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From sink to Sea: Microplastic release from kitchen sponges and potential environmental effects

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ABSTRACT

Microplastics (MPs) are released during the extraction, production, usage, or end-of-life stage of plastic products and cause negative effects for organisms and humans. Kitchens have recently been identified as hot spots for MP release, however, only few household items have been analyzed for their MP release quantities and related environmental effects. Here, we determined the MP release of kitchen sponges based on a combination of citizen science and laboratory experiments and performed life cycle assessment (LCA) to evaluate the resulting ecosystem damage. The results show the plastics in the sponges are released into the wastewater as MPs due to abrasion. The total MP release from different sponges into the wastewater stream ranges between 0.682 ± 0.566 and 4.212 ± 3.039 g of MP per year and person. The damage to ecosystem quality per 100 hours of dishwashing ranges between 6.26 PDF*m2*yr and 9.73 PDF*m2*yr. Most of the damage is caused by the water usage during manual dishwashing rather than material production or abrasion. A sponge declared as organic contained the least amount of plastic (15.9 w%), which led to the lowest MP release and lowest damage to the ecosystem compared to a sponge with a higher plastic share (59.3 w%). Therefore, a lower plastic share in kitchen sponges can significantly reduce MP release and related negative effects in the environment.

1. Introduction

Microplastics (MPs, 0.1 µm - 5 mm) and Nanoplastics (NPs, <0.1 µm) are released during the extraction, production, use, or end-of-life stages of plastic products (Maga et al., 2022). Once released, they can enter the human body through food, water, inhalation, and skin contact (Yang et al., 2022; Yang et al., 2023), depending on personal lifestyle, dietary habits, and levels of environmental pollution (Yang et al., 2023; Liu et al., 2024). Human health may then be affected by toxic chemical components, vectors of contaminants, and physical damage. On a cellular level, MPs and NPs can lead to changes in gene expression and cause oxidative stress. On a system level, MPs and NPs can influence the digestive, respiratory, endocrine, reproductive, and immune systems (Yang et al., 2022). Many of these suspected adverse effects are derived from animal studies, and further studies are required to close knowledge gaps concerning human exposure and health impacts (Yang et al., 2022; Yang et al., 2023).

Kitchens have recently been identified as hot spots for human MP exposure (Liu et al., 2024; Snekkevik et al., 2024). Objects like cutting boards, tableware, food packaging, plastic containers, or plastic kettles release MPs due to abrasion during food preparation, storage, serving, or cleaning (Liu et al., 2024; Snekkevik et al., 2024). Carrot chopping on polyethylene boards leads to an estimated MP release of 7.4–50 g per person and year, which, based on particle sizes around 100 µm, corresponds to approximately 14.5 to 71.9 million MP particles (Yadav et al., 2023). However, MP release from many sources in kitchens has not been quantified under real-use conditions (Liu et al., 2024).

In this article, we selected kitchen sponges as an example to show how consumer-related MP release can be quantified through a combination of citizen science and laboratory methods. Kitchen sponges are a

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ubiquitous tool in kitchens (Liu et al., 2024) and are designed specifically for scrubbing surfaces, which may release large amounts of MPs during abrasion (Su et al., 2024). In past laboratory experiments, kitchen sponges released hundreds of MP particles and fibers within 30 s when scrubbing a glass surface (Luo et al., 2022), but the total amount per sponge or per dishwashing session was not determined. A study in Denmark estimated annual MP release from kitchen sponges based on sales figures and a non-experimentally derived abrasion factor, resulting in 10–100 tons of MP released per year (Lassen et al., 2015).

Our study aims to quantify MP release from kitchen sponges through a combination of citizen science and laboratory studies and use a bottom-up, data-driven approach to review previous estimates of MP release from kitchen sponges. As MPs from sponges are likely mainly released during user handling, i.e., scrubbing and cleaning, we chose a citizen science approach to ensure realistic handling of kitchen sponges and gain further information about use routines through a survey. We selected two kitchen sponge types that are commonly available in German and North American supermarkets, and a third sponge type that is labelled as organic. The laboratory experiments were conducted as abrasion tests to investigate abrasion over time with no other influences, such as detergents, food residues, or different users. Additionally, we performed a life cycle assessment (LCA) to evaluate potential environmental impacts of the production, use, and end-of-life of the sponges, including plastic pollution, which is commonly not considered in LCAs (Galafton et al., 2025).

2. Methods

2.1. Selected materials

Three kitchen sponges were selected: a conventional European sponge (EU-con), a conventional North American sponge (Am-con), and an organic sponge (EU-org) (Fig. 1). EU-con has three layers: a blue cloth layer (80 % viscose, 20 % polypropylene), a yellow foam layer in the middle (100 % polyurethane), and a black scrubbing layer (51 % quartz, 24 % resin, 17 % polyamide, 7 % polyester, 1 % silicon carbide). EU-con is a commonly sold kitchen sponge in German supermarkets and represents a typical product type for household cleaning. Am-con sponge consists of two layers: a scrubbing layer (100 % recycled polyester) and a yellow sponge layer (100 % cellulose). This sponge is widely used for dishwashing and cleaning in North America and is readily available through common retail outlets. EU-org has two layers, a green foam layer (100 % cellulose) and a beige scrubbing layer (60 % sisal, 40 % recycled polyethylene terephthalate). All material information was gained from manufacturer data on the sponge packaging or found in online product descriptions by the manufacturer.

To determine the plastic content of each sponge type, three sponges per type were cut to separate the layers and weigh them individually. Each weight was then multiplied by the material share indicated by the manufacturer and added up. The total plastic share is 59.3 % for EU-con, 15.9 % for EU-org, and 41.9 % for Am-con (n = 3 each).

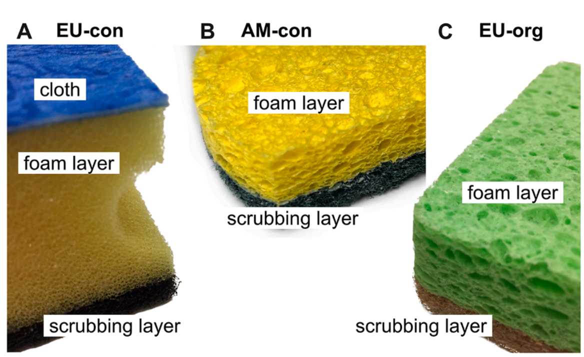

Fig. 1. (one column): Material layers for each sponge type: A) EU-con, B) AM-con, and C) EU-org. Microscopic images of each layer can be found in SM-1A.

For this study, we understand MPs' releases as the material loss that arises from the source, i.e., at the start of the MPs' pathway into the environment, e.g., during rinsing and cleaning, and before potential retention in wastewater treatment plants (WWTP). Microplastics that are not captured by wastewater treatment plants (i.e., total releases minus retained MPs) are considered environmental emissions, following Maga et al. (2021).

2.2. Determination of microplastic release

For the citizen science experiment, 30 sponges of EU-con and EU-org were rinsed with clean tap water and dried for 24 h in a heating cabinet at 42 °C. Afterwards, they were numbered, weighed, and handed out randomly to volunteering participants in Germany between March and May 2021. Each participant received a short survey regarding the daily usage of the sponge (see SM-1B). After the sponges were returned, they were rinsed again with clean tap water, dried in a heating cabinet at 42 °C for 24 h, and weighed again to determine the weight loss. No further cleaning was performed. We did not analyze the sponges under the microscope to determine types of degradation or dirt retention, which may have influenced weight loss or gain. The experiment was repeated in 2022 in Germany with 30 sponges of the two types each, but with other participants. In 2023, 50 Am-con sponges were distributed in Florida and Minnesota, USA, and New Brunswick and Alberta, Canada, to volunteer participants with the same survey translated into English.

In the laboratory experiment, we wanted to determine MP release over usage time in addition to the citizen science experiments. Therefore, we constructed an automatic test stand called “SpongeBot” to repeatedly compress the sponges mimicking sponge usage and material abrasion under laboratory conditions (see SM-1C for design and detailed parts description). The SpongeBot v1.0.0 release, with its manual, CAD files, and code, is deposited at Zenodo (10.5281/zenodo.17579830). The machine was placed in a tank (dimensions 500 × 300 × 300 mm) that was equipped with an aquarium filter (Eheim Type 2213, 440 l/h) to constantly clean the water from abraded material and a heater (Hydor Hydromatic T50) to keep the water temperature at 30 °C to mimic cleansing temperatures (SM-1D). The tank was filled with warm water, and the filter pump and heater were started at least one hour before each trial. Positioned in Spongebot, the sponges are compressed and fully submerged in water to simulate scrubbing conditions in a filled sink.

To determine the starting weight of the three sponge types, eight sponges of each sponge type were rinsed with tap water, squeezed five times, and dried in a heating cabinet (Vevor Scientific Lab Incubator 25 L) at 42 °C for 24 h before being weighed. During each trial, the numbered sponges (1–8) of one type were randomly distributed in the two compression levels of the machine. The levels were then moved up and down to find the minimum and maximum compression of each sponge type (SM-1D). Because the sponges became softer after more trials, this procedure was repeated each time, and the compression distance was adapted accordingly. The number of iterations (one iteration equals one up and down movement) was entered into the software, and the trial started. In total, 5 trials were done with 100, 200, 400, 800, and 1600 iterations. Since we observed no further material loss after the total of 3100 iterations for each sponge, we stopped the experiments. After each trial, the sponges were removed from the tank and squeezed five times over the sink to remove excess water. Afterwards, the sponges dried for at least 24 h. The final weight was taken when no change in weight was measured. The water of the tank was changed after all sponge types were tested for each iteration step. The laboratory experiments were all performed by one operator. The raw data for the citizen science and laboratory experiments can be found in the data repositories outlined in SM-1C.

2.3. Data analysis, statistics, and upscaling

The experimental data were analyzed and visualized using the R programming environment (R Core Team, R version 4.2.2, 2022). The data were used to calculate the material loss of each sponge, i.e., the weight before the experiment minus the weight after the experiment. For further analysis, we excluded sponges that gained weight, i.e., the sponge weight after the citizen science experiments was larger than the starting weight. We assume that this net weight gain is probably due to large amounts of retained food waste or dirt rather than any material properties or other processes. MP release was calculated based on the material loss multiplied by the share of plastic used in each sponge layer. Results are reported as mean with standard deviation. To assess the suitability of parametric tests, we evaluated the assumptions of normality and homogeneity of variances for the variables absolute daily release (weightloss_day) and relative total release (release_rel_total) across the three sponges (EU-con, Am-con, and EU-org). Normality within each group was tested using the Shapiro–Wilk test, which revealed significant deviations from normality in several groups (e.g., EU-con and EU-org; $p < 0.05$). Homogeneity of variances was assessed using Levene's test, which indicated no significant differences in variance among the groups for either variable ($p > 0.45$). Given the violation of the normality assumption, but homogeneity of variances and unequal sample sizes, we proceeded with the non-parametric Kruskal–Wallis followed by post hoc Dunn test with Bonferroni correction for group comparisons. Effect sizes for Kruskal–Wallis tests were quantified using epsilon squared (ε^2). To compare slopes of material loss over time between sponge types without relying on parametric assumptions, we used Theil–Sen regression, a non-parametric method robust to outliers and non-normal residuals. Slopes were calculated separately for each material, and differences between slopes were evaluated using permutation tests (10,000 iterations, seed set for reproducibility). We first tested for an overall difference in slopes among the three sponges (variance of slopes as the test variable) and, if significant, conducted pairwise permutation tests. Holm's method was applied to adjust p-values for multiple comparisons.

We also calculated MP release in grams per person and year (g/cap*a) to draw comparisons to other MP sources. Furthermore, we used the mean of the data to exemplarily extrapolate the total MP releases from kitchen sponges for Germany (population: 84.3 million, Federal Statistical Office, 2025). To deduct MP emissions from the release data, we calculate annual MP emissions for Germany by including a retention rate of 90 % through wastewater treatment (Iyare et al., 2020).

2.4. Assessment of the environmental impacts of the sponges

We conducted an LCA to analyze the environmental impacts of the three types of kitchen sponges throughout their lifetimes, including the potential impacts of plastic pollution resulting from material loss during usage. The aim was not to compare their impacts, but to determine environmental hotspots and analyze the contribution of MP pollution to the overall impacts. The function of the system under study is to clean dishes by manual dishwashing. According to the objective of the study, the functional unit is defined as 100 h of manual dishwashing. The reference flows are: 100 h of manual dishwashing with either kitchen sponge EU-con, EU-org, and Am-con. The main characteristics of the three types of sponges are given in Table S-1 in supplementary material SM-1F. The indicated lifetime is based on the citizen science experiment.

The scope of the study covers all relevant life cycle stages from cradle to grave (see SM-1E), including raw material extraction, the production of the sponges, the transport of materials and products, the use stage (material abrasion), and the end-of-life (EoL). Infrastructure, such as extruders, is excluded. The geographical scope for the study is Germany and North America, and the data are representative for the year 2023. Regional differences in impact scores may arise from variations in production technology, transport, and end-of-life practices. In the case of the fossil-based plastic materials (PA, Polyester, PU, PP), raw material extraction and virgin granulate production include crude oil extraction, refining, and granulate transport. For biobased materials (sisal, cellulose), raw material extraction and virgin fiber production comprise biomass cultivation, processing, and transport to the production facility.

The LCA scenarios are modeled using the software openLCA (version 2.5) and the ecoinvent APOS database (version 3.9) with plastic litter extension (PLEX) (Wernet et al., 2016). The life cycle inventory (LCI) can be found in supplementary material SM-1F. We assume an electricity consumption of 1 kWh per kg of sponges for all sponges based on the average of different production processes in the ecoinvent APOS 3.9 database (Wernet et al., 2016). For the distribution, the population-weighted average of the most direct distance of the production facility to the different states within the country (Germany/USA) is used (see SM-1F). Based on data obtained from the citizen science experiment (see results), 1.04–1.66 % of the material is lost during the use phase, entering the sewage system as part of the wash water. We assume that 90 % of the material is retained at the wastewater treatment plant (WWTP) (Wolff et al., 2018 and Iyare et al., 2020 for Germany, Gies et al., 2018 and Conley et al., 2019 for North America) although the share may be as high as 99.9 % (Carr et al., 2016, Mintenig et al., 2017, Talvitie et al., 2017) for WWTPs with tertiary treatment such as additional sand filters or membranes). We are presenting results for retention efficiencies of 85 % and 99.9 % as a sensitivity analysis. In Germany, 81 % of the microplastic contained in the WWTP sludge is thermally treated, 14 % emitted to agricultural soil as fertilizer, and 6 % landfilled (Federal Statistical Office, 2024)many. In the case of the USA, 59.8 % of the WWTP sludge is applied to land, 14 % is incinerated, 23.9 % is landfilled, and 2.3 % receives another type of treatment (United States Environmental Protection Agency, 2025). The remaining 10 % at the WWTP are emitted to freshwater as part of the effluent. For the other life cycle stages, the ecoinvent plastic litter extension (PLEX) (Cilleruelo Palomero and Ciroth, 2024) enables the quantification of plastic emissions.

Thermal treatment is assumed at the end-of-life for the intact material. When modeling waste treatment that yields valuable co-products, the substitution approach is applied to address multifunctionality. In this approach, credits are assigned for the recovered energy based on the respective electricity and district heat grid mixes.

In LCA, environmental flows (i.e., resource extraction and emissions) are assigned a so-called characterization factor in accordance with the substance's contribution to specific impact categories, such as climate change or land use. That way, the substance's environmental or human health impact can be measured and compared. Due to a lack of data, we focused on environmental rather than human health impacts. In our case, the environmental impacts were assessed using the ImpactWorld+ 2.1 methods package (Bulle et al., 2019; Agez et al., 2024), which quantifies impacts on ecosystem quality as potentially disappeared fraction of species (PDF)*m^2*year. This metric indicates the potential loss of species in a specific area over a defined amount of time (usually 1 year) due to environmental pressures, representing the impact of human activities on biodiversity. The ImpactWorld+ method package does not contain a method for plastic pollution impacts. Therefore, in line with USEtox (Rosenbaum et al., 2008; Fantke et al., 2017), the environmental impact of the plastic emissions was assessed by calculating own characterization factors consisting of fate factors (Maga et al., 2022), an exposure factor, and effect factors (Corella-Puertas et al., 2023; Saadi et al., 2025; Tunali and Nowack, 2025; Vázquez-Vázquez et al., 2025.; see also SM-1H). For the fate, we made assumptions regarding the shape and size of the abraded particles per sponge layer based on macro images of the SpongeBot residues displayed in SM-1G. Since details regarding the polymer types, shapes, and sizes of the emissions quantified in the PLEX are unavailable, it is currently not possible to characterize the impacts of emissions in the background system and include them in the overall assessment. We, therefore, indicate emission quantities for both the background and the foreground system (use stage), and plastic pollution impacts only for the use stage.

3. Results

3.1. Microplastic release

By the end of the experiment, 32 EU-con, 21 Am-con, and 39 EU-org sponges were returned, which equals a return rate of 53.3 %, 42.0 % and 65.0 %, respectively (Table 1). Some sponges were removed from the study because of incomplete surveys or too much mold on a sponge. Additionally, sponges were removed if they gained weight during the citizen science study. These sponges likely also lost weight due to material abrasion. However, it is impossible to estimate the mass lost and gained. Including these sponges and recording the material loss as 0 or as a negative value to account for the weight gain would lead to an underestimation of material loss. Therefore, we excluded the sponges from the analysis.

In the end, 27 EU-con, 13 Am-con, and 28 EU-org sponges were used for further analysis, which resulted in a final return quote of 45.0 %, 26.0 % and 46.8 %, respectively. The number of persons per household was 1.96 ± 0.81 for the EU-con, 1.77 ± 1.01 for the Am-con, and 2.14 ± 0.89 for the EU-con but this difference between the sponges was not significant (Kruskal-Wallis test, $\chi^2(2) = 2.5016$, $p = 0.2863$, $\varepsilon^2 = 0.023$). Most households had a dishwasher (>61.5 %), used the sponge for rinsing (>53.8 %), and used detergents (>84.6 %). For all three factors, the Am-con showed the lowest share. However, the slight differences regarding these household factors had no significant effect on the average usage time per day (Kruskal-Wallis test, $\chi^2(2) = 4.4808$, $p = 0.1064$, $\varepsilon^2 = 0.053$), which were 8.33 ± 4.14 min/day for EU-con, 11.15 ± 5.46 min/day for Am-con, and 11.15 ± 8.76 min/day for EU-org. They also had no effect on the total days of usage (Kruskal-Wallis test, $\chi^2(2) = 1.7805$, $p = 0.4106$, $\varepsilon^2 = 0.012$).

On average, the EU-con sponges lost 0.033 ± 0.016 g/day of material, the AM-con sponge lost 0.013 ± 0.010 g/day, and the EU-org lost 0.023 ± 0.019 g/day (Fig. 2A). This corresponds to 0.0028 ± 0.0014 %, 0.0013 ± 0.0011 % and 0.0025 ± 0.0020 % material loss per day, respectively (Table 1). Post-hoc pairwise comparisons using Dunn's test with Bonferroni correction reveal that EU-con differed significantly from Am-con ($z = 3.258$, $p = 0.0034$, $\varepsilon^2 = 0.146$), while no significant differences were found between EU-con and EU-org ($p = 0.578$) or AM-con and EU-org ($p = 0.077$). When considering the plastic share of the sponges, the EU-con sponges release 0.019 ± 0.01 g MP/day, the Am-con releases 0.005 ± 0.004 g MP/day, and the EU-org releases 0.004 ± 0.003 g MP/day (Table 1, Fig. 2B). Post-hoc pairwise comparisons using Dunn's test with Bonferroni correction show significant differences between EU-con and Am-con ($z = 3.921$, $p = 0.00025$, $\varepsilon^2 = 0.561$) and EU-con and EU-org ($z = 2.793$, $p < 0.0001$), while there is no significant difference between Am-con and EU-org ($p = 1$). On an annual basis, the MP release sums up to 4.212 ± 3.039 g/cap*a for EU-con, 1.171 ± 0.951 g/cap*a for Am-con, and 0.682 ± 0.566 g/cap*a for EU-org. Similar to MP release per sponge, post-hoc pairwise comparisons using Dunn's test with Bonferroni correction show significant differences between EU-con and Am-con ($z = 3.474$, $p = 0.0015$, $\varepsilon^2 = 0.544$) and EU-con and EU-org ($z = 5.967$, $p < 0.0001$), while there is no significant difference between Am-con and EU-org ($p = 0.58$).

The overall permutation test did not indicate a statistically significant difference in the rate of MP release among the three sponge types (Theil-Sen regression of relative MP release over usage time, $p = 0.13$). Pairwise comparisons showed that EU-con had a slightly steeper negative slope than EU-org (observed difference = 1.54×10^{-5}, unadjusted $p = 0.044$, Holm-adjusted $p = 0.132$) and Am-con (observed difference = 1.54×10^{-5}, unadjusted $p = 0.208$, Holm-adjusted $p = 0.416$), but these differences were not statistically significant after adjustment (Table S-6 in supplementary material SM-1I). The slope difference between EU-org and Am-con was negligible (2.13×10^{-8}, $p = 0.994$).

Based on the assumption that everyone in Germany used the EU-con sponge, the total MP release for Germany is around 355.1 t/a. If everyone used the Am-con, the annual release for Germany would be 98.7 t/a and 57.5 t/a if everyone used EU-org (Table 1). MPs from kitchen sponges are likely released into the sewage system, where wastewater treatment plants retain around 90 % of MPs (Iyare et al. 2020); hence, MP emissions into freshwater would amount to 10 % of the release, i.e., 35.51 t/a, 9.87 t/a, and 5.75 t/a, respectively. Additional MPs emitted to agricultural soil as part of sewage sludge would amount to 44.7 t/a, 12.4 t/a, and 7,2 t/a, respectively.

The laboratory abrasion tests with the SpongeBot show that all three sponge types lose material. After 3100 iterations, EU-con had lost 3.82 ± 0.35 % of its starting weight, Am-org had lost 5.61 ± 2.65 %, and EU-org had lost 9.2 ± 2.65 % (Fig. 3C). Most of the material was lost after 300 iterations, with 60.61 % of the total weight loss for the EU-con sponge, 78.92 % for Am-con, and 85.40 % for EU-org.

Table 1
Overview of survey and weighing results regarding the three sponge types.

	EU-con	Am-con	EU-org	Kruskal-Wallis test
Returned sponges considered for analysis (n)	27	13	28	-
Final return quote	45.0 %	26.0 %	46.8 %	–
Number of persons per household	1.96 ± 0.81	1.77 ± 1.01	2.14 ± 0.89	not significant
Household with a dishwasher	18 (66.6 %)	8 (61.5 %)	24 (85.7 %)	-
Usage "rinse"	23 (85.2 %)	7 (53.8 %)	26 (92.9 %)	-
Used with detergents	25 (92.6 %)	11 (84.6 %)	27 (96.4 %)	-
Average usage time (min/day)	8.33 ± 4.16	11.15 ± 5.46	12.14 ± 8.76	not significant
Days of usage	44.30 ± 28.46	73.38 ± 73.95	53.46 ± 40.28	not significant
Absolute weight loss per day [g/day]	0.033 ± 0.016	0.013 ± 0.010	0.023 ± 0.019	-
Relative weight loss per day [%/day]	0.0028 ± 0.0014	0.0013 ± 0.0011	0.0025 ± 0.0022	significant
Plastic share of material	59.3 %	41.9 %	15.9 %	-
Absolute MP release per day [g/day]	0.019 ± 0.01	0.005 ± 0.004	0.004 ± 0.003	significant
MP release per person and year [g/cap*a]	4.212 ± 3.039	1.171 ± 0.951	0.682 ± 0.566	significant
Total MP release in Germany [t/a]	355.1	98.7	57.5	-

3.2. Environmental impacts

The impact categories contained in the chosen impact assessment method package are displayed in Fig. 3A. Since all impacts in all impact categories are converted into the unit PDF*m^2*yr, they can be combined into a single score indicator, and the contributions of different impact categories to this indicator can be assessed. The total impact on ecosystem quality is 9.73 PDF*m^2*yr for EU-con, 6.26 PDF*m^2*yr for EU-org, and 17.91 PDF*m^2*yr for Am-con, respectively. The main contributing impact categories for all three sponge alternatives are climate change (51.0–52.9 %), freshwater ecotoxicity (13.6–16.1 %), and marine acidification (15.6–16.3 %) (Fig. 3A). Plastic pollution (use stage only) contributes 0.002 %, 0.009 %, and 0.004 %, respectively.

For all three most contributing impact categories, the scores are largely driven by the provision of water for dishwashing and the treatment of wastewater from upstream activities and the use phase, including the construction of water supply and sewage systems. These processes cause the emission of aluminum, copper, and iron to groundwater, which are responsible for the freshwater ecotoxicity scores, as well as the emission of carbon dioxide and other greenhouse gases, which contribute to climate change and marine acidification

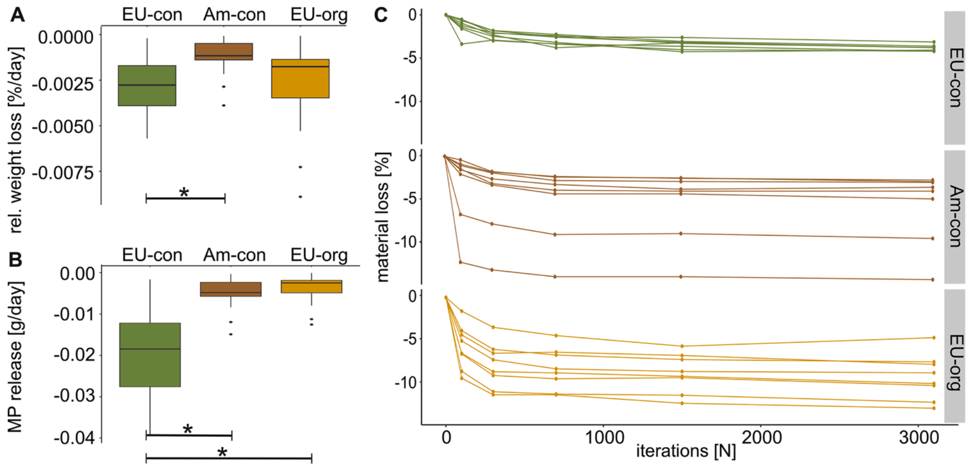

Fig. 2. MP release from three different kitchen sponges (EU-con, Am-con, EU-org) based on citizen science experiments. Statistically significant differences are indicated with an asterisk (Dunn's test with Bonferroni correction). A) Comparison in relative weight loss of the three sponge types and B) Daily MP release.C) Results of laboratory abrasion tests using Spongebot with all three sponge types (n = 8). One iteration is one up and down movement of the compression levels.

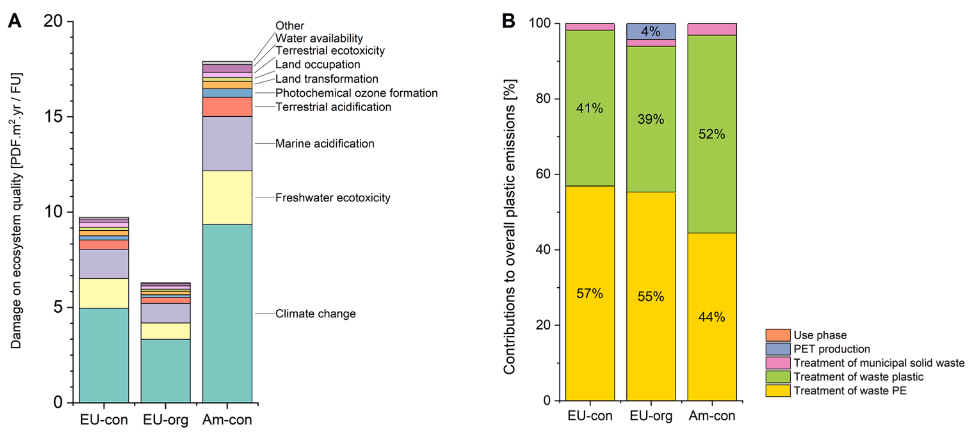

Fig. 3. A) Environmental impacts of the three sponge alternatives on ecosystem quality, B) Process contributions to plastic emissions.

impacts. Water use causes 85.7–96.8 % of the damage to ecosystem quality caused by the sponges. For Am-con, the environmental impacts are influenced by our choice of tap water provider in the LCA model, which represents a global average and causes 3.5 times the damage on ecosystem quality as the same process for Europe. EU-con causes additional freshwater ecotoxicity impacts via the treatment of sulfidic tailings from the silver mine operations, which is related to the small amount of silver chloride contained in the product to reduce bacteria. Plastic pollution causes 0.01–0.06 % of the freshwater ecotoxicity impacts, less than 0.01 % of the terrestrial ecotoxicity impacts, and

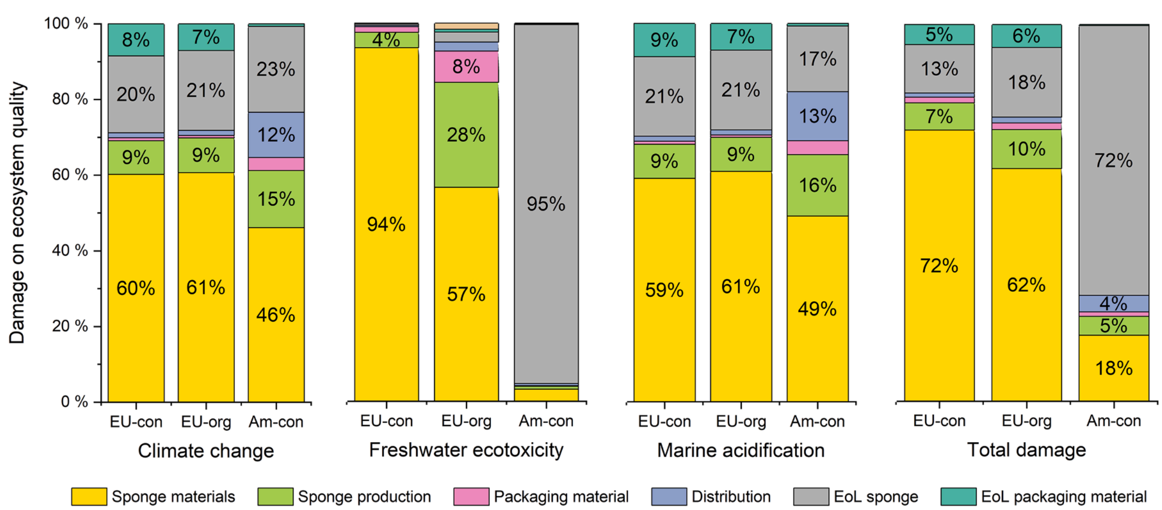

Fig. 4. Contribution of different processes to climate change, freshwater ecotoxicity, and marine acidification impacts (excluding water use).

78.7–93.4 % of the marine ecotoxicity impacts (note that the impacts of plastic emissions of the background system are not included in the ecotoxicity assessment).

The production of the sponge materials, such as PU, sisal, etc., contributes 0.6–10.5 % to the damage to ecosystem quality. Fig. 4 shows the contributions of the different processes to the damage on ecosystem quality, without the impacts of water use, which mask all other details. While the sponge materials are the second highest contributor to climate change and marine acidification (after water use), the EoL treatment of the sponges plays a larger role for Am-con for freshwater ecotoxicity and the overall impact scores. The freshwater ecotoxicity scores of the Am-con sponge are tainted by the fact that we chose a generic municipal waste treatment process due to a lack of more representative data. As a result, the chosen process includes open burning and open dumping/unsanitary landfill, which is not accurate for the USA.

The sponges lead to plastic emissions of 150 g, 110 g, and 290 g per FU, respectively, for EU-con, EU-org, and Am-con. The emissions occur almost exclusively in the background system (99.98–99.99 %), especially during (plastic) waste treatment and, to a lesser extent, plastic material production (Fig. 3B). Plastic emissions from the use phase contribute less than 0.02 % to the overall plastic emissions. Despite the lower emission quantity, the impacts of the use phase emissions of the EU-org sponge are higher than the EU-con sponge due to the significantly slower degradation of PET microplastics compared to PA, PU, and PP microplastics (see SM-1J). Depending on the retention efficiency of microplastics at the WWTP, the emissions from the use phase may be shifted to freshwater in the case of a lower efficiency or to agricultural soil for a higher efficiency (see SM-1J). Depending on the degradation speed of the specific material, this can increase or decrease the effects. For example, in the case of Am-con and EU-org, a lower efficiency increases the impacts of the use phase emissions from 0.91 and 0.58 PDF.m^2.year to 1.36 and 0.86 PDF.m^2.year because the effects on terrestrial organisms are much lower than aquatic organisms (0.0626 PAF.m^3.$kg^{-1}_{emitted}$ in soil based on Tunali and Nowack (2025) compared to 16.2 PAF.m^3.$kg^{-1}_{emitted}$ in freshwater sediment (Saadi et al. 2025)).

The environmental impacts are influenced by the number of sponges necessary to fulfil the reference flow, although the differences in the data from the citizen science regarding the average usage time per day and the average days of usage are not statistically significant. When assuming the same lifetime of 50 days for all three sponge types, the total impact on ecosystem quality is 0.66 PDF*m^2*yr for EU-con, 0.64 PDF*m^2*yr for EU-org, and 1.69 PDF*m^2*yr for Am-con, respectively.

4. Discussion

There are two dimensions of plastic pollution: emitted quantities on the one hand and their impact on the environment on the other hand. All three types of kitchen sponges lose material during use, both in a citizen science and laboratory setup. If these sponges contain plastics, they inevitably also release MPs into the wastewater system. In this study, EU-org causes the lowest amount of plastic release during dishwashing (the use phase) with 0.682 ± 0.566 g/cap*a and EU-con the highest with 4.212 ± 3.039 g/cap*a (Table 1). Only one estimate of MP release from kitchen sponges was reported for Denmark, with a total of 8.57 g/cap*a (Lassen et al. 2015), which was not based on experimental data. Due to the assumed 90 % efficiency of the WWTP, the amount of direct plastic emissions (sponge abrasion) into treated freshwater for our study ranges from 0.068 to 0.421 g/cap*a (Table 1). This equals 0.003–0.011 g during 100 h of dishwashing. The MPs that are retained in the WWTP sludge may be emitted to agricultural soil depending on the country's legislation and practices regarding WWTP sludge application. In our case, sludge application leads to indirect emissions of 0.003–0.021 g to agricultural soil per FU (100 h of dishwashing). The actual value may be higher for WWTPs with tertiary treatment. Still, in comparison, the emissions during upstream and downstream activities, especially plastic waste treatment, are far greater (110–290 g per FU). As a consequence, plastic emissions of the use stage contribute less than 0.02 % to the overall plastic emissions (Fig. 3B). For comparison, Saadi et al. (2025) calculated plastic emissions into the marine environment and corresponding impacts of textiles for high-income countries such as Germany and the USA, as 0.023–4.744 g per kg textile during washing, causing 4.52E-06–140.0 PDF.m^2.yr of damage to ecosystem quality (marine environment only). For the same region, Corella-Puertas et al. (2023) calculated plastic emissions of food packaging made of polylactic acid (PLA) and polypropylene (PP) as 6.00E-08 kg due to pellet loss during production and 4.3E-06 - 4.48E-05 kg during the end-of-life, considering different fragmentation scenarios. The end-of-life (fragmentation of macroplastic litter into microplastic particles) caused 9.55E-05 - 9.55E-06 PDF.m^2.yr of damage to ecosystem quality per FU for the made of PP and 5.3E-07 - 5.3E-08 PDF.m^2.yr of damage to ecosystem quality per FU for the PLA packaging. Other estimates for MP release in the household are polyethylene chopping boards with 7.4–50.7 g/cap*a (Yadav et al. 2023) and cosmetics with estimates varying from 2.4–11 g/cap*depending on region, year, and method. Based on this comparison, we suspect that kitchen sponges are a minor to medium MP source in the household.

However, the MP release from kitchen sponges during the use stage in Germany still amounts to 57.5 to 355.1 t/a, depending on sponge type, if the analyzed sponge were used in every household on a national scale. Bertling et al. (2018) identified and quantified 51 MP sources that add up to around 4 kg MP emissions/cap*a for Germany. The top ten of these quantified sources, such as tire wear abrasion, releases during waste management, pellet losses, losses from artificial turfs, or abrasion of MPs fibers from textiles, account for approximately 65 % of the total released MPs, leaving 41 minor sources for the remaining 35 %. Kitchen sponges were not considered in that study. Therefore, it is important to identify and quantify also minor MP sources. When doing so, it is important to avoid double-counting. For example, in our LCA study, we included emissions from upstream and downstream processes such as plastic and waste management, which are separately reported in studies such as Bertling et al. (2018).

MP release from sponges could be reduced by replacing the plastic content in the sponges. This can be seen when comparing the EU-org and EU-con sponges. The material loss in those two sponge types is similar (see Table 1), while the MP release is significantly higher for EU-con, which has the higher plastic share. However, the impact of the plastic emissions on the environment is higher for EU-org based on the materials used and emitted. Note that when considering the impacts of plastic pollution as part of LCA, data are insufficient to characterize emissions during upstream and downstream activities. Therefore, we can only consider the impacts of direct emissions. The difference in sponge materials is also reflected in the environmental impacts. Overall, EU-org causes less damage to ecosystem quality than EU-con per FU (6.26 PDF*m^2*yr vs. 9.73 PDF*m^2*yr). Beyond that, it is difficult to compare the German to the North American alternative because different distances and background processes are used. For all sponge types, the overall damage to ecosystem quality is much more dependent on the water usage during manual dishwashing than on material choice or abrasion. In fact, the water use overshadows all other impacts. As a consequence, sponges that clean more effectively and reduce the amount of water needed to obtain clean dishes offer an advantage over sponges that require more water. Besides, the sponges' lifespan also influences the results. Although the differences in usage times of the different sponge types were not statistically significant, we used the individually measured usage times calculated from the citizen science experiment for the LCA. If we used the same usage time for all sponges, the damage on ecosystem quality would be very similar between EU-con and EU-org (0.66 and 0.64 PDF*m^2*yr, respectively) but Am-con would still have the highest impact score of 1.69 PDF*m^2*yr. Therefore, increasing the sponges' lifespan could ameliorate the environmental impacts, but not as a stand-alone measure.

When quantifying MP releases, data quality is of high importance. In most cases, quantifications are based on available stock data, models, assumptions, and mass balances to calculate the range of MP releases and emissions depending on different scenarios. For example, the MP release from sponges in Denmark is based on sales figures and an estimated abrasion factor (Lassen et al., 2015). By multiplying all these factors, the propagation of uncertainty increases, and the estimation accuracy decreases (van Wezel et al., 2016). A data quality assessment of MP quantifications by the EU (Hann et al., 2018) includes a scoring system with the following criteria on a scale from 1 (best) to 5 (worst): reliability, completeness, temporal, and geography. Applying this quality assessment, most published quantifications of MP releases achieve 'intermediate certainty' scores (Hann et al., 2018). We would give the MP quantification performed here a total score of 6 points (Reliability: 1; Completeness: 2; Temporal: 1; Geography: 2), which means that the data has a high certainty and is likely to be of good quality.

There are some shortcomings in the methods that need to be mentioned. Some sponges gained weight during the citizen science experiments; probably caused by retaining food waste or dirt. This was not found in the sponges in the laboratory experiments, where they consistently lost weight. Assuming this process occurs for all sponges under real-life conditions, it is likely that the actual material weight loss in all sponges is higher than measured. A comparison of sponges after the citizen science and the Spongebot lab experiments showed differences in the outer appearance of the sponges (see SM-1K). The citizen science sponges show loss of material, fading of material colors, partly separation of sponge layers and residues of dirt or food waste, whereas the lab sponges still looked almost new. This discrepancy underlines that the results of the two experiments cannot be quantitatively compared, as the lab experiments lack factors that would resemble real life conditions. In future lab experiments, factors such as detergents, abrasion of different materials and surface structures, food residues and dirt, and compression in all dimensions could be added to. Additionally, the extrapolation from citizen science measurements to national scale are based on the assumption that all households consistently use one of the three sponges for rinsing. This may not represent real conditions and should be determined in future surveys. Data quality is also important for all other aspects of the LCA. In this case, we were unable to use country-specific data for tap water production and waste and wastewater management in the USA. Instead, we had to use generic, global data, which does not accurately represent water provision and waste management infrastructure. The plastic emissions of upstream and downstream activities are estimated based on a probabilistic approach and may need to be validated. Due to the small level of detail regarding these data (no polymer types, no sizes and shapes), it was not possible to apply characterization factors to these estimates. In the future, it would be useful to add the required level of detail to enable a more thorough comparison.

5. Conclusion

The combination of citizen and laboratory experiments to quantify MP release shows that very high-quality data can be gained, aiding in prioritizing measures to reduce MP release compared to other MP sources. The data can also be used in LCA to determine environmental impacts and derive efficient reduction measures. The LCA in this study shows that most of the damage is caused by the water usage during manual dishwashing rather than material production or abrasion. Focusing on the effects of MPs, the sponge declared as organic contained the least amount of plastic (15.9 w%), which led to the lowest MP release and lowest damage to the ecosystem compared to a sponge with a higher plastic share (59.3 w%). Therefore, a lower plastic share in kitchen sponges can significantly reduce MP release and related negative effects in the environment. We recommend using a similarly holistic approach to identify, quantify, and evaluate other MP sources and emissions to contain microplastic pollution. To further increase the data quality of this approach, more scenarios over a greater geographic area should be studied, for example, by using more sponge types, increasing the number of distributed sponges to increase return rates and statistical power, and repeating the citizen science experiments in more countries, including the Southern hemisphere.

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CRediT authorship contribution statement

Leandra Hamann: Writing – original draft, Visualization, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Christina Galafton:** Writing – original draft, Visualization, Methodology, Investigation, Formal analysis, Data curation. **Peter T. Rühr:** Writing – review & editing, Software, Methodology, Data curation. **Alexander Blanke:** Writing – review & editing, Supervision, Resources, Project administration, Conceptualization. **Nils Thonemann:** Writing – review & editing, Validation, Supervision, Resources, Project administration, Methodology, 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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Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.envadv.2026.100693.

Data availability

All data is shared

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