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Role of Biodegradable and Non-Biodegradable Microplastic in Modulating the toxicological Effects of Organic Pollutants in the Soil Organism *Folsomia candida*

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Abstract The ecotoxicological effects of microplastics in soil ecosystems are complex, particularly in areas of intensive agriculture and livestock production, where plant protection products and veterinary drugs commonly coexist with plastic residues. In this study, we investigated the impact, under laboratory conditions, of 3 MP types (non-biodegradable low-density polyethylene (LDPE) and biodegradable polybutylene adipate terephthalate-based (PBAT-based) and a starch-based polymer) on the soil-dwelling species *Folsomia candida* (Willem, 1902) in soils contaminated with the anthelmintic albendazole and the fungicide pyraclostrobin. These organic pollutants (OPs) are frequently found in areas of intensive agriculture and livestock production. *F. candida* individuals were exposed for 28 days to soils contaminated by the OPs at 0.0001 w/w% (1 mg/kg), with and without MPs at 0.01 and 0.1 w/w% concentrations (100 and 1000 mg/kg respectively), under laboratory conditions (21 ± 1 C°, $80\% \pm 1$ RH). Adults' survival, egg production, and juveniles' occurrence were recorded as endpoints. Our findings indicate that microplastics alone did not significantly affect the survival and reproductive outcomes of *F. candida*. However, in soils contaminated with albendazole and pyraclostrobin, the presence of biodegradable MPs resulted in significant effects compared to the control and the treatment with only microplastics. Specifically, PBAT-based MPs significantly impacted adult survival, juvenile occurrence, and egg counts, while starch-based MPs primarily affected egg counts. On the contrary, co-exposure to OPs and LDPE MPs did not show significant effects. These results suggest that different MPs influence the bioavailability and toxicity of co-occurring fungicides and veterinary drug in soil ecosystems in different ways, with implications for assessing the ecological risks of biodegradable and non-biodegradable plastics in contaminated soils. The potential of MPs to influence the spatial distribution and bioavailability of organic pollutants for soil mesofauna needs further investigation.

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1 Introduction

Over recent decades, the utilization of plastics has significantly advanced public health, safety, energy efficiency, and food preservation. However, their extensive use has concurrently introduced a major environmental challenge, with plastic pollution exerting substantial impacts on both aquatic and terrestrial ecosystems. In soil environments, plastic contamination originates from numerous sources, including sewage sludge, compost, irrigation practices, plastic mulching, contaminated organic fertilizers, street runoff, atmospheric deposition, landfills, and tire wear (Kumar et al., 2020; Meng et al., 2021; Nizzetto et al., 2016). Plastics degrade over time through processes like exposure to UV radiation, mechanical wear, and chemical reactions such as hydrolysis, forming smaller particles, known as microplastics (MPs). Due to their small size and high surface-area-to-volume ratio, these particles can adsorb and vector other pollutants (Brennecke et al., 2016; Camacho et al., 2019; Gigault et al., 2021; Guo et al., 2022; Li et al., 2018; Torres et al., 2021; Wang & Wang, 2018; Zhu et al., 2022). Absorption occurs through chemical and physical interactions (Fred-Ahmadu et al., 2020; Torres et al., 2021; Yu et al., 2019). For example, hydrophobic polymers such as polyethylene (PE) and polystyrene (PS) have strong affinity for non-polar compounds, facilitating their accumulation in the environment; electrostatic forces can enhance the sorption of ionic or polar contaminants in more hydrophilic polymers; porous structures of certain polymers can trap contaminants within their pores; and Van der Waals forces and π-π interactions can facilitate the sorption of organic contaminants, especially those with aromatic rings. Interaction between MPs and contaminants can be also influenced by environmental factors. For example, weathering processes can alter the surface properties of microplastics, potentially increasing their reactivity and sorption capacity (Torres et al., 2021). Such interactions can affect the distribution and bioavailability of chemical contaminants in the environment, raising concerns about their ecological and health impacts (Chen et al., 2023; Hüffer et al., 2019; Jiang et al., 2020; Shang et al., 2024; Tan et al., 2022; Wan et al., 2022; Zhang et al., 2023; Zuo et al., 2019). The interaction between MPs and organic pollutants is also influenced by soil properties such as porosity, organic matter content, and pH which may affect the sorption, transport and distribution of MPs and associated pollutants in the soil (Scott-Fordsmand et al., 2024; Zhang et al., 2024).In areas of intensive agriculture and livestock production, plant protection products, such as pesticides and herbicides, and veterinary drugs, like antibiotics and anthelmintics respectively, are frequently applied to enhance productivity and the control of pests and diseases. These chemicals commonly coexist and interact with microplastics in agricultural soils (Kumar et al., 2020; Nizzetto et al., 2016; Okoffo et al., 2021; Piehl et al., 2018). Organic fertilizer is produced in large quantities by the European Union (EU), and 1.4 billion tonnes of manure from livestock animals were produced annually in the period 2016–2019 in the EU27 and UK (Eurostat, 2021).

Plastic residues may derive from both non-biodegradable and biodegradable polymers. One of the most widely used conventional polymers is low-density polyethylene (LDPE), produced in large quantities to make greenhouse plastic, stretch, mulching films, and miniature tunnels (Lwanga et al., 2022).

Recently bioplastics, also known as biodegradable plastics, have drawn a lot of attention as a promising substitute for traditional non-biodegradable polymers (mostly petroleum-based plastics) (Haider et al., 2019). In agriculture they are being used for the production of mulch films, plant pot, wood-plastic composites, composites with natural fibres, agriculture fibres, food packaging, and seed coatings. (Shruti & Kutralam-Muniasamy, 2019). Common bioplastics comprise biodegradable polylactic acid (PLA), polyhydroxybutyric acid (PHB), polybutadiene and starch blends, one of the emerging biodegradable material compositions employed for the production of mulching films (van Loon et al., 2024a). Biodegradable plastics are designed to decompose more easily than traditional ones but may still pose environmental risks during their degradation process (Li et al., 2024). The ability of degradable plastics to absorb contaminants is also higher than that of non-degradable plastics (Torres et al., 2021).

We investigated, under laboratory conditions, the impact of three types of MPs: non-biodegradable low-density polyethylene (LDPE) and two biodegradable polymers, polybutylene adipate terephthalate-based (PBAT-based) and a starch-based polymer, on the soil-dwelling species *Folsomia candida* (Willem, 1902), in soils contaminated with the anthelmintic compound albendazole (ABZ) and the fungicide pyraclostrobin (PYR). These two pollutants are frequently found in soils adjacent to areas of intensive agriculture and livestock production (Geissen et al., 2021; Navrátilová et al., 2021, 2023). In particular, ABZ is excreted in livestock urine or faeces, thus it can enter soil ecosystems through the application of contaminated manure (Navrátilová et al., 2021). PYR is a broad-spectrum strobilurin fungicide, widely used in agriculture to control a variety of fungal diseases in crops (Li et al., 2024). In 2014, PYR was the second best-selling fungicide among SFs, following Azoxystrobin, (Mao et al., 2020), and is mainly applied to wheat, soybean, and corn cultivation (Kumar et al., 2020).

Till now ABZ and PYR toxicity have been tested on a range of soil organisms, including macro and mesofauna, where both the compounds showed harmful effects. For example, under laboratory conditions PYR disrupts the antioxidant defence system of the earthworm *Eisenia foetida*, leading to oxidative stress and DNA damage (Ma et al., 2019) and in *Enchytraeus crypticus* it exhibits significant toxicity (Kovačević et al., 2021). Reproductive effects, weight decrease and alterations in key enzyme activities have been demonstrated for *E. foetida* exposed to ABZ under laboratory conditions (Gao et al., 2007a, 2007b, 2013, 2015).

Data on the toxicity of ABZ and PYR on soil mesofauna (e.g., collembola, nematodes, mites, etc.) are limited. Springtails (Collembola) are the most abundant soil-dwelling insects, contributing to organic matter decomposition, nutrient cycling, soil fertility, energy flow, and regulation of soil microbial communities (Bhagawati et al., 2021; Ju et al., 2019; Potapov et al., 2023). The decline of collembola populations is considered a signal of broader ecological risks, affecting soil health and the organisms' dependent on soil ecosystems (Fiera, 2009; Potapov et al., 2020). Due to their high sensitivity and rapid response to environmental pollutants, springtails are recognized as standard test organisms for use in ecotoxicological assessments (OECD 2016; ISO 2011, 2014). *F. candida*, in particular, is widely employed as a model species in soil ecotoxicity studies, with endpoints such as survival, growth, behaviour, and reproduction serving as reliable indicators of soil contamination (Crouau & Pinelli, 2008; Jager et al., 2007). Preliminary findings suggest that ABZ can impair *F. candida* reproduction at environmentally relevant doses (Forestieri et al., 2023). Indeed, ABZ and its metabolite, ABZ sulfone, which is readily formed in soils, can target the endosymbiont *Wolbachia*, an obligate symbiont essential for reproduction in *F. candida* (Forestieri et al., 2023; Negri, 2012; Serbus et al., 2012). Studies on the effects of commercially available products containing PYR on *F. candida* have shown both lethal and sub-lethal impacts (Giordani et al., 2020). Additionally, exposure of *F. candida* to other strobilurin fungicides at high doses has been linked to reproductive alterations (Kovačević et al., 2023).

In the present work, we explored the effects of three MP types, one non-biodegradable (low-density polyethylene—LDPE) and two biodegradables (polybutylene adipate terephthalate – PBAT- based and Starch-based), alone and in combination with ABZ and PYR, on the springtail *F. candida*.

Low-Density Polyethylene (LDPE) is extensively used in agricultural settings, including in mulch films, irrigation systems, and greenhouse coverings. Due to its high resistance to degradation, LDPE persists in soil for extended periods, contributing to long-term environmental contamination (Ohtake et al., 1998). Over time, LDPE fragments can accumulate, and in regions where crop cultivation overlaps with livestock farming, LDPE plastic residues may co-occur with pesticides and veterinary medicines for long time. PBAT-based and Starch-based polymers are biodegradable and decompose more easily than LDPE, but may pose environmental risks in soils during their degradation process (Li et al., 2024; Shang et al., 2024; Zhou et al., 2024). Furthermore, biodegradable MPs may act as more effective carriers for organic pollutants compared to non-biodegradable ones, increasing pollutant retention, bioavailability, and toxicity through prolonged exposure for living organisms (Jiang et al., 2020; Zhang et al., 2023; Zuo et al., 2019).

This study aims to enhance our understanding of how MPs, veterinary drugs, and pesticides affect soil ecosystems, providing valuable insights into the ecological risks linked to their presence and exposure.

2 Materials and Methods

2.1 Test Organisms

F. candida (Collembola, Isotomidae) was reared at Università Cattolica del Sacro Cuore. The cultures were housed in a box (capacity 1,2 L) with moistened plaster of Paris and graphite (9:1 w/w) as the substrate (2 cm deep) at 20 ± 1 °C and 80% relative humidity. As a feeding supply, dried granulated beaker yeast was added twice a week. Adults of *F. candida* were moved from the culture to Petri dishes to deposit eggs to produce synchronized juveniles. To allow them to hatch, the eggs were kept apart from the adults at 20 ± 1 °C and 80% relative humidity in the dark. According to ISO guidelines, synchronized juveniles between the ages of 10 and 12 days were used for the tests (11267, 2014).

2.2 Test Chemicals and Soil

LDPE, PBAT-based, and starch-based MPs were tested in the present study. Data on particle size distribution were provided by the Wageningen University and Research, in the frame of the project MINAGRIS—MIcro- and NAno-Plastics in AGRIcultural Soils: sources, environmental fate and impacts on ecosystem services and overall sustainability. MPs were obtained from plastic films (30 µm thick for LDPE, and 15 µm for PBAT and starch-based), which were cryomilled and sieved as described in Meng et al. (2024). The size distribution of the fractions used in our study was characterized through Mastersizer 3000 particle size analyser from Malvern Instruments Ltd. with a dry powder dispersion unit, measured according to Dris et al. (2015). MPs size distribution was between 271 ± 2.4 µm (10th percentile) and 780 ± 14.7 µm (90th percentile), with a median mean diameter of approximatively 466 ± 4.4 µm and a density of 0.937 g cm $^{-3}$ for LDPE; 245 ± 2.3 µm (10th percentile) and 752 ± 2.6 µm (90th percentile), with a median mean diameter of approximatively 439 ± 2.0 µm and a density of 1.449 g cm $^{-3}$ for PBAT-based; 210 ± 6.8 µm (10th percentile) and 733 ± 11.6 µm (90th percentile), with a median mean diameter of approximatively 405 ± 7.5 µm and a density of 1.276 g cm $^{-3}$ for starch—based.

ABZ (CAS 54965–21-8), PYR (CAS 175013–18-0) and ethanol (CAS 64–17-5) were purchased from Sigma Aldrich with a purity ≥ 98% for ABZ and PYR and a purity ≥ 99.5% for ethanol. All the experiments were performed using soil containing no detectable levels of MPs, which was collected from the fields of Unifarm, Wageningen University & Research (the Netherlands). The control soil was air dried and then passed through a metal sieve (2 mm). It was sandy loam soil (3.2% clay, 50.0% silt and 46.8% sand, with pH of 6.2 (1:5, w/v, extracted with water) and organic matter content of 3.8% (loss on ignition).

2.3 Test Soil Preparation

To identify possible toxicological effects of MPs on *F. candida*, LDPE, PBAT-based, and starch-based MPs were mixed to control dry soil at final concentrations of 0.1% (w/w%) (1000 mg/kg) and 0.01% (w/w%) (100 mg/kg) per MPs type. These concentrations are within the MPs range of 0.0004–28% w/w, which has been used in tests with soil invertebrates (Huerta Lwanga et al., 2016, 2017; Kwak et al., 2024; Rodríguez-Seijo et al., 2018; Vaccari et al., 2022; Zhu et al., 2018). The plastic size tested falls within the range tested on soil invertebrates, namely 0.02–1400 µm (Vaccari et al., 2022). This mixture was left to stabilize for three days before adding *F. candida* individuals.

To test the toxicity of organic pollutants (OPs) plus MPs, soil treated with the higher concentration of plastics, 0.1% (w/w%) (1000 mg/kg), was chosen. This mixture was left to stabilize for three days before adding PYR and ABZ, as an ethanol solution, to achieve a nominal soil concentration of 0.0001% (w/w%) (1 mg/kg) per each pollutant. Dry soil was treated with the ABZ and PYR solutions in Petri dishes, then left under a fume hood for 24 h to allow ethanol evaporation. The treated soil was transferred to Petri dishes (5 g/each, three replicates) containing plaster of Paris and graphite (10:1), 0.5 cm deep, and distilled water was added to reach 60% water holding capacity. LDPE, PBAT-based, and Starch-based MPs untreated soils (three replicates each) were used as negative controls. Preliminary experiments with solvent (ethanol) control (4 mL) showed no significant differences from untreated soils. For ABZ, the nominal concentration used is within the range often reported in agricultural soils and it matches with the expected exposure scenario where the agricultural soils are amended regularly with contaminated manure or employed as contaminated manure dumping sites (Gkimprixi et al., 2023). The PYR concentration was selected based on data regarding residues found in agricultural soils following fungicide application for fungal disease management. Studies indicate that PYR residues show concentrations ranging from 0.05 to 1.04 mg/kg, with other reports indicating levels up to 3.153 mg/kg (Han et al., 2022; Wang et al., 2018).

2.4 MPs Exposure Tests and MPs-OPs Co-Exposure Tests

The ecotoxicological tests were performed following the ISO guidelines 11,267 (2014) for collembola reproduction test in soil with some modifications. Three replicates were performed for each treatment, and three specific controls were performed for each set of three replicates. 2.5 mg of granules of dehydrated baker's yeast were added in random points of the test soil surface as food source for Collembola. In accordance with ISO 11267 guidelines (2014), ten 10–12 age-synchronized *F. candida* juveniles were used for the experiments. The test Petri dishes were in a climate chamber with 21 ± 1 C °and 80% ± 1 relative air humidity, under 12–12 light dark. Two times a week, distilled water was added to maintain the moisture and some granules of dehydrated baker's yeast was inserted as food source.

After the exposure period (28 days), using an excess of water the contents of each Petri dish were carefully stirred with a small spatula and transferred to a larger container to allow floatation of springtails. The number of adults, juveniles and eggs was recorded.

As exposure tests with the three types of MPs (LDPE, PBAT, and Starch-based) at concentrations of 0.01% and 0.1% (w/w%) showed no significant differences compared to the control, subsequent tests with MPs and OPs were conducted using only the higher 0.1% plastic concentration.

2.5 Data Analysis

The experimental data were analysed using R software for statistical analysis. Two types of models were explored: Linear Mixed-Effect Models (LMMs) and Generalized Linear Mixed-Effect Models (GLMMs). These types of models are particularly suited for non-independent data and incorporate both fixed and random effects that may influence the data (Bryk & Raudenbush, 1992). In R, the models were constructed using the *glmmTMB* and *lme4* packages (Bates et al., 2015; Brooks et al., 2017). Three models were tested:

a. Modelling of MP Effects Only: The counts of adults, juveniles, and eggs were modelled separately, with each specific count (adults, juveniles, or eggs) as the response variable. The test type (either LDPE, PBAT, Starch, or Control) and concentration levels were included as fixed effects, while batch number was treated as a random effect. In this case, batch numbers ranged from 1 to 6, with the first three batches corresponding to tests conducted at an MP concentration of 0.1% and the remaining three at 0.01%.
b. Modelling of Combined MPs-OPs Effects: The counts of adults, juveniles, and eggs were again modelled separately, with each specific count (adults, juveniles, or eggs) as the response variable. The test type was included as a fixed effect (either LDPE-OPs, PBAT-OPs, Starch based-OPs, or control), and batch number was treated as a random effect. Here, batch numbers ranged from 1 to 3, as the concentration was fixed at 0.1% for both MPs and OPs.
c. Modelling the Effects of OPs Versus MPs: In this case, count data from both experiments were merged into a larger dataset to assess the impact of OP application compared to the presence of MPs alone, using data with a concentration of 0.1% and batch numbers from 1 to 3. Counts of adults, juveniles, and eggs were modelled separately, with each specific count as the response variable. The test type was a fixed effect (either LDPE, PBAT, Starch, LDPE-OPs, PBAT-OPs, Starch based-OPs, or Control), while the presence or absence of OPs was treated as a random effect.

Notably, all control groups were consolidated into a single test category.

To assess the goodness of the models fit on the data, statistical tools for diagnostics of residuals and evaluation of pairwise significance were adopted: (i) a quartile-quartile plot (QQ plot), (ii) a DHARMa residuals plot, and (iii) a plot of model's predictions for the specific variable subject of the test [computed using DHARMa, performance, and emmeans R packages (Hartig, 2024; Lenth, 2024; Lüdecke et al., 2021)].

The QQ plot is useful to visualize residuals deviations from the expected distribution, the data dispersion, and the presence of outliers. When computing the plot, the DHARMa package runs three tests: (i) a dispersion test, (ii) a Kolmogorov–Smirnov (KS) test (Massey, 1951) to assess the correctness of the applied distribution in the model, and (iii) an outliers' test. For each test, a p-value is provided and written in the corresponding graph for quick reference.

The DHARMa residuals plot represents the residuals plotted against the predicted value for each quantile (25%, 50%, 75%) and, if the simulation produces outliers, the graph will highlight them in red colour. The model's prediction plot represents the mean prediction as a solid dot and its 95% confidence intervals (CIs) as vertical bars. Moreover, statistical significance of the results was analysed by conducting a pairwise test with post-hoc Tukey analysis, with α set at 0.05.

3 Results

All validity criteria outlined in the ISO 11267 test guideline for chemical testing with Collembola (ISO, 2014) were satisfied. In particular, control survival exceeded 80% for exposures to both MPs alone and MPs combined with OPs, over 100 juveniles were produced in each control test, and the coefficient of variation in juvenile counts within the controls remained below 30% (Fig. 1).

3.1 Effects of MPs

Figure 1 shows the number of survived adults (Fig. 1a), juveniles (Fig. 1c) and eggs (Fig. 1e) of *F. candida* counted after 28 days of exposure to LDPE, PBAT-based, and Starch-based MPs at concentrations of 0.01% and 0.1% (w/w%), respect to controls (plastic-free soil). No significant effect can be observed by varying concentrations with respect to the controls.

The models developed to analyse count data for each sample type (adults, juveniles, and eggs) demonstrate a good distribution of residuals, and the DHARMa residual diagnostics indicate no concerns (Figure S1. Supplementary Materials).

The model predictions for the three populations (adults, juveniles, and eggs) are shown in Fig. 2. In general, statistical significance is not reached for any group and the effects of the two concentrations seems generally the same, with a reduction in adult and egg counts for 0.1% concentration.

3.2 Combined Effects of MPs and OPs

Figure 1 shows the counts for the adult (b), juvenile (d), and egg (f) populations of *F. candida* from the experiment on co-exposure to MPs and OPs, paired with the corresponding counts from the previous experiment (MPs only) to better visualize the impact.

Consistent with the previous findings, the models trained to analyse count data for the samples show well-distributed residuals, and DHARMa residuals diagnostics indicate no concerns for the counts of juveniles and eggs (Figure S3. Supplementary Materials). The only exception being the model for adult counts, for which DHARMa residual diagnostics highlight quantile deviations. However, this is not cause of concerns since the residuals histogram and quantile–quantile plot are excellent (Figure S2. Supplementary Materials).

Considering the model predictions for the adult population (Fig. 3a) and juvenile population (Fig. 3b), the prediction plots indicate a significant negative effect of the PBAT-OPs combination group compared to the control group. In the case of egg count (Fig. 3c), the model predictions indicate a negative effect for both PBAT based-OPs and Starch based-OPs groups relative to controls, both reaching statistical significance.

3.3 Comparative Effects of MPs vs MPs and OPs

The models trained to analyse count data for adults and juveniles demonstrate well-distributed residuals (Figure S4. Supplementary Materials), but the DHARMa residuals diagnostics indicate quantile

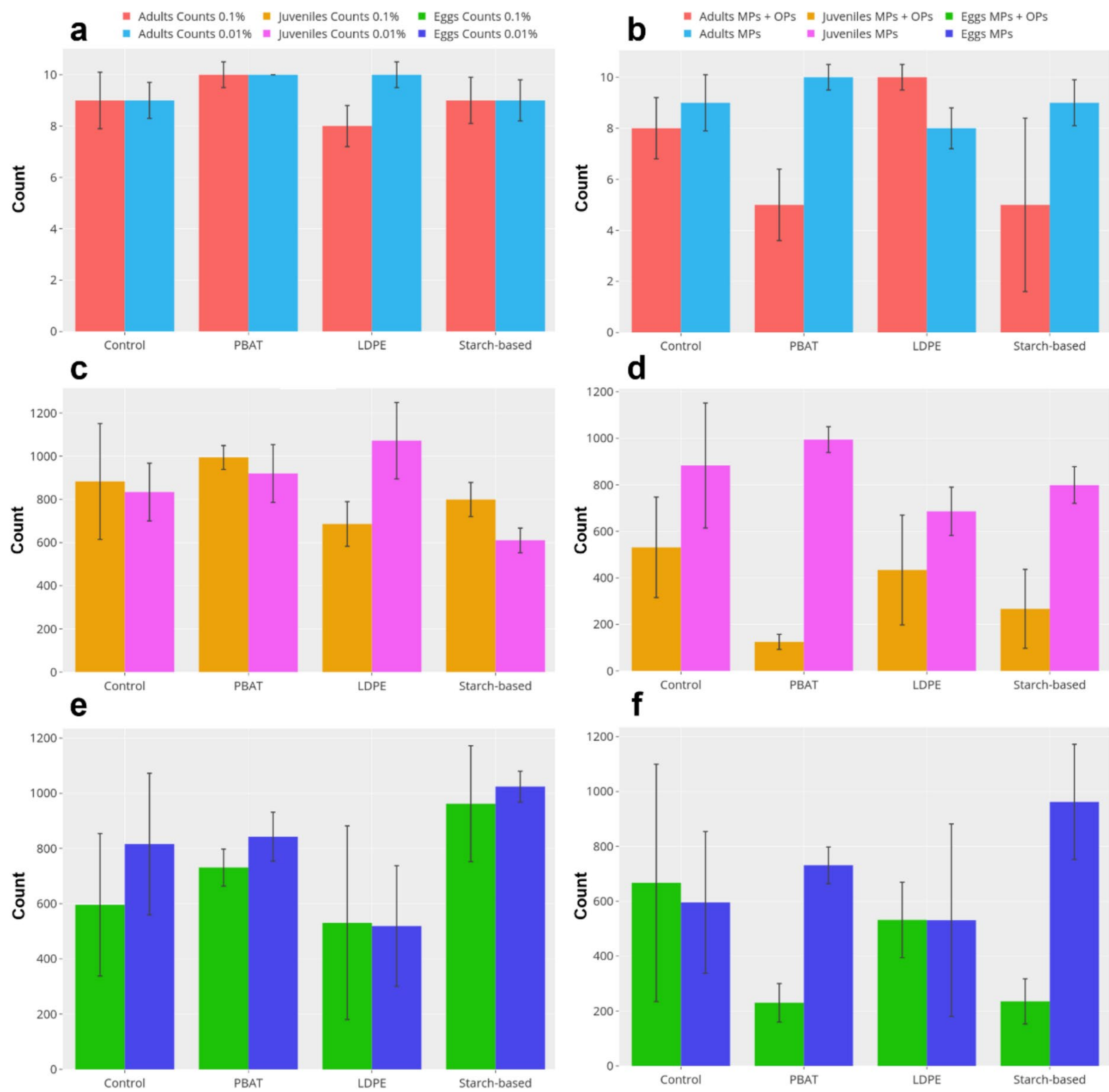


Fig. 1 (a, c, e) Bar plots of *F. candida* adults (**a**), juveniles (**c**), and eggs (**e**) counted at the end of the first experiment (MPs only) for the two concentrations. Data are grouped according to test type (control batch number and MP tested) and according to concentration (0.1 and 0.01 w/w%). (b, d, f) Bar plot of *F. candida* adults (**b**), juveniles (**d**), and eggs (**f**) counted at the end of the MPs only experiment compared with the combined MPs-OPs experiment. Data are grouped according to the applied MPs (PBAT, LDPE, and Starch-based). Bars indicate standard errors

deviations for the egg count (Figure S5. Supplementary Materials). However, the quantile–quantile plot and the residuals distribution for this model do not show concerns (Figure S5. Supplementary Materials).

Looking at the model predictions in Fig. 4, it is evident that PBAT in combination with OPs is statistically significant for all the counts (adults, juveniles, eggs), showing a distinct negative effect. This trend is observable also for the Starch-based MPs in combination with OPs, statistically significant only for the egg counts.

Fig. 2 Model predictions for the adult (**a**), juvenile (**b**), and egg (**c**) counts at the end of the experiment for the MPs model. Red and blue colours refer to 0.01 and 0.1 w/w% MPs concentrations, respectively. Confidence intervals are represented by the vertical bars. Same letters indicate no statistical difference ($p > 0.05$)

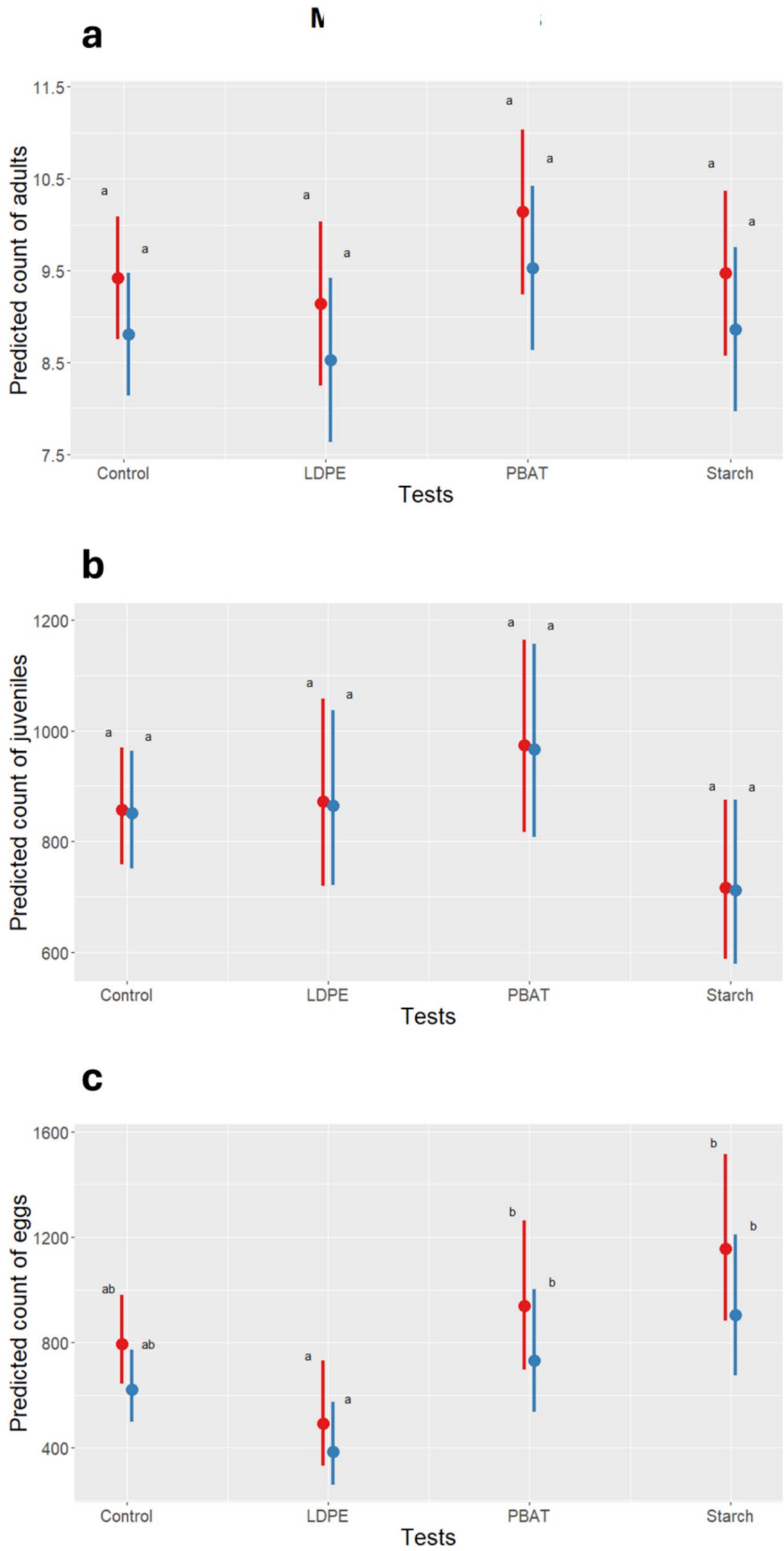
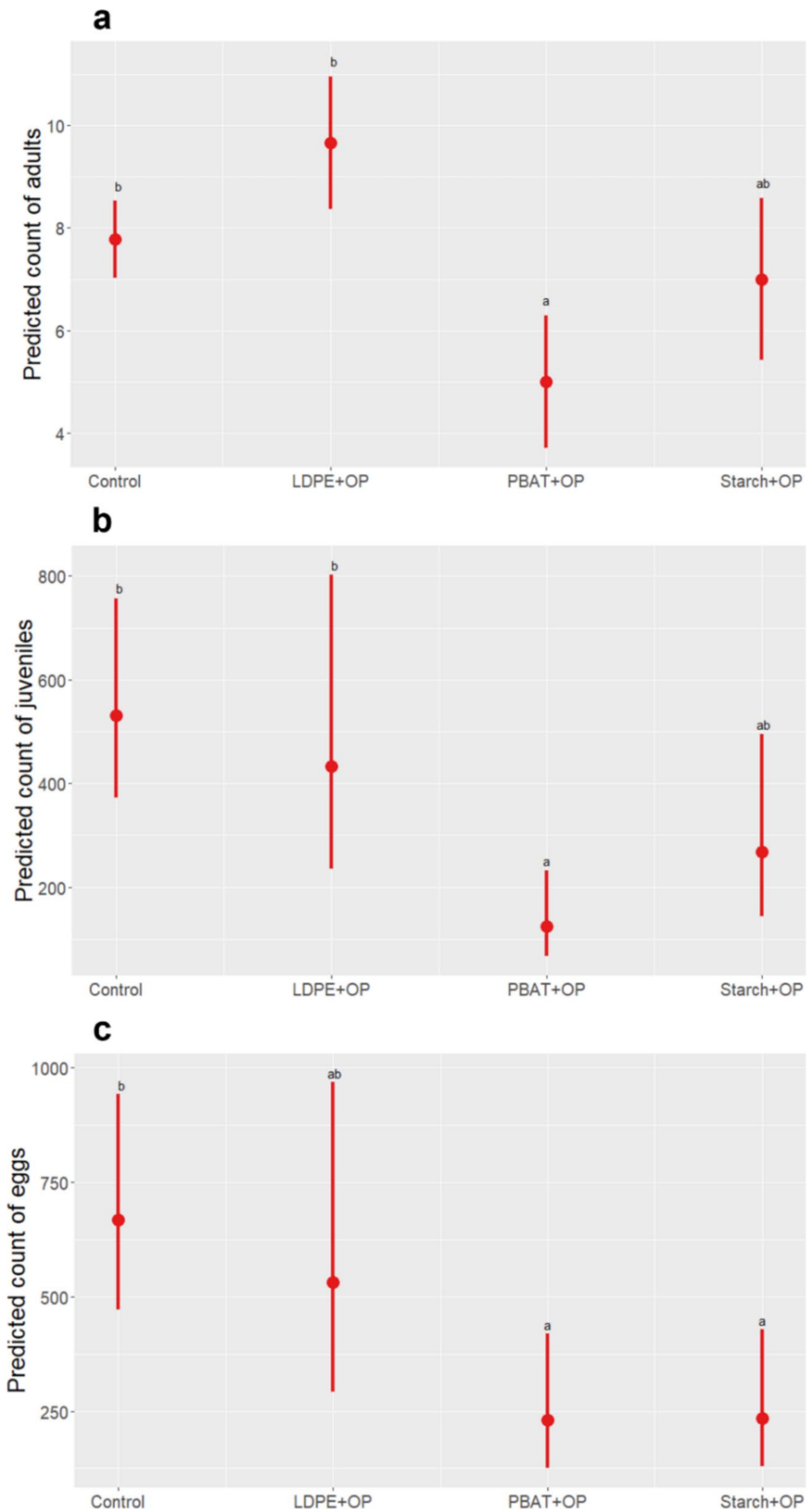


Fig. 3 Model predictions for the adult (**a**), juvenile (**b**), and egg (**c**) counts at the end of the experiment for the combined MPs-OPs experiment model at MPs concentration of 0.1 w/w%. Confidence intervals are represented by the vertical bars. Same letters indicate no statistical differences



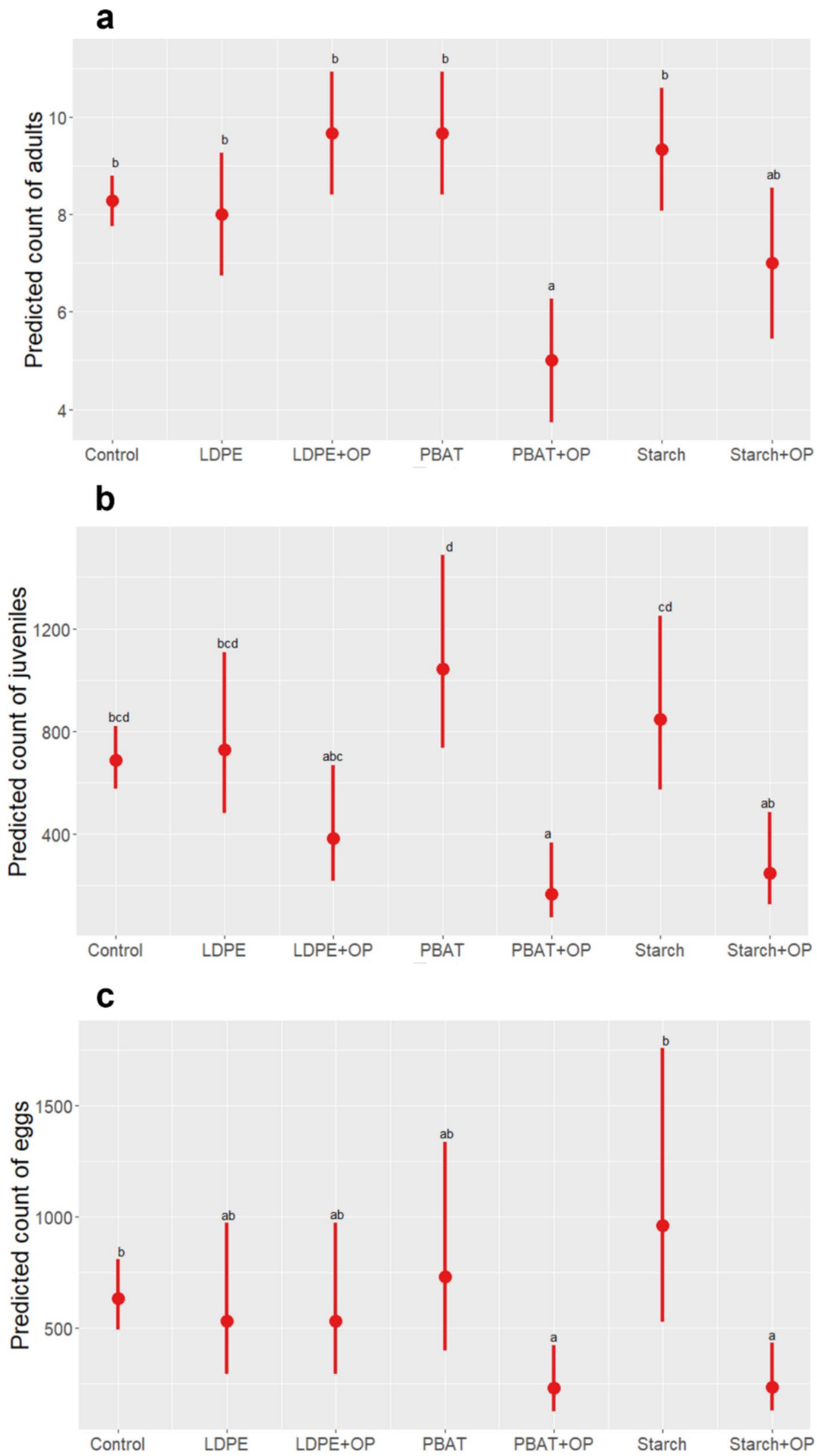


Fig. 4 Model predictions for the adult (**a**), juvenile (**b**), and egg (**c**) counts at the end of the experiment for the MPs versus MPs-OPs model at MPs concentration of 0.1 w/w%. Confidence intervals are represented by the vertical bars. Same letters indicate no statistical differences

4 Discussion

Our findings indicate that MPs alone with the concentrations tested do not significantly impact *F. candida* reproduction. This finding is consistent with previous studies suggesting that MPs are generally non-toxic to this species or are likely avoided if the particle size is larger than the edible size range recorded for *F. candida* (< 66.0 ± 10.9 µm; Kim & An, 2020; van Loon et al., 2024). The MPs used in the present study had a size distribution between 271 ± 2.4 µm and 780 ± 14.7 µm, thus substantially exceeding the edible particle size range for *F. candida*.

In exposure tests involving LDPE, PBAT-based, or starch-based MPs combined with OPs, toxicological effects were observed only with biodegradable plastics. Specifically, the reduction in the number of adults, juveniles, and deposited eggs recorded with PBAT-based MPs indicates both direct (acute) and reproductive toxicity, suggesting a broad spectrum of adverse effects. In contrast, the reduction in the number of deposited eggs recorded with starch-based MPs indicates only reproductive toxicity, pointing to a more limited interaction with OPs. Our findings suggest that biodegradable plastics are more likely to degrade or react in the presence of OPs, potentially releasing harmful byproducts or enhancing the toxicity of the OPs. For instance, it has been demonstrated that during degradation in aquatic environments, PBAT can release toxic byproducts, including monomers such as adipic acid, 1,4-butanediol, and terephthalic acid, as well as low-molecular-weight carboxylic acids. These compounds pose risks to aquatic ecosystems before undergoing metabolization (Ali et al., 2024).

Microplastics (MPs) can also interact with chemical contaminants in the environment through various physicochemical mechanisms, particularly hydrophobic/hydrophilic interactions, which influence the spatial distribution and bioavailability of organic pollutants (Torres et al., 2021). Notably, PBAT is a hydrophilic compound, as indicated by its water contact angle below 90°, reflecting its water-attracting properties (Wang et al., 2024). In contrast, LDPE has a contact angle ranging from 90° to 110°, signifying greater hydrophobicity (Yuan & Lee, 2013). For the starch-based polymer, the reported contact angle is 94.7°, suggesting intermediate hydrophilic properties between LDPE and PBAT (Aldas et al., 2020). These differences in contact angles likely influence the plastics' affinities for hydrophobic compounds such as PYR and ABZ, both of which are apolar and have very low water solubility. LDPE, being more hydrophobic, is more likely to adsorb these compounds, potentially reducing their bioavailability. Conversely, the more hydrophilic nature of PBAT may limit its ability to adsorb apolar compounds, leading to greater bioavailability of the OPs in its presence. The starch-based polymer, with its intermediate contact angle, may exhibit adsorption characteristics between those of LDPE and PBAT which could help explain the reproductive – but not acute—toxicity observed within the experimental time frame. Additionally, differences in degradation rates, surface functional groups, and interactions with soil properties (e.g., porosity, organic matter content, pH, and cation exchange capacity) among the three microplastics may also contribute to the observed differential effects, further influencing the extent of pollutant bioavailability and toxicity. Further studies are needed to fully understand the physical and chemical interactions between the microplastics (MPs), the organic pollutants (OPs) and the soil used in this study. This includes investigating how MPs adsorb, retain, or release PYR and ABZ in different environmental conditions, as well as identifying the factors that influence these processes, such as the size, composition, and surface properties of the different MPs. Additionally, it is crucial to explore whether these interactions alter the stability, bioavailability, or degradation rates of the OPs, potentially increasing their environmental persistence or toxicity. Understanding these mechanisms is essential to evaluate the role of MPs as carriers for OPs, their potential to facilitate co-exposure, and the implications for ecosystem and human health.

5 Conclusions

The findings highlight that toxicological effects in exposure tests occur only when biodegradable plastics (PBAT-based or starch-based MPs) are combined with OPs, while no effects are observed with the conventional plastics (LDPE). This suggests that biodegradable plastics may interact with OPs, either through degradation processes releasing harmful byproducts or enhancing the bioavailability of OPs. PBAT-based MPs exhibit both direct (acute) and reproductive toxicity, whereas starch-based MPs primarily affect reproduction, indicating distinct mechanisms of interaction. The absence of effects with LDPE and OPs underscores the inert nature of conventional plastics under these conditions. Importantly, the data reveal that biodegradable plastics, often perceived as environmentally safer, could pose unexpected risks when co-exposed to pollutants. This information will improve the assessment of ecological risks posed by both biodegradable and non-biodegradable plastics in soils contaminated with OPs.

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Data Availability The data supporting the findings of this study are available from the corresponding author upon reasonable request.

Declarations

Ethics Approval and Consent to Participate Not applicable.

Conflict of interest The authors declare that they have no conflict of interest.

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