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Assessing Plastic Brittleness to Understand Secondary Microplastic Formation on Beaches: A Hotspot for Weathered Marine Plastics

Astrid E. Delorme[1,2,3*], Laurent Lebreton[3,4], Sarah-Jeanne Royer[2,3], Kimeona Kāne[5], Mael Arhant[1], Maelenn Le Gall[1] and Pierre-Yves Le Gac[1]

Abstract

The degradation and fragmentation of plastic debris into secondary microplastics pose significant environmental challenges, particularly on beaches where mechanical abrasion and chemical weathering accelerate plastic fragmentation. This study contributes to the understanding of secondary microplastic formation by addressing key questions related to plastic embrittlement, mechanical behavior, and degradation processes. We investigate the brittleness of polyethylene (PE) and polypropylene (PP) particles collected from Hawaiian beaches, focusing on their mechanical and chemical degradation states. Conventional tensile testing methods are impractical for irregularly shaped, small field-recovered particles. Therefore, we used a simple fragmentation test to evaluate brittleness under fixed applied pressure, enabling large-scale statistical analysis of PE and PP samples from beach field surveys. We show that the brittle samples have a very low molecular weight (*Mw*). Such low *Mw*, coupled with the appearance of oxidation products, suggests an advanced degradation state of the sampled plastics. Through our fragmentation test, we provide a large-scale, field-based quantification of plastic brittleness in beach samples, underscoring an increased propensity for further fragmentation and highlighting the severity of coastal and ocean plastic pollution.

Introduction

Numerical models have shown that even with an 80% reduction in plastic inputs from 2020, surface ocean plastic debris is projected to continue increasing until 2040 [1, 2]. Exacerbating this increasing input of ocean plastic pollution is that the degradation of these legacy plastics continues [1], leading to further fragmentation, abrasion, and breakdown into smaller fragments, generating secondary microplastics— plastic particles ≤5 mm formed from the physical and chemical weathering of larger plastics [3–5]. The continuous influx of microplastic fragments increases the bioavailability of plastic particles to marine organisms [6, 7], ultimately jeopardizing the ecological integrity, cultural practices, wellbeing and livelihood of communities reliant on healthy ocean and coastal ecosystems. Thus, with the world currently navigating the final negotiations of the Global Plastics Treaty, it must prioritize stopping the flow of plastic into the ocean while also addressing legacy plastic pollution, particularly in accumulation hotspots [2]. Understanding where, how and when legacy plastic fragments, can inform targeted interventions such as beach cleanup and open-ocean cleanup efforts (e.g. location, frequencies and mitigation methods) and can provide a 'last chance' to remove plastics before they further degrade

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*Correspondence:
Astrid E. Delorme
astrid.delorme@ifremer.fr
[1] Ifremer RDT, Research and Technology Development Unit, 1625 Route de Sainte-Anne, 29280 Plouzané, France
[2] Center for Marine Debris Research, Hawai'i Pacific University, Waimānalo, HI, USA
[3] The Ocean Cleanup, Rotterdam, The Netherlands
[4] The Modelling House Limited, Raglan, New Zealand
[5] Community Scholar, Cultural Practitioner, Chairperson of the Waimānalo Neighborhood Board, Waimānalo, Hawaii, USA

into the much more challenging secondary microplastics [1, 8, 9].

Notably fragmentation of plastic particles onshore has been observed to be faster than in seawater due to high temperature (heat-build up in the sand), high oxygen levels, high exposure to solar radiation (due to little vegetation coverage) and mechanical forces from waves and sand movements, which accelerates the cracking and fragmentation of plastic debris [10–12]. These transformations affect the mechanical properties and density of plastic particles, influencing its transport, accumulation, and buoyancy [13, 14]. Beaches, which are at the land–ocean interface, are thus critical zones for understanding factors driving secondary microplastic formation. Furthermore, Hawai‘i, situated near the North Pacific Garbage Patch, is heavily impacted by floating ocean plastic pollution, with beaches receiving substantial amounts of highly weathered plastic [15–19]. Hawaiian shores are not only important hotspot of weathered ocean plastic pollution, but also living ecosystems that form the foundation of Hawai‘i's identity and wellbeing. The risk of continued microplastic inputs further threatens these environments, disrupting ecosystems, cultural practices, and community livelihoods. For example, loko i‘a (Hawaiian fishponds) are currently actively restored to revitalize traditional aquaculture practices, where fish feed on nutrient-rich waters flowing from the land [20–22]. However, these same waters now carry microplastic pollution, increasing the exposure of juvenile and coastal fish—key to food security and cultural traditions—to pervasive and bioavailable plastic particles. To better assess the current risk of beached plastics on Hawaiian shores fragmenting into secondary microplastics, we examined key polymer characteristics—such as brittleness and degradation state—that influence their susceptibility to fragmentation.

However, investigating the"brittleness"of plastics—a key precursor to fragmentation— in the environment remains challenging due to the irregular shapes, weathered conditions, and small sizes of environmental samples, which often fail to meet the requirements for standardized mechanical testing [23]. Over 70 years of polymer degradation research have established that embrittlement of polymers, which is defined as the loss of ductility and when they transition from ductile to brittle materials, can be quantified by a critical molecular weight, commonly referred to as the critical molar mass, *M'c* [24, 25]. Despite the fact that *M'c* does not depend on the initial molecular mass, it is worth noting that the *M'c* value depends on the experimental characterization conditions, such as temperature or loading rate. When the average molecular weight (*Mw*) of a polymer falls below *M'c*, it becomes brittle; above this threshold, it remains ductile as can be confirmed in stress–strain curves from standardized tensile testing [24]. Semicrystalline polymers such as polyethylene (PE) and polypropylene (PP) have been shown, through accelerated aging tests, to undergo a decrease in *Mw* with aging due to oxidation, or UV-induced oxidation [4, 10, 24, 26–29]. These chemical degradation processes lead to chain scissions (and thus lower *Mw*) in the amorphous phase of the polymer, reducing entanglements and increasing the crystalline phase due to the thickening of crystalline lamellae driven by the increased mobility of macromolecular chains—a process known as chemicrystallization—leading to polymer embrittlement [24, 30, 31]. Changes in crystallinity ratio also influence the thickness of the amorphous layer, which could serve as relevant factor in more detailed studies of polymer fracture. A notable advantage is that, regardless of a polymer's initial state, chemical aging typically causes a transition from ductile to brittle, with *M'c* serving as a key measure of this change and is therefore independent of the type of chemical degradation, initial molecular weight, or presence of stabilizers in the polymer formulation [32–34]. For PE *M'c* is 70 kg/mol, and for PP it is 200 kg/mol with values derived from accelerated aging tests [24]. These *M'c* values depends on the testing temperature, due to the increase in mobility with temperature [35], and the embrittlement criteria which is commonly the standardized tensile testing [23, 36]. While these principles of embrittlement are well-established in controlled conditions, their translation to environmental plastics requires an approach that accounts for the impossibility to apply standard tensile testing methods directly on environmental samples of plastic particles.

This study aims to advance the understanding of secondary microplastic formation by addressing key questions related to the mechanical and chemical degradation of plastics on beaches. First, are the plastic particles found on beaches brittle or ductile? Conventional mechanical tensile testing methods to test the brittleness are impractical due to the small, irregular sizes of field-recovered plastic particles. Therefore, a simple fragmentation test is employed to determine whether beach-sampled plastic particles break or remain intact under a fixed pressure, enabling the collection of a large dataset for statistical analysis. Additionally, the study investigates whether the brittleness of plastics correlates with the carbonyl index, a common measure of polymer oxidation evolution on the surface of the material [37, 38]. Another question explored is whether polymer embrittlement under our fragmentation test is associated with low *Mw*, providing a physicochemical perspective for understanding of the brittle plastics on beaches. The potential influence of brittleness on the size distribution of plastics found on beaches is also examined.

We also critically reflect on our study which was conducted on Native Hawaiian land, yet we did not involve or inform the local communities about our research during its initial phases. This oversight and subsequent connections with a cultural practitioner, enabled us to reflect more deeply on the cultural implications of our research and recognize the potential harm caused by our initial lack of engagement with the local communities. Drawing from Liboiron et al. [39] concept of reconciliation science, we integrate our reflections directly into the scientific process rather than separating them into distinct social science or opinion pieces. Through this lens, we assess our research methods and findings in the Results and Discussion section, with the hope to prevent further harm and ensure our work is conducted with greater respect for the cultural and environmental values of the communities our work is intended to serve.

Methods

Study area

The field sampling for this study was conducted on Oʻahu, the third-largest and most populated island in the Hawaiian archipelago, located in the North Pacific Ocean [40]. The three beaches selected for this research—Kahuku (21°42′09.0"N 157°57′36.0"W), Kokololio (21°37′41.2"N 157°55′15.6"W), and Waimānalo (21°20′06.0"N 157°41′45.6"W)—are all situated on the island's windward side, which is consistently exposed to easterly trade winds. These trade winds generate short-period waves year-round, contributing to the transport and accumulation of marine debris. The location of Oʻahu near the North Pacific Garbage Patch and the persistent influence of the trade winds, result in the windward side being particularly vulnerable to the influx of plastic debris [41–43]. The beaches were selected due to their locations on the windward side of Oʻahu, and their varying beach characteristics and plastic accumulation patterns, with Kahuku known for accumulating the largest amounts of plastics at the surface [41, 44], however not buried [45].

Permits

The beach site at Kahuku was located just outside the James Campbell National Wildlife Refuge, requiring a permit to access. This permit was secured from the US Fish and Wildlife Department prior to the commencement of fieldwork and remained valid for the duration of the study. No additional permits were necessary for plastic removal activities on the beach, as confirmed by the Department of Land and Natural Resources (DLNR) and the Division of Aquatic Resources (DAR), provided that only plastic was removed.

Analysis of methods for reconciliation science

In our effort to align with reconciliation science as conceptualized by Liboiron et al. [39], we critically reflected on our methods through collaboration with cultural practitioner Kimeona Kāne. Reconciliation science, as described by Liboiron et al. (and how we interpret it), emerges from the recognition that scientific research often perpetuates colonial frameworks and extractive practices that often overlook or undermine local cultural practices and culture-ecological interconnections. This, together with our collaboration with Kāne provided guidance for how we reflected on our work in Hawaiʻi, especially after realizing that our initial methods failed to adequately consider its cultural and ecological implications for the local Hawaiian community.

Beach plastic sampling

All sites considered in this study were sampled every three months over 15 months, starting in November 2022, with subsequent samplings in February 2023, May 2023, August 2023, November 2023, and February 2024. Sampling was carried out as we describe in detail in Delorme et al. [45]. In summary, sampling involved triplicate quadrats (60 × 60 cm) placed 10 m apart (measured from the quadrat centers) and aligned parallel to the shoreline along the drift line (berm). The top 2 cm of sand were excavated to collect surface debris (through floating separation). Sand from successive 10 cm layers was then excavated and floating debris was collected every 10 cm layers down to a depth of 102 cm. In total, 594 samples were collected across 11 depth layers, six sampling events, three beaches, and three quadrats per beach. It should be noted that only buoyant polymers were recovered using the Buoyancy Separating Device (BSD) in the field to separate the plastic from the sand using seawater. However, as noted in Brignac et al. [41], > 90% of particles recovered on the windward side of Oʻahu are less dense plastic materials and buoyant plastics. Thus it should be noted that the occasional detection of denser polymers (e.g., Nylon and PVC) does not accurately reflect their true environmental frequency.

Plastic sampling processing

The mesh bags (250 μm) with the floating debris collected per stratified layer were air-dried indoors for at least a week after collection, after which samples were sieved through a 500 μm stainless steel sieve to remove smaller particles. The plastic debris was manually and visually separated from the natural debris and the plastic debris was then laid out on a blue background sheet alongside a reference coin of 37 mm diameter and were photographed from a 1 m distance with a resolution of at least 10 pixels in diameter, with roughly a Ground Sampling Distance of 0.1 mm/pixels. The image was processed using the Segmentation Model developed by The Ocean Cleanup, where the model workflow is described by Royer et al. [46]. This model classifies each plastic particle into four classes (hard fragment, pellet, line, and foam), into 13 colors (black, white, blue, green, red, orange, salmon, yellow, lightblue, lightgreen, indigo, turquoise, and lightgray), and measures the size of the particles (minimum and maximum length), among other things.

Polymer identification

Attenuated Total Reflectance/Fourier Transform Infrared spectroscopy (ATR/FTIR) was used to determine the bulk polymer identity of sampled plastic particles collected during field surveys. Sample sizes ranged from 0 to 2,079 particles per layer sampled in the sand-column. When a sample contained more than 60 particles, a random subset of 60 particles was selected for polymer identification analysis. This threshold was determined based on FTIR analysis of four subsample sets, totaling 680 particles. The results showed that randomly selecting 60 particles provided a 95% match to the overall polymer distribution, ensuring sample representativeness [45]. The background and sample spectra were recorded between 4000 cm^{-1} and 550 cm^{-1} with a resolution of 4 cm^{-1} and 16 scans on a Nicolet iS5 FTIR spectrometer equipped with an iD5 ATR module. The main peaks in the spectra were identified, using the OMNIC™ Spectra Software, to determine the functional groups present and establish the polymer identity, following the method described by Jung et al. [47]. For samples requiring further verification or those that could not be identified manually, spectral libraries within the OMNIC™. Spectra Software were consulted, provided the search score was ≥ 0.90 (on a scale from 0 to 1). Samples that remained unidentified after these steps were labeled as UnI (UnIdentified).

Carbonyl index

The carbonyl index (CI) was calculated following the method described by Almond et al. [37] and Celik et al. [38], using the Specified Area Under Band (SAUB) technique. Accordingly, the CI as calculated from the ratio of the integrated band absorbance of the carbonyl absorption band (1850–1600 cm^{-1}) relative to the methylene scissoring peak (1500–1420 cm^{-1}), as expressed by Eq. 1. The areas for both the carbonyl and methylene bands were extracted using the peak analysis tool in Spectragryph 1.2 spectroscopy software (Friedrich Menges, Oberstdorf, Germany). To minimize manual error and bias, baseline corrections were standardized to "y = 0" using Spectragryph 1.2, as described by Celik et al. [38].

$$CI = \frac{Integrated\ Carbonyl\ Band\ Area\left(1600 - 1850\ cm^{-1}\right)}{Integrated\ Methylene\ Band\ Area\left(1420 - 1500\ cm^{-1}\right)} \tag{1}$$

Analysis of brittle vs ductile under ATR/FTIR pressure

For the 16,322 plastic particles that were analyzed using ATR/FTIR analysis we noted whether each particle fractured or remained intact under applied ATR pressure by visual inspection and labeled them as “brittle” and “ductile”, respectively. The standard ATR/FTIR analysis involves pressing the sample using a pressure tower equipped with either a concave tip or a flat surface tip. In our case the ATR was equipped with a smooth concave tip of 4 mm diameter. In conventional ATR/FTIR instruments, the applied pressure is set within the pressure device so that it stops automatically via a slip clutch mechanism to prevent over-pressurization and ensure reproducibility with consistent pressure on all the samples analyzed [48]. As we collected the IR spectra, the pressure knob was always tightened until the audible click was heard, indicating that the preset maximum pressure had been reached. Samples that fracture under this applied load were labeled as brittle, while those that did not were labeled ductile. This setting was consistent for all samples analyzed under the ATR/FTIR. An image of the ATR module used in our study is shown in Figure S1 in the SI.

To determine the pressure exerted by the ATR/FTIR pressure tower on the sample, we measured the resistance experienced when the preset pressure was applied using a Fluke 83 multimeter and resistance sensors from Ohmite Force Sensing Resistors (type: FSR07). We used four sensors and measured the resistance five times per sensor, the resistance values recorded during each test were averaged to obtain the mean resistance for each sensor and the results are summarized in Table S2 in the SI. The force exerted by the ATR tip was then calculated through calibration curves, generated using an Instron machine and equipping it with a 4 mm concave screw. Two sensors were used to measure the resistance felt at different known applied pressures. From the calibration curves (Figure S2), we calculated the force and pressure applied of the ATR pressure tower. We calculate a 4.5 Newton force, thus 1.1 MPa on a 4 mm2 diameter area. For comparison, this pressure is similar to the force exerted by a fingertip pressing on hard surfaces [49].

Data analysis

To evaluate the influence of polymer type (PP and PE) and fragmentation state (brittle and ductile) on particle size, a two-way analysis of variance (ANOVA) was conducted. The fixed effects in the model were polymer type (PE or PP) and fragmentation state (brittle or ductile), with their interaction term also considered. The ANOVA model was designed to assess how particle size varied with polymer type and fragmentation state, as well as to determine whether the interaction between these factors was significant. Residual variance was assessed as part of the model fit, ensuring that the differences in particle size could largely be attributed to the main effects and their interaction. All statistical analyses were performed using R software (version 2024.04.2), with the ANOVA conducted using the Car package to evaluate the significance of the fixed effects.

Gel Permeation Chromatography

Gel Permeation Chromatography (GPC) was used to analyze the chain length distribution and obtain the *Mw* of a subsample of 17 PE and PP particles, randomly selected from a field sampling conducted on 25/02/2023 at Kokololio Beach. The selected particles included four ductile PE particles, five brittle PE particles, four ductile PP particles, and four brittle PP particles. For each sample, approximately 20 mg of particle was dissolved in stabilized trichlorobenzene (TCB) at 160 °C for two hours. The solution was then injected into a GPC system equipped with a set of two 8.0 mm ×300 mm Agilent Mixed-B ID columns at a flow rate of 1 ml/min. The analysis was performed at a temperature of 150 °C, and a differential refractive index detector was used for detection. Calibration was carried out using narrow polystyrene standards across a molar mass range of 500 to 80,000,000 g/mol. The standard error of the experiment was 5% for *Mw* values.

Results and discussion

Fragmentation of sampled plastic particles

A total of 77,033 plastic particles were recovered from 594 samples collected across three beaches—Kahuku, Kokololio, and Waimānalo. This resulted in the FTIR/ATR analysis of 16,322 particles, for which we recorded whether each particle was brittle or ductile under the applied pressure of the ATR pressure tower. Of the 16,322 particles analyzed, 89% were classified as brittle. The distribution of particle counts, polymer identification, shape, and size for the entire sample collection can be found in Delorme et al. [45]. For this study, we focused on PE and PP particles, which accounted for 92% (15,061 particles) of the total sample, to explore the correlation between brittleness in these semi-crystalline materials whose embrittlement have been extensively studied in laboratory settings [24, 25, 30, 31] and also represent the majority of plastics found on Hawaiian east-facing beaches [41]. The percentages of brittle and ductile PE and PP particles under ATR pressure are summarized in Table 1 (see Table S2 in the SI for details on all polymers identified in our analysis).

This relatively low pressure (1.1 MPa) caused a significant number of particles to fracture during ATR/FTIR analysis, indicating that 91.8% and 92.6% for PE and PP particles, respectively, would fracture under this pressure. This finding aligns with our field sampling using the Segmentation Model, which revealed that 93% of recovered particles were small fragments, with mean sizes ranging from 5.4 to 7.9 mm [45], highlighting their potential for further fragmentation into smaller particles in dynamic coastal environments.

Analysis of brittleness-inducing properties

To assess the correlations of brittleness with the CI, *Mw* and their sizes of the sampled PP and PE particles, we investigated a subsample of 17 PE ($n = 9$) and PP ($n = 8$) samples, including both brittle and ductile particles, focusing on their physicochemical traits that might related to their extent of oxidation, weathering and links to their observed brittleness. Images of the 17 analyzed particles are summarized in Table S3 in the SI.

Infrared spectroscopy analysis

PP and PE undergo chemical changes when exposed to environmental stressors such as UV radiation and oxygen, leading to the formation of various degradation products. These chemical changes can be observed in their infrared spectra. For example, weathered samples often exhibit a broad peak in the carbonyl region indicating the presence of oxidation products such as lactones, esters, ketones, and carboxylic acids [50–52]. Such peaks were also present in the spectra of PP and PE samples collected on the beaches, as illustrated in Fig. 1.

Figure 2 shows the calculated CI values for the 17 analyzed PE ($n = 9$) and PP ($n = 8$) samples. Samples that

Table 1 Percentages and counts of PE and PP particles collected on Hawaiian beaches that were determined as brittle or ductile under ATR pressure (4.5 N over 4 mm^2)

Polymer ID	Brittle/Ductile	Count	Percentage of Polymer ID
PE	Brittle	9804	91.8
PE	Ductile	880	8.2
PP	Brittle	4055	92.6
PP	Ductile	322	7.4

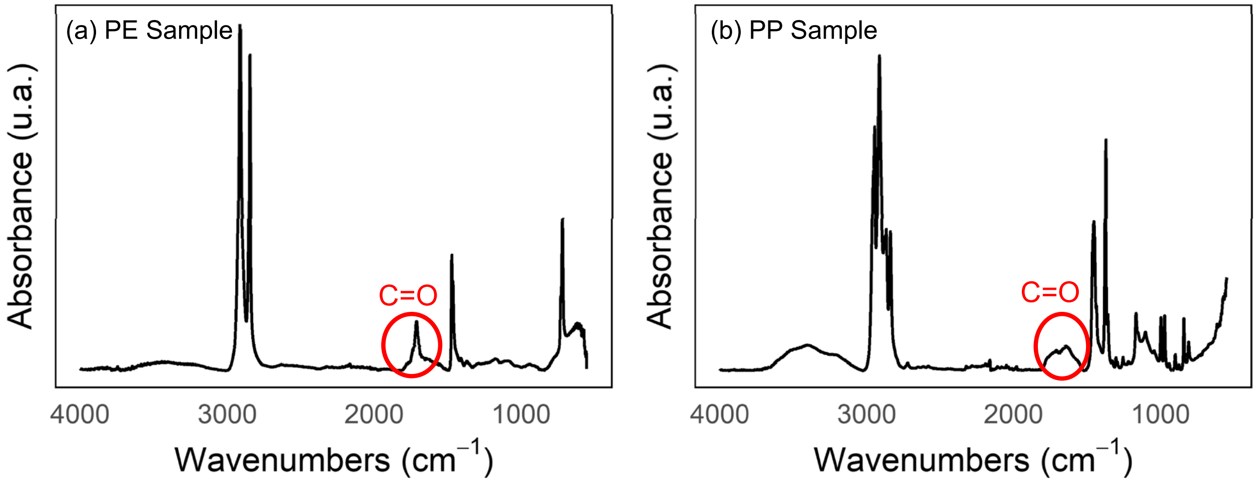

Fig. 1 FTIR spectra of collected plastic samples from field sampling. (**a**) PP samples and (**b**) PE samples, showing absorbance across wavenumbers (4000–550 cm^{-1}). The carbonyl band for each spectrum is highlighted

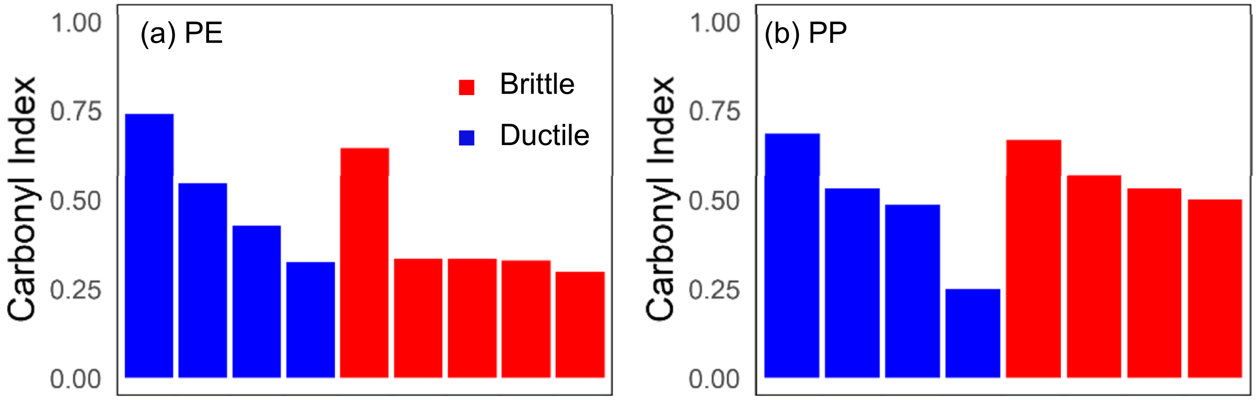

Fig. 2 The CI for the 17 subsampled PE and PP particles, calculated from the ratio of the integrated band absorbance of the carbonyl absorption band (1850–1600 cm^{-1}) relative to the methylene scissoring peak (1500–1420 cm^{-1}) in the respective ATR/FTIR spectra. Samples that were ductile under ATR pressure are represented by blue bars and those that were brittle are shown as red bars

were ductile under ATR pressure are represented by blue bars, while those that were brittle are shown as red bars.

The CI ranged from 0.74 to 0.30 for PE particles and 0.68 to 0.25 for PP, this is similar ranges that Celik et al. [38] reported for their samples collected from the west coast of the Sea of Japan (although they use a slightly smaller window of 1650–1850 cm^{-1}). No clear trend is observed between CI and whether particles are brittle or ductile. This absence of correlation may be partly due to the nature of the measurements, as fracturing under pressure captures both surface and interior material, which may have undergone different extents of weathering and oxidation. Another explanation is that while CI serves as an indicator of oxidation, it does not fully capture the overall degradation state of the polymer. For example, a high CI may reflect extensive oxidation but does not necessarily correspond to a low chain length, as chain length is influenced by both the polymer's initial molecular structure and the extent of degradation [24, 30–32, 34]. These same 17 samples were subjected to GPC analysis in order to investigate the chain length and molecular weight of the polymers.

GPC analysis

GPC is commonly used to assess changes in molecular chains in polymers subjected to aging and weathering; however, its application to field-recovered samples remains limited. The results of the GPC analyses for PP and PE particles that were brittle or ductile under the applied pressure are shown in Fig. 3.

From Fig. 3 we can see that, for both PE and PP, the brittle particles are at higher elution times than the ductile particles indicating shorter chains. From the GPC response we can determine the *Mw*, which represents the

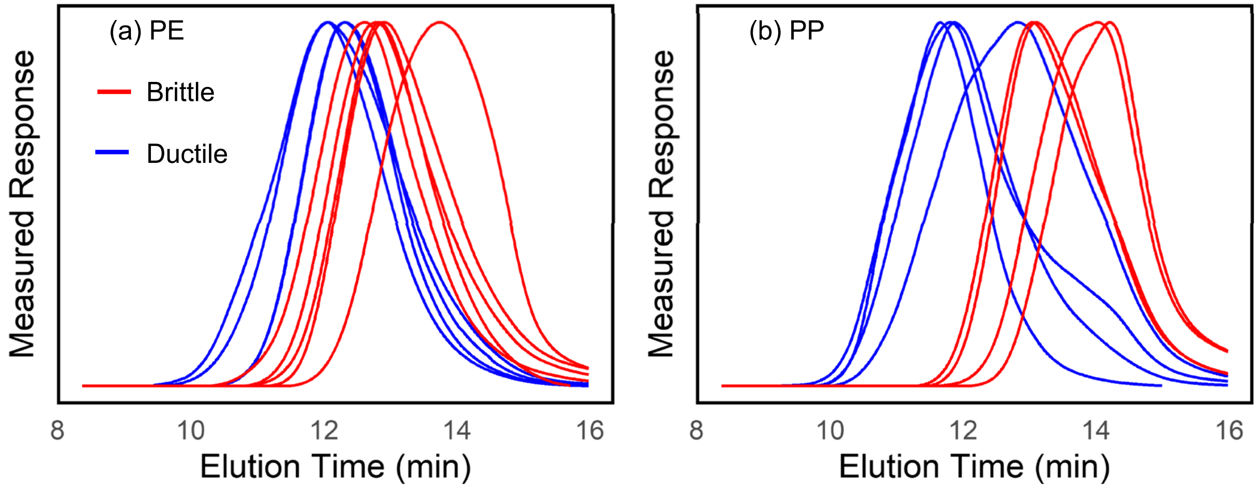

Fig. 3 GPC response of (**a**) PE particles and (**b**) PP particles that are brittle (red line) and ductile (blue line)

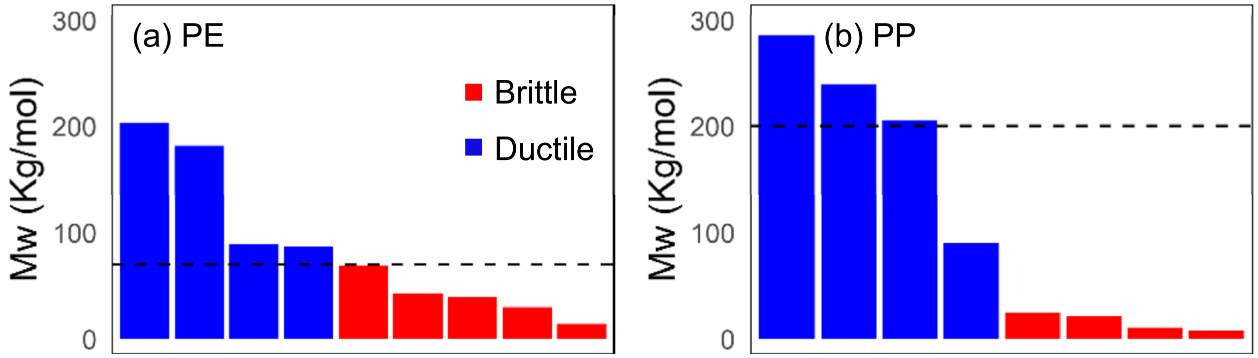

Fig. 4 *Mw* derived for (**a**) PE and (**b**) PP particles that are ductile (blue bars) and brittle (red bars). The *M'c* of PE (70 kg/mol) and PP (200 kg/mol) is shown as a dashed line [24]

average molecular mass of polymer chains weighted by their mass [25]. Crosslinking increases *Mw* while chain scission decreases *Mw* by breaking polymer chains into smaller chains [24]. Figure 4 shows the *Mw* determined from the GPC results for the PP and PE particles.

For both polymers a wide range of *Mw* values spanning from 280 kg/mol to 7 kg/mol, is observed from Fig. 4. These results highlight the presence of a variety of molecular weights among the sampled PP and PE particles from the Hawaiian beaches. As a comparison Ceccarini et al. [53], reported *Mw* values ranging from 73.4 to 173.4 kg/mol for PE, PP and PS samples collected from beach sediment collected in Tuscany, Italy. While polymer production naturally yields a broad range of *Mw* values, unweathered PE and PP used in commercial applications typically exhibit *Mw* values above 50 kg/mol [24, 25, 32–34]. In our dataset ($n = 17$), four of nine PE samples (44%) and four of eight PP samples (50%) showed *Mw* values below 50 kg/mol. Furthermore, five PE samples had *Mw* below the critical molar mass (*M'c*) of 70 kg/mol, and five PP samples fell below *M'c* for PP, defined as 200 kg/mol. Notably, we measured minimum *Mw* values of 14 kg/mol for PE and 7 kg/mol for PP. To our knowledge, such low *Mw* values have not been reported for plastics collected in the environment. While these 17 particles represent a targeted subset selected to investigate the *Mw* of ductile and brittle samples, 92% of all collected particles in our broader dataset were classified as brittle—and brittle particles tended to have lower *Mw*. Thus, we could expect that a significant proportion of plastics present on Hawaiian beaches also exhibit low molecular weights.

Furthermore, for both PE and PP samples the higher *Mw* appear to exhibit ductile behavior whereas samples with the lower *Mw* are brittle. Thus, the embrittlement of polymers appears to be linked to their *Mw* values, which is consistent with findings from previous laboratory studies [24, 25, 36]. In Fig. 4, the black dotted line represents the *M'c* value determined in laboratory-controlled

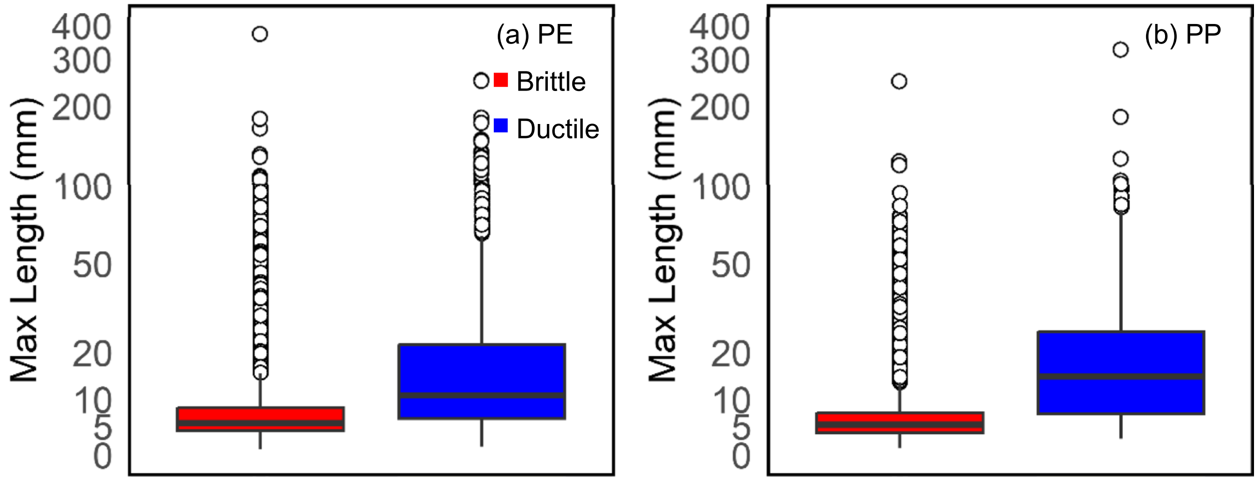

Fig. 5 Whiskers plot of plastic particle maximum lengths for (**a**) PE and (**b**) PP particles that were brittle (red) or ductile (blue). The plot shows the distribution of particle max lengths across each group. Note that the y-axis is in pseudo-log scale

accelerated aging experiments, using standardized mechanical tensile testing [24, 25, 32–34, 36]. For PE (Fig. 4a), all brittle samples fall below this value, while all ductile samples fall above it. However, for PP (Fig. 4b), the brittle particles exhibit significantly lower *Mw* values than the laboratory-derived *M'c*. This discrepancy may stem from differences in brittleness criteria (our assessment is based on an arbitrary pressure), variations in crystalline structure, or the specific PP composition (e.g., homopolymer, random copolymer, or block copolymer, with block copolymers being more ductile than homopolymers). Further investigation is needed to clarify these factors. Additionally, a broader *Mw* distribution in the ductile PP samples could also contribute to the observed differences, for instance, in Fig. 3b, the wider curve suggests a more extensive *Mw* distribution, which may influence the mechanical properties of the material. The advanced degradation, coupled with dynamic coastal conditions such as wave action, sediment abrasion, and wind, creates an environment that favors fragmentation of already brittle material and thus continuous formation of secondary microplastics.

Fragmenting state and size of particles

Given that fragmentation is a key pathway for secondary microplastic generation, with smaller particles resulting from the breakdown of larger debris [4, 5, 54, 55] we investigate the correlation between particle size and their brittleness. The particle maximum lengths, determined using the Segmentation Model (the sizes were measured before the ATR/FTIR analysis), of all 15,061 PP and PE particles that were brittle and ductile are summarized in Fig. 5.

Figure 5 shows the mean maximum length for brittle particles being smaller than the ductile particles. The mean maximum length of brittle PE particles is 8.5 ± 10.7 mm, while for brittle PP particles, it is 7.7 ± 9.1 mm. In contrast, the ductile PE particles had a mean maximum length of 20.0 ± 25.6 mm, and the ductile PP particles had a mean maximum length of 22.8 ± 28.2 mm. Figure 6 further illustrates the size distribution of PP and PE particles across brittle and ductile groups. To enable meaningful density plots and determine relative frequencies, particle sizes were grouped into size classes (bins) by rounding maximum lengths to the nearest 0.1 mm. The density plots reveal that brittle particles exhibit a more compact size distribution, with a pronounced peak at smaller particle sizes. In comparison, ductile particles display a broader distribution, with peaks at slightly larger sizes and lower relative frequencies. It is worth noting that the smaller size of brittle particles may result from more advanced degradation leading to fragmentation, or it could indicate that the brittle/ductile transition measured through our method could be influenced by particle size. In our ATR measures, the loading conditions were kept constant across samples to reduce the impact of particle size on the measurements.

The ANOVA results further reinforce these observations, revealing statistically significant differences between particle sizes and fragmentation state ($F = 1211.78$, $p < 0.001$), with brittle particles being significantly smaller than ductile ones. The correlation between size and brittleness observed in our study suggests that more degraded polymers, which are more brittle, tend to be smaller in size. These findings align with the expectation that more brittle and weathered plastics are more susceptible to fragmentation and thus form smaller

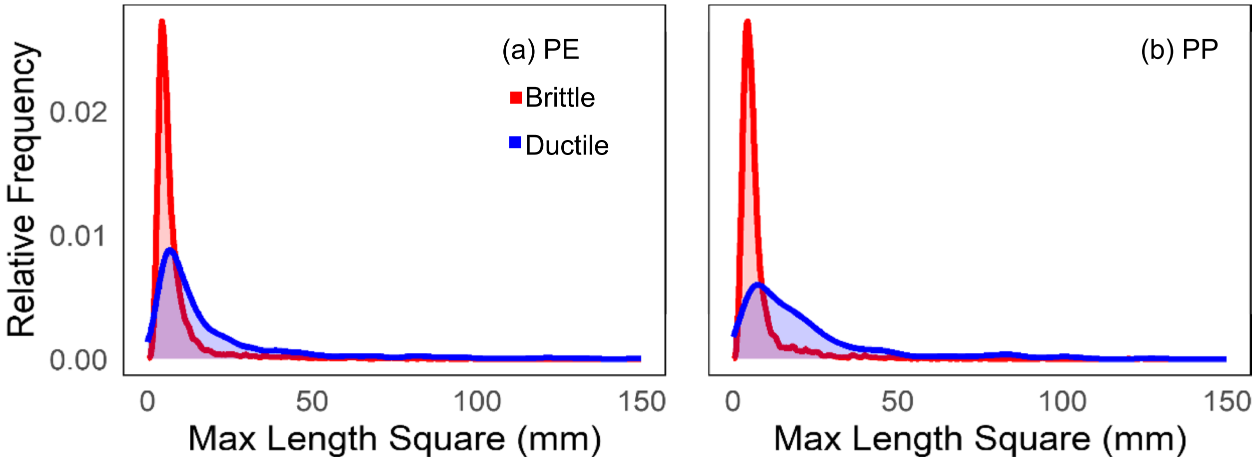

Fig. 6 Density plots of max length (rounded up to 0.1 mm) of (**a**) PP samples and (**b**) PE samples; the red curve represents brittle particles, and the blue curve represents ductile particles

particles [3, 10], highlighting the important role of weathering processes in shaping particle size distributions. Additionally, our analysis reveals a statistically significant main effect of polymer type on particle size ($F = 19.89$, $p < 0.001$) indicating that overall particle sizes differ between polymer types (PE or PP). Further research is currently being conducted on expanding the understanding of the size-fragmentation relationship, particularly by investigating how environmental factors such as UV radiation, mechanical stress, and exposure to different weathering conditions influence this relationship.

Reconciliation science and the path towards an understanding of plastic pollution risks to Hawaiian shores

Our research, which involved excavation of beaches risked causing unintentional harm in disturbing iwi (ancestral remains) [56]. This realization emerged only at the later stages of the research when Delorme connected with Kanaka ʻŌiwi (Native Hawaiian) leader and cultural practitioner, Kimeona Kāne. Although we argue that the immense burden of plastic pollution in Hawaiʻi is very disruptive —causing ecological harm and interrupting ways of life—we must also confront the possibility that our research methods might be equally, if not more, disruptive. Such unintended harms are not isolated to this study but reflect a broader historical pattern in Western research approaches across Hawaiʻi and Moana Nui (the Pacific). This pattern illustrates a systemic tendency in Western scientific inquiry to position cultural considerations as secondary to environmental data [57]. As Alegado [58] points out, in the context of the contested construction of a thirty-meter telescope on Mauna Kea, “in our excitement to herald new avenues of research, we must ensure that we are doing so within appropriate bounds.” This call underscores the importance of ethical science that is responsive to the intricacies of societal and cultural contexts. Alegado [58] further reminds us, “No field of science exists outside the sphere of culture.”

When scientific inquiry is detached from cultural considerations, it risks causing harm [59]—not only through its methods but also by sidelining the lived realities, worldviews, and true impacts on the places it studies. Our study initially focused on environmental metrics, isolating plastic particles from the sand column to assess them as indicators of plastic pollution and hence environmental degradation. However, Hawaiʻi’s beaches are more than environmental sites,they are deeply interconnected with cultural practices, livelihoods, and identity. The increasing burden of plastic pollution disrupts these relationships— from declining populations of Peʻeone (marbled mole crabs), once abundant in the sand column and used as bait in subsistence fishing, to the contamination of loko i ʻa (Hawaiian fishponds) and the erosion of food sovereignty. The loss of species, practices, food systems, are not just a loss of a single species, practice or food; it is a loss of relational knowledge and the cultural practices embedded in their use. Yet, as long as we pursue an illusion of “unbiased” or “objective” science [60], we risk overlooking these deeper connections, failing to fully understand or address the impacts of plastic pollution on both ecosystems and the communities that rely on them.

As Ngata & Liboiron [61] argue, the solution is not simply"more and better science"—such as researching histories, cultures, or conducting additional studies on the impact of plastic pollution on sand turtles. Rather, it might require scientists to apply the same critical curiosity they use within their disciplines to question the perceived neutrality and universality of their frameworks [62]. This includes recognizing Western science's historical role in the erasure and marginalization of Indigenous knowledge systems, its complicity in unethical, non-consensual research on Indigenous peoples, and its ongoing benefit from access to Indigenous lands, knowledge, and communities without proper consent [63]. Another pathway is to also acknowledging that knowledge is contextually rather than universally relevant fosters a deeper appreciation for diverse knowledge systems. Such recognition is particularly important in research on global challenges like pollution and climate change, where longstanding Western dominance has often imposed research agendas and priorities that fail to align with the lived experiences and cultural values of Indigenous Peoples—who disproportionately bear the burden of all three crises within the "triple planetary crisis" [62].

Our research has shown the reality that a significant portion of plastic particles on the shores of Hawai‘i are brittle, making them prone to further fragmentation. Measuring plastic brittleness helps us understand where and how secondary microplastics are generated, ultimately to guide mitigation efforts. However, to ensure that science and mitigation actions truly serve the communities most impacted by pollution equitably, we must also confront the systemic inequities embedded in scientific research itself. Science must serve—not sideline—communities. This shift calls for integrating and empowering Indigenous scientists and communities, who already hold critical knowledge and solutions, to lead the way and (re)design existing systems of scientific knowledge [61].

Conclusion

Our findings indicate that 92% of the particles analyzed exhibited brittleness under a small applied pressure, making them prone to further fragmentation into secondary microplastics. This brittleness was associated with low *Mw* values, with values as low as 14 kg/mol for PE and 7 kg/mol for PP. Such low *Mw* values, combined with the presence of oxidation products, suggest advanced polymer degradation. To our knowledge, these low *Mw* values have not been previously reported for plastics collected in the environment. Particles classified as brittle in our method were smaller than their ductile counterparts, highlighting the potential role of fragmentation in particle size distribution. Overall, this study confirms our understanding of secondary microplastic formation by linking polymer embrittlement to reduced chain length and degradation. By investigating the embrittlement behavior of naturally aged plastics in Hawai‘i's coastal environments, we highlight the substantial risks posed by degraded, brittle plastics in dynamic beach systems—particularly in regions like Hawai‘i, which receives a significant influx of highly weathered plastic. Through this study, we aim to emphasize the urgency of considering legacy plastics in ongoing global negotiations to prevent further exacerbation of plastic pollution's burden on Hawaiian communities. We also acknowledge that to truly understand the impacts of plastic pollution and to develop sustainable, equitable, and just solutions, scientific research must also reflect on its own practices. We recognize that reconciliation science is not a solution in itself, nor is it an endpoint. Rather, it is an ongoing process of reflection—one that requires naming our missteps and failures, interrupting ingrained habits, and actively (un)learning from them [64]. Incorporating these reflections into our research is an opportunity for growth and improvement, ensuring that future studies are more inclusive of cultural and ecological contexts as well as diverse ways of knowing.

Supplementary Information

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Supplementary Material 1.

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Authors' contributions

Astrid E. Delorme: Conceptualization, Formal analysis, Methodology, Investigation, Resources, Writing – original draft, Visualization, Project administration, Funding acquisition. Laurent Lebreton: Methodology, Supervision, Writing – review & editing. Sarah-Jeanne Royer: Methodology, Investigation, Writing – review & editing, Supervision, Project administration, Funding acquisition. Kimeona Kāne: Conceptualization, Writing – review & editing. Mael Arhant: Methodology, Investigation, Writing – review & editing. Maelenn Le Gall: Methodology, Investigation, Writing – review & editing Pierre-Yves Le Gac: Methodology, Investigation, Supervision, Project administration, Writing – review & editing.

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Data availability
No datasets were generated or analysed during the current study.

Declarations

Competing interests

S.J.R. and L.L. are employed by The Ocean Cleanup, a non-profit organization aimed at advancing scientific understanding and developing solutions to rid the oceans of plastic, headquartered in Rotterdam. A.E.D., K.K., M.L.G., P.Y.G. and M.A. declare no competing interests.

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References

1. Kvale K, Andrews ZA, Egger M. Mind the fragmentation gap. *Nature*. Communications. 2024;15:9831. https://doi.org/10.1038/s41467-024-53962-3.
2. Richon C, Kvale K, Lebreton L, Egger M. Legacy oceanic plastic pollution must be addressed to mitigate possible long-term ecological impacts. Microplastics Nanoplastics. 2023;3:25. https://doi.org/10.1186/s43591-023-00074-2.
3. Chubarenko I, Efimova I, Bagaeva M, Bagaev A, Isachenko I. On mechanical fragmentation of single-use plastics in the sea swash zone with different types of bottom sediments: Insights from laboratory experiments. Mar Pollut Bull. 2020;150:110726. https://doi.org/10.1016/j.marpolbul.2019.110726.
4. Liro M, Zielonka A, van Emmerik THM. Macroplastic fragmentation in rivers. Environ Int. 2023;180:108186. https://doi.org/10.1016/j.envint.2023.108186.
5. Thompson, R. C., Courtene-Jones, W., Boucher, J., Pahl, S., Raubenheimer, K., & Koelmans, A. A. Twenty years of microplastic pollution research—what have we learned? Science. 2024; 386. https://doi.org/10.1126/science.adl2746
6. Botterell ZLR, Beaumont N, Cole M, Hopkins FE, Steinke M, Thompson RC, Lindeque PK. Bioavailability of microplastics to marine zooplankton: Effect of shape and infochemicals. Environ Sci Technol. 2020;54(19):12024–33. https://doi.org/10.1021/acs.est.0c02715.
7. Kooi M, Primpke S, Mintenig SM, Lorenz C, Gerdts G, Koelmans AA. Characterizing the multidimensionality of microplastics across environmental compartments. Water Res. 2021;202:117429. https://doi.org/10.1016/j.watres.2021.117429.
8. Liro M, Zielonka A. Towards a geography of plastic fragmentation. Microplastics & Nanoplastics. 2025;5:12. https://doi.org/10.1186/s43591-025-00120-1.
9. Manzari Kushwaha S, Shankar S, Goel D, Singh S, Rahul J, Rachna K, Singh J. Microplastics pollution in the marine environment: A review of sources, impacts and mitigation. Mar Pollut Bull. 2024;209:117109. https://doi.org/10.1016/j.marpolbul.2024.117109.
10. Andrady AL, Barnes PW, Bornman JF, Gouin T, Madronich S, White CC, Zepp RG, Jansen MAK. Oxidation and fragmentation of plastics in a changing environment; from UV-radiation to biological degradation. Sci Total Environ. 2022;851:158022. https://doi.org/10.1016/j.scitotenv.2022.158022.
11. Kalogerakis, N., Karkanorachaki, K., Kalogerakis, G. C., Triantafyllidi, E. I., Gotsis, A. D., Partsinevelos, P., & Fava, F. (2017). A review of the mechanisms of plastics biodegradation in the marine environment. Front Marine Sci. 2017; 4. https://doi.org/10.3389/fmars.2017.00084
12. Morales-Caselles C, Viejo J, Martí E, González-Fernández D, Pragnell-Raasch H, González-Gordillo JI, Montero E, Arroyo GM, Hanke G, Salvo VS, Basurko OC, Mallos N, Lebreton L, Echevarría F, van Emmerik T, Duarte CM, Gálvez JA, van Sebille E, Galgani F, García CM, Ross PS, Bartual A, Ioakeimidis C, Markalain G, Isobe A, Cózar A. An inshore–offshore sorting system revealed from global classification of ocean litter. Nature Sustainability. 2021;4:484–93. https://doi.org/10.1038/s41893-021-00720-8.
13. Schwarz AE, Lensen SMC, Langeveld E, Parker LA, Urbanus JH. Plastics in the global environment assessed through material flow analysis, degradation and environmental transportation. Sci Total Environ. 2023;875:162644. https://doi.org/10.1016/j.scitotenv.2023.162644.
14. van Emmerik T, Schwarz A. Plastic debris in rivers. *WIREs*. Water. 2020;7:e1398. https://doi.org/10.1002/wat2.1398.
15. Berg CJ, Hafner J, Lamson MR, Maximenko NA, Welti CW. Interannual variability in marine debris accumulation on Hawaiʻian shores: The role of North Pacific Ocean basin-scale dynamics. Mar Pollut Bull. 2024;203:116484. https://doi.org/10.1016/j.marpolbul.2024.116484.
16. Blickley LC, Currie JJ, Kaufman GD. Trends and drivers of debris accumulation on Maui shorelines: Implications for local mitigation strategies. Mar Pollut Bull. 2016;105(1):292–8. https://doi.org/10.1016/j.marpolbul.2016.02.007.
17. Carson HS, Lamson MR, Nakashima D, Toloumu D, Hafner J, Maximenko N, McDermid KJ. Tracking the sources and sinks of local marine debris in Hawaiʻi. Mar Environ Res. 2013;84:76–83. https://doi.org/10.1016/j.marenvres.2012.12.002.
18. Kaandorp MLA, Lobelle D, Kehl C, Dijkstra HA, van Sebille E. Global mass of buoyant marine plastics dominated by large long-lived debris. Nat Geosci. 2023;16(8):689–94. https://doi.org/10.1038/s41561-023-01216-0.
19. Lebreton L, Slat B, Ferrari F, Sainte-Rose B, Aitken J, Marthouse R, Hajbane S, Cunsolo S, Schwarz A, Levivier A, Noble K, Debeljak P, Maral H, Schoeneich-Argent R, Brambini R, Reisser J. Evidence that the Great Pacific Garbage Patch is rapidly accumulating plastic. Sci Rep. 2018;8(1):4666. https://doi.org/10.1038/s41598-018-22939-w.
20. Beamer K, Elkington K, Souza P, Tuma A, Thorenz A, Köhler S, Kukea-Shultz K, Kotubetey K, Winter KB. Island and Indigenous systems of circularity: How Hawaiʻi can inform the development of universal circular economy policy goals. Ecol Soc. 2023;28(1):9. https://doi.org/10.5751/ES-13656-280109.
21. Delevaux JMS, Whittier R, Stamoulis KA, Bremer LL, Jupiter S, Friedlander AM, Poti M, Guannel G, Kurashima N, Winter KB, Toonen R, Conklin E, Wiggins C, Knudby A, Goodell W, Burnett K, Yee S, Htun H, Oleson KLL, Wiegner T, Ticktin T. A linked land-sea modeling framework to inform ridge-to-reef management in high oceanic islands. PLoS ONE. 2018;13(3):e0193230. https://doi.org/10.1371/journal.pone.0193230.
22. Innes-Gold AA, Madin EMP, Stokes K, Ching C, Kawelo H, Kotubetey K, Reppun F, Rii YM, Winter KB, McManus LC. Restoration of an Indigenous aquaculture system can increase reef fish density and fisheries harvest in Hawaiʻi. Ecosphere. 2024;15(3):e4797. https://doi.org/10.1002/ecs2.4797.
23. ISO 527–1:2019(en): Plastics — Determination of tensile properties — Part 1: General principles. International Organization for Standardization, 2019.
24. Fayolle B, Richaud E, Colin X, Verdu J, et al. Review: degradation-induced embrittlement in semi-crystalline polymers having their amorphous phase in rubbery state. J Mater Sci. 2008;43:6999–7012. https://doi.org/10.1007/s10853-008-3005-3.
25. Laot R, Le Gac P-Y, Le Gall M, Broudin M, Ovalle C, Laiarinandrasana L. Embrittlement induced by oxidation in polyethylene: Role of initial bimodality. Polym Degrad Stab. 2024;230:111025. https://doi.org/10.1016/j.polymdegradstab.2024.111025.
26. Andrady, A. L. Degradation of plastics in the environment. In A. L. Andrady (Ed.), Plastics and Environmental Sustainability. 2015:151–168. https://doi.org/10.1002/9781119009405.ch6
27. Arhant M, Le Gall M, Le Gac P-Y, Davies P. Impact of hydrolytic degradation on mechanical properties of PET - Towards an understanding of microplastics formation. Polym Degrad Stab. 2019;161:175–82. https://doi.org/10.1016/j.polymdegradstab.2019.01.021.
28. Chamas A, Moon H, Zheng J, Qiu Y, Tabassum T, Jang JH, Abu-Omar M, Scott SL, Suh S. Degradation rates of plastics in the environment. ACS Sustainable Chemistry & Engineering. 2020;8(9):3494–511. https://doi.org/10.1021/acssuschemeng.9b06635.
29. Fayolle B, Audouin L, Verdu J. Oxidation induced embrittlement in polypropylene—a tensile testing study. Polym Degrad Stab. 2000;70(3):333–40. https://doi.org/10.1016/S0141-3910(00)00108-7.
30. Fayolle B, Audouin L, Verdu J. Initial steps and embrittlement in the thermal oxidation of stabilised polypropylene films. Polym Degrad Stab. 2002;75(1):123–9. https://doi.org/10.1016/S0141-3910(01)00211-7.
31. Ojeda T, Freitas A, Birck K, Dalmolin E, Jacques R, Bento F, Camargo F. Degradability of linear polyolefins under natural weathering. Polym Degrad Stab. 2011;96(4):703–7. https://doi.org/10.1016/j.polymdegradstab.2010.12.004.

32. Deshoulles Q, Le Gall M, Dreanno C, Arhant M, Stoclet G, Priour D, Le Gac PY. Origin of embrittlement in Polyamide 6 induced by chemical degradations: Mechanisms and governing factors. Polym Degrad Stab. 2021;191:109657. https://doi.org/10.1016/j.polymdegradstab.2021.109657.
33. Deshoulles Q, Le Gall M, Benali S, Raquez J-M, Dreanno C, Arhant M, Priour D, Cerantola S, Stoclet G, Le Gac PY. Hydrolytic degradation of biodegradable poly(butylene adipate-co-terephthalate) (PBAT) - Towards an understanding of microplastics fragmentation. Polym Degrad Stab. 2022;205:110122. https://doi.org/10.1016/j.polymdegradstab.2022.110122.
34. Reano AF, Guinault A, Richaud E, Fayolle B. Polyethylene loss of ductility during oxidation: Effect of initial molar mass distribution. Polym Degrad Stab. 2018;149:78–84. https://doi.org/10.1016/j.polymdegradstab.2018.01.021.
35. Okamba-Diogo O, Richaud E, Verdu J, Fernagut F, Guilment J, Fayolle B. Investigation of polyamide 11 embrittlement during oxidative degradation. Polymer. 2016;82:49–56. https://doi.org/10.1016/j.polymer.2015.11.025.
36. Arhant M, Le Gall M, Le Gac P-Y. Fracture test to accelerate the prediction of polymer embrittlement during aging – Case of PET hydrolysis. Polym Degrad Stab. 2022;196:109848. https://doi.org/10.1016/j.polymdegradstab.2022.109848.
37. Almond J, Sugumaar P, Wenzel MN, Hill G, Wallis C. Determination of the carbonyl index of polyethylene and polypropylene using specified area under band methodology with ATR-FTIR spectroscopy. e-Polymers. 2020;20(1):369–81. https://doi.org/10.1515/epoly-2020-0041.
38. Celik M, Nakano H, Uchida K, Isobe A, Arakawa H. Comparative evaluation of the carbonyl index of microplastics around the Japan coast. Mar Pollut Bull. 2023;190:114818. https://doi.org/10.1016/j.marpolbul.2023.114818.
39. Liboiron M, Zahara A, Hawkins K, Crespo C, de Moura Neves B, Wareham-Hayes V, Edinger E, Muise C, Walzak MJ, Sarazen R, Chidley J, Mills C, Watwood L, Arif H, Earles E, Pijogge L, Shirley J, Jacobs J, McCarney P, Charron L. Abundance and types of plastic pollution in surface waters in the Eastern Arctic (Inuit Nunangat) and the case for reconciliation science. Sci Total Environ. 2021;782: 146809. https://doi.org/10.1016/j.scitotenv.2021.146809.
40. Fletcher, C. H., Romine, B. M., Genz, A. S., Barbee, M. M., Dyer, M., Anderson, T. R., Lim, S. C., Vitousek, S., Bochicchio, C., & Richmond, B. M. National assessment of shoreline change: Historical shoreline change in the Hawai'ian Islands: U.S. Geological Survey Open-File Report. 2012.
41. Brignac KC, Jung MR, King C, Royer SJ, Blickley L, Lamson MR, Potemra JT, Lynch JM. Marine Debris Polymers on Main Hawaiʻian Island Beaches, Sea Surface, and Seafloor. Environ Sci Technol. 2019;53(21):12218–26. https://doi.org/10.1021/acs.est.9b03561.
42. Cooper DA, Corcoran PL. Effects of mechanical and chemical processes on the degradation of plastic beach debris on the island of Kauai. Hawaiʻi Marine Pollution Bulletin. 2010;60(5):650–4. https://doi.org/10.1016/j.marpolbul.2009.12.026.
43. Kubota M. A Mechanism for the Accumulation of Floating Marine Debris North of Hawaiʻi. J Phys Oceanogr. 1994;24(5):1059–64. https://doi.org/10.1175/1520-0485(1994)024%3c1059:AMFTAO%3e2.0.CO;2.
44. Young AM, Elliott JA. Characterization of microplastic and mesoplastic debris in sediments from Kamilo Beach and Kahuku Beach, Hawaiʻi. Mar Pollut Bull. 2016;113:477–82. https://doi.org/10.1016/j.marpolbul.2016.11.009.
45. Delorme AE, Poirion OB, Lebreton L, le Gac P-Y, Kāne K, Royer SJ. A year-long field study of buried plastics reveals underestimation of plastic pollution on Hawaiian beaches. Mar Pollut Bull. 2025;214:117712. https://doi.org/10.1016/j.marpolbul.2025.117712.
46. Royer S-J, Wolter H, Delorme AE, Lebreton L, Poirion OB. Computer vision segmentation model—deep learning for categorizing microplastic debris. Front Environ Sci. 2024;12:1386292. https://doi.org/10.3389/fenvs.2024.1386292.
47. Jung MR, Horgen FD, Orski SV, Rodriguez C, V., Beers, K. L., Balazs, G. H., Jones, T. T., Work, T. M., Brignac, K. C., Royer, S.-J., Hyrenbach, K. D., Jensen, B. A., & Lynch, J. M. Validation of ATR FT-IR to identify polymers of plastic marine debris, including those ingested by marine organisms. Mar Pollut Bull. 2018;127:704–16. https://doi.org/10.1016/j.marpolbul.2017.12.061.
48. Thermo Fisher Scientific. iD5 ATR Accessory for the Nicolet™ iS™ 5 Spectrometer (Catalog number IQLAADGAAGFAJAMAYY). Retrieved December 11, 2024, https://www.thermofisher.com/order/catalog/product/IQLAADGAAGFAJAMAYY
49. Hale N, Valero M, Tang J, Moser D, Jiang L. A preliminary study on characterisation of finger interface kinetics using a pressure and shear sensor system. Prosthet Orthot Int. 2018;42(1):60–5. https://doi.org/10.1177/0309364617728121.
50. Achimsky L, Audouin L, Verdu J. Kinetic study of the thermal oxidation of polypropylene. Polym Degrad Stab. 1997;57(3):231–40. https://doi.org/10.1016/S0141-3910(96)00167-X.
51. Grause G, Chien M-F, Inoue C. Changes during the weathering of polyolefins. Polym Degrad Stab. 2020;181:109364. https://doi.org/10.1016/j.polymdegradstab.2020.109364.
52. Mellor DC, Moir, AB, Scott G. The effect of processing conditions on the UV stability of polyolefins. Eur Polymer J. 1973; 9(3), 219–225.
53. Ceccarini A, Corti A, Erba F, Modugno F, La Nasa J, Bianchi S, Castelvetro V. The hidden microplastics: New insights and figures from the thorough separation and characterization of microplastics and of their degradation byproducts in coastal sediments. Environ Sci Technol. 2018;52(10):5634–43. https://doi.org/10.1021/acs.est.8b01487.
54. Kaandorp MLA, Dijkstra HA, van Sebille E. Modelling size distributions of marine plastics under the influence of continuous cascading fragmentation. Environ Res Lett. 2021;16(5):054075. https://doi.org/10.1088/1748-9326/abe9ea.
55. Liro M, Zielonka A, Mikuś P. First attempt to measure macroplastic fragmentation in rivers. Environ Int. 2024;191:108935. https://doi.org/10.1016/j.envint.2024.108935.
56. Hall, D. N. (2010). Sovereign ground. In C. Howes & J. K. K. Osorio (Eds.), *The value of Hawaiʻi: Knowing the past, shaping the future* (pp. 195–200). University of Hawaiʻi Press. http://www.jstor.org/stable/j.ctt6wqxjc.29
57. Borrelle SB, Koch JB, McDonough MC, Ingeman KE, McGill BM, Lambert MR, Belasen AM, Dudney J, Chang CH, Teffer AK, Wu GC. Corrigendum 2 (published 29 Oct 2021) to: What does it mean to be for a Place? Pac Conserv Biol. 2021;27:507–8. https://doi.org/10.1071/PC20015_C1.
58. Alegado R. Opponents of the Thirty Meter Telescope fight the process, not science. Nature. 2019;572:7. https://doi.org/10.1038/d41586-019-02304-1.
59. Bang M, Marin A, Medin D. If Indigenous peoples stand with the sciences, will scientists stand with us? Daedalus. 2018;147(2):148–59. https://doi.org/10.1162/DAED_a_00498.
60. van Eck, C.W., Messling, L. & Hayhoe, K. Challenging the neutrality myth in climate science and activism. npj Clim. Action 3, 81 (2024). https://doi.org/10.1038/s44168-024-00171-9
61. Ngata T, Liboiron M. A Māori approach to starting research from where you are. Catalyst Feminism Theory Technoscience. 2021;7, *2*, 1–7. https://doi.org/10.28968/cftt.v7i2.37648
62. Stein S, Ahenakew C, Valley W, Sherpa PY, Crowson E, Robin T, Mendes W, Evans S. Toward more ethical engagements between Western and Indigenous sciences. FACETS. 2024;9:1–14. https://doi.org/10.1139/facets-2023-0071.
63. Liboiron M. Pollution is colonialism. Duke University Press. 2021. https://doi.org/10.2307/j.ctv1jhvnk1.
64. Arshad-Ayaz, A., Naseem, M. A., & Mohamad, D. Engineering and humanitarian intervention: learning from failure. International Journal of Humanitarian Action. 2020;5(7). https://doi.org/10.1186/s41018-020-00073-5

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