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REVIEW & ANALYSIS

Microplastics in composts, digestates, and food wastes: A review

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Abstract

Diverting food waste from landfills to composting or anaerobic digestion can reduce greenhouse gas emissions, enable the recovery of energy in usable forms, and create nutrient-rich soil amendments. However, many food waste streams are mixed with plastic packaging, raising concerns that food waste-derived composts and digestates may inadvertently introduce microplastics into agricultural soils. Research on the occurrence of microplastics in food waste-derived soil amendments is in an early phase and the relative importance of this potential pathway of microplastics to agricultural soils needs further clarification. In this paper, we review what is known and what is not known about the abundance of microplastics in composts, digestates, and food wastes and their effects on agricultural soils. Additionally, we highlight future research needs and suggest ways to harmonize microplastic abundance and ecotoxicity studies with the design of related policies. This review is novel in that it focuses on quantitative measures of microplastics in composts, digestates, and food wastes and discusses limitations of existing methods and implications for policy.

1 | INTRODUCTION

Food waste constitutes approximately a quarter of all material landfilled in the United States (USEPA, 2020) and is readily converted to methane—a potent greenhouse gas—under the anaerobic conditions found in landfills (Buzby et al., 2014). Diverting food waste from landfills to anaerobic digestion and composting could reduce methane emissions and enable the recovery of nutrients and energy in usable forms (USEPA, 2021a,2021b). Both processes produce soil amendments—digestate and compost, respectively—that can be applied to agricultural lands to support soil health and fertility (Cheong et al., 2020; Kelley et al., 2020; Roy, 2017). Anaerobic digestion provides the additional benefit of recovering usable energy from food waste in the form of biogas (F. Xu et al., 2018). Growing recognition of these cobenefits has prompted recent legislation regarding the diversion of food waste from landfills (Golwala et al., 2021). In the United States, food waste disposal bans have been enacted in four states (New York, Massachusetts, Rhode Island and Vermont) and diversion requirements established in three others (California, Connecticut and New Jersey) (Ryen & Babbitt, 2022). The state of Vermont's *Universal Recycling Law* (2012) is the most stringent in the United States, mandating the diversion of all food residuals—including household food waste—from landfills in 2020, whereas other policies only apply to large volume commercial and industrial food waste generators (Ryen & Babbitt, 2022). These types of policies could help to close resource loops and facilitate a more circular economy of resource use; however, contamination from plastic packaging has emerged as a significant challenge in their implementation (O'Connor et al., 2022; USEPA, 2021a).

Conventional petroleum-based plastics are used in a wide variety of food packaging applications (Table 1) due to their low cost and chemical barrier properties (Ncube et al., 2020). As a result of their ubiquitous use in packaging, plastics are often mixed in with many pre- and post-consumer food waste streams (USEPA, 2021a). Substantial fractions of wasted food from industrial and commercial settings can remain packaged for a variety of reasons (e.g., expiration, off-specification, contamination). In the U.S. state of Vermont, for example, an estimated 38% of food waste is still in packaging (DSM Environmental Services Inc, 2018). Recovering food waste in these cases requires some form of depackaging, using either mechanical depackagers or human labor, both of which are likely to achieve variable and imperfect separation efficiency (do Carmo Precci Lopes et al., 2019; Edwards et al., 2018). Source-separated post-consumer food waste may also contain mis-sorted plastic packaging, with varying levels of contamination that may be influenced by factors such as population density (Friege & Eger, 2021) and degree of public engagement (Dai et al., 2016).

Despite efforts to separate packaging from food waste streams, early evidence suggests that macro- (>5 mm) and micro- (<5 mm) plastics may be present in many food waste-derived composts and digestates (Figure 1) and could be transferred to agricultural soils when these amendments are land applied (Figure 2; Kawecki et al., 2020; Weithmann et al., 2018). Due to their highly stable chemical structure, most conventional petroleum-based plastics are resistant to total degradation and may persist in the environment for centuries (Ali et al., 2021). Through time, plastics may accumulate in soils (Y. Yu & Flury, 2021), with macroplastics fragmenting into microplastics or even nanoplastics due to physicochemical and biological degradation (Ali et al., 2021). This partial degradation can release additives and impurities that may be harmful to human and ecosystem health (Rillig et al., 2021). In addition to the potential risks posed to human and ecosystem health, there is early evidence to suggest that some microplastics have an inhibitory effect on the composting and anaerobic digestion processes (J. Zhang et al., 2020a; Y. Zhou et al., 2022), thereby possibly reducing the intended benefits of food waste diversion initiatives. Furthermore, plastic contamination can impede circular economy efforts by making composts and digestates less attractive to farmers and consumers (Friege & Eger, 2021; Roy et al., 2021). "Biodegradable" and "compostable" plastics have been touted as a more environmentally friendly alternative to conventional petroleum-based plastics (European Commission, 2018; Folino et al., 2020; Shaikh et al., 2021), but are problematic for multiple reasons and do not yet represent a clear solution (Calabrò & Grosso, 2018; Haider et al., 2019; Markowicz & Szymańska-Pulikowska, 2019; Serrano-Ruiz et al., 2021).

Core Idea

- Microplastic presence is documented in composts, digestates, and food wastes.
- Lack of standardized methods for these materials complicates comparison between studies.
- Existing regulations establish w/w limits on contamination which is inconsistent with many studies.
- Focus on maximizing benefits of food waste diversion and minimizing risk of microplastic pollution is needed.

Microplastic contamination in organic materials is receiving increasing attention as food waste diversion policies continue to proliferate, leading to a growing number of entities imposing regulatory thresholds for microplastics in composts and digestates (USEPA, 2021a). However, relatively little is known about the abundance of microplastics in composts, digestates, and food wastes and their downstream effects in the environment. Since microplastics were first reported to be accumulating in the oceans in 2004 (Thompson et al., 2004), studies in the field have largely focused on marine and other aquatic environments (e.g., Besseling et al., 2019; Bond et al., 2018; Markic et al., 2020). It was not until 2012 that the presence of microplastics in terrestrial environments began to receive attention (Rillig, 2012), and studies focusing on terrestrial environments still represent a small fraction of all microplastic publications (i.e., 5% as of 2019) (R. Qi et al., 2020). Recent reviews have focused on the abundance and sources of microplastics in soils, the challenges of detecting and characterizing microplastics in soils and complex organic matrices, and the documented effects of microplastics on soil–plant systems (e.g., Iqbal et al., 2020; J. Li et al., 2020; Ng et al., 2018; J. Sun et al., 2019; R. Qi et al., 2020; Ruggero et al., 2020; Q. Sun et al., 2022; J. Wang et al., 2019; W. Wang et al., 2020; B. Xu et al., 2020; Y. Zhou et al., 2020; F. Zhu et al., 2019). Few peer-reviewed studies have focused further upstream on a likely source of microplastics in soils: composts and digestates derived from food waste (Golwala et al., 2021). One recent review article includes microplastics among other emerging contaminants in food waste-derived composts and digestates (O'Connor et al., 2022). However, to the best of our knowledge, no peer-reviewed literature reviews to date have focused on quantitative measures of microplastics in composts, digestates, and food wastes along with the implications for policy. Here, we fill that gap in the

TABLE 1 Recycling identification code, molecular structure (IUPAC ID), and food packaging applications of commonly used plastic polymers

Recycling identification code[a]	Name (abbreviation)	IUPAC ID	Common food packaging applications[b]
1	Polyethylene terepthalate (PET)	Poly(ethyl benzene-1,4-dicarboxylate)	Bottles, jars, tubes, trays, blisters, bags, and snack food wrappers
2	High density polyethylene (HDPE)	Polyethene	Containers, caps, covers, container labels
3	Polyvinyl chloride (PVC)	Poly(1-chloroethylene)	Rigid and flexible films, closures
4	Low density polyethylene (LDPE)	Polyethene	Films for frozen foods, bakery products, fresh meat and poultry, caps, covers, container labels
5	Polypropylene (PP)	Poly(propene)	Rigid food packaging
6	Polystyrene (PS)	Poly(1-phenylethene-1,2-diyl)	Disposable cups, trays for meat and produce, clam shell packaging for eggs, tubs for preserves, yogurt containers, breathable films
7	Other resins[c]	Variable	Shopping bags, cups, films, containers, bottles, wrapping, stirrers, cutlery, straws, foams

[a] Marsh and Bugusu (2007).
[b] Dybka-Stępień et al. (2021).
[c] This includes biodegradable plastics such as polylactic acid (PLA).

FIGURE 1 Visible plastic contamination in (a) organic municipal solid waste compost windrows prior to screening (credit: E.D. Roy, S. Asia), (b) screw-press separated solid digestate from co-digestion of dairy manure and food waste (credit: E.D. Roy, United States), and (c–f) Putative microplastics found in food waste digestate (credit: K.K. Porterfield, United States).

literature and provide a starting point for scientists, practitioners, and policy makers in the solid waste management field who are engaging with the issue of microplastics contamination in food waste streams.

First, we present a summary of existing peer-reviewed reports of microplastic abundance in composts, digestates, and food wastes. Next, we provide an overview of the different methods that have been used to measure microplastics in these materials and discuss limitations associated with the lack of standardized methods. We then briefly discuss the various inputs of microplastics to agricultural soils and summarize the documented impacts of microplastics on soil–plant systems. Finally, we provide a roadmap to harmonize efforts to quantify microplas-

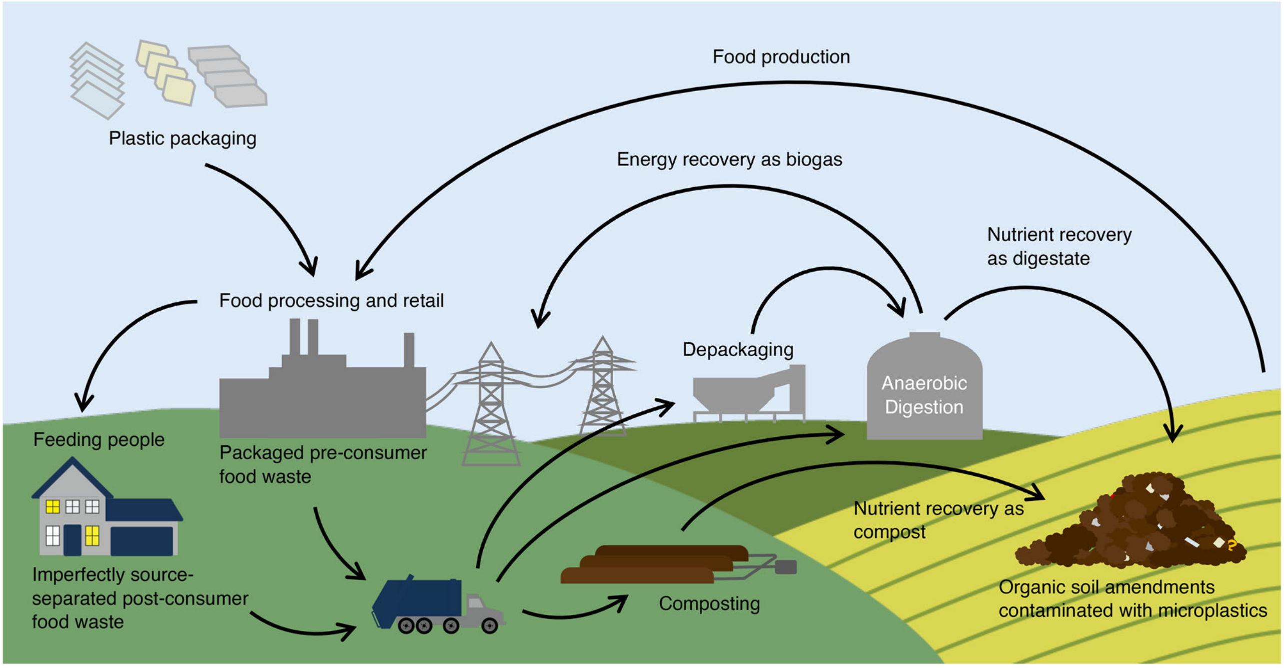

FIGURE 2 Conceptual diagram showing flows of food waste and microplastics to composting and anaerobic digestion and on to agricultural soils

tics in food waste-derived materials, understand the effects of microplastics in agricultural soils, and establish related policy.

2 | MICROPLASTIC ABUNDANCE IN FOOD WASTES, COMPOSTS, AND DIGESTATES

We used a systematic literature search to identify scientific articles providing primary data on microplastic abundance in food waste-derived composts and/or digestates (Table S1). For a full description of the systematic review methods, see the Supplementary Materials. We intentionally excluded studies focusing on biosolids-derived organic amendments unless there was codigestion with food waste because microplastic occurrence in wastewater has been reviewed elsewhere (J. Sun et al., 2019). We include studies of green waste-derived composts (e.g., yard and landscape trimmings) for comparison with food waste-derived composts. The studies that report microplastic abundance in terms of particles per weight (standardized to particles kg^{-1} dry material where possible) are summarized in Table 2 and the studies that report microplastic abundance in terms of w/w (standardized to w/w dry material where possible) are summarized in Table 3. For composts, digestates, and food wastes, we report plastic abundance values that include all size fractions measured for a given study. In some instances, this includes or is solely comprised of macroplastics.

All the studies we reviewed reported finding plastics in composts, digestates, and/or food wastes, even in cases where the compost was derived exclusively from green waste. The most frequently identified polymers included polyethylene (PE), polypropylene (PP), and polystyrene (PS) (Tables 2 and 3), which are also some of the most common plastics used in food packaging (Ncube et al., 2020). “Biodegradable” and “compostable” bioplastics, including polylactic acid (PLA), Mater-Bi®, and cellulose-based polymers, were identified as well (Tables 2 and 3).

Plastic abundance in food waste alone spanned five orders of magnitude on a count per weight basis (Table 2) and three orders of magnitude on a w/w basis (Table 3). Values for homogenized food waste ranged from 36 (Schwinghammer et al., 2020) to 1400 ± 150 particles kg^{-1} dry material (Ruggero et al., 2021); however, the former study only considered larger particles (1–5 mm) and the latter only considered smaller particles (0.1–2 mm). A study of grocery waste in the United States found 300,000 particles kg^{-1} dry material, but no information about the size fraction was available (Golwala et al., 2021). On a mass basis, plastic abundance ranged from ~0.025% w/w in homogenized food waste (Schwinghammer et al., 2020) to 5.6% w/w in source-separated household biowaste (do Carmo Precci Lopes et al., 2019).

Reported values also varied widely both within and between studies measuring plastic abundance in composts—spanning seven orders of magnitude on a count per mass basis (Table 2), and five orders of magnitude on a w/w basis (Table 3). Plastic abundance ranged from 12 ± 8 (Braun et al.,

TABLE 2 Plastic abundance in composts, digestates, and food wastes on a count basis

Feedstock	Abundance (particles kg^{-1} dry)	Size (mm)	Polymer types	Location	References
Compost					
Green waste	12 ± 8 to 46 ± 8	>0.3[b]	n/a	Germany	Braun et al. (2021)
Green waste	82,800 ± 17,400[a]	1–5	PLA	Netherlands	Huerta-Lwanga et al., 2021
Green waste	1253 ± 561	0.03–2	PE, PP	Netherlands	van Schothorst et al., 2021
Green waste	5733 ± 850 to 6433 ± 751	0.05–5	Mostly PP, also PE, nitrile rubber, PES	Lithuania	Sholokhova et al., 2021
Green and household waste	20–24	>1	Mostly styrene-based polymers (PS etc.) & PE, also PES, PP, PET, PVC	Germany	Weithmann et al., 2018
Food waste	3783 ± 351 to 4066 ± 658	0.05–5	Mostly PE & PS, also PET, PP	Lithuania	Sholokhova et al., 2021
Household biowaste	32 ± 20[b]	>0.3[b]	n/a	Germany	Braun et al., 2021
Rural domestic waste	2400 ± 358	0.05–5	Mostly PES, PP & PE, also PVC, PS, PE:PP, PU	China	Gui et al., 2021
OFMW digestate	39–102	1–5	Mostly PE & PVC, also PET, PS, PES, PUR, Other	Germany	Schwinghammer et al., 2020
OFMW	2800 ± 616	0.03–2	PE, PP	Netherlands	van Schothorst et al., 2021
OFMW	10,000–30,000	>0.025	Mostly PE, also PS, PP, PES, PVC, ACR	Spain	Edo et al., 2022
Unknown	5.2–42.8 (15.4) Mil[a]	0.005–1	n/a	Austria	Meixner et al., 2020
Digestate					
OFMW	75–240[b]	1–5	Mostly PES and PVC, also PP, PE, PET, PS, PA, EVA	Germany	Schwinghammer et al., 2020
Commercial biowaste	895	>1	n/a	Germany	Weithmann et al., 2018
Household biowaste	70–146	>1	Mostly styrene-based polymers (PS etc.), also PES, PE, PP, PET, PVC, PVDC, PA, PUR, latex, and cellulose-based polymers	Germany	Weithmann et al., 2018
Food waste and dairy manure	1670	>1	n/a	USA	O'Brien, 2019
Unknown	0.6–38.7 (7.1) Mil[a]	0.005–1	n/a	Austria	Meixner et al., 2020
Food waste					
Grocery store	300,000[a]	n/a	n/a	USA	Golwala et al., 2021
Pulped food waste	1400 ± 150[a]	0.1–2	Mostly Mater-Bi®, also PP, PE, PS, CE	Italy	Ruggero et al., 2021
Homogenized food waste	36	1–5	Mostly PE, also PP, PS	Germany	Schwinghammer et al., 2020

Abbreviations: OFMW, organic fraction municipal waste; ACR, acrylic polymers; CE, cellophane; PA, polyamide; EVA, ethylene vinyl acetate; PE, polyethylene; PES, polyester; PET, polyethylene terephthalate; PLA, polylactic acid; PP, polypropylene; PS, polystyrene; PU/PUR, polyurethane; PVC, polyvinyl chloride; PVDC, polyvinylidene chloride.

[a] Dry/as-is not reconciled.

[b] Estimated from figure.

TABLE 3 Plastic abundance in composts, digestates, and food wastes on a w/w basis

Feedstock	Abundance (% w/w dry)	Size (mm)	Polymer types	Location	References
Compost					
Green waste	0.00024–0.0065	>0.5	n/a	Germany	Bläsing & Amelung, 2018
Green waste	0.0048 ± 0.0089 to 0.065 ± 0.06[b]	>0.3[b]	n/a	Germany	Braun et al., 2021
Green waste	0.82 ± 0.11 to 1 ± 0.51[a]	>1	PLA	Netherlands	Huerta-Lwanga et al., 2021
Green waste	0.0237	1–5	Mostly PP, also PE, nitrile rubber, PES	Lithuania	Sholokhova et al., 2021
Food waste	0.0845	1–5	Mostly PE & PS, also PET, PP	Lithuania	Sholokhova et al., 2021
Biowaste	0.018	>0.5	n/a	Germany	Bläsing & Amelung, 2018
Household biowaste	0.1358 ± 0.0596	>0.3[b]	n/a	Germany	Braun et al., 2021
Organic waste	0.001–0.0102[a]	All	PET	Germany	Müller et al., 2020
OFMW digestate	0.005–0.05[b]	1–5	Mostly PE and PVC, also PET, PS, PES, PUR, Other	Germany	Schwinghammer et al., 2020
Digestate					
Kitchen and green waste	0.12 ± 0.12[c]	>6	n/a	Switzerland	Kawecki et al., 2020
Organic waste	0.0209–0.0776[a]	All	PET	Germany	Müller et al., 2020
Food waste + dairy manure	0.25	>1	n/a	USA	O'Brien, 2019
OFMW	0.01–0.0350[b]	1–5	Mostly PES & PVC, also PP, PE, PET, PS, PA, EVA	Germany	Schwinghammer et al., 2020
Food waste					
Kitchen and green waste	0.5 ± 0.46[c]	>6	n/a	Switzerland	Kawecki et al., 2020
Homogenized food waste	0.025[b]	1–5	Mostly PE, also PP and PS	Germany	Schwinghammer et al., 2020
Household biowaste	3.0–5.6[d]	>2	n/a	Austria	do Carmo Precci Lopes et al., 2019
Household biowaste (mechanically sorted)	0.04–2.9	>2	n/a	Austria	do Carmo Precci Lopes et al., 2019

Abbreviations: OFMW, organic fraction municipal waste; PA, polyamide; EVA, ethylene vinyl acetate; PE, polyethylene; PES, polyester; PET, polyethylene terephthalate; PLA, polylactic acid; PP, polypropylene; PS, polystyrene; PUR, polyurethane; PVC, polyvinyl chloride.

[a] Dry/as-is not reconciled.

[b] Estimated from figure.

[c] as-is.

[d] Calculated by mass balance.

2021) to 82,800 ± 17,400 (Huerta-Lwanga et al., 2021) particles dry kg^{-1} green waste-derived composts and from 20 (Weithmann et al., 2018) to 30,000 (Edo et al., 2022) particles dry kg^{-1} of composts made with food waste, with one study reporting 4.28×10^7 particles dry kg^{-1} of a compost of unknown origin (Meixner et al., 2020). On a mass basis, plastic abundance ranged from 0.00024% w/w (Bläsing & Amelung, 2018) to 1 ± 0.5% w/w (Huerta-Lwanga et al., 2021) in green waste-derived composts and from 0.001% w/w (Müller et al., 2020) to 0.1358 ± 0.0596% w/w (Braun et al., 2021) in composts made with food waste.

Plastic levels in digestates were comparable to those found in composts in both magnitude and variability—also spanning seven orders of magnitude on a count per mass basis (Table 2), and just two orders of magnitude on a w/w basis (Table 3), albeit with fewer studies. Plastic counts typically ranged between 70 and 1670 particles dry kg^{-1} in digestates derived from commercial organic waste and codigested manure and food waste, respectively (O'Brien, 2019; Weithmann et al., 2018), with one study reporting up to 3.87×10^7 particles dry kg^{-1} of a digestate of unknown origin (Meixner et al., 2020). On a w/w basis, plastic estimates ranged from 0.01% w/w in digestate derived from the organic fraction of municipal waste (Schwinghammer et al., 2020), to 0.25% w/w in digestate derived from codigested dairy manure and food waste (O'Brien, 2019).

With a limited number of studies reporting microplastic abundance in composts, digestates and food wastes, caution should be taken when drawing any conclusions. Nonetheless, we observed the following patterns: (1) estimated microplastic abundance varies widely both within and between studies of food wastes, composts, and digestates; (2) methods used to quantify microplastics vary widely, and likely exert a strong influence on abundance estimates (see Section 3). (3) The overlapping ranges of microplastic abundance in food waste-derived composts and digestates indicates that neither practice necessarily produces contaminant-free soil amendments and (4) the presence of microplastics in green-waste-derived composts indicates that packaging from food waste is not the only possible source of plastics in organic soil amendments.

3 | MICROPLASTIC MEASUREMENT

To date, there are no standardized methods for measuring microplastics in composts, digestates, and food waste. Methods for measuring microplastics in solid organic matrices such as these typically involve a sequence of steps aimed at isolating, identifying, and characterizing the microplastics in each sample (Ruggero et al., 2020). Isolation methods include flotation, elutriation, centrifugation, digestion (with e.g., H_2O_2, Fenton's reagent), and sieving (Junhao et al., 2021; Ruggero et al., 2020). Identification methods include fluorescence microscopy, thermal degradation (e.g., TED–GC–MS, PY–GC–MS), spectroscopy (e.g., Fourier transform infrared spectroscopy (FTIR), Raman) and visual analysis (with or without light microscopy) (Junhao et al., 2021; Ruggero et al., 2020). It is common for multiple isolation and identification methodologies to be combined in series (Ruggero et al., 2020). Studies of microplastic abundance in food wastes, composts, and digestates largely report values on a count per weight basis (Table 2), with a smaller number of studies reporting values on a weight per weight (w/w) basis (Table 3). Only a third of studies reviewed report values in both units (Braun et al., 2021; Huerta-Lwanga et al., 2021; O'Brien, 2019; Schwinghammer et al., 2020; Sholokhova et al., 2021). Below, we summarize the most common methods used to quantify microplastics in food waste, compost and digestate, as well as some of the challenges that arise due to the lack of standardized methods.

A limited number of studies have measured microplastic abundance in food waste alone (Tables 2 and 3). In these studies, microplastic isolation was achieved by organic matter oxidation with 30–35% H_2O_2 (Ruggero et al., 2021; Schwinghammer et al., 2020), density separation with a saturated salt solution (Golwala et al., 2021; Ruggero et al., 2021), and/or wet sieving (do Carmo Precci Lopes et al., 2019; Kawecki et al., 2020; Schwinghammer et al., 2020). Microplastics were identified using fluorescence microscopy (Ruggero et al., 2021), visual analysis (do Carmo Precci Lopes et al., 2019; Golwala et al., 2021; Kawecki et al., 2020; Schwinghammer et al., 2020), and/or FTIR (Golwala et al., 2021; Ruggero et al., 2021; Schwinghammer et al., 2020).

More studies (albeit still a relatively small number) have examined microplastics in food waste-derived composts or digestates than in food waste itself (Tables 2 and 3). Among studies reporting microplastic abundance on a count per weight basis, isolation strategies included sieving (Edo et al., 2022; O'Brien, 2019; Schwinghammer et al., 2020; Weithmann et al., 2018), organic matter oxidation with 30% H_2O_2 (Edo et al., 2022; Gui et al., 2021; Meixner et al., 2020; Schwinghammer et al., 2020) or Fenton's reagent (Sholokhova et al., 2021), density separation with a saturated salt solution (Braun et al., 2021; Edo et al., 2022; Gui et al., 2021; Meixner et al., 2020; Sholokhova et al., 2021), and centrifugation (van Schothorst et al., 2021). Light microscopy was used in most cases to identify and count putative microplastics based on morphology, color, and response to heat, resulting in values on a count per weight basis. Subsequently, FTIR was used to confirm and identify the polymer type of some or all the putative microplastics (Edo et al., 2022; Gui et al., 2021; Schwinghammer et al., 2020; Sholokhova et al., 2021; van Schothorst et al., 2021; Weithmann et al., 2018). Studies reporting microplastic abundance in composts and digestates on a w/w basis employed more variable methods, including quantification of a single polymer type using alkaline extraction followed by liquid chromatography with UV detection (Müller et al., 2020), direct weighing of larger size fractions (Bläsing & Amelung, 2018; Braun et al., 2021; Kawecki et al., 2020; O'Brien, 2019; Schwinghammer et al., 2020), and estimation based on polymer densities for smaller size fractions (Braun et al., 2021).

There are several challenges associated with current approaches to quantifying microplastics in the scientific literature. The most significant one is that two different units are being used (count per weight and w/w), and there is no consistent way to convert between them without knowing or assuming shape, size, and polymer type (Braun et al., 2021; Leusch & Ziajahromi, 2021). This is problematic not only because it prevents comparison between studies, but also because microplastic ecotoxicity thresholds and regulatory limits are typically delineated on a w/w basis (Leusch & Ziajahromi, 2021; USEPA, 2021a), while 44% of the studies we reviewed reported microplastic abundance in composts and/or digestates exclusively on a count per weight basis (Table 2). This results in a mismatch between science and policy whereby a large fraction of the existing body of knowledge cannot effectively inform regulatory limits. This disconnect makes it difficult to design studies that evaluate microplastic ecotoxicity risk at real world concentrations, or in ways that can contribute directly to existing policy. There is also an inherent challenge in the use of count per weight units

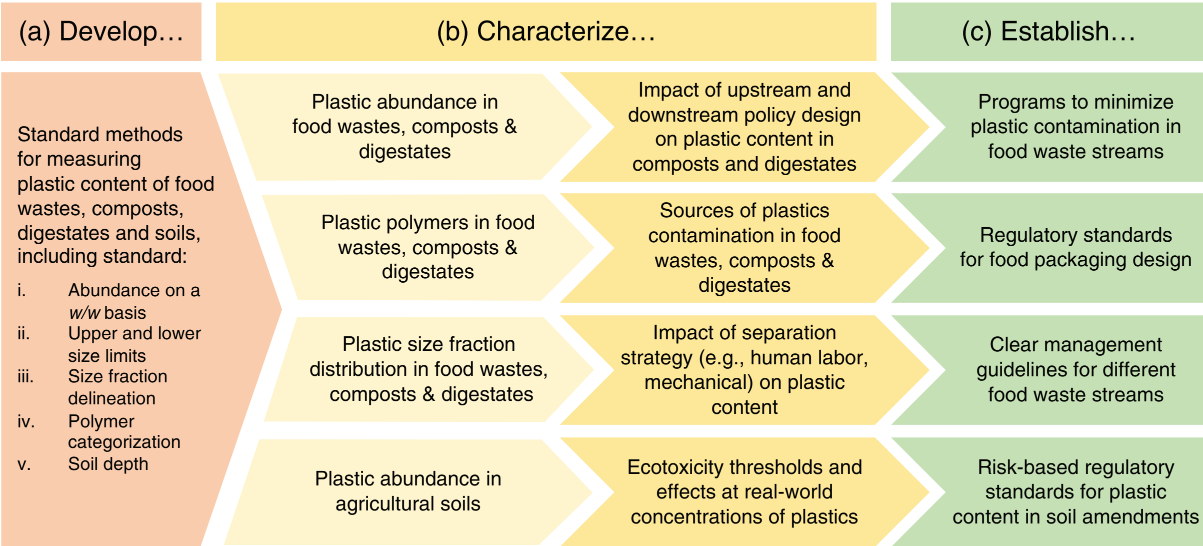

FIGURE 3 Schematic illustrating a design process to harmonize food waste microplastics science and policy

caused by the propensity of plastics to fragment in the environment (Ali et al., 2021). Given that a single macroplastic can fragment into an indeterminate number of micro- or nanoplastics, vastly different abundances could be concluded from the same starting amount of plastic depending on the degree of fragmentation undergone. Variation in microplastic size fraction and other categorizations (e.g., shape, polymer type, color etc.) complicates comparison among studies as well. For example, while it is widely accepted that microplastics are defined as particles <5 mm in size, there is far less consensus on other size-based delineations (Gigault et al., 2018). Macroplastics are sometimes defined as plastic particles >5 mm (L. Zhang et al., 2018), although other studies further divide into meso- (5–25 mm) and macro- (>25 mm) plastics (Braun et al., 2021; Golwala et al., 2021; Gui et al., 2021). The term "nanoplastic" remains under debate as well and has been used to refer to plastic particles less than 0.1, 1, or even 1000 µm throughout the literature (Gigault et al., 2018; R. Qi et al., 2020). Most of the studies reviewed here focused on microplastics >1 mm (Tables 2 and 3). However, some studies reported lower bounds as small as 30 µm (van Schothorst et al., 2021), while others report no lower limit of detection at all (Table 2). On the opposite end of the spectrum, some studies include or even exclusively measure macroplastics (e.g., Kawecki et al., 2020). One final challenge is that the most common methods used to isolate plastics from complex organic matrices may not be appropriate for all polymer types. High-density plastics (e.g., PVC, PET) may not be recovered with density separation and flotation methods (M. Liu et al., 2018), and organic matter oxidation with 30% H_2O_2 has been shown to cause visual changes to PA, PP, PC, PET, and linear LDPE (Nuelle et al., 2014). These methodological differences likely exert a strong influence on total counts of microplastic abundance and underscore the need to develop standardized methods for measuring microplastics in composts, digestates, and food wastes. This should include standard sampling, isolation, and identification protocols as well as known lower thresholds and efficiencies.

4 | IMPLICATIONS FOR AGRICULTURE

Microplastics have been widely documented in agricultural soils. Reported abundance values typically range from the 10s to 1000s of particles per dry kg of soil (Table S2). Land application of contaminated organic amendments is one of multiple potential pathways by which microplastics may enter agricultural soils. Primary microplastics—those that are intentionally engineered to be small (Golwala et al., 2021)—are directly applied to agricultural soils in the form of plastic-coated controlled-release fertilizers, treated seeds, and capsule suspension plant protection products (ECHA, 2020; Stubenrauch & Ekardt, 2020). Secondary microplastics—which form from the breakdown of macroplastics—can be unintentionally added to soils in the form of contaminated soil amendments (e.g., biosolids, composts, digestates) or through the breakdown of plastic mulching (Bläsing & Amelung, 2018; Corradini et al., 2021; F. Zhu et al., 2019). Plastic mulching made with LDPE or biodegradable polymers is often used in agriculture to boost crop yields, suppress weeds, retain water and fumigants, and reduce fertilizer and herbicide requirements (Brodhagen et al., 2017; Serrano-Ruiz et al., 2021). However, plastic mulch can also fragment through time and release microplastics into agricultural soils, and in some cases is even tilled into soils intentionally at the end of the season (Brodhagen et al., 2017; Feng et al., 2021; Serrano-Ruiz et al., 2021; B. Zhou et al., 2020). Other sources of secondary microplastics include irrigation water (B. Zhou et al., 2020), roads (Chen et al., 2020; Sommer et al., 2018), litter (de Souza Machado et al., 2018a), and atmospheric deposition (Bianco & Passananti, 2020; Scheurer & Bigalke, 2018; J. Zhang et al., 2020b). Not all potential sources will necessarily influence microplastic abundance at a specific site (Corradini et al., 2021; L. Yu et al., 2021). More research is needed to understand the relative importance of different pathways of microplastics introduction to agricultural soils, including the use of soil amendments derived from food waste. Effective mitigation will require knowledge of the magnitudes of existing microplastic inputs from all possible sources and the use of reference soils (i.e., experimental controls) to help delineate microplastic inputs from various sources (e.g., distinguish between microplastics introduced by soil amendments versus atmospheric deposition) (Harms et al., 2021; Kumar & Sheela, 2021).

Several recent reviews summarize documented effects of microplastics on soil physical properties, biota and crops (e.g., Iqbal et al., 2020; Ng et al., 2018; R. Qi et al., 2020; J. Wang et al., 2019; B. Xu et al., 2020; Y. Zhou et al., 2020; F. Zhu et al., 2019). These authors largely conclude—as do we based on our review—that the long-term impacts of microplastics in agricultural soils are still poorly understood. A selection of documented effects of plastics (macro-, micro-, and/or nano-) in agricultural systems is summarized in Table S3. Briefly, physical impacts include increased water repellence and porosity, as well as decreased soil bulk density and aggregate size (e.g., de Souza Machado et al., 2018b, 2019; Kim et al., 2021; Y. Qi et al., 2020; see Table S3 for additional references). Delayed or reduced seed germination, reductions in plant growth, and uptake into plant tissues have been documented in multiple crop varieties (e.g., Boots et al., 2019; Pflugmacher et al., 2020; Tympa et al., 2021; see Table S3 for additional references). Effects on soil microbes include changes in biomass, species dominance, diversity, and richness (e.g., Blöcker et al., 2020; Fei et al., 2020; Ren et al., 2020; see Table S3 for additional references). Oxidative stress, abnormal gene expression, gut microbiota perturbation, and movement inhibition have been observed in soil macrofauna (e.g., Cheng et al., 2020; Kim & An, 2019; D. Zhu et al., 2018; see Table S3 for additional references). Microplastic ingestion and bioaccumulation has also been reported in some livestock species (Beriot et al., 2021; Huerta Lwanga et al., 2017; J. Yang et al., 2021). These impacts tend to vary by polymer type, size and shape, soil characteristics, microplastic dose, and exposure time (de Souza Machado et al., 2018b; Lozano et al., 2021; Zhao et al., 2021). For instance, plant biomass reductions were only observed for certain polymer types but not others (de Souza Machado et al., 2019; Y. Qi et al., 2018; F. Wang et al., 2020; M. Yang et al., 2021), at certain sizes but not others (Z. Li et al., 2020; M. Yang et al., 2021), or under certain soil pH conditions (Y. Liu et al., 2021). In one study, for example, the dry biomass of spring onion (*Allium fistulosum*) bulbs decreased with exposure to polyamide beads when compared with an untreated control, but nearly doubled with exposure to polyester fibers (de Souza Machado et al., 2019). These effects may not be limited to conventional petroleum-based microplastics either—there have been reports of biodegradable plastics having ecotoxic effects in soils as well (Boots et al., 2019; Iqbal et al., 2020; Y. Qi et al., 2018; Serrano-Ruiz et al., 2021).

While several studies report potential negative effects of microplastics in soil–plant systems, existing data are not sufficient to fully evaluate the risks of microplastics in agricultural soils (Gouin et al., 2019; USEPA, 2021a). The lack of common units between many microplastic ecotoxicity and abundance studies precludes evaluation of the environmental relevance of the microplastic doses at which negative effects are observed (Leusch & Ziajahromi, 2021). Connors et al. (2017) suggest nine areas of improvement to advance the quality of environmental microplastic research, which we suggest should be applied in the context of food waste-derived soil amendments and agricultural soils:

> 1. Environmental relevance of test concentrations, 2. Provision of sufficient detail for converting particle concentrations, 3. Thorough characterization and/or description of test particles, 4. Detailed reporting of particle preparation techniques and [stability], 5. Analytical verification of test concentrations, 6. Consideration of the environmental relevance of particle size, 7. Inclusion of appropriate controls… 8. Consideration of endpoint applicability to environmental risk assessment framework… [and] 9. Reporting findings accurately, without conjecture beyond experimental limits.
>
> (p. 1702)

Additional research is also needed to determine remediation options for soils that have already been contaminated with microplastics.

5 | HARMONIZING SCIENCE AND POLICY

Prevailing scientific uncertainty creates a challenging context for policy design related to microplastics and food waste diversion efforts. Scientists continue to debate the risk posed by microplastics generally and the best course of action for risk management, with differing viewpoints (Backhaus & Wagner, 2020; Burton, 2017; Coffin et al., 2021; Gouin et al., 2019; Hale, 2018; Kramm et al., 2018). Most scientists continue to frame microplastic risks as uncertain, which stands in contrast to the prevailing media narrative that microplastics are emphatically harmful to humans and the environment (Völker et al., 2020). Despite the lack of scientific consensus on the risks posed by microplastics in soils and the relative input from organic amendments, a growing number of entities have imposed regulatory thresholds for microplastics in composts and digestates (USEPA, 2021a). Thirteen U.S. states (California, Iowa, Maryland, Minnesota, Montana, New Hampshire, New York, North Carolina, Ohio, Rhode Island, South Carolina, Washington, and Wisconsin) have enacted regulatory limits on physical contaminants in compost, and the state of California regulates physical contaminants in both composts and digestates (USEPA, 2021a). Total physical contaminant limits (a category encompassing glass, metal, and other human-made inert materials in addition to plastics) range from 0.5 to 6% w/w with most falling in the 1–2% w/w range (USEPA, 2021a). Four of the thirteen states (California, Maryland, Ohio, Washington) have additional limits specifically for plastics or film plastics ranging from 0.1 to 2% w/w (USEPA, 2021a). Only five states specify a lower size threshold for consideration—4 mm in all cases—though testing requirements and detection limitations may implicitly determine the size fractions measured (USEPA, 2021a).

Regulations tend to be more stringent outside the United States, with limits largely falling between 0.25 and 0.5% w/w for total physical contaminants and between 0.05 and 0.5% w/w for plastics or film plastics (USEPA, 2021a). Most countries set the lower size threshold for consideration at 2 mm except for Germany, which regulates particles >1 mm (USEPA, 2021a).

There are multiple limitations to the existing regulatory approach to microplastic contamination in composts and digestates. First, regulatory standards are in units of w/w, while 44% of the studies we reviewed reported microplastic abundance in composts and/or digestates exclusively on a count per weight basis (Table 2). This results in a mismatch between science and policy whereby a large fraction of the existing body of knowledge cannot effectively inform regulatory limits. Second, due to an incomplete understanding of the risks posed by microplastics in soils under different conditions (e.g., dosing rates, edaphic factors, polymer types, size distributions etc.), allowable contamination levels and lower particles size thresholds may instead be determined by aesthetic concerns and detection limits rather than known risk (USEPA, 2021a). Third, regulating microplastics content in finished products, without considering the fertilizer value of the material or application rate, does not limit the ultimate flow of microplastics to soils via organic amendments. For example, under the current regulatory structure, it may be permissible to land apply a large amount of microplastics in a dilute form, but not a smaller amount of microplastics in a more concentrated form. Finally, regulating contamination levels in organic amendments alone may be insufficient to fully mitigate the flow of microplastics into agricultural soils given the existence of other entry points (see Section 4).

There are other examples of narrowly focused microplastics policy that similarly do not address multiple pathways of introduction to the environment. For example, current or proposed policies in the United States, European Union, China, and South Korea restrict the use of primary microplastics in cosmetic products, but exclude other sources of microplastics (e.g., plastic mulching, plastic packaging, tires) (Mitrano & Wohlleben, 2020). There are, however, existing regulations that could be applicable to microplastics and should be considered in current discussions. Certain heavy metals in biosolids, for example, underwent rigorous toxicity assessments to determine allowable contamination thresholds grounded in scientific evidence (Lu et al., 2012). Currently, the same is not true for microplastics in composts and digestates; thus, current regulatory thresholds lack a scientific basis, and the benefits of such regulations are largely unknown. Given the persistence of microplastics, uncertainties regarding toxicity, and the upward trend in both plastic production and environmental detection, some have argued for a more precautionary approach than the traditional regulatory paradigms for threshold contaminants (Coffin et al., 2021). Depending on the degree of precaution taken, this type of approach could create undesirable tradeoffs in the context of present-day food waste diversion efforts. If, for example, extremely strict limits for microplastic presence in soil amendments are put in place as a precautionary measure, this could potentially lead to a return to landfilling most food waste streams and subsequent methane emissions to the atmosphere. Therefore, it is critical to consider counterfactual scenarios: The net benefit of diverting food waste from landfills must be weighed against the potential cost of sending additional microplastics to agricultural soils. Future studies incorporating microplastics into life cycle analyses of food waste management strategies could help to elucidate these tradeoffs.

We propose the following path forward to better align efforts to quantify microplastics in organic amendments, understand their effects in soils, and establish related policy. First, standardized methods for measuring microplastics in food wastes, composts, digestates, and soils must be developed, ideally at the national or international level to enable collaboration and data comparison (Figure 3a). We recommend the development of methods that generate abundance values on a w/w basis, given that these are the units that are used in ecotoxicity studies and existing regulatory structures (Leusch & Ziajahromi, 2021; USEPA, 2021a). Using these standard methods, future studies should characterize both the extent of microplastic contamination in food wastes, composts, digestates, and soils as well as the sources, impacts, and most effective strategies to mitigate this contamination (Figure 3b). Third, if toxicity is well established, evidence- and risk-based regulatory measures can be implemented to reduce microplastic contamination from all sources (Figure 3c).

This will take time and precautionary steps in the interim are, in our opinion, justified to help limit contamination. For example, in Germany, a limit of 0.1% w/w has been established for film plastics >1 mm in fertilizers (USEPA, 2021a). Based on the range of microplastic contamination values available to date (Table 3), this would theoretically eliminate land application of the most contaminated materials. Plastic contamination in food waste-derived composts and digestates could also be reduced through innovations in mechanical depackaging technology and improved source separation. However, given the challenges of achieving perfect separation in both instances (Dai et al., 2016; do Carmo Precci Lopes et al., 2019; Edwards et al., 2018; Friege & Eger, 2021), the most transformative solutions may lie in redesigning the way we package food in the first place. Biodegradable plastics are a promising alternative to conventional petroleum-based plastics because they can be broken down by microbes into nontoxic compounds like carbon dioxide and water (Folino et al., 2020; Shaikh et al., 2021). However, the current array of biodegradable plastics on the market come with remaining challenges (Calabró & Grosso, 2018; European Commission, 2018; Haider et al., 2019; Markowicz & Szymańska-Pulikowska, 2019; Serrano-Ruiz et al., 2021). First, the terminology used to label these plastics can be misleading. “Bioplastics” can be bio-based (made from renewable carbon sources) or biodegradable (able to be broken down by organisms) or both (Folino et al., 2020). Therefore, not all bioplastics are bio-based or biodegradable and some biodegradable plastics are actually petroleum-based (Folino et al., 2020). Some biodegradable plastics also meet ASTM criteria established for compostable materials, meaning that they can be broken down within a similar timeframe as natural materials (e.g., food scraps) under controlled composting conditions (Brodhagen et al., 2017; Shaikh et al., 2021). However even these standards allow for the persistence of fragments <2 mm, therefore compostable microplastics may still accumulate in soils over time (Brodhagen et al., 2017). In fact, most biodegradable plastics (including compostable plastics) do not fully degrade under all field or operational conditions they might be subject to (e.g., compost windrows, anaerobic digesters, and soils) (Brodhagen et al., 2017; Calabró et al., 2020; Folino et al., 2020; Haider et al., 2019; Huerta-Lwanga et al., 2021). Furthermore, compostable products are not all environmentally benign—some are treated with per- and polyfluoroalkyl substances and can have environmental footprints larger than those of noncompostable alternatives (Mistry et al., 2018). Finally, consumers can mistake noncompostable plastic and plastic-coated foodware for compostable versions, increasing contamination in some instances (Mistry et al., 2018). Clearly, this generation of biodegradable and compostable plastics is not a panacea for the issue of microplastics contamination in food waste (Folino et al., 2020); however, advances in the field could enable a more circular economy of resource use in the future.

Technological advances that improve separation of food waste from packaging in conjunction with evidence-based regulations on microplastics content in food waste-derived composts and digestates could help to limit the flow of microplastics to soils. Both upstream and downstream interventions such as these should be analyzed in future studies to determine their effectiveness and the resultant benefits and burdens for the environment. Ultimately, plastic packaging that was developed within a linear economy is not designed to function within a circular economy model. Advances in green chemistry for packaging (Deng et al., 2021; Kramm et al., 2018), stricter regulations of biodegradable and compostable plastics (Brodhagen et al., 2017) and elimination of unnecessary packaging should all be pursued as part of a comprehensive approach to reducing microplastic pollution, including that originating from food waste diversion programs.

6 | CONCLUSIONS

Microplastic abundance varies widely within and among studies of food wastes, composts, and digestates. There is some evidence that microplastics may adversely affect soils and plants; however, lack of common units between microplastic ecotoxicity and abundance studies precludes rigorous assessment. Existing regulations establish weight-based limits for finished composts and digestates, which is incongruent with many scientific studies that use count-based estimates of microplastic abundance. Further work is necessary to elucidate tradeoffs associated with diverting food waste from landfills and to design policies that maximize the benefits of recovering food waste while minimizing the risk of microplastic pollution in soils.

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AUTHOR CONTRIBUTIONS
Katherine K. Porterfield: Conceptualization; data curation; formal analysis; investigation; methodology; visualization; writing –original draft; writing – review & editing. **Sarah A. Hobson**: Conceptualization; data curation; formal analysis;

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CONFLICT OF INTEREST

The authors declare no conflict of interest.

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