

# Comparative analysis of phthalate leachate in U.S., Oceania, and European recycled consumer PET bottles: assessing the impact of regulatory policies and recycled content.

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## Research Article

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# Abstract

Polyethylene terephthalate (PET) bottles are commonly used as packaging for consumer beverages with regulations on the use of plastic additives (often phthalates) within these bottles depending on the region. European legislation enforces stricter controls on these additives compared to the United States and authorizes that PET bottles must contain 30% recycled plastic content. Due to growing concern regarding health risks associated with phthalate exposure, solid-phase microextraction (SPME) in tandem with GC-MS was used to quantify and compare the concentrations of several phthalates of interest that leached from consumer plastic water bottles; butyl benzyl phthalate (BBP), dibutyl phthalate (DBP), di(2-ethylhexyl) phthalate (DEHP), diethyl phthalate (DEP), and dimethyl phthalate (DMP) in PET bottles manufactured in the U.S., Europe, and Oceania region. There was significant variation between phthalate leachate concentrations across countries. European bottles were found to have the lowest total phthalate concentration on day 7 ( $205 \pm 18 \mu\text{g/L}$ ) followed by Oceania bottles ( $459 \pm 38 \mu\text{g/L}$ ) and U.S. bottles ( $629 \pm 34 \mu\text{g/L}$ ). The goal of this project was to understand how differences in regulatory policies and the use of recycled materials affect phthalate concentration over time. The findings of this research will help contribute towards understanding the effect of regulatory policies in decreasing chemical contaminants and exposure to consumer plastic products.

## 1) Introduction

Polyethylene terephthalate (PET) bottles are widely used for beverage packaging, accounting for 67% of market use (Benyathiar et al., 2022). However, concerns have risen over the presence of phthalates and plastic additives added to bottles to increase structural integrity post-consumer use during manufacturing and recycling processes that may pose health risks (Amiridou & Voutsas, 2011; Gerassimidou et al., 2022; Jeddi et al., 2016; Keresztes et al., 2013). With global plastic production rising from 359 million metric tons (Mt) in 2018 to 367 Mt in 2020 and estimated to reach 884 Mt by 2050, plastic waste has become a global challenge (Benyathiar et al., 2022; Kasuske, Arole, et al., 2025; Tumu et al., 2024). As a result, there has been a renewed focus on recycling as a viable solution. However, with global recycling rates of 9% and mismanagement of 60–90 million tons of plastics each year, the result is progressively more plastic within the environment (Ekvall et al., 2022; Singh & Walker, 2024).

Another concern to consider is the disparity of recycling policies between countries. Currently 80% of EU countries implement an extended producer responsibility (EPR) policy which requires companies to participate in circular economies by reacquiring products post-consumer use (Tumu et al., 2023). However, even among countries these policies differ. In Italy the policy is a collective producer responsibility (CPR) in which an independent third party, a consortia of CONAI, CORIPET, and CORPELA in this instance, act as regulators for recycling (Emma Watkins, 2017; System, 2024). Denmark and a majority of other countries participate in an individual producer responsibility (IPR), where producers are responsible for end-of-life management (collecting, recycling, and disposal) of their products. Regardless, according to European Union Directives, the Directive on Packaging and Packaging Waste requires that by 2025 all PET bottles contain at least 25% recycled content. It also further requires that by 2030 all plastic packaging should be 55% recycled with PET containing at least 30% (Parliament, 2019).

Within the US there is currently not a nationwide EPR policy in place, instead EPR legislation has been enacted at the state level leading to varied policies in the states that participate in these policies. Currently in 2025, twelve states (California, Oregon, Colorado, Maine, Minnesota, New Jersey, New York, Hawaii, Massachusetts, Washington, Nebraska, and Illinois) have introduced legislation for EPRs with 7 EPR packaging bills having passed (Coalition, 2025). Several of these bills also require producers to enter into a Producer Responsibility Organization (PRO) to report data and pay fees to fund recycling programs. Following in the footsteps of Europe, in 2020 US EPA set a new target for plastic waste management aiming to increase recycling to 50% by 2030 (EPA, 2020). They also updated their Safer Choice and

Design for the Environment (DfE) Standards in 2024. In order for consumer plastic packaging products to earn a Safe Choice label or the DfE logo they must contain a minimum 15% post-consumer recycled content by weight (EPA, 2024). However, this is not a general mandate for all plastic packaging.

In areas of the globe where infrastructure for processing and recycling is lacking for a multitude of reasons these policies and implementation of the policies become more difficult. For example, in parts of Oceania, recycling is largely a voluntary citizen program. The Deposit Return Scheme (DRS) allows informal collectors to collect bottles and return them to the factory for 5 cents per bottle (March A., 2025). As major producers present within Oceania do not have the infrastructure or recycling centers within the region to facilitate this, the bottles are packaged and exported to China for recycling (IUCN, 2020; March A., 2025). As of July 2025, China's National Health Commission (NHC) and China National Center for Food Safety Risk Assessment (CFSA) drafted amendments to GB 9685, which is their standard on additives for food-contact materials and articles. In these amendments, six phthalates have been deemed "high risk" or had been previously eliminated from use by industry. These amendments suggest that China is tightening use towards additives inclusion in consumer products whose regulation has been historically limited (LLC, 2025; Straková, 2022).

As previously discussed, regulatory policies and implementation of these policies differ regionally and globally. The purpose of this research is to quantify and compare phthalate concentrations in PET bottles from the U.S., Europe, and Oceania. By assessing the influence of regulatory differences and recycled content, this study aims to provide insights into the effectiveness of existing policies in reducing chemical contaminants. Findings may help inform future regulations and promote safer plastic manufacturing practices worldwide.

## **2) Methods and materials**

### **2.1 Reagents and standards**

BBP, DMP, DEP, and DEHP standards were purchased from Scientific Polymer Products Inc. (Ontario, NY). DBP and BPA standards were purchased from Sigma-Aldrich (St. Louis, MO). All neat standard compounds were of 98% purity or higher. Stock standard solutions of 1 mg/mL and 0.1 mg/mL in acetonitrile were prepared for each compound. Working individual and mixture standards of 1000 µg/L were prepared weekly in acetonitrile and used for subsequent dilution. All standards were prepared and stored in autoclaved borosilicate cylinders and stored at 4 °C. Acetonitrile (99.9% purity) and LC-MS grade water were purchased from Sigma-Aldrich (St. Louis, MO). Helium carrier gas (ultra-high purity 200) was purchased from AirGas (Lubbock, TX).

### **2.2 Sample preparation**

U.S. and Oceania single-use PET consumer plastic bottles were purchased from a local supermarket in Lubbock, TX. European bottles were purchased from a local supermarket in Glasgow, Scotland and shipped to Lubbock, TX. All bottles were 16.9 oz and composed of 100% recycled PET (rPET). For sampling and analysis, experimental units (rPET bottles) were housed in darkness, at room temperature, within a biosafety chamber equipped with a laminar flow hood. Air flow was constant throughout the experiment duration with temperature and relative humidity monitored daily via a Govee Smart Thermo-hygrometer (Model #H5075).

To quantify phthalate leachate from consumer plastic bottles over the period of 7 days, a repeated measure design was implemented. Bottles were not emptied and original water was used for sampling. A sample period of 7 days was chosen as survey data indicates that a majority of families shop once per week (Koppes, 2026; Ou, 2025). For each country, a respective triplicate was designated. On days 0, 1, 3, and 7, water from each plastic bottle (2 mL) was removed and placed in a 2 mL GC autosampler vial with a PTFE/silicone septum and analyzed for phthalate leachate. Day 0 samples served as controls for their respective countries throughout the study as access to the natural water

prior to bottling was unobtainable thus a reference point for initial phthalate levels was unavailable. Samples were stored in a covered container at 4 ° C in a refrigerator and were analyzed within 12 h of being removed.

## 2.3 Solid-Phase Microextraction (SPME)

For phthalate extraction a polydimethylsiloxane/divinylbenzene (PDMS/DVB) SPME fiber with a 65 µm coating from Supelco (Bellefonte, PA) was implemented. Previous work from our lab, as well as, several optimization studies have shown this fiber to be an ideal fiber for phthalate analysis (Kasuske, Medrano, et al., 2025; Liu, 2008; Luks-Betlej et al., 2001; Peñalver et al., 2001; Perestrelo et al., 2021; Polo et al., 2005). Prior to sampling, all fibers were conditioned for three 30 min sessions in a 250 ° C oven. A blank fiber instrument run followed to ensure the absence of contaminants on the fiber. For phthalate extraction, sample vial water (2 mL) was allowed to acclimate at 40 ° C in a dry bath block heater (Cole-Palmer, Vernon Hills, IL) for 5 min. Prior to SPME fiber injection sample vials were sonicated for 30 seconds to ensure a homogenous mixture. The fiber was then directly immersed into the sample vial for 30 min at 40 ° C, based on previous optimization studies. After 30 min of fiber immersion the fiber was then manually injected into the GC/MS column for sample runs. Blank fiber instrument runs were performed prior to each sampling period to ensure cross contamination and carryover between samples did not occur. LC-MS water was used as a blank between countries and respective samples. Glassware was cleaned with acetone, DI-water, and baked in an oven for 8 h at 100 °C to avoid contamination. Linearity of analytes was determined via external calibration standards ranging from 1–500 µg/L. Supplemental Information (SI) Table 1 shows the concentration ranges and correlating coefficients for respective phthalates. SPME extraction efficiency was conducted in accordance with EPA methodology (SW-846 and Method 8061A). Briefly, for recovery seven individual LC-MS grade water replicates were spiked with 250 µg/L of phthalate mixture and then processed and analyzed in the same manner as experimental samples. Average recoveries were calculated and are presented in SI Table 2. Due to some extractions exceeding 100%, an extraction efficiency factor was applied to all subsequent phthalate concentrations, and all values are recovery and blank corrected.

## 2.4 GC/MS

GC/MS analysis was performed using an Agilent Technologies 7890A GC system equipped with a 5975C inert XL MSD with a Triple-Axis Detector (Santa Clara, CA) and a Rtx-5 capillary column 30 m x 250 µm x 0.25 µm (Restek Corporation, Bellefonte, PA, USA). The GC oven temperature program consisted with methodology described in Kasuske et al., 2025. Briefly, initial temperature of 50 ° C; increased to 260 ° C at a rate of 10 ° C/min with a hold time of 3 min. Total run time was 24 minutes. The following temperatures were used: inlet at 270 ° C, detector at 280 ° C, MS Source at 230 ° C, and the MS Quad at 150 ° C. Helium was the carrier gas with a flow rate of 1.3 mL/min. The analysis was conducted in splitless mode with a 3 min solvent delay.

The mass spectrometer was operated in electron ionization (EI) mode at 70 eV and scanned over a mass range of 40 – 400 m/z. Dwell time was 30 ms and acquisition was conducted using synchronous Selective Ion Monitoring (SIM) mode, see SI Table 3 for respective ions used within SIM mode and other parameters. Data was acquired using the Agilent ChemStation (MSD ChemStation F.01.03.5357) equipped with the NIST 17 library. External calibration of GC-MS was carried out prior to sampling, with calibration points ranging from 1 – 500 µg/L with a correlation coefficient ( $R^2$ ) ≥ 0.99. Sample volume for direct injection was 1 µL. Limit of detection (LOD) and LOQ (limit of quantification), Equation 1 and 2 respectively, were calculated in accordance with International Conference on Harmonization (ICH) guidelines (HARMONISATION, 1994) (SI Table 4).

$$\text{LOD} = 3.3 \cdot \sigma / S \quad \text{Eq. 1}$$

$$\text{LOD} = 3.3 \cdot \sigma / S \quad \text{Eq. 2}$$

$\sigma$  = standard deviation of response, S = slope of calibration curve

## 2.5 Statistical analysis

Data was analyzed using JMP Pro 16 (version 16.0.0). Mean values are expressed as mean  $\pm$  standard error of the mean (SEM). A repeated measures ANOVA and Tukey's HSD post-hoc multiple comparisons tests were conducted to look at statistical differences between phthalate leachate at respective timepoints and respective countries. A p value < 0.05 was considered significant and all tests  $\alpha$  = 0.05.

## 3) Results and Discussion

### 3.1 Total phthalate leachate

It was found that there was significantly more total phthalate leached on day 7 for US bottles than on days 0, 1, or 3. Whereas for the Oceania and Scotland bottles, there were no significant differences in leaching of total phthalate over the 7-day period (Fig. 1).

However, between day 0 and day 7 there was a 63.8% increase, 20.4% increase, and a 55.1% decrease, in total phthalate leachate for the US, Oceania, and Scotland bottles, respectively. Repeated measure ANOVA results (Wilks' Lambda, Pillai's Trace, and Roy's Max Root, all  $p < 0.0001$ ) also showed significant interactions between total PAE leachate and timepoint, as well as all within interactions. It was also found that there was significantly ( $p < 0.05$ ) more total PAE leachate detected within US day 7 samples ( $629 \pm 97 \mu\text{g/L}$ ,  $34 \pm 5 \mu\text{g/g/bottle}$ ) relative to Scotland day 7 ( $198 \pm 60 \mu\text{g/L}$ ,  $11 \pm 5 \mu\text{g/g/bottle}$ ) and Oceania day 3 ( $267 \pm 76 \mu\text{g/L}$ ,  $16 \pm 4 \mu\text{g/g/bottle}$ ) samples. One explanation for the decrease in leachate within the Scotland samples is that the manufacturing, storage, travel, bottling, and shelf conditions prior to sampling in this study are unknown. It is entirely possible that conditions prior to this study were not ideal (temperature or UV) and accelerated diester conversion. Given that these samples were additionally shipped from overseas it is entirely possible. This is discussed further below. Regardless, these results still highlight differences between policy, inclusion of additives, and human exposure to phthalates within countries.

Interestingly there have been some recent updates regarding inclusion rates of phthalates within consumer products. As of 2022 Scotland has implemented a ban on single-use polystyrene plastics ranging from cutlery to beverage containers rather than managing additive inclusion through their Environmental Protection Regulations in 2021 (Government, 2021). As of right now this ban only covers polystyrene, hopefully in the future this will be expanded to cover other single-use plastics like PET. However, as of March 2026 a new UK REACH regulatory framework regarding phthalate inclusion within consumer products has gone into effect. Phthalates like DEHP, DBP, BBP, DIBP are now restricted to less than 0.1% by mass in materials, whereas DINP, DIDP, and DNOP are completely restricted from children's toys, goods, and food contact materials (Government, 2026). Regarding the broader region of Oceania there is not a region-specific policy currently in place, however countries like New Zealand and Australia are adopting the new inclusion rates (0.1% by weight) now in place in the UK. Additionally, the US seems to be following suit, as of early 2026 Section 108 of the Consumer Product Safety Improvement Act through the Consumer Product Safety Commission (CPSC) now prohibits the manufacture, sale, distribution, or importation of children's toys and child care articles that contain more than 0.1% (1000 ppm) of DEHP, DBP, BBP, DINP, DIBP, DPENP, DHEXP, and DCHP (Commission, 2026). So there seems to be a global movement to ban or significantly reduce inclusion of harmful plastic additives within consumer and child products. With regard to drinking water guidelines and standards for human exposure, the current FDA allowable limit for DEHP is 6 ppb ( $\mu\text{g/L}$ ) in bottled water following EPA's Maximum Contaminant Level (MCL) for drinking water. However, there is currently not an allowable limit set for any of the other phthalates detected within this study. While some DEHP values presented within this study do exceed the allowable 6 ppb ( $\mu\text{g/L}$ ) level, exposure may pose some unintended risk to consumers, but subsequent risk and hazard assessments are necessary to verify.

### 3.2 Individual phthalate leachate

While BPA was not detected in any samples, there were several significant findings regarding individual phthalates across countries. In Oceania samples there was significantly more total DEHP ( $383 \pm 131 \mu\text{g/L}$ ) detected across the 7-day sampling period relative to DMP ( $177 \pm 30 \mu\text{g/L}$ ). Within Scotland samples, BBP was not detected at any timepoints. However, over the sampling period, it was found that there was significantly more total DEP detected ( $712 \pm 81 \mu\text{g/L}$ ) relative to DBP ( $214 \pm 43 \mu\text{g/L}$ ), DEHP ( $203 \pm 115 \mu\text{g/L}$ ), and DMP ( $141 \pm 35 \mu\text{g/L}$ ). For the U.S. samples, there were no significant differences found between individual total phthalate concentration over the 7-day sampling period. However, there were similar trends observed regarding frequency of occurrence of both DEP and DEHP at specific timepoints. There was significantly more DEHP detected on day 7 ( $204 \pm 34 \mu\text{g/L}$ ) than day 0 ( $65 \pm 39 \mu\text{g/L}$ ). There was also significantly more DEP detected on day 7 ( $163 \pm 19 \mu\text{g/L}$ ) than on day 0 ( $86 \pm 11 \mu\text{g/L}$ ) (Table 1).

Table 1  
Total PAE leachate over time ( $\mu\text{g/L} \pm \text{SEM}$ )

| USA      | Timepoint | DMP         | DEP           | DBP          | BBP          | DEHP         | Total         |
|----------|-----------|-------------|---------------|--------------|--------------|--------------|---------------|
|          | 0         | $34 \pm 8$  | $86 \pm 11$   | $66 \pm 13$  | $71 \pm 22$  | $65 \pm 39$  | $324 \pm 96$  |
|          | 1         | $67 \pm 22$ | $119 \pm 29$  | $84 \pm 6$   | $61 \pm 21$  | $52 \pm 22$  | $386 \pm 102$ |
|          | 3         | $5 \pm 3$   | $78 \pm 3$    | $62 \pm 4$   | $40 \pm 25$  | $106 \pm 41$ | $293 \pm 77$  |
|          | 7         | $65 \pm 8$  | $163 \pm 19$  | $96 \pm 6$   | $100 \pm 29$ | $204 \pm 34$ | $629 \pm 97$  |
| Oceania  | Timepoint | DMP         | DEP           | DBP          | BBP          | DEHP         | Total         |
|          | 0         | $36 \pm 3$  | $80 \pm 3$    | $60 \pm 1$   | $58 \pm 29$  | $138 \pm 38$ | $374 \pm 75$  |
|          | 1         | $67 \pm 16$ | $113 \pm 7.2$ | $78 \pm 3$   | $69 \pm 24$  | $155 \pm 25$ | $483 \pm 77$  |
|          | 3         | $14 \pm 2$  | $99 \pm 11$   | $79 \pm 8$   | $45 \pm 10$  | $38 \pm 31$  | $267 \pm 76$  |
|          | 7         | $60 \pm 9$  | $153 \pm 20$  | $109 \pm 14$ | $74 \pm 27$  | $60 \pm 37$  | $459 \pm 110$ |
| Scotland | Timepoint | DMP         | DEP           | DBP          | BBP          | DEHP         | Total         |
|          | 0         | $49 \pm 10$ | $159 \pm 9$   | $72 \pm 6$   | 0            | $55 \pm 7$   | $336 \pm 62$  |
|          | 1         | $54 \pm 19$ | $194 \pm 20$  | $56 \pm 11$  | 0            | $69 \pm 4$   | $375 \pm 96$  |
|          | 3         | $38 \pm 6$  | $214 \pm 30$  | $57 \pm 17$  | 0            | $47 \pm 30$  | $357 \pm 83$  |
|          | 7         | 0           | $145 \pm 22$  | $29 \pm 9$   | 0            | $23 \pm 22$  | $198 \pm 60$  |

The results indicate that while all bottles exhibited some phthalate leaching over time, the U.S. bottles showed a greater overall increase in total phthalate concentration by day 7. Similarly, *Cao et al. (2008)* quantified phthalate leachate from PETE bottles from Canada using headspace SPME analysis. They found DEP, DBP, and DEHP at concentrations of 80, 138, and 118 ng/L, respectively (Cao, 2008). While some of the values presented in this work are higher than reported here, this is likely due to inclusion of higher additive and recycled material in these bottles than those used in 2008. Kawamura et al., (2000), analyzed nonylphenols and phthalates within consumer plastic toys and products. They found several phthalates (DBP, DEP, and DEHP), with DEHP being the most prevalent phthalate at a median concentration of 350 ng/L (Kawamura et al., 2000) which agrees with the concentrations found in the present study. Casajuana and Lacorte (2003), found that phthalate leachate from PET bottles increased over the course of a 10-week period with mean concentrations ranging from 0.002–0.214  $\mu\text{g/L}$  (Casajuana & Lacorte, 2003). Jeddi et al., (2016), examined the effects that temperature has on phthalate leachate in PET bottles. It was found that DEHP was

the most abundant phthalate detected across all samples and temperatures with  $0.418 \pm 0.196 \mu\text{g/L}$  (outdoor conditions),  $0.917 \pm 0.342 \mu\text{g/L}$  ( $40^\circ\text{C}$ ), and  $0.411 \pm 0.161 \mu\text{g/L}$  (room temperature)(Jeddi et al., 2016). Xu et al., (2020) conducted similar studies looking at phthalate leachate from 10 different PET brands found in Beijing. DEP, DMP, and DBP were detected in all 10 bottles and had concentrations ranging from  $101.97 \mu\text{g/kg}$  to  $707.87 \mu\text{g/kg}$  which are within the ranges detected in this study. Values detected within the water itself ranged from  $0.19\text{--}1.0 \mu\text{g/L}$ (Xu et al., 2019). However, recycled content was not discussed within the study so while room temperature and storage conditions were similar it is possible that increased recycled content is responsible for higher values presented within this study. Massahi et al., (2025) studied the effects that sunlight and temperature have on DEHP, DBP, and BBP migration from PET bottles. The authors found that exposure to sunlight induced significant phthalate leachate relative to bottles stored in the shade. Samples exposed to sunlight had DEHP, DBP, and BBP concentrations ranging from  $2,042\text{--}8,432 \text{ ng/L}$  whereas shaded samples had concentrations ranging from  $1,377\text{--}4,191 \text{ ng/L}$  (Massahi et al., 2025). With respect to the bottles used within this study, the authors are unaware of processing and storage conditions prior to sampling. It is possible that prior to sampling the bottles were exposed to sunlight or increased temperatures, both have been shown to increase phthalate leachate and could explain why higher values were observed in this study. Another reason that could explain higher phthalate concentration could be contributed to potential phthalate accumulation post-manufacturing. There could have been significant time between which these bottles were manufactured, filled, shipped, and stocked and subsequently sampled for this study. So, while a 7-day period is applicable based on consumer habits, it only represents a snapshot within the lifetime of the bottle and longitudinal studies should be considered. Recently, using EPA methodology Consumer Reports published data regarding phthalates detected in 67 US consumer food products. Several common beverage containers were analyzed, and it was found that values ranged from  $0\text{--}20,452$  total phthalate (ng) per serving (Reports, 2024).

## 4) Conclusion

This study highlights how inconsistencies or differences in global regulatory policies can affect phthalate inclusion rates and contaminant migration from PET bottles. This issue does not just apply to phthalates though, previous work by Li et al., (2025) shows that contaminants detected within rPET and virgin PET bottles are highly variable and differ from region to region within the US. Within rPET bottles, benzene was frequently detected along with several organophosphate esters (OPEs) while virgin materials had higher ethylene glycol and 2-methyl-1,3-dioxolane levels (Li et al., 2025). These findings support the hypothesis that stricter regulations may help reduce phthalate and other contaminants in consumer plastics. While the values presented within this work are well below the 1000 ppm (0.1%) inclusion rate that is being adopted globally, there is additional pressure to reduce global plastic production as well. One of the proposed interventions to curb plastic production is recycling. However, it has been shown that as recycling content increases so does additive or phthalate content and this aspect is often missing from Life Cycle Assessments (LCAs) as discussed extensively by Logan et al., (2024) (Logan et al., 2024). With this in mind, future research should explore long-term exposure effects, additional environmental factors, and broader sampling to further assess the effectiveness of regulatory measures. These insights can help inform policymakers and industry leaders in promoting safer plastic packaging practices worldwide.

## Declarations

## Conflict of Interest

The authors declare no conflicts of interest.

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There are no funding sources supporting the presented work.

## Author Contribution

ZK: Conceived and designed the experiments, Analyzed and interpreted the data, Wrote the paper. JCC: Reviewed and edited paper, supervised experiment. All authors whose names appear on the submission: approved the version to be published.

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## Data Availability

Supporting Information can be found in the online version of the manuscript. More information can be made available upon request from the corresponding author (jaclyn.e.canas@ttu.edu).

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# Figures

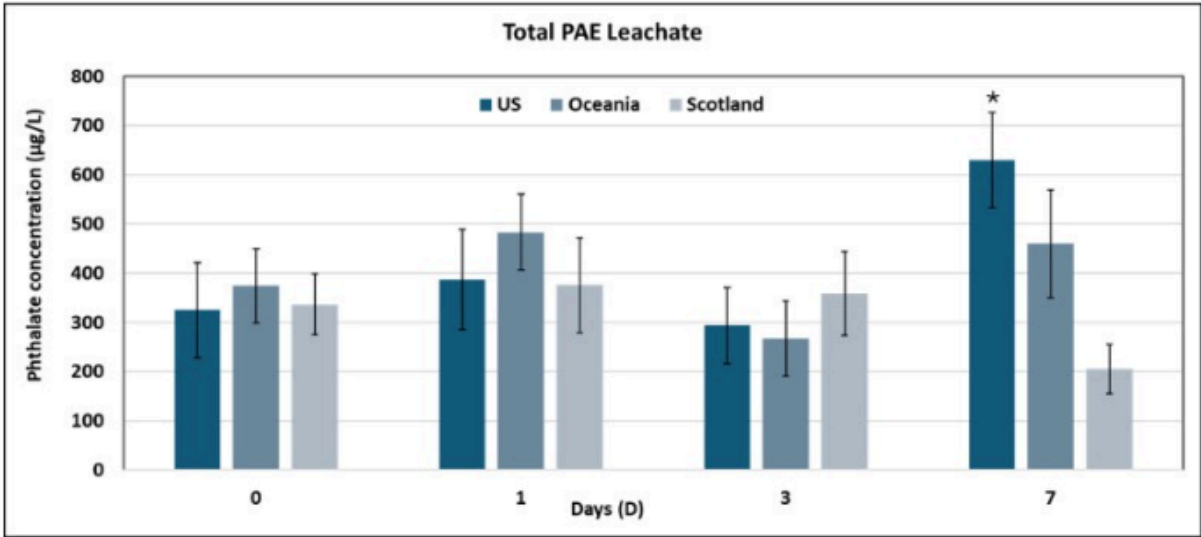


Figure 1

Total average phthalate leachate per country over 7-day period, error bars represent ± SE (n = 3 per group). Asterisk (\*) represent Tukey’s comparisons denoting a statistically significant (p < 0.05) within countries.

## Supplementary Files

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