks from each batch that did not contain bees, serving as 'blank' samples (Taurozzi et al., 2024). After the incubation period, the flasks were allowed to cool to room temperature before their contents were filtered through a

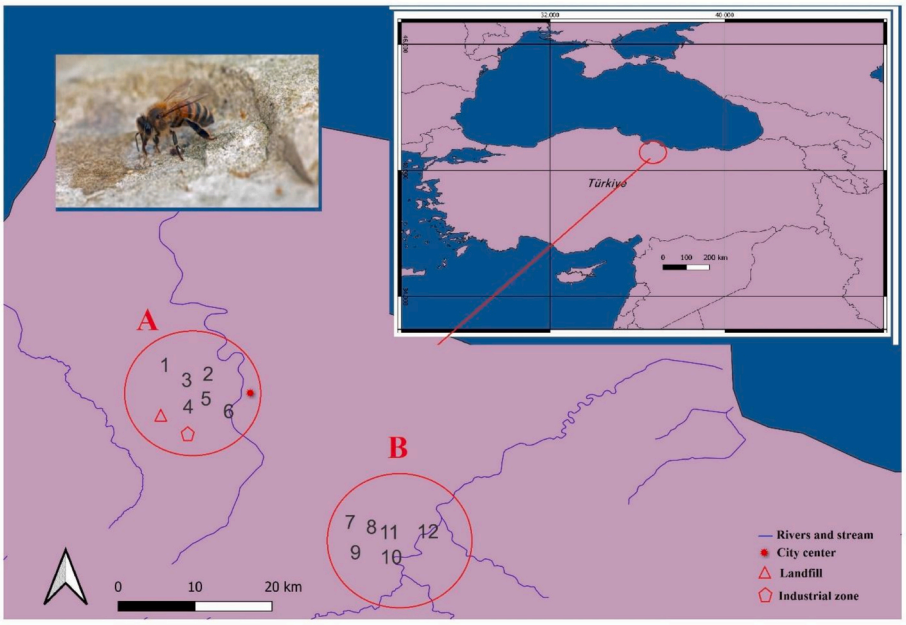


Fig. 1. Sampling stations of *Apis mellifera* in urban and rural zones in the Çarşamba Plain in Türkiye. A: Urban region, B: Rural region.

Whatman GF/C filter using a vacuum pump. The resulting filters were then transferred to glass.

2.3. Contamination control of microplastics

A robust set of measures was implemented during the experimental phase to ensure the highest precision and to reduce the risk of MP contamination (Taurozzi et al., 2024). Among these measures, meticulous maintenance of laboratory hygiene was of paramount importance. Throughout the experiments, the researchers wore cotton lab coats and polymer material-free gloves at all times. Furthermore, all liquids, including dH_2O, H_2O_2, and ethanol, were subjected to rigorous filtration using Whatman GF/C filter paper with a pore size of 1.2 μm and a diameter of 47 mm. In addition, laboratory equipment such as beakers, glass jars, aspirator bottles, flasks, and Petri plates, which are crucial for the sampling process, were carefully cleaned with filtered water and protected with aluminium foil before use (Tatlı et al., 2025b). To ensure reliable detection of MPs and assess potential airborne contamination during microscopy, a separate, sterile Petri plate (blank) filled with appropriately filtered water was placed near the microscope throughout the observation period. Blank samples (n = 5) were processed using the same digestion, filtration, and stereomicroscopic protocols as the bee specimens, resulting in an average of 0.2 ± 0.1 MP particles per blank, indicating low background contamination. Recovery experiments were conducted using pre-weighed microplastic fragments of known polymer types (PET, PE, and PP; 150–212 μm), yielding average recovery rates of 92.8 % for PET, 90.1 % for PE, and 95.6 % for PP, confirming the reliability of the extraction and identification procedures.

2.4. Microscopic examination

A comprehensive examination of filter papers was conducted using a Leica S6D microscope to classify MPs according to the physical characteristics outlined by Hidalgo-Ruz et al. (2012). Particles suspected to be MPs were carefully transferred onto fresh filter papers using a fine needle and documented with a digital camera attached to the microscope. Their morphology was used for classification into fibers, fragments, or other categories. ImageJ software (https://imagej.nih.gov/ij/) was employed to measure particle lengths. Based on those measurements, MPs were grouped into predefined size classes (e.g., 30–100 μm, 101–200 μm, 201–300 etc;), as commonly reported in MP literature (Edo et al., 2021; Rodrigues et al., 2024). From the 93 suspected MPs identified under the stereoscope, a subset of 34 (approximately 36 %) was selected for further analysis. Given that literature usually analyses a subsample of 10–30 % of total sample, we selected more the 30 % to have a reliable and robust analysis (see De Frond et al. 2023; Cowger et al. 2024; Gallitelli et al., 2024a). The selection process was initially based on color and later refined to ensure representation across different provinces and sampling stations. Following the thorough examination of all filter papers, the Petri plate, maintained as a blank control, was subjected to microscopic analysis. If any MPs were identified in the blank sample, a background correction was applied by subtracting the detected MPs from the respective experimental set to account for possible contamination.

2.5. Polymer verification

A PerkinElmer Spectrum 100 FT/IR (Fourier Transform Infrared) spectrophotometer equipped with an Attenuated Total Reflectance (ATR) instrument was used for sample characterization. The FTIR data collection involved 32 scans for each reading, spanning the range of 4000–650 cm^{-1}, with a resolution set at 2.0 cm^{-1} (Fig. 2). Subsequently, the acquired spectra were compared against FT–IR spectra from reference polymer materials and subjected to library searches using Spectrum Search Plus Software (Spectrum Search Plus, ver. 2.0, PerkinElmer; Massachusetts, USA). Only particles showing a match of more than 70 % were deemed MPs, provided the data were aligned with available spectral information in the library (Altunışık, 2025; Tatlı and Altunışık, 2025; Tatlı et al., 2022).

2.6. Data analysis

Normality and variance homogeneity were evaluated using the Shapiro-Wilk and Levene tests, respectively. Given that the data did not follow a normal distribution and the sample size was below 30, non-parametric tests were employed. Unless otherwise specified, all reported mean values are accompanied by their standard deviation (SD) to indicate the variability within the data. MP data are expressed as the number of MPs per individual across different locations. To compare MP prevalence between locations, Mann-Whitney U tests were conducted.

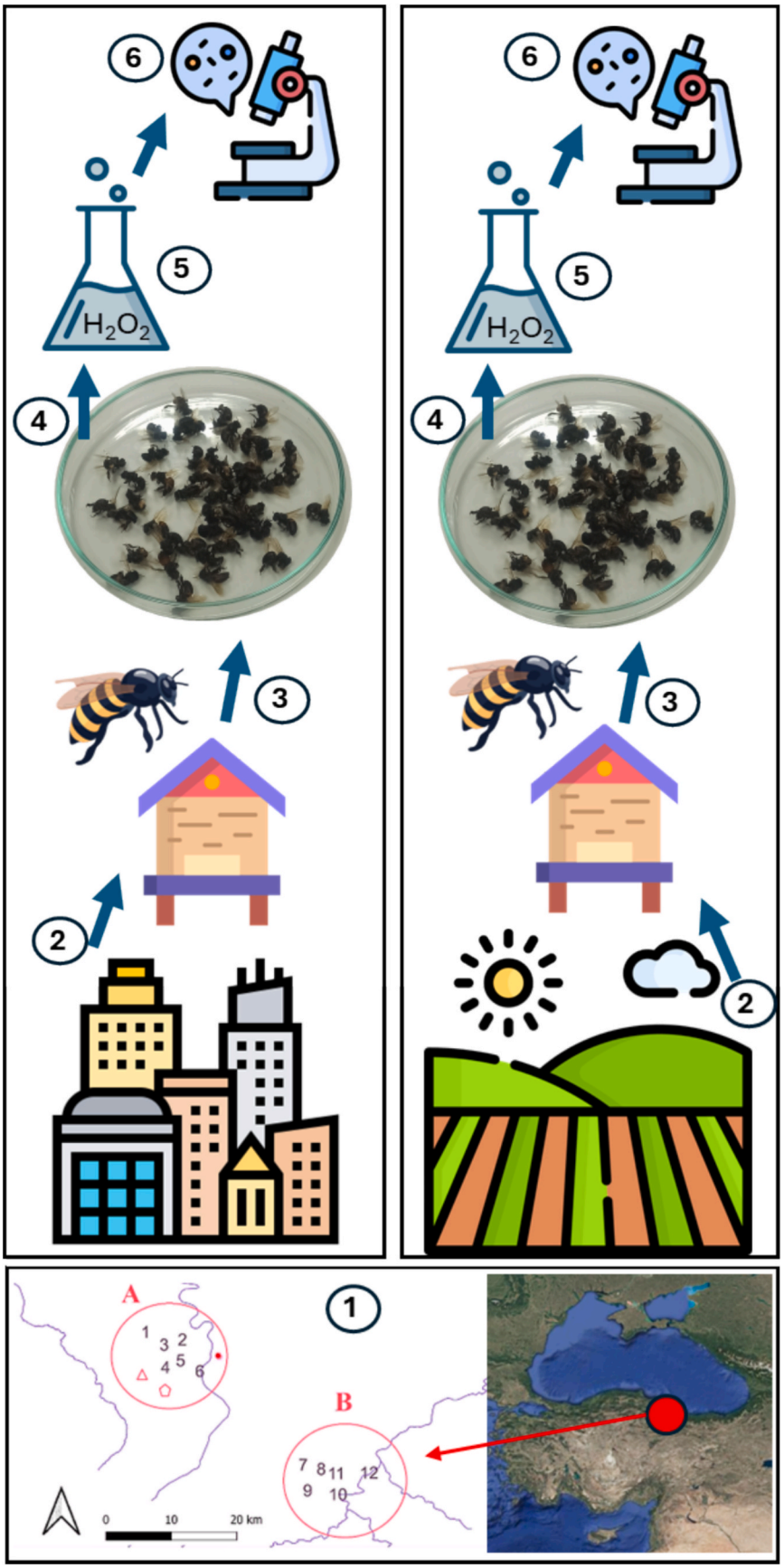


Fig. 2. Schematic overview of MP analysis: honeybee sampling (urban/rural) (1–4), digestion with H_2O_2 (5), and stereomicroscopic examination (6).

The "Health Index" (HI, hereafter) is expressed as the number of the ratio of dead bees (DB) on the total bees (TB) at all stations. Then, this result is subtracted by 1 and provided a value ranging from 0 (i.e., not healthy = dead) to 1 (i.e., healthy = alive), expressed by Equation 1:

$$HI = 1 - \frac{DB}{TB}$$

Additionally, Spearman correlation analysis was performed to examine the relationship between honeybee weight and MP abundance. All statistical analyses were carried out using SPSS version 29.00.

3. Results

3.1. General characterization of microplastics in sampling sites

A total of 93 suspected particles were sub-sampled (n = 34, 36 %). The ATR-FTIR analyzed the 34 particles suspected to be MPs, and 21 of them (61.7 %) were identified as synthetic polymers. Based on their physical characteristics, the MPs were divided into fragments, fibers, and film (Fig. 3). The dominant shapes of MPs were fibers (76 %) followed by fragments (19 %) and film (5 %). In the rural sites, most of the MPs were fibers (75 %, n = 6), while a small number were fragments (25 %, n = 2). On the other hand, the most prevalent type of MPs detected in the urban sites was once again fiber (77 %, n = 10) followed by fragment (15 %, n = 2) and film (8 %, n = 1).

The presence of MPs was confirmed in 11 of the 12 sampling stations for honeybees, with an observed range of 0.00 to 0.30 MPs (mean 0.21 ± 0.03) per individual. In the rural sites, MPs were found in 5 out of 6 stations, and the number of MPs per individual ranged from 0.00 to 0.23 (mean 0.15 ± 0.09). In contrast, urban areas, where MP was detected in all stations, demonstrated a considerably narrower range, varying between 0.22 and 0.30 MPs (mean = 0.28 ± 0.04) per individual. More MPs were detected in the urban area than in the rural area regarding MP abundance per individual (Mann-Whitney *U* test, U = 2.562, p < 0.01).

The detected MPs varied in a range of sizes, spanning from 30 to 720 μm (mean: 228.19 ± 43.10 μm). The sizes of the MPs detected in the rural zone (range: 35–542 μm, mean: 185.50 μm) did not differ from the sizes of the MPs detected in the urban zone (range: 30–720 μm, mean: 254.46 μm) (Mann-Whitney *U* test, p = 0.595).

In the study, five distinct polymer types (Fig. 4) were identified and listed in descending order of percentage of presence as follows: Polyethylene terephthalate (PET, 38 %), Polyethylene (PE, 24 %), Ethylene–vinyl acetate (EVA, 19 %), Polyamide (PA, 14 %), and Polypropylene (PP) at 5 %. In the rural sites, four polymer types were detected in descending order as follows: PET (37 %, n = 3) > PE% (25 %, n = 2) = EVA (25 %, n = 2) > PA (13 %, n = 1). On the other hand, in the urban sites, a total of five distinct polymer types were identified, with the following prevalence in descending order: PET (39 %, n = 5) > PE% (23 %, n = 3) > EVA (15 %, n = 2) = PA (15 %, n = 2) > PP (8 %, n = 1).

These polymers exhibited four distinct color variations. The most prevalent color was blue (76 %), followed by transparent (14 %) and green–red (10 %) tones (Fig. 5). In the rural sites, the colors of the MPs detected in the sampled honeybees were as follows: blue (75 %, n = 6) > red (12.5 %; n = 1) = transparent (12.5 %; n = 1). In the urban sites (see Fig. 6), a total of three different colored MPs were found, with the prevalence of each in descending order as follows: blue (77 %, n = 10) > transparent (15 %; n = 2) > green (8 %; n = 1).

The weight of the honeybees collected from the various locations exhibited a range of 2.14 to 3.76 g, with an average of 2.82 ± 0.14 g. In the rural region, the weights of the samples varied between 2.14 g and 3.18 g (mean: 2.65 ± 0.14). On the other hand, weights ranged from 2.24 g to 3.76 g (mean: 2.97 ± 0.23) for samples taken from the urban region. No correlation was observed between the honeybees' weight and the MP levels per individual (Spearman correlation, rs = 0.37, p = 0.226).

In total, 90.5 % of bees were found dead, while 9.5 % were alive

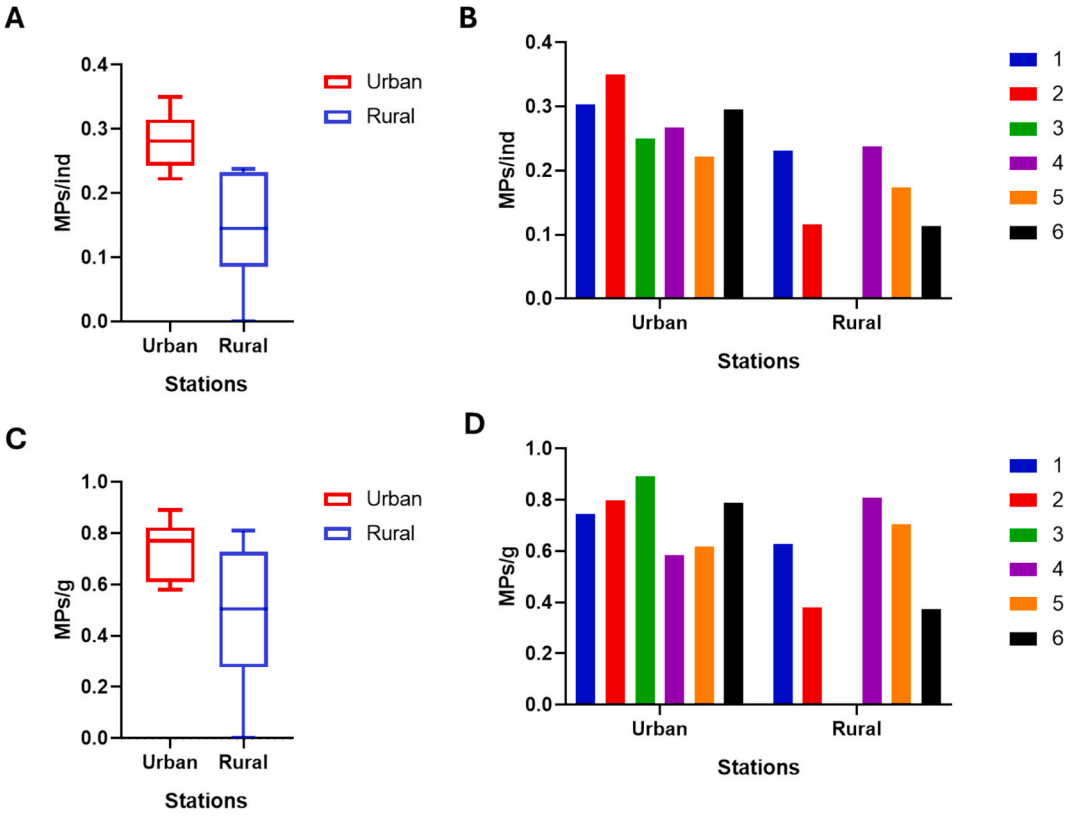


Fig. 3. The concentration of MPs per individual (A and B) and MPs per gram (C and D) is shown here per area (i.e., urban and rural) and station. Boxplots were used with line at mean and standard deviation (min–max).

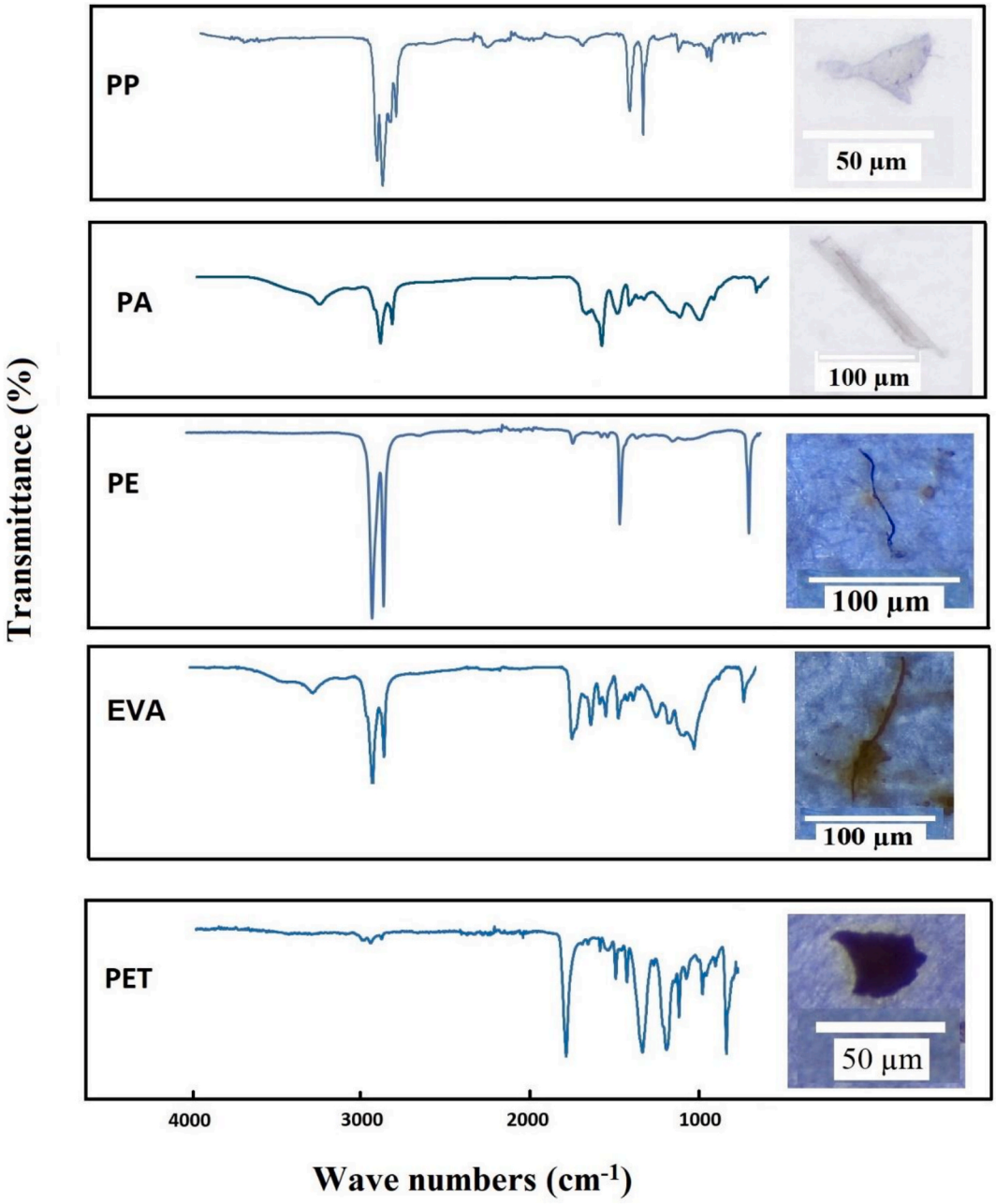


Fig. 4. FTIR spectra and stereomicroscope photographs of microplastics detected in the sampled honeybees.

(Fig. 5). Alive bees were found in urban sites (i.e., 2 and 6) than rural sites (i.e., 7). Although we found a moderate positive Spearman correlation between MP concentration per biomass (g) and the health index (i.e., dead bees) ($rs = 0.54$), the association was not statistically significant ($p = 0.08$), and thus no robust monotonic relationship can be concluded.

4. Discussion

In this study, we provided valuable insights into the presence and characteristics of MPs in honeybees, with significant differences observed between urban and rural areas. ATR-FTIR analysis revealed that 61.7 % of the particles thought to be MPs were synthetic polymers. This implies that MPs are present in the nearby environment and can be consumed by or integrated into the bodies of animals such as honeybees, which could have significant effects on the ecosystem and human health.

4.1. MP concentration in rural and urban zones

The greater abundance of MPs in the urban zone as opposed to the rural zone was a noteworthy trend in our investigation. A statistically significant difference between the average number of MPs per honeybee in contaminated areas (ranging from 0.22 to 0.30 MPs per individual) and rural areas (ranging from 0.00 to 0.23 MPs per individual) has been observed. This supports earlier research that showed a link between environmental pollution and MP accumulation in biota, indicating that anthropogenic pollution has a role in the increased prevalence of MPs in organisms within these sites. Moreover, we should consider that some bees from rural locations did not contain MPs, indicating that honeybees do not consume MPs unless they are found in more concentrated environmental sources, or it may indicate that contamination levels in these areas are relatively low.

Our study confirmed the widespread presence of MPs in honeybees, with higher levels detected in urban areas (0.28 MP/bee) compared to

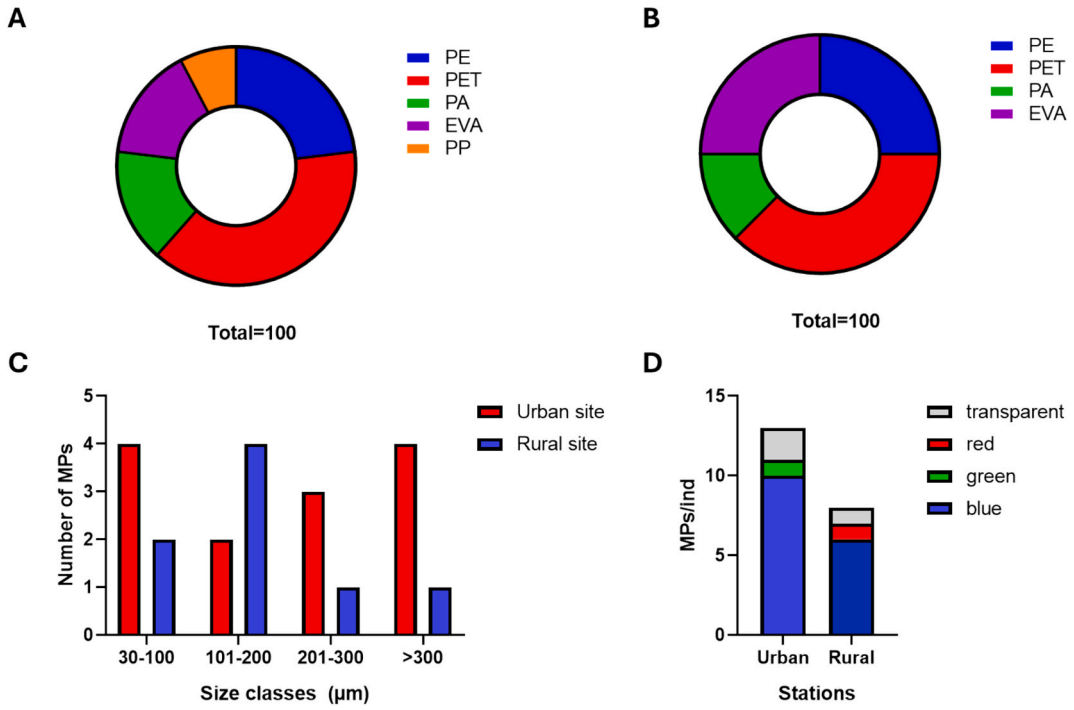


Fig. 5. MP characteristics in honeybees, including polymer types (urban: A; rural: B), size distribution (C), and color composition (D).

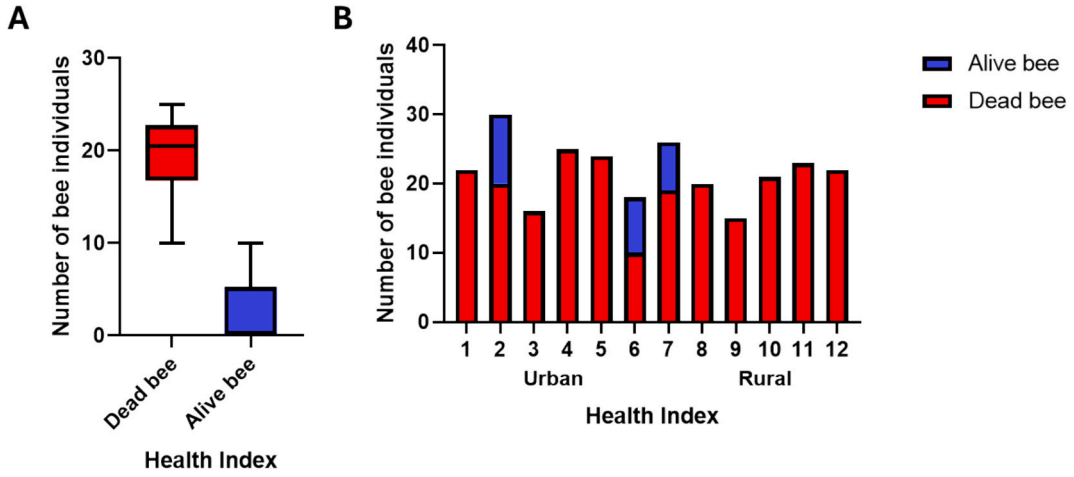


Fig. 6. Mean number of dead and alive bees expressed as "Health Index" in (A) all stations and (B) as the number of bees alive or dead per station.

rural sites (0.15 MP/bee). While MPs were found in most sampling stations, their abundance varied between locations, with more contamination observed in areas exposed to anthropogenic activities. For instance, Edo et al. (2021) reported lower MP burdens per bee, with values around 0.02 MP/bee in urban areas and slightly lower in rural (0.015 MP/bee) and suburban (0.01 MP/bee) zones. In contrast, our study found a higher overall MP load per bee, particularly in urban locations. Rodrigues et al. (2024) reported significant MP contamination in bees from different landscapes in Brazil, detecting a total of 505 MP particles. Given that 40 to 50 worker bees were collected from each of the 12 sampled hives, we estimate that approximately 540 individuals were analyzed. They reported an even greater MP load per bee (~0.94 MP/individual), which is considerably higher than both our study and that of Edo et al. (2021). These variations may be attributed to differences in sampling methods, environmental conditions, and pollution sources at the study sites. Similarly, Rodrigues et al. (2024) found that *peri*-urban and rural areas exhibited considerable MP contamination, suggesting multiple potential sources, including agricultural activities and atmospheric transport. Edo et al. (2021) found no statistically significant difference in MP loads between rural and urban apiaries, attributing this to the extensive foraging range of bees and the influence of wind in dispersing MPs. Our results align with these findings, as MP concentrations were detected in both urban and rural areas, with a higher but not exclusive presence in urban sites. Rodrigues et al. (2024) also reported a surprisingly high MP presence in rural areas, emphasizing the role of agricultural practices and inadequate waste disposal as potential contributors to MP contamination. This is further supported by our findings, which showed that MPs were present in five out of six rural sites and all urban sites, with significantly higher MP loads per bee in urban locations.

4.2. *On the shape, size, and polymers of MPs found in honeybees*

Regarding the shape of MPs, the honeybees ingested mostly fibers

(76 %), followed by pieces (19 %) and films (5 %). Fibers are frequently the most prevalent MP shape in terrestrial and aquatic ecosystems, possibly as a result of their persistence and extensive environmental distribution. The dominance of fibers further supports the hypothesis that textile-related sources contribute significantly to MP pollution in these environments (Edo et al., 2021; Rodrigues et al., 2024). The average size of MPs in the urban zone (mean 254.46 μm) was marginally greater than in the rural zone (mean 185.50 μm), but there were no discernible changes in size distribution between the two zones. These findings are consistent with Edo et al. (2021), who reported MPs in honeybees with a size range of 64 ± 39 μm for fibers and 234 ± 156 μm for fragments. Similarly, a study in China found MPs ranging in size from 1 to 100 μm, with the majority being in the 1–10 μm range. PS particles of 0.5, 5.0, and 50 μm were specifically investigated in an experimental setup, where smaller particles were more likely to accumulate in the gut and translocate to the hemolymph and other tissues (Deng et al., 2021). In addition, Rodrigues et al. (2024) observed MPs within a comparable size range in their study across different Brazilian landscapes. Although MPs may be more prevalent in urban locations, the size range of the particles may not be significantly impacted by the pollution level, according to the lack of size variation between the zones. Regarding the polymers, we mostly found polyethylene terephthalate being the most prevalent, followed by polyethylene, polyamide, ethylene–vinyl acetate, and polypropylene. In the rural sites, four types of polymers were identified, with PET being the most prevalent. This was followed by PE and EVA, which were equally represented, while PA was the least abundant. In contrast, the urban sites showed the presence of five distinct polymer types. Among these, PET remained the most dominant, followed by PE. EVA and PA exhibited equal prevalence, while PP was the least abundant polymer in these sites. A recent study conducted in China found that MPs were present in 66.7 % of honeybee samples, including polystyrene (PS), PE, polycarbonate (PC), and PET (Deng et al., 2021). On the other hand, Rodrigues et al. (2024) conducted their study in Piedade, São Paulo, Brazil, and reported PET as the most dominant polymer, followed by PA and PE, with polyester appearing predominantly in rural areas. Their study attributed the prevalence of PET to its high production volume and widespread environmental presence. Our study observed a similar pattern, with PET being the most frequently identified polymer. This consistency suggests that the dominance of PET in environmental samples is likely linked to its extensive usage and persistence in different ecosystems. The results of the present study align with and extend previous findings in the field (Schiano et al., 2024; Rodrigues et al., 2024), demonstrating that MPs are not only present in urban zones but also reach remote areas through atmospheric pathways (Gómez-Méndez et al., 2024). The predominance of fibers in both urban and rural sites further supports the hypothesis that textile-related emissions significantly contribute to MP pollution. This consistency across multiple studies highlights the urgent need for a broader assessment of airborne MP contamination and its ecological consequences. These findings demonstrate the variety of synthetic polymers found in the environment, which probably corresponds to the MPs frequently found in textiles, packaging, and other goods. PET is a common plastic used in bottles and textiles, and its prevalence in environmental plastic pollution is commensurate with this. In 2024, total plastic raw material production in Türkiye is estimated to be around 2.6 million tonnes, with PET accounting for approximately 52 % of this volume (PAGEV, 2024). This makes PET the most widely produced and consumed plastic in the region, especially in packaging sectors such as beverage bottles and food containers. Despite its higher density, PET fragments—especially in micro- or nano-sized forms—can still become airborne through wind resuspension, human activity, or secondary fragmentation processes (Liro et al., 2024; Leal et al., 2025). The combination of high production volume and the durability of PET increases its likelihood of environmental dispersal and detection in multiple compartments, including air (Thompson et al., 2024). Furthermore, the MPs' color distribution—blue, transparent, and green–red—highlights the possible origins of these particles, with blue polymers frequently connected to textiles and packaging. Finding particular polymers in the honeybees can provide important information on the types of environmental pollutants in the regions under study. It may also reveal the routes by which these MPs enter the food chain.

4.3. Ecological implications

In this study, we did not find a significant relationship between honeybee weight and the number of MPs per individual. This implies that the honeybees' weight or size has no bearing on the MPs found in them, which would have been predicted if MPs were disproportionately building up in larger or smaller individuals. The generalist aspect of honeybees' foraging behavior, or the fact that MPs are distributed so that their consumption is not always connected to specific bee traits, is a possible explanation for the lack of association. Understanding the wider ecological effects would benefit from more research on elements including bee behavior, environmental MP distribution, and the possibility that MPs could influence bee development or health. Although weight did not correlate with MP burden, the overall presence of MPs in honeybees suggests that these organisms are still effective bioindicators of environmental contamination. The impact of MP contamination on honeybee health is a crucial aspect that requires further investigation. Experimental research in China demonstrated that ingestion of PS MPs led to physiological stress in honeybees, including damage to midgut tissues and increased susceptibility to viral infections such as Israeli Acute Paralysis Virus (IAPV) (Deng et al., 2021). Smaller MPs (0.5 μm) were more likely to translocate beyond the gut, leading to potential systemic effects. These findings highlight the need for further research into the impact of MPs on honeybee immune responses and overall colony health. Regarding ecological implications, some studies (e.g., Al Naggar et al., 2023) have proposed health indices to assess the physiological and ecological effects of MP exposure on pollinators. However, neither Rodrigues et al. (2024) nor Edo et al. (2021) explicitly addressed a Health Index for honeybees in their assessments. Future research should aim to integrate health indicators such as oxidative stress levels, immune responses, and behavioral changes to understand better the broader implications of MP contamination on honeybee populations.

These findings suggest that honeybees are exposed to MPs during their foraging activities, either through atmospheric deposition, contact with contaminated plant surfaces, or ingestion of contaminated resources such as water and nectar. The presence of MPs within the digestive tracts of bees indicates that ingestion is a significant route of exposure. Additionally, differences in MP abundance across landscapes imply varying levels of environmental contamination, potentially driven by differences in urbanization, agricultural practices, and prevailing atmospheric transport patterns (Hinsch et al., 2024; Lin et al., 2024; Shavali-Gilani et al., 2025). The observed discrepancies between studies could also be influenced by regional differences in land use, atmospheric MP transport, and the specific industries present in each study location.

Our findings confirm the presence of MPs in honeybees from both urban and rural environments, with a higher concentration in urban sites. Considering that the study area is located near the Yeşilırmak River, it is worth discussing the potential contribution of riverine transport to MP contamination. Previous studies have shown that rivers serve as key conduits for MPs, transporting them from industrial, urban, and agricultural sources (Wagner et al., 2014; Blettler et al., 2018; Gallitelli et al., 2024b; Xia et al., 2025). While some MPs reach the marine environment, many remain suspended or settle in freshwater ecosystems, where they may become available to terrestrial organisms, including pollinators (Azevedo-Santos et al., 2021; Gallitelli and Scalici, 2023). These results highlight the need for further research into the role of freshwater systems in MP transport and potential availability for pollinators (Gallitelli and Scalici, 2023; Sheng et al., 2024), particularly in regions with high agricultural and industrial activities. Furthermore, future research needs to focus on exploring the long-term effects of MP

on bee health to prevent from reduction of the pollination ecosystem service.

4.4. Limitations

This study has several limitations that should be acknowledged. First, while the sample size provided a foundational understanding of microplastic contamination in honeybees, a broader sampling effort across additional locations and seasons would improve representativeness. Second, although our findings document the presence and characteristics of microplastics, we did not assess physiological or behavioral impacts on the bees. Lastly, it is important to note that our methodology may not efficiently detect microplastic particles smaller than 30 μm. The identification of such small-sized particles typically requires more advanced techniques, such as micro-FTIR spectroscopy or pyrolysis-GC/MS, which were beyond the scope of this study. Future studies incorporating biomarkers, histopathological analyses, or controlled exposure experiments are essential for understanding health consequences. Finally, our results reflect local conditions within the Çarşamba Plain; expanding the study to different ecological zones would allow for a more comprehensive assessment of regional variation in microplastic exposure.

5. Conclusions

This study provides valuable data on the presence and characteristics of MPs in honeybees, highlighting differences in MP abundance between urban and rural zones and offering insight into the polymer types and physical characteristics of these particles. The consistent detection of MPs across diverse environments in our study highlights the complex interplay of urbanisation, agricultural activities, and atmospheric dynamics in MP distribution. The necessity of ongoing monitoring of MP contamination in ecosystems and its possible impacts on wildlife health is further supported by the strong correlation found between pollution levels and MP abundance. Given the role of honeybees in pollination, understanding the pathways through which MPs enter their systems is pivotal to evaluating the wider broader ecological and environmental risks associated with MP pollution. These findings highlight the potential of honeybees as effective bioindicators for monitoring microplastic contamination in terrestrial environments. Policymakers should consider implementing stricter regulations on plastic waste management, especially near agricultural landscapes and urban settlements, to reduce environmental exposure. From an agricultural perspective, chronic MP contamination in apiaries situated near urban or industrial areas may translate into measurable declines in pollination-dependent crop productivity over time, with knock-on effects on farm incomes and rural livelihoods. For ecosystems, shifts in pollinator behavior or population dynamics could destabilise plant–pollinator networks, reducing biodiversity and resilience to other stressors (e.g., pesticides, pathogens, habitat loss). We recommend integrating MP monitoring into apicultural surveillance, guiding apiary siting and management to minimize exposure, and testing mitigation measures—such as buffer strips and water filtration—to limit MP ingress into foraging areas. Further research are needed to explore the potential effects of MPs on honeybees and other pollinators, as well as the implications for food security and ecosystem health.

CRediT authorship contribution statement

Abdullah Altunışık: Writing – review & editing, Writing – original draft, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Damià Barcelò:** Writing – review & editing, Validation, Supervision. **Luca Gallitelli:** Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Software, Conceptualization.

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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Data availability

No data was used for the research described in the article.

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