Imperial College London

Myth or reality? – Assessing the suitability of biodegradable plastics within a circular bioeconomy framework

A thesis submitted for the degree of
Doctor of Philosophy

by

Sarah Kakadellis

Centre for Environmental Policy

Imperial College London

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Pour méng φέτα rodzina

“*This permanent doubt, the deep source of science.*”
— Carlo Rovelli, Reality Is Not What It Seems

“*When we first begin […], we have no experience and make many mistakes. The secret of life*[1]*, though, is to fall seven times and to get up eight times.*”
— Paulo Coelho, The Alchemist

[1] And, by extension, of a PhD

Declaration of Originality

I declare that this thesis, entitled *Myth or reality? – Assessing the suitability of biodegradable plastics within a circular bioeconomy framework*, is entirely my own work and that where any material could be construed as the work of others, it is fully cited and referenced and with appropriate acknowledgement given.

Sarah Kakadellis

Copyright Declaration

Abstract

The proposition of a circular bioeconomy framework was introduced as a means of moving from a fossil-based to bio-based economy. With an emphasis on resource efficiency and waste valorisation, it has supported the development of biodegradable bioplastics (BBPs), notably in food packaging applications. Designed to be treated alongside organic waste, BBPs open new streams for plastic waste management within the food-energy-waste nexus, but their suitability in the current social, policy and sustainability landscape remains to be determined. Taking a systems-thinking approach, this thesis explores the compatibility of (certified) BBP packaging under a circular bioeconomy framework, focusing on a co-mingled food and BBP waste stream for anaerobic co-digestion. It uncovers major technical, policy and social challenges and urges for caution when deploying these novel plastic packaging materials on the consumer market.

Chapter 1 provides a brief introduction to BBPs and their framing in the wider context of plastic sustainability and organic waste management, followed by aims and objectives in **Chapter 2**.

Chapter 3 provides a comprehensive literature review, depicting the importance and ubiquity of plastics, their environmental impact and the role BBPs could play in a circular bioeconomy framework from a systems-thinking perspective.

Chapter 4 details the anaerobic co-digestion treatment of different BBPs with food waste and the impact of BBPs on biogas and methane yields and on microbial communities. The need for consistent experimental design of co-digestion trials is also discussed.

Guided by these results, **Chapter 5** presents the outcomes of a stakeholder study on attitudes towards BBPs in the current waste management infrastructure and policy landscape to explore how BBPs are perceived and managed on-the-ground in the United Kingdom.

Chapters 6 & 7 build on a major finding from the stakeholder study, which outlined the importance of consumers in enabling circularity in the system. **Chapter 6** covers a systems framework developed to identify and structure systemic factors that influence how consumers interact with BBP packaging, with a focus on disposal routes. The framework is then applied in practice, based on a survey conducted at two academic institutions, and the role of contextual setting is explored through a comparative case study presented in **Chapter 7**.

Chapter 8 extends the debate on the suitability of BBPs further upstream in the value chain and consumption system and addresses the functional properties of BBP packaging in the context of a shelf-life study, anchoring BBPs in the food system they are embedded within.

Chapter 9 summarises key findings and suggests future research related to this thesis.

The **Appendix** contains supplementary figures and data for Chapters 4-8.

Acknowledgements

First and foremost, my thanks go to Zoe for being an incredible, supportive and, it has to be said, kick-ass supervisor. I know I have tended to get carried away at times (and utterly demotivated at others) and I truly appreciate your commitment to keep me focused while allowing me to develop as a researcher (and science communicator!). A warm thank you also to my co-supervisors, Henry and Jem, as well as my collaborators Beth, Gail, Marco and Ned, for your support while encouraging me to explore new research avenues.

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Villmools merci to my kochającej family for never ceasing to believe in me (#warrior), despite not becoming the next Marie Skłodowska–Curie or Lady Fleming. The next 200 pages will probably be as soporific as one of those *Curia* translations lying around the house, but it is thanks to your unconditional love and educational values (and μπουμπουναρίες) that I came to be where I am today.

A special shoutout goes to my beshtie Glo, for all the wonderful memories we built over the course of our PhDs and for keeping up with my mood swings and occasionally abysmal silences. Being able to grow and exchange together (as peas from the same pod do), both personally and academically, has truly been one of the highlights of this journey.

To all the wonderful friends I have made along the way, thank you. It takes a village to raise a child – it takes a bunch of bike rides, Parkland/Thames walks, wild camping and hiking trips, strolls to garden centres and coffee shops, joint Spotify sessions, Fernbank retreats, cosy sleepovers, trivia nights, cooking seshes and a ludicrous amount of chocolate (and other *shnacks*) to complete a PhD, and I couldn't have done it without you all.

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Abbreviations

Abbreviations used once or twice and/or limited to a single paragraph are not displayed.

AD	Anaerobic digestion
ANOVA	Analysis of variance
ASTM	American Society for Testing & Materials
BBP(s)	Biodegradable bioplastic(s)
BEIS	Department for Business, Energy & Industrial Strategy
BioN	Natural (undyed) biodegradable plastic
BioP	Printed (dyed) biodegradable plastic
BMP	Biochemical methane potential
C	Carbon
CH_4	Methane
C:N	Carbon-to-nitrogen ratio
CO_2	Carbon dioxide
co-AD	Anaerobic co-digestion
CO_2eq	Carbon dioxide equivalent
COM-B	Capability, opportunity and motivation behavioural model
DEFRA	Department for Environment, Food & Rural Affairs
EN 13432	European standard/norm for industrially compostable packaging
EoL	End-of-life
EU	European Union
FMCG(s)	Fast-moving consumer good(s)
FW	Food waste
GGM	Gaussian graphical model
GHG(s)	Greenhouse gas(es)
H	Hydrogen

H_2	Dihydrogen
HDPE	High-density polyethylene
H&F	London Borough of Hammersmith & Fulham
H_2O	Water
HRT	Hydraulic retention time
ICL	Imperial College London
I:S	Inoculum-to-substrate ratio
ISO	International Standardisation Organisation
K&C	Royal Borough of Kensington and Chelsea
LCA	Life-cycle assessment
LDPE	Low-density polyethylene
MAP	Modified atmosphere packaging
MP	Macro-perforated (film)
miP	Micro-perforated (film)
MSW	Municipal solid waste
(M)t	(Million) metric tons
N	Nitrogen
NFU	National Farmers Union
NH_3	Ammonia
NH_4^+	Ammonium
NGO(s)	Non-governmental organisation(s)
n_X	Sample size of population X
N_X	Network of population X
O	Oxygen
O_2	Dioxygen
OFMSW	Organic fraction of municipal solid waste

PA(s)	Polyamide(s)
PAS 110	Publicly available specification for digestate in the United Kingdom
PBSA	Polybutene succinate adipate
PE	Polyethylene
PET	Polyethylene terephthalate
PHA(s)	Polyhydroxyalkanoate(s)
PLA	Polylactic acid
PP	Polypropylene
PS	Polystyrene
PVC	Polyvinyl chloride
P_X	Population X
SDG(s)	Sustainable Development Goal(s)
SE	Standard error
SFW	Synthetic food waste
TPB	Theory of planned behaviour
TPS	Thermoplastic starch
TS	Total solids
UCD	University of California, Davis
UK	United Kingdom
UN	United Nations
UNEP	United Nations Environment Programme
US	United States
WRAP	Waste and Resources Action Plan
WTP	Willingness-to-pay
WVTR	Water vapour transmission rate
w/w	Weight by weight

Definitions of key terms

Anaerobic digestion: Process by which organic material, such as garden, animal or food waste is broken down by microorganisms in the absence of dioxygen (O_2) to produce biogas, which can be upgraded to biomethane, and digestate, a natural fertiliser.

Bioplastics (biopolymers): Umbrella term capturing a range of plastics that are bio-based, biodegradable (including certified industrially and home compostable), or both.

> **Bio-based**: (Partly or fully) made from biological and renewable resources.
>
> **Biodegradable**: Degradable by microorganisms into water (H_2O), carbon dioxide (CO_2), methane (CH_4) and inorganic compounds under certain conditions.
>
> **Industrially compostable**: Biodegradable under well-defined conditions representative of industrial composting practices according to harmonised standards.

Circular bioeconomy: Framework based on two fundamental pillars, the use of renewable natural capital (**bioeconomy**) and the design of closed material, component and product loops (**circular economy**), to achieve sustainable wellbeing within planetary boundaries.

Consumer behaviour chain: Methodological approach that maps individual behaviours performed sequentially across acquisition, use and disposal stages (the consumption phase).

Conventional plastics (polymers): Umbrella term for a large family of synthetic materials made from by-products of the oil industry, moulded under heat and pressure. The term is often used interchangeably with polymers, which refers to long chains of repeating units (monomers).

Food wastage: Discarded food that would be suitable for human consumption and consists of both **food loss**, which captures wastage from agricultural production, post-harvest handling or processing, and **food waste**, wastage that occurs further down the food supply chain during distribution, retail sale and consumption at the consumer level – note that this thesis focuses on the food waste component of food wastage.

Organic waste (also referred to as **biowaste**): Waste derived from plant or animal resources that can be broken down and assimilated (i.e. biodegraded) by microorganisms.

Systems-thinking: Approach to study a given topic by looking at system components and their relationships as a whole rather than individually. This approach is particularly suitable for addressing complex problems, such as sustainability issues.

Shelf-life: The length of time during which a product remains fit for consumption or meets marketability criteria.

Note to the reader

COVID-19 impact statement

The direction of this PhD research was heavily influenced by the COVID-19 pandemic. Experimental data collection from co-digestion assays related to **Chapter 4**, conducted at the University of Oxford in collaboration with the Department of Engineering Science, was interrupted in March 2020 and further data analysis could not be subsequently resumed due to the timeline of the collaboration. Thus, digestate quality, as well as physical and chemical properties of co-digested bioplastic fragments originally planned could not be analysed.

Research activities were adjusted to adapt to remote working, while still aligning with the overarching research question outlined at the start of the PhD. Influenced by findings from the stakeholder study presented in **Chapter 5**, a new focus on consumer behaviour was introduced, with an emphasis on systems-thinking (see **Chapters 6 & 7**), in collaboration with the Dyson School for Design Engineering at Imperial College London. The restoration of international travel and the opportunity to build upon an established collaboration within the Department of Plant Sciences at the University of California, Davis contributed to an additional experimental study, presented in **Chapter 8**.

Due to the aforementioned access restrictions to laboratory space and the collaborations that ensued, this thesis is highly interdisciplinary and addresses the suitability of biodegradable plastics within a circular bioeconomy framework from biochemical, logistical, social, behavioural, technical and policy angles.

Thesis content-related publications

Content from the following chapters has been adapted and expanded from peer-reviewed publications related to or stemming from research conducted for this thesis (further signposted at the start of individual chapters). All publications are open access under a Creative Commons Attribution International Licence (CC BY 4.0). Some figures have been directly reproduced from these publications and have been appropriately referenced in the thesis.

All publications were led by the author of this thesis, with the entirety of analyses conducted by her alone. Most co-authors are PhD supervisors/advisors. None of the following publications have been/are planned on being used by any other student as part of their own thesis and all authors agreed on these publications contributing towards this thesis solely.

Publications related to/stemming from this work (in chronological order):

Kakadellis, S. & Harris, Z. M. (2020). Don't scrap the waste: The need for broader system boundaries in bioplastic food packaging life-cycle assessment – A critical review. *Journal of Cleaner Production*, 274, 122831.

Kakadellis, S., Woods, J. & Harris, Z. M. (2021). Friend or foe: Stakeholder attitudes towards biodegradable plastic packaging in food waste anaerobic digestion. *Resources, Conservation and Recycling*, 169, 105529.

Kakadellis, S. & Rosetto, G. (2021). Achieving a circular bioeconomy for plastics: reframing the debate. *Science*, 373, 6550, 49-50.

Kakadellis, S., Lee, P.-H. & Harris, Z. M. (2022). Two Birds with One Stone: Bioplastics and Food Waste Anaerobic Co-Digestion. *Environments*, 9 (1), 9.

Kakadellis, S., Muranko, Ż., Harris, Z. M. & Aurisicchio, M. (2023). *Closing the loop: enabling circular biodegradable bioplastic flow through a systems-thinking framework*. Manuscript submitted for publication and in peer-review.

Kakadellis, S., Lingga, N., Zhu, A., Bikoba V. N., Taylor, G., Mitcham, B. J. & Harris, Z. M. (2023). *Effects of biodegradable plastic packaging and perforation rates on the shelf life of leafy greens*. Manuscript in preparation.

Thesis abbreviation and referencing style

As per College guidelines, the Harvard referencing style was adopted. Recurrent key terms were abbreviated after first use, apart from headings and figure legend titles (excluding "BBP(s)", which was abbreviated throughout to facilitate legibility).

Chapter 1 – Introduction

Part of the content presented in **Chapters 1 & 3** appears in the following publications:

Kakadellis, S. & Harris, Z. M. (2020). Don't scrap the waste: The need for broader system boundaries in bioplastic food packaging life-cycle assessment – A critical review. *Journal of Cleaner Production*, 274, 122831.

Kakadellis, S. & Rosetto, G. (2021). Achieving a circular bioeconomy for plastics: reframing the debate. *Science*, 373, 6550, 49-50.

Within a century, in the wake of petrochemical alliances (Freinkel, 2011), plastics have become the most ubiquitous and controversial class of man-made, synthetic materials. The visual nature of plastic pollution and the scandals of plastic waste exports to developing nations have prompted a shift in how plastics are made, used and disposed of (Williams & Gregory, 2021). Plastic waste remains poorly managed, with as many as 12,000 million tonnes (Mt) of plastic waste projected to have accumulated in landfills or the natural environment by 2050 (Geyer, Jambeck & Law, 2017). Whilst mechanical recycling was initially promoted as the solution to rising levels of post-consumer plastic waste, its failure to do so over the past decades has exposed the severity and scale of the plastic waste management crisis (d'Ambrières, 2019).

Recognising the significant economic, social and environmental costs associated with a linear take-make-use-dispose approach traditionally applied to the plastics value chain, the circular economy has aimed to redefine growth by focusing on keeping materials and products in use, turning waste into a resource and regenerating natural systems (Ellen MacArthur Foundation, 2015). The concept of circular economy is often accompanied by the bioeconomy, which promotes the sustainable use of renewable biological resources and includes supporting new sources of renewable energy and producing 'smarter', 'greener' and cheaper materials (Tan & Lamers, 2021).

Shunning conventional fossil-based plastics has provided a fertile ground for the emergence of alternative materials, loosely referred to as bioplastics, an umbrella designation that captures a range of polymer chemistries, properties, and application sectors. There is a clear need for divesting from fossil fuels, and thus bio-based sources are necessary. However, life-cycle analyses have uncovered complexities in the system, in part due to agricultural inputs for bioplastic feedstock production (Kakadellis & Harris, 2020). In the context of a circular bioeconomy, biodegradable bioplastics (BBPs) have received particular attention, due to their ability – at least conceptually – to extend bio-circular design to the end-of-life (EoL).

The biggest advantage of BBPs may not be their biodegradability per se but their compatibility with food and other organic wastes (Kakadellis & Harris, 2020). This benefit opens new EoL streams for plastic waste management uniquely positioned around the treatment of the organic fraction of municipal solid waste (OFMSW). Initially, Industrial composting appeared as the most common EoL route, leading to certified BBPs being commonly referred to as [industrially] 'compostable' plastics. More recently, anaerobic digestion (AD), which enables the conversion of OFMSW into renewable energy and the creation of a circular carbon and nutrient cycle within the food-energy-waste nexus, has gained increasing attention. BBP waste could, in theory, be collected and treated together with OFMSW in AD, including commercial and household food waste (FW).

Nevertheless, issues associated with feedstock separation and contamination in existing mechanical recycling streams for conventional, non-biodegradable plastics and concerns over theoretical vs observed biodegradability of BBPs in the current organic waste management infrastructure continue to be raised (Kakadellis, Woods & Harris, 2021), especially in the absence of a certification for 'AD-able' BBPs. Despite favourable public opinion, consumer awareness and understanding of the subtleties in bioplastic terminology remain poor (Dilkes-Hoffman et al., 2019a), reflected by misaligned behavioural patterns for BBP disposal (Taufik et al., 2020). In addition, research on bioplastics tends to occur in isolation, with little consideration for the wider social, behavioural and policy context. The recent public consultations on bio-based, biodegradable and compostable plastics and global commitments to frame the plastics value chain within a circular bioeconomy, including the United Nations (UN)'s draft resolution for an international legally binding instrument on plastic pollution published in March 2022 (UNEA, 2022), reflect the relevance and need for further research in this field.

A more holistic view of the complex system within which BBP food packaging exist is needed. A seemingly benign task at the end of the food supply chain – disposing and treating BBPs through AD in a co-mingled stream alongside FW – involves a range of stakeholder groups (e.g. consumers, local authorities, AD operators), locations (e.g. home, on-the-go, AD plant site) and scales (e.g. local, national, international) (Allison et al., 2022a). Adopting a systems perspective would help to better characterise existing challenges related to the use of BBP food packaging and their disposal in AD and identify relevant intervention policies and design strategies to support policymakers, regulators, plastic manufacturers, the waste management industry and, pivotal when addressing disposal, consumers, on their path towards a sustainable, closed-loop economy for plastics.

Chapter 2 – Aims & Objectives

BBPs represent one potential solution to address plastic pollution by contributing towards closed cycles within the food-energy-waste nexus, where BBP food packaging could become part of an integrated waste management strategy for the treatment of OFMSW, including commercial and household FW. However, **Chapter 1** highlighted some of the complexities and pitfalls associated with BBPs, which need to be investigated to ensure BBPs address, rather than exacerbate, plastic pollution.

So far, research efforts have been dedicated to the development of BBPs and, more recently, to the evaluation of the extent and mechanism of their biodegradability in aerobic environments (composting and soil). While important, this highly siloed approach, focused on the (bio)chemistry of BBPs, presents two challenges. First, study design is often conducted hermetically, with little consideration for relevant waste streams and/or commercial waste management practices. Second, the contribution of actors, in particular the role consumers play in enabling appropriate disposal (and thus capture) of BBP waste, is poorly characterised.

This research explores the suitability of BBPs within a circular bioeconomy framework in the context of municipal organic waste management through anaerobic digestion. It is uniquely embedded within a systems-thinking perspective, distinguishing itself from a highly siloed traditional research approach. Through this interdisciplinary approach, it will contribute towards a better characterisation and analysis of the main policy, technical and behavioural barriers and opportunities in the system within which BBPs are embedded, focusing on the United Kingdom (UK), European Union (EU) and to some extent Unites States (US). The aims of this work can be summarised as follows:

1. Assess the impact of BBPs on AD performance (EoL design) and on produce shelf-life (functionality design);
2. Investigate the compatibility of BBP packaging under current and projected organic waste management infrastructure;
3. Characterise the system within which BBP food packaging exist;
4. Investigate the role of consumers as enablers of BBP packaging flow across the consumption phase, with an emphasis on disposal behaviour.

To achieve these aims, a set of objectives and corresponding thesis chapters were identified:

1. Quantify changes in gas composition and microbial community structure and function when individual BBP materials are introduced (**Chapter 4**);
2. Undertake interviews with industry, policy and non-profit sectors to explore current waste management practices and stakeholder attitudes towards BBPs (**Chapter 5**);

3. Develop a systems framework capturing the system elements and interdependencies in the BBP value chain, framed as complex adaptive system (**Chapter 6**);
4. Apply the framework developed to explore consumer disposal behaviour under distinct contextual settings based on a comparative case study approach (**Chapter 7**);
5. Compare the effect of conventional plastics and BBPs on the shelf life of fresh food, using leafy greens as example (**Chapter 8**).

The research aims and associated objectives mentioned above will contribute towards an improved understanding of the systemic impact of emerging plastic alternatives that will inform both industry and policymakers. Ultimately, this research strives to ensure that the promotion of BBPs within a circular bioeconomy framework is based on environmentally sound evidence and is compatible with circularity ambitions.

1. Statement of Novelty

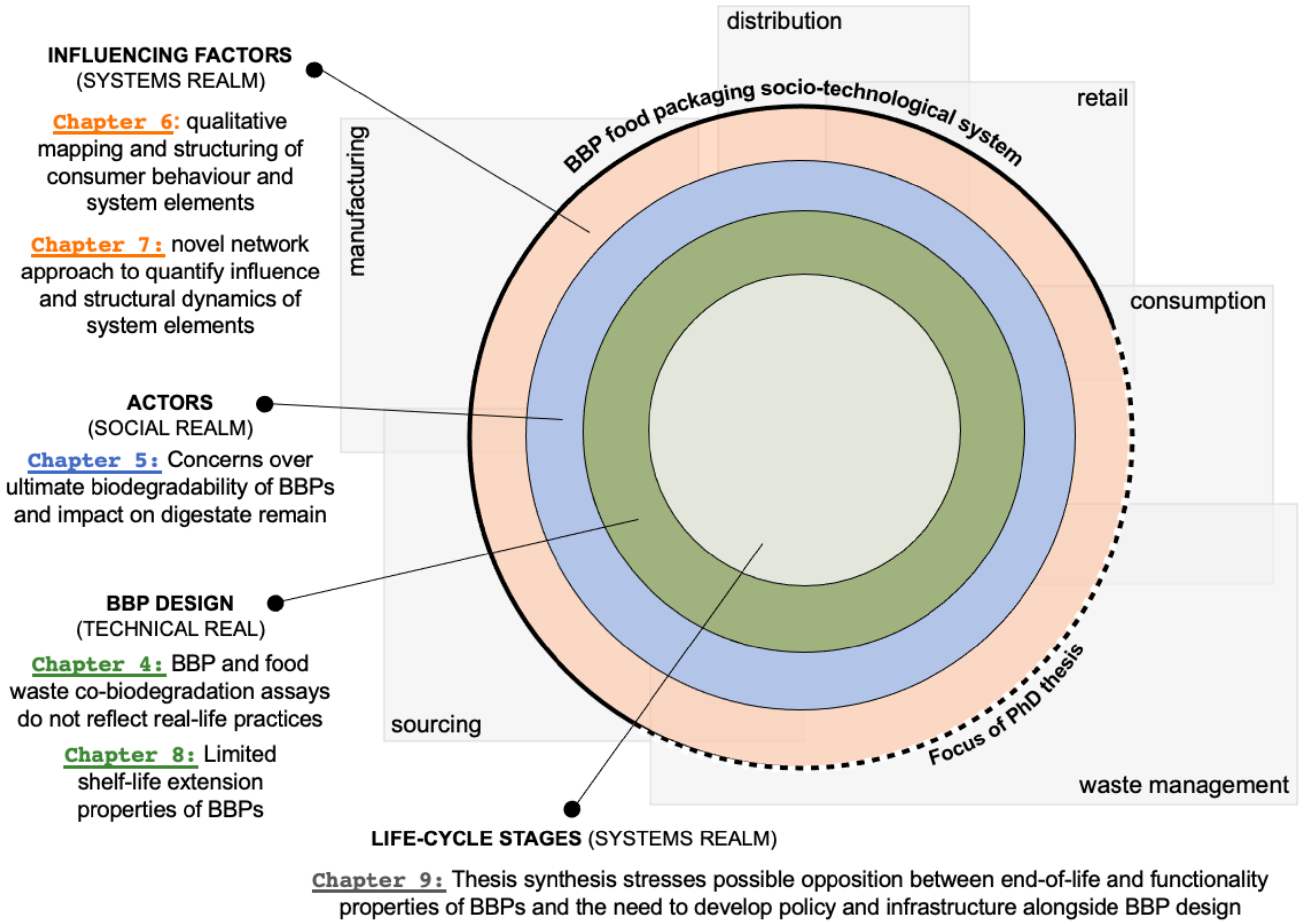
While some of the findings of the studies conducted as part of this PhD are not in themselves novel, the integration of individual studies and their associated chapters in an interdisciplinary way represents a novel contribution to the field. This integration has enabled the appraisal of BBP food packaging from a range of academic disciplines and methodologies not commonly studied together, thereby providing a synthesis novelty.

For example, biochemical and microbiological characterisation presented in **Chapter 4** is often conducted in biodegradation studies that tend to consider the sustainability of a given BBP solely from a biochemistry/material science perspective while omitting practical routes of disposal outside of the controlled laboratory environment. The latter point, however, may play a critical role in determining how sustainable a given material will truly be in practice, i.e. how suitable it is within its relevant waste management context. Aligning theoretical BBP biodegradability with a more realistic on-the-ground assessment was achieved in **Chapter 5**, where experimental results were compared and contrasted against practical cases of food and BBP waste co-digestion through semi-structured stakeholder interviews. This, in turn, enabled a novel and critical appraisal of the limitations of current experimental design in the scientific literature.

Chapters 6 & 7 address the overall research question from further perspectives: design and psychology. In this case, a novel methodology rooted in behaviour chain mapping was developed and applied in practice to investigate consumer behaviour in the context of BBP packaging waste disposal more holistically. Some of the study results confirmed the role of some key factors influencing appropriate disposal of BBPs by consumers previously identified in the literature. Nevertheless, the integration of both qualitative and quantitative data through focus groups and surveys coupled with a comparative case study helped shine a light on the underlying mechanisms, explaining *how* identified factors may exert their influence.

Lastly, climbing up the food supply chain and addressing functionality design in **Chapter 8** through shelf-life experimental data contributed towards extending the sustainability debate beyond the consumption and waste management stages. Comparing both functionality and EoL side-by-side by drawing from experimental chapters (**Chapters 4 & 8**) contributed towards a novel perspective of what it means for BBPs to be sustainable in the context of sustainable food and food packaging systems.

Importantly, adopting an interdisciplinary approach has led to a more comprehensive and robust characterisation of the socio-technological system BBPs are embedded within, which encompasses multiple life-cycle stages (e.g. manufacturing, consumption, waste management), actors (e.g. legislators, regulators, consumers and waste management practitioners) and influencing factors, or system elements (e.g. intrinsic drivers, infrastructure, policy). This socio-technological system can be visualised in the graphical abstract below, which also synthesises the contribution of individual chapters towards the overall narrative.



The system is represented as a combination of layers in an onion-like arrangement. The BBP life-cycle (centre grey circle, with individual life-cycle stages depicted as grey rectangles) lies at its core. Each of the individual life-cycle stages can then be further characterised based on technical, social and systemic attributes (green, blue and salmon pink circles, respectively). While focus was placed on consumption and waste management stages, adopting an interdisciplinary approach also enabled to tap into additional life-cycle stages. For example, investigating shelf-life extension properties of BBP food packaging (**Chapter 8**) provided insights into the manufacturing stage (i.e. by providing recommendations for the design and manufacturing of novel BBP packaging materials based on experimental findings).

Chapter 9 brings findings from the individual chapters into a single, unified discussion on the challenges and opportunities associated with the development, consumption and disposal of BBP food packaging in the context of organic waste management. By considering multiple life-cycle stages (see above) under both social (**Chapters 5, 6 & 7**) and technological (**Chapters 4 & 8**) angles, it has weighed evidence from one discipline with that of another (e.g. theoretical biodegradability with on-the-ground practices, functionality vs EoL design, intended vs actual disposal intentions) to provide a novel, more nuanced and, arguably, more representative view of the system, while developing tools for interdisciplinary science (e.g. consumer behaviour chains, systems framework for material flow).

Chapter 3 – Literature review

The following sections provide the 'backstory' to BBPs with some historical background on the rise of plastics in modern society and the plastic management crisis that ensued, before moving onto the emergence of circular bioeconomy frameworks in the context of global warming and resulting climate change. In this light, the promotion and development of BBPs and their role in circular organic waste management practices are reviewed. Finally, the importance of adopting a systems-thinking approach and addressing consumer behaviour when investigating barriers and opportunities in the food-energy-waste nexus are presented.

1. Plastics: a brief history

1.1. Plastics and the 20th century – a love story

From cars to food packaging, from medicine to electronics, plastics have brought significant changes to daily life, revolutionising modern society. The development of plastics started in the late 1850s while seeking a suitable alternative to ivory, amid growing concerns around the stark decline in elephant population (Freinkel, 2011). Cellulose nitrate, a material derived from cotton fibres dissolved in nitric acid, first patented by Alexander Parkes in 1862 as *Parkesine*, became the world's first man-made plastic. In 1869, John Wesley Hyatt developed *Celluloid*, a solid, stable version of nitrocellulose, which became the first commercially successful man-made plastic. It wasn't until 1907 that the first fully synthetic plastic – as opposed to plastic derived from biological resources, such as natural fibres –, *Bakelite*, a polymer of formaldehyde and phenol, was invented by Leo Baekeland. As with *Celluloid*, this new polymer was developed to replace a scarce natural material, shellac, a resin secreted by the female lac beetle with excellent electrical insulation properties. By substituting bio-derived materials that were either in short supply or expensive, plastics contributed to the democratisation of consumer goods and culture in an increasingly consumption-oriented society (Freinkel, 2011).

Bakelite was a steppingstone for the emerging petrochemical industry and catapulted the 20th century into an era of synthetic plastics, driven by the desire to turn by-products from crude oil and natural gas processing into higher-value products (Geyer, 2020). Material shortages during World War II fuelled the development of modern plastics and the unprecedented economic growth of the post-war period contributed to the mass production of plastic fast-moving consumer goods (FMCGs) (Geyer, 2020). Polyethylene (PE), discovered in 1933, was first used in military applications as cable coating and radar insulator, before its application in FMCGs, including plastic bags and Tupperware® (Freinkel, 2011). PE is now the most common plastic in the world (Geyer, Jambeck & Law, 2017). Other iconic products/brands and their corresponding plastics, such as nylon 66 (a polyamide) and Teflon® (polytetra-fluoroethylene, or PTFE), further anchored plastic products as highly desirable FMCGs.

The proliferation of affordable FMCGs in an increasingly linear economic model framed around convenience culminated in the commercialisation of polyethylene terephthalate (PET) bottles in 1973. Initially developed for the carbonated beverages sector, PET became a popular container for non-carbonated soft drinks and still beverages, with as many as 500 billion PET bottles sold around the world every year (Science Museum, 2019).

Today, plastics form a large and growing family of synthetic polymers and an even larger set of plasticisers (Geyer, 2020), chemical additives added during the manufacturing process that help acquire the optimal polymer properties sought for a specific application. The most widely used plastics, which include PE, PET, polypropylene (PP), polystyrene (PS), polyvinyl chloride (PVC) and polyamides (PAs), can all be re-melted at higher temperatures and re-moulded into a new shape, a key property known as thermoplasticity. Because of this intrinsic property, all thermoplastics are, in theory, recyclable (Geyer, 2020). In contrast, thermosets, including polyurethane (PUR), epoxy, silicone and phenolic resins form an irreversible three-dimensional polymer matrix upon synthesis; they cannot be subsequently re-melted and are thus not easily recyclable (Geyer, 2020).

The sharp growth in plastic production is unprecedented (Geyer, Jambeck & Law, 2017). Reflective of modern linear production and consumption patterns and epitomised by the PET bottle, plastics' largest sector is packaging (PlasticsEurope, 2021). In 2020, demand for packaging reached 44.8% of the 367 Mt of plastics produced globally, of which 19.9 Mt were destined to the European market (Law & Narayan, 2022; PlasticsEurope, 2021) (**Figure 1**).

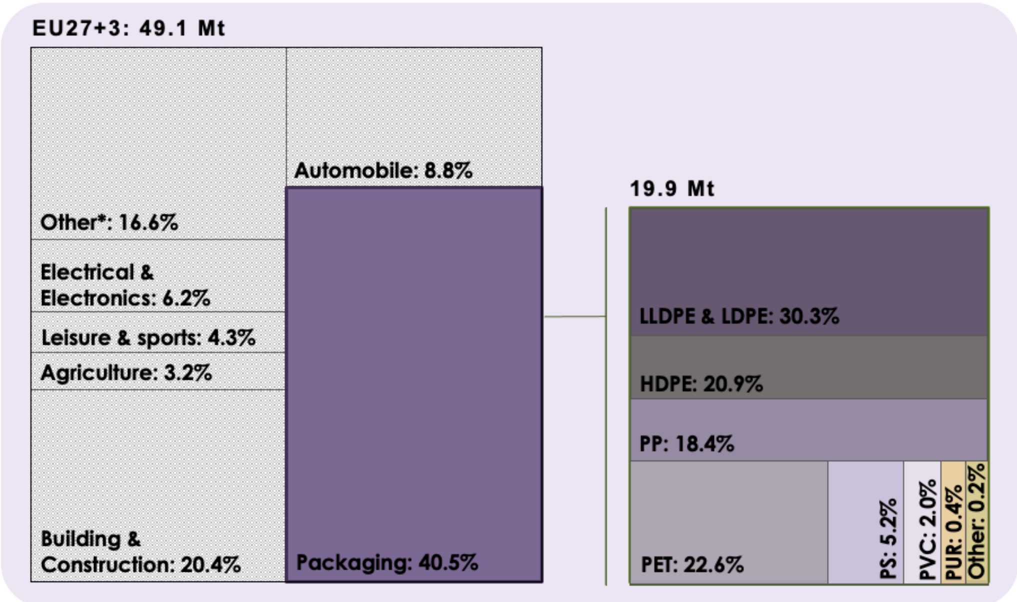


Figure 1 | Share of European plastic production per application sector and polymer type. Data covers the 27 European Union Member States, Norway, Switzerland and the United Kingdom. Other application sectors (*) include plastics used in furniture, medical applications, machinery and mechanical engineering. (L)LDPE: (linear) low-density polyethylene, HDPE: high-density polyethylene; PP: polypropylene; PET: polyethylene terephthalate; PS: polystyrene; PVC: polyvinyl chloride; PUR: polyurethane. Source: Geyer, Jambeck & Law, 2017; PlasticsEurope, 2021.

The most commonly used plastics in packaging are PE, PET, PP and PS, which together represent over 60% of the global plastics demand (Geyer, Jambeck & Law, 2017; Narancic & O'Connor, 2019) (**Figure 1**). Plastic packaging has found a particularly useful application in the food and beverage industry (**Figure 2**), especially plastic films and thermoformed plastics – plastics heated to moderately high temperatures and moulded into a specific shape –, engendering the development of complex food supply chains.

Figure 2 | Most common plastic types in food packaging and other fast-moving consumer goods. Each polymer type is associated with its corresponding resin identification number. Blue: commonly recycled; grey: recyclable at certain local collection points; black: currently not recyclable. LDPE: low-density polyethylene, HDPE: high-density polyethylene; PP: polypropylene; PET: polyethylene terephthalate; PS: polystyrene; PVC: polyvinyl chloride. Adapted from Hocevar, 2020.

From the large array of food produce consumers have gained access to throughout seasons and across geographic locations, to the way food is processed, transported and displayed, purchasing food has never been more convenient (Ellen MacArthur Foundation, 2019). With over half of the world's population now living in cities, with projections reaching 68% by 2050 (United Nations Department of Economic and Social Affairs, 2018), effective packaging design ensures food quality – and quantity – is preserved from production to consumption, while minimising waste (Verghese et al., 2013). In 2018, over 1.13 trillion items of packaging – most of them plastic-based – were used for food and drinks in the EU alone (Fuhr, Matthew & Schächtele, 2019). Sales of packaged produce and food deliveries have skyrocketed during the COVID-19 pandemic (Vanapalli et al., 2021), highlighting the role plastic packaging has played in ensuring food hygiene standards across the food supply chain. However, the continuous growth of the single-use packaging has brought new challenges for the waste management sector and there is now an imperative to address the environmental, social and economic conundrum it has brought.

1.2. Plastic pollution – the plot thickens

The very properties that have made plastics the most ubiquitous materials they are today – their durability, hydrophobicity and affordability – have also contributed to poorly reversible plastic pollution (MacLeod et al., 2021). The global weight of accumulated plastics, including those currently in use and plastic waste, is now greater than that of all terrestrial and marine fauna combined, with 8,000 Mt estimated to have been produced to date (Elhacham et al., 2020; Geyer, Jambeck & Law, 2017). In that same time window, 6,300 Mt of plastic waste have been generated (Geyer, Jambeck & Law, 2017), highlighting the linearity of the plastics value chain. Further fragmentation into microplastics and nanoplastics, plastic fragments invisible to the human eye induced by weathering processes[1], poses an additional threat to both human and ecosystem health (MacLeod et al., 2021) and threatens the integrity of planetary boundaries (Persson et al., 2022), a safe operating space for both society and the planet (Rockström et al., 2009).

Although the majority of plastics are theoretically recyclable, in practice the potential for recycling plastic waste remains largely unexploited (Narancic & O'Connor, 2019). How plastic waste is handled varies significantly from country to country, but globally, recycling as an EoL option is still uncommon (d'Ambrières, 2019), including for plastic packaging (**Figure 3**). Despite a post-consumer plastic recycling rate of 35% in the EU (PlasticsEurope, 2021), as little as 9% of cumulative plastic waste has been recycled globally, of which only 10% recycled more than once (Geyer, Jambeck & Law, 2017). In contrast, 79% have accumulated in landfills or in the natural environment (Geyer, Jambeck & Law, 2017).

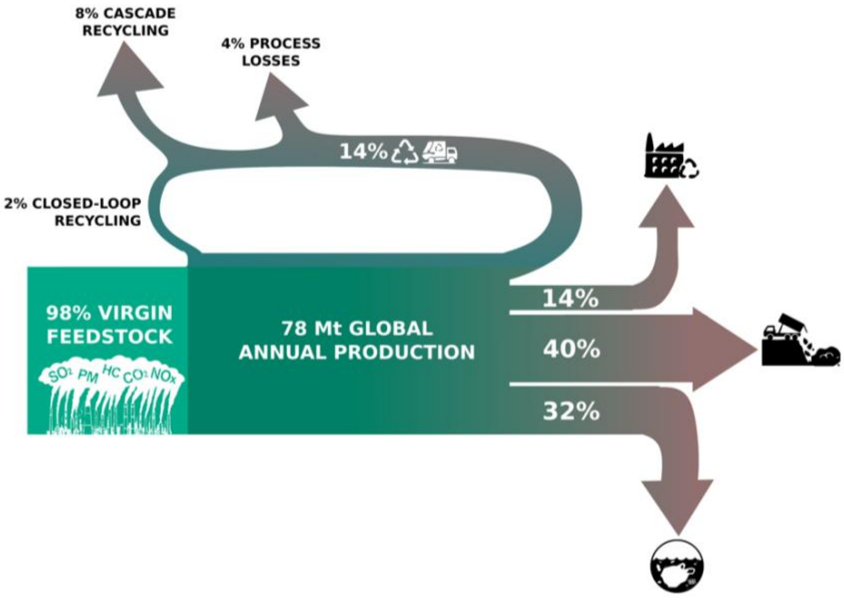


Figure 3 | The linear flow of plastic packaging. In 2016, only 14% of the 78 Mt plastic packaging produced globally were collected for recycling, 14% were incinerated (with or without energy recovery), 40% were landfilled and 32% leaked into the environment. Plastic packaging accounted for 40% (141.2 Mt) of the 353 Mt of global plastic waste generation in 2019 (OECD, 2022a). Cascade recycling: recycling into lower-value applications; closed-loop recycling: recycling into similar quality applications. Adapted from World Economic Forum, Ellen MacArthur Foundation & McKinsey, 2016.

[1] Microplastics may also originate directly from commercial products (e.g. microbeads in personal care products, synthetic textiles), which are referred to as primary microplastics, while secondary microplastics result from the breakdown of larger plastics from weathering.

The biggest sources of plastic pollution originate from regions with insufficient waste management infrastructure (Jambeck et al., 2015), particularly those undergoing steep economic development and societal change, although the potential contribution of improper waste management (accidental or intentional) from mature economies with adequate and sufficient infrastructure is non-trivial (Law & Narayan, 2022). Mismanaged plastic waste contaminates terrestrial, freshwater and marine ecosystems (Law & Narayan, 2022), the seafloor being the primary site for the accumulation of plastic debris (MacLeod et al., 2021) and exhibiting some of the highest concentrations of microplastics (Tekman et al., 2020).

Despite the promise of mechanical recycling – whereby plastic waste is recovered via mechanical processing to be used in the manufacturing of new products – to curb plastic pollution, recycled plastics currently account for less than 10% of total demand for plastics worldwide (d'Ambrières, 2019), their growth compounded largely by the lack of economic incentives (Voulvoulis & Kirkman, 2019). Though there are compelling arguments for the replacement of virgin plastics by recycled ones, the means of achieving this shift are not always financially competitive (Voulvoulis & Kirkman, 2019), especially without a regulatory framework (Geyer, 2020). If resources are cheap – as is the case for naphtha, the oil fraction from which most conventional, fossil-based plastics are synthesised from – the incentive to develop single-use products is high (Voulvoulis & Kirkman, 2019). This in turns makes it challenging to steer away from current consumption trends and to turn plastic materials into new valuable products.

A range of regulatory and economic policy instruments, including taxes on virgin plastics, plastic waste collection and recycling targets, bans on certain plastic items and restrictions on plastic waste exports have been instituted by both state governments and international bodies (**Table 1**). Collection targets alone may not truly reflect recovery of valuable materials, as collecting plastics for recycling does not guarantee their reuse (Voulvoulis & Kirkman, 2019). They also often conceal the reality of plastic waste exports from Western nations to the Global South (Bergmann et al., 2022). For example, Germany, the world's recycling leader, is also among the top waste exporters (Fuhr et al., 2019). Nevertheless, extra-EU plastic waste exports have dropped by 16% between 2018 and 2020 (PlasticsEurope, 2021), following China's and other Southeast Asian nations' waste import bans on several types of waste, including low-grade plastics, which has compelled developed countries to manage their plastic waste domestically (Wang et al., 2020).

Table 1 | Policy instruments addressing plastic production, use and waste management.

Policy Pillar	Function	Policy Instrument	Example of Instrument	Corresponding Governing Body
Harmonise baseline reporting	Measure plastic waste flows to map sources of emissions and environmental impacts	Country-level data collection and evaluation tool	National Assessment and Modelling (NAM)	Global Plastic Action Partnership (GPAP)
			National Guidance for Plastic Pollution Hotspotting and Shaping Action	United Nations Environment Programme (UNEP)
Restrain supply of virgin polymers	Disincentivise the production and use of plastics	Plastic packaging tax	£200/tonne tax on plastic packaging with less than 30% recycled plastic content	United Kingdom (UK)
		Production cap	25% reduction in sales of plastic packaging by 2032	California
Enhance circularity	Enhance the durability of plastic products and maximise their value	Eco-design for durability and repair	Eco-design for Sustainable Products Regulation (under revision)	European Union (EU)
		Tax breaks on repair/reuse	15p reusable cup levy	Imperial College London
			52% reduction in value-added tax (VAT) on certain repairable consumer goods	Sweden
		Landfill bans/ reduction targets	≤10% of municipal waste landfilled by 2035	EU
Enhance recycling	Where reduction and reuse are not possible, promote and develop routes for plastic recovery and recycling	Recycled content target	30% recycled content across plastic packaging	UK Plastics Pact
		Extended Produced Responsibility (EPR) schemes	EPR Act of 2022	Philippines
		Recycling rate target	50% of plastic packaging recycled by 2025	EU
		Direct fiscal contribution	Contribution of €0.80/kg tax for non-recycled plastic packaging waste produced by each Member State	EU
Close leakage pathways	Decrease and eliminate mismanaged plastic waste	Bans on plastic items	Plastic bag ban	Kenya
		Waste exports	Ban on 24 types of solid waste (including low-grade plastic waste)	China
			Basel Convention on the Control of Transboundary Movements of Hazardous Wastes (including certain types of plastic waste)	UNEP

In contrast with pre-consumer plastic waste, which is generated during manufacturing processes and generally easy to recycle both technically and economically, post-consumer plastic waste provides additional challenges to mechanical recycling (Geyer, 2020). Whilst straightforward conceptually, sorting target polymers from non-target materials represents a major challenge in an increasingly diverse and complex plastic research and development space (Law & Narayan, 2022), often with little consideration for EoL design. In addition, plastic waste needs to be separated by polymer type to maintain the value of the recycled polymer (Geyer, 2020). Moreover, recycled plastics exhibit inferior technical properties compared to virgin plastics, due to polymer degradation throughout their life-cycle and during reprocessing (Ragaert, Delva & van Geem, 2017) and thus tend to be used in lower-value applications, an issue known as downgrading, downcycling, or cascade recycling (Law & Narayan, 2022).

Recycling can only contribute towards limiting further plastics entering the natural environment if it also inhibits the production at the front-end – otherwise disposal is simply postponed (Zink & Geyer, 2018). Boosting the recycling sector and the market for recycled plastics does not guarantee a reduction in primary consumption (Law & Narayan, 2022), and in some cases may lead to unintentional rebound effects by either failing to compete effectively with primary production or by lowering prices and therefore increasing consumption, thereby undermining potential environmental benefits gained from secondary production (Zink & Geyer, 2017). Furthermore, the entire life-cycle of plastics accounted for 4.5% of global greenhouse gas (GHG) emissions in 2015 and 6% of global coal electricity was used for plastics production that same year (Cabernard et al., 2022). The growing production of plastics, set to double by 2045 (Geyer, 2020), and the inevitable release of plastics into natural ecosystems, which could triple in the next decades to reach 265 Mt per year by 2060 (Lebreton & Andrady, 2019), will exacerbate these problems (MacLeod et al., 2021).

2. The emergence of circular bioeconomy frameworks

2.1. Global warming, fossil fuels and plastics: a growing paradox

Given the unprecedented rates of global warming, ecosystem collapse and resource depletion and the threat they all pose, there is an imperative to decarbonise the economy (IRENA, 2019; United Nations, 2019). Limiting global warming to 1.5°C above pre-industrial levels will only be feasible if global GHG emissions peak within the current decade and then start to decline rapidly, halving by 2030 and reaching net zero by 2050 (IPCC, 2018). In many cases, the tools for delivering a carbon neutral economy already exist and include shifting away from fossil fuels to renewable energy sources, cutting emissions from industrial, agricultural and shipping processes, and increasing energy efficiency (Shah et al., 2013).

While an increasing number of nations are committing to achieve net zero GHG emissions over the coming decades (IEA, 2021), the demand for plastics continues, paradoxically, to grow (IEA, 2018) (**Figure 4**). As traditional applications for crude oil – mostly vehicle fuels – are beginning to experience significant disruption from the electrification of the transport sector, petrochemicals, of which plastics represent the largest segment (45%), are expected to exert a major influence on future economic investments by the fossil fuel industry (Bauer & Fontenit, 2021). Petrochemicals currently account for 12% of oil use worldwide and their projected growth is predicted to drive half of global oil demand by 2050 (IEA, 2018), led by a growing demand in emerging economies in Africa and Asia (OECD, 2022b). Thus, calls have been made for a global cap on virgin plastic production (Bergmann et al., 2022), guided by alternative economic frameworks to eliminate, rather than simply reduce, plastic pollution.

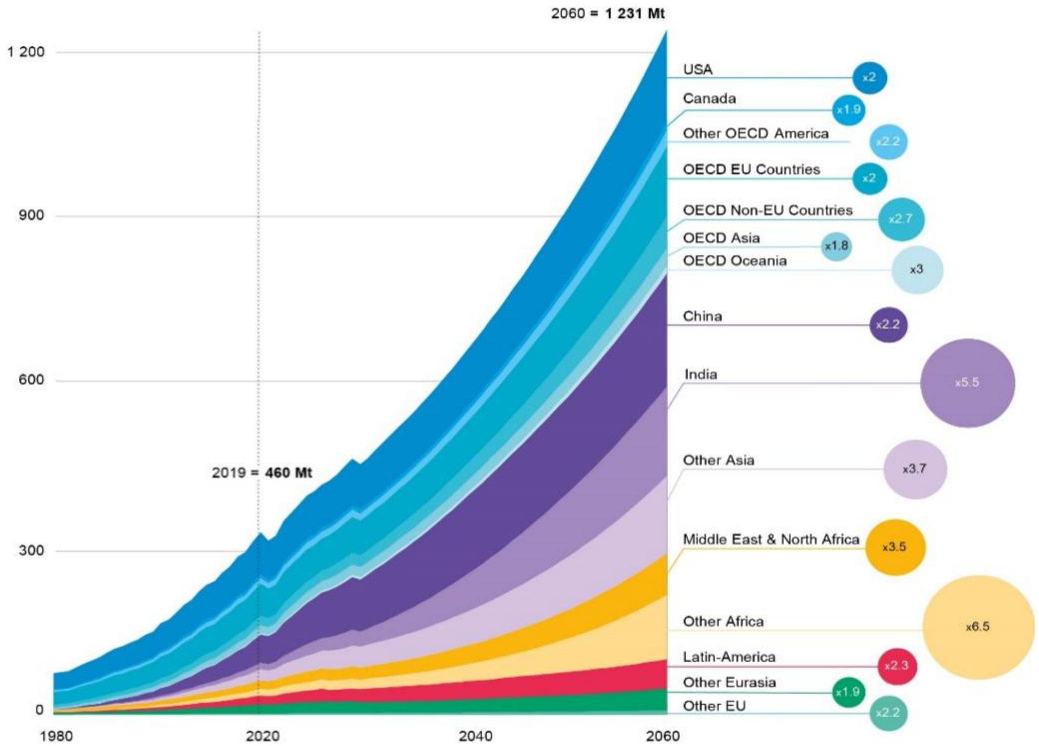


Figure 4 | Historical and projected plastic use globally. Reproduced from OECD, 2022b.

2.2. Circular economy: closing the loop

The global plastic pollution crisis represents one of the starkest examples of the pervasive take-make-use-dispose mindset. Growing public outcry and the gradual recognition that the current linear economic model is pushing natural ecosystems to the brink of collapse (Rockström et al., 2009) have led to calls for a systemic rethink of how resources, materials and products are managed in society. Looking beyond the current extractive economic model, a circular economy aims to redefine growth (Ellen MacArthur Foundation, 2015), decoupling economic prosperity from the mere consumption of materials and products (Jackson, 2009). Both restorative and regenerative by design, framing waste out of the system is one of its core principles, aiming to ‘close the loop’ (**Figure 5**).

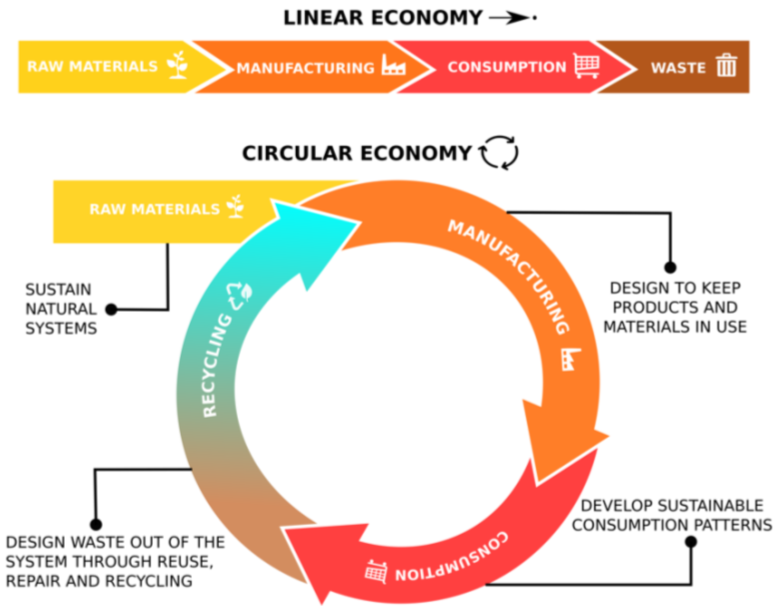


Figure 5 | Schematics for linear and circular economy frameworks.
Adapted from Ellen MacArthur Foundation, 2015.

A central strategy to achieve closed loops lies in the concept of the waste hierarchy, where the value of resources, materials and products is maintained in the economy for as long as possible. The waste hierarchy ranks waste management options according to their environmental benefits (**Figure 6**), with the aim to extract the maximum value from products while minimising waste generation. The inclusion of a hierarchical decision-making tool is paramount for the creation of a truly circular system, in that it ensures waste reduction goals are incorporated into product design and manufacturing by applying a life-cycle approach. Criticisms have been made regarding a systematic focus on lower stages of the waste hierarchy, particularly around recycling and extracting energy from waste, rather than directing efforts upstream, where environmental benefits are highest (Simon, 2019).

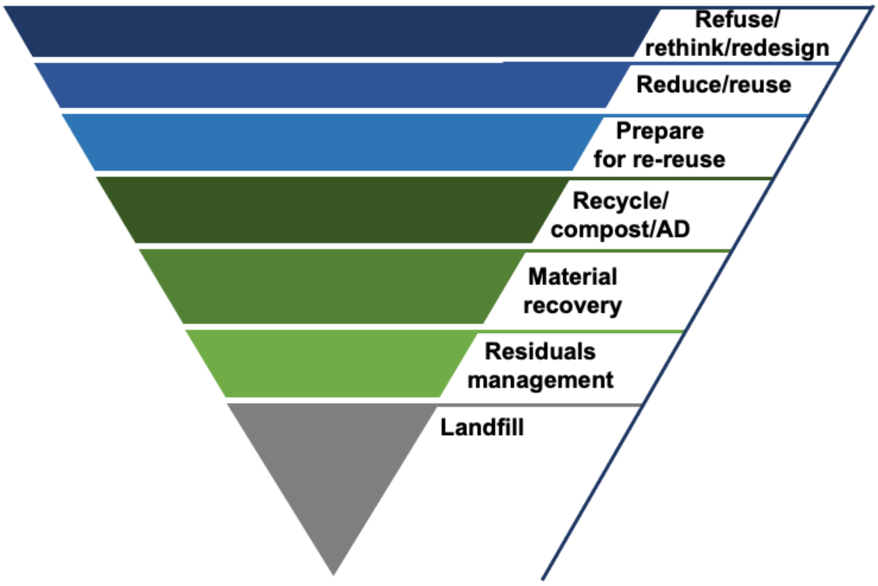


Figure 6 | The waste hierarchy. The original waste hierarchy pioneered by the European Union was subject to redefinition to a 'Zero Waste' hierarchy, which focuses on value and energy preservation. Adapted from Simon, 2019.

Conceptually, a circular economy model provides social, economic and environmental gains. The focus on increased resource efficiency benefits industry and businesses while ensuring the provision of resources for current and future generations (European Commission, 2015). It also creates cross-sectoral synergies such as agriculture and food manufacturing, allowing nutrients to flow back to the soil, thus forming a truly circular system. Together, these measures contribute towards meeting the UN Sustainable Development Goals (SDGs), a set of intergovernmental, multi-dimensional objectives to build a sustainable, just and peaceful society while safeguarding the Earth's diverse ecosystems (United Nations, 2015).

As a pioneer in circular economy, the Ellen MacArthur Foundation has advocated for a circular economy for plastics. In 2018, in collaboration with the UN Environment Programme (UNEP), it launched the New Plastics Economy Global Commitment. With over 500 signatories representing 20% of all plastic packaging produced globally – a laudable, yet relatively small number –, it aims to unite businesses, governments and other organisations to eliminate, innovate and circulate plastics (Ellen MacArthur Foundation, 2022). In that same year, the EU published its Circular Economy Package, new legislation strengthening the waste hierarchy by introducing new measures for waste prevention, restricting the amount of municipal waste sent to landfill, as well as recycling targets, including for plastic packaging (**Table 2**), and linking the circular economy to the bioeconomy.

Table 2 | Targets and timelines under the European Union's Circular Economy Package. Separately collected food waste will have to be either recycled or prepared for reuse. Numbers are in percentage (%) of total waste generated per category. Reproduced from Kakadellis, Lee & Harris, 2022.

Material Targeted	By 2023	By 2025	By 2030	By 2035
Plastic packaging recycled	-	50%	55%	-
Municipal waste recycled	-	55%	60%	65%
Municipal waste landfilled	-	-	-	≤10%
Household food waste	Separate collection and landfill ban	-	-	-

2.3. Bioeconomy: closing the (fossil) tap

Partially overlapping with the concept of circular economy, the bioeconomy at its fundamental level refers to the 'biologisation' of industrial value creation (Carus, 2017) and focuses on the substitution of fossil fuels with renewable alternatives, in particular the shift from fossil to biogenic carbon sources from biomass (Tan & Lamers, 2021). The term 'bioeconomy' is characterised by a certain level of confusion, and definitions and interpretations abound (Aguilar & Twardowski, 2022; Giampietro, 2019; Tan & Lamers, 2021). Vivien et al. (2019) outlined and compared three main interpretations of the bioeconomy: a historical entropic narrative of 'bio-economics' developed by Georgescu-Roegen (1975) that considers the economy within the limits of the biosphere, a science- or knowledge-based bioeconomy, heavily reliant on biotechnology as driver of industrial innovation, and a biomass-oriented narrative based on the concept of biorefining, whereby multiple value-added products are obtained from the valorisation of biomass (**Table 3**). The science-based narrative is prevalent in the US, which focuses more narrowly on synthetic biology (D'Amato et al., 2017), while the biomass-based narrative dominates the bioeconomy discourse, especially within the EU (D'Amato et al., 2017; Vivien et al., 2019).

Table 3 | Summary of three bioeconomy narratives. Adapted from Vivien et al., 2019.

Bioeconomy Narrative	Definition	Identified Tensions
Biophysics/entropy-based	An economy that is compatible with (and constrained by) the biophysical limits of the biosphere	Directly challenges the sustainability of the other two bioeconomy narratives
Science-based	A science- and knowledge-based economy driven by industrial biotechnology	Highly technocratic society; social resistance, e.g. to genetically modified organisms (GMOs)
Biomass-based	An economy focused on the use of biomass to replace fossil resources, based on the concept of biorefineries	Increased pressure on resources and land

The latter two narratives (**Table 3**) have been criticised for their semantic and conceptual 'hijacking' of its original meaning and their reliance on the neoclassical economics adage that technological innovation can overcome external ecological constraints (Giampietro, 2019; Vivien et al., 2019). Nonetheless, all three interpretations concur on framing the bioeconomy as the utilisation of renewable biological resources for the production of food and energy and the achievement of economic, environmental and social benefits (Tan & Lamers, 2021). It goes beyond the flow of biomass itself, with an emphasis on the development of new chemical building blocks and processing routes and the creation of synergies in material and energy flows across industrial systems (Venkatesh, 2022), a principle known as industrial ecology. This includes innovation in agriculture and forestry (precision farming, genome editing), new processing pathways with lower toxicities, chemicals and materials with new properties and functionalities as well as more nature-compatible, healthy bio-based products (Carus, 2017).

While a bioeconomy can, in principle, be easily developed – and to some extent already has been –, a sustainable bioeconomy will require more than the replacement of fossil fuels with renewable alternatives from the biosphere (Tan & Lamers, 2021). Resource scarcity and competition still apply to biological raw materials (Venkatesh, 2022), arguably even more so in an economy constrained by biophysical processes (Giampietro, 2019). For a bioeconomy to truly support the transition towards a sustainable society, circularity must be incorporated into the design of the supply chains, conversion processes and products (Venkatesh, 2022).

2.4. Circular bioeconomy: biogenic carbon cycles and biowaste revalorisation

Recognising the need for circularisation of a bioeconomy framework, the EU Bioeconomy Strategy, first published in 2012, was updated in 2018 to include an action plan towards a "*sustainable, circular bioeconomy*" (European Commission, 2018, p.10). Although critics of modern bioeconomy and circular economy frameworks have cautioned against the use of a circular bioeconomy, which in their view represents an oxymoron of two incompatible goals (Giampietro, 2019, Vivien et al., 2019) and fails to question economic growth (D'Amato et al., 2017), others have highlighted the potential synergistic effect from combining both concepts, resulting in a more sustainable framework overall (Tan & Lamers, 2021; Yadav et al., 2021).

While both the bioeconomy and circular economy aim to achieve a sustainable, resource-efficient society in line with decarbonisation goals, a circular bioeconomy can be interpreted as an overarching framework that brings together a desirable outcome (circular economy) and the means to achieve it (bioeconomy) in a single package, combining the *what* with the *how* of a sustainable society (Giampietro, 2019). Alternatively, a circular bioeconomy can also be defined as an efficient use and management of biological, renewable resources through the integration of circular economy principles into the bioeconomy (D'Amato et al., 2017).

Multi-faceted and still evolving, the circular bioeconomy aims to create a more sustainable future, with closed biogenic carbon loops at its core (Tan & Lamers, 2021). Since a life-cycle approach is not necessarily inherent to the bioeconomy, incorporating circular principles into biological and biotechnological systems will help mitigate unintended consequences of increased biomass demand and ensure the consideration of reuse, repair, recycle and waste prevention in resource use and product design (Yadav et al., 2021). In addition, some have stressed the importance of bio-circular metrics, including new socio-economic indicators (D'Adamo, Falcone & Morone, 2020; Kardung et al., 2021), for identifying and prioritising pathways and monitoring progress on the circular bioeconomy. Developing comprehensive terminology and a framework of concrete strategies for closed-loop product design and business models will further support the move to a circular bioeconomy (Bocken et al., 2016).

Aiming to ensure that the bioeconomy truly delivers an efficient use and sustainable management of biomass, Muscat et al. (2021) propose a set of ecological principles for a circular bioeconomy: (1) protecting and regenerating ecosystems health, (2) prioritising biomass flows for basic human needs, (3) avoiding or at the very least minimising waste, (4) utilising and recycling unavoidable by-products or waste, and (5) shifting to renewable energy sources while minimising overall energy use. Together, these principles represent an opportunity to address interconnected societal challenges (Yadav et al., 2021) and call for a transformation of the current economic model (Muscat et al., 2021).

A cascading use of biomass and waste revalorisation lie at the heart of this social and economic transformation (D'Amato et al., 2017; Yadav et al., 2021). Given the significant amount of solid waste generation of biological origin (i.e. organic waste, or biowaste), resulting from a linear economic model, and building upon cycles naturally present in the bioeconomy, e.g. nutrient cycles, revalorising organic waste and bio-based by-products would strongly contribute towards a circular bioeconomy (Yadav et al., 2021). At the interface between organic waste management, renewable energy production and sustainable agriculture, AD is an important pillar of a circular bioeconomy and represents a unique opportunity to kill two birds with one stone in the food-energy-waste nexus (Kakadellis, Lee & Harris, 2022).

3. Biowaste: from rubbish to resource

3.1. Organic waste management: an untapped potential

Valorising biowaste is a key strategy for ensuring the efficiency and sustainable use of biomass, while meeting renewable energy targets and contributing towards the replenishment of nutrient and organic matter in increasingly depleted agricultural soils (European Commission, 2018). The OFMSW, which includes food and garden waste from households and household-like sources (e.g. institutional canteens, food markets, and businesses), has been increasingly recognised as resource rather than as waste, with the potential to move from linear to circular and energy-positive waste management strategies (Kakadellis, Lee & Harris, 2022).

Food wastage undermines the sustainability and efficiency of current food supply chains, both in the developed and developing world. Food wastage is broadly defined as food loss, generated at early stages of the food supply chain (e.g. in-field and harvest loss, spoilage, transport) and FW, which occurs at the retail and consumer level and is generally linked to behavioural practices (FAO, 2011). More recently, the term 'food waste' was expanded to cover "*food intended for human consumption, either in edible or inedible status, removed from the production or supply chain to be discarded, including at primary production, processing, manufacturing, transportation, storage, retail and consumer levels, with the exception of primary production losses*" (European Parliament, 2017, p.6), though nuances in its definition remain, e.g. surrounding inedible food parts and what those might be (Patel et al., 2021).

Globally, one third of all food produced for human consumption is lost or wasted, corresponding to 1,300 Mt a year (FAO, 2011). If considered a country, FW would represent the third largest GHG emitter in the world (FAO, 2019), its associated emissions (**Figure 7**) outstripped only by the US and China (Crippa et al., 2021). Food wastage has environmental, social and economic impacts (Girotto, Alibardi & Cossu, 2015), and represents a waste of resources used in food production including land, water and energy use (FAO, 2011).

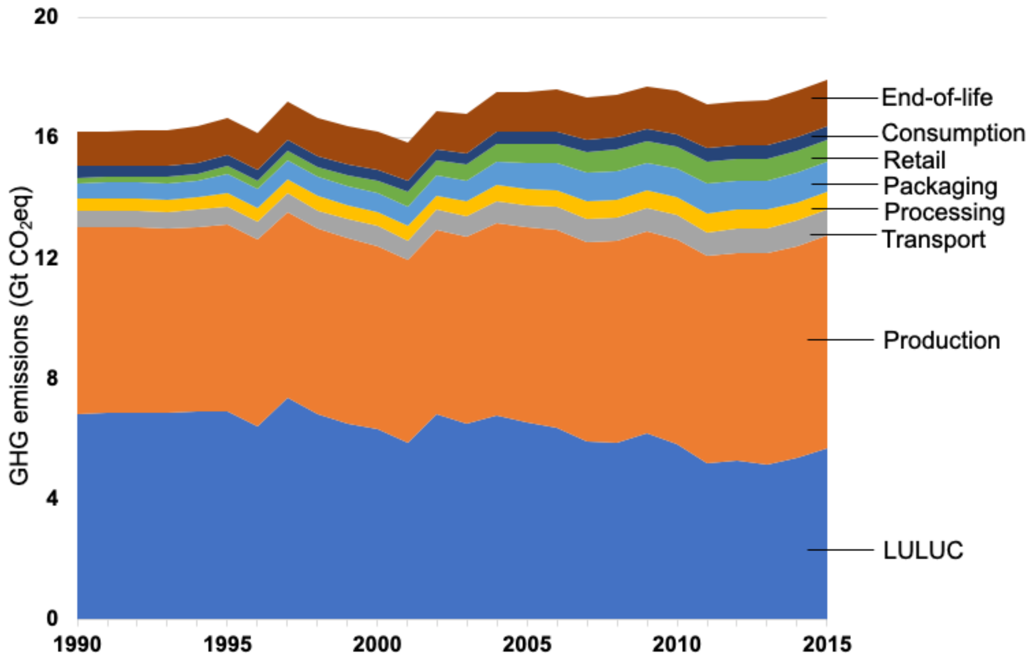


Figure 7 | Greenhouse gas emission trends of the food sector. The following life-cycle stages were identified: land use and land-use change (LULUC); primary production of food commodities; food processing; food distribution including packaging, transport and retail; food consumption including domestic food preparation; and end-of-life of food, including food residues management and management of non-food residues used in previous food-system stages. Source: Crippa et al., 2021.

Producing food that will not be consumed not only leads to unnecessary GHG emissions, it also represents a significant loss of economic value of the food produced (FAO, 2011). With a global population projected to reach nine billion people by 2050, demand for abundant and nutritious food is peaking (Searchinger et al., 2018). This is further compounded by rising consumption patterns, driven by increased wealth and higher living standards. Climate change, competing land uses, soil erosion and diminishing supplies of clean water are already threatening food production worldwide (Verghese et al., 2014).

Considering that the production, distribution and consumption of food produce is a major source of GHG emissions, any measure to reduce or divert FW, even to a small extent, may have a significant effect on the overall environmental footprint of food supply chains (Williams & Wikström, 2011). Thus, preventing FW should remain a priority for sustainable resource management (reflecting the relative contributions in **Figure 7**). Yet, as FW is still generated at high volumes and is in some cases inevitable, it is worth dedicating efforts to utilise this waste as a valuable resource to generating circular products and value chains, thereby demonstrating a repurposing at the EoL and maximising sustainability benefits (Iacovidou & Voulvoulis, 2018).

Across the EU, between 118 and 138 Mt of biowaste are generated annually (EEA, 2020), 100 Mt of which are OFMSW (FUSIONS, 2016). Currently, only up to 25% of this biowaste are captured and recycled into compost and digestate through separate organic waste collections (EEA, 2020). The majority is captured through residual waste, which is either landfilled or incinerated (EEA, 2020), both streams effectively acting as carbon and nutrient lock-in, keeping them from flowing back to the soil as bio-available components. The Landfill Directive (1999/31/EC) addressed this partly, obliging EU Member States to reduce the amount of OFMSW they landfill to 35% of 1995 levels by 2016. The Circular Economy Package (**Table 2**) further addressed the untapped potential of OFMSW, introducing mandatory separate FW collections from households and businesses from 2023 across the EU (European Commission, 2020).

At a global scale, major cities across the world have also introduced separate FW collections, including Milan, Copenhagen, Paris, New York, San Francisco, Auckland, Cajicá and Seoul (WBA, 2018). Though these decisions have taken place at a city level, they are often enabled by a supportive national legislative framework (WBA, 2018). Remarkably, following a ban on direct landfilling of FW in 2005, FW recycling rates in South Korea rose from a mere 2% to over 90% within 20 years (Ju et al., 2016). A number of countries have since then introduced policies aimed at the separation and treatment of FW, including Japan, Malaysia, Thailand and China (Awasthi et al., 2020). In light of legislation targeting the capture and recovery of FW, AD provides a unique opportunity to move from a simple waste management strategy to a more holistic, energy-producing system.

3.2. Anaerobic digestion as food waste recycling strategy

Waste management strategies can be broadly split into separately collected FW treatment and technologies that treat both organic and inorganic fractions from unsegregated residual waste collections. Though treating a mixed waste stream collectively, such as through gasification, incineration with or without energy recovery, landfill with or without gas collection, pyrolysis and mechanical biological treatment are associated with lower operational costs and remove the need for separate collection systems, their circularity is limited (WBA, 2018). Instead, source-separated FW treatment technologies offer a number of environmental benefits that those treating mixed wastes do not, including maximising energy recovery, fertiliser production and improved soil health by recirculating organic matter (WBA, 2018). The separate treatment of FW varies from one waste collection scheme to another and includes in-vessel composting, windrow composting and AD (WBA, 2018), each presenting advantages and limitations (**Table 4**). AD is increasingly recognised as the most adequate and sustainable technology to tackle the significant amount of FW generated each year, due to its environmental, social and economic advantages (ADBA, 2020; WBA, 2018, WRAP, 2019).

Table 4 | Characteristics of waste treatment strategies for separately collected organic waste. Adapted from Kakadellis, Lee & Harris, 2022.

Technology		Process Description	Advantages	Disadvantages
Anaerobic digestion (AD)		Degradation of organic waste by micro-organisms in the absence of O_2 in a closed chamber	- Allows for energy production alongside nutrient and organic matter recovery - Products are substitutes for fossil-based natural gas and synthetic fertilisers and can be sold for agricultural purposes through the PAS 110 quality standard scheme	- Capital and operational costs can be prohibitive - Highly sensitive process - CH_4 content of biogas can be low for some substrates - Possible restrictions on digestate application timings - Digestate storage (e.g. lagoons) can be costly - Poor plant design/operation can lead to CH_4 leakage - May spread plastics to soil
Composting	In-vessel (industrial) composting (IVC)	Degradation of organic waste by micro-organisms in the presence of O_2 in a silo or concrete-lined chamber	- High organic matter compost can be sold through PAS 100 - Allows FW to be collected alongside garden waste - Simple, predictable and naturally occurring process - Relatively cheap	- Does not recover energy - Produces more CO_2 than AD (relatively little CH_4 produced) - Leachate must be treated - May spread plastics to soil
	Windrow composting	Degradation of organic waste by micro-organisms in the presence of O_2 in windrow (i.e. heaps laid out to dry outdoors)		- See disadvantages for IVC - Cannot be used in some countries (e.g. UK) to treat wastes containing catering and animal waste under the Animal By-Products Regulation

The term ‘anaerobic digestion’ refers to the degradation and stabilisation of organic waste in the absence of dioxygen (O_2). It is a natural biological process, dependent on an ensemble of microorganisms, the microbiota, which process the organic matter. The resulting products consist of biogas (composed of 50-70% methane (CH_4)), 30-50% carbon dioxide (CO_2) and traces of other species, depending on the substrate), a valuable source of renewable energy, and digestate, a nutrient-rich sludge that can be used as natural fertiliser (**Figure 8**).

Biogas can be used directly for renewable heat and electricity production. If used as biofuel or injected into the natural gas grid, CO_2 is removed in a process called upgrading to produce biomethane, which consists almost exclusively of CH_4 and is approximately equal to natural gas in quality. While focus should be placed on FW minimisation to achieve the highest savings in GHG emissions over the life-cycle of food produce (Dilkes-Hoffman et al., 2018), the biggest advantage of AD over composting is its ability to recover the chemical energy stored in FW (alongside nutrient recovery).

Figure 8 | Anaerobic digestion: at the interface between organic waste management, renewable energy production and sustainable agriculture. Agricultural land provides food, which is distributed through the food supply chain to consumers. Food waste is then treated through anaerobic digestion. The resulting biogas can be used for heating and electricity through a combined heat and power (CHP) plant or upgraded into biomethane, a substitute for natural gas. The digestate, rich in nutrients, can be used as natural fertiliser. Reproduced from Kakadellis, Lee & Harris, 2022.

The AD process is characterised by four main stages, associated with distinct bacterial communities that drive the biochemical transformations (**Figure 9**):

- (1) Hydrolysis: from complex biopolymers (i.e. carbohydrates, proteins and lipids) to smaller, soluble monomers;
- (2) Acidogenesis: from soluble sugars, amino acids and long-chain fatty acids to short chain fatty acids – also referred to as volatile fatty acids due to their low molecular weight and high volatility –, alcohols, CO_2, dihydrogen (H_2), as well as nitrogenous and sulphurous compounds;
- (3) Acetogenesis: from alcohols and volatile fatty acids to acetic acid, H_2 and CO_2;
- (4) Methanogenesis: from acetic acid, CO_2 and H_2 to CH_4 (and some CO_2).

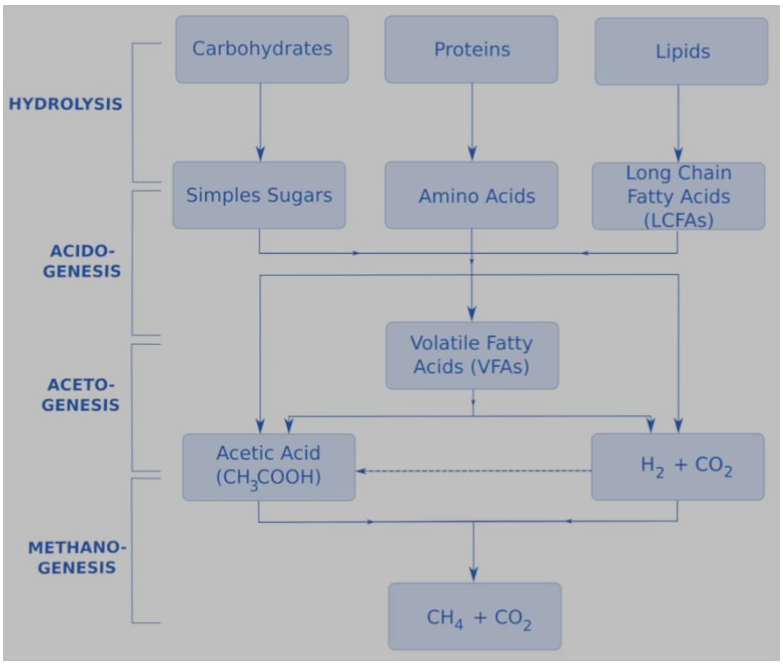


Figure 9 | Main biochemical stages of anaerobic digestion.

AD is a flexible process that can be configured in multiple ways, according to which organic materials are used as feedstocks, which outputs are sought, space and infrastructure. As an efficient waste and wastewater treatment technology, AD has been traditionally used for treating municipal wastewater, organic industrial wastes and agricultural wastes, due to availability and compositional uniformity (Trabold & Babbitt, 2018). More recently, it has gained popularity for the treatment of OFMSW (Angelonidi & Smith, 2015).

As a system, AD can be categorised based on a range of operational parameters, including:

- Temperature: mesophilic or thermophilic AD, based on the temperature range (25-45°C and 50-60°C or above, respectively). Thermophilic systems have a faster throughput, faster biogas production and more thorough destruction of pathogens, but the capital costs are higher, and the process is more energy demanding, particularly for water-rich substrates characterised by high heat capacity (NNFCC, 2021);
- Feedstock moisture: wet or dry AD, based on the total solid and liquid contents of the feedstock (around 10% and 20-40% solids, respectively). As there is less water content to heat, dry systems are often used in thermophilic AD (Angelonidi & Smith, 2015);
- Feedstock supply frequency: continuous, semi-continuous or batch AD, referring to whether the flow of feedstock input is run continuously, partially uninterrupted or whether all required substrates are added at the beginning of the process without any further material flow into or out of the process until the desired fermentation state is achieved. Batch AD has the advantages of comparatively low investment costs, simpler maintenance and operations, but biogas production is periodic (Uddin & Wright, 2022);

- Digester tank number: single or multiple digesters, where some systems have multiple digesters to ensure each stage occurs sequentially and is as efficient as possible, thereby increasing productivity, but also capital and operating costs (NNFCC, 2021);
- Mixing type: reactor types abound, but the most common ones are vertical, stirred-tank reactors and horizontal plug-flow reactors. Feedstock mixing can be achieved through mechanical stirring, biogas recirculation, or with the assistance of a pump or nozzle (Uddin & Wright, 2022). Examples of non-mixing digesters do exist and include covered lagoons (also used for storage), although the mixing process is vital to provide a uniform distribution of bio-available substrates in the tank and to prevent the accumulation of toxic compounds. While the design of complex mixing reactor tanks is associated with more expensive production and operating costs, these costs can be offset by higher biogas yields (Uddin & Wright, 2022).

The nature of the feedstock (alongside budgetary constraints) will inform which AD operating choice to choose from. For FW AD, the preferred option tends to be mesophilic wet AD, due to the high moisture content associated with FW feedstocks (Angelonidi & Smith, 2015). Most digesters are single or double digesters and operate in continuous or semi-continuous flow (NNFCC, 2021; Uddin & Wright, 2022). The reactor tank shape will depend on the financial investment available (NNFCC, 2021).

3.3. Anaerobic digestion in the circular bioeconomy policy landscape

The speed of uptake of AD as FW recycling strategy will depend on the policy landscape and the support it receives in the circular bioeconomy discourse. In 2017, substituting some of its natural gas use for biogas, the EU was able to cut 61 Mt of CO_2 equivalent (CO_2eq), saving the equivalent annual GHG emissions of Bulgaria and representing 1.3% of EU's annual GHG emissions (Bioenergy Europe, 2019). Given the exponential growth of the biomethane industry, with a 40% increase between 2020 and 2021, biomethane has the potential to meet up to 40% of EU gas demand expected for 2050 (EBA, 2021). This demonstrates how biogas can help the EU cut its GHG emissions by 40% by 2030 (Bioenergy Europe, 2019). In this context, a legislative framework enabling the large-scale deployment of sustainable biogas and biomethane and the establishment of reliable value chains is of critical importance to speed-up the decarbonisation of the energy sector (EBA, 2021; Camia et al., 2018).

In 2020, 18,774 biogas and biomethane units were operating in Europe, of which 880 were biomethane plants (**Figure 10**), with a combined biogas and biomethane production of 18 billion cubic meters (bcm), representing 4.6% of the EU's annual gas consumption (EBA, 2021). The development of the biogas industry and subsequent upgrading into biomethane varies across countries, but there is a clear trend towards biomethane production (EBA, 2021).

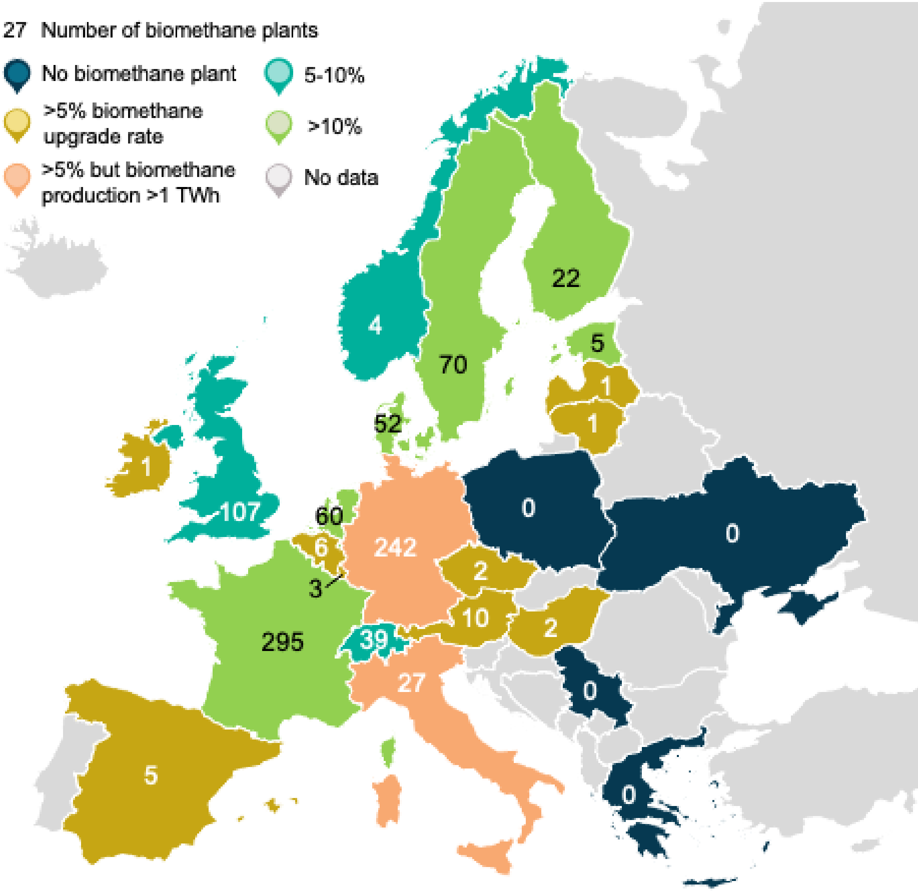


Figure 10 | Biomethane plants and upgrading rates in Europe. Source: EBA, 2021.

The biomethane sector is experiencing faster growth than other traditional biogas valorisation processes, such as heat and/or electricity (Scarlat et al., 2018; Tercinier et al., 2022), with the strongest growth exhibited by the UK, Denmark, France, Italy and the Netherlands (EBA, 2021). While Germany has historically exhibited the largest upgrading capacity for biogas in Europe, with 11,200 to 11,837 biogas plants (EBA, 2021; Tercinier et al., 2022), a favourable national policy framework has led France to establish itself as a European leader in the biomethane sector (EBA, 2021; Tercinier et al., 2022). The example of France demonstrates that by introducing a favourable legislative framework, biomethane production can be dramatically increased (Bioenergy Europe, 2019).

Most producing countries choose to inject biomethane into the natural gas grid (Scarlat et al., 2018), with some exceptions, including Sweden, Finland and Norway (Tercinier et al., 2022). This Nordic trend can be attributed to a relatively small gas transport and distribution network within a relatively vast territory (Tercinier et al., 2022). Nonetheless, Sweden is the world leader in the use of biomethane as transport fuel, accounting for 75% of the biomethane use for transport in the EU (Scarlat et al., 2018).

To achieve further growth in the bioenergy sector, including AD, fossil fuels subsidies should be phased out in favour of measures promoting a credible carbon price able to internalise the negative externalities of GHG emissions (Bioenergy Europe, 2019; Camia et al., 2018). According to the EU Directive 2009/73/EC related to common rules for the internal natural gas market, EU Member States should take concrete measures to assist the wider use of biogas, the producers of which should be granted non-discriminatory access to the gas system, provided that such access is compatible with the relevant technical rule and safety standards.

Main support mechanisms include both supply-side and demand-side schemes (**Table 5**), such as feed-in tariffs – a system providing guaranteed renumeration for each unit of renewable energy produced –, investment support for the building phase of new biogas/biomethane plants, fiscal incentives through carbon taxes, as well as quotas and certificates, whereby governments set a target contribution of renewable energy in a given sector, which can incentivise a trading platform (Tercinier et al., 2022). There has been a push within the EU towards demand-side incentives, including through quotas and the implementation of Renewable Energy Guarantees of Origin (REGO) schemes for renewable electricity (Tercinier et al., 2022). The record gas prices following Russia's invasion of Ukraine and its ripple effects on global energy markets have accentuated the importance of EU's ambitions to produce 35 bcm of biomethane by 2030 (Pronczuk, 2022).

Table 5 | Policy mechanisms for biogas production and use. Adapted from Tercinier et al., 2022.

	Main Support Scheme	**Mechanism**
Supply-side	Feed-in tariffs and premiums	Guaranteed renumeration for each unit of renewable energy produced, usually by offering long-term contracts
	Investment support	Fixed amount received for the building phase of the plants, independently of the amount of renewable energy produced
Demand-side	Fiscal incentives	Tax exemption or reduction on the energy product
	Quotas and certificates	Governmental obligation to reach a proportion of renewable energy in a given sector alongside the issuance of certificates that can be traded between actors in the sector

In the UK, AD has until recently been limited to small on-farm digesters (DEFRA, 2015). Decentralised biogas production and use in rural areas offers the opportunity to green the agricultural sector, provides rural communities with a sustainable source of energy and can be a source of income diversification for on-farm AD plants (DEFRA, 2015). Expanding the market to process significant amounts of organic waste from urban and municipal sources is important to generate increased opportunity for the reuse of biowaste and production of bioenergy (Rigby & Smith, 2011), as we move towards an integrated system for OFMSW.

3.4. Food and food packaging: towards an integrated (organic waste) system

The physical quality of the resulting digestate is equally critical to secure a sustainable AD market (Rigby & Smith, 2011). This remains a challenge for municipal FW, characterised by highly heterogeneous and often contaminated feedstocks (ADBA, 2020). Plastics films are particularly present in FW because most food produce are wrapped in these (WBA, 2018). Prior to AD, the feedstock goes through a mechanical depackager, which separates non-organic matter (i.e. packaging, including plastics) from organic waste through compression and filtering processes. The organic fraction is then fed into the reactor as a purée. The presence of plastic films in digestate at the end-gate, resulting from inadequate depackaging or subsequent screening of the incoming purée, represents a technical, environmental and economic challenge (WRAP, 2019). Though packaging has played a key role in cutting down GHG emissions across complex supply chains (Dilkes-Hoffman et al., 2018), the resulting rise in packaging waste has brough new challenges for the sustainability of food supply chains. Packaging is not the only problem: agriculture uses around 6.5 Mt of plastics worldwide each year, including mulches, nets and tree guards (Fuhr, Matthew & Schächtele, 2019) to protect plants and boost yields. The application of compost and digestate onto agricultural land can also contribute to the unintentional introduction of plastics to soils (MacLeod et al., 2021).

While plastic pollution in aquatic ecosystems, especially in marine environments, has been extensively studied and publicised, its scale and impacts on soil ecosystems remains largely unexplored (Fakour et al., 2021, MacLeod et al., 2021). Based on estimates of plastic fragments in sewage sludge alone, the extent of plastic pollution in agricultural soils globally is likely to exceed that caused by plastic debris on surface waters (Galafassi et al., 2019). Of concern is the vertical migration of plastic particles in agricultural soils and plastic-induced enhancement of pesticide transport towards underlying groundwater systems (Wanner, 2021). Food and its packaging – most of which of plastic origin – must thus be considered as an integrated system as we explore alternatives for diverting FW from landfill and incineration (Trabold & Babbitt, 2018).

Since mechanical recycling of plastic packaging remains highly challenging for multi-layered and food-contaminated plastics (Schyns & Shaver, 2021), designing materials that are compatible with FW processing strategies is an attractive option in building a circular society. Coercive governmental measures to limit fossil-based activities have favoured a shift from fossil-based to bio-based plastics, supported by a circular bioeconomy framework (Spierling et al., 2018).

4. The rise of bioplastics

4.1. Navigating the bioplastics terminology

Despite increasing attention, the term ‘bioplastic’ is commonly misunderstood, due to the ever-growing number of alternative polymers emerging on the market and a lack of well-defined characteristics (Brockhaus, Petersen & Kersten, 2016). Bioplastics are not all made from one single material; just as conventional plastics, they comprise a family of materials from differing feedstocks and capture a range of polymer chemistries, properties and application sectors (Kakadellis & Rosetto, 2021). The term ‘bioplastics’ encompasses two distinct concepts (**Figure 11**):

- Bio-based plastics: plastics (partly or fully) made from biological and renewable resources such as grains, starchy root vegetables, sugarcane or vegetable oils;
- Biodegradable plastics: plastics that can be degraded by naturally occurring microorganisms into water, CO_2 and/or CH_4 and inorganic compounds under certain conditions. The process of biodegradation depends on the surrounding environmental conditions (e.g. location or temperature), on the material and on the application (European Bioplastics, 2016).

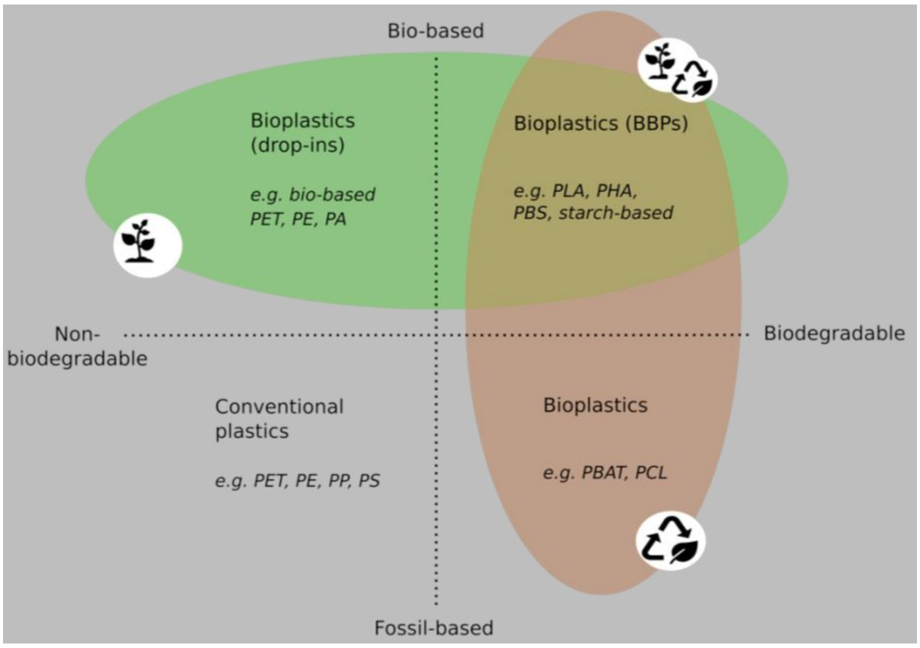
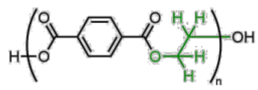
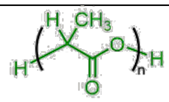
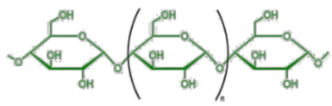
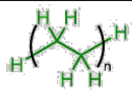
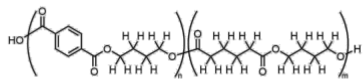
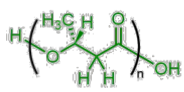
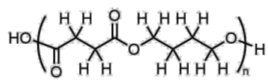
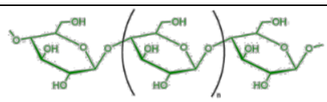
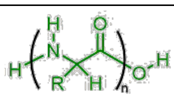


Figure 11 | Bioplastics: material origin and biodegradability properties. PE: polyethylene; PET: polyethylene terephthalate; PA: polyamide; PP: polypropylene; PS: polystyrene; PLA: polylactic Acid; PHA: polyhydroxyalkanoate; PBS: polybutylene succinate; PBAT: polybutylene adipate terephthalate; PCL: polycaprolactone. Reproduced from Kakadellis & Harris, 2020.

A plastic material is defined as a bioplastic if it is either bio-based, biodegradable, or both (European Bioplastics, 2016). **Table 6** shows the most common industrially produced bioplastics.

Table 6 | Most common industrially produced bioplastics. PET: polyethylene terephthalate; PLA: polylactic acid; PE: polyethylene; PBAT: polybutylene adipate terephthalate; PBS: polybutylene succinate; PH3B: poly-3-hydroxybutyrate; PHA: polyhydroxyalkanoate; Y: yes; N: no. Atoms from biological resources are displayed in green, while black atoms reflect a fossil origin. [†]The biodegradability of PLA as a polymer is still contested. It requires an initial activation temperature above 60°C to trigger degradation, due to its comparatively high glass transition temperature (Tg). [#]PBAT and PBS are bioplastics in class transition, since partially bio-based versions of these compounds are currently being developed. Therefore, in a near future, PBAT and PBS are expected to be both bio-based and biodegradable. [§]PH3B is the simplest and most commonly occurring form of PHA, which consists of a large class of polyesters highly prevalent in nature and synthesised by a range of microorganisms. Adapted from Kakadellis, Lee & Harris, 2022.

Polymer	Monomer/ Subunit	Common Feedstocks	Bio-Degradable	Chemical Structure	Market Share
bio-PET	Ethylene glycol & terephthalic acid (TPA)	Corn, sugar beet, wheat, sugarcane, (ethylene glycol) & fossil-based (TPA)	N		19.0%
PLA	Lactic acid	Corn, sugarcane	Y[†]		16.2%
Starch-based polymer	Starch (α-linked D-glucose)	Corn, potato, wheat, cassava, sugarcane	Y		15.8%
bio-PE	Ethylene	Corn, sugar beet, sugar cane, wheat	N		9.1%
PBAT	Adipic acid, 1,4-butanediol (BD) & dimethyl-terephthalate	Fossil-based[#]	Y		7.7%
PH3B (PHA)[§]	Hydroxy-alkanoate	Corn, vegetable oils, food waste, wastewater (through microbial fermentation)	Y		4.5%
PBS	Succinic acid & BD	Fossil-based[#]	Y		3.8%
Cellulose-based polymer	Cellulose (ß-linked D-glucose)	Wood pulp	Y		< 1%
Protein-based polymer	Amino acid	Wheat gluten, soy protein, milk casein	Y		< 1%

While not all biodegradable plastics are bio-based, the vast majority are (European Bioplastics, 2021). This work focuses on bioplastics that are both biodegradable and bio-based, hereafter referred to as biodegradable bioplastics (BBPs), unless stated otherwise (specifically, certified for industrial composting, as outlined below). Non-biodegradable bio-based plastics, referred to as drop-ins, such as bio-based PE (bio-PE) or PET (bio-PET), possess properties identical to their conventional counterparts. Since they are chemically and physically identical to their corresponding fossil-based versions they can be treated in the same existing recycling infrastructure, including mechanical recycling (European Bioplastics, 2016) and are not the focus of this work.

Though less common, an alternative classification of bioplastics can also be made based on the feedstock origin (Queiroz & Collares-Queiroz, 2008):

- Polymers extracted directly from biomass, with or without modification, e.g. starch-modified polymers and polymers derived from cellulose;
- Polymers produced by microorganisms in their natural or genetically modified state, e.g. polyhydroxyalcanoates (PHAs);
- Polymers obtained indirectly from biomass, requiring further biomass processing, e.g. polylactic acid (PLA).

Plastics labelled as 'biodegradable' have become a frequent sight in the retail sector, however few display recognised industry standards to support their claims (WRAP, 2020). Only certified compostable materials according to international standards can be composted in industrial composting plants (European Bioplastics, 2022). These include the harmonised European standard EN 13432 for compostable plastic packaging, the International Organisation for Standardisation (ISO) 17088 and American Society for Testing Materials (ASTM) D6400 standards for compostable plastics and the ASTM D6868 standard for items that incorporate plastics and polymers as coatings or additives with paper and other substrates (**Table 7**).

Of relevance to the European market, a major producer and user of BBPs (European Bioplastics, 2021), the EN 13432 requires compostable plastics to disintegrate after 12 weeks and completely biodegrade after 6 months, with 90% or more of the plastic material needed to have been converted to CO_2 (European Bioplastics, 2022). The UK is primarily certified to EN 13432 by two European certification bodies, TÜV Austria (formerly Vinçotte) and DIN CERTCO (Association for Organics Recycling, 2011). BBPs certified according to EN 13432 can be recognised by conformity labels such as the *Seedling*, *DIN Geprüft*, or *OK compost* industrial logos for industrial composting (**Figure 12**).

Table 7 | Standard specifications for biodegradation of plastic materials.

Environment	Standard Specification	Specified Material/Product	Country/ Region	Certifying Body
Industrial Composting	EN 13432	Biodegradable plastic packaging	EU	DIN CERTCO TÜV Austria
	D6400	Biodegradable plastic materials and products	US	Biodegradable Products Institute (BPI)
	D6868	Materials and products (incl. packaging) that incorporate a biodegradable plastic film or coating (through lamination, extrusion or mixing)		
	AS 4736	Biodegradable plastics	Australia	Australasian Bioplastics Association (ABA)
	ISO 17088	Plastic materials and products incorporating plastic materials	Global	International Organisation for Standardisation
Home Composting	prEN 17427	Biodegradable plastic packaging	EU	DIN CERTCO TÜV Austria
	NF T51-800	Biodegradable plastics	France	DIN CERTCO
	AS 5810	Biodegradable plastics	Australia	ABA

Figure 12 | Certification labels for industrial compostability. The following labels are used across Europe (top), the United States (bottom left), Japan (bottom centre) and Australia (bottom right).

These standards and corresponding labels do not, however, cover industrial AD. Both the ISO and the ASTM provide test methods for evaluating the anaerobic degradation of plastics under high-solid (> 20% total solids (TS) concentration, which reflects dry AD) and low-solid (< 15% TS, as is the case for wet AD, commonplace for FW treatment) conditions for thermophilic and mesophilic ranges, including ISO 14853 and ASTM D5210 for aqueous mesophilic processes under low TS (< 5%), or ISO 13975 for slightly higher TS (< 15%). However, to date, no certification scheme for 'AD-able' material exists (Kakadellis, Woods & Harris, 2021). Nevertheless, BBPs are increasingly used in short-lived food-oriented applications, where their biodegradable properties are most meaningful (WRAP, 2020), and BBP waste is thus likely to be found in AD streams.

4.2. Biodegradable bioplastic packaging in the current landscape

Currently, bioplastics represent about 1% of the 367 Mt of plastics produced annually (European Bioplastics, 2021). The global bioplastics production capacity is set to increase from around 2.42 Mt in 2021 to approximately 7.59 Mt in 2026 (European Bioplastics, 2021). Bioplastics are used in an increasing number of markets, including electronics, agriculture, horticulture and catering products (European Bioplastics, 2021). As for conventional plastics, packaging remains the largest application segment for bioplastics, with almost 48% (1.15 Mt) of the market in 2021 (European Bioplastics, 2021). Certified compostable BBPs are estimated to account for approximately 0.5% of consumer plastic packaging in the UK, corresponding to 8,000 tonnes (t), of which 80% are expected to be flexible packaging and 20% rigid packaging (Ricardo Energy & Environment, 2019). The steady growth of BBPs drives the need to define EoL alternatives (Briassoulis, Pikasi & Hiskakis, 2019).

BBPs can open new possibilities for the post-consumer management of plastics (Kakadellis, Lee & Harris, 2022), but if not carefully managed, they could increase GHG emissions (e.g. through landfill emissions) or exacerbate plastic pollution (e.g. through increased littering and incomplete degradation of some materials) (Gómez et al., 2013). The waste management landscape for certified industrially compostable plastics, let alone BBPs more broadly, is complex, and the environmental consequences of BBPs depend on the waste disposal stream adopted (WRAP, 2020). **Table 8** summarises the main disposal waste BBPs might follow and the suitability of each at the time of writing in the current UK waste management context.

Further expanding the use of BBPs and the development of new markets will depend on a combination of extended product design, improvements in labelling schemes and upgrading the existing waste management infrastructure (WRAP, 2020). The deployment of BBPs should target plastic packaging where effective recycling measures are failing due to the current challenges that remain for treating and recycling materials made of multiple and/or highly food-contaminated layers (Kakadellis & Harris, 2020). Understanding which plastics can be recycled and which ones are economically viable are thus important considerations that need to be taken into account when assessing the potential for the BBP market (Rossi et al., 2015).

The substitution of conventional plastics by certified compostable BBPs should be focused in the area of multi-layered (or composite) food packaging as well as flexible packaging likely to be food contaminated, due to the challenges that still exist for treating and recycling materials made of multiple layers, including plasticisers and adhesives (WRAP, 2020). In addition, food contamination represents a serious issue undermining the viability of conventional plastic packaging recycling as additional cleaning steps make the process more expensive (Kaiser, Schmid & Schlümmer, 2017). The Waste and Resources Action Programme (WRAP), a UK climate action non-governmental organisation (NGO) with an extensive collaborative network and global outreach, published a list of preferred applications for compostable BBPs (WRAP, 2020). The list includes FW caddy liners, fruit and vegetable stickers, tea bags, coffee pods and food ware, especially in closed-loop situations, such as festivals and within the hospitality sector (WRAP, 2020).

Altogether, BBPs can help build a circular economy supported by a bioeconomy, helping governments and citizens of the world achieve the UN's SDGs, in particular SDG 12 on responsible consumption and production. Yet switching from conventional, non-biodegradable plastic to BBP packaging does not necessarily indicate an improvement in overall sustainability, especially when considering their efficacy at preventing FW (Dilkes-Hoffman et al., 2018). In the case of bioplastics, it is important to ensure that they provide genuine environmental benefits compared to their non-renewable counterparts.

Table 8 | End-of-life streams for biodegradable bioplastics. Source: WRAP, 2020.

End-Of-Life Stream	Suitability	Potential Advantages of Using BBPs (if any)	Key Challenges
Home Composting	Compatible only for certified home compostable BBPs (different from industrial composting)	Plastic pollution represents a key challenge for the application of compost and digestate to agricultural soils, especially their fragmentation into microplastics. Provided they are compatible with the relevant stream, BBPs may offer a solution to this pressing and ongoing issue	Compost must be properly managed. The variability in home composting strategies raises questions about the relevance of a standard for home compostable BBPs
Organics Recycling – Industrial (In-Vessel) Composting	Most favourable waste disposal stream		Challenge in distinguishing between compostable and conventional plastics and thus to remove the latter before treatment
Organics Recycling – Open Air Windrow	Compatible only for certain non food-contaminated BBPs approved by the Animal and Plant Health Agency (APHA), e.g. coffee cups		Similar issues to in-vessel composting. In addition, the Animal By-Product Regulations restrict the use of food and food-contaminated waste in open air windrow
Organics Recycling – Anaerobic Digestion (AD)	Ambiguous		Same as for industrial composting. The absence of a standard for ‘AD-able’ BBPs is a major concern
Mechanical Recycling	Problematic	N/A	The risk of BBPs entering the conventional plastics recycling stream is a major concern
Paper Recycling	Problematic	N/A	While paper recycling can tolerate some plastic contamination, it is not optimal and the introduction of BBPs can undermine subsequent separation (e.g. recyclable plastic-lined drink cartons and cups)
Residual Waste – Energy-From-Waste (EfW)	Optimal stream in the absence of suitable organic infrastructure	Biodegradability has little merit here, but if the material is bio-based and substitutes a conventional non-recyclable plastic, incineration with energy recovery is the most environmentally sound option for residual waste	None, although this option is criticised for both conventional and biodegradable plastics alike for its lack of circularity and contribution to air pollution
Residual Waste – Landfill	Problematic	Prevent plastic waste from leaking into the environment	Unlike conventional plastics, BBPs are not inert and may emit GHG emissions if anaerobic degradation takes place
Open Environment (mismanaged)	Problematic	In some (very rare) cases, BBPs may be able to fully biodegrade in the open environment and may represent a solution to fugitive plastics	Huge variability in soil conditions raises questions about the relevance of a standard for BBPs than can biodegrade in the natural environment

4.3. *Bioplastics and environmental sustainability: a nuanced debate*

Adopting a life-cycle approach when assessing the environmental impacts of BBPs can help clarifying the trade-offs that may occur between carbon intensity of polymer production, FW prevention and waste management at the EoL (Kakadellis & Harris, 2020). A few reviews have performed an analysis of bioplastic life-cycle assessments (LCAs) in packaging applications (Hottle et al., 2017; Kakadellis & Harris, 2020; Yates & Barlow, 2013). The results suggest that BBPs are not as environmentally friendly as they are currently perceived by the general public (Kakadellis & Harris, 2020) (**Table 9**). They highlight that the benefits of a lower carbon footprint are often compounded by agricultural inputs associated with crop cultivation for feedstock production (Kakadellis & Harris, 2020). The further environmental costs of bioplastic granule production and film processing point out the need for process optimisation and alternative feedstocks (Leceta et al., 2014).

Table 9 | Main advantages and disadvantages of biodegradable bioplastics.

Advantages	Disadvantages
Made from renewable resources (i.e. bio-based)	Environmental cost associated with agricultural inputs for crop production
Biodegradability supports a join food and organic waste management system	Represent a contaminant in the conventional plastic recycling stream
Contribute towards a circular bio-based economy (conceptually)	May lead to a rise in littering due to public perception that bioplastics will break down in the natural environment
More distributed resources (unlike oil and gas) able to boost rural communities	Energy intensive material production and processing
New material properties with novel chemistries	Few bioplastics are currently market competitive compared to their fossil-based counterparts
Potential solution to address hard-to-recycle plastics	Biodegradability properties are only relevant within the appropriate waste management infrastructure

The optimisation potential of BBPs is significant (European Bioplastics, 2017). The biopolymer and oleochemical industry – which supplies the chemicals for the modification and compounding in biopolymer production – is still young and thus currently has a lower optimisation degree than the petrochemical industry (Vidal et al., 2007). This potential should be taken into consideration when assessing material preference; otherwise LCA can become a tool that tends to hinder innovation by favouring already optimised material streams (European Bioplastics, 2017). Innovation opportunities include alternative feedstocks that exhibit high agro-ecological adaptation (Casarejos *et al.*, 2018). There has been a growing interest in bioplastic production from waste and by-products, such as soy protein, chitosan and seaweed (Casarejos et al., 2018; Leceta et al., 2013; 2014). The development of more effective and efficient bioplastics is particularly promising, notably in multi-layer film packaging applications. The introduction of additives e.g. nanoclays and surfactants, can enhance the performance of some BBPs close to that of conventional plastics and reduce food wastage by extending shelf-life and preventing microbial growth (Lorite et al., 2017).

Opting for BBPs over conventional plastics in food packaging applications may thus not be guided by their assumed inherent eco-friendliness, but by the advantages they confer to the food and packaging waste management system (Kakadellis & Harris, 2020). Considering the environmental impact of FW on food packaging LCAs, the biodegradable properties of BBPs offer an unprecedented solution to divert FW from landfill while also preventing plastics from leaking into the environment, thereby contributing to a circular bioeconomy (Kakadellis & Harris, 2020). However, despite the certification standards that have attempted to systematise the labelling of BBPs for specified EoL streams (**Table 7**), uncertainties around their biodegradability in various managed and unmanaged environments still exist, particularly for AD processes.

4.4. Biodegradability – myth or reality?

Despite a growing body of scientific literature, little is known about BBP biodegradation efficacy in given waste treatment streams and their influence on terrestrial and aquatic ecosystems is still poorly understood, particularly for microbial communities (Sander, 2019; Zettler, Mincer & Amaral-Zettler, 2013). The biodegradability of BBPs depends on their distinct chemical and physical structure; generally, polymers with shorter chains, more amorphous content (i.e. lacking a clearly defined molecular arrangement, as opposed to a crystalline one) and less complex formulations are more prone for biodegradation (Emadian, Onay & Demirel, 2017).

Polymer biodegradation takes place in three steps through microbial activity (**Figure 13**):

- Bio-deterioration: the modification of the mechanical, chemical and physical properties of the polymer;
- Bio-fragmentation: the conversion of polymers to shorter and simpler chains (i.e. oligomers and monomers);
- Bio-assimilation: the conversion of organic matter, energy and nutrients into CO_2, water and biomass (Emadian, Onay & Demirel, 2017; Kjeldsen et al., 2018).

It is essential to ensure that biodegradation does not stop at the fragmentation stage, as this leads to the accumulation of microplastics (Kjeldsen et al., 2018). The environment plays a crucial role in determining how a given BBP will biodegrade; factors such as pH, temperature, moisture and O_2 content are amongst the most significant (Emadian, Onay & Demirel, 2017).

In the ocean, low temperature and O_2 levels hinder biodegradation (Emadian, Onay & Demirel, 2017). For example, PLA, the most common BBP (European Bioplastics, 2021), performs well in industrial composting and thermophilic anaerobic treatments, but is currently unsuitable for home composting or mesophilic AD (Kolstad et al., 2012; Narancic et al., 2018). These treatments do not reach the initial activation temperature above 60°C required to trigger degradation, due to the comparatively high glass transition temperature (Tg) of PLA (57°C) (Zaaba & Jaafar, 2020).

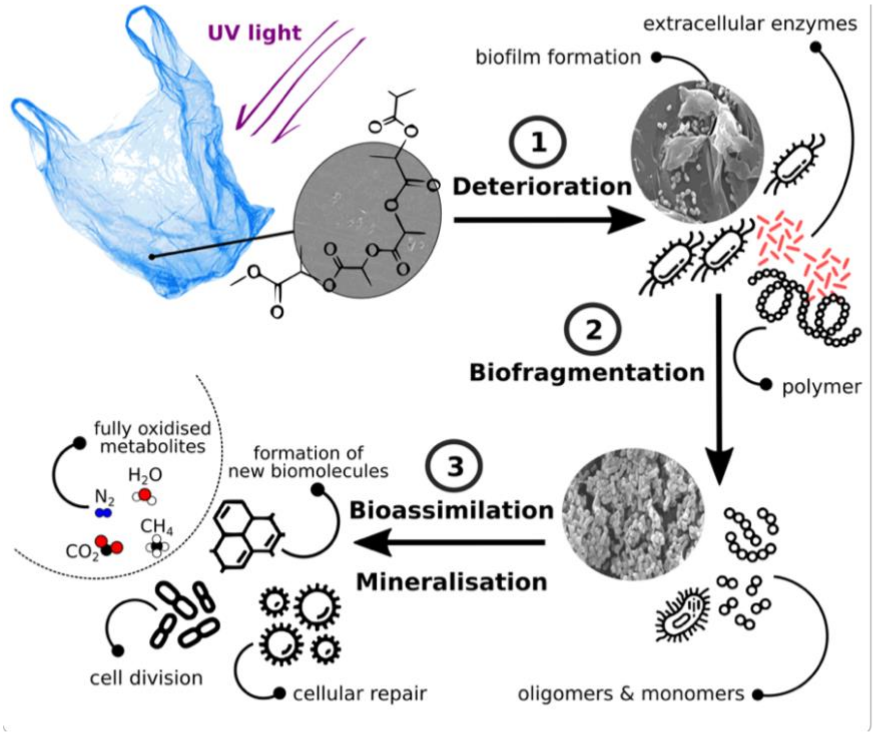


Figure 13 | Plastic biodegradation steps. Electron microscopy image sources: Rasheed et al., 2021; Quiroz-Castillo et al., 2014; Zettler, Mincer & Amaral-Zettler, 2013.

In addition, blending various polymers, which is commonly done in industry, can lead to synergistic or antagonistic effects (Narancic et al., 2018). The addition of polycaprolactone (PCL) to PLA not only improves the material properties of the blend, but also makes the material home compostable (Narancic et al., 2018). Similarly, the thickness of any given material will influence its rate of biodegradation; a thin BBP film will biodegrade while a coffee cup of the same material may fail to pass the test (van den Oever et al., 2017), underlining the importance of testing materials in their commercial stage, rather than as raw materials.

There are still gaps in knowledge on the impact of BBPs on microbial community structure and function and how this impacts the wider ecosystem (Bandopadhyay et al., 2019; Pinnel & Turner, 2019; Pinto et al., 2019; Sander, 2019). The extent to which microorganisms incorporate carbon into their biomass is species- and substrate-specific and dependent on experimental conditions, e.g. the presence of alternative carbon sources (Sander, 2019).

Characterising microorganism communities present in AD sludge could help clarify the differences in performance of various BBP materials (Narancic et al., 2018) and identifying hotspots for further process optimisation. Some studies have aimed to characterise a species (or genus) of bacteria or fungi that would show higher rates of (bio)plastic biodegradation (Liu et al., 2019; Morohoshi et al., 2018; Zettler, Mincer & Amaral-Zettler, 2013). Among the numerous microbial species associated with BBP biodegradation, those belonging to bacterial *Pseudomonas*, *Streptomyces*, *Arthrobacter* and *Rhodococcus* and fungal *Aspergillus* and *Fusarium* genera are commonly cited (Danso, Chow & Streit, 2019; Pathak & Navneet, 2017). Spiking of the microbial inocula with known plastic degraders could be applied to enhance biodegradation, through research suggests that the addition of non-indigenous cultures to inocula causes competition with microorganisms present in the consortium and may lower biodegradation activity (Muhonja et al., 2018).

While the previous sections have uncovered some key technical challenges in the use of BBPs in food packaging – and applicable to the FMCG sector in general –, these issues are deeply intertwined with their social dimension, which are explored next.

5. Plastics, people and behavioural patterns – a socio-technical system

5.1. Behavioural models

Unlike other environmental issues, such as the release of CO_2 into the atmosphere that are both anthropogenic and from natural biophysical sources, the accumulation of plastics in the environment stems directly and solely from human decisions and behaviour (Pahl, Wyles & Thompson, 2017). The path towards sustainability requires changes in governance and technological innovation, but also a major shift in environmental behaviour (Schill et al., 2019). It is therefore not sufficient to describe environmental problems without considering the role of people in the process (Pahl & Wyles, 2017).

Environmental behaviour encompasses all types of behaviours that change the availability of resources or energy from the environment or affect ecological or biophysical processes (Stern, 2000). When those behaviours aim to reduce environmental harm, or even bring environmental benefits, they are defined as pro-environmental behaviour (Steg & Vlek, 2009). Steg & Vlek (2009) outlined four key stages for studying pro-environmental behaviours and monitoring the progress of behavioural interventions: (1) choosing the target behaviour(s) to address a given environmental issue, (2) identifying the factors driving the relevant behaviour(s), (3) selecting interventions most effective in encouraging the relevant behaviour(s) and (4) systematically evaluating the consequences of these interventions on the target behaviour(s) and environmental and social outcomes.

By adopting pro-environmental behaviours aimed at achieving resource efficiency and circularity, referred to as circular behaviours (Muranko et al., 2020), individuals can contribute towards closing resource loops necessary for the revalorisation and recircularisation of materials, components and products (Wastling, Charnley & Moreno, 2018). Understanding the cognitive, motivational and contextual factors and mechanisms underlying behavioural actions is thus pivotal in implementing interventions aimed at building circular production and consumption systems (Steg & Vlek, 2009).

A range of behaviour models has been developed and revised over the past decades. While these models were generally not designed to address pro-environmental behaviours in mind, they have been applied in this context, including circular (i.e. reuse and recycling) behaviour (Allison et al., 2022a; Tassel & Aurisicchio, 2021), conservation behaviour (Vining & Ebreo, 2002) and travel mode choice (Steg & Vlek, 2009). The norm-activation model considers behaviour to be primarily pro-social and driven by subjective norms (Schwartz, 1970; 1973). The value-belief-norm theory attempts to link the former norms with general values and environmental beliefs in a causal chain (Stern, 2000). Taking a broader stance (though still in the realm of motivational factors), the theory of planned behaviour (TPB) (Azjen, 1991) is arguably the most influential behavioural framework to date (Steg & Vleg, 2009; Tassel & Aurisicchio, 2021). Built on the assumption that individuals make rational choices based on a personal cost-benefit analysis (akin to Adam Smith's rational choice theory omnipresent in classical economics), the TPB stipulates that behavioural intentions are shaped by attitudes, subjective norms (perceived social pressure) and perceived behavioural control (perceived agency of a behavioural action) (Ajzen, 1991).

These behavioural models can be limited because of their tendency to focus on psychological factors and their assumption of linear causal chains (Tassel & Aurisicchio, 2021; Whitmarsh, Poortinga & Capstick, 2021), thus providing only a snapshot of drivers of change. Human behaviour is not dependent on motivational factors alone; many contextual factors may enhance or hinder motivational factors, thereby influencing the likelihood of the desired behavioural outcome (Steg & Vlek, 2009). For example, the TPB only considers how contextual factors are *perceived*, rather than their *actual* effects on behavioural intentions and behavioural outcomes (Steg & Vlek, 2009). Contextual factors can affect behaviour directly, they can mediate or moderate the relationship between behaviour and motivational factors (Steg & Vlek, 2009). For example, one cannot sort their household waste without the provision of a separate waste collection scheme; the latter may also boost positive attitudes towards recycling because its availability makes recycling more convenient but may only result in actual recycling behaviour among individuals who are already sensitised to environmental issues (Steg & Vlek, 2009).

Efforts have been made to increase the reliability of established models through the addition of extrinsic variables in revised TPB models (Ertz et al., 2017; Klöckner, 2013; Tonglet, Phillips & Read, 2004), and newer models that do not assume behaviour as a linear process are emerging (Fogg, 2009). According to Fogg's model, behaviour arises from the convergence of motivation (physical, emotional and social), ability (time, cost, physical effort, mental effort and habit) and prompts, or nudges (a behavioural cue) (Fogg, 2009). However, these models still assume that individuals act in isolation and provide limited information on the system in which these individuals evolve (Tassel & Aurisicchio, 2021) and neither theory is linked to a systematic method for designing interventions (Gainforth et al., 2016).

A model of behaviour, the capability-opportunity-motivation and behaviour model, or COM-B model, has been developed by Michie, van Stralen & West (2011) with the aim to assist behaviour change interventions in identifying appropriate targets for those interventions. The COM-B model is founded upon the observation that at any given moment, a particular behaviour will occur only when the person concerned has the capability and opportunity to engage in the behaviour and is more motivated to enact that behaviour than any other behaviour (Michie, van Stralen & West, 2011). The COM-B system helps understand what needs to change for the desired behaviour to occur and has since been expanded to the Behaviour Change Wheel for characterising and designing behaviour change interventions. It consists of three parts: 1) an inner hub, the COM-B model, 2) a middle layer of intervention types, and 3) an outer layer of policy options targeting the relevant types of intervention (**Figure 14**).

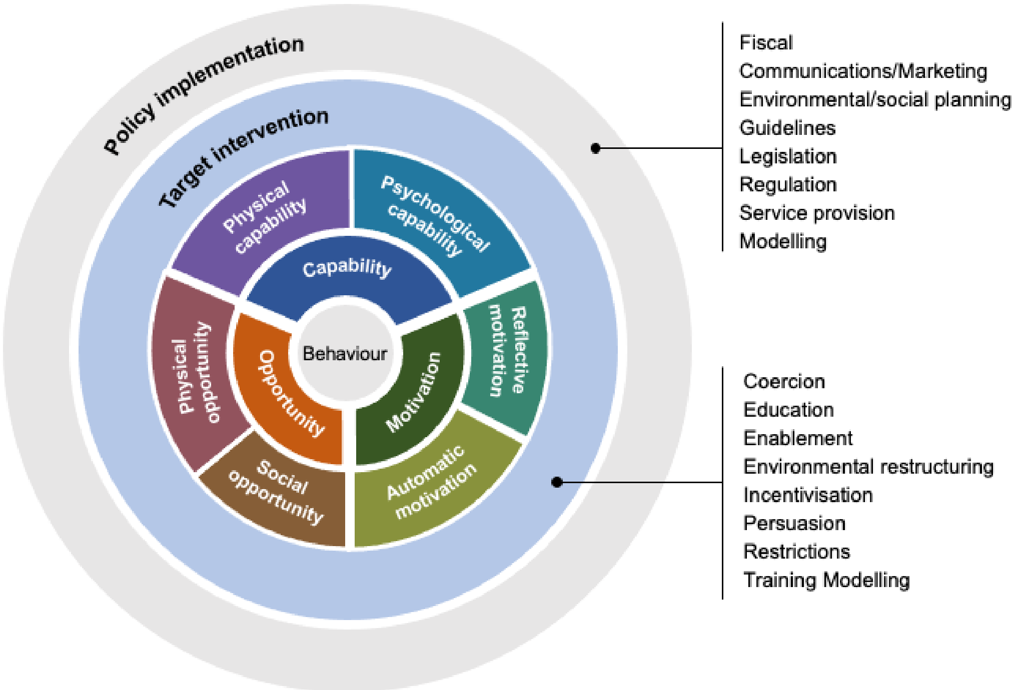


Figure 14 | The Behaviour Change Wheel framework. Source: Michie, van Stralen & West, 2011.

5.2. Consumer attitudes towards bioplastics

Consumer attitudes towards bio-based, biodegradable and compostable plastics have been subject to increased research interest by social and behavioural scientists. A pan-European study surveying the general public has shown that 'bio-based' as a concept remains unfamiliar and is most often associated with positive environmental terms, such as 'naturalness' and 'eco-friendliness', but also with negative environmental issues and to some extent with technological and health issues (Sijtsema et al., 2016). Similarly, an Australian study showed that the public's knowledge of bioplastics is low, but their perception, particularly of BBPs, is positive; biodegradable plastics were perceived as better for the environment than conventional plastics and even easily recyclable plastics (Dilkes-Hoffman et al., 2019a).

The preference for BBPs and other seemingly environmentally friendly materials may be strengthened by their perceived advantages in light of ever-growing plastic pollution. Indeed, the risk perception of plastic pollution has changed in the recent decades, in part due to increasing awareness of both environmental and human health impacts of plastics and microplastics (Heidbreger et al., 2019). While plastic pollution awareness may have heightened, changes in behavioural practices may not necessarily reflect this, and in some cases lead to rebound effects (Heidbreger et al., 2019; Ansink, Wijk & Zuidmeer, 2022). Some scholars have warned against the self-licensing effect of BBPs, whereby the 'green' credentials of BBPs may drive consumers to pay less attention to plastic reduction behaviour (Longoni et al., 2014).

Intriguingly, although BBP packaging has strong environmental appeal to consumers, paradoxically this does not translate in appropriate disposal of the packaging waste (Taufik et al., 2020). This was reflected in another study, with 62% of respondents saying they would dispose of bioplastic items in the recycling bin (Dilkes-Hoffman et al., 2019a). Consumers with a stronger familiarity with bio-based products more often correctly disposed of compostable bio-based packages, but not recyclable bio-based packages, relative to fossil-based packages. These studies highlight both the complexity and a lack of familiarity with the relevant terminology.

There is currently no standardised guidance in the UK covering the communication of biodegradability claims of BBPs (WRAP, 2020). Current food packaging does not seem to communicate information on disposal methods adequately and appears to be designed as a container before and during the consumption of the contents, rather than a facilitator for separating and sorting the packaging after the contents are consumed (Nemat et al., 2022). However, clear labelling is key to minimise the risk of contamination (to either the organic or conventional plastic recycling stream) (WRAP, 2020), especially for products that are purchased regularly and that are sorted habitually (Nemat et al., 2022).

Labelling exists only in the form of certifications for home and industrial composting (and a handful of controversial ones for various unmanaged natural environments). The public is generally unaware of (and to some extent insensitive to) the meaning and implications of such labels (Ansink, Wijk & Zuidmeer, 2022; Taufik et al., 2020), which provide little information about disposal and rarely account for the waste collection infrastructure available at the local level (WRAP, 2020). Given the significance of contextual factors for pro-environmental behaviours (Steg & Vlek, 2009) and the role of the latter in addressing major sustainability issues (Steffen et al., 2015), characterising the wider system in which BBPs exist is an important step towards achieving a circular bioeconomy and anchors plastics sustainability research in interdisciplinarity and transdisciplinarity characteristic of 21st century research.

5.3. Behaviour in the 21st century science: from silos to systems

The current global food system, which encompasses activities across the entire food supply chain and includes the disposal and management of food and food packaging waste, remains defined by research conducted in disciplinary silos, with little consideration for the diversity of actors and actions involved (Rosenzweig et al., 2020). Historically, the dominant view of human behaviour has relied on a reductionist approach across disciplines, from economic and social sciences to natural resource management, whereby individuals were viewed as self-interested, perfectly disciplined and rational economic agents (Schill et al., 2019). However, the gradual recognition that humans are socially and culturally wired and confined to the limits of the biosphere they ultimately depend on has shifted the research space towards a more integrated and dynamic view of human behaviour (Schill et al., 2019).

This transition from silos to systems can be seen through the lens of design for sustainability, which has moved from a techno-centric to people-centric approach, incorporating environmental and social dimensions into design questions (Ceschin & Gaziulusoy, 2016). Design for sustainability approaches have been categorised into four ‘innovation levels’, each of which aims to address environmental problems through innovation: product-oriented (designing new and better materials/products), product-service system (new business models, e.g. from car ownership to car sharing), spatio-social system (focus on the spatial and social dimensions of human communities, e.g. community gardens for food security) and socio-technical system (focus on the relationship between technologies, ecosystems and social and cultural practices) (Ceschin & Gaziulusoy, 2016).

As an alternative material to conventional plastics, BBPs fall under the first category and are currently limited in addressing consumerism or synergies and potential trade-offs within the food-energy-waste nexus. Yet, materials as technical innovations imply behavioural changes because individuals need to accept, understand and use them appropriately (Steg & Vlek, 2009). As we shift from a linear to circular economy model, with resources flowing beyond consumption stages to reuse, recycling and redesign streams, consumers will become key players in the value chain (Wastling, Charnley & Moreno, 2018). As noted by Selvefors, Strömberg & Renström (2016), it is not possible to design behaviour; only the conditions driving behaviour can be designed to nudge it into a desired direction. Thus, in order to fully capitalise on the benefits BBPs may bring, understanding and characterising their wider social context is essential.

5.4. Framing BBPs within a complex adaptive system

The design of circular products and circular production and consumption models comes hand in hand with circular behaviours, which need to be accounted for (Wastling, Charnley & Moreno, 2018). Behaviour is a part of a system of behaviours that interact and compete with each other (Allison et al., 2021); if circular behaviours are to be adopted, design strategies to assist consumers in adopting these behaviours must be addressed (Wastling, Charnley & Moreno, 20218). The socio-technical system within which BBPs exist is characterised by a complex and dynamic set of interacting elements, where consumers' behaviours are influenced by a range of systemic factors (**Figure 15**), including intrinsic drivers, infrastructure and the social structure they are part of, ultimately enabling or hindering the flow of BBPs throughout the consumption phase, from product acquisition and utilisation to disposal.

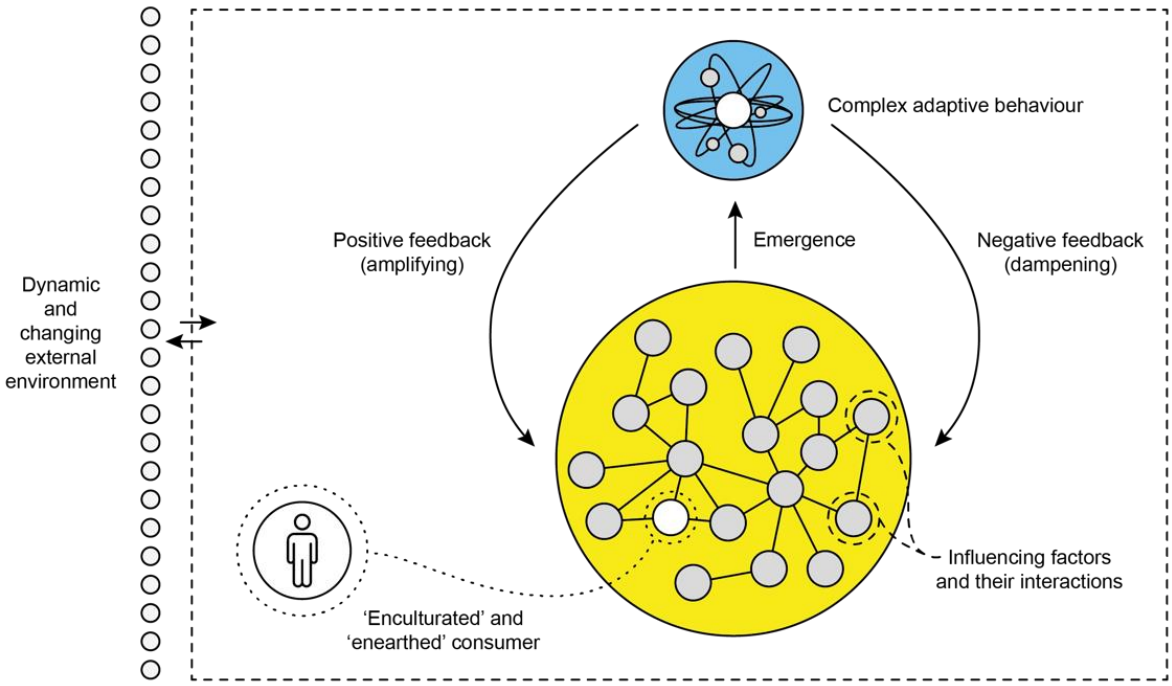


Figure 15 | Complex adaptive system. Individuals in their broader socio-cultural contexts ('enculturated') and embedded within the biosphere ('enearthed') (Schill et al., 2019) are elements of the system and influence and are influenced by a range of system elements (or factors), and their interactions. Reproduced from Kakadellis et al. (in review).

Renner & Giampietro (2020) have also called for the consideration of the biophysical feasibility, techno-economic viability and social desirability in the design of paths to sustainable society and strategies to achieve those. The transdisciplinary approach of complex adaptive systems lends itself well to addressing these challenges through an integrated application of disciplines not commonly studied together. A complex adaptive system is a framework for systems that cannot be determined by linear (however intricate) causal relationships. It is complex in that it is a dynamic, self-organised network of interacting elements, where the behaviour of the whole may not be solely predicted by summing the behaviour of individual system elements, as is the case of a reductionist approach. It is adaptive in that both individual and collective behaviours evolve over time in response to a triggering event or collection of events.

Adopting a complex adaptive system approach enables a deeper understanding of human behaviour, the broader social, technical and bio-physical context shaping it, and their continuous co-evolution (Schill et al., 2019), and represents the starting point for identifying where best to intervene and how. Changes in current consumption patterns will inevitably require structural changes within current power and social dynamics (Renner & Giampietro, 2020). Understanding how consumers adapt to intervention strategies and how and why changes occur over time is critical to ensure that BBPs deliver on their environmental promise.

6. Summary

In summary, the literature review presented in this chapter outlined the following key points: (1) a historically fossil-based economy has propelled plastics into a consumption-oriented society; (2) the sharp growth in disposable plastics as FMCGs has led to a plastic pollution and management crisis; (3) aiming to address both issues, the circular bioeconomy proposes a novel framework to achieve sustainable wellbeing within planetary boundaries based on the use of renewable natural capital and the design of closed material, component and product loops; (4) under this framework, the role of AD for the treatment of OFMSW, including FW, has gained increasing attention; (5) equally promoted by this framework, the development of BBPs compatible with organic waste management streams represents an opportunity to address plastic pollution; (6) harnessing this opportunity requires BBPs to fulfil certain design and EoL requirement, and, lastly; (7) it also requires actors along the BBP value chain to adopt certain behaviours/practices, highlighting the socio-technical system within which BBPs are embedded and the need for a systems-thinking approach to address the complexities within that system.

The following chapters cover different disciplines purposefully, so that the compatibility of BBP food packaging within a circular bioeconomy framework can be appraised holistically.

The challenges and opportunities associated with treating BBP food packaging through a co-mingled organic waste stream for AD are explored from a range of perspectives, including biochemical (**Chapter 4**), technical (**Chapters 4, 5 & 8**), policy (**Chapters 5, 6 & 7**), social (**Chapters 5, 6 & 7**) and design (**Chapters 6 & 7**) lenses. Designing the research approach in such a way and combining the findings from each discipline into a single, cohesive picture enables this thesis to make a truly interdisciplinary contribution to the fields of plastic sustainability, circular bioeconomy and system design.

Chapter 4 – Biochemical and microbial characterisation of biodegradable plastics and food waste anaerobic co-digestion

"*Science never solves a problem without creating ten more.*" – George Bernard Shaw

Part of the content presented in this chapter appears in the following publication:

Kakadellis, S., Lee, P.-H. & Harris, Z. M. (2022). Two Birds with One Stone: Bioplastics and Food Waste Anaerobic Co-Digestion. *Environments*, 9 (1), 9.

Microbial 16S rRNA bioinformatics analysis was carried out by Johanna de la Cruz as part of an undergraduate summer research project at Imperial College London. This only related to the bioinformatics pipeline and did not include sample preparation nor data interpretation.

1. Introduction

The development and commercialisation of BBPs represent an opportunity to address some of the issues outlined in **Chapter 3** (**Section 3.4**) and move towards an integrated food and food packaging waste stream for AD. Plastic packaging, in particular plastic films, are prevalent in source-separated FW collections (ADBA, 2020); conventional PE carrier bags account for 6% weight by weight (w/w) of separately collected FW and supermarket food produce past their sell-by date are often disposed of in their packaging (Banks et al., 2018). The design of a co-mingled waste management strategy for AD, in which both BBP food packaging and FW are digested side-by-side and referred to as anaerobic co-digestion (co-AD), could simplify collection and processing and help reduce contamination levels in both incoming feedstocks and AD products generated downstream (Abraham et al., 2021).

BBPs have high theoretical biochemical methane potential (BMP) (Narancic et al., 2018), a standard test and corresponding metric that provide an indication of the biodegradability of a given substrate and its potential to produce CH_4 under anaerobic conditions (Jingura & Kamusoko, 2017). The chemical composition of BBPs (mostly carbon (C), oxygen (O) and hydrogen (H)) could contribute towards optimal C to nitrogen (N) ratio (C:N), a key parameter for efficient CH_4 production (Narancic et al., 2018). However, it is now recognised that the biodegradability of BBP films – and BBPs more broadly – is dependent on environmental conditions (Banks et al., 2018).

A range of methodologies have been developed to monitor the rate and extent of biodegradation of BBPs in various environments (Ruggero, Gori & Lubello, 2019). Although the scientific literature has primarily focused on aerobic conditions, such as industrial composting (Quecholac-Piña et al., 2020), composting alone as a disposal method for FW and food packaging waste is currently limited (see **Chapter 3 Section 3.2**).

More recently, the degradation of BBPs under anaerobic conditions has been the subject of dedicated reviews (Bátori et al., 2018; Stagner, 2015; Quecholac-Piña et al., 2020). However, these reviews focused on the process of biodegradation itself, rather than considering bioplastics within the existing organic waste management infrastructure. Moreover, biodegradation assays of BBPs have traditionally been performed either alone or in co-digestion with sewage sludge, while the suitability of BBPs for FW co-AD remains underexplored (Bátori et al., 2018; Gómez et al., 2013). Given the recent policies aimed at FW recycling outlined in **Chapter 3 Sections 3.1 & 3.3**, as well as the growing proportion of plastic packaging in the food supply chain, addressing BBP co-AD with FW is paramount to ensure they are studied in their most appropriate context.

2. Study aims

To address this research gap, a co-AD experimental study was designed, with current waste composition and management practices in mind. A synthetic food waste (SFW) recipe reflective of current household FW composition in the UK was first developed. Biogas and CH_4 yields of co-AD experiments were quantified to assess the impact of several BBPs on AD performance, followed by characterisation of microbial community structure and putative function. While undertaking this study, a lack of consensus in experimental study design was identified. This led to a formal examination of the design of industry-relevant research on BBP and FW co-AD, informed by a rapid literature review, which is also discussed here.

3. Materials & Methods

3.1. Rapid review of literature

A rapid review of the relevant literature on BBP and FW co-AD was conducted between May and June 2020 and was subsequently updated in May 2021 and October 2022. A rapid review follows the same methodology as a systematic review, although some of the components of the review process are simplified or omitted, thereby considerably reducing its timeframe (Higgins et al., 2019). Rapid reviews may be conducted for new and emerging topics (James Cook University, 2022). Given the relatively niche field of BBP and FW co-AD, a rapid review was thus deemed most suitable.

The original search was conducted on May 17th, 2020, using Scopus at the citation index service and based on the following search string, using Boolean operators AND and OR:

TITLE (bioplastic* OR plastic* OR polymer OR film OR waste) AND TITLE (bio-based OR biobased OR biodegradable OR hydroxyalkanoate OR polylactic OR starch) AND TITLE (anaerobic OR AD OR digest* OR biogas OR methane) AND TITLE-ABS-KEY (food OR organic OR municipal)

Technical papers on bioplastic chemical formulations were excluded. Inclusion criteria were:

- Polymer type: bioplastics (bio-based, biodegradable, and/or compostable)
- Treatment condition: anaerobic co-digestion with FW
- Article type: original research article

References were recorded and processed in the citation manager Mendeley. The latest updated search on October 11th, 2022, retrieved 81 documents, with additional entries since the original search screened for relevance to the research question using the inclusion criteria. Ten studies were kept for critical appraisal, with six additional ones obtained through snowballing. All retained studies were in English and were published between 2012 and 2022.

3.2. Substrate preparation

3.2.1. <u>Synthetic food waste recipe</u>

A SFW recipe was developed to reflect current UK household FW composition as realistically as possible and to avoid contamination from non-food materials. Individual food items were ordered online from Tesco, a major retail supermarket. The quantity and type of individual food items are displayed in **Table 10**. Pasta, rice and all vegetables except lettuce were cooked first before being blended with the remaining ingredients with an immersion blender. The resulting purée was stored in a freezer at -20°C in 500 g resealable plastic freezer bags and defrosted on an ad hoc basis.

Packaging associated with the food items purchased was also recorded (**Table 11**). Overall, for 10 kg of SFW, 364 g of food packaging were needed (3.5% of the total weight), slightly less than the SFW recipe developed by Zhang et al. (2019a), who recorded 4.5% of the total weight stemming from packaging. Plastics accounted for 65.1% of the total packaging weight, of which 50.6% and 49.4% were recyclable and non-recyclable plastics, respectively. These values helped inform the concentration of plastic fragments (both conventional and EN 13432 certified compostable) inoculated in subsequent co-AD trials.

Table 10 | Materials used for preparation of synthetic food waste. Weight values are on a fresh weight basis and were informed by: Zhang, Heaven & Banks, 2018; FUSIONS, 2016 & WRAP, 2019.

Food Category	Food Item	Quantity (g)	Of Which Peel/Shell (g)	Sub-Total (g)	Sub-Total (%)
Bakery	Wholemeal sliced bread	800		1796	18.0
	White sliced bread	400			
	Pitta bread	348			
	Tortilla wraps	248			
Vegetables	Potatoes	1275	160	2625	26.3
	Veg mix (frozen)	600			
	Broccoli	330			
	Tomatoes	160			
	Lettuce	140			
	Carrots	120	14		
Fruit	Apples	670	90	1815	18.2
	Satsumas	600	125		
	Bananas	380	218		
	Pears	165	120		
Eggs	Eggs	100	11	100	1.0
Dairy	Semi-skimmed milk	568		1068	10.7
	Greek yoghurt	200			
	Single cream	150			
	Cheddar (grated)	150			
Meat & Fish	Pork & beef meatballs	360		670	6.7
	Chicken breast	170			
	Fish fingers (frozen)	140			
Confectionary	Baked beans	210		520	5.2
	Jam sandwich creams	160			
	Crisps	150			
Ready meals	Pizza (frozen)	314		914	5.2
	Rice (cooked)	300			
	Pasta (cooked)	300			
Drinks	Water	245		492	9.1
	Lemonade (sparkling)	247			
Total				**10000**	**100**

Table 11 | Packaging content from food produce for the synthetic food waste recipe.

Packaging Type	Sub-Total (g)	Sub-Total (%)	With Food Waste (%)
Plastics	237	65.1	2.3
Of which recyclable	120	32.9	1.2
Of which non-recyclable	117	32.1	1.1
Cardboard	127	34.9	1.2
Total	**364**	**100**	**3.5**

3.2.2. Plastic samples

Commercially developed conventional and biodegradable plastic films were used as plastic substrates in the co-AD trials. Both types of the BBP films selected for the study were cellulose-polybutylene succinate adipate (PBSA) polymer blends in their raw natural, undyed (BioN) and final, printed (BioP) form. The conventional polymer used was made of unprinted PET. All polymers were supplied by a packaging manufacturer specialising in sustainable flexible packaging and were of food-grade quality. The BBPs films were certified for industrial composting based on the European EN 13432 standard. The films were cut into 10 x 10 mm^2 pieces before being added to the glass bottles (**Figure 16**).

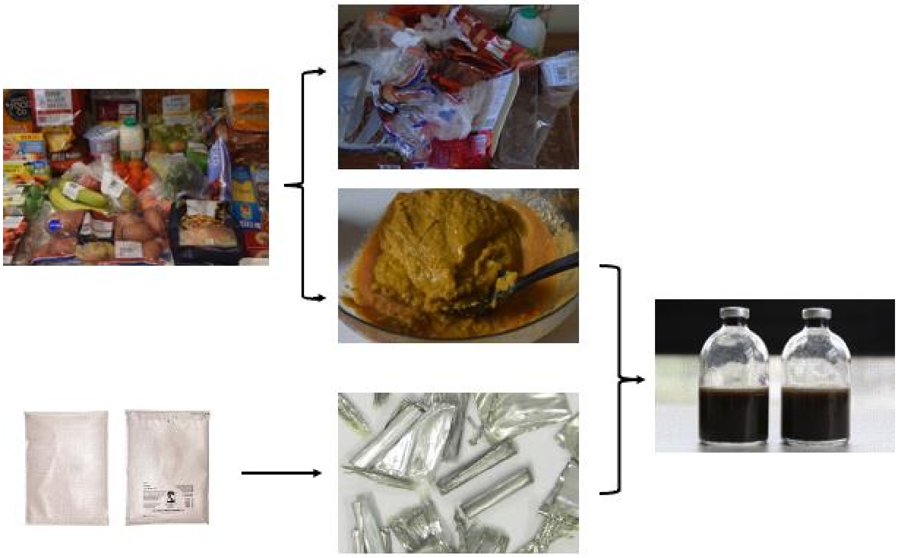


Figure 16 | Photographic summary of substrate preparation. Image sources: bottom centre, Zhang et al., 2019a; all others, author.

3.3. Co-digestion batch trials

3.3.1. Substrate and inoculum preparation

The sludge inoculum was sourced from a local commercial AD plant dedicated to household FW treatment in Oxfordshire, UK and stored at 4°C until inoculation took place on the same day. Total solid (TS) and volatile solid (VS) contents (**Table 12**) were determined following standards analytical procedures (e.g. Peces, Astals & Mata-Alvarez, 2014). Drying was performed in a laboratory oven at 105°C and ignition at 550°C.

The inoculum to substrate ratio (I:S) was set to 2:1 on a VS basis. 150 ml of sludge inoculum were poured into 250 ml glass bottles, followed by SFW and one of the polymers. Cellulose (Sigma Ltd, UK) was used as basic sludge activity control on a 0.5 g per 30 ml basis. Pieces of individual polymer types were added to the SFW and sludge inoculum on a 0.2%, 0.5%, 1%, 2% and 5% w/w of SFW, the 5% cap reflecting the maximum packaging content displayed in **Table 11**. Each experimental condition (including individual polymer categories and plastic concentrations) was prepared in triplicates. The resulting 51 bottles were sealed using a rubber stopper and Parafilm M (Bemis, US) and put on a shaking incubator at 100 revolutions per minute (rpm) at 37°C for 35 days. The UV conditions were not recorded.

Table 12 | Total solids and volatile solids of sludge inoculum, synthetic food waste and polymers used in co-digestion batch experiments. SFW: synthetic food waste; BioN: natural polybutylene succinate adipate (PBSA)-based biopolymer; BioP: dyed PBSA-based biopolymer; PET: polyethylene terephthalate; TS: total solids; VS: volatile solids. TS and VS contents were averaged from triplicates.

Component	%TS	%VS (of TS)	Volume (ml) or Mass (g)	VS (g)	Total VS Added (g)
Inoculum	6.18	67.96	150	6.30	6.30
SFW	29.95	96.70	10.87	3.15	9.45
BioN	97.07	99.13	0.02	0.02	9.47
			0.05	0.05	9.50
			0.11	0.10	9.55
			0.22	0.21	9.66
			0.54	0.52	9.97
BioP	69.82	99.59	0.02	0.02	9.47
			0.05	0.04	9.49
			0.11	0.08	9.53
			0.22	0.15	9.60
			0.54	0.38	9.83
PET	56.29	99.71	0.02	0.01	9.46
			0.05	0.03	9.48
			0.11	0.06	9.51
			0.22	0.12	9.57
			0.54	0.31	9.76

3.3.2. Physical and chemical parameters sampling and characterisation

Biogas production and pH were measured on the first 5 days following incubation and every 4-7 days after that until the end of the experimental period (35 days). Biogas was collected in an inverted graduated cylinder inside a water tank and measured through water displacement (e.g. Narancic et al., 2018) and recorded manually. The relative concentrations of CH_4, CO_2 and dinitrogen (N_2) were obtained by sampling 1 ml from the headspace of each bottle and run through a gas chromatograph and TotalChrom software (PerkinElmer, 2004). Digestate pH measurements were measured using a glass electrode and meter calibrated in buffers at pH 4, 7 and 9.

3.3.3. Data analysis

Data analysis was undertaken in R 4.1.1 (R Core Team, 2021) via RStudio (https://www.rstudio.com). Analysis of variance (ANOVA) was used as statistical approach to investigate gas production and yield, based on treatment means. Given that the same bottles were sampled over the trial period, a repeated-measures two-way ANOVA was initially used to compare means of groups classified by two types of factor variables: a between-subjects factor, which have independent categories (i.e. treatment condition), and within-subjects factor, which have related categories, or repeated measures (i.e. sampling day). A simpler one-way ANOVA was then used to investigate overall treatment effect and is reported below.

3.4. DNA extraction, sequencing and analysis

3.4.1. Sample DNA extraction and sequencing

Five samples were collected for microbial analysis. These were:

- SFW only (mono-digestion) on day 0 (start of incubation period)
- SFW only on day 35 (end of incubation period)
- SFW + undyed biopolymer (BioN) on day 35
- SFW + dyed biopolymer (BioP) on day 35
- SFW + undyed conventional polymer (PET) on day 35

The microbial DNA extraction for each sample was carried out by using the Dneasy PowerSoil kit (Qiagen, Germany). Samples were amplified through polymerase chain reaction (PCR) using the (forward and reverse) primer pairs 341F (CCTAYGGGRBGCASCAG) and 806R (GGACTACNNGGGTATCTAAT) targeting the hypervariable V3-V4 region of bacterial and archaeal 16S ribosomal RNA (rRNA) genes (Bahram et al., 2019). Sequencing was performed through the Illumina HiSeq PE250 high-throughput sequencing system (Illumina, US).

3.4.2. Bioinformatics pipeline

Sequenced libraries were quality checked and trimmed including primer sequence removal using the DADA2 algorithm. The trimmed reads were then merged, constructing an amplicon sequence variant (ASV) table from which chimeric sequences were removed. Taxonomy was assigned to the ASVs using a naïve Bayesian classifier method, based on SILVA 138.1 as the reference database (Quast et al., 2012). Further visualisation and statistical analysis of the bioinformatics output was conducted using the `phyloseq` R package (McMurdie & Holmes, 2013) in R 4.1.1 (R Core Team, 2021) via RStudio. Microbial community composition was visualised using stacked bar plots. Alpha diversity was assessed based on Chao1 richness and Shannon-Weaver diversity indices. Beta diversity was investigated using principal coordinate analysis (PcoA) derived from Euclidian distances of the provided distance matrix, providing insights into how the samples relate to each other.

4. Results & Discussion

4.1. Biogas and methane yields

4.1.1. Cumulative biogas production and biogas yields

First, cumulative biogas production over the incubation period (35 days) was quantified (**Figure 17**). Experiments yielded between 2.74 l and 2.94 l of biogas (excluding sludge activity control with cellulose, orange line). All polymer co-digestion treatments were above the baseline (SFW only, black dashed line in bold). The highest biogas production was observed from the printed biopolymer (BioP) at 0.5% w/w. Both BBP incubations yielded over 2.80 l, regardless of concentration, while all PET treatments were below 2.80 l, but still above the baseline, and none of the cumulative biogas production after 35 days differed significantly from the baseline across all treatments ($F(14, 27) = 1.120$, $p = 0.378$).

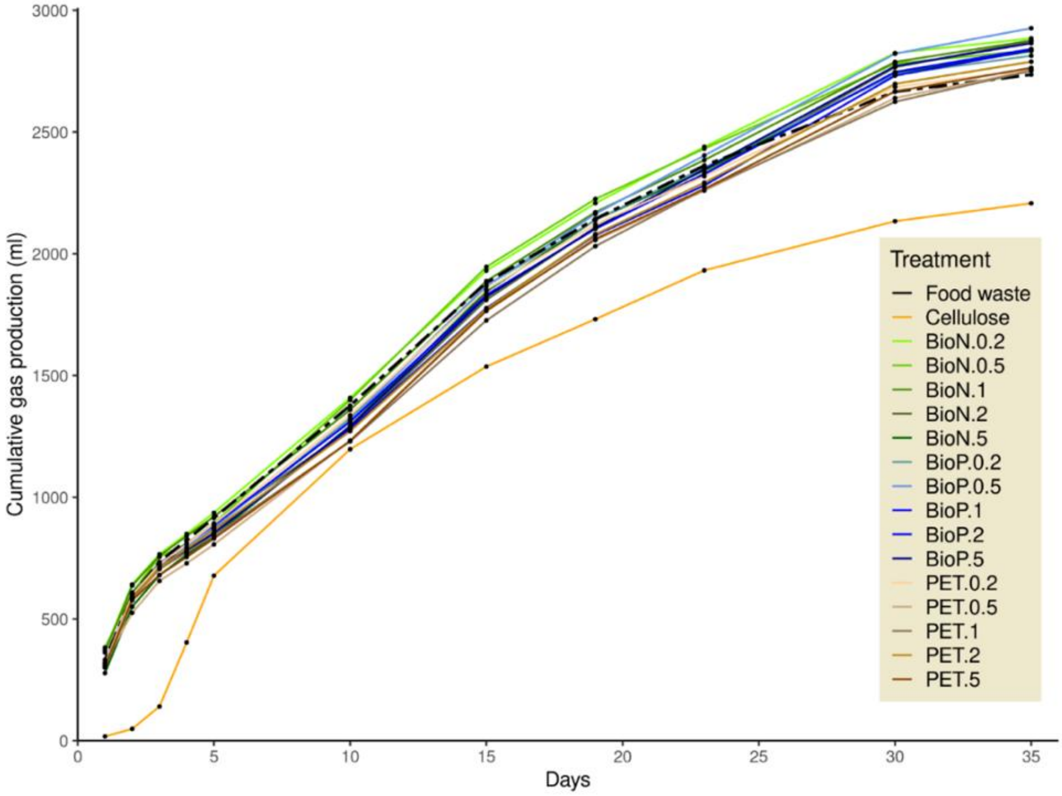


Figure 17 | Cumulative biogas production over experimental run. Cellulose treatment (orange line) corresponds to sludge activity control and was not included in statistical analysis. BioN: natural polybutylene succinate adipate (PBSA)-based biopolymer; BioP: dyed PBSA-based biopolymer; PET: polyethylene terephthalate. Numbers in the legend correspond to the weight of polymer added relative to the weight of synthetic food waste (w/w) added in percentage (e.g. PET.5 corresponds to 5% w/w of PET). Each data point represents the arithmetic mean of triplicates. Error bars are not shown to facilitate visualisation.

Plotting the biogas yields (i.e. adjusting the volume of biogas produced per weight of total VS inoculated) did not alter the general trends (**Figure 18**). Total cumulative biogas yields ranged from 238.18 l/kg VS (PET at 5% w/w) to 308.39 l/kg VS (BioP at 0.5% w/w), with a baseline average of 289.52 l/kg VS, consistent with typical values for the substrate used (Zhang, Banks & Heaven, 2012). Nevertheless, none of the treatments differed significantly from each other

($F(14, 27) = 1.600$, $p = 0.378$). These results suggest that at relatively low concentrations (0.1-5% w/w), the presence of plastic fragments (biodegradable or not) does not affect biogas production and biogas yields in mesophilic co-AD over 35 days.

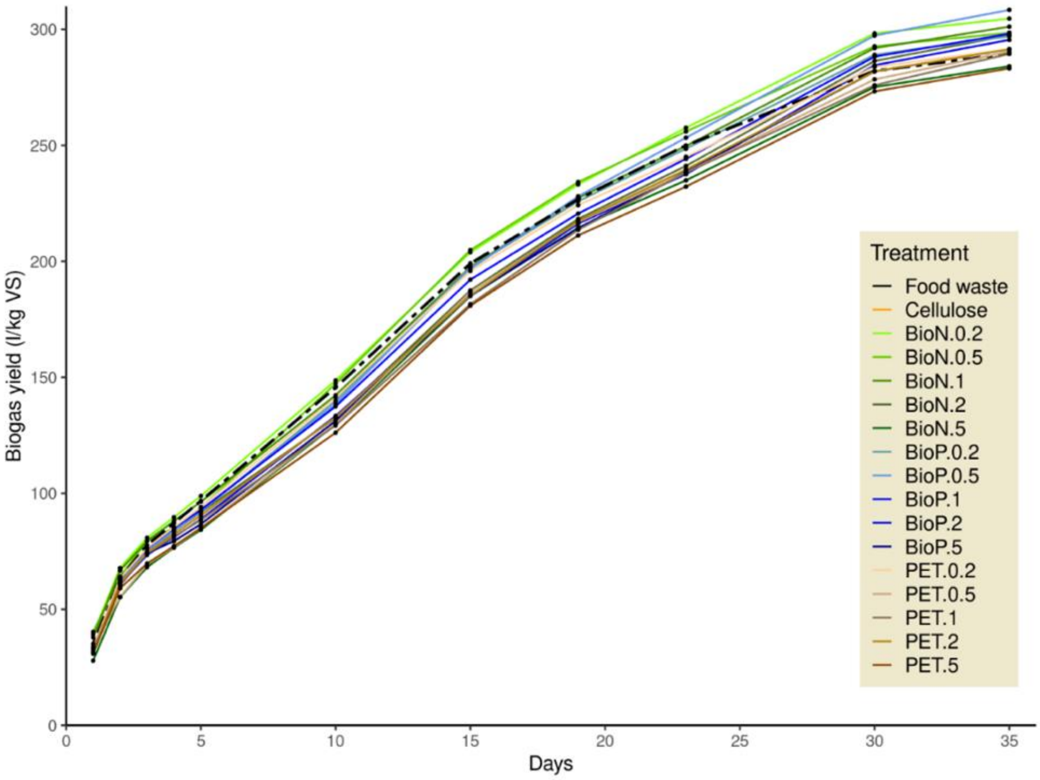


Figure 18 | Cumulative biogas yields over experimental run. Cellulose data is not displayed as the total solids and volatile solids were not determined.

4.1.2. Cumulative methane yields

While general biogas volumetric quantification provides a broad overview of co-AD performance, biodegradability assays tend to focus on the conversion of organic matter into CH_4 (BMP tests). The proportion of CH_4 in biogas will reflect the quality of biogas and will determine its heating value (IEA, 2020). To be used as renewable natural gas, biogas needs to be upgraded, whereby CO_2 and other contaminants present in the biogas are removed to produce biomethane, a near-pure source of CH_4. Thus, the higher the production of CH_4 per unit of VS added (i.e. the CH_4 yield), the more biomethane can be generated per unit of biogas produced and the more economically attractive a given AD system is.

Gas chromatography allowed to quantify the relative gas composition on days 1, 2, 4, 5, 10 and 30 and derive CH_4 yields (**Figure 19**). Similar to biogas yields, total cumulative CH_4 yields did not differ significantly across experimental conditions ($F(14, 27) = 1.281$, $p = 0.281$), despite a spike observed for the BioP at 1% after 10 days of incubation (lapiz blue line). Final cumulative CH_4 yields ranged from 149.29 l CH_4/kg VS (BioN 5%) to 168.06 l CH_4/kg VS (BioP 0.2%), with a baseline of 163.53 l CH_4/kg VS.

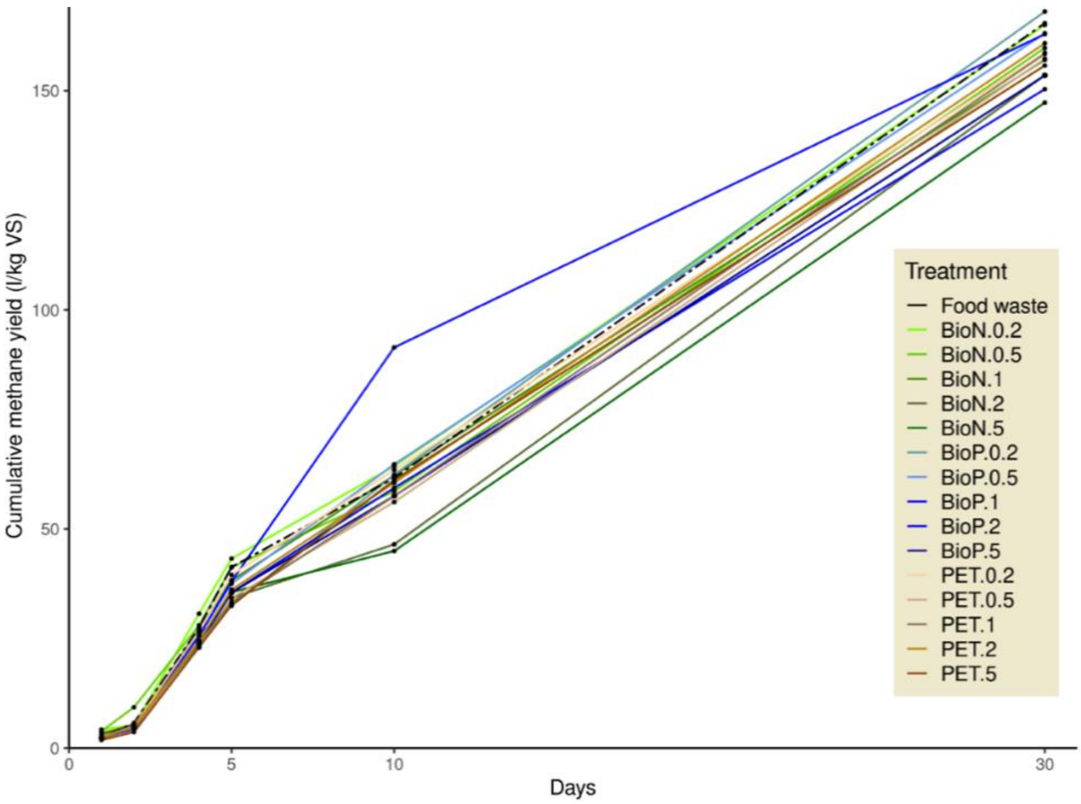


Figure 19 | Cumulative methane yields over experimental run. Cellulose data is not displayed as the total solids and volatile solids were not determined.

Together, these results suggest that the presence of both conventional and biodegradable plastic fragments at concentrations ranging from 0.2 to 5% w/w (relative to SFW) does not hinder biogas and CH_4 production or yields. Given the relatively low BBP to SFW ratios used in this study, a higher biogas production/yield would be unexpected, unless a much higher plastics loading than is likely to occur in a real mixed FW stream had been used (Zhang, Heaven & Banks, 2018).

The presence of additives such as dyes and plasticisers in composite materials can have an impact even on the most biodegradable BBPs (Battista, Frison & Bolzonella, 2021; Vardar, Demirel & Onay, 2022). For this reason, both printed (BioP) and natural (BioN) polymers were investigated in this study, but no discrepancies were found between the two conditions, regardless of concentration ($p > 0.05$ across all pairwise comparisons). While this is encouraging, without a physical and chemical characterisation of polymer samples before and after incubation (e.g. through microscopy analysis and Fourier-transform infrared spectroscopy (FTIR)), polymer biodegradation cannot be formally assessed (Quecholac-Piña et al., 2020).

Unfortunately, polymer fragmentation and potential biodegradation could not be investigated (see **Note to the reader**). Thus, the possibility that the absence of discrepancies in CH_4 yields across treatment conditions was due to a lack of polymer biodegradation cannot be ruled out and is, in fact, highly likely. Lu et al. (2022) found limited morphological changes in PLA-based fragments at mesophilic conditions. Furthermore, in a co-digestion study with SFW, Zhang, Heaven & Banks (2018) found that despite displaying substantial anaerobic biodegradability under mesophilic conditions, none of the EN 13432 certified BBPs met the criteria for physical contaminants under the UK publicly available specification for digestate quality (PAS 110), an industry standard for the use of digestate on agricultural land. This means that while BBPs may indeed undergo some level of biodegradation, they do not break down into sufficiently small particles for those to be no longer considered as contaminants. This poses challenges for industrial AD plants and the commercial viability of the digestate output, which may not be deemed of sufficient quality to receive PAS 110 certification, not to mention its associated environmental impacts (MacLeod et al., 2021).

4.2. Microbial characterisation

4.2.1. Diversity characterisation

Next, potential impacts of polymer fragments on the microbial community were investigated. Inoculum and SFW samples on days 1 and 35 (SFW_i and SFW_f, respectively) as well as samples with the highest polymer concentration on day 35 (i.e. BioN 5%, BioP 5% and PET 5% w/w) were chosen for DNA extraction, sequencing and subsequent bioinformatics analysis based on 16S rRNA gene identification (V3-V4 hypervariable region).

First, within-sample diversity was investigated through Chao1 and Shannon alpha diversity indices. Chao1 estimates how many individual species there are in each sample (richness), taking rare species into account and based on how many species are represented by a single individual (or, in this context, a single read belonging to a given amplicon sequence variant, or ASV) or two (Chao, 1984). Shannon (sometimes referred to as Shannon-Wiener) combines both combines both richness and the inequality between species representation (abundance) (Shannon & Wiener, 1963). A large Shannon value indicates the presence of many species with balanced abundances, and values can range from 1 (a single dominant species) to the total number of all species (if all species are represented in equal proportions) (Shannon & Wiener, 1963).

Alpha diversity indices SFW_i differed most markedly from the other samples (**Figure 20**) and scored highest for both alpha diversity indices (Chao1 = 273, Shannon = 2.81). In contrast, SFW_f showed a relatively lower alpha diversity (Chao1 = 198, Shannon = 2.12). Nevertheless, in absolute terms discrepancies between all five samples remained low.

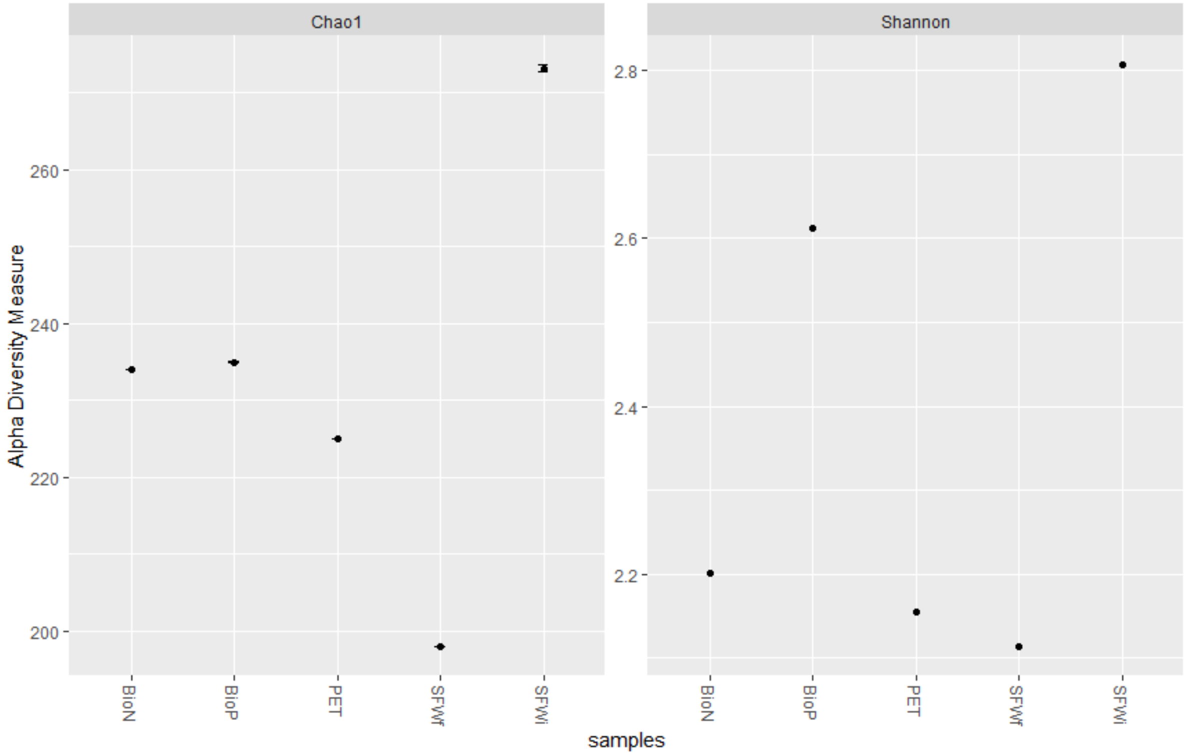


Figure 20 | Alpha diversity measures for sequenced mono- and co-digestion samples. Chao1 is a diversity index based on richness, which estimates the number of individual species in a community. Shannon-Wiener (Shannon in short) combines both the number of species (richness) and the inequality between species representation (abundance). BioN: BioN 5% w/w, BioP: BioP 5% w/w, PET: PET 5% w/w, SFW_i: synthetic food waste-only on day 1; SFW_f: synthetic food waste-only on day 35.

Next, between-sample diversity was investigated through PcoA. PcoA revealed that both baseline conditions differed from any other (**Figure 21**), while samples containing plastic fragments were more closely related, particularly for BBPs.

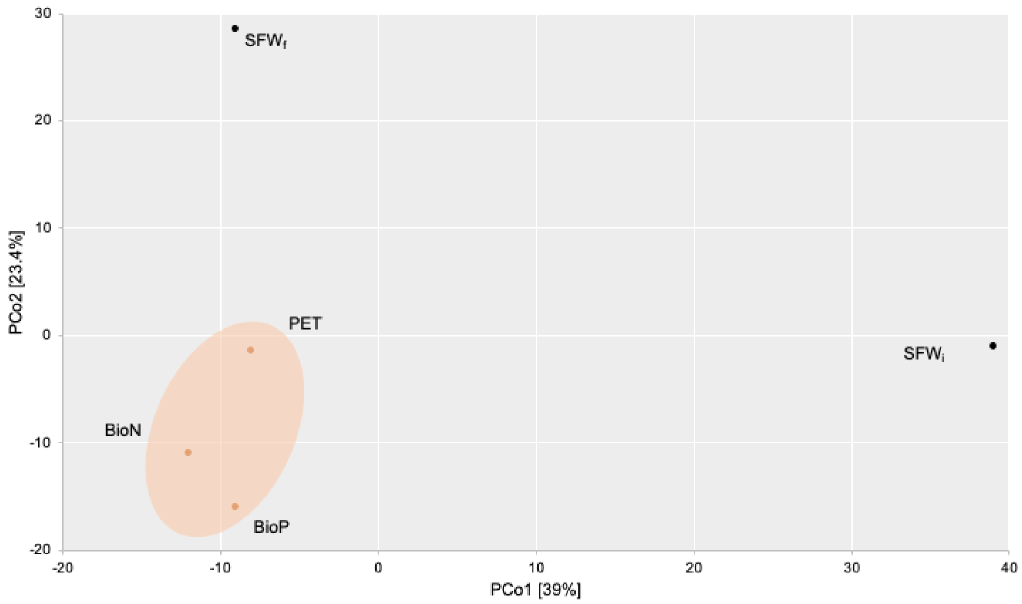


Figure 21 | Principal coordinate analysis (PCoA) for sequenced mono- and co-digestion samples. Principal coordinates 1 (PCo1) and 2 (PCo2) accounted for 39% of variation among samples, respectively.

4.2.2. Microbial taxonomic characterisation

Table 13 shows the proportion of reads belonging to bacterial and archaeal kingdoms. Despite evidence that the 341F-806rB primer pair, which covers the hypervariable V3-V4 region of the small ribosomal RNA (16S rRNA) gene, is suitable for both Bacteria and Archaea (Bahram et al., 2019), Archaea were poorly represented, and samples showed a strong bias towards Bacteria. This trend was previously observed in the 515F-806rB primer pair covering the V4-V5 region (Bahram et al., 2019), which was shown to discriminate against both *Crenarchaeota* and *Thaumarchaeota* (or *Nitrososphaerota*), two important environmental archaeal phyla (Walters et al., 2016). Given that the V4 and adjacent regions have been popular targets for bacterial metabarcoding (Bahram et al., 2019) and that V1-V2 regions of the 16S rRNA gene provide the greatest resolution among archaeal taxa (Hartmann et al., 2010), primer bias may have played a role in their relative representation. The thermophilic tendency among Archaea, at least from an evolutionary perspective, may also explain why the majority the 16S rRNA gene reads belonged to bacterial taxa, which are more widely adapted to mesophilic environments (López-García et al., 2022).

Table 13 | Proportion of 16S rRNA gene reads belonging to Bacteria and Archaea.

Treatment	Bacteria (%)	Archaea (%)
SFWi	99.87	0.13
SFWf	98.56	1.44
BioN (5%)	97.63	2.37
BioP (5%)	96.56	3.44
PET (5%)	98.37	1.63
All-sample average	98.20	1.80

At the phylum level, *Firmicutes* dominated the microbial community across all samples (62.99-81.56%), followed by *Cloacimonadota* (9.44-26.26%) and *Bacteroidota* (4.65-10.00%) (**Table 14 & Figure 22**).

Table 14 | Top six bacterial phyla recovered from 16S rRNA gene reads. Values are in percentage of total read count per sample.

Major Phyla	SFW_i	SFW_f	BioN (5%)	BioP (5%)	PET
Firmicutes	81.56	63.88	67.59	62.99	70.45
Cloacimonadota	9.44	26.26	19.69	16.62	16.83
Bacteroidota	5.27	4.65	7.30	10.00	5.59
Thermotogota	0.48	2.01	1.85	3.21	2.27
Actinobacteriota	1.34	1.74	0.95	3.30	2.19
Synergistota	0.40	0.33	0.70	0.95	1.08

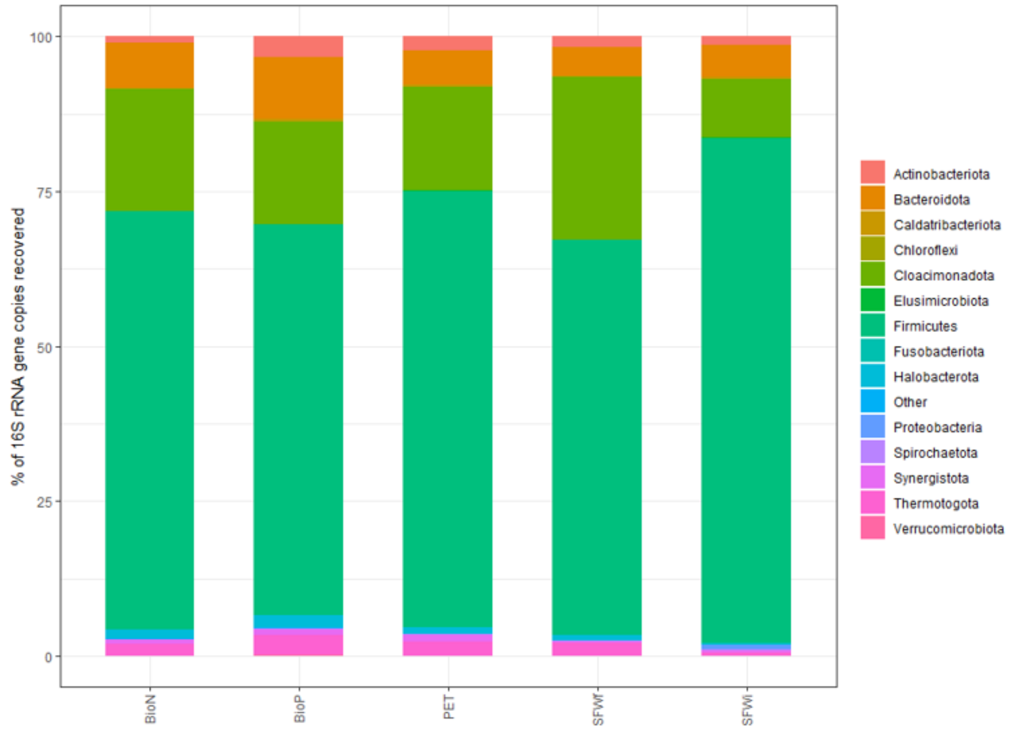


Figure 22 | Taxonomic profiles at the phylum level. Values are in percentage of total read count per sample.

Bacterial classification at the genus level showed more pronounced differences (**Table 15** & **Figure 23**). *Fastidiosipila* dominated the community, accounting for over half of total reads among all samples apart from SFW_i (48.31%), in which it still represented, however, the most abundant genus. *Gallicola* was the second most represented genus (6.71-14.03%), although once again SFW_i differed from the other treatment conditions, in that it reported a higher abundance of the genus *Clostridium stricto 1* than of *Gallicola.* Furthermore, while *Clostridium stricto 1* accounted for 16.49% of the total read count in SFW_i, the same genus was significantly less represented in all other samples (> 1%). *W5053*, *Acetomicrobium* and *Caldicoprobacter* were also represented in low percentages across all samples (0.4-3%). All most common genera recovered belonged to the phylum *Firmicutes*, the class *Clostridia* and the order *Eubacteriales*, apart from *Caldicoprobacter*, from the phylum *Synergistota*, the class *Synergistia* and the order *Synergistales*.

Table 15 | Top six bacterial phyla recovered from 16S rRNA gene reads. Values are in percentage of total read count per sample.

MAJOR GENERA	SFW_i	SFW_f	BioN (5%)	BioP (5%)	PET
Fastidiosipila	48.31	70.87	71.05	59.69	71.61
Gallicola	14.03	12.65	6.71	11.61	7.86
Clostridium stricto 1	16.49	0.46	0.57	0.45	0.72
W5053	1.82	0.82	1.29	2.85	1.25
Acetomicrobium	0.55	0.52	1.00	1.50	1.48
Caldicoprobacter	0.82	0.37	0.47	0.62	0.40

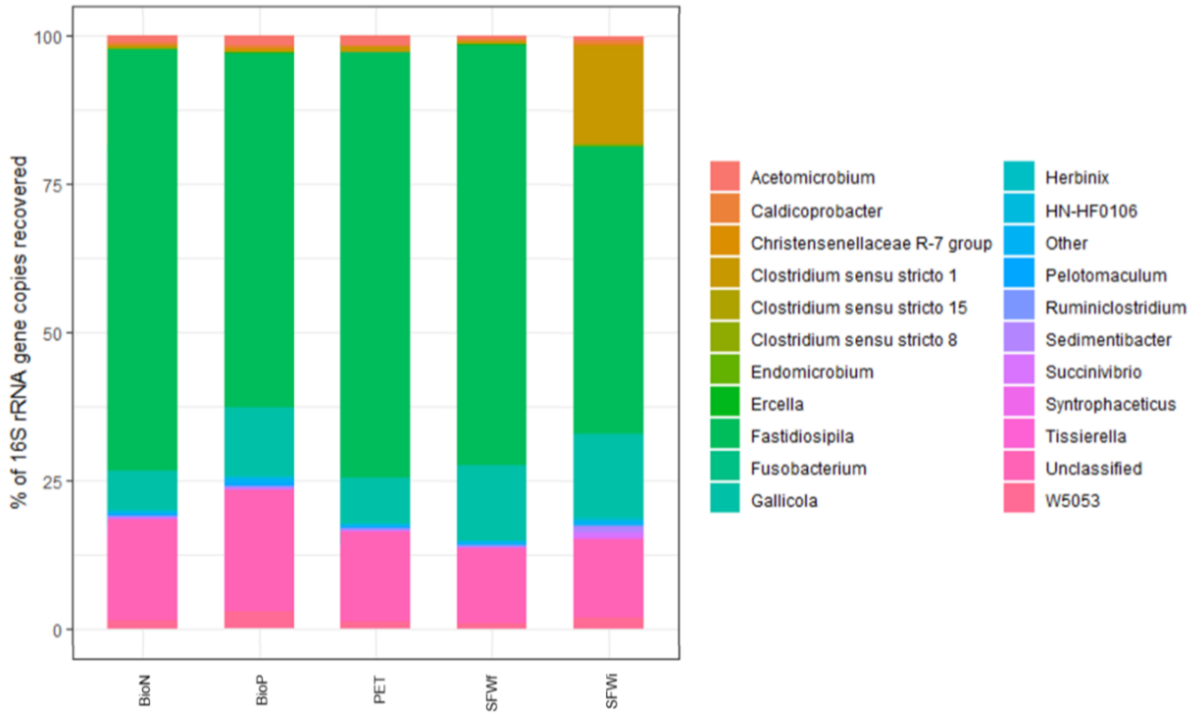


Figure 23 |Taxonomic profiles at the genus level. Values are in percentage of total read count per sample.

Findings from microbial (predominantly bacterial) 16S rRNA characterisation suggest that microbial communities remain unaffected by the presence of both conventional and biodegradable plastic fragments, with the biggest (although still relatively minor) discrepancies observed between SFW_i and the remaining samples. Once again, given the relatively low plastics loadings in the experimental design, a noticeable shift in microbial communities was unlikely to occur (Peng et al., 2022), unless the polymers studied exhibited toxic properties. The microbial profiles seem to indicate that the inoculation of SFW, a major source of easily biodegradable substrate, drove the shift in the microbial community, rather than either conventional of biodegradable polymer fragments (Peng et al., 2022).

Echoing this substrate bias, microorganisms grown in the presence of simple sugars and a more recalcitrant substrate have shown decreased expression of carbohydrate active enzymes (CAZymes) – which initiate the hydrolysis step – compared to those grown on the latter substrate only (Blair, Dickson & O'Malley, 2021). Nevertheless, PBSA (which BioN and BioP are partially based from) was shown to have a unique microbial niche distinct from the microbiome of surrounding soils (Puharong et al., 2021). In a separate study, based on morphological and chemical analysis, degradation of PBSA was assumed to depend on bulk erosion, with a change to internal chemical bonds (Jin et al., 2022). The prolonged exposure to the biopolymer and the absence of a readily available alternative energy source may have contributed to the formation of a specialised local niche in the vicinity of PBSA samples. Intriguingly, in a short-incubation study on low-density PE (LDPE) degradation, a putative plastic-specific microbial profile emerged within two days of incubation and was no longer distinguishable after nine days (Erni-Cassola et al., 2020), suggesting that such microbes are quickly outnumbered as the community matures.

The dominance of the phylum *Firmicutes* aligns with the literature (Lear et al., 2021; Lu et al., 2022; Patil et al., 2021; Peng et al., 2022). While distinct microbial ‘plastispheres’ were not found in this study, spikes in *Firmicutes* have been observed following the introduction of BBPs, suggesting that this phylum is well adapted to such substrates (Lear et al., 2021; Peng et al., 2022). The most abundant bacterial genera identified in this study all belong to the order *Clostridiales*, which plays an important role in hydrolysis and fermentation in AD (De Vrieze et al., 2015). The presence of members of the *Clostridiaceae 1* family has been reported in FW AD (Zhang et al., 2019b); this family can metabolise a range of substrates to produce volatile fatty acids. Jin et al. (2022) reported several orders, including *Clostridiales* and *Synergistales*, as key players in BBP biodegradation under mesophilic conditions. Members of these bacterial orders initiate the degradation process in hydrolysis by producing CAZymes (Blair, Dickson & O’Malley, 2021).

Several limitations may restrict the how much can be inferred from microbial characterisation. First, due to budgetary constraints, a single sample per treatment condition was used for sequencing, thereby preventing any subsequent statistical analysis. Microbial characterisation was performed through 16S rRNA metabarcoding, which targets only bacterial and archaeal kingdoms and excludes fungal species, which rely on separate metabarcoding targeting the internal transcribed spacer (ITS) region from the eukaryotic nuclear rRNA cistron (Schoch et al., 2012). Lastly, it is also possible that some of the unclassified bacterial and archaeal species play a role in plastic-specific metabolism, due to the abundance of microorganisms that have yet to be characterised, as well as the difficulty of characterising rare species (Blair, Dickson & O’Malley, 2021).

4.3. Towards an integrated design for anaerobic co-digestion

While the results of this experimental study on BBP and SFW co-AD suggest that the introduction of BBPs does not present any detrimental impact on biogas and CH_4 yields or on microbial consortia, their limited ability to provide a detailed picture of how BBPs behave in the system (e.g. by characterising morphological and chemical changes in surface structure) also points at the need for the systematic development and application of adequate research design and methods. Indeed, while reviewing the literature to inform the design of the experimental trials presented in this chapter, it became apparent that limited consideration was given to the relevance of study design beyond its scientific discipline.

This led to a questioning of current experimental design in scientific studies addressing BBP and FW co-AD and their relevance to full-scale AD design and operation. A rapid review was conducted, and relevant study design parameters for BBP co-AD in the context of the treatment of OFMSW were investigated.

So far, only a limited number of studies have addressed FW and BBP co-AD. **Table 16** provides an overview of relevant studies. While technical aspects of co-AD are discussed, it is not the ambition of this section (and of **Chapter 4** more broadly) to provide a comprehensive review of the underlying biochemical details, which has been the subject of dedicated reviews (Bátori et al., 2018; Stagner, 2015; Quecholac-Piña et al., 2020). Instead, it aims to frame the technical challenges faced by BBP and FW co-AD within the wider sustainability context, with an emphasis on on-the-ground organic waste management practices, which is often missing when taking a siloed approach.

Table 16 | Summary of studies on biodegradable bioplastics and food waste anaerobic co-digestion. BBP: Biodegradable bioplastic; FW; Food waste; AD: Anaerobic digestion; PHA: Polyhydroxyalkanoate; PLA: Polylactic acid; HDPE: High-density polyethylene; PP: Polypropylene; PS: Polystyrene; VS: Volatile solid; COD: Chemical oxygen demand; CSTR: Continuous stirred tank reactor; SBR: Semi-batch reactor; M: Mesophilic (35-37°C); T: Thermophilic (55°C), hT: Hyperthermophilic (80°C); §Not directly FW but sludge inoculum obtained from an AD plant running on the organic fraction of municipal solid waste; †Not BBPs but conventional plastics; *Associated data article supporting primary research article. Adapted from Kakadellis, Lee & Harris, 2022.

#	First Author	Year	Basis	FW:BBP	BBP Tested	BBP Size	HRT (days)	Reactor	°C	Findings
1	Wang	2012	COD	4:1	PLA bags	2×2 mm^2	25-41 (run over 592)	CSTR	T & hT	Introducing a hyperthermophilic treatment improved the overall co-AD performance. The microbial communities in both conditions were dominated by similar genera.
2	Gómez	2013	Weight	15:2	Range of BBP films	10×10 mm^2	50	Batch	M	Materials have different degradation rates under different end-of-life scenarios. Most BBPs biodegraded only to a limited extent under AD.
3	Vasmara	2016	Volume/weight	50 ml: 1 g	Mater-Bi (starch) bags & PLA cups	10×10 mm^2	98	Batch	M & T	Synergistic effect of BBP co-digestion with pig slurry or cheese whey on CH_4 and H_2 yields. Thermophilic treatment further increased yields by roughly 50%.
4	Lim	2018	VS	2:1	HDPE bags, PS boxes, & PP trays†	10×10 & 5x5 mm^2	30-35	Batch	M	Plastics inhibited CH_4 production. PS and PP were found to inhibit CH_4 production from FW more than HDPE. Inhibition was more likely due substrate competition, which intensified with increased plastic surface area.
5	Narancic	2018	Weight	200:3§	Range of BBP films	25×4 mm^2	Up to 127	Batch	T	The majority of the tested BBPs and their blends degraded by thermophilic AD with high biogas output, but degradation times were 3–6 times longer than the retention times in commercial AD plants.
6 & 6*	Zhang	2018/2019	VS	4:1	PLA-, starch-, cellulose-based films & PLA blend pellets	10×10 mm^2	147	Batch & SBR	M	Of the 9 BBPs tested, only 4 showed substantial biodegradability under AD conditions. Even the most degradable materials would not break down sufficiently to meet the physical contaminant criteria of the UK PAS 110 specification standard for anaerobically digested material, if fed to a digester at 2% of the input load on a VS basis.

#	First Author	Year	Basis	FW:BBP	BBP Tested	BBP Size	HRT (days)	Reactor	°C	Findings
7	Hobbs	2019	Volume/weight	200 ml: 1.1 g	PLA bags	20×20 mm^2	70	Batch	M	Alkaline pre-treatment with sodium hydroxide (NaOH) has the greatest solid reduction of PLA and maximum CH_4 production.
8	Samitthi-wetcha-rong	2019	VS	1:1	PLA film	20×20 mm^2	60	CSTR	M	The results found that NaOH concentration and reaction time were two main parameters influencing PLA degradation. Optimum pre-treatment conditions were at 0.5 M NaOH and at a temperature of 60°C over 24 hours.
9	Bandini	2020	Weight	From 10^4:1 to 65:1	PLA bottles & Starch-based bags	20×20 mm^2	23	CSTR	T	Lack of degradation for PLA bottles, while starch-based bags achieved significant disintegration. Phytotoxicity test on compost (for aerobic conditions) revealed negative effects on seed germination for PLA.
10	Cucina	2021	Weight	10:1	Starch-based plastic bags & PLA tableware	50×50 mm^2	35	Not speci-fied	M	Starch-based BBPs were shown to degrade faster than PLA, but both types showed limited degradability in co-AD and subsequent composting of the digestate. The resulting compost yielded a relatively high BBP content, which did not meet current regulatory requirements. Intrinsic biodegradability in soil was demonstrated (though very slow).
11	Hedge	2021	VS	1:1	PHA film	10×10 mm^2	60	Batch	M & T	PHA co-AD with FW significantly improved the consistency of degradation under thermophilic conditions and slightly increased PHA degradation, which reached 43% over 60 days. Mesophilic conditions enabled PHA degradation to reach 70%.

#	First Author	Year	Basis	FW:BBP	BBP Tested	BBP Size	HRT (days)	Reactor	°C	Findings
12	Cucina	2022	Volume/weight	300 ml: 3 g	Starch-based bags and PLA-based crockery	25×25 mm^2	90	Batch & full-scale	T	PLA-based crockery showed significant CH_4 conversion under anaerobic conditions, while starch-based bags resulted in limited degradation only. In a full-scale AD reactor, PLA-based cutlery degraded by 70% on a weight basis, followed by PLA-based dishes (60%) and starch-based bags (39%). Post-AD composting enhanced biodegradation.
13	Cucina	2022	Volume/weight	300 ml: 3 g	Starch-based bags and PLA-based crockery	25×25 mm^2	60	Batch & full-scale	T & M	Similar experimental design to **12** but within a shorter time frame and followed by mesophilic maturation. BBPs tested showed an average degradation of 22-32% on a weight basis, with starch-based bags performing better in this study. Thermophilic temperatures were needed to achieve a significant level degradation.
14	Lu	2022	Weight	Unclear: 5 g PLA /kg total TS	PLA micro-plastics from film	1.5 mm Ø	30	CSTR	M & T	The spiked BBP did not increase the CH_4 yield of kitchen waste but exhibited physical deformation and fragmentation at mesophilic and thermophilic conditions, respectively. Thermophilic incubation resulted in higher rates of biodegradation and CH_4 yields than the mesophilic one, although only 1.5% of PLA was degraded under thermophilic AD, based on CH_4 yields.
15	Peng	2022	Weight	7:1	PBAT/PLAbag	Not speci-fied	100	CSTR	M & T	The addition of the polymer to FW co-AD did not enhance overall biogas production, with no discernible degradation. Disintegrated fragments were observed under thermophilic conditions, which facilitated degradation in subsequent aerobic treatment.

4.3.1. Co-digestion feedstocks

Combining feedstocks with different compositions has been promoted as a means of enhancing process stability and efficiency through the equilibration of the nutrient balance, particularly when dealing with complex types of substrates, such as manure or FW, rich in nitrogen. Yet, this question can only be addressed if the relevant co-substrates are being investigated and further characterisation of BBP degradation under co-AD conditions with FW is needed to expand upon recent efforts in this field.

Commercial or household FW used as co-substrate in co-AD studies includes FW from university canteens or catering (Hedge et al., 2021; Hobbs et al., 2019; Lim et al., 2018; Lu et al., 2022; Peng et al., 2022; Samitthiwetcharong & Chavalparit, 2019), food markets (Bandini et al., 2020; Samitthiwetcharong & Chavalparit, 2019), artificial household FW i.e. SFW (Wang et al., 2012; Zhang, Heaven & Banks, 2018; Zhang et al., 2019a), OFMSW (Cucina et al., 2021; 2022a; 2022b; Gómez et al., 2013; Narancic et al., 2018) and industrial food processing waste (Vasmara & Marchetti, 2016). However, FW can be a challenging co-substrate to study, due to its high nitrogen content and its heterogeneity (Okoro-Shekwaga et al., 2020), especially for municipal and household FW. Designing synthetic municipal/household FW recipes for research purposes, representative of a given geographical and societal context, can help towards more consistent system characteristics, thereby strengthening the reliability and reproducibility of the data (Wang et al., 2012; Zhang, Heaven & Banks, 2018). Robust experimental design should also consider the compatibility of the microbial sludge inoculum with the incoming substrate. Some studies used sludge from wastewater treatment plants (Cucina et al., 2022a; 2022b; Gómez et al., 2013; Hobbs et al., 2019), despite using FW as substrate. Nevertheless, one study chose sludge from palm oil mill effluent rather than from a wastewater treatment plant because of proven more consistent data with FW as a substrate (Lim et al., 2018), arguably because of the more constant characteristics of the palm oil-derived sludge since the plant treated a specialised substrate (Lim et al., 2018). When using inocula developed in laboratory environments, feeding for a prolonged period is equally important to ensure the microbial consortium can adapt to its substrate (Wang et al., 2012; Zhang, Heaven & Banks, 2018).

4.3.2. Feedstock ratios

Co-substrate ratios are often determined on a VS basis (**Table 16**). Nevertheless, they also need to reflect current and projected rates of plastic packaging in the organic waste stream. For example, if using an FW-to-BBP VS ratio of 1:1, 2:1 and 4:1 (Lim et al., 2018; Samitthiwetcharong & Chavalparit, 2019; Zhang, Heaven & Banks, 2018), the resulting plastic content (by weight) would roughly correspond to 30%, 15% and 7.5%, respectively (based on average TS and VS characteristics of both substrates). Yet, currently, total plastic content is estimated to account for 2-5% of household FW (Zhang et al., 2014) and BBPs are expected to account for up to 7% of the OFMSW in the coming years (Cucina et al., 2021). Studies need to reflect that, especially when assessing the potential for CH_4 yield enhancement, as this can be an attractive selling point for AD plant operators, but which needs to be realistic.

In some cases, existing biogas and biomethane plants operate below capacity due to lack of feedstock availability, with detrimental effects on operational costs (ADBA, 2020). While diversifying sources of organic material is needed, the role of feedstock ratios on process performance should be carefully monitored. Ammonia (NH_3) – or ammonium (NH_4^+), its ionised form – is produced through biological degradation of nitrogenous matter; a high concentration of NH_3 leads to the accumulation of volatile fatty acids, which inhibits methanogenesis, resulting in low CH_4 yields (Shi et al., 2017). Based on the chemical composition of BBPs and FW, a theoretical co-digestion ratio can be easily determined to achieve an optimal C:N ratio for efficient CH_4 production (around 20-30:1) (Narancic et al., 2018). How this then translates into practical terms is an important consideration, both in terms of actual FW and BBP proportions in waste arisings and the level of BBPs tolerated by the system to ensure biodegradation and meet the quality standards of the resulting digestate (Cucina et al., 2021; Zhang, Heaven & Banks, 2018). This was directly addressed by one study (Cucina et al., 2021), in which the amount of BBPs added in the experimental assays was determined based on current and projected trends.

4.3.3. Hydraulic retention time

The hydraulic retention time (HRT), which corresponds to the average time that digester contents sit in the tank, is an important parameter to consider when assessing the real-life suitability of BBPs in AD. A number of studies have already highlighted that although some BBPs have the ability to fully biodegrade in AD, few fulfil the HRT of operating AD plants (Bátori et al., 2018; Gómez et al., 2021; Narancic et al., 2018) with degradation times 3-6-fold longer than current industrial HRT (Narancic et al., 2018). Though it is of scientific interest to run experiments for as long as biodegradation takes place, more emphasis needs to be placed on relevant HRT (Bandini et al., 2020; Bátori et al., 2018; Cucina et al., 2021), as well as digester operating mode (Wang et al., 2012). A typical biogas plant treating OFMSW operates with an HRT of 15-30 days (Bátori et al., 2018), though longer HRTs up to 100 days at commercial facilities treating source-separated FW have been reported (Angelonidi & Smith, 2015). Therefore, a BBP suitable for FW collection should be able to degrade within these timeframes. Some BBPs have been shown to biodegrade at an HRT usually applied at industrial scales, such as materials made of PHAs, starch, cellulose and pectin, so no possible contamination would occur (Bandini et al., 2020; Bátori et al., 2018), although some of these results were contested elsewhere (Cucina et al., 2021; Lu et al., 2022).

BBP biodegradation could benefit from longer retention times typically observed in wet mesophilic AD (Angelonidi & Smith, 2015; Zhang et al., 2014). Wet AD systems, in which water liquid is added or recycled to the feedstock to yield a more pumpable slurry with lower TS concentration (TS < 15%), are commonly used for the AD of organic wastes (Angelonidi & Smith, 2015; Rocamora et al., 2020), including FW (Angelonidi & Smith, 2015). The longer HRT characteristic of FW treated in wet AD systems reinforces the suitability of FW as substrate for BBP co-digestion (Zhang, Heaven & Banks, 2018). As the VS content of BBPs is high (Narancic et al., 2018), the addition of BBPs could also increase the organic loading rate, with little effect on the overall HRT.

Dry AD (typically TS ≥ 20%) may offer several advantages over wet AD due to lower water use, more favourable energy balances and a more robust system (Angelonidi & Smith, 2015; Rocamora et al., 2020). The high solids content of FW feedstocks and the presence of additional pre- and post-treatment steps in dry AD processes (Angelonidi & Smith, 2015) could make dry AD an attractive new avenue to explore for FW and BBP co-digestion, which was addressed in some recent co-AD studies (Cucina et al., 2021; Peng et al., 2022). In any case, the systematic deployment of pre-treatment steps, such as pasteurisation or autoclaving, ahead of AD, could accelerate initial hydrolysis (the rate limiting step for BBP biodegradation), thereby reducing the HRT required for effective biodegradation of further BBP materials.

4.3.4. Polymer pre-treatment

There is a growing interest in pre-treating BBP waste to enhance its biodegradability (and thus biogas and CH_4 recovery) in AD, including PLA (Battista, Frison & Bolzonella, 2021; Hobbs et al., 2019; Samitthiwetcharong & Chavalparit, 2019). A 15-day pre-treatment incubation with sodium hydroxide (NaOH) promoted PLA degradation and yielded between 97 and 99% solubilisation, effectively removing PLA aggregates left in the digestate (Hobbs et al., 2019). Nevertheless, as any additional step in the process will come at a cost (both energetic and financial) to the plant operator, it is worth asking whether such a strategy is a practical option, given that BBPs represent a minor fraction of the total feedstock stream (Cucina et al., 2021). Importantly, the use of corrosive agents to accelerate hydrolysis of BBPs could have environmental implications and result in the waste liquid from the pre-treated fraction to be classified as hazardous.

A number of studies looked at the effect of temperature on CH_4 yield and BBP degradation. Operating at thermophilic (55°C) conditions increased CH_4 yields by 51% compared to a mesophilic range (35-37°C) (Vasmara & Marchetti, 2016), although one study found no noticeable change in CH_4 yield and only observed disintegration (as opposed to degradation) of fragments of a PLA polymer blend (Peng et al., 2022). Hyperthermophilic treatment (80°C) after or before thermophilic incubation further increased CH_4 conversion and PLA transformation ratios, achieving an 80% conversion from PLA to lactic acid (Wang et al., 2012). However, mesophilic AD currently represents the most practical and financially viable system for BBP co-digestion with FW; the characteristically high water content of FW makes it costly to operate at thermophilic ranges (Angelonidi & Smith, 2015) (**Chapter 3 Section 3.2**). In addition, at thermophilic temperatures, NH_3 toxicity is increased, and the addition of trace elements is no longer effective in enabling metabolic pathway switching, so that other methods are necessary (Banks et al., 2018). The longer retention times typically observed in wet mesophilic AD would also enhance further BBP biodegradation (Angelonidi & Smith, 2015; Wang et al., 2012). In one study, mesophilic conditions were indeed found to be more favourable for PHA degradation (Hedge et al., 2021). Given that pasteurisation is often a legal requirement for AD plants treating FW, moving this step at the front end could represent a compelling switch, which would allow for a thermal pre-treatment step at no extra operational cost (a theme explored further in **Chapter 5**).

4.3.5. Polymer properties

All studies reviewed except one cut their plastics to obtain plastic fragments between 0.4 and 4 cm^2. While this is often necessary due to the lightweight nature of the materials and the limited volume of small lab-scale batch reactors (including the experimental design presented in this study), this drastically increases the number of edges available for surface erosion during microbial polymer biodegradation. While this will yield only a marginal increase on the overall surface for a single-layered plastic film, it could have more profound implications for multi-layered films. Indeed, the extra edges provide additional sites for micro-organisms to reach inner layers, which could alter the mode and rate of biodegradation, as hinted by scanning electron microscopy seen in the literature (Bandini et al., 2020; Zhang et al., 2014). Thus, experimental data may not match real-life AD performance and biodegradability rates of BBPs being tested. Notably, in a recent study the BBP fragment size was set at 25 cm^2, to reflect the size used to sieve OFMSW in commercial dry AD (Cucina et al., 2021), indicating that real-life conditions are being increasingly considered in study design.

On the polymer front, some BBP blends have been shown to have higher BMP (as a proxy for ultimate biodegradability) than individual BBP polymers found to have limited biodegradability in previous experiments (Narancic et al., 2018). It is possible that the better performance of PLA-based crockery to starch-based bags observed by Cucina et al. (2022a) was due to the use of additional polymers other than PLA. The synergistic effect of blending various polymers may thus represent a fruitful avenue to explore. In addition, commercially available products come with a range of additives, plasticisers and dyes, introducing further variability from the original raw material and greater uncertainties for AD plants handling these materials. It is therefore important to make the distinction between the polymer itself, i.e. its inherent physical, chemical and biochemical properties, and the product, the shape of which, alongside its thickness, number of layers, etc. will vary from one product to another, even if both products are made from the same given polymer. In practice, more data and mechanistic characterisation are needed to understand how a full plastic bag or rigid container will impact the process as well as assess the technological adjustments required to process complex mixtures of BBPs and other organic materials. The constantly changing composition of both incoming packaging and FW itself represent a significant challenge for the AD industry (DEFRA, 2015).

4.3.6. Microbial communities

Often perceived as the ‘black box’ of AD, the role of microbial communities has started to be increasingly recognised as a powerful indicator of AD performance (Zhang et al., 2014), and the addition of specialised microorganisms could enhance FW and BBP co-digestion through ‘prebiotic dosing’, or bioaugmentation. Different BBPs are degraded by different microorganisms, and different microbial communities will colonise the digester depending on the composition of the waste available. Among the numerous microbial species associated with BBP biodegradation, those belonging to bacterial *Pseudomonas*, *Streptomyces*, *Arthrobacter* and *Rhodococcus* and fungal *Aspergillus* and *Fusarium* genera are commonly cited (Danso, Chow & Streit, 2019; Pathak & Navneet, 2017).

In one study, PLA degradation increased over experimental runs, indicating an acclimatisation of the AD microbiome to PLA (Wang et al., 2012). Despite this, very little is studied in an industrial context, including with the relevant substrates. The non-trivial, time-consuming and costly nature of microbial analysis (i.e. meta-omics techniques) is arguably a contributing factor to the paucity of microbial characterisation in the field, although the cost and complexity of genetic sequencing has dropped sharply since the early 2000s with the emergence of next-generation sequencing (Dickson, 2021). Nevertheless, the ability to link taxonomic data with functional insights remains limited (Rahman et al., 2021), which is needed before a comprehensive picture of metabolic pathways occurring within a given AD system can be drawn. Strengthening public-private partnerships could accelerate knowledge transfer to and put research into practice.

Intriguingly, in one study investigating the impact of conventional plastic contamination on FW AD performance, scanning electron microscopy results suggested that the reduction in CH_4 yield was due to the interference between microorganisms and FW for effective biodegradation, and that the biological processes of AD were not affected by the plastics per se (Lim et al., 2018). Greater reductions in CH_4 yields were also observed when the surface areas of the plastic materials were increased (Lim et al., 2018), supporting the idea of a mechanical inhibition. It remains to be determined whether some BBPs do not present a similar barrier in an industrial AD context.

The set of parameters discussed in the previous sections (**Sections 4.3.1-4.3.6**) are condensed and illustrated in **Figure 24**. Together, they ensure the study of the biodegradation of BBPs is performed within an industry-relevant fashion, reflective of real-life treatment of BBPs and in line with current and future policy trends.

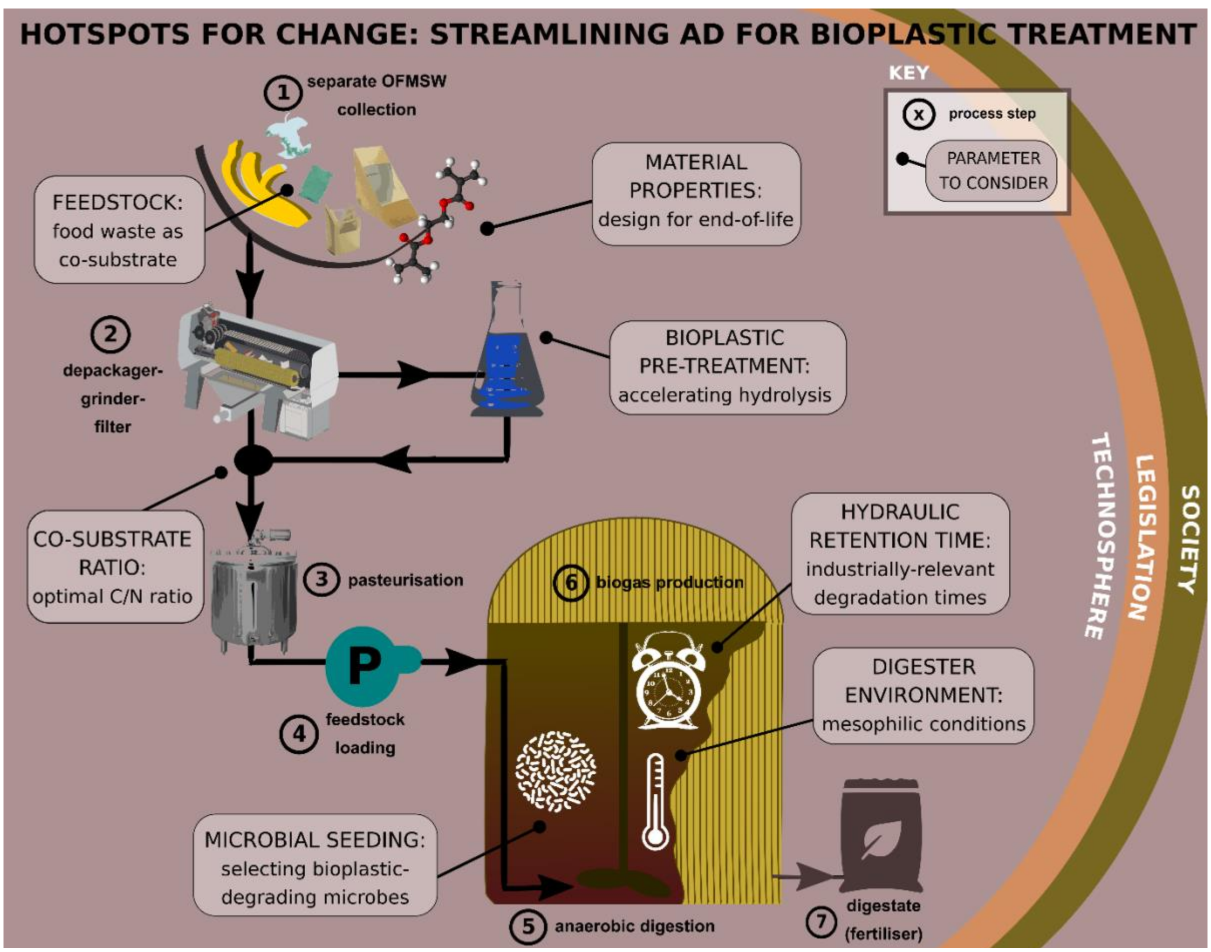


Figure 24 | Process steps (1-7) and key study parameters for the co-digestion of food waste and biodegradable bioplastics. In the UK, the digestate obtained in step 7 can be sold for agricultural purposes after meeting PAS 110 requirements. Source: author.

5. Conclusions & Future work

This chapter investigated the impact of conventional and BBP polymers on FW co-AD. Based on biogas and CH_4 yields, the introduction of (bio)polymers at relatively low concentrations (ranging from 0.1% to 5% w/w of SFW added) does not have any detrimental impact on AD performance, including biogas and CH_4 production. Biological characterisation through bacterial 16S rRNA metabarcoding supported these findings, which displayed relatively consistent microbial profiles and suggested the slight community shift observed between the start and the end of the incubation period was driven by the inoculation of SFW, rather than of (bio)polymer fragments.

Nevertheless, the precautionary principle should be applied to avoid unintended consequences. Given that microplastic leakage is expected to increase by 1.3-2.5 times by 2040 under a business-as-usual scenario (Lau et al., 2020), ensuring BBPs do not exacerbate this issue must be prioritised. As the BBP market share continues to grow, the study of BBP biodegradation needs to adequately reflect their intended EoL under current and future waste management practices.

By framing co-AD experimental design and research under the umbrella of a circular bioeconomy in which both FW and BBP food packaging waste are treated in a single organic waste stream, this chapter aimed to provide a paradigm shift for future co-AD studies. It highlighted the need for more harmonised experimental study designs and key study parameters and associated reporting units. Further research is needed to characterise the best conditions required to ensure optimal biodegradation of BBPs in FW co-AD and how these compare to real-life practices.

Results from co-AD trials pointed at the possibility that while they do not harm the performance of AD, EN 13432 certified BBPs may not undertake extensive biodegradation under anaerobic conditions and may find themselves in partially degraded states (i.e. physically fragmented, but not necessarily biodegraded by microorganisms) in digestate fractions. Future research should focus on digestate quality and consider the impacts of BBP on soil communities and plant physiology once introduced into agricultural soils, which remain poorly understood compared to those on marine vertebrates (Lear et al., 2021).

Achieving sustainability requires a systems-thinking approach. While this chapter and the wider scientific discipline it sits within suggest that BBPs may not undermine FW AD from a biochemical perspective, many operational, legislative and socio-economic challenges (the outer circles of **Figure 24**) remain to be addressed. Informed by empirical results and moving beyond the theoretical approach of the rapid review conducted in this chapter, on-the-ground AD practices in the context of OFMSW treatment are explored in the following chapter.

Chapter 5 – Stakeholder attitudes towards biodegradable bioplastic packaging in food waste anaerobic digestion

"*In theory, theory and practice are the same. In practice, they are not.*" – Albert Einstein

Part of the content presented in this chapter appears in the following publication:

Kakadellis, S., Woods, J. & Harris, Z. M. (2021). Friend or foe: Stakeholder attitudes towards biodegradable plastic packaging in food waste anaerobic digestion. *Resources, Conservation and Recycling*, 169, 105529.

1. Introduction

The potential of AD and biodegradation more broadly as waste management strategy for the treatment of OFMSW has benefitted from both societal and political support. The shift toward more circular, bio-based economic frameworks in national and international legislation has provided a fertile ground for a thriving innovation space and the development of an array of BBPs, which have become one of the fastest growing segments of the global plastics market (Meeks et al., 2015). The public has started to value biodegradability properties of packaging more highly than mechanical recyclability (Dilkes-Hoffman et al., 2019a); packaging manufacturers and retailers are increasingly switching to biodegradable alternatives to conventional plastic packaging, often with little consideration for the local context and waste management practices (Meeks et al., 2015). This inclination towards biodegradability might cause a problem if the pre- and post-consumer stages are not aligned (Dilkes-Hoffman et al., 2019a), with non-trivial consequences for local authorities and the waste management sector dealing with those materials once they have reached their EoL.

As separate FW collections are increasingly being implemented worldwide (WBA, 2018), developing a co-mingled collection for both FW and BBP food packaging represents a unique opportunity to 'kill two birds with one stone' within the food-energy-waste nexus (Babbitt et al., 2022; Kakadellis, Lee & Harris, 2022). In 2018, the EU announced an ambitious mandate for Member States to implement separate FW collections from businesses, households and household-like sources (e.g. corporate offices, canteens) by 2024 under Directive (EU) 2018/851 (amending Directive 2008/98/EC). The Directive clearly states that packaging waste with similar biodegradability properties to other organic waste streams and compliant with relevant EU or national standards may be collected alongside OFMSW. In the UK, the Waste and Resources Strategy published in 2018 (HM Government, 2018) outlined plans to make separate municipal

FW collections mandatory by 2023 and was enacted in the 2021 Environment Act (HM Government, 2022). However, the Government's Net Zero Strategy published in late 2021 announced £295 million of capital funding to support local authorities in England ahead of the collections from 2025 (HM Government, 2021), implying a two-year delay in the implementation of the scheme for households. Nonetheless, the availability of household FW collection schemes across devolved administrations is expanding and as many as 93% of Welsh households already had access to such schemes a decade ago (Research Service, 2013). The majority of local councils across Scotland and Northern Ireland also provide FW collection and recycling services (nidirect, 2022; The Scottish Government, 2019).

In light of upcoming changes in the amount of commercial and municipal organic waste and growing concerns over the accumulation of plastics in agricultural soils and waterways (MacLeod et al., 2021), understanding the practical implications of a joint biowaste management scheme is critical.

2. Study aims

Experimental data from **Chapter 5** showed that the introduction of BBP fragments at relatively low concentrations (0.1%-5% w/w of synthetic FW) does not harm overall biogas or CH_4 yields of AD processes. Encouraged by these results, but mindful of the wider environmental, social and economic implications of treating BBP waste alongside OFMSW in industrial AD, this study explores on-the-ground AD practices and stakeholder attitudes towards the promotion and waste management of BBPs. Based on semi-structured interviews, views from stakeholders across the civil service, waste management and non-governmental sectors were gathered to understand common issues, barriers and opportunities associated with the treatment of BBPs in FW AD. The study is based on three main research axes:

I. What are the attitudes towards the treatment of BBPs in AD among stakeholders?
II. How suitable is the current infrastructure for AD industry and what are the barriers?
III. How do various stakeholder groups' views relate to each other, in particular between the waste management industry and legislative/regulatory bodies?

Though it focuses on the British legislative and waste management landscape, the implications and recommendations identified in this study are relevant to a wider circular economy audience, in light of recent global incentives to valorise and harness the potential of biowaste.

3. Methodology

3.1. Study design and participant recruitment

A semi-structured interview-based approach was chosen in order to gather an in-depth understanding of stakeholder views and emerging issues related to BBP waste management in AD. This method enables interviewees to identify and expand on any information they believe to be noteworthy through responses to open-ended questions (Kvale, 2008). Semi-structured interviews allow for new topics not previously identified in the research question to emerge, thus bringing further insights to the study (Kuckartz, 2014).

While a narrow view sees stakeholders as groups or individuals who have power over the realisation of an enterprise's objectives, a wider view extends stakeholders to those groups or individuals who can affect or be affected by the attainment of such objectives (Gold, 2011). In the context of this study, stakeholders are those who can impact or are impacted by a decision to promote BBPs in the FW AD stream. Participants were chosen to cover the breadth of stakeholders directly or indirectly involved with the waste management of BBPs. An initial scoping stage identified relevant stakeholder groups and included AD plant operators, waste contractors, representatives of trade associations related to the bioplastics and AD sectors, bioplastic food packaging manufacturers, retail, environmental charities, local authorities as well as civil servants and environmental regulators (**Figure 25**). Unfortunately, no representative from local authorities was successfully recruited, which may have limited insights into logistical and financial aspects of FW collections, especially regarding the use of BBPs as FW caddy liners (see **Chapter 7 Section 2** for an overview of BBPs as caddy liners in separate FW collections). Farmers were also identified as a relevant stakeholder group, through the National Farmers' Union (NFU). However, when reaching out to the NFU, the scope of the request was deemed outside the remit of what the NFU felt capable of commenting on and the invitation to participate in an interview was declined. In spite of this, some views from the agricultural sector were captured through interviews with agriculture-related charities and environmental regulators (**Section 4.5.2**). Consumers were excluded from this study; though their views and influence are by no means trivial, their attitudes have been studied elsewhere (Dilkes-Hoffman et al., 2019a; Herbes, Beuthner & Rammer, 2018; Mehta et al., 2020; Sijstema et al., 2016) and are the focus of research presented in **Chapters 6 & 7**. In addition, this study focused on technical and legislative aspects of BBP waste management, given consumers' positive attitudes towards biodegradable packaging (Dilkes-Hoffman et al., 2019a). Academic stakeholders were excluded, as the study focused on on-the-ground practices. Nevertheless, main academic concerns emerging in the relevant scientific literature reflect priorities of the scientific community and are discussed throughout this chapter.

Participant recruitment was initiated at the 'Everything is Connected' conference hosted by the Bio-based & Biodegradable Industries Association (BBIA) on February 25th, 2020, in London, which gathered representatives of the biodegradable food packaging supply chain. Further participants were contacted directly by email, based on a gap-filling exercise following the aforementioned conference. Overall, 19 interviews were carried out between February and July 2020. Interviews were carried out by the author in person or online (Skype/phone call) and lasted between 15 and 30 minutes. The interviews were recorded on the researcher's phone/laptop with the participant's permission. Transcription was supported by the transcription software Happy Scribe and transcripts were later checked manually before proceeding to data analysis. It was decided to stop further participant recruitment and data collection when all identified stakeholder groups had been interviewed and preliminary data assessment did not yield any new themes.

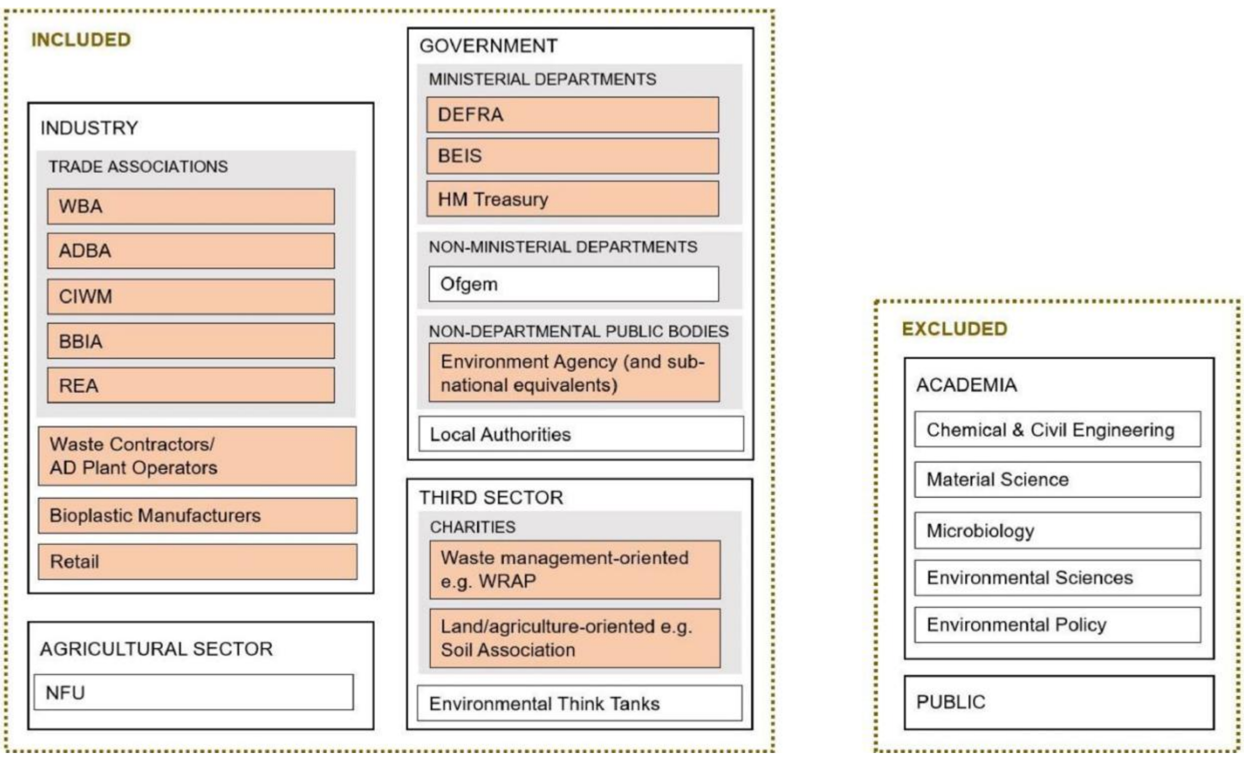


Figure 25 | Stakeholder map showing groups relevant to the scope of this project and those beyond the scope. Stakeholders relate to the British context**.** Orange boxes represent stakeholder groups who took part in the interviews (i.e. participants). WBA: World Biogas Association; ADBA: Anaerobic Digestion & Bioresources Association; REA: Renewable Energy Association; CIWM: Chartered Institute of Waste Management; BBIA: Bio-based and Biodegradable Industries Association; NFU: National Farmers Union; WRAP: Waste & Resources Action Program; DEFRA: Department for Food and Rural Affairs; BEIS: Department for Business, Energy & Industrial Strategy; Ofgem: Office of Gas and Electricity Markets. Reproduced from: Kakadellis, Woods & Harris, 2021.

3.2. Research ethics

Research ethics are increasingly recognised as a key stage of research design. The sensitivity of the topic was considered in the research design (Eynon, Fry & Schroeder, 2008). AD is not a highly sensitive topic compared to e.g. health data, so privacy issues were inherently reduced.

Ethical approval by the researchers' institutional ethics committee was granted ahead of the study (registration number: 20IC5758) and informed consent from participants was sought prior to data collection. Following standard procedures, a unique reference number was assigned to each participant during the interview and for further data analysis to ensure identifiable information remained pseudonymised. Recordings and transcriptions of the interviews were kept in a secure folder in the online storage platform provided by the researchers' institution (OneDrive) and only accessible to the researchers. The original audio recordings were destroyed once transcription was completed.

3.3. Topic guides

When conducting the interviews, participants were split into two categories: civil servants, and others. This was based on the third research axis, which aimed to address the potential conflicts between policy incentives and on-the-ground practices. Two a priori topic guides (see **Appendix Section 2**) were developed and refined with the support of experienced interviewers to ensure the terminology used by the interviewer was as neutral as possible to reduce interviewer bias. Questions were designed specifically to address the research aims outlined in the introduction and the information sought. For this reason, they were directly related to the thematic nodes in subsequent data analysis. Given the nature of semi-structured interviews, not all questions were asked, acting more as a guide to steer the conversation whilst ensuring the relevant research question was addressed.

3.4. Data analysis

The trustworthiness of qualitative research depends upon the integrity of data gathering and analysis, the robustness of processes and the demonstration of thoroughness (Kuckartz, 2014). The aim is to ensure that the research questions are answered from the relationships emerging out of the data being searched (Kuckartz, 2014). Transcripts were analysed using the computer-assisted qualitative data analysis software (CAQDAS) NVivo 12 (QSR International). Open-ended questions were analysed based on inductive and deductive strategies, whereby transcript extracts were assigned to predetermined or emerging thematic folders, respectively, called *nodes*. The main thematic nodes were selected based on an iterative process in line with CAQDAS methodology (Ackrill & Abdo, 2020; Kuckartz, 2014) and following NVivo's user guides (QSR

International, 2021). Each main node corresponded to a broad theme directly relevant to the research aims, or which emerged from transcript analysis. These broad nodes, or parent nodes, were further divided into child and grandchild nodes where relevant.

Any given text extract, called *reference*, can belong to multiple nodes. Relationships between nodes were then explored using in-built interrogational tools, which allowed the researcher to conduct word frequency searches, follow coding patterns based on stakeholder characteristics (e.g. stakeholder group) and explore relationships across codes (e.g. the link between gate fees and policy incentives). To support the reader in identifying some of the nuances, certain references were cited, followed by the pseudonymised respondent, referred to as 'R' followed by its associated number (e.g. R7 for respondent #7) in the text.

3.5. Methodological limitations

One limitation of the study is the lack of direct evidence from all relevant stakeholder groups, particularly from the farming sector. To further increase the robustness of the results, further stakeholder groups would need to be recruited. In addition, many participants were confused by the term 'biodegradable plastics' (see **Section 4**), which may have altered their perception on the suitability of BBPs in AD.

The content analysis was performed by one researcher only. Whilst this may lead to a bias issue, the iterative nature of the qualitative research process aimed to ensure consistency among question formulation, participant recruitment, data collection and analysis. In that sense, researcher responsiveness, rather than external scrutiny of the completed analysis, supports the trustworthiness of qualitative research (Morse et al., 2002). Nevertheless, the coding strategy was scrutinised and discussed amongst all authors at multiple stages to further avoid bias and strengthen the validity of data interpretation.

4. Results & Discussion

The research design and questions focused on a qualitative understanding of key issues around BBP uptake in industrial AD plants treating FW. Interview transcripts were analysed with the NVivo 12 software and classified into thematic nodes and sub-nodes based on both inductive and deductive strategies, leading to the classification displayed in **Figure 26**.

Qualitative data were first queried through an initial word frequency search (**Table 17**). The search revealed that alongside the expected most frequent words (e.g. 'plastic', 'biodegradation' and 'AD'), the term 'people' was also used frequently, which led to the exploration of a new theme exploring participants' views of other groups not included but relevant to this study (**Section 4.5**).

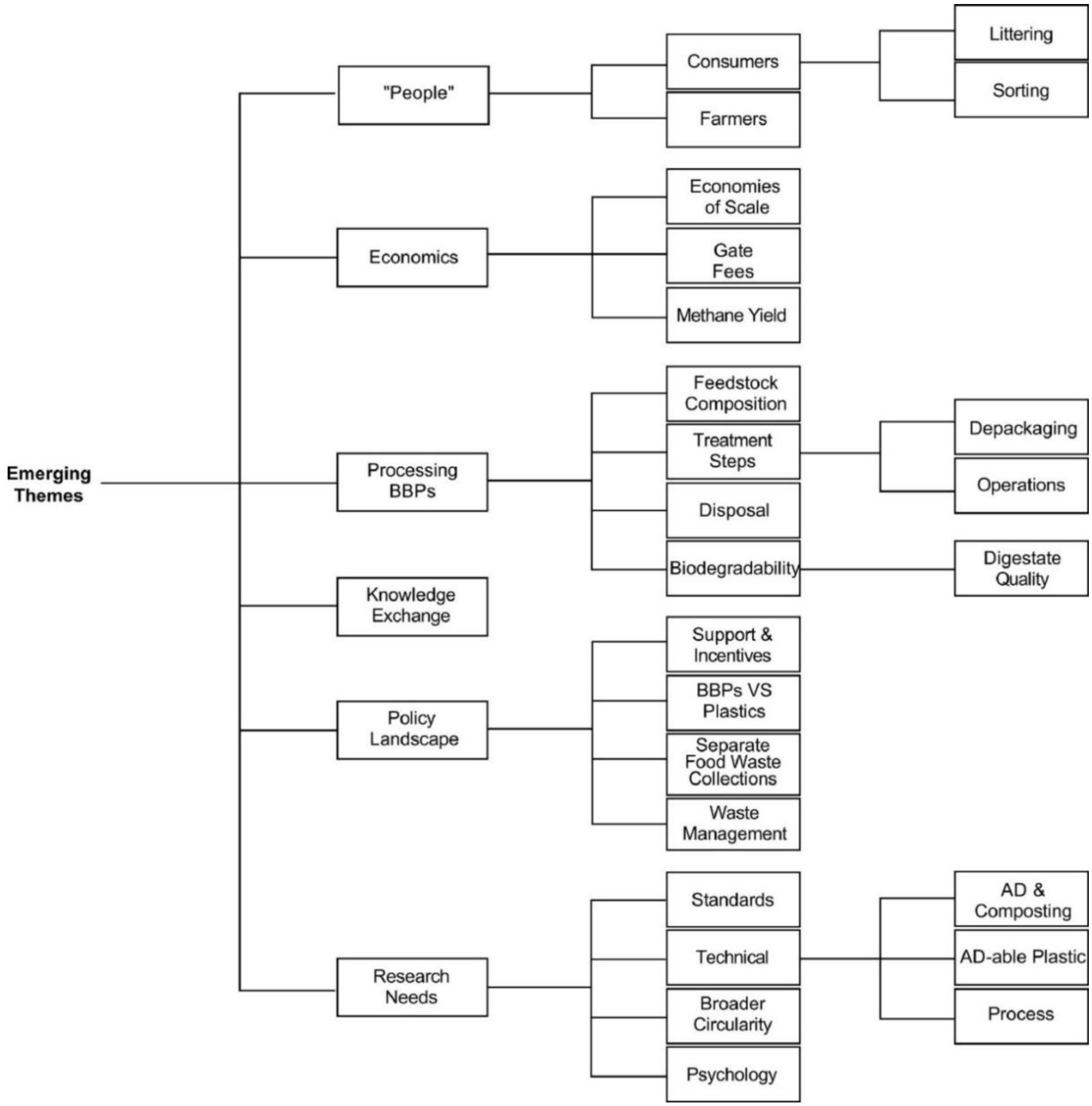


Figure 26 | Emerging themes from transcript analysis. The coding process was a continuous and iterative process, starting with a deductive approach for broad nodes, with new nodes being created/merged based on emerging themes from the analysis, following an inductive strategy. Parent nodes were further divided into child and grandchild nodes where relevant. BBPs: Biodegradable bioplastics. Reproduced from: Kakadellis, Woods & Harris, 2021

The majority of the questions (and associated answers) focused on the capability of the current AD infrastructure to process BBPs alongside FW, which was reflected in the extent of coding for the thematic node 'Processing BBPs' (**Figure 27**), covered in **Section 4.1**. There was often some confusion and/or disagreement among interviewees, particularly within the waste management sector, as to what 'biodegradable' referred to, and some respondents assumed all BBPs were oxo-degradable, which are now banned from the EU (EU, 2019).

Table 17 | Most frequent words used across all interviews. Based on a Word Frequency Query with stemmed words and minimum length of 2 letters across all 19 interviews. Irrelevant terms (e.g. verbs 'think', 'go', 'know') were removed and the query was run again. An asterisk (*) indicates the root term for similar words (right column). Reproduced from: Kakadellis, Woods & Harris, 2021.

Rank	Word	Count	Similar words (if relevant)
1	Plastic(s)	295	
2	Waste(s)	210	
3	Compost*	196	Compostability, compostable(s), composter, composting
4	Food(s)	177	
5	AD	137	
6	Biodegrad*	125	Biodegradability, biodegradable(s), biodegradation, biodegrade(s)
7	Digest*	103	Digestible, digestate, digested, digester(s), digesting, digestion
8	Material(s)	90	
9	People	85	
10	Plant(s)	81	
11	Process*	80	Process, processed, processes, processing
12	Bag(s)	74	
13	Collect*	69	Collect(s), collectable, collected, collecting, collection(s)
14	Industr*	62	Industrial, industry
15	Packag*	62	Package, packaged, (de)packaging

While the study aimed to assess the suitability of the current AD infrastructure to process BBPs from a technological point of view, the economic and policy landscape was also addressed in order to investigate deployment/processing barriers from a wider circular perspective, reflected in the homonymous themes in **Figures 26 & 27**. Despite a push from bioeconomy-oriented policies to shift towards biodegradable packaging alternatives as well as circular waste management approaches, respondents highlighted potential conflicts between political ambitions and financial support, as well as between siloed and holistic circularity in the food and packaging supply chain (**Sections 4.2 & 4.3**).

The nature of individual nodes and their implication are analysed and discussed in more depth in the following sections and sub-sections. As research opportunities were identified and discussed across all nodes, the content for the node 'Research Needs' is discussed through the remaining nodes rather than as a separate section. Representative extracts from the reviews are presented throughout to illustrate emerging themes. Quantitative indicators of frequency of individual themes have been included for descriptive purpose to support the analysis.

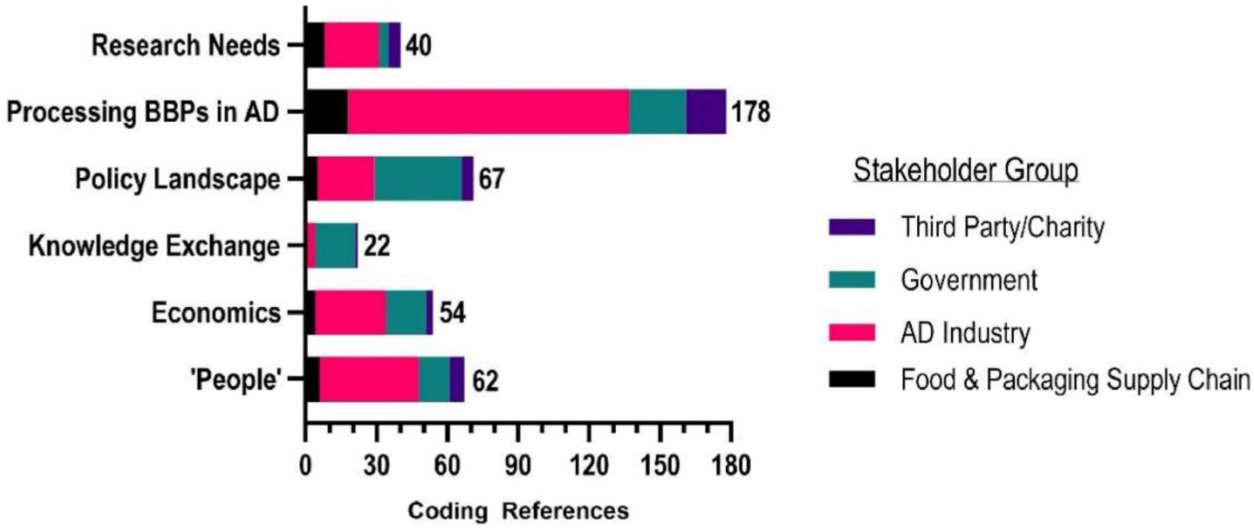


Figure 27 | Coding references of emerging themes by stakeholder group. Every text extract matched to a given node counted as one reference, and one reference may belong to multiple nodes. AD: Anaerobic digestion; BBPs: Biodegradable bioplastics. Reproduced from: Kakadellis, Woods & Harris, 2021.

4.1. Processing BBPs

Broadly speaking, responses on BBPs tended to be more negative (61%) than positive (39%) (**Figure 28**), due to the low incoming volume to treat, the difficulty in distinguishing them from conventional, non-biodegradable plastics, the lack of an 'AD-able' degradation standard, systematic depackaging and concerns over their impact on process operations and on digestate quality. Nevertheless, some positive comments from plant operators were supported by practical case studies, which suggests the identified barriers may be overcome.

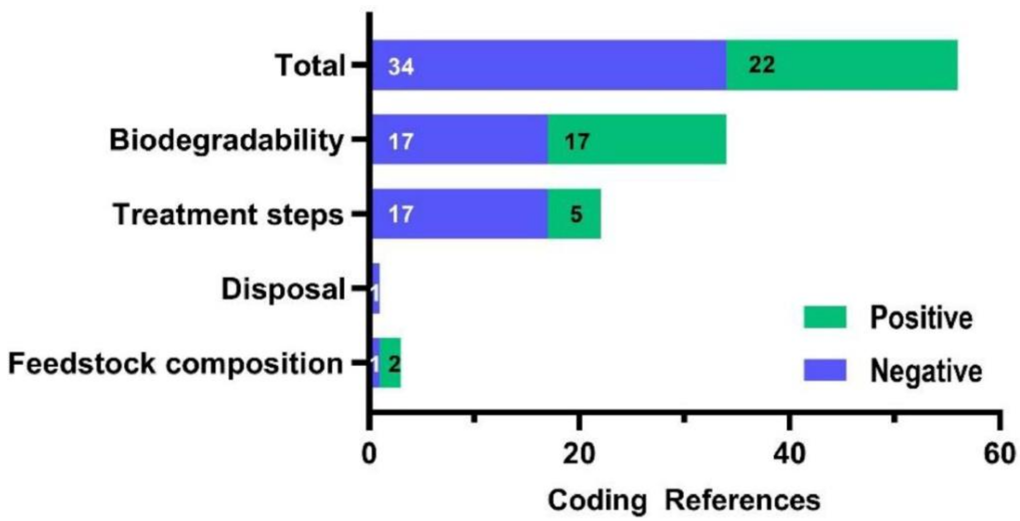


Figure 28 | Disposition towards BBPs for the node 'Processing BBPs in AD'. Disposition ('Sentiments' in NVivo) refers to references with a distinct positive or negative quality. Not all coded content was associated with a 'sentiment', hence the number of total references for this node (178, cf. **Figure 27**) is not equal to the total shown here. Reproduced from: Kakadellis, Woods & Harris, 2021.

4.1.1. Feedstock composition

Where relevant, feedstock composition was explored to determine in which proportion plastics were present in the incoming feedstock. The data provided by plant AD operators indicated that plastic packaging represents 5-9% of incoming feedstock by weight, slightly higher than the percentages adopted in co-AD trials in **Chapter 4**. Respondents indicated that these values may rise to 20-25% in highly packaged streams, such as supermarket waste, which is delivered to AD facilities without any pre-processing (i.e. depackaging). A small proportion of these plastics are made of EN 13432 certified compostable packaging, mostly because of local authorities providing caddy liners to households. Due to the challenge of distinguishing between biodegradable and non-biodegradable plastics, an accurate quantification of BBPs in the feedstock is missing and would require more visible labelling and/or sorting technology. The similarity between both categories of plastics also explains why conventional plastics may find their way in the FW bin, with households notably substituting compostable caddy liners with non-biodegradable plastic shopping bags (CIC, 2017), further discussed in **Section 4.5.1**.

4.1.2. Disposal

One respondent alluded to the costs associated with dealing with plastic waste following feedstock depackaging and screening. The plastic materials (including BBPs) captured at these stages are either landfilled or incinerated due to food contamination and thence of poor quality – at the cost of the operator. This stands against circular waste management principles and brings additional costs to running the AD plant:

> "*The plastics are an issue because we need to pay to dispose of them. With a 5% contamination rate (…), that's roughly £455,000 per year just to dispose of plastics.*" (R17)

The Italian biowaste sector currently spends €90-120 million annually to extract plastics from its organic feedstocks, despite having the most extensive FW collections in the EU with remarkably low plastic contamination levels (1.5% on a weight basis) (BBIA, 2020). Plastic contamination must be reduced to avoid elevated costs for the biowaste industry, which currently suffers from a limited ability to set minimum quality thresholds for feedstocks, due to economic obstacles (**Section 4.2**). One drastic solution would be to mandate all flexible food packaging to be biodegradable (through certified standards), although the question of their suitability in AD remains to be addressed (**Section 4.1.4**). A more moderate approach could target consumers by making BBPs more visible (**Section 4.5.1**), with multiple stakeholders highlighting the need to consolidate current knowledge on consumer behaviour.

4.1.3. Operational hurdles

One of the major concerns perceived by plant operators related to operational hurdles across stages of the AD process ('Treatment steps' in **Figure 28**). Depackaging appeared as the biggest barrier, since most plants treating FW in the UK currently go through systematic depackaging, regardless of packaging material and including certified compostable caddy liners used in some household FW collections. As one respondent summarised:

> "*Unless you can say with a 100% certainty that all the packaging was biodegradable, then you would never get rid of that first process of depackaging*." (R1)

Several stakeholders agreed that the viability and credibility of BBPs depend on their disposal route. If no distinction is made between conventional plastics and BBPs at the sorting stage, there is little point designing biodegradable alternatives, especially since biodegradability is a key component of their 'green' credentials (Kakadellis & Harris, 2020).

In addition, compostable and other biodegradable polymers were reported to stretch out in the process and may cause clogging of pumps and wrap around stirrers. However, provided the volume of BBPs going through AD continue to increase, plants would adapt their technology and processes to accommodate this new material stream. Notably, two plant operators indicated no operational issues despite accepting a range of certified compostable plastics in their plants. One said:

> "*They [plastics] can get wrapped around the screws that transfer some of the products. But our system is quite resilient and stuff that isn't degradable (…) settles down at the bottom of the tank and then we just periodically remove that and dispose of it. (…) I haven't seen any plastics wrapped around the mixing blades or anything like that*." (R17)

This example provides practical evidence that with an appropriately designed system, BBPs can indeed be suitable for anaerobic treatment. Pre-treatment strategies and effective product design could further enhance the suitability of BBPs for AD (Bátori et al., 2018; Narancic et al., 2018), as discussed below.

4.1.4. Biodegradability and digestate quality

Another major concern identified centred around the extent of biodegradability and its inherent link to digestate quality and further impacts on soil health. This concern was prevalent among environmental regulators and charities. There was a shared apprehension amongst stakeholders that BBPs are not fully adapted to anaerobic biodegradation, given that the only certification standards for industrial waste management of biodegradable products apply to in-vessel composting (EN 13432). Indeed, compostable packaging is not designed to perform under anaerobic conditions (Kale et al., 2007). This may not exclude dual aerobic/anaerobic biodegradability properties for some biopolymers (Narancic et al., 2018), but partial biodegradation may lead to the release of microplastics, especially under colder weather conditions characteristics of the UK climate, where soil temperature will likely hinder further biodegradation, leading to unintended environmental consequences downstream of AD (Aspray, Dimambro & Steiner, 2017).

Some more informed stakeholders noted the importance of looking at individual products and applications, rather than any given polymer per se. Indeed, a compostable caddy liner may go through the process smoothly, while a thicker, waxed coffee cup might not be suitable, even though they are made from the same starch-based biopolymer (Kakadellis, Lee & Harris, 2020). This nuance was also reflected in a certain mistrust towards scientific research and the available body of academic literature dedicated to BBP biodegradation studies, deemed unrealistic:

> "*I'd want some serious proof of real-world degradation in the type of feedstock that we are collecting regularly. I don't want no lab study. I'm not interested in a little cell somewhere that's all perfectly designed (…). What I'd want is: here is a real-world mix of home food waste and we've dropped in a decent proportion of this stuff [biodegradable packaging] as a substitute for other packaging types, now let's see how it [performs]. That's the only way we run our [AD] plants.*" (R3)

This further reinforces the need for relevant study design that truly reflects current and projected waste management practices around OFMSW advocated in **Chapter 4** and more broadly the need for cross-sectoral knowledge exchange and collaboration (**Section 4.4**).

Two potential solutions were identified in discussion with stakeholders: the development of an industrial standard for 'digestible' or 'AD-able' packaging and the introduction of pre-treatment steps. Both these solutions are compelling, because practical examples already exist. Thermophilic and alkaline pre-treatments have been shown to improve biodegradability of a range of BBPs at a laboratory scale (Hobbs et al., 2019; Samitthiwetcharong & Chavalparit, 2019). From an industrial perspective, one of the respondents noted they have introduced autoclaving (a sterilisation technique alike pasteurisation) upstream of AD, which effectively acts as thermo-mechanical pre-treatment step, whilst conforming with legal requirements under the Animal By-Products Waste Regulation (ABPR). Enhancing biodegradability comes hand in hand with the provision of a biodegradability standards for AD. As an example, Cré, the Composting & Anaerobic Digestion Association of Ireland, has recently introduced a label for packaging compatible with separate FW collections (BBIA & Futamura, 2020). Together, these measures would support the viability of the AD outputs, especially given the growing interest in urban waste-derived fertilisers in the farming sector (Case et al., 2017).

4.2. Economics

While none of the questions in the topic guides specifically referred to economic factors (see **Appendix Section 2**), the activities of commercial AD plants are evidently influenced by economic viability, which was reflected in the coding frequency (**Figure 27**). Three sub-themes dominated the analysis: biogas (and biomethane) yield, gate fees and economies of scale.

Initially, the research question focused on assessing participants' awareness and interest in the potential of BBPs to contribute towards biogas production as suggested in the literature (Narancic et al., 2018; Vasmara & Marchetti, 2016). However, it became apparent that although plant operators were inclined to consider any feedstock with biogas potential, this was not of primary concern. Firstly, as several respondents pointed out, although BBPs may indeed release CH_4 upon anaerobic degradation, the relatively small volume of BBPs would be negligible compared to the incoming FW. Experimental data from co-AD trials conducted in **Chapter 4** supported this view. For this reason, economies of scale seemed to dictate respondents' attitudes towards BBPs, whereby the volumes of BBP in current FW streams are relatively low and are thus not considered a priority by AD plant operators. This suggests that plant operators may be willing to consider retrofitting their facilities in the future insofar as the share of BBPs increases to a point where it affects biogas production levels. This may indeed be the case, given the incoming policy on separate FW collections from 2023-2025 across the UK and EU (DEFRA, 2021; EU, 2020) and the ever-expanding BBP market in food packaging (European Bioplastics, 2021; Meeks et al., 2015), provided an industrial standard for 'AD-able' packaging becomes available.

Secondly, the risk of digestate contamination was perceived as a more pressing issue to address (**Section 4.1.4**), given its potential to replace petrochemical fertilisers:

> "*Whereas actually it's more important to get the right quality of organics to go to land as well, as well as, of course, generating gas*." (R2)

This is especially important considering increasing pressure on environmental regulators to tighten the contamination threshold (especially on plastic fragments) of PAS 110 and that this quality standard enables the organic waste management sector to sell digestate at a premium price to farmers and agricultural contractors (WRAP, 2016). Failing to adhere to PAS 110 criteria can therefore severely undermine the application of FW-derived digestate for agricultural purposes. Thus, the decision to process BBPs alongside FW appears as a trade-off between potentially improved biogas yields and increased feedstock contamination, especially since distinguishing between ‘AD-able’ and conventional plastics remains challenging. Nevertheless, the co-benefit of increased CH_4 yields – provided proven biodegradation – was deemed, at least conceptually, positive.

Respondents were highly divergent on the nature of economic drivers, across stakeholder groups. From eleven quotes on the subject, biogas production was perceived as the major source of income through feed-in-tariffs by seven respondents. In contrast, three respondents quoted gate fee – a charge for a given quantity of waste received at a waste processing facility – as primary economic driver, in part due to increasing landfill taxes (£98.60/t since April 1st, 2022; HM Revenue & Customs, 2022). It seems that early tariffs allowed AD operators to operate on a ‘zero’ gate fee, which is no longer the case with the phasing out of the Renewable Heat Incentive (RHI), which could explain the divergence of opinions (Röder, 2016). New incentives beyond the RHI, such as the Green Gas Support Scheme and Green Gas Levy, should provide a new subsidy stream for the future support of low carbon heat (BEIS, 2020a; 2020b).

4.3. Policy landscape

Focus is often put on developing alternatives to conventional plastics with little consideration for the readiness of the relevant EoL infrastructure. Thus, this study sought to capture the emerging policy agenda regarding the treatment of BBPs and its link to AD as FW treatment strategy and explore the balance between the promotion of BBPs and the increase in recycling rates for conventional plastics.

One respondent from the civil service noted that any debate and information transfer within the AD sphere can be challenging due to its very nature, as AD is dealt both by the Department for Business, Energy and Industrial Strategy (BEIS) – as a source of renewable energy – and the Department for Food, Environment and Rural Affairs (DEFRA) – as a waste management strategy. In addition, whilst plastic pollution management is in the remit of DEFRA’s activities, the development of alternative materials such as BBPs fits within BEIS’ Industrial Strategy and Bioeconomy, not to mention that Treasury oversees the Plastic Packaging Tax. The same applies to regulatory bodies, where gas and electricity-related markets are regulated by the Office of Gas and Electricity Markets (Ofgem), while the Environment Agency (and devolved bodies) is the regulator for (but not limited to) aspects of waste management and land management practices.

4.3.1. Balancing end-of-life scenarios

When addressing the biodegradability vs recyclability debate, a liberal 'laisser-faire' approach among civil service respondents seemed to be favoured, in that the industry would drive one market or another based on the soundest packaging application. In terms of terminology, there was an acknowledgement that clearer and simpler signposting is needed. In light of this, some respondents were concerned that the addition of terms such as 'biodegradable' or 'compostable' may complicate the recycling picture (Dilkes-Hoffman et al., 2019a), just as a binary 'recycle/don't recycle' labelling scheme for plastics in being developed. This echoes the position of Burgess et al. (2021), who argue for a 'one bin to rule them all' for effective plastic waste management.

Furthermore, the recent Plastic Packaging Tax, which came into effect on April 1st, 2022, introduced a £200/t tax on UK manufacturers or importers of plastic packaging components containing less than 30% recycled plastic content (HM Revenue & Customs, 2021) and thus applies to any biodegradable plastics, including EN 13432 certified compostable plastics. This policy is in direct contradiction with the UK Plastics Pact, a consortium of businesses from the entire plastics value chain, governmental bodies and NGOs and led by WRAP and the Ellen MacArthur Foundation, which aims to achieve 100% of plastic packaging to be reusable, recyclable or compostable by 2025 (WRAP, 2022).

There is also uncertainty over how BBPs will fit in the revised PAS 110 standard for digestate (BSI, 2014) (**Section 4.5.2**). The substitution of hard-to-recycle plastics with BBPs may represent a step towards tackling plastic contamination in agricultural soils, but whether such polymers do not further contribute to microplastics pollution remained to be determined (SEPA, 2017).

4.3.2. Aligning the promotion of BBPs with anaerobic digestion incentives

The use of BBPs as caddy liners for FW collections was seen as beneficial by all stakeholders because of its role in enhancing both the rate and the quality of consumer participation in FW sorting. Practical examples from both European and international cities validate that view (EEA, 2020). AD is stated as the preferred treatment option for separately collected FW, although, in line with the European Waste Framework Directive, there is some leeway depending on the local context (ADBA, 2020). There was a consensus amongst stakeholders that mandating separate FW collections, coupled with a ban on biodegradable waste sent to landfill, will boost the AD sector.

Yet, there is no direct line of action that directly incentives AD linked to separate FW collections. Nonetheless, in line with achieving a net-zero economy, there is a big push to inject biomethane in the current gas grid (EBA, 2021; see **Chapter 3 Section 3.3**) – a strategy already widely in place in other countries, such as Germany, France and Austria, where AD has been primarily portrayed as a renewable energy technology (Röder, 2016). In the UK, subsidies have gradually shifted from electricity (in the early feed-in tariffs) and heat (through the RHI) to biomethane (through the Green Gas Support Scheme). Since AD facilities for the treatment of commercial and household FW are most likely to be located in peri-urban areas, which tend to be closer to the national grid pipelines than rural AD plants (agricultural wastes), this may promote municipal wastes as feedstocks. One respondent noted that due to recent concerns over direct/indirect land use changes a cap has been recently imposed on new AD plants, with no more than 50% of their biogas to come from dedicated energy crops (equivalent to roughly 10% of incoming feedstock).

While recognising the role of biogas as an important contribution to renewable energy targets, representatives from environmental charities, regulatory bodies and biogas industry called for an emphasis on ensuring feedstock quality as well as adherence to circular economy principles, with a focus on resource efficiency across the system, include waste prevention:

> "*You're trying to gain as much methane as you can because that's where your incentives come from, which has always been the problem (…). I think [AD] should have been built as a process to get as much out of everything so [AD] shouldn't have just been built for the methane and the subsidies*." (R11)

> "*There's been a huge push [for] biogas (…) We should capture everything we possibly can, but we must also recognise we shouldn't waste it in the first place*." (R2)

4.4. Knowledge exchange

In line with the third research objective ('How do various stakeholder groups' views relate to each other, in particular between the waste management industry and legislative/regulatory bodies?'), a node was created to evaluate existing information transfer and engagement between stakeholder groups. Content analysis revealed that substantial crosstalk takes place between relevant government departments – BEIS and DEFRA – and trade associations – BBIA, Renewable Energy Association (REA) and Anaerobic Digestion and Bioresources Association (ADBA) – and to a lesser but non-trivial extent with academia, including through public consultations. Trade bodies and regulators also engage with the farming industry; however, there seems to be a lack of engagement across the supply chain as a whole, especially at the retail level. As a representative of the waste management trade body expressed:

> "*I would like retailers to have discussions with trade bodies, like the National Farmers' Union or the Anaerobic Digestion and Bioresources Association, but I don't think they will*." (R15)

These insights echo the growing perception among the public that manufacturers and the retail sector ought to take more responsibility for their products (Changing Markets Foundation, 2020). This is indeed crucial, given that increased awareness on environmental issues by consumers does not directly translate into an increase in individual pro-environmental actions (Dunn, Mills & Veríssimi, 2020). There is a hope that the revised Extended Producer Responsibility (EPR) scheme for packaging, originally planned for 2023 but delayed to 2024 or 2025 (Langley, 2022), will address this issue by requiring producers to increasingly contribute towards the net costs of disposal of packaging they place on the market.

4.5. Views on other stakeholder groups

When conducting an initial word frequency search to explore the content of respondents' transcripts, it became apparent that the term 'people' was coined frequently (in 9^{th} position in **Table 17**). This new thematic node allowed for attitudes towards groups that were not included in this study (e.g. consumers) to be captured, and to frame the debate within a wider picture.

4.5.1. Consumers

When enquiring on the suitability of BBPs in FW AD, a number of respondents directly linked the uptake of such materials to consumer behaviour, which shows that the robustness of the industry is heavily dependent on the quality of the feedstock. Consumer behaviour was split into two sub-themes: ease of sorting and littering.

Respondents emphasised that only if it is clear and easy for the consumer to understand, distinguish and separate various packaging materials will the waste management sector be able to collect a clean FW stream. This is inherently linked to the issue of terminology, as consumers are often confused about terminology and consumer awareness on bioplastics – and variations of – is poor (Dilkes-Hoffman et al., 2019a; Sijtsema et al., 2016; Taufik et al., 2020). As one respondent commented:

> "*How do we make it easy and visible to the consumer to ensure that the right material ends up in the right bin?*" (R3)

This is crucial since consumers are most likely to dispose of their biodegradable packaging in the recycling bin (Dilkes-Hoffman et al., 2019a). Respondents also called for supportive legislation to ensure that 'green' credentials are backed up by certification standards. This is reflected by the number of calls by the industry and circular economy-orientated organisations for political action to create an effective after-use plastics economy, including the ban of evasive terminology (ADBA et al., 2020; World Economic Forum, Ellen MacArthur Foundation & McKinsey, 2016).

Several stakeholders were concerned that BBPs may act as a license for littering. It has been reported that biodegradable products contribute to an improper disposal of litter due to a perceived lower responsibility on the part of the consumer (Haider et al., 2019). Nevertheless, in the context of compostable caddy liners for FW collections, the example of the Italian composting industry contradicts this, showing that the substitution of conventional bags with compostable liners increased consumer awareness of quality issues and non-compostable contamination was reduced from 9% to 2% w/w (Ricci, 2020).

4.5.2. Farmers

Although farmers were not interviewed (see **Section 3.1**), references to the agricultural sector were gathered here by respondents directly involved with farmers and landowners. One of the respondents stated that farmers' leverage on the question was limited, once again demonstrating the need for strengthened communication and co-operation within the supply chain:

> "*Really, they [farmers] don't necessarily have an awful lot to say in how food is packaged (…) because the use of plastic is essentially driven by retailers. Retailers do what consumers want, or what they think consumers want.*" (R9)

There also seemed to be a general fear amongst farmers that digestate from non-crop-based feedstocks (i.e. urban waste-oriented streams, such as household FW) will result in more plastic contamination, due to the relatively higher proportion of plastic packaging in these streams. An example was given of the Scottish meat, dairy and whisky industry, which lobbied for a tightening of plastic contaminant threshold in PAS 110 (BSI, 2014). The Scottish Environment Protection Agency has since amended its PAS 110 regulation (SEPA, 2017). In an effort to safeguard soil quality on agricultural land, it reduced the limit of plastic contaminants allowable in compost and digestate outputs across its administrative territory to 8% of the PAS 110 limit for digestate and 50% of the PAS 100 limit for compost. In 2021, the Environment Agency announced that the Compost and Anaerobic Digestate Quality Protocols (QPs) need to be revised before they can be further supported – that is, for the end of waste statuses of compost and anaerobic digestate to remain. Plastic contamination was deemed the most pressing issue that needs addressing in the review of both Quality Protocols and needs to be minimised to raise industry standards and build market confidence. Thus, despite not having direct control over what goes through the supply chain, farmers have some influence over the quality of AD outputs – which may not be surprising, given that the agricultural sector represents the largest end-user of soil enhancement products (WRAP, 2016).

It must be noted, however, that interest in OFMSW-derived fertiliser is growing (Case et al., 2017). Yet issues on both demand and supply sides, such as unreliable nutrient contents or high costs and lack of availability, represent barriers to the adoption of such organic fertilisers (Case et al., 2017). In addition, as one respondent noted, the uptake of digestate from FW AD by the farming industry is ultimately governed by legislation. Because of ammonia emissions ceiling, for example, there are some strict rules as to when farmers are legally allowed to spread liquid digestate on land (by surface application), a window that is further narrowed in the case of organic agriculture (WRAP, 2016). Thus, the capture of BBPs for FW AD sits within the broader political landscape of digestate use in the context of sustainable agricultural practices. This further underlines the need for co-operation between stakeholders from across the food supply chain (including FW) to guarantee a viable and sustainable digestate market from FW feedstocks (WRAP, 2016).

5. Conclusions & Future work

The incoming policy mandating separate FW collections across both the UK and the EU offers an unprecedented opportunity for the AD sector to expand its feedstock stream and demonstrate its circular ambitions. In this context, BBPs can contribute towards increasing both the quantity and quality of FW from household and household-like sources. However, this study shows that the current waste management infrastructure presents a number of barriers to the immediate and widespread uptake of BBP materials. By gathering attitudes towards BBPs in AD from a range of stakeholders through semi-structured interviews, it was found that the waste and AD sectors remain generally concerned about the suitability of BBPs in anaerobic treatment, despite acknowledging the merits of biodegradability properties. Interview content analysis revealed that BBPs are still perceived as a contaminant by waste contractors and plant operators, largely due to the relatively low volume of such materials in the waste stream and uncertainties around their proven biodegradability. Systematic depackaging was another issue identified, underlying the need for clearer distinction between conventional and certified BBPs. Impacts on digestate quality and downstream effects on soil health were identified as primary concerns for environmental charities and regulators. These issues highlight the need for the development of an industrial standard for AD-compatible packaging materials. Some examples showed that with the right incentives and technological innovations, BBPs can be treated alongside FW, without detrimental consequences on the AD process. Whether BBPs increase the capture of FW, thereby providing cleaner feedstock and securing a viable market for plastic-free digestate, could be further clarified by gathering insights from local authorities responsible for implementing separate FW collections.

This study also highlighted the role of consumer education, supported by enabling legislation. Providing consumers with simple and clear labelling as well as educational campaigns aimed at clarifying the often evasive terminology would help households to be better informed and enable the system to capture their waste streams more consistently. Further research should aim to better characterise the drivers of consumer disposal behaviour and the wider system they are embedded within, which represent the focus of **Chapters 6 & 7**.

Ultimately, the development and adoption of BBPs for food packaging applications sits within the wider circular and bioeconomy debate. The drivers behind the growth of bioplastics are complex and vary across countries, and directions and effects of activities in the bio-based sector remain unclear. In the UK, policies aimed at BBPs from a bioeconomy perspective show a lack of connected thinking with wider climate mitigation strategies. In the context of AD specifically, this issue is further undermined by the fact that AD is both a waste management and an energy production system. The link between plastic substitution, increased FW, environmental protection and the provision of clean energy was generally not articulated clearly by civil servants. Priorities set by the waste management/agricultural and business/energy branches of Government could thus come into conflict if a coordinated approach is not adopted soon.

It is imperative that policies promoting the development of biodegradable polymers consider the intended EoL of these materials and how to ensure they are treated accordingly. For BBPs that currently perform well in AD (and industrial composting), efforts should focus on enabling a robust waste management stream to accommodate the increasing volume of incoming biopolymers while ensuring a clean and viable digestate without microplastics. Strengthened dialogue between Government departments (e.g. by setting a hybrid task force) is needed to ensure effective implementation of circular bioeconomy ambitions. Collaboration between industry and academic partners will also ensure that scientific research focuses on addressing real-world issues, effectively blurring the line between theory and practice. Working holistically across academia, industry and government will be key to deliver truly circular practices in the food-energy-waste nexus.

Chapter 6 – Systems framework for biodegradable bioplastic packaging flow

"*You cannot hope to build a better world without improving the individuals.*" – Marie Skłodowska Curie

Part of the content presented in this chapter appears in the following publication:

Kakadellis, S., Muranko, Ż., Harris, Z. M. & Aurisicchio, M. (in review). Closing the loop: enabling circular biodegradable bioplastic packaging flow through a systems-thinking framework.

1. Introduction

So far, the focus of this thesis has been on the technical compatibilities (whether related to ultimate biodegradability or operational processes) of BBPs in co-AD. Yet, as highlighted by the prevalence of the term 'people' during semi-structured interviews with stakeholders conducted in **Chapter 5**, materials as technological innovations imply behavioural changes as well because individuals need to accept, understand and use them appropriately (Steg & Vlek, 2009). A joint food and food packaging waste stream would require a clear separation between biodegradable and non-biodegradable plastics at their EoL (Rujnić-Sokele & Pilipović, 2017). Similar technical properties and visual appearances represent a challenge to the consumer and may contribute to inappropriate disposal behaviours (Taufik et al., 2020), such as placing BBPs in conventional plastic recycling streams or mixing non-biodegradable plastics with FW.

Research on consumer attitudes towards bio-based and biodegradable bioplastics has shown that while public knowledge of bioplastics is low, the perception, particularly of BBPs, is mostly positive (Dilkes-Hoffman et al., 2019a; Sijtsema et al., 2016; Zwicker et al., 2020). However, a strong environmental appeal does not translate into consistent disposal of bioplastic packaging at the EoL (Ansink, Wijk & Zuidmeer, 2022; Dilkes-Hoffman et al., 2019a; Herbes, Beuthner & Rammer, 2018; Taufik et al., 2020), highlighting the complexity and a lack of familiarity with the relevant terminology. Understanding and facilitating the role of consumer behaviour in tackling post-consumer BBP waste, especially for food packaging, is therefore pivotal in designing closed-loop material flows (Camacho-Otero, Boks & Pettersen, 2018; 2020).

To date, literature on pro-environmental behaviour, including recycling, reuse and the adoption of bioplastics has focused on understanding these actions as individual circular behaviours. However, these often fail to consider the chains within which individual behaviours occur (Muranko et al., 2020) and neglect key features of circular behaviour, including the parallel occurrence of alternative behaviours (Ertz et al., 2017; Tassell & Aurisicchio, 2021) (e.g. using both reusable and single-use cups) and the presence of both direct and indirect causal dependencies (Muranko et al., 2020). Mapping individual behaviours taking place across the consumption phase using a behaviour chain approach is important to investigate consumer behaviour and frame it within closed-loop systems (Zeeuw van der Laan & Aurisicchio, 2021).

In addition, non-linear interactions between parts of complex systems, such as the food-energy-waste nexus, are increasingly recognised, calling for systems-thinking in decision-making (Levy, Lubell & McRoberts, 2018) and human behaviour (Schill et al., 2019). As outlined in **Chapter 3 Section 5.4**, BBPs are embedded in a socio-technical system characterised by a complex and dynamic set of interacting elements. Consumers' behaviours both influence and are influenced by a range of systems factors (e.g. convenience, infrastructure, social norms, etc.), in turn enabling or inhibiting the progression of BBPs across the consumption phase, from product acquisition and utilisation to disposal.

2. Study aims

This chapter aims to capture the dynamic relationships that occur across stages of BBP acquisition and utilisation leading to circular disposal behavioural intention, defined as the intention to dispose of BBPs in a FW bin, as a steppingstone towards the following question: how can systems design encourage the realisation of circular behavioural intention as a means of facilitating the flow and value preservation of BBPs?

Rooted in a behaviour chain and complex adaptive system approach (**Figure 15)** and building on previous work on circular behaviours (Muranko et al., 2020; Tassel & Aurisicchio, 2021) and resource flow systems (Zeeuw van der Laan & Aurisicchio, 2021), this study presents a framework of the system elements that enable or hinder consumers from establishing the flow of BBPs across the consumption phase. Informed by focus groups and a systematic literature review, it places the behaviour chain at the heart of a stratified framework to map (1) the consumer interaction with the flow of BBPs as technological materials, and (2) the circular system elements that enable consumers to establish this flow. The emerging model is then expanded upon and quantified (3) through a comparative network analysis based on a survey conducted within two academic institutions in the UK and in California, US to identify which elements are most likely to influence appropriate disposal behaviour in each setting (**Chapter 7**). Thus, while presented separately, this chapter acts as the methodological and structural foundation for **Chapter 7** and implications for policy and system design interventions are mainly discussed in the latter.

3. Methodology

The methodological approach for each step contributing towards building the overall framework is described. Behaviour chain mapping was completed through focus groups, system elements mapping was achieved through a systematic literature review and validated through focus groups.

3.1. Behaviour chain mapping

3.1.1. Theoretical background

A behaviour chain approach was adopted to identify and map the sequence of behaviours performed by consumers throughout the consumption phase (acquisition, use and disposal). A behaviour chain represents a set of individual actions (i.e. behaviours) performed sequentially, where the completion of one action drives the next one in the chain (Muranko et al., 2020). This approach can be a useful tool when investigating the role of consumers in circular material flows (Zeeuw van der Laan & Aurisicchio, 2019). In the context of this study, the aim was to investigate the impact of consumers on the flow of BBPs and ultimate circular disposal, as defined above.

Behaviour chains are typically modelled using a sequence of nodes (i.e. individual behavioural actions) and paths between them (**Figure 29a**). In addition, they are characterised by a number of key attributes (**Figure 29b-e**), which provide information about the nature and direction of paths (e.g. forking or colliding, forward or return), the nature of behaviours (e.g. primary or secondary), their hierarchy (e.g. macro-, meso- or micro-level), the interdependencies between behaviours (e.g. direct or indirect feedback loops) and performance indicators (e.g. intensity, time, distance) (Muranko et al., 2020). Here, focus was placed on meso-level behaviours, which provide spatio-temporal insights into distinct classes of behaviours (e.g. ‘travel’, ‘read label’, ‘consider EoL’) of interest for policy interventions, without delving into in-depth sequences of individual steps (e.g. ‘hold door handle’, ‘open door’, ‘enter building’). The aim lay predominantly in the representation of the diversity of behaviours, with a clear distinction between primary (i.e. essential) and secondary (i.e. non-essential) behaviours, where the former represents actions required for the behaviour chain to progress, while the latter refers to behaviours that may or may not take place.

Acquisition, use and disposal are all part of the consumption phase, which occurs after the production phase, once a product is placed on the market (Muranko et al., 2020). The end of the consumption phase marks the end of a consumer’s interaction with a product. The disposal behaviour will determine which subsequent waste management route may follow. In some cases, the product may re-enter the consumption phase (e.g. reuse and repurpose models). The consumption phase is divided into six stages, consisting of pre-acquisition, during acquisition, post-acquisition, pre-utilisation, during-utilisation and post-utilisation (i.e. disposal).

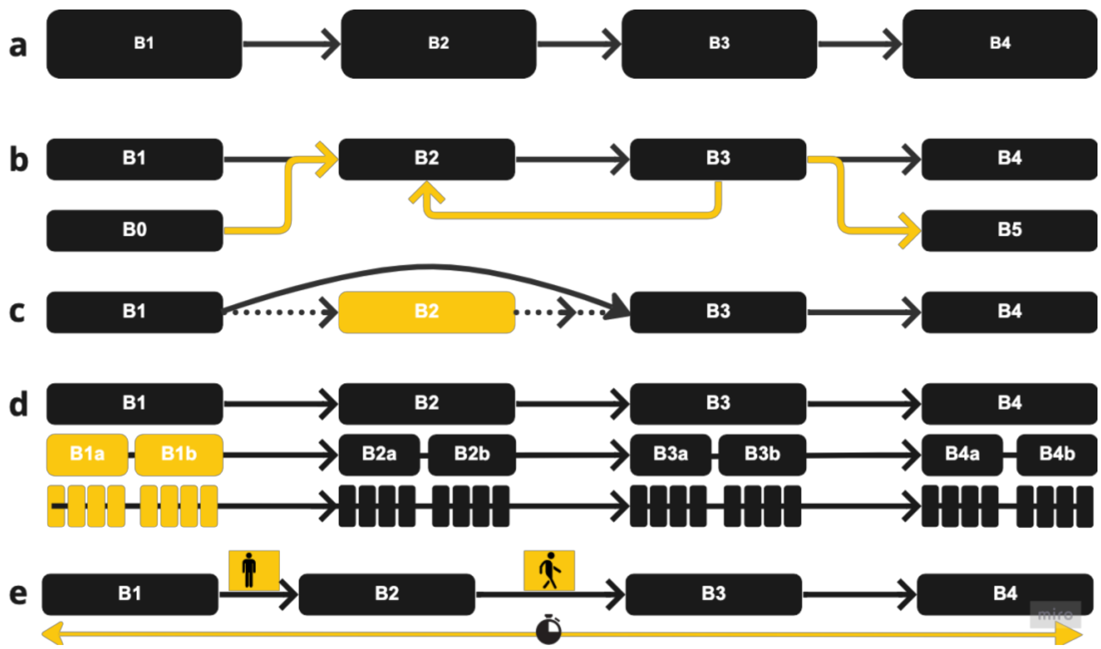


Figure 29 | Behaviour chain key attributes. a: Basic behaviour chain, consisting of four behaviours (B1-B4), performed sequentially from left to right and connected by arcs, or paths. b: Path type and direction; collapsing (left), return (centre) and forking (right). c: Behaviour type; behaviour 2 (B2) is optional and performing this behaviour will not increase the complexity but not the progression of the chain. d: Level of behaviour; B1-B4 (in this case only) high-level, macro-behaviours, and can be further broken down into meso- (B1a, B1b, etc.) and micro-behaviours (bottom). e: Chain performance indicators. As an example, time and distance are represented here. Adapted from Muranko et al., 2020.

3.1.2. Focus group participant recruitment and data collection

In focus groups, the goal is to capture a homogeneous audience, but with sufficient variations among participants to allow for a broad range of views and experiences (Onwuegbuzie et al., 2009). Therefore, participants were recruited during a sustainability workshop focused on plastic pollution as part of a science and arts festival in autumn 2021 at Imperial College London. This was to ensure that participants were somewhat familiar with the notion of BBPs prior to the workshop, but not necessarily knowledgeable on biodegradation and waste management streams for food and organic waste.

Ethical approval by the authors' institutional research ethics committee was granted ahead of the study (registration number: 21IC7191). Both online and in-person focus groups were recorded and a unique reference number was assigned to each participant in the transcription. The original audio recordings and written content were destroyed once transcription was completed. A participant information sheet and a consent form were sent to each participant ahead of the focus group. One facilitator and one observer attended each focus group. Each focus group lasted an hour and a half and was split into two parts. In the first part, participants were shown a behaviour chain template and asked to reflect on their own experience through the chain, based on a product of their choice (or prompted by the facilitator if no example came to mind). The template only showed assumed essential behaviours (e.g. ‘travel to shop’, ‘choose product’, ‘pay for product’). Input from participants confirmed these behaviours and enabled to enrich the sequence and explore deviations from the theoretical baseline chain. A group discussion followed, which helped transition to the second part of the focus group, where participants were asked to reflect on their behaviour chain and identify factors influencing it.

Departing from the sequence of individual consumer behaviours recorded by participants (online: n_1 = 4, in-person: n_2 = 6), comprehensive chains for each BBP category were mapped (**Figure 31**). The chains represented a key methodological step in the framework development. Indeed, the primary aim of the focus groups (through behaviour chains) was to validate/complement factors identified through the literature review and helped refine the terminology adopted in the subsequent survey (**Chapter 7**). Nevertheless, the chains also helped gain first-hand behavioural insights in the context of BBP purchase, use and disposal, thereby contributing to the novelty of this research study and supporting the interpretation of the comparative network analysis.

3.2. System elements mapping

3.2.1. Systematic literature review

To formally capture the system elements identified in the focus groups, systematic literature review was conducted (Collaboration for Environmental Evidence, 2013) in the context of consumer behaviour and BBP disposal. The search was conducted on June 21st, 2021, using Scopus as the citation index service. After several rounds of refining the search string was set as:

TITLE (bio*) AND TITLE (role* OR perception* OR attitude* OR barrier* OR challenge* OR issue*) AND TITLE (plastic* OR bioplastic* OR packaging OR product* OR polymer*) AND TITLE-ABS-KEY (consumer* OR stakeholder* OR public* OR people* OR citizen*) AND TITLE-ABS-KEY (closed OR closing OR loop OR circular* OR flow OR sustainab*)

All references were recorded and processed by the citation manager Mendeley. Articles retrieved from the search (103) were screened for relevance using a priori inclusion criteria. Following refining, 22 studies were kept for critical appraisal and qualitative data extraction. All references were in English and were published between 2011 and 2021. Technical papers on material properties and chemical formulations of bioplastics were excluded. Inclusion criteria were:

- Packaging material: bioplastics (bio-based, biodegradable, and/or compostable)
- Application: food packaging
- Abstract/discussion: included a consumer behaviour component

3.2.2. Structuration of systems elements

For each paper, factors influencing (i.e. enabling or hindering) consumers from establishing the flow of BBPs across the consumption phase were identified. These system elements identified went through several rounds of refining and were categorised according to their broad architectural identity, based on a structural model developed by Müller & Stark (2010) and adapted by Zeeuw van der Laan & Aurisicchio (2020). The architectural types were further adapted to reflect the context of this present study (**Figure 30**), with the aim of modelling the structure that delivers the flow of BBPs across the consumption phase. They provided a high-level structure for the characterisation of system elements and consisted of resources, intrinsic, data, value, infrastructure and policy categories. How and why system elements impact behaviour and ultimately the performance of the chain (i.e. the effective flow of BBPs through the chain) can then be further explored through behavioural models.

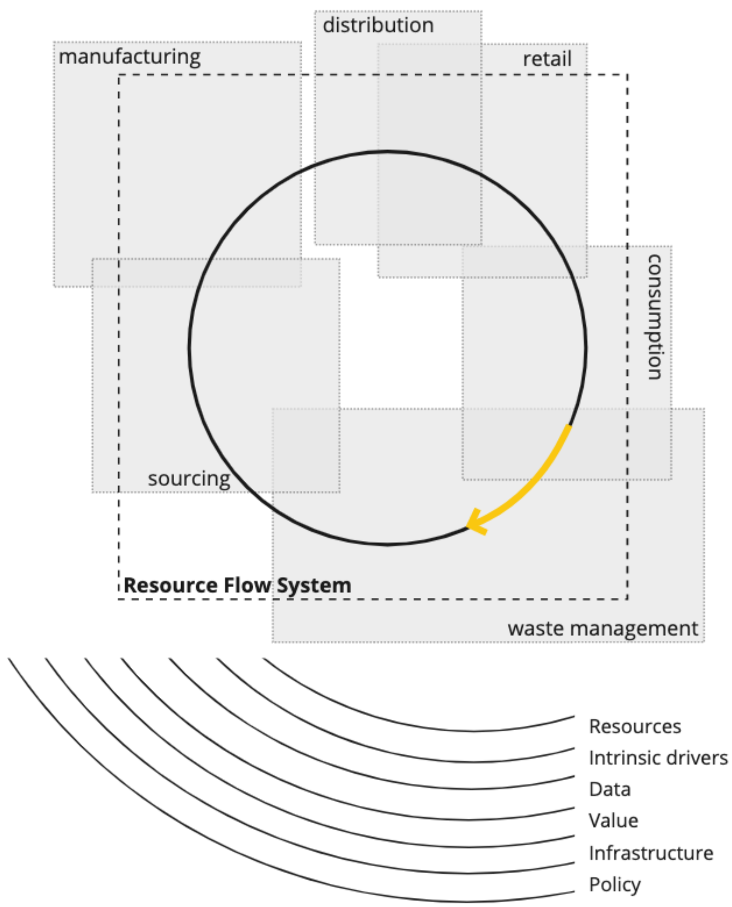


Figure 30 | Resource flow system and structural model of systems elements influencing the flow of resources through the system. While the resource flow system encompasses, sourcing, manufacturing, distribution, retail, consumption and waste management (grey boxes), this study focuses on parts related to consumer disposal, spanning consumption and waste management stages (yellow arrow). Adapted from Zeeuw van der Laan & Aurisicchio, 2021.

4. Results & Discussion

4.1. Consumer behaviour chain

The first stage in building the framework consisted in mapping comprehensive consumer behaviour chains for soft packaging (e.g. films), take-away packaging (e.g. disposable coffee cups or lunch containers) and dual-purpose plastic bags/FW caddy liners. Packaging products represent the highest share of the bioplastics market (European Bioplastics, 2021), and both focus groups and survey data validated these products as the predominant applications, with the majority of respondents interacting with BBPs on-the-go.

The sequence of individual consumer behaviours drawn by participants were merged into a final chain to include all possible behaviours for each BBP category. The resulting chains are displayed in **Figure 31**. Each chain unfolds from left to right and contains both essential, or primary (black circles) and non-essential, or secondary behaviours (black circles with blue contour). The chains exhibited common attributes of circular behaviour chains (**Figure 29b-e**). The range of secondary behaviours points at the complexity of the chain, inasmuch as the inclusion or omission of any of the secondary behaviours will lead to a distinct behaviour chain. The presence of forking paths shows the non-linearity of behaviour chains, which has been identified as a common attribute of circular behaviour chains (Muranko et al., 2020). It is worth distinguishing between secondary behaviours and alternative behaviours (black circles with yellow contour); whilst the inclusion of a secondary behaviour will simply add another action required for the progression of the chain, the presence of an alternative behaviour leads to a separate path, and thus a different outcome.

An alternative behaviour will commonly interrupt the progression of the chain, which will end prematurely, since the chain is mapped from the perspective of BBP flows. For example, if a consumer decides to opt for plastic-free packaging or brings their own reusable take-away crockery (e.g. a reusable coffee cup), there will be no further flow of BBP for that scenario. These alternative behaviours highlight that multiple types of circular behaviour can be performed by consumers (Ertz et al., 2017). In addition, consumers may not always have the choice of choosing their preferred packaging (Fogt Jacobsen, Pedersen & Thøgersen, 2022), e.g. only conventional plastics may be provided for take-away and/or the produce of interest may already be pre-packaged, so that alternative paths may also take place passively. Mapping these alternative paths is important, because it represents the first step in identifying the likely factors responsible for these parallel paths to take place (Tassel & Aurisicchio, 2021).

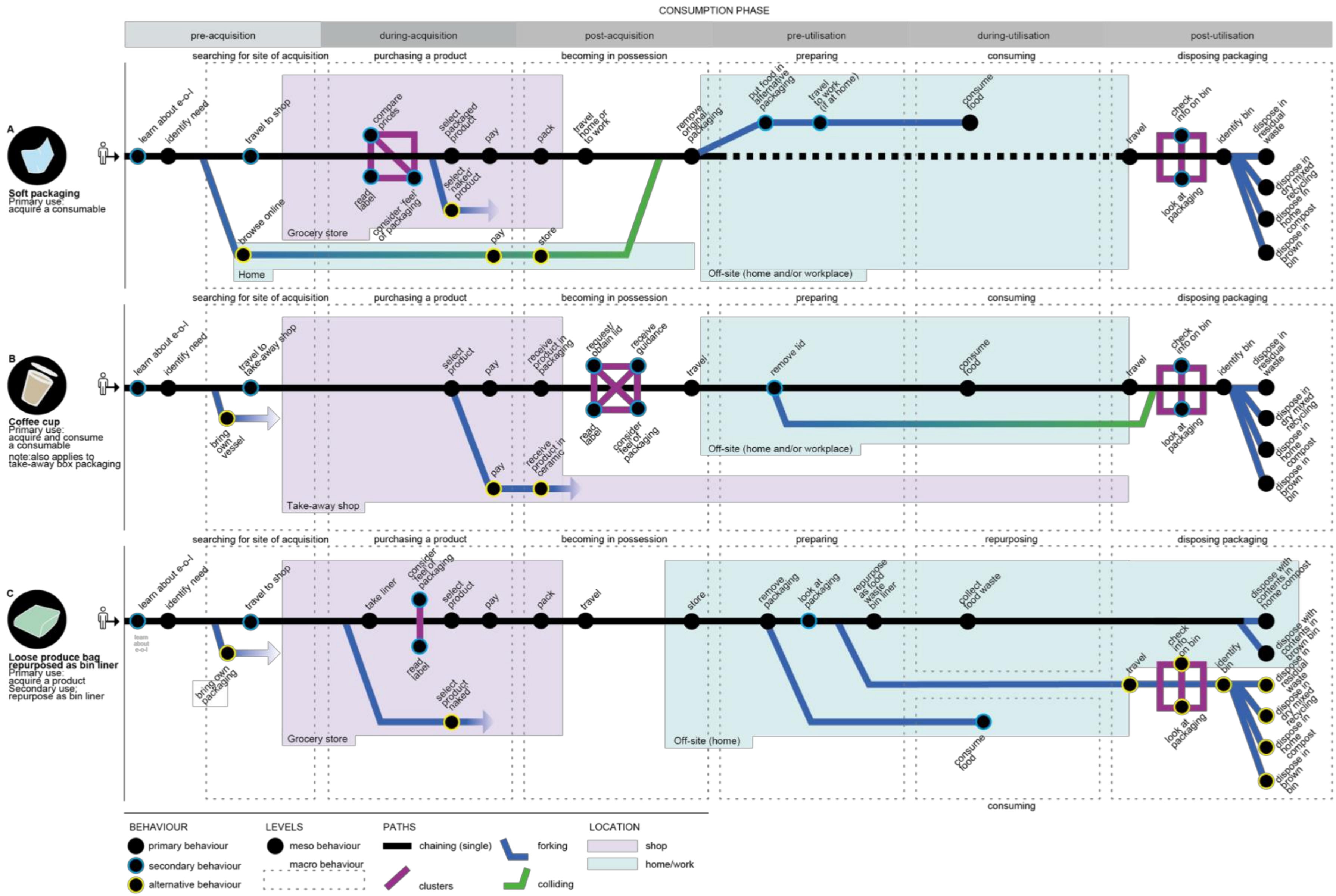


Figure 31 | Consumer behaviour chains for three biodegradable plastic food packaging products under different scenarios. Comprehensive behaviour chains were informed by individual chains drawn by focus group participants (by hand or digitally recorded on Miro) and refined based on logic. Chain attributes were informed by previous work on behaviour chain characterisation and visualisation (see **Figure 29** and Muranko et al., 2020). First built in Miro and refined in Adobe Illustrator. Reproduced from Kakadellis et al. (in review).

All chains mapped involved some degree of travel; acquisition on one hand and consumption and disposal on the other hand often take place at different times and locations (Muranko et al., 2020). This factor adds another layer of complexity and may make appropriate disposal behaviour more challenging, especially for take-away packaging, where consumption and subsequent disposal often take place on-the-go with unpredictable waste infrastructure (Fogt Jacobsen, Pedersen & Thøgersen, 2022). Whilst the first stages of the COVID-19 pandemic may have initially reduced overall on-the-go sales, thereby reducing the need for take-away packaging, many businesses have shifted to take-away or home deliveries, which may have resulted in an increased use of single-use packaging (EEA, 2021). Furthermore, increasingly catering services are shifting to single-use disposable packaging even for on-site consumption (Kochańska, Łukasik & Dzikuć, 2022). This may represent an opportunity to enhance appropriate disposal of BBPs by combining acquisition, use and disposal in a closed-loop system. Nevertheless, this may be a controversial and misleading direction, given that the pandemic has already led to many delays in policies related to the reduction of single-use plastics (Vanapalli et al., 2021) and that traditional tableware e.g. ceramics already offer a circular approach for on-site consumption.

The focus groups showed that participants tended to consider packaging EoL only at the post-consumption stage, unless a packaging-free or reusable packaging option was deliberately chosen, in which case this consideration preceded consumption. The more engaged and informed a participant appeared, i.e. the more they assumed responsibility for reducing their use of plastic (especially single-use) and understood the nuances of bioplastic terminology, the earlier in the behaviour chain they considered EoL. Intriguingly, one participant said they paid attention to whether a product was biodegradable or not (e.g. biodegradable wet wipes), but not in the case of packaging (where the primary function is to protect the product itself), suggesting a cognitive separation between biodegradable products on one side and packaging on the other.

Disposal may also take place more than once at various stages of the behaviour chain, when a given item of packaging is composed of multiple components (e.g. container with lid or sleeve) or when transferring packaging (e.g. unwrapping fresh produce before storing it and/or wrapping it again in alternative packaging at a later stage). The EoL route chosen for BBP disposal varies among consumers and included all possible waste management options (except for littering, which is not considered a waste management strategy, but as environmental pollution). Quantifying the likelihood of each of these disposal behaviours, as well as identifying potential factors influencing these behaviours, was further explored through a network approach **Chapter 7**.

4.2. System elements framework

Mapping behaviour chains represented a key stage in framing BBPs in their social and behavioural context; it provided an entry point to identifying system elements associated with consumer behaviour and helped a practical interpretation of the resulting network of system elements. Following the characterisation of behaviour chains, a formal appraisal of its influencing factors (i.e. system elements) was sought. A systematic literature review covering consumer behaviour and circularity in the context of BBPs was conducted and 22 studies were retained for qualitative data extraction to validate and complement the system elements identified through focus groups. The resulting hierarchical classification of system elements gathered from focus groups and the literature review can be seen in **Table 18** and consists of:

- 6 overarching *system categories*, which represent the broad class of system elements based on a fundamental system structure (macro level) from previous work on resource flow mapping (see **Section 3.2.2**);
- 18 *system elements* (meso-level), stemming from the literature review and terminology of which was some cases refined through behavioural models, such as the TPB (Azjen, 1991) and the COM-B models (Michie, van Stralen & West, 2011), which also contributed towards the classification of sub-elements;
- 35 *system sub-elements*, which further specify the underlying constructs, where applicable (micro-level). In some instances, these were split to ensure content validity (e.g. personal obligation was split into personal responsibility and personal morality).

Table 18 | Structural framework of system elements. The framework consists of six broad categories, which are mutually exclusive. Where applicable, system elements were further divided into sub-elements. The constructs and corresponding questionnaire items are listed on the far-right column. The classification of elements and sub-elements within the *intrinsic consumer drivers* category was supported by well-characterised behavioural models, including the theory of planned behaviour (TPB) and capability-opportunity-motivation behavioural model (COM-B). The *policy* category was not included in the subsequent survey due to the difficulty of capturing its system elements through questionnaire items and was instead addressed through a case study approach. Reproduced from Kakadellis et al. (in review).

Categories	Category definition	System elements	System sub-elements	Graphical model constructs (Chapter 7)
Resources	Captures design features for both material and product states	Sensorial features	Texture	1
			Smell	2
			Visual appearance	3
		Technical features	Durability	4
			Geometry	5

Categories	Category definition	System elements	System sub-elements	Graphical model constructs
Intrinsic consumer drivers	Refers to psychological factors, as opposed to all other categories that are contextual factors	Effort	Simplicity	6
			Convenience	7
			Time	8
		Knowledge	Knowledge of terminology	9
			Knowledge of disposal	10
		Attitudes	Ambiguity	11-12
			Apprehension	13-14
			Trust	15
		Social norms	Perceived social pressure	16
			Social membership	17
		Personal norms	Self-identity	18-19
			Personal obligation	20-21
		Habit	N/A	22
Data	Information provided to the consumer to increase awareness, communicate and assist them through the behaviour chain	Marketing	Problem/mission-oriented	23
			Material/product-oriented	24
		Labelling	Written content	25
			Visual content	26
		Signposting	N/A	27-29
		Consumer education	N/A	31-32
Value	Economic worth of biodegradable plastics and the market space they operate in	Cost consideration	N/A	33-34
		Broader value chain	N/A	35
Infrastruc-ture	Encompasses the basic physical structures and logistics needed for capturing and treating post-consumer waste	Treatment system	N/A	36
		Collection system	Access	37
			Uniformisation	38
Policy	Overarching legislative and regulatory framework (at international, national and institutional/business level)	Market interventions (taxes and subsidies)	Landfill taxes	N/A
			Plastic packaging taxes	N/A
			Renewable heat/electricity subsidies	N/A
		Soft instruments (voluntary agreements)	Public-private partnerships	N/A
			Institutional goals	N/A
		Direct regulation (bans and caps)	Mandatory separate food waste collections	N/A
			Extended Producer Responsibility	N/A
			Caps on non-recyclable and/or non-biodegradable (plastic) packaging	N/A
			Technical/biological recycling targets	N/A

While the policy category and its constituting elements form an integral part of the system studied, as depicted in **Table 18**, they differ from all other categories inasmuch as they remain intangible to consumers at the time of acquisition, use or disposal. They exert their influence *indirectly* through the other system elements (e.g. new legislation on mandatory separate food waste collections translates, at the consumer level, to the introduction of food waste bins, which consumers will *directly* interact with). Importantly, they also represent intervention leavers sought *as a result of* the identification and characterisation of the most important system elements acting upon disposal circular disposal intention for BBP disposal, as will be explored in **Chapter 7**.

5. Conclusions & Future work

This chapter presented a systems framework for mapping and structuring (1) the consumer behaviour chain across stages of acquisition, use and disposal of BBP food packaging and (2) the system elements that enable or hinder the material flow to progress across the chain, through focus groups and a systematic literature review. Mapping the consumer behaviour chain lay the foundation of the framework and provided an entry-point to identify system elements that exist within that system. The framework enabled the characterisation of both consumers' actions and the resource of interest (BBPs), as well as their embedment within the wider system in which they exist.

The focus groups conducted in this study were limited to a small and partially opportunistic sample. While the range of behaviours and disposal routes for BBP waste considered by participants pointed at data saturation, additional focus groups should be conducted to consolidate the behaviour chains mapped and ensure they can be applied more broadly. Nonetheless, the identification of system elements in the context of BBP disposal were corroborated by a systematic literature review, thus increasing the reliability of the framework.

So far, the framework has provided qualitative insights only, with limited insights into which system element(s) might be most effective in driving the flow of resources forward, and which behaviour(s) in the chain they are associated with. Gaining a more dynamic view of how system elements interact with each other and how they influence behaviours, in particular circular disposal behavioural (i.e. the disposal of BBPs alongside FW/organic bin), would help inform policy and enable the design of intervention strategies.

Thus, while this chapter (**Chapter 6**) focused on structural and methodological components of the system, the next chapter (**Chapter 7**) builds on the emerging model and aims to capture the dynamic interactions that occur in the system by applying the model in practice through two case studies conducted at two academic institutions, Imperial College London and the University of California, Davis.

Chapter 7 – The role of system elements in circular disposal behavioural intentions: a comparative case study analysis

"*Is there anyone so wise as to learn by the experience of others?*" – Voltaire

Part of the content presented in this chapter appears in the following publication:

Kakadellis, S., Muranko, Ż., Harris, Z. M. & Aurisicchio, M. (in review). Closing the loop: enabling circular biodegradable bioplastic packaging flow through a systems-thinking framework.

The survey conducted at the University of California, Davis was supported by Professor Ned Spang and Professor Gail Taylor, as part of an Overseas Institutional Visit (OIV).

1. Introduction

The emerging systems framework developed in **Chapter 6** on consumer behaviour and BBP disposal (hereafter referred to as 'the framework', unless stated otherwise) highlighted the complexity and diversity of behaviours taking place across the consumption phase, as well as the plethora of system elements that act upon these behaviours. In line with the non-deterministic theoretical background underlying the framework, these system elements influence behaviour in a non-linear fashion (Iacovidou, Hahladakis & Purnell, 2021). Thus, exploring the interactions within the system through system dynamics, based on a Gaussian graphical model (**Section 4.4**) represents a more practical approach for identifying the system elements most likely to influence the completion of a given behaviour chain and ultimately appropriate disposal behaviour for BBP waste.

Two academic institutions, Imperial College London and the University of California, Davis were chosen as medium-size case studies for comparative network analysis, each with a distinct geographical, socio-cultural and political context and selected for their representative and critical/atypical nature respectively (**Section 4.1**). Using a public academic institution as study boundary provides several advantages. As a public body, they both help forge and are forged by the norms and policy directions of a given cultural, geographical and temporal context (Vogt & Weber, 2020). At a wider societal scale, a university represents a relatively small, tangible entity that can nonetheless provide a significant enough sample size for appropriate statistical power in subsequent analysis. At a more local scale, a university can be seen as a relatively closed-loop system, with a distinct culture and institutional strategy that will have a direct impact on internal activities, including – relevant to this research – material procurement, catering and waste management. In the next section, the contextual setting of each case study site is reviewed ahead of the network analysis.

2. Social, cultural and policy landscapes of case study sites

2.1. Imperial College London

Imperial College London (ICL), officially Imperial College of Science, Technology and Medicine, is a public research university located in South Kensington, London, UK. It was established in 1907 by royal charter under Victorian influence in an aim to create an education and cultural hub following the success of the Great Exhibition that took place in Hyde Park in 1851. Since then, ICL has achieved world-class reputation in science, engineering, business and medicine and is particularly renowned for its international community and global network of collaborations within academia and industry, as well as a flourishing entrepreneurial space (The Times Higher Education, 2022). In 2021, ICL was home to 1,348 academic staff, 2,409 research staff, 4,168 support and/or administrative staff and 22,445 students, of which 11,285 undergraduates and 15,040 postgraduates (Imperial College London, 2021).

In December 2020, following the UK's announcement (in 2019) of its commitment to achieve net-zero GHG emissions by 2050 (HM Government, 2021), ICL published its first Sustainability Strategy 2021-2026 (Imperial College London, 2020). The strategy outlines how research, training and innovation will be fostered under the institution's 'Transition to Zero Pollution' theme as well as how it intends to transform the campus and manage its resources to become a net-zero carbon institution by 2040 (Imperial College London, 2020). As part of this strategy, a Sustainable Food & Drink Policy was also developed to guide food procurement and the menus developed as well as to design programs to increase awareness on the environmental impact of food (Imperial College London, 2022) – FW generation, however, is not discussed. Intriguingly, the strategy provides no tangible targets and instead lists several commitments to "*reduce waste sent to landfill from all College sources by being efficient in use of [its] resources and in reusing and recycling unwanted materials*" (Imperial College London, 2020). This includes plans to:

- Continue to replace plastic and single-use items throughout catering operations;
- Encourage the use of refillable drinks containers by continuing the 15p disposable coffee cup levy (introduced in November 2018) and introducing more water fountains;
- Reduce the number of food deliveries made to the College to a minimum by 2030.

Progress will be monitored through the following measures:

- Reduction in the number of plastic and single-use items used and sold through ICL's catering outlets;
- Reduction in waste produced from catering operations (in t/year).

As a waste stream, FW represents 4% of total waste generation at ICL (Imperial Estates Facilities, 2022a), though this value only captures post-consumer waste. Currently, pre- and post-consumer FW from South Kensington's main catering outlets is collected and turned into a liquid in a maceration unit over three days before taken to an AD plant in Buckinghamshire (Imperial Estates Facilities, 2021). The system does not allow any packaging waste, including compostable plastic food packaging. Despite ICL's Sustainability Strategy, there are no plans to extend the collection to other spaces on campus (or beyond the main campus in South Kensington) due to the relatively small volumes, based on the argument that the benefits gained from treating FW through AD would be negated by logistical requirements (Imperial Estates Facilities, 2021). FW from student halls is collected separately by a third party and treated through a decentralised composting system.

The Sustainability Strategy adopted by ICL reflects UK waste management policies, which are themselves historically built on the concept of the waste hierarchy introduced under the EU's Waste Framework Directive (2008/98/EC and its revised version (EU) 2018/851). The waste hierarchy requires that waste management practices first consider prevention, preparation for reuse and recycling, followed by other methods of recovery (e.g. energy recovery through incineration) and, provided there are no sound alternatives, disposal (i.e. landfilling). More recently, following its departure from the EU, the UK published its Environment Bill, which became law in 2021 with the Environment Act (HM Government, 2022). The Act commits the Government to roll out separate FW collections from households and businesses by 2023, in a pledge to eliminate FW entering landfills by 2030 and to recycle at least 65% of municipal solid waste (MSW) by 2035 (see **Chapter 5 Section 1**). Taking a further step, Wales and more recently Scotland announced a moratorium on the building of new incineration plants (The Scottish Government, 2022; Welsh Government, 2021). Similarly, the European Parliament voted for incinerators to be included in the EU Emissions Trading System from 2026 (European Parliament, 2022), which will make incinerators subject to carbon taxes alongside other major emitters, indirectly incentivising higher tiers of the waste hierarchy.

On a local scale, the city of London has set a net-zero carbon target by 2030 (Greater London Authority, 2018). While consumption-based emissions, including those associated with post-consumer waste, are beyond the scope of this target, the Mayor of London has partnered with London boroughs to improve waste and resource management (ReLondon, 2022). The waste management ecosystem within the Greater London urban area is a particularly complex one, given that each of its 33 boroughs runs its waste management scheme independently, meaning there is no single, uniform recycling standard (**Figure 32**). ICL alone (excluding its North West London Hospitals and Silwood Park campuses) is located within three boroughs: Kensington & Chelsea (K&C) to the South, Westminster to the North and Hammersmith & Fulham (H&F) to the West, with the administrative boundary between K&C and Westminster cutting through the main campus.

Westminster introduced a separate FW recycling service in November 2019. From the initial 7,000 households benefiting from the scheme, the service has been expanded to include every household by the end of 2022 (City of Westminster, 2022). K&C trialled a similar scheme in 2018, collecting FW from selected households through weekly collection (The Royal Borough of Kensington & Chelsea, 2022). In November 2021, H&F followed suit, rolling out its trial across a small number of designated streets. Both trials have since then been extended to more areas, although they remain limited to local areas and street-level properties only (Hammersmith & Fulham, 2022).

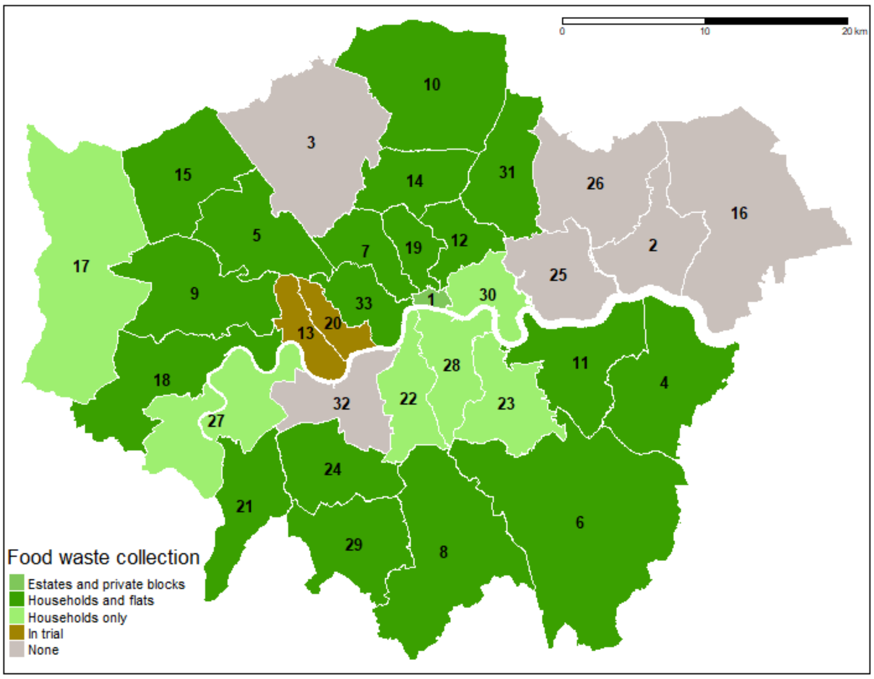


Figure 32 | Food waste collection schemes across London boroughs. Dark green: separately collected food waste from houses and flats; light green: separately collected food waste from houses and flats within houses only (except 1: estates and certain private blocks only); khaki: separate food waste collection scheme trials; grey: food waste not collected separately. 1: City of London; 2: Barking and Dagenham; 3: Barnet; 4: Bexley; 5: Brent; 6: Bromley; 7: Camden; 8: Croydon; 9: Ealing; 10: Enfield; 11: Greenwich; 12: Hackney; 13: Hammersmith and Fulham; 14: Haringey; 15: Harrow; 16: Havering; 17: Hillingdon; 18: Hounslow; 19: Islington; 20: Kensington and Chelsea; 21: Kingston upon Thames; 22: Lambeth; 23: Lewisham; 24: Merton; 25: Newham; 26: Redbridge; 27: Richmond upon Thames; 28: Southwark; 29: Sutton; 30: Tower Hamlets; 31: Waltham Forest; 32: Wandsworth; 33: Westminster. Built in R with shapefile data from gov.uk and food waste collection data from londonrecycles.co.uk and cross-checked via individual borough websites.

All three schemes are limited to FW only and exclude both packaging (including plastics, applicable to all London boroughs) and garden waste – the latter is collected alongside residual waste in Westminster or separately through a subscription-based fortnightly collection or on-demand booking system in K&C and H&F, respectively. In addition to these services, all three boroughs incentivise households to home compost their organic waste, with information available online on getting started with the composting process.

The collected FW is treated through AD in Hertfordshire, North London and Southwest London in Westminster, K&C and H&F, respectively. However, *how* FW is collected differs, reflecting the aforementioned lack of uniformity in waste management practices. K&C provides compostable bin liners to all households taking part in the collection scheme, with additional rolls of liners available free of charge upon request. H&F requires FW to be disposed in compostable liners or paper bags; two rolls of compostable liners are provided free of charge when a household first joins the scheme but must then be purchased by the household. The borough of Westminster provides residents with one initial roll of liners made of 100% recycled plastic as part of the FW recycling roll-out. The borough allows residents to use both compostable and conventional plastics; these liners are later removed before treatment and sent to an energy-from-waste facility. This may be a source of confusion for households, given that no other plastic packaging items are accepted otherwise.

2.2. The University of California, Davis

The University of California, Davis (UCD), is a public research university in the County of Yolo in Northern California, US, a relatively rural region with extensive agricultural land (**Figure 33**). First established in 1905 as a 779-acre University Farm for the University of California (UC) system, the now 5,300-acre university has kept a strong foundation in agriculture and plant sciences ever since, consistently ranking first in the US and second in the world in agriculture and forestry (QS, 2022). It has expanded over the past century to include a broad range of disciplines, including both natural and social sciences, as well as engineering, humanities, medicine, education and business management. Its School of Veterinary Medicine, founded in 1948, is the largest of its kind in the US and has become a world leader in this field (US News & World Report, 2022).

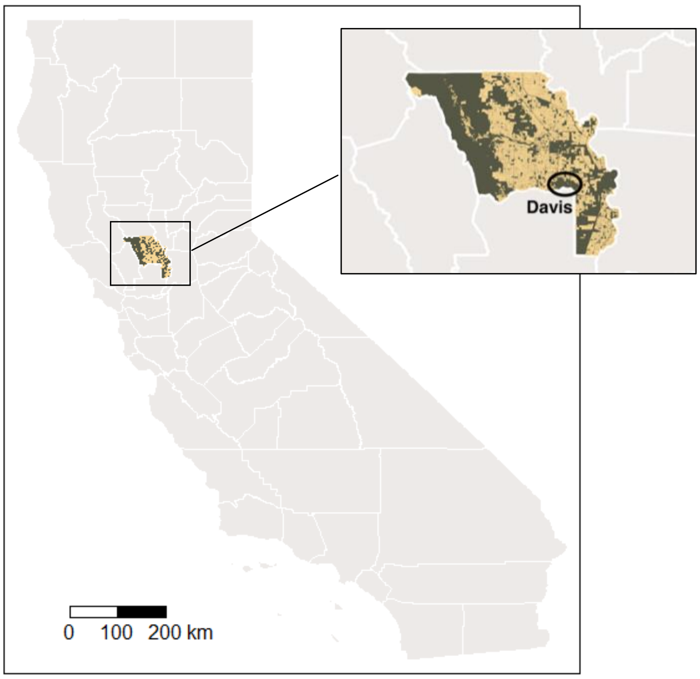


Figure 33 | Californian county boundaries. The city of Davis is in the County of Yolo (coloured area) in Northern California. Yolo is predominantly agricultural land, as displayed in yellow. Map built in R with shapefile obtained from gis.data.ca.gov and yodata-yolo.opendata.arcgis.com.

UCD has also built a reputation for its sustainability ethos, ranked first nationally for its sustainable university campus (UI Green Metric, 2021); the campus core is closed to vehicular traffic, with cycling being the preferred mode of transportation both on campus and in the city of Davis. In 2021, UCD was home to 2,144 academics, 21,486 staff (which include research, support and administrative staff) and 40,031 students, of which 31,162 undergraduates and 8,869 postgraduates (UC Davis Business Intelligence Office, 2022).

As one of the ten student campuses of the UC system, UCD's policies are informed by a centralised UC governing board (the UC Regents). The UC first published a set of sustainability policy principles in 2003, preceding ICL by nearly two decades (University of California, 2022). These principles have gone through multiple revisions, with the latest Sustainable Practices Policy released in March 2022. Guided by a UC-wide pledge to become a carbon neutral institution by 2025, the document establishes goals in 12 areas of sustainable practices: green building, clean energy, climate protection, transportation, sustainable operations, zero-waste, procurement, foodservice, water, health care, performance assessment, and health and well-being (University of California, 2022). As part of its efforts to become a zero-waste institution, it has set to divert 90% of MSW from landfill and reduce per capita MSW by 25% from 2015-2016 levels by 2025 and 50% by 2030 (University of California, 2022). In line with these goals, the UC committed through its policy to the reduction and elimination of single-use items, with a particular focus on plastic packaging, by taking the following actions:

- Immediate ban on packaging foam or expanded polystyrene for take-away food containers;
- Eliminate plastic bags in all retail and foodservice facilities (since January 2021);
- Replace single-use plastics foodware accessory items (e.g. utensils, napkins, cup lids and sleeves, straws, stirrers, etc.) in all foodservice facilities with reusable or locally compostable alternatives at to-go facilities (since July 2021);
- Provide reusable foodware items for food consumed on-site at dine-in facilities and to-go facilities by July 2022 (postponed to 2023 due to the COVID-19 pandemic);
- Replace single-use plastic foodware items with reusable or locally compostable alternatives at to-go facilities by July 2022 (equally postponed);
- Phase out the procurement, sale and distribution of single-use plastic beverage bottles by January 2023, supported by the installation of water refill stations.

The waste hierarchy is clearly acknowledged in the Sustainable Practices Policy. In addition, there is an emphasis on locally compostable, where locally compostable products must be compostable in the local facilities that provide service to the campus, with acceptable products varying by facility.

The increased attention given to zero-waste goals, including waste diversion and minimisation targets, which were first introduced in 2007 and expanded upon in 2018 and 2020, comes in timely, with the state of California introducing the Senate Bill 54, a new legislation requiring all packaging to be recyclable or compostable by 2032 (Allen et al., 2022). SB 54 also establishes an extended producer responsibility programme to fund reuse, recycling and composting throughout the state.

In addition, the 2016 Senate Bill SB 1383, recognising the significant contribution of landfilled organic waste to CH_4 emissions, introduced mandatory separate FW collection from all residents and business from January 2022 (Lara, 2016). This landmark policy has put California at the forefront of sustainability in the context of MSW management, becoming the second state in the US after Vermont to introduce such policy. These two states, alongside Connecticut, Massachusetts and Rhode Island, had already adopted FW disposal bans targeting the commercial and industrial sector, including manufacturers, food wholesalers and supermarkets (WBA, 2018). The new targets include reducing the amount of organic waste sent to landfill by 75% by 2025 and redistributing at least 20% of currently disposed surplus food by 2025 (Lara, 2016).

However, similarly to the UK, given the lack of legislation mandating compost to be of sufficient quality for the farming community, some have warned against the quality of the resulting compost and digestate (Ronayne, 2021). This issue is not trivial, as the usability and marketability of these organic outputs are greatly dependent on the quality of the input and resulting output (WBA, 2018).

The city of Davis' joint food and garden waste collection scheme is run by Recology, a private waste management company. Residents are advised to put FW in compostable bin liners or wrap it in paper bags/newspapers before placing them in the designated mixed organics bin (Recology, 2022). Food soiled paper, waxed carboard and certified compostable plastic packaging are also accepted in the scheme (Recology, 2022). The combined organic waste is transported for anaerobic treatment to the Yolo County landfill site located in the nearby town of Woodland, which includes infrastructure for anaerobic treatment of organic waste, as well as a small composting unit. Depending on the quality of the incoming FW, the feedstock is sent directly to AD or goes through a depackager first, in which case any packaging, including compostable plastics, will be removed from the stream and either landfilled or recycled depending on the material. Since July 2022, a new covered aerobic static pile has been in operation at the landfill site, with part of the incoming feedstock (especially garden or mixed organic waste) being composted directly, instead of undergoing prior anaerobic treatment.

Since 2014, UCD has been treating part of the organic waste generated across catering and research facilities on-site in the Renewable Energy Anaerobic Digester (READ). The plant processes pre- and post-consumer FW (including napkins) from student dining halls, organic feedstocks from animal facilities and grounds as well as from commercial food businesses. Food and food-related waste (e.g. compostable plastics, food-soiled paper and cardboard) from other retail services on campus is collected in mixed organics waste bins and treated through AD at the Yolo County landfill site (UD Davis Biological and Agricultural Engineering, 2022). A small fraction of pre-consumer FW from the Coffee House (a popular student-run catering site on campus) and organic waste from the Pomology Department, the Department of Plant Sciences' greenhouses and several other locations on campus (about 1 t in total) is collected by the student-run Project Compost initiative, which processes the waste through windrow composting at the Student Farm (UC Davis Student Housing and Dining Services, 2022). The resulting compost is sold to the campus and the wider Davis community.

Furthermore, in line with SB 1383, UCD started introducing separate composting bins across campus (i.e. beyond catering sites). From the 19 buildings involved in the pilot study in 2021, the scheme has now been extended to 46 buildings (UC Davis Facilities Management, 2021). Consumer behaviour is a key consideration in the scheme; bins are grouped together at strategic locations to facilitate disposal for all waste streams equally and posters are placed on and above each bin to guide individuals (UC Davis Facilities Management, 2021). Echoing the scheme run throughout the city of Davis, food soiled paper, cardboard and certified compostable plastic packaging can also be disposed of in the composting bins. The collected food and packaging waste is sent to the Yolo County landfill site for AD treatment.

3. Study aims and hypotheses

The contextual setting of ICL and UCD is summarised in **Table 19**. As outlined in **Chapter 6**, the aim of this explorative study was to address the dynamic relationships that occur across the consumption phase of BBP food packaging, with a focus on circular disposal behavioural intentions (as a proxy for disposal behaviour) and investigate the role of contextual setting on these intentions across ICL's and UCD's populations, referred to as P1 and P2, respectively. While the overarching research question and research design are exploratory in nature (Cresswell & Clark, 2006; Maki et al., 2019), the following hypotheses were also tested:

Hypothesis 1: The rate of circular disposal intention for BBP waste will be higher among P2. Informed by literature on barriers to recycling behaviour (Allison et al., 2022a; 2022b; Tonglet, Phillips & Read, 2004), it was hypothesised that UCD's sustainability leadership (**Table 19**) provides an exemplary setting for enabling circular disposal behaviour, leading to a statistically higher likelihood of intending to adopt such behaviour among P2 than P1.

Hypothesis 2: FW recycling habit will be stronger among P2. In psychological research, habit refers to *how* behavioural choices are made rather than their frequency (Steg & Vlek, 2009); habits can only develop if contextual factors remain stable (Klöckner & Blöbaum, 2010). Since P2 is characterised by a harmonious waste management context (**Table 19**), P2 was expected to be statistically more used to recycling their organic waste than P1.

Hypothesis 3: A deeper knowledge of BBP terminology and disposal routes will be exhibited by P2. Following from hypothesis 1 and based on recent findings showing that psychological capability (of which knowledge is a key attribute) predicted household FW recycling (Allison et al., 2022b) and that stronger familiarity with bio-based products increased the correct disposal rate of compostable bio-based packaging (Taufik et al., 2022), P2 was anticipated to exhibit statistically higher knowledge of BBPs and of their disposal than P1.

Hypotheses were first tested through traditional statistical analysis; case study and network analysis approaches were then adopted to gain insights into potential drivers behind them.

Table 19 | Case study contextual settings. Information marked † was obtained from interviews with relevant managers. [1] Imperial College London, 2020; [2] UC Office of the President, 2013; [3] University of California (2022); [4] Imperial Estates Facilities, 2021; [5] Imperial Estates Facilities, 2022b; [6] UC Davis Biological and Agricultural Engineering, 2022; [7] UC Davis Student Housing and Dining Services, 2022; [8] DEFRA, 2021; [9] London Recycles, 2022; [10] Lara, 2016. Reproduced from Kakadellis et al. (in review).

University	Imperial College London of Science, Technology and Medicine (ICL)	University of California, Davis (UCD)
Surveyed population	Population 1 (P1), n_{P1} = 457	Population 2 (P2), n_{P2} = 284
Location	London (UK)	Davis (US)
Tangible target for reduction in (conventional) plastic and single-use items in campus catering	No – minimisation only[1], although most to-go facilities have switched to compostable take-away packaging (cardboard and/or BBP)	Yes – multiple targets, including the replacement of single-use plastic foodware items with reusable or locally compostable alternatives at to-go facilities by July 2022 (postponed to 2023 due to the COVID-19 pandemic) [2]
First institutional sustainability strategy	2020[1]	2003 (last updated 2022)[3]
On-site designated bins for food waste and compostable packaging	Partially – in theory, separate bins are present at the main catering site[4], but none were found at the time of writing. Pre-consumer waste is indeed recycled[5]	Yes
On-site organic waste processing	Partially – Preparation, over-production, and out-of-date (i.e. pre-consumer) food waste† is pre-treated in a small fermentation unit and sent to an off-site anaerobic digestion plant[4]	Yes – anaerobic digestion plant treating food waste from student catering halls and organic waste from animal facilities[6]. Some of the waste from retail services is also composted on-site[7]. The remaining waste is sent to an off-site anaerobic digestion plant†

(Continued on p. 140)

University	Imperial College London of Science, Technology and Medicine (ICL)	University of California, Davis (UCD)
Local municipal context – separate organic waste collection scheme	Mandated nationally from 2023-2025[8], though many London boroughs already have individual schemes in place/in trial. Biodegradable packaging waste is not accepted (other than, in some cases, a compostable bin liner)[9]	Mandated across California since 2022[10], single scheme offered across the city of Davis. The scheme includes garden waste, food waste and any certified compostable packaging waste[†]

4. Methodology

4.1. The case for case studies

Two academic institutions were chosen as medium-size case studies for comparative network analysis, each with a distinct geographical, socio-cultural and political context. Comparative research is an established research strategy that can be helpful for identifying causal and explanatory patterns (Pierre, 2005). It therefore complements the exploratory nature of the GGM and enables further insights to be gained through a deeper understanding of individual case studies and their unique context. The fact that knowledge cannot be formally generalised does not mean it cannot enter the collective process of knowledge accumulation in a given field or in society (Flyvbjerg, 2006). Some have argued that formal generalisation is overvalued as a source of scientific development, while the insights gained from bounded, context-dependent examples (i.e. case studies) remain undervalued (Flyvbjerg, 2006). Context-dependent knowledge can help uncover similarities and differences across distinct settings and inform policy makers in the decision-making process when faced with and addressing similar societal challenges (Krehl & Weck, 2019).

The case study sites were selected for their representative and critical/atypical nature respectively. While a representative case study provides a baseline scenario, a critical case study allows for logical deductions and is well suited to the falsification of proposition (Flyvbjerg, 2006). This approach can identify leverage points most effectively across the consumption phase in order to inform design strategies aimed at supporting consumers as they transition towards more pro-environmental behaviours in the context of plastic sustainability.

4.2. Survey design

The framework described in **Chapter 6** helped develop a valid survey design. Following several rounds of refining within the author's research group and a pilot study on 13 individuals to ensure accuracy and validity of the questionnaire items (see **Appendix Section 3**) while taking survey fatigue into account, a final questionnaire was developed, which comprised of five consecutive sections:

I. An introductory page, outlining the aims of the research and the overall study, the option to enter a draw before/after completing the survey, a participant information sheet and a consent box, which was required for proceeding with the survey;
II. A section exploring types of BBP food packaging commonly used (tick boxes) and the likelihood of choosing a disposal route for these items (5-point Likert scale, from “extremely unlikely” to “extremely likely”);
III. The third section, which constituted the bulk of the questionnaire, presented a series of statements about BBP food packaging and/or their disposal (5-point Likert scales, from “strongly disagree” to “strongly agree”). The statements were split into sub-sections based on their respective high-level categories;
IV. A section on demographics, including age, affiliation to university (e.g. postgraduate student, academic staff member, etc.), gender, access to green infrastructure (e.g. home garden, allotment) and access to a FW collection service;
V. An optional section for participants who wished to enter a draw. For legal requirements under Californian legislation regarding monetary incentives, this section was placed after the introductory page on the UCD questionnaire.

As outlined in **Chapter 6 Section 4.2**, system elements related to the *policy* category in the framework (**Table 18**) differed from all other categories, in that they cannot be directly measured via a questionnaire. They were not included in the subsequent survey and did thus not appear as nodes in the resulting networks. Instead, the role played by policy was addressed by means of all other categories and, importantly, by characterising the contextual setting of each case study site (**Section 2**).

4.3. Survey administration

The survey was conducted within the bounds of ICL and the UCD as two medium-size case studies and administered through the online survey software Qualtrics (https://www.qualtrics.com/uk/) in December 2021 and June 2022, respectively, for a period of 3 weeks. Survey participants were recruited by email through departmental newsletters and internal communications. The survey was open to any member of the university, including undergraduate and postgraduate students, academic, research, teaching, technical/scientific support, professional/administrative and operational staff members. Participants were given the opportunity to enter a draw to win one of ten vouchers valued at £/$10, 20, 50 and 100.

Ethical approval by either institutional research ethics committee was granted ahead of the study (registration number: 21IC7191 NoA1 and 1897980-1 for ICL and UCD, respectively). Responses were anonymous and optional email addresses provided to enter the draw were deleted as soon as the vouchers were distributed and were not included in the data analysis.

4.4. Data preparation and analysis

While traditionally exploratory analyses are carried out through correlation matrices, here a network approach was adopted, using a Gaussian graphical model (GGM) to provide a more visual and meaningful representation of the interactions between systems elements (Bhushan *et al.*, 2020). In a weighted, non-directional network, the structure of the underlying data can be seen through a system of nodes that are connected and interact with each other with varying strength of relationship represented by lines, called edges (Zwicker et al., 2020). In a behavioural context, nodes represent measured variables (questionnaire items, or multiple items aggregated into underlying theoretical constructs), and the edges represent the partial correlations between them (Dalege et al., 2017).

The GGM provides a novel perspective to gain an understanding of the structure of the system studied and insights into when and how a range of factors might influence behavioural intentions and subsequent behaviours (Bhushan et al., 2019; Dalege et al., 2017; Epskamp et al., 2012; Epskamp, Borsboom & Fried, 2018; Zwicker et al., 2020). It is a useful tool in interdisciplinary research, in that it allows the analysis of large datasets that include variables from multiple theories not commonly studied together (Bhushan et al., 2020), as was the case in the present research.

Statistical data analysis was undertaken in R 4.1.1 (R Core Team, 2021) via RStudio (https://www.rstudio.com). Prior to data analysis, any participant who completed the survey in under 270 seconds (4'30 minutes) was removed from the data set to reduce noise, based on the minimal expected time it would take to complete the survey while truthfully responding to all sections. First, trimmed questionnaire data were imported as data frames consisting of 43 columns (one per questionnaire item corresponding to individual system elements and behavioural intentions) with 457 (P1) and 284 (P2) rows each (one per participant), and with values ranging from 1 to 5. A correlation matrix of questionnaire was computed at the item level and fed as input for the network model using the *glasso* algorithm (Friedman, Hastie & Tibshirani, 2008) and visualised using the R package `qgraph` (Epskamp et al., 2012). The *glasso* method estimates partial correlations between each pair of variables conditioning on all other variables. Items that are strongly correlated appear spatially close to each other and form a cluster.

Looking beyond the global structure of the network, the structural importance of individual nodes can be investigated through centrality measures (Dalege et al., 2017). Three centrality measures were computed: strength, betweenness and closeness. Strength represents the direct influence of a given node on the network and is calculated by summing the absolute values of all edge weights a given node has (Dalege et al., 2017). Betweenness and closeness are based on the mathematical concept of shortest path length. The shortest path length between two given nodes refers to the shortest distance between these two nodes based on the edges that directly or indirectly connect them (Dalege et al., 2017). Betweenness measures the number of times a node lies on the shortest path between other nodes (Epskamp, Borsboom & Fried, 2018). Closeness measures how well a node is directly and indirectly connected to all other nodes (Epskamp, Borsboom & Fried, 2018). Betweenness thus represents how the ability of a given node to disrupt the flow of information within the network, while closeness helps identify ‘broadcasters’, nodes that allow information to reach the whole network quickly (Dalege et al., 2017).

Graph density is controlled by a tuning parameter, which forces partial correlation coefficients below a certain threshold to zero (Bhushan et al., 2020), meaning that only coefficients above the given threshold will be displayed. The extended Bayesian information criteria (EBIC) method is commonly used to determine the optimal tuning parameter (Foygel & Drton, 2010), decreasing the number of spurious (i.e. erroneous) partial correlations and thereby leading to sparser (and thus more visually comprehensible) graphs (Friedman, Hastie & Tibshirani, 2008). As with any data analysis approach, a network analysis requires certain assumptions to be made. In this case, the EBIC *glasso* algorithm was chosen to build the partial correlation matrix for the graphical model. Having taken a more conservative approach in order to minimise spurious correlations (i.e. false positives, or type I errors), true correlations may have been dismissed, thereby omitting insightful interactions between individual nodes in the network. However, these limitations do not lessen the potential value of network analyses to better understand the role of system elements on consumer behaviour and to design effective strategies for behavioural change (Zwicker et al., 2020).

Network comparison was investigated by computing the structural Hamming distance in the R package `bnstruct` (Sambo & Franzin, 2009), which represents the distance, in edge terms, between the two network structures. The lower the output, the more similar the networks are. A permutation-based hypothesis test was also conducted to assess the difference between both networks based on several invariance measures (network structure invariance, global strength invariance and edge invariance), using the `NetworkComparisonTest` R package (van Borkulo et al., 2022). Finally, the stability of the estimated network and centrality measures was tested using the R package `bootnet` (Epskamp, Borsboom & Fried, 2018).

4.5. Study limitations

While providing behavioural and structural insights related to circular disposal of BBPs, the study did have several limitations. First, the surveys were limited to members of academic institutions; the data emerging from both networks may not apply to the wider society. Given that different countries have varying levels of familiarity with composting and recycling (Taufik et al., 2020) and educational attainment was shown to influence levels of awareness on the indirect impact of plastics on human health (Barbir et al., 2021), replicating the study across representative UK and US populations and across countries would help strengthen the study.

Certain comments provided by certain participants during focus groups (**Chapter 6**) suggested that respondents may not have understood the scope of the study and may have failed to distinguish between biodegradable from bio-based concepts as well as between food and non-food packaging, which in turn may have distorted the resulting networks. This phenomenon was previously reported by Zwicker et al. (2020), who found that 58% of study participants thought bio-based plastics are biodegradable. Nevertheless, this finding is in itself a valuable outcome and calls for improved education and availability of information to the public, as discussed further below.

To ensure consistency in survey design and enable a side-by-side network comparison, the surveys used were virtually identical across both case studies (apart from minor changes to reflect British/American terminology, e.g. garden/yard, Senior Lecturer/Assistant Professor, etc.). To minimise survey fatigue, questionnaire items were kept to a minimum. In retrospect, some questionnaire items may have lacked specificity, undermining (although only slightly) content validity. For example, the question on access to a FW collection scheme ("Do you have access to a FW collection scheme?") does not provide geographical boundaries (e.g. on/beyond campus) and raises the question of whether constraining the case studies to an institutional setting is meaningful in this case. However, the qualitative part of this study (i.e. the description and analysis of the social, cultural and political landscape of each case study site) helped contextualise each network and distinguish between institutional and higher-level variables (e.g. FW infrastructure on-site vs at home).

5. Results and Discussion

5.1. Survey participant demographics

Overall, 557 and 361 participants completed the survey at ICL and UCD, respectively, of which 457 and 284 were kept post quality filtering (completion time > 270 seconds). The survey captured the following demographics: position at university (**Figure 34**), age (**Figure 35**) and gender (**Figure 36**). In both cases, students accounted for over half of respondents (52.9% and 57.7%, respectively), with postgraduate students representing the largest demographic group (29.5% and 36.6%) (**Figure 34**). The absence of any research staff members from UCD (in grey) may be attributed to the terminology used in the survey ("post-doctoral fellow/research assistant"), which may not have adequately depicted the American university system and the fact that research staff in the US are often highly engaged in teaching.

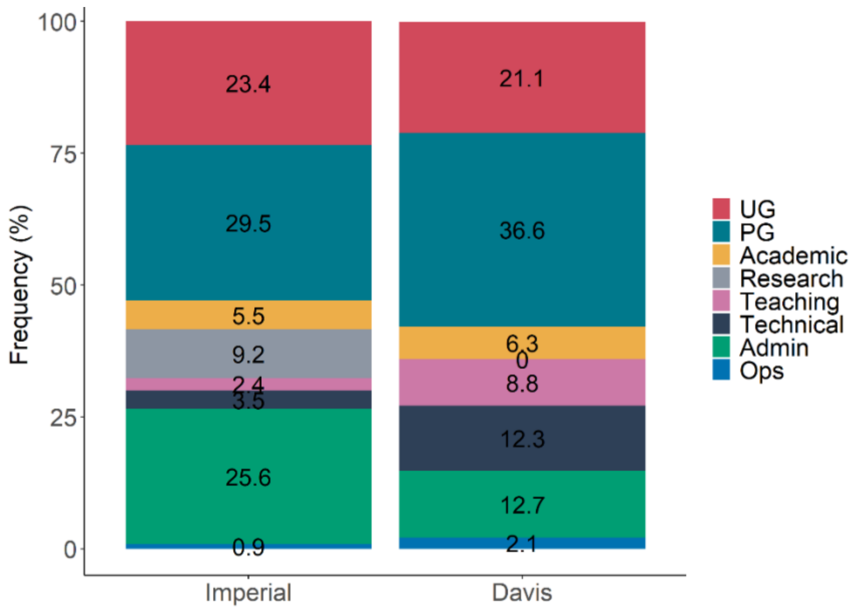


Figure 34 | University role of survey participants (by frequency). UG: undergraduate students; PG: postgraduate students (includes taught and research postgraduates); Academic: academic staff (e.g. lecturer, senior lecturer/assistant professor (UK/US), professor); Research: research staff (e.g. research assistant, postdoctoral researcher); Teaching: teaching staff; Technical: technical/scientific support staff; Admin: administrative staff; Ops: operational staff. n_{P1} = 457, n_{P2} = 284.

The participants belonged predominantly to younger generations (65.2% and 66.5% under 35, respectively) (**Figure 35**), as expected given the size of the student body and the majority frequency of students completing the surveys. ICL participants tended to be younger than UCD's, with 36%.1 under 25, compared to 26.4% at UCD, which could be attributed to more students in the UK undertaking doctoral studies directly after their bachelor's degrees and a more mature postgraduate body in the American system. The age skewness could have also been reinforced by the monetary incentive. All other age groups were similar across both case study sites.

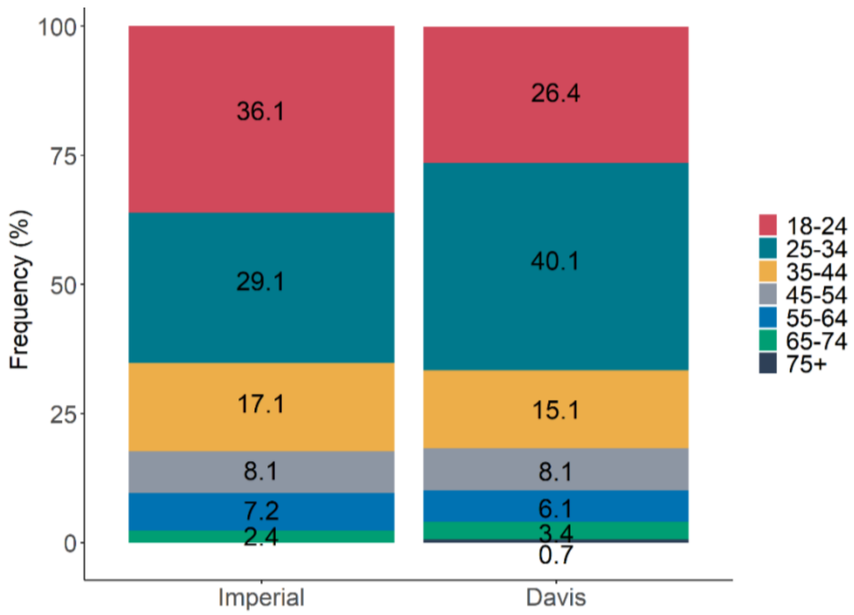


Figure 35 | Age range of survey participants (by frequency). n_{P1} = 457, n_{P2} = 284.

The gender profile revealed a significant female bias, with over 60% of participants self-identifying as female in both surveys (**Figure 36**). While this reflects the female majority (61%) at UCD (based on data for full-time UCD undergraduates) (UC Davis Business Intelligence Office, 2021), it does not adequately portray the male majority (58.2%) of ICL's student body (Imperial College London, 2021). Nevertheless, this was somewhat expected, due to the association of environmentalism with stereotypically female gender roles in line with social role theory (Swim, Gillis & Hamaty, 2020), and given that women are more likely to engage in pro-environmental behaviours (Gatersleben, Murthagh & Abrahamse, 2012).

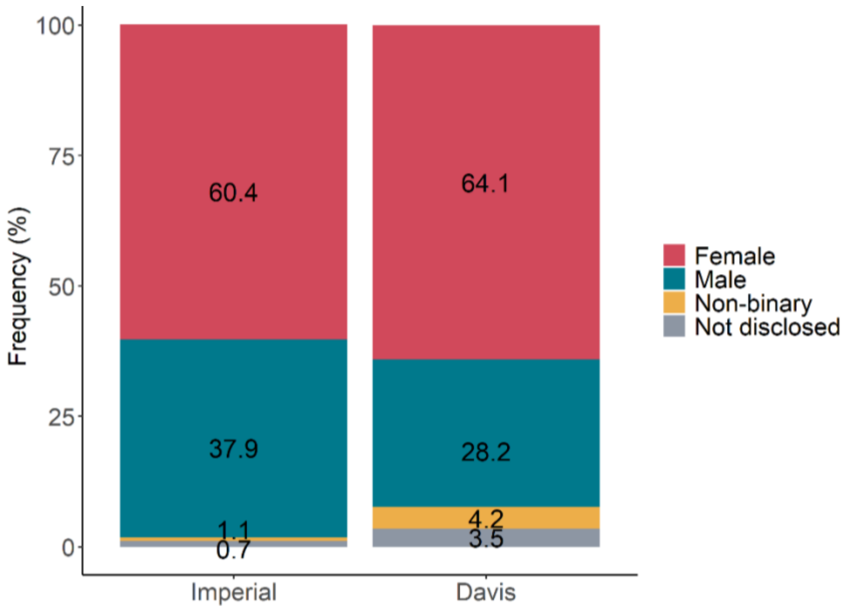


Figure 36 | Gender of survey participants (by frequency). Not disclosed was phrased as 'prefer not to say' in the questionnaire. n_{P1} = 457, n_{P2} = 284.

5.2. Metrics related to organics and organic waste management

Two questions focused on access to organics and organic waste infrastructure: access to a garden/allotment (**Figure 37**), which offers the potential for home composting, and access to a separate FW collection scheme (**Figure 38**). 42.5% of ICL participants indicated that they had access to either a home garden, community garden or allotment (**Figure 37**). This value is uncharacteristically high, given that 21% of households in London do not have gardens (Office for National Statistics, 2020) and that there has been a stark decline of allotment provision, approximately a third of that of a decade ago (Fletcher & Collins, 2020). The fact that nearly half (48.6%) of UCD participants have access to such infrastructure is somewhat less surprising, given that the population density of the city of Davis (2,588 residents/km^2) is more than half that of London (5,666 residents/km^2). There was no statistically significant difference in the frequency distributions of both case study sites (χ^2 = 2.671, df = 1, p = 0.102).

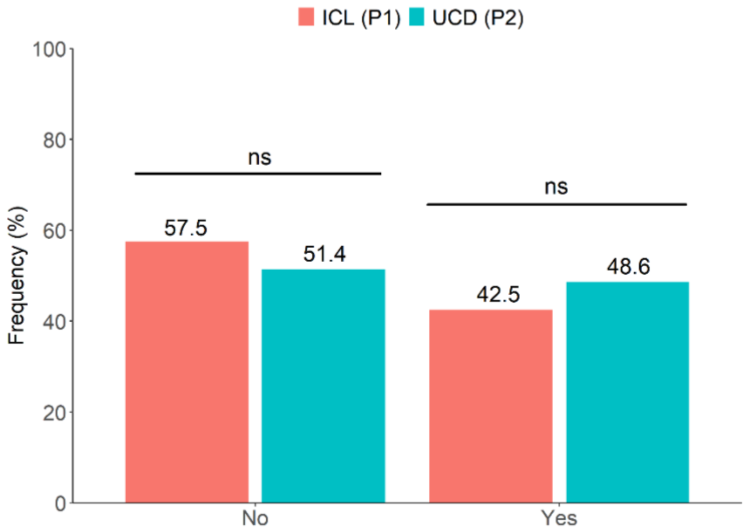


Figure 37 | Access to garden, community garden or allotment (by frequency). Significance of difference in frequency distribution between ICL and UCD populations was tested with Pearson's chi-test (χ^2 = 2.671, df = 1, p = 0.102). ns: non-significant

However, P2 was significantly more likely to have access to a FW collection scheme than P1 (77.8% vs 56.9%, χ^2 = 34.315, df = 1, p < 0.001) (**Figure 38**), as expected from the contextual setting (**Table 19**). While two of the closest boroughs surrounding ICL (where many students reside) do not have comprehensive schemes in place, other boroughs within a larger radius do provide them (**Figure 32**), and separate FW bins are available in university-owned student accommodation. On campus, however, only the main food catering outlets offer separate FW bins. In contrast, in line with SB 1383 legislation introduced in 2016, separate FW collections from all residents and businesses (including academic institutions) have been mandated in California since January 2022. The theoretical 100% frequency was yet not observed, arguably because of the time lag between a policy being introduced, its implementation on the ground and its assimilation by the local community, which requires residents to adopt new waste sorting behaviour.

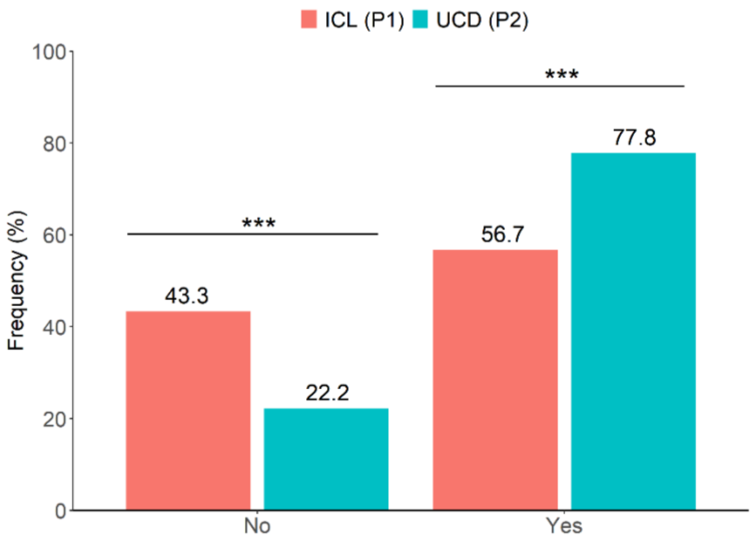


Figure 38 | Access to a separate food waste collection scheme (by frequency). Significance of difference in frequency distribution was tested with Pearson's chi-test (χ^2 = 34.315, df =1, p < 0.001).

5.3. Most commonly used BBPs

As hinted in the focus groups (**Chapter 6**), the most commonly used BBPs reported by respondents were take-away packaging and FW caddy liners (**Figure 39**), though their relative contributions differed between the two populations ($\chi^2 = 48.438$, df = 5, $p < 0.001$). Respondents interacted most frequently with soft packaging (69.6%) and take-away packaging (66.8%) at ICL and UCD, respectively (post-hoc Bonferroni correction tests $p < 0.01$ and $p < 0.001$, respectively). P1 was also more likely to use FW caddy liners ($p < 0.01$) than P2. A minority of respondents stated they were unsure (2.4-5.6%) or did not interact with BBPs on a weekly basis (3.9-5.5%). Other BBPs mentioned by respondents included magazine wrappers (outside the remit of food packaging) and plastic bottles (likely to be bio-based but not biodegradable, e.g. bio-PET or bio-PE bottles). The fact that participants mentioned non food-related and non-biodegradable packaging items suggests there may be some confusion about what the term 'biodegradable plastics' refers to, as highlighted by previous research (Dilkes-Hoffman et al., 2020, Taufik et al., 2020).

5.4. Disposal behavioural intentions

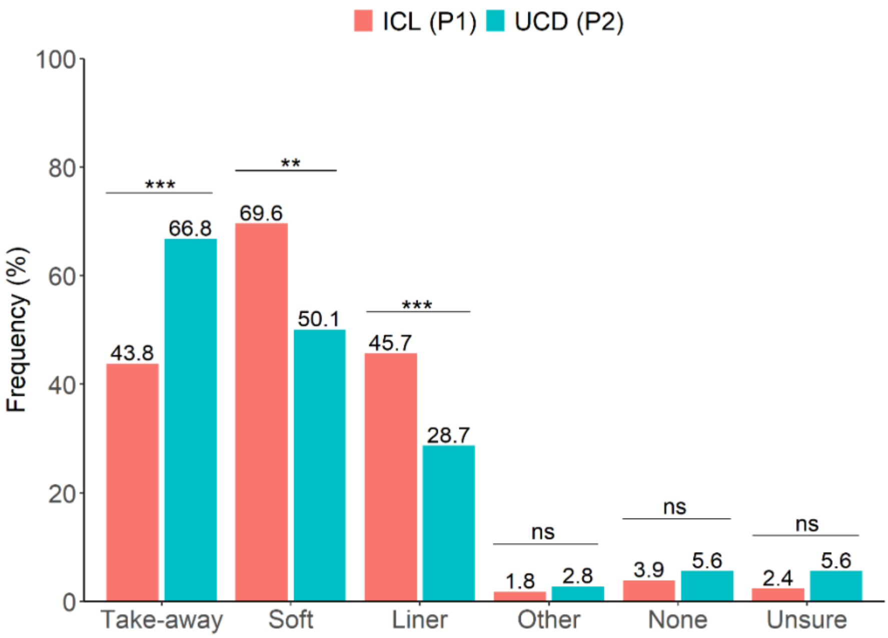


Figure 39 | Most commonly used BBPs among survey participants (by frequency). Take-away: take-away packaging; Soft: soft packaging. Significance of difference in frequency distributions between ICL and UCD populations was tested with Pearson's chi-test ($\chi^2 = 48.438$, df = 5, $p < 0.001$) with post-hoc Bonferroni correction post-hoc tests. ns: non-significant; ** $p < 0.01$; *** $p < 0.001$.

The survey also aimed to independently quantify behavioural intentions for BBP disposal, measured by the likelihood of adopting a given EoL route, including general waste, FW recycling, dry mixed recycling, home composting and the open environment (i.e. littering). Ultimate disposal behaviour is also a complex decision-making process, reflected by the range of EoL options considered by participants across both populations (**Figure 40**). 67.4% [P1] and 60.2% [P2] of respondents reported they were somewhat to extremely likely to dispose of

BBPs in residual waste, while 60.0% [P1] and 59.5% [P2] scored favourably for BBP disposal in dry mixed recycling. The fact that general waste was the most likely disposal route among P1 (and only second to FW recycling by 3.4% among P2) was expected, and is to some extent welcome, since general waste is the preferred option in the absence of a separate FW collection stream (WRAP, 2020). The non-trivial predisposition for dry mixed recycling as disposal route was also expected, given the public's generally low level of knowledge of bioplastic-related concepts (Dilkes-Hoffman et al., 2020). Their resemblance to conventional, recyclable plastics (Taufik et al., 2020) also encourages consumers to resort to habitual disposal routes regardless of plastic type, as previously demonstrated in field experiments (Taufik et al., 2020; Zwicker et al., 2023). Nevertheless, 21.2% [P1] and 18.7% [P2] were extremely unlikely to dispose BBPs in dry mixed recycling, suggesting that a growing level of consumers are grasping the nuances of BBP terminology.

In contrast, frequency distributions differed for FW recycling and home composting (χ^2 = 194.22, df = 4, $p < 0.001$ and χ^2 = 18.681, df = 4, $p < 0.001$), with P2 statistically more likely to opt for FW recycling as EoL than P1 (70.8% vs 45.3%, $p < 0.001$), confirming hypothesis 1. These findings align with the reported frequencies of access to a separate FW collection scheme (**Figure 38**) suggesting that the provision of an appropriate collection infrastructure may foster the adoption of circular behaviour for BBP disposal, especially given that food-contaminated cardboard and certified compostable take-away packaging are accepted in FW bins across UCD's campus and through the municipal waste collection scheme in Davis.

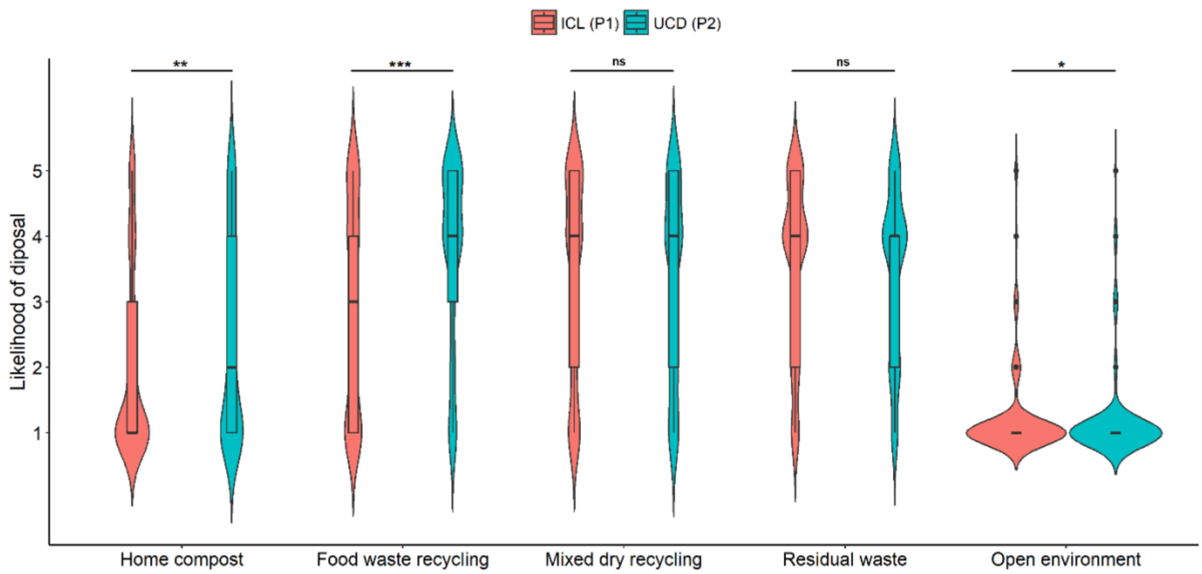


Figure 40 | Disposal routes for BBP waste (by frequency). Significance of difference in frequency distributions between ICL (P1) and UCD (P2) populations was tested with Pearson's χ^2 test with post-hoc Bonferroni correction. The y-axis corresponds to the 5-point Likert scale from the survey, where 1 = extremely unlikely, 2 = somewhat unlikely, 3 = neither likely nor unlikely, 4 = somewhat likely, 5 = extremely likely. n_{P1} = 457, n_{P2} = 284. Reproduced from Kakadellis et al. (in review).

While both populations were reluctant to home compost their BBP waste, P1 was significantly less inclined to do so than P2 (58.0% vs 46.1%, somewhat to extremely unlikely, $p < 0.01$). It is important to note that not all industrially compostable BBPs are home compostable, and thus the reluctance of opting for home composting as disposal route for unspecified BBPs may be legitimate, and even desirable. A recent citizen science-based study has shown that 60% of BBPs certified as 'home compostable' did not disintegrate (Purkiss et al., 2022), strengthening the argument that home composting is not a suitable EoL for biodegradable or compostable packaging in the UK.

Finally, whilst 86.4% [P1] and 90.1% [P2] were extremely unlikely and another 7.4% and 2.5% somewhat unlikely to dispose of BBPs in the open environment, 2.6% and 3.6% were at least partly likely to do so. Whilst low, at wider scale such frequency can have devastating consequences on the natural environment (MacLeod et al., 2021). It is possible that the neutral term 'open environment' may have been misunderstood by some participants, and the more loaded term 'littering' may have led to a more conservative number. Participants may have also perceived BBPs as capable of being fully assimilated by natural organisms regardless of environment conditions, which remains a common misconception (Dilkes-Hoffman et al., 2019a). Nonetheless, it must be noted that littering is not a waste management strategy and should not be confused with logistical issues (e.g. infrastructure provision and design) and was thus not included in the network analysis.

5.5. Comparative network analysis

The behavioural intentions for BBP waste management quantified above were combined with questionnaire items covering system elements (**Table 18**) and analysed through a network analysis, based on a partial correlation matrix. The graphical representations of the resulting networks are displayed in **Figure 41A** (ICL) & **B** (UCD), hereafter referred to as N1 and N2.

Overall, nodes related to the same construct (indicated by nodes of the same colour) tended to form a cluster and displayed stronger links between them than with other nodes, a common phenomenon known as network homophily (Şimşek & Jensen, 2008). The networks shared a similar structure overall, as shown by comparable clustering patterns. The fact that the P2 network exhibited more disconnected nodes could be attributed to P2's smaller sample size (n_{P1}: 457, n_{P2}: 284); this did not, however, impact network connectivity. Despite a structural Hamming distance of 231 (corresponding to the number of edge insertions, deletions or weight changes needed to transform one graph to another), the differences in global strength and edge weights were not significant ($p > 0.05$ resulting from the permutation tests for each measure), suggesting that interactions between system elements remained consistent across networks.

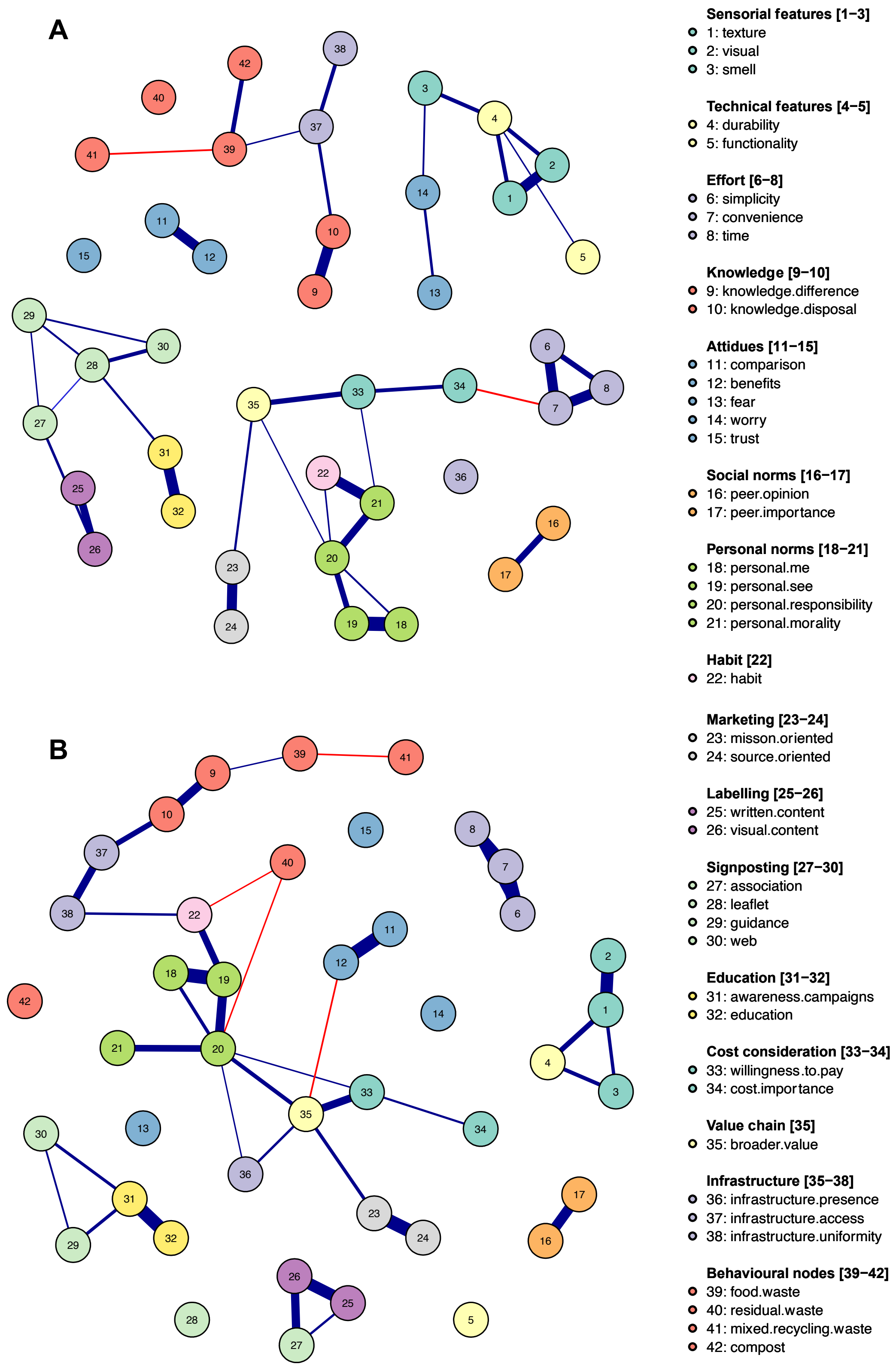


Figure 41 | Gaussian graphical models for ICL (A) and the UCD (B) networks. The networks are based on partial correlation matrices using the *glasso* algorithm and visualised using the R package `qgraph`. Each node (circle) represents a questionnaire item and its corresponding system sub-element. The thickness of the edges (lines) is directly proportional to the strength of the correlation. Blue and red edges represent positive and negative partial correlations, respectively. Nodes are coloured to reflect the construct family they are part of. Note that only 12 colours were available in the 'pastel' colour scheme, so that some unrelated categories share the same colour. Reproduced from Kakadellis et al. (in review).

In both cases, the nodes related to social norms (#16 & 17) were only linked internally and were otherwise isolated from the network (in addition to the nodes related to ambiguity (#11 & 12) in N1. The lack of connectivity between social norms and other nodes is somewhat surprising, given the importance of descriptive normative beliefs on pro-environmental behaviour, including recycling (Steg, Perlaviciute & van der Werff, 2015; Thomas & Sharp, 2013). However, in a study on the role of social norms on energy conservation, participants rated such norms as least important, in contrast to a subsequent field experiment showing that these norms achieved greatest behavioural change (Nolan et al., 2008). Therefore, peer pressure can be a powerful driver of change, yet its influence tends to be underrated by self-reporting respondents in a survey-based research approach.

Nodes representing personal norms formed a strong cluster with habit in both networks. Personal norms have indeed been shown to exert a strong influence on habits, driven by an individual's value system (Klöckner & Blöbaum, 2010). The link between habit and items framed around a sense of self-ethical obligation (personal responsibility and morality, #20 & 21) in N1, while in N2 self-perception nodes (sustainability as integral part of self, #18 & 19) mediated the link between habit and the personal norms cluster. While these singularities are minor compared to the overall network structure, they may suggest that P1's habitual sorting disposal practices tend to be driven by moral integrity, compared to personal integrity for P2. The strong sustainability ethos of UCD as an institution may partially mediate this variation.

No statistical difference was found when conducting Pearson's χ^2 test for FW recycling habit between populations ($p > 0.05$ based on a single-node comparison), refuting hypothesis 2 that P2 would display higher FW recycling habit. However, statistical analysis was undermined by a low *n* for one factor of the scale, resulting in low power and thus limiting any reliable position on hypothesis 2. Nevertheless, habit was linked to uniformity of waste management infrastructure (#38) in N2, suggesting that while hypothesis 2 could not be tested reliably, an association between recycling habit and a stable environment through the access to consistent waste collection schemes may indeed exist in the system. In line with literature on habitual behaviour (Gkargkavouzi, Halkos & Matsiori, 2019; Klöckner & Blöbaum, 2010), the single, consistent waste management infrastructure across UCD and the city of Davis may have enabled individuals to adopt repetitive behaviours and integrate them into their daily routines. In contrast, the lack of uniformity both within the ICL campus and across London boroughs may represent a barrier to the formation of EoL sorting habits for BBP waste. Indeed, one focus group participant (**Chapter 6**) stated that they would not dispose BBPs with FW on-the-go (even when such an option were to be provided), but then reverted to such recycling habit when visiting their parental home, where FW sorting and separate collection are the norm.

The correlation between willingness-to-pay (WTP, #30) and morality (#21) and personal responsibility (#20) for N1 and N2, respectively, resonates with the findings from Zwicker et al. (2020), who found guilt was most strongly connected to WTP. WTP was also positively correlated with cost consideration (#34), which was negatively correlated with the effort cluster (simplicity, convenience and time-efficiency, #6, 7 & 8) in N1. This suggests that the more price-conscious an individual is, the more likely they are to seek easy waste sorting behaviour, which could lead to inappropriate or sub-optimal disposal of BBPs. The call for simplification (and uniformisation) of household and household-like waste streams could therefore also contribute towards lifting additional barriers and achieving higher perceived behavioural control (Burgess et al., 2021). The introduction of BBPs on the already crowded plastic packaging market may – and to some extent already did (Dilkes-Hoffman et al., 2019a; Taufik et al., 2020) – exacerbate the challenge of plastic waste recycling, but in the long run could prove beneficial. For example, by providing a joint FW and packaging disposal stream for highly contaminated plastics that require consumers to clean them prior to sorting them could help overcome the 'yuck' factor (Kakadellis, Woods & Harris, 2021).

The link between marketing (#23 & 24) and the value of BBPs within the broader value chain (#35) might imply that brand communications, especially mission-oriented marketing (corresponding to the node #23), could strengthen consumers' understanding and approval of the role BBPs play in the wider plastic sustainability arena. Given that the network is undirected, this relationship could also mean that consumers who are relatively aware of the role of BBPs in the broader value chain expect brands to state their contribution towards plastic sustainability. This is an important point for building trust and legitimacy, given that the transition to sustainable packaging in the food and beverage industry has been shown to be slow and inconsistent, with a tendency for corporations to overlook systems-level solutions (Phelan et al., 2022). The negative correlation displayed in N2 between #35 and #11 & 12 further supports the link between trust and legitimacy, whereby the more sceptical/ambiguous an individual feels towards BBPs, the less likely they are to see the benefits of BBPs in the wider sustainability sphere.

Constructs of labelling, signposting and education formed a cluster (partially disjointed in N2, though still spatially close), but were not linked to other nodes. The lack of interaction between labelling and the circular behavioural node of interest (#39) may be linked to the lack of familiarity with compostability symbols, as opposed to recycling symbols (Boesen, Bey & Niero, 2019). Increasing consumer familiarity with such symbols and ensuring logos guide consumers in their disposal decision-making process could help enhance appropriate recycling rates (Taufik et al., 2020).

The combination of familiarity, trust and habit can have far-reaching consequences on appropriate BBP waste disposal and undermine behaviour change strategies. In a study based on the Behaviour Change Wheel framework, which provided a step-by-step methodology for the development of intervention strategies to improve BBP disposal, Allison et al. (2022a) identified the creation of a disposal instruction label as the optimal intervention. However, in a field experiment conducted in the Netherlands investigating consumer recycling behaviour for bio-based, compostable and conventional plastics, Ansink, Wijk & Zuidmeer (2022) found that participants were not responsive to the presence of logos nor to the information displayed on them, as suggested by the networks in **Figure 41**. This observation resonated with a point raised by focus group participants, who admitted they did not pay attention to bioplastic-related logos and even expressed scepticism towards them due to concerns over 'greenwashing'. Nevertheless, they showed support for a universal label they could immediately recognise and trust, stating the Fairtrade logo as an example. This suggests that parallel policies, such as stricter regulation on bioplastic terminology and packaging labelling, would be required first to build trust among consumers. Governments and manufacturers must pay more attention to consumer behaviour and attitudes towards bioplastics in the design of logos and disposal instruction labels for them to prove effective (Ansink, Wijk & Zuidmeer, 2022).

In N1, sensorial and technical features (#1-5) were linked to fear (#13) and worry (#14), with the olfactory node (#3) acting as bridge. Here again, given that edges are undirected, it is difficult to determine a causal relationship between, however one could argue that the more apprehensive consumers are of their (potentially misaligned) disposal behaviour and its unintended consequences, the closer attention they will pay to material and product properties. This link was not observed in N2 and remains to be determined, though a link between material properties and attitudes has been reported (Karana, 2012). It is perhaps not too surprising that attitudes are not strongly related to appropriate disposal, as previous research has already shown that a positive perception of BBPs does not match with consumer disposal behaviour (Ansink, Wijk & Zuidmeer, 2022; Dilkes-Hoffman et al., 2019a; Taufik et al., 2020). Furthermore, Barr's model of recycling behaviour suggests that convenient access to a recycling bin plays a key role in predicting sorting behaviour (Barr, 2006) and Stern's attitude-behaviour-context model shows that behavioural outcomes are less impacted by attitudes once the environmental factors are optimal for recycling (Stern, 2000).

The most meaningful implications for this study lie in the cluster formed around the behavioural intentions, infrastructure and knowledge nodes across both networks. Knowledge of differentiation (#9) and of disposal (#10) was positively correlated with higher access (#37) and uniformity (#38) of waste collection infrastructure. It is likely that consumers' increased familiarity with BBPs and understanding of disposal routes enhances their awareness of the available infrastructure, with positive knock-on effects on the likelihood of performing circular disposal behaviour (#39). Taufik et al. (2020) indeed found that increased familiarity of bio-based concepts was linked to appropriate disposal. The possibility that access to a uniform and accessible waste collection scheme could lead to better informed citizens – that is, the opposite directionality – should not be dismissed. However, no difference was found between populations for knowledge-related constructs ($p > 0.05$ for both single-node comparisons), refuting hypothesis 3. This suggests that while an understanding of the relevant concepts and disposal routes is important for circular disposal intentions, providing consumers with the relevant infrastructure – the physical opportunity, as described in the COM-B model (Michie, van Stralen & West, 2011) – remains essential in enabling the translation of such knowledge into action, emphasising the role of the contextual setting.

Disposing BBPs with FW (#39) was negatively correlated with mixed dry recycling (#41). This implies that the more engaged an individual is in circular disposal behaviour, the more they understand BBPs are different from conventional plastics and thus should not recycle them in the traditional recycling stream. FW disposal behaviour was also linked directly to access to infrastructure in N1 and mediated by the knowledge nodes in N2. Research has shown that access to waste collection services plays a large role in participation in recycling behaviours (Thomas & Sharp, 2013). FW disposal was also correlated with home composting in N1, but not in N2. At first glance, this may be surprising, given that P2 was more likely to choose home composting as EoL for BBP disposal than P1 (**Figure 40**). This correlation is likely due to the significantly higher proportion of P2 engaging in FW disposal (**Figure 38**), while composting rates were consistently low across both populations.

Opting for general/residual waste for BBP disposal (#40) was not linked to any other behavioural node but was negatively correlated with personal responsibility (#20) and habit (#22) in N2. Once again, this confirms the common idea in behavioural models that the more habitual a given behaviour is, the more likely it will be completed – and, consequently, the less likely competing behaviours are to take place (Tonglet, Phillips & Read, 2004). This link between pro-environmental intentions on the one hand and habit and personal factors on the other, notably self-identify, has been shown to increase the predictive ability of established behavioural models in both private and public spheres (Gkargkavouzi, Halkos & Matsiori, 2019; Klöckner & Blöbaum, 2010). However, situational variables can override personal norms, leading to what is commonly referred to as the value-action gap, whereby an individual's values (and behavioural intentions) do not correlate with their actions (Barr, 2006). Both logistics and perceived behaviour control over a given action can undermine the success of turning behavioural intention into action (Barr, 2006). This may explain why none of the personal norms nodes were linked to circular disposal intentions, as was previously observed in field experiments on European consumers (Herbes, Beuthner & Ramme, 2018; Taufik et al., 2020). Here again, the mandate to separately collect FW from UK households and businesses by 2023 and the call for a uniform national waste recycling scheme (Burgess et al., 2021) should reduce the contextual barriers to FW recycling, and in the long-term could contribute towards developing habitual behaviours centred around FW and BBP recycling.

5.6. Centrality measures

While community analysis can be used to investigate the global structure of a network, centrality measures can provide further insights into the structural importance of individual nodes on network cohesiveness and their ability to act as 'bridges' (Dalege et al., 2017). Here, three centrality measures were computed in `qgraph`, and visualised using the CentralityPlot functions (**Figures 42 & 43**). First, the strength measure of centrality was calculated to investigate the direct influence of each node on each network (Dalege et al., 2017). Both networks exhibited similar strength profiles overall, with personal responsibility and self-perception scoring highest across N1 and N2, alongside convenience, the broader value chain and awareness campaigns, as well as morality [N1] and visual content [N2]. This pattern was driven predominantly by the substantial edge weights within the personal norms and effort clusters, as well as the high number of links to other nodes for personal responsibility.

Next, betweenness and closeness were computed. Betweenness can be interpreted as the ability to disrupt information flow within the network, while nodes that can diffuse information rapidly are characterised by a high closeness score (Dalege et al., 2017). The highest betweenness-scoring nodes were the broader value chain and personal responsibility/morality in both networks, as well as convenience, simplicity and knowledge of disposal in N1, and habit, self-perception and awareness campaigns in N2. Closeness scores exhibited less variability than for the other two centrality measures in N1 (apart from the social norms cluster), although relatively higher scores were observed for the nodes related to personal norms, habit and the broader value chain, similarly to N2, as well as WTP [N1 only].

Centrality analysis suggests that nodes representing the broader value chain and personal norms, in particular personal morality [N1] and responsibility [N2] were the most central clusters overall, closely followed by convenience [N1] and habit [N2]. These nodes are well connected and/or characterised by thick edge weights within their respective clusters and act as information bridges, meaning that tapping into these nodes can have far-reaching changes on nodes across the network, especially those in the same cluster (Dalege et al., 2017). Intriguingly, none of these nodes were connected to the behavioural node of interest (FW recycling, #39), although both habit and personal responsibility, which were negatively correlated to general/residual waste in N2, also performed well in centrality measures.

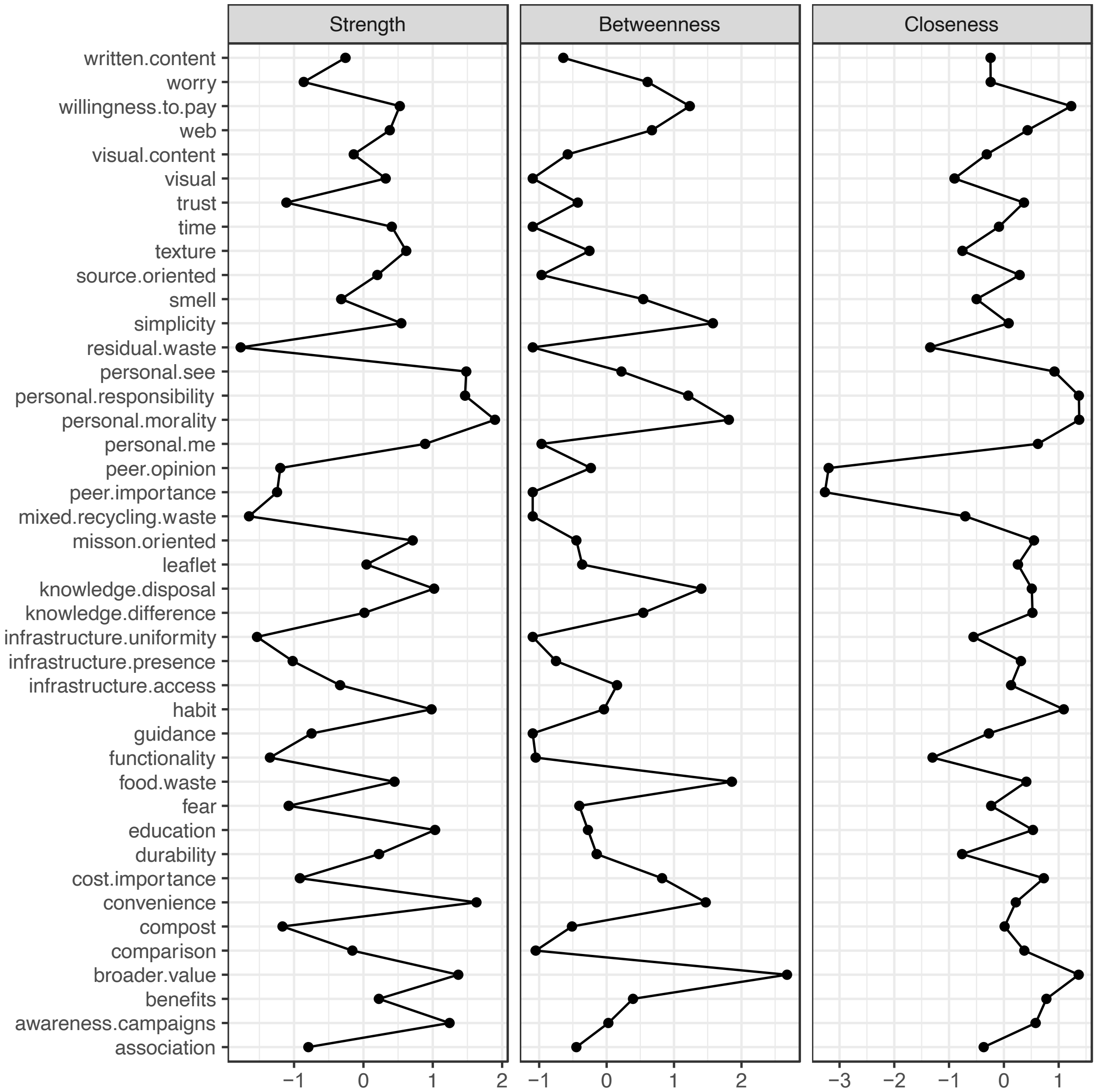


Figure 42 | Centrality plot of the ICL network. The left, centre and right panels show the strength, betweenness and closeness estimates for each node of the network. Measures were computed in R using the package `qgraph`. Reproduced from Kakadellis et al. (in review).

Knowledge of disposal, whilst not as central, represents the only moderately central node in N1 also connected to the circular behaviour node (through the intermediary of access to infrastructure). Thus, knowledge of disposal might be a promising potential target to influence circular behaviour, though only to a certain extent, as discussed in **Section 5.5**.

Social norms were major outliers in centrality analysis and their isolation from the network was such as to prevent the computation of a closeness score in N2, as hinted by their position in the network (**Figure 41**). Their exclusion of inter-cluster dynamics was partly unexpected, although the role of social norms on behaviour was shown to be underrated (Nolan et al., 2008). Other socially oriented factors, such as a concern for the local community, may play a more important role (Tonglet, Phillips & Read, 2004), but were not investigated in this study.

The stability of the estimated network and centrality measures was tested using the R package `bootnet` (Epskamp, Borsboom & Fried, 2018), the results of which can be found in the **Appendix Section 4.** Consistency in edge-weight order contributed towards validating the accuracy of the network structure (**Figure 48**), while stability of centrality estimates showed a reliable strength index (**Figure 49**), while closeness and betweenness should be interpreted with care (**Figure 49**).

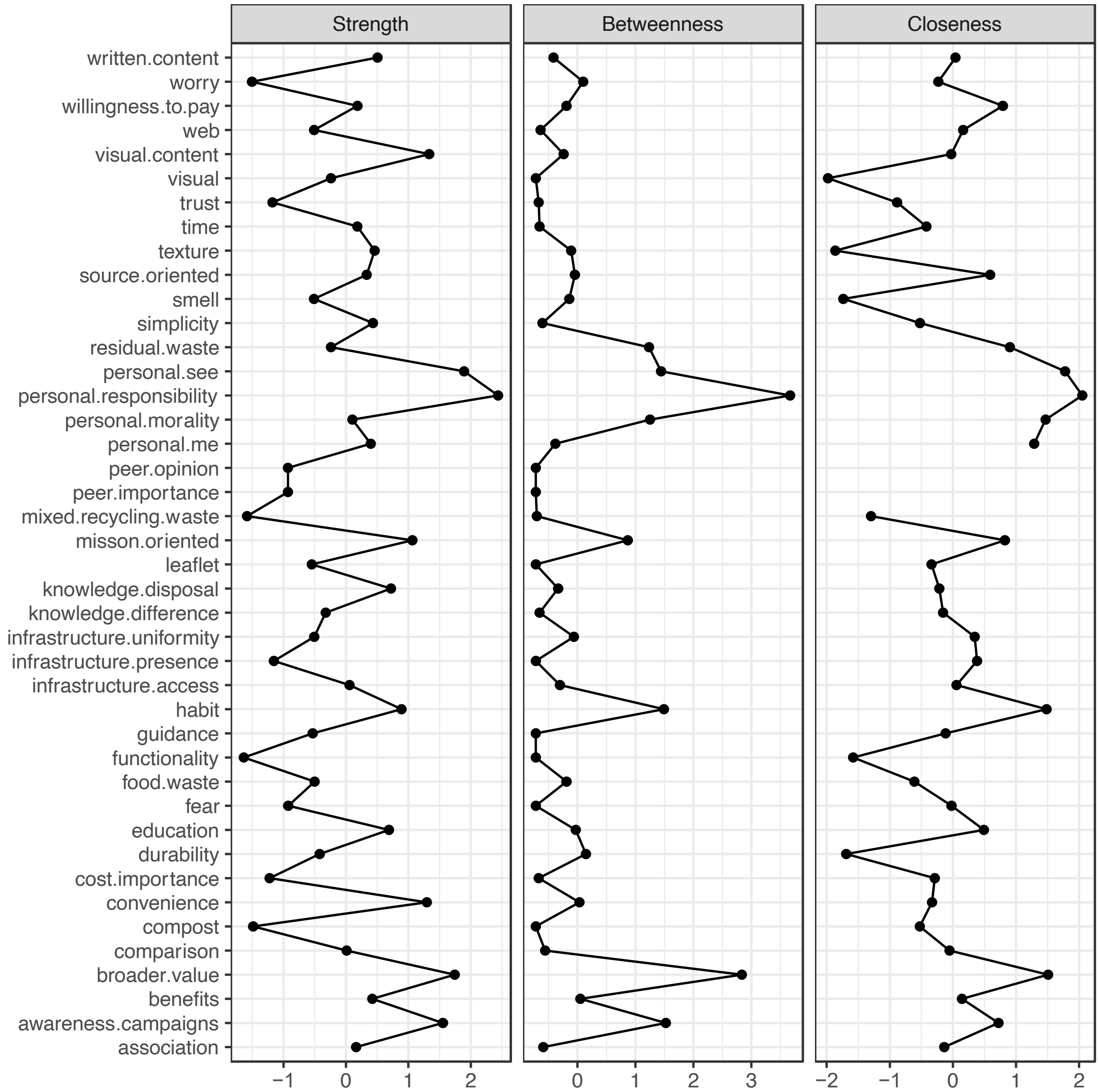


Figure 43 | Centrality plot of the UCD network. The left, centre and right panels show the strength, betweenness and closeness estimates for each node. Measures were computed in the R package `qgraph`. Missing values for social norms constructs (peer.opinion and peer.importance, #16 & 17 in the network) are linked to their unconnectedness in the network, whereby the shortest path length can be infinite, leading to 0 closeness for all items. The CentralityPlot function shows the closeness in the giant component, hence the missing values. Reproduced from Kakadellis et al. (in review).

6. Conclusions & Future work

This chapter covered a comparative case study analysis undertaken in the context of circular behaviour and the disposal of BBP food packaging. The unique social, cultural and policy landscape of the two case study sites, ICL in the UK and UCD in the US were reviewed in order to gain an understanding of the contextual setting ahead of the survey dissemination and analysis. The survey, which was conducted on 457 and 284 participants at ICL and UCD, respectively, was subsequently analysed through a network approach. UCD participants were significantly more likely to have access to a separate FW collection scheme, as well as to dispose of BBP waste in FW recycling, confirming hypothesis 1.

Network analysis revealed that both the presence and consistency of the relevant infrastructure and knowledge-related constructs play a key role in enhancing circular behaviour intention (and, hypothetically, circular behaviour). While no statistical difference was found in FW recycling habit between populations, thereby refuting hypothesis 2, the analysis was undermined by a low statistical power and would require a bigger sample size for a reliable position on hypothesis 2. Hypothesis 3, which tested whether UCD's population exhibited a deeper knowledge of BBP terminology and disposal routes, was rejected, emphasising the importance of contextual setting on disposal intentions. It suggests that while a higher likelihood of appropriate disposal of BBP was associated with a higher level of familiarity with BBPs, providing consumers with the relevant infrastructure enables the translation of such knowledge into action.

This comparative approach provided insights into which system elements from the framework developed in **Chapter 6** are most likely to facilitate appropriate disposal and highlighted the need for the relevant organics infrastructure, as well as fostering consumer understanding of bioplastic-related concepts and the range of waste disposal streams. Designing policy interventions framed around these elements would contribute towards building habitual recycling behaviour as society transitions to more circular consumption and waste management practices.

While some of the findings (e.g. those related to the infrastructure-knowledge-behavioural intentions cluster) are not in themselves novel, the research strategy adopted in this study provides a multimethod and novel network approach in the field of circular system design and environmental psychology. This approach reaches similar conclusions to previous literature on recycling behaviour and BBP waste disposal based on a network approach that is not limited to a single behavioural or design theory (Bhushan et al., 2019), which enabled the exploration of new relationships and the identification of antagonistic effects.

It is important to acknowledge that the survey captured behavioural intentions rather than actual behaviour. While behavioural intentions, measured by the likelihood of opting for a given waste stream, constitute a valuable predictor of actual disposal behaviour, in practice behavioural actions may be undermined by situational variables, leading to the value-action gap (Barr, 2006). Capturing disposal behaviour more truthfully would have required a behavioural experiment (e.g. Taufik et al., 2020) or a waste audit, as conducted at the Harvard Medical School (Meier, 2017). A post-survey interventional study, following the research design of Zwicker et al. (2020), would complement and help validate the findings of the network analysis. Furthermore, a longitudinal study would provide valuable insights into the impact of policy on circular disposal behaviour. Given that California's mandate for a separate FW collection from residents and businesses took effect in January 2022 (with fines for noncompliance starting in 2024), with a similar scheme yet to be fully implemented in London (from January 2023, though the scheme has been postponed until 2024 and possibly 2025), changes waste management practices among the public are likely to change over the next few years. Studying these changes over time would enable an exploration of the link between consumer behaviour and policy implementation and investigate which policy interventions are most effective in achieving the desired behavioural outcomes in a given context.

Chapter 8 – Moving upstream: shelf-life study of fresh leafy greens under different packaging films

"*Every opportunity has a shelf-life*." – Margaret Atwood

The experimental study presented in this chapter has been conducted at the University of California, Davis as part of the OIV under the supervision of Professor Gail Taylor and Professor Elizabeth Mitcham, with the support of Aiwei Zhu for produce collection (**Section 3.1**), Veronique Nzigire Bikoba and Nico Lingga for packaging preparation (**Section 3.2**) and Dr. Nicholas Reitz for water exchange measurement (**Section 3.4.2**). The experimental work, data collection, analysis and interpretation were undertaken by the PhD candidate solely.

1. Introduction

So far, **Chapters 4-7** have addressed the contribution of BBPs towards a joint waste stream for the treatment of OFMSW including commercial, institutional and household FW. However, given that food production accounts for 30% of global GHG emissions and that an estimated 8-10% of all human induced GHG emissions could be avoided by tackling FW (UNEP, 2021), FW prevention remains an essential climate mitigation strategy in the design of sustainable food systems.

While efforts to increase the efficiency of food supply chains have traditionally focused on minimising food losses during food cultivation and harvesting, preserving and extending the quality of food produce post-farmgate plays an equally vital role in ensuring food sustainability (Briassoulis et al., 2013). This is particularly the case for higher-income countries, where most of FW originates at post-consumer stages, with as little as 3% lost at early stages of the food supply chain (Schweitzer et al., 2018). At the same time, demand for fresh-cut, ready-to-eat leafy greens, which include lettuce, cabbage, kale, chard, endive, rocket, endive and spinach, is on the rise (Bergquist, Gertsson & Olsson, 2006). While such produce enhance convenience by minimising preparation steps for consumers, in return their shelf-life will be reduced, thereby directly impacting FW generation.

Recent advances in material science and plant physiology have led to the development of effective packaging designs, which not only protect fresh produce across stages of transportation and distribution – the primary function of packaging – (Silvenius et al., 2014), but also maximise their shelf-life (Verghese et al., 2013). Among these designs, active packaging encompasses a range of 'smart' packaging materials and chemicals aimed at modifying the environmental conditions surrounding packaged fresh produce, including fruits and vegetables, to improve their quality and/or extend their shelf-life (Salgado et al., 2021).

Active packaging expands the means of protection and preservation of traditional food packaging (Fuertes et al., 2016) and includes antibacterial agents, antioxidants, moisture and smell absorbers, O_2 and ethylene (C_2H_4) scavengers, CO_2 generators and modified atmosphere packaging (MAP).

By sealing the packaged food from its environment and through the selection of polymer films with distinct permeability properties, MAP allows for the creation of optimal, produce-specific atmospheric conditions surrounding fresh food aimed at slowing down plant metabolism. These typically correspond to lower O_2 and higher CO_2 concentrations relative to ambient air (Qu et al., 2022). In-package relative humidity is also an important consideration, given the high water content of fresh produce (Briassoulis et al., 2013). Fruits and vegetables continue to lose water post-harvest through the process of transpiration and, to a lower extent, respiration, with minimally processed food produce (i.e. fresh-cut produce) exhibiting higher water loss (Ayala-Zavala et al., 2008). On the other hand, excessive accumulation of water vapour inside the packaging may favour microbial growth, thus accelerating the deterioration of fresh food (Qu et al., 2022).

The optimal atmospheric and humidity conditions in MAP depend on the physiological characteristics of individual produce types, their respiration rates and gas tolerance levels (Owoyemi, Rodov & Porat, 2021). Anaerobic respiration uses carbohydrates (a source of energy for plant embolism) much faster than aerobic respiration and therefore contributes to premature spoilage of produce. Keeping an O_2 concentration at the lowest physiologically tolerable level for plant aerobic respiration while avoiding a switch to anaerobic metabolism contributes significantly towards maximising shelf-life (Ellis, Knowles & Knowles, 2019). However, as O_2 and CO_2 diffusion across the polymer is limited, their concentration will change over time as respiration takes place, eventually depleting in-package O_2 (Qu et al., 2022).

The introduction of micropores (diameter < 200 μm) into polymer design has enabled the development of passive, or equilibrium MAP (EMAP) by drastically improving the permeability of packaging films (Gonzalez et al., 2008). Steady-state conditions are obtained when the exchange rate of gases through the pores is in equilibrium with the production of CO_2 or consumption of O_2 under given respiration and transpiration rates (Mistriotis et al., 2016). The permeability of a given micro-perforated film can be adjusted by altering the number of pores, their diameter, length, area and distribution (Qu et al., 2022). Which polymer is used as packaging material will also influence the shelf-life of the food produce it contains; while the respiration rate can be regulated by perforation, water vapour exchange takes place mainly through the polymer itself (Mistriotis et al., 2011).

The more FW is generated, the more food, packaging and transportation are required to ensure a certain amount of produce is delivered to customers (Lorite et al., 2017). For bioplastic food packaging to deliver on their sustainability promise, they must fulfil their primary function (Dilkes-Hoffman et al., 2018; Kakadellis & Harris, 2020). This is especially relevant for foods exhibiting high environmental production cost per unit (e.g. animal products) or per total waste (e.g. fruits and vegetables, which account for the largest losses by weight at the consumer level, 19% and 22% of total FW, respectively) (Verghese et al., 2014).

The inherent hydrophilicity of BBPs provides both challenges and opportunities for horticultural packaging applications. The water vapour transmission rate (WVTR) of BBP films at 25°C is 20–300 g/m^2/day depending on polymer type, much higher than the 1 g/m^2/day exhibited by PE (Briassoulis et al., 2013). While this could potentially prove beneficial to prevent decay in leafy greens, whose high water content makes them prone to damage and microbial spoilage (Fang & Wakisaka, 2021), such permeability to water vapour may increase water loss.

2. Study aims

In this study, the effectiveness of BBPs at extending the shelf-life of fresh leafy greens is investigated, based on an assessment of visual quality and physical properties of produce and packaging. Given that leafy greens are among the most widely consumed and wasted vegetable class (FDA, 2022) and that California, where this study was undertaken, produces an estimated 74% of the total production of fresh market spinach in the US (California Leafy Greens Research Board, 2022), baby spinach (*Spinacia oleracea L.*) was used as case study.

Informed by previous research on the impact of BBP packaging on the shelf-life of bell peppers (Owoyemi, Rodov & Porat, 2021) and cucumbers (Owoyemi, Porat & Rodov, 2021), it was hypothesised that spinach packaged in macro-perforated BBP films have the potential to match the shelf-life of macro-perforated (but not micro-perforated) conventional plastic bags.

3. Materials & Methods

3.1. Material cultivation, sourcing, transportation and storage

Fresh-cut, washed and ready-to-eat baby spinach was supplied by Braga Fresh Family Farms located in Salinas, California, US. The produce was collected in bulk in macro-perforated plastic bags (1 lb, or 0.45 kg, per bag) on May 30th, 2022 (trial 1) and September 8th, 2022 (trial 2) and transported in coolers to the UCD Postharvest Laboratory and stored at 5°C overnight before the start of the experiment on the following day.

The macro- and micro-perforated conventional polymers films were sourced from Braga Fresh Family Farms and a polymer manufacturer based in Israel, respectively. Three types of EN 13432 certified compostable plastics were provided by a polymer manufacturer specialising in biodegradable food packaging based in Atlanta, Georgia, US. The polymers films to be used as packaging in the experimental trials were provided as ready-to-use bags, A4 sheets or rolls. Individual material types and film properties are summarised in **Table 20.**

Table 20 | Material properties for the (bio)plastic films used in the study. OTR: O_2 transmission rate; WVTR: Water vapour transmission rate. N/A: information not provided by the supplier. *OTR of biodegradable films show values for unperforated films as supplied by the manufacturer and do not reflect actual OTRs of the perforated films used in the experimental design.

Treatment	Material	Thickness (mm)	Application	OTR (cc/m^2/day)	WVTR (g/m^2/day)
Micro-perforated conventional plastic (**miP**)	Polypropylene (PP)	35	Parsley	13,000	5
Macro-perforated conventional plastic (**MP**)	PP	N/A	Tender leaves	310,000	10
Biodegradable bioplastic (**NE**)	Cellulose-based	30	Smoked dairy and meat	5*	5
Biodegradable bioplastic (**NK**)	Cellulose-based	30	Dairy	5*	15
Biodegradable bioplastic (**NVS**)	Cellulose-based	30	Fresh fruit and vegetables	5*	200

3.2. Experimental design and preparation

Polymer sheets and rolls were cut to create sheets of identical dimensions to the ready-to-use bags, using a manual sheet cutter. The sheets were then heat-sealed on three sides using a portable direct heat sealer (KF-200CS, Sealer Sales Inc., US) at an intensity level of 2 for 5 seconds. The dimensions of the experimental bags were: 175 x 210 mm^2 (outer dimensions) and 160 x 190 mm^2 (inner dimensions), with an internal volume of 880 ml. A preliminary test for leakiness was conducted on a separate set of tester bags by adding 100 ml of water into the bags and gently shaking them.

Since it was not possible to source micro-perforated BBP films and given their near total impermeability to O_2 (**Table 20**), BBP bags were perforated with a 25 gauge needle (0.5 mm outer diameter) on both sides, with 5 rows of 4 holes each in the first experimental trial (e.g. Owoyemi, Rodov & Porat, 2021). In the second experimental trial, this design was repeated and complemented with two additional perforation patterns (for NE and NVS polymers only), consisting of 5 rows of 8 holes each and a combination of the two patterns (i.e. 5 rows of 4 holes and 5 rows of 8 holes), yielding 20, 40 and 60 holes per side.

In total, spinach was assigned to five treatments in the first experimental trial (µP, MP, NE with 20 perforations (NE_20, NK_20 & NVS_20), with 3 technical replicates for each sampling day (60 in total). Informed by results from the first trial, NE and NVS were selected for additional perforation rates in the second trial and eight individual treatments were evaluated (µP, MP, NE_20, NE_40, NE_60, NVS_20, NVS_40 & NVS_60), again in triplicates (96 in total).

Each bag was weighed and tared before 50 g of spinach was added into each experimental bag, after which the open side was heat-sealed. Bags were then transferred to a cold room at 5°C for a maximal duration of 15 days, with sampling occurring on days 0 (once bags were sealed and let in the cold room for one hour), 5, 10 and 15. The ideal storage conditions for spinach and other fresh produce are close to 0°C (Batziakas et al., 2020), however the actual storage temperature over the shelf-life of the packaged produce is likely to be higher (Owoyemi, Rodov & Porat, 2021), especially once the produce has been purchased by the consumer. Thus, adopting a slightly higher temperature in the study is more reflective of fluctuations across the supply chain. In addition, given that baby spinach has a high respiration rate that is about five times higher at 10°C than at 0°C (Allende et al., 2004), adopting a higher-than-optimal temperature shortened the shelf-life of baby spinach, which supported the measurement of visual quality and physical variables within the relatively short experimental timeframe.

3.3. Visual quality scoring

A range of visual characteristics, including overall visual quality, discolouration, decay (i.e. spoilage) and wilting were scored using qualitative scales. Overall visual quality was scored on a 9 to 1 scale, where 9 = excellent, 7 = good (some leaves displaying slight discolouration, decay or wilting), 5 = fair (limit of marketability), 3 = poor (most leaves displaying discolouration, decay or wilting) and 1 = extremely poor (unfit for human consumption) (Bergquist, Gertsson & Olsson, 2006). Discolouration, decay and wilting were scored on a 1 to 5 scale where 1 = none, 2 = slight (1-5% of leaves affected, corresponding to one or two leaves), 3 = moderate (6-10%, around five leaves), 4 = considerable (11-30%, 10-12 leaves) and 5 = severe (> 30%).

3.4. Physical variables measurement

3.4.1. <u>Weight loss</u>

The weights of the packaging and of the produce were recorded prior to the experimental start and at their designated sampling date. Weight loss was measured using the following formulas:

$$\text{Weight }(\mathrm{S})_{\mathrm{t}} = \text{Weight}(\mathrm{S}+\mathrm{P})_{\mathrm{t}} - \text{Weight}(\mathrm{P})_{\mathrm{i}}$$

$$\text{Weight loss }(\%) = \frac{\text{Weight}(\mathrm{S})_{\mathrm{t}} - \text{Weight}(\mathrm{S})_{\mathrm{i}}}{\text{Weight}(\mathrm{S})_{\mathrm{i}}} \times 100,$$

where $\text{Weight}(\mathrm{S})_{\mathrm{t}}$ corresponds to the weight of spinach at time *t*, $\text{Weight}(\mathrm{S}+\mathrm{P})_{\mathrm{t}}$ the weight of spinach and packaging at time *t*, $\text{Weight}(\mathrm{P})_{\mathrm{i}}$ the weight of the packaging at the start of the experiment (*i* for initial) and $\text{Weight}(\mathrm{S})_{\mathrm{i}}$ the weight of spinach at the start of the experiment.

3.4.2. Water exchange

The water exchange rate across packaging films was determined using a moisture chamber developed by Reitz & Mitcham (2022) and consisting of a sealed PET box connected to a sensor (BME280, Bosch Sensortech, Germany) and an Arduino Uno R3 microcontroller board (www.arduino.cc). The packaged produce was placed inside the sealed chamber; the sensor then monitored the temperature, relative humidity and pressure over 30 seconds outside the packaging but inside the sealed chamber. A fan inside the container ensured proper air mixing. The water exchange rate was calculated based on the readings of all three variables.

3.4.3. Temperature and relative humidity

HOBO data loggers were added to each sample destined to be measured on day 15 to measure both temperature and relative humidity throughout the experimental timeframe. Measurements were recorded in one-minute increments. Two additional loggers were also used to monitor the room environment. Measurements were visualised and prepared for data analysis through the HOBOware software (Onset, US).

3.4.4. In-package relative O_2 and CO_2 compositions

In-package relative O_2 and CO_2 concentrations were measured with a MAP gas analyser (model 900141, Bridge Analyzers Inc., US). Measurements were made after moisture exchange and weight measurements to avoid interfering with these values. The device was first reset and measurements were taken once ambient CO_2 and O_2 concentrations were reached (0.03% and 20.81%, respectively) by poking the sample needle through the packaging and waiting a few seconds for the values to stabilise.

3.5. Data analysis

The data were analysed through two-way ANOVA and Tukey's Honestly Significant Difference (HSD) test at $\alpha < 0.05$ in R through RStudio (https://www.rstudio.com/). Beside basic R functions (R Core Team, 2021), the following packages were also used: `agricolae` (de Mendiburu, 2021) and `car` (Fox, Weisberg & Price, 2019) for further statistical tools specific to agricultural research and regression analysis, and `dplyr` (Wickham et al., 2022), `ggplot2` (Wickham, 2016) and `RColorBrewer` (Neuwirth, 2022) for data visualisation. Means from triplicates for each sampling date and standard errors (SE) were plotted for data visualisation.

4. Results & Discussion

4.1. Packaging shelf-life performance

Visual quality is an important factor for the marketability of fresh-cut, ready-to-eat produce such as baby spinach leaves (Bergquist, Gertsson & Olsson, 2006). Visual quality and physical properties of spinach and packaging for both June and September trials were evaluated (**Figures 44, 45 & 46**), with mean effects summarised in **Tables 21-24**. Both trials aligned on the ranking of the packaging films in terms of their overall shelf-life performance, leading to the following order: (1) micro-perforated conventional plastic (miP), (2) BBP film with the lowest WVTR (NE), (3) BBP with intermediate WVTR (NK, June trial only), (4) macro-perforated conventional plastic (MP) and (5) BBP with enhanced WVTR (NVS).

Spinach from the miP treatment was the only one to still meet marketability criteria after 15 days across both trials (**Figures 45a & 46a**) and spinach leaves remained hydrated and green (**Figures 45c-d & 46c-d**). While miP pouches showed slight condensation (**Figure 44**) on the internal surface of the film, they remained firm and impermeable. In contrast to findings by Owoyemi, Porat & Rodov (2021), NE and NK films exhibited acute condensation (**Figure 44**) and were thinner, more elastic and starting to stick to the produce after 10 days of storage, suggesting a change in chemical properties, as would be expected from inherently hydrophilic BBP materials susceptible to hydrolysis (Di Bartolo, Infurna & Dintcheva, 2021). This made it relatively difficult to empty the bag, with spinach leaves sticking to the film. Similar condensation issues were also reported in PLA trays (Botondi et al., 2015), thereby affecting the marketability of the packaged produce, despite acceptable visual quality. Neither MP nor NVS films displayed any visible condensation but resulted in severely wilted leaves (**Figures 45d & 46d).** However, they presented no difficulty in emptying the content of the pouches.

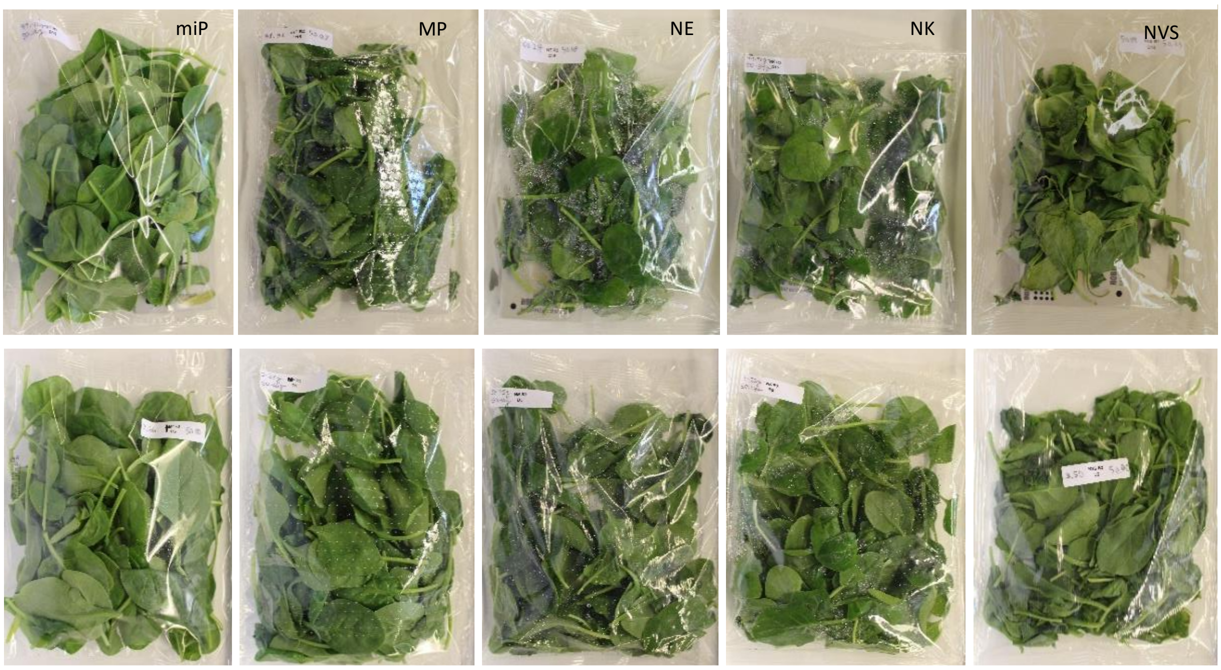


Figure 44 | Representative photographs of packaging treatments at day 5 (bottom) and day 15 (top). Bags were emptied before proceeding to visual quality scoring.

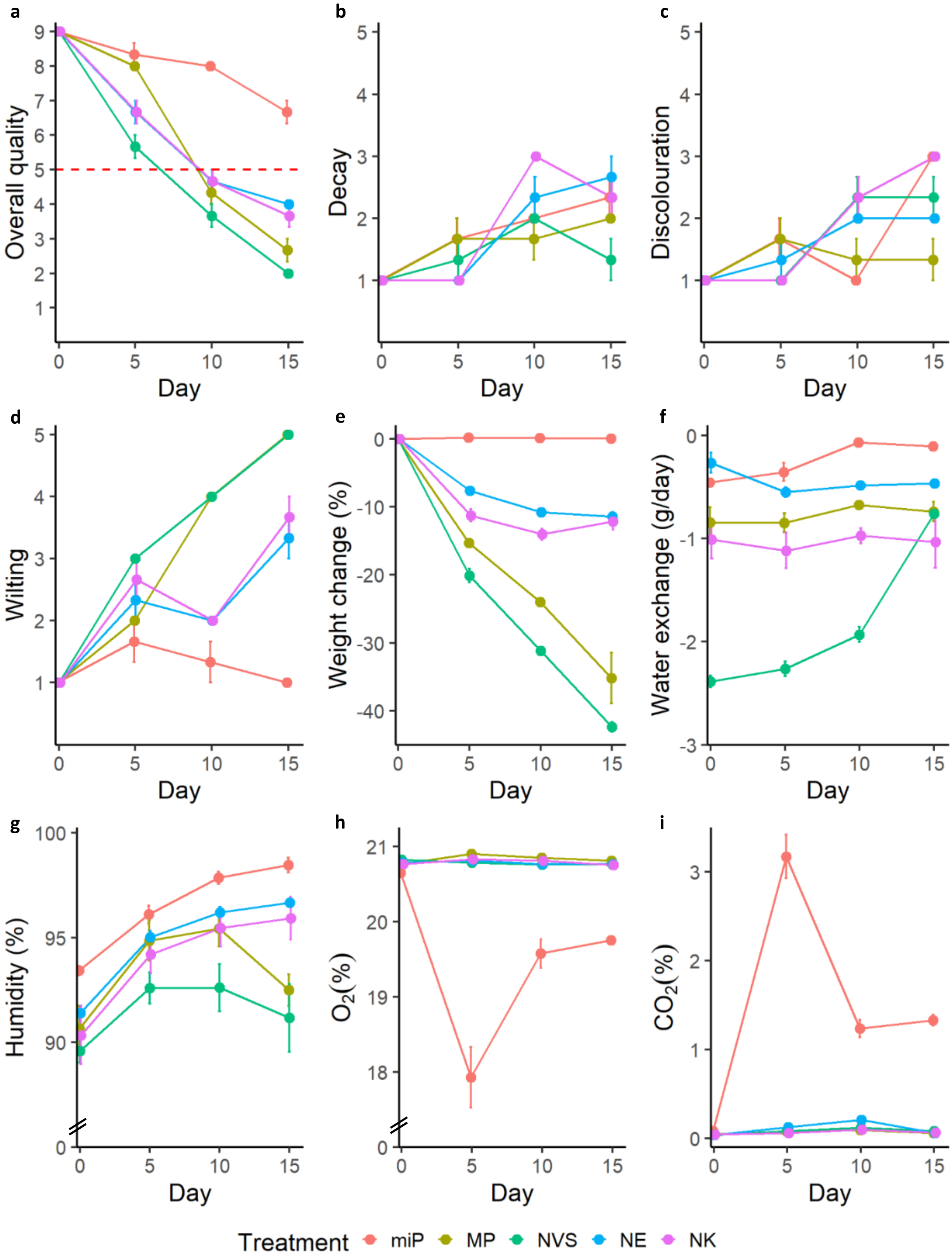


Figure 45 | Graphs of individual variables assessed over experimental timeframe for the June trial. a: overall visual quality; b: decay; c: discolouration; d: wilting; e: weight change; f: water exchange; g: relative humidity; h: relative O_2 concentration; i: relative CO_2 concentration. Each point represents the mean from triplicates, bars represent standard errors. The red dashed line in (a) represents the limit of marketability.

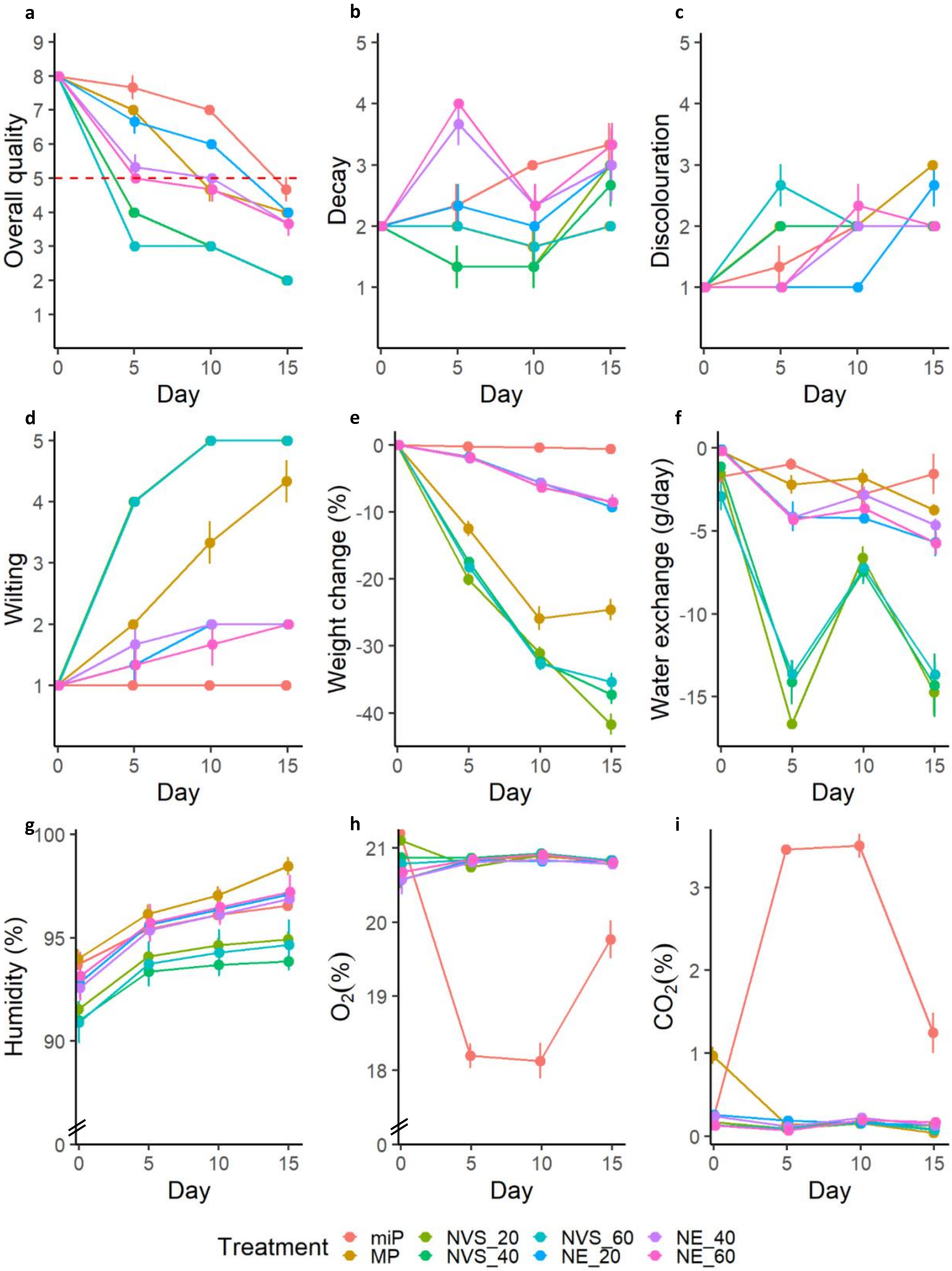


Figure 46 | Graphs of individual variables assessed over experimental timeframe for the September trial. a: overall visual quality; b: decay; c: discolouration; d: wilting; e: weight change; f: water exchange; g: relative humidity; h: relative O_2 concentration; i: relative CO_2 concentration. Each point represents the mean from triplicates, bars represent standard errors**.** The red dashed line in (a) represents the limit of marketability.

The clear outperformance of the miP film was likely the result of the creation of a distinct in-package atmosphere compared to ambient air and the other treatments (**Figures 45h-i & 46h-i**). This corresponded to significantly lower O_2 and higher CO_2 concentrations (**Tables 22 & 24**), which spiked to 17.93% and 18.20% O_2 and 3.17% and 3.46% CO_2 after 5 days in June and September, respectively, compared to an average 20.83% and 19.83% O_2 and 0.09% and 0.49% CO_2 across all other treatments at the same time point. Low to moderate O_2 concentrations (0.8-10%) have been associated with lower plant respiration (Tudela et al., 2013) and slower consumption of carbohydrate resources (Mudau et al., 2018), resulting in reduced weight and antioxidant loss (Batziakas et al., 2020; Gil, Ferrers & Tomás-Barberán, 1999; Ko et al., 1996; Mudau et al., 2018; Tudela et al., 2013).

However, while significant, the effect size of the reduction in O_2 levels observed is unlikely to have affected plant respiration alone; the elevated CO_2 atmosphere may have been a contributing factor (Allende et al., 2004). Spinach stored in a 5% O_2 and 15% CO_2 atmosphere at 4°C were the only ones to meet commercial acceptability after 12 days of storage, compared to those stored under ambient air conditions (Mudau et al., 2018). MAP was also effective at higher temperatures, where samples stored in micro-perforated MAP reached 4-6% O_2 and 8-11% CO_2 at 13-21°C and extended shelf-life by up to 2 days compared to those stored in conventional, non-perforated plastic bags (Batziakas et al., 2020). Nevertheless, Tudela et al. (2013) highlighted some trade-offs; increasing O_2 concentration from 1 to 10% limited the development of off-odours, a key factor for consumers and thus for marketability of spinach, but reduced shelf-life by increasing the respiration rate.

Relative humidity remained consistently above 90% across all treatments and throughout the experimental timeline (**Figures 45g & 46g**). In the June trial, significantly higher values were observed among the best performing treatments (miP & NE, close to 95-96.5%), followed by NK & MP (in the 93-94% range) and NVS (91.5%) (**Table 22**). This categorisation was not visible in the September trial, although NVS treatments displayed statistically lower values than all other treatments (**Table 24**) and exhibited the highest level of wilting (**Table 23**). The produce conditions at the start of the experimental trial and ambient relative humidity may have contributed toward these discrepancies (**Section 4.3**). The results suggest that a 95% (and above) relative humidity is preferable for preserving visual quality and extending the resulting shelf-life of fresh-cut spinach, similarly to other leafy greens, such as lettuce (Agüero et al., 2011). Nonetheless, Medina et al. (2012) have noted the benefits of exposing baby spinach to short-term (36 hours) low relative humidity (72%) at postharvest followed by rehydration (97%), reporting highest leaf stiffness and lower mechanical damage due to lower initial water content.

The miP film treatment also resulted in minimal weight loss (**Figures 45e & 46e**) and differed significantly from all other treatments, with less than 1% weight loss after 15 days. In contrast, both MP and NVS treatments exhibited major weight loss with the NVS treatment resulting in a 42.37% average cumulative weight loss, significantly more than the second-to-last MP treatment (35.21%) in the June trial. NE and NK treatments displayed similar performances, with 11.41% and 12.15% weight loss, respectively, in the same trial.

Similar trends in weight loss have been reflected elsewhere. In a shelf-life study on lettuce, an eight-fold increase in weight loss was observed between a starch-based polymer film and a conventional PVC film after eight days at 6°C (Brandelero, Brandelero & de Almeida, 2016). Red bell peppers and cucumbers wrapped in PP films yielded 2.6% and 5% weight loss after up to 4 and 2 weeks of simulated supply chain and home storage conditions, respectively, compared to 8-10% for all other BBP treatments (Owoyemi, Rodov & Porat, 2021; Owoyemi, Porat & Rodov, 2021). However, the PP films used in these experiments were macro-perforated and present a contrasting performance to the macro-perforated PP films (MP treatment) used in this study. The significant weight loss presented by the MP treatment is somewhat unexpected, given that the WVTR of the polymer was 10 g/m^2/day and that the average water exchange rate was -0.78 g/day in the June trial, significantly less than for the NK treatment (-1.03 g/day) (**Table 22**). It is possible that the WVTR value provided by the supplier referred to unperforated film and although water vapour exchange takes place mainly through the polymer itself (Mistriotis et al., 2011), this may only refer to micro-perforations and may not apply to films with a large number of macro-perforations (> 1500 per MP film pouch).

The only noticeable difference between NE and NK treatments was found in moisture loss (**Figure 45f**), with the NE film exhibiting statistically lower water exchange rate (-0.44 g/day) compared to its NK counterpart (-1.03 g/day) (**Table 22**), reflecting the minor differences in their WVTRs (5 and 15 g/m^2/day, respectively). The moderate water exchange rate of NE and NK films compared to that of the NVS treatment might explain the discrepancies in both weight loss and condensation level. While a higher permeability to water vapour prevented acute weight loss, unlike the miP treatment where a modified atmosphere slowed plant metabolism, ambient respiration rates still took place under NE and NK treatments, leading to the accumulation of water inside the packaging. Minimising condensation is important because such accumulation of free water may represent a food safety concern and enhance microbial spoilage (Owoyemi, Porat & Rodov, 2021). Given that spinach quality is particularly sensitive to water loss (Batziakas et al., 2020) and is determined by a fresh appearance and crisp texture (Cantwell & Kasmire, 2002), modulating that WVTR to reduce the condensation rate without compromising on weight loss thus represents a priority for BBPs (Owoyemi, Porat & Rodov, 2021).

Table 21 | Mean effect size of treatment on visual quality (June). Significance levels were obtained by conducting Tukey's test; means with different letters are statistically different from each other.

Treatment	Overall Visual Quality	Decay	Discolouration	Wilting
miP	8.00^{a}	1.75^{a}	1.67^{a}	1.25^{c}
MP	6.00^{b}	1.58^{a}	1.33^{a}	3.00^{a}
NE_20	6.08^{b}	1.75^{a}	1.58^{a}	2.17^{b}
NK_20	6.00^{b}	1.83^{a}	1.83^{a}	2.33^{b}
NVS_20	5.03^{c}	1.42^{a}	1.67^{a}	3.25^{a}

Table 22 | Mean effect size of treatment on weight loss, water exchange, relative humidity and O_2 and CO_2 concentrations (June). Significance levels were obtained by conducting Tukey's test; means with different letters are statistically different from each other. Weight loss: day 15 values only.

Treatment	Weight Loss (%)	Water Exchange (g/day)	Humidity (%)	$[O_2]$ (%)	$[CO_2]$ (%)
miP	0.05^{a}	-0.25^{a}	96.46^{a}	19.48^{b}	1.46^{b}
MP	-35.21^{c}	-0.78^{b}	93.36^{bc}	20.83^{a}	0.07^{a}
NE_20	-11.41^{b}	-0.44^{a}	94.82^{ab}	20.78^{a}	0.11^{a}
NK_20	-12.15^{b}	-1.03^{c}	93.98^{b}	20.79^{a}	0.07^{a}
NVS_20	-42.37^{d}	-1.83^{d}	91.49^{c}	20.78^{a}	0.08^{a}

Table 23 | Mean effect size of treatment on visual quality (September). Significance levels were obtained by conducting Tukey's test; means with different letters are statistically different from each other.

Treatment	Overall Visual Quality	Decay	Discolouration	Wilting
miP	6.83^{a}	2.67^{ab}	2.00^{a}	1.00^{c}
MP	5.92^{ab}	1.92^{bc}	1.58^{ab}	2.67^{b}
NE_20	6.17^{ab}	2.33^{abc}	1.42^{b}	1.58^{c}
NE_40	5.50^{b}	2.75^{a}	1.50^{ab}	1.67^{c}
NE_60	5.33^{b}	2.92^{a}	1.58^{ab}	1.50^{c}
NVS_20	4.25^{c}	1.92^{bc}	1.75^{ab}	3.75^{a}
NVS_40	4.25^{c}	1.83^{c}	1.75^{ab}	3.75^{a}
NVS_60	4.00^{c}	1.92^{bc}	1.92^{ab}	3.75^{a}

Table 24 | Mean effect size of treatment on weight loss, water exchange, relative humidity and O_2 and CO_2 concentrations (September). Significance levels were obtained by conducting Tukey's test; means with different letters are statistically different from each other. Weight loss: day 15 values only.

Treatment	Weight Loss (%)	Water Exchange (g/day)	Humidity (%)	$[O_2]$ (%)	$[CO_2]$ (%)
miP	-0.59^{a}	-1.76^{a}	95.44^{a}	19.32^{b}	2.10^{a}
MP	-24.59^{c}	-1.97^{a}	96.41^{a}	20.78^{a}	0.33^{b}
NE_20	-9.31^{b}	-3.54^{a}	95.48^{a}	20.76^{a}	0.18^{b}
NE_40	-8.55^{b}	-2.96^{a}	95.24^{a}	20.75^{a}	0.18^{b}
NE_60	-8.58^{b}	-3.48^{a}	95.64^{a}	20.81^{a}	0.14^{b}
NVS_20	-41.69^{d}	-9.88^{b}	93.81^{b}	20.89^{a}	0.14^{b}
NVS_40	-37.27^{d}	-9.25^{b}	92.98^{b}	20.87^{a}	0.12^{b}
NVS_60	-35.36^{d}	-9.37^{b}	93.40^{b}	20.84^{a}	0.12^{b}

4.2. Effect of added perforation on BBP shelf-life performance

The June trial showed that only the miP film succeeded in creating a modified atmosphere inside the packaging. While perforating BBP films with macro-perforations did not affect O_2 and CO_2 concentrations relative to ambient air, the impact of increased perforation levels on BBP shelf-life performance was investigated in the second trial (conducted in September) by doubling or tripling the number of perforations per pouch, focusing on NE and NVS treatments.

Increasing the perforation rate had no significant impact on any of the variables investigated on either BBP film type (**Figure 46, Tables 23 & 24**). This suggests that for variables directly influenced by the nature and level of perforations, such as O_2 and CO_2 concentrations, the lowest perforation rate was already above the range at which it may have regulated the flow of gas across the packaging film, which would have required more precise, fine-tuned micro-perforations (Mistriotis et al., 2016; Qu et al., 2022). On the other hand, for variables more influenced by the type of polymer, such as those related to moisture exchange (Mistriotis et al., 2016), changes in macro-perforations were not major enough to interfere with diffusion across the polymer surface.

4.3. Effect spinach growing season on packaging shelf-life performance

Since the trials were conducted three months apart, with spinach leaves from the second trial harvested and transported to the research facility during a major heatwave (> 40°C over 5 consecutive days) in September 2022, the impact of the spinach growing and harvest season on shelf-life performance of each treatment was considered.

A two-way ANOVA revealed that there was a statistically significant interaction between trial and treatment for all variables except for O_2 and CO_2 concentrations (**Figure 47 & Appendix Table 49**). This difference can be attributed to elevated temperatures during growth, harvest and transportation for the September trial, resulting in sub-optimal visual quality of produce at the start of the experiment, with some decay already present on day 0 **(Figure 47b**). This was consequently reflected in the quality of the spinach at the end of the experiment (**Figure 47a-c**). Conducting Tukey's multiple comparison tests (**Tables 25 & 26**) revealed that water exchange rates were generally higher in September for all treatments, although only significant for BBP films (NE & NVS) (**Figure 47**). The fact that spinach from the September trial displayed wetter leaves upon collection, likely driven by enhanced respiration rates due to elevated temperatures (Batziakas et al., 2020) exhibited during transportation of the spinach produce from the harvesting site to the laboratory, may explain the increased water movement across the film.

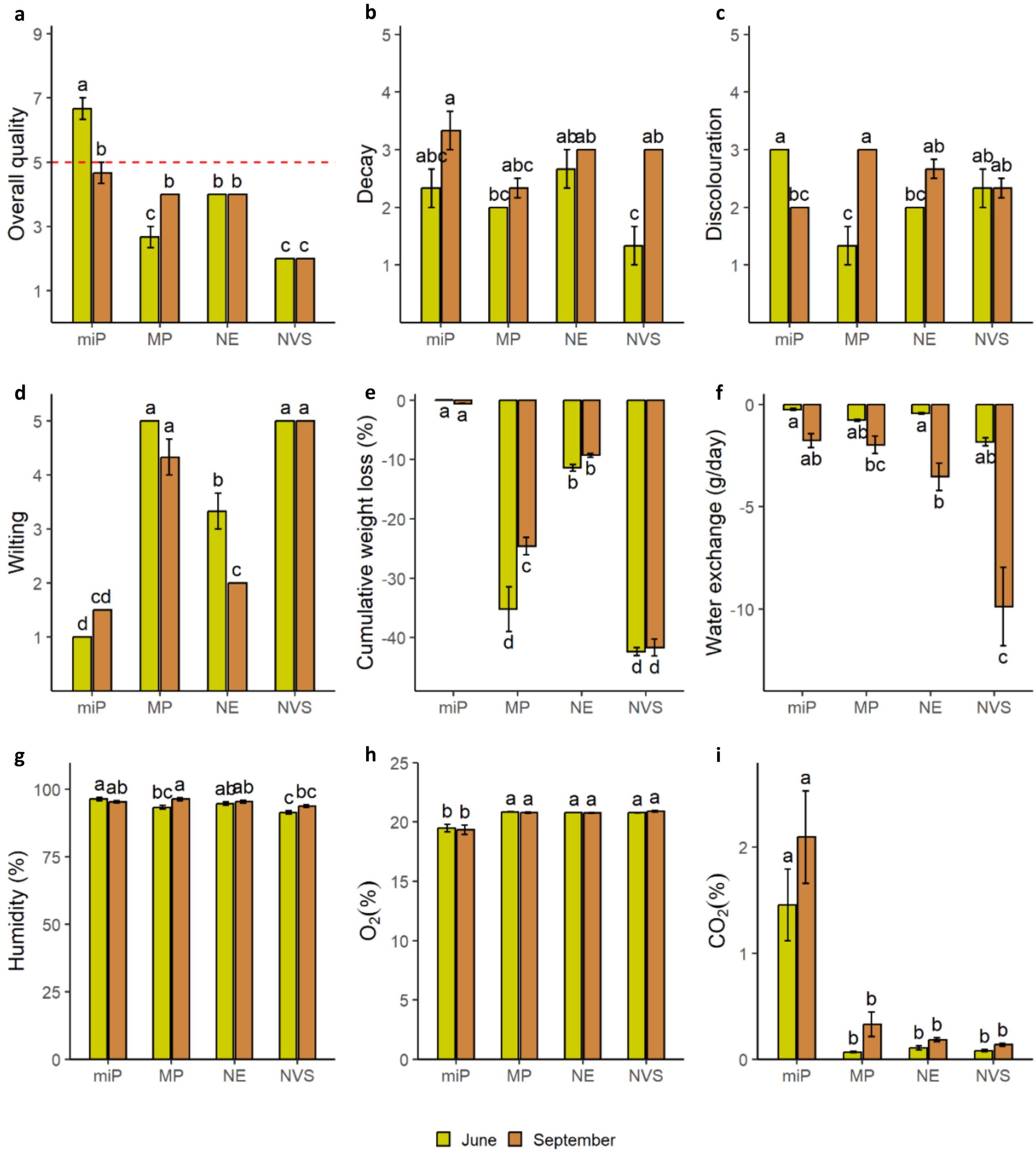


Figure 47 | Mean effect size of individual study variables across trials. a: overall visual quality; b: decay; c: discolouration; d: wilting; e: weight change; f: water exchange; g: relative humidity; h: relative O_2 concentration; i: relative CO_2 concentration. Each point represents the mean from triplicates, bars represent standard errors. Values for overall visual quality, decay, discolouration, wilting and weight loss were taken from day 15 samples only; for the remaining variables, all time points were used. The red dashed line in (a) represents the limit of marketability. Significance levels were obtained by conducting Tukey's test on all possible pairs and are represented through compact visual display where means with different letters are statistically different from each other.

The wetter conditions of the produce upon collection may also explain the difference in relative humidity (**Figure 47g**), which was significantly higher in September than in June for the MP treatment (**Table 26**), reaching levels of the best performing treatments (> 95%). Furthermore, it is likely that the warmer conditions at the start of the September trial resulted in warmer spinach leaves upon sealing the pouches, trapping additional moisture inside the packaging and thus influencing the relative humidity. Samples from the MP treatment also displayed significantly lower weight loss in September compared to June, (35.21% in June vs 24.59% in September, **Figure 47e**). The higher relative humidity for the MP treatment in September may have mediated the difference in weight loss (Gil & Garrido, 2020), given that spinach leaves exposed to lower relative humidity conditions lose water content more easily compared to those stored under high humidity (Medina et al., 2012).

Table 25 | Mean effect size of treatment on overall visual quality, discolouration, decay and wilting for treatments conducted in both trials. Values and compact letter display for overall visual quality, decay, discolouration, wilting and weight loss are from day 15 samples only; for the remaining variables, all time points were used. Sept: September.

Treatment	Overall Visual Quality	Decay	Discolouration	Wilting
miP: June	6.67^{a}	2.33^{abc}	3.00^{a}	1.00^{d}
miP: Sept	4.67^{b}	3.33^{a}	2.00^{bc}	1.50^{cd}
MP: June	2.67^{c}	2.00^{bc}	1.33^{c}	5.00^{a}
MP: Sept	4.00^{b}	2.33^{abc}	3.00^{a}	4.33^{a}
NE_20: June	4.00^{b}	2.67^{ab}	2.00^{bc}	3.33^{b}
NE_20: Sept	4.00^{b}	3.00^{ab}	2.67^{ab}	2.00^{c}
NVS_20: June	2.00^{c}	1.33^{c}	2.33^{ab}	5.00^{a}
NVS_20: Sept	2.00^{c}	3.00^{ab}	2.33^{ab}	5.00^{a}

Table 26 | Mean effect size of treatment on weight loss, water exchange, relative humidity and O_2 and CO_2 concentrations for treatments conducted in both trials. Values and compact letter display for overall visual quality, decay, discolouration, wilting and weight loss are from day 15 samples only; for the remaining variables, all time points were used. Sept: September.

Treatment	Weight Loss (%)	Water Exchange (g/day)	Humidity (%)	[O_2] (%)	[CO_2] (%)
miP: June	-0.05^{a}	-0.25^{a}	96.46^{a}	19.48^{b}	1.46^{a}
miP: Sept	-0.59^{a}	-1.76^{ab}	95.44^{ab}	19.32^{b}	2.10^{a}
MP: June	-35.21^{d}	-0.78^{ab}	93.36^{bc}	20.83^{a}	0.07^{b}
MP: Sept	-24.59^{c}	-1.97^{bc}	96.41^{a}	20.78^{a}	0.33^{b}
NE_20: June	-11.41^{b}	-0.44^{a}	94.82^{ab}	20.78^{a}	0.11^{b}
NE_20: Sept	-9.31^{b}	-3.54^{b}	95.48^{ab}	20.76^{a}	0.18^{b}
NVS_20: June	-42.37^{d}	-1.83^{ab}	91.49^{c}	20.76^{a}	0.08^{b}
NVS_20: Sept	-41.69^{d}	-9.88^{c}	93.81^{bc}	20.89^{a}	0.14^{b}

4.4. Shelf-life considerations beyond modified atmosphere packaging

The previous sections have highlighted the contribution of MAP towards shelf-life extension by slowing the rate of plant respiration and exposed trade-offs that may emerge, such as between weight loss minimisation and condensation prevention. Efficient packaging should enable the formation of optimal storage conditions by lower O_2 and higher CO_2 concentrations relative to ambient air, while preventing reaching extreme levels (< 0.4-8% O_2 and > 12-15% CO_2 in spinach), which induce hypoxic fermentation and CO_2-mediated NH_3 accumulation (Cantwell, Hong & Nie, 2010), encouraging the growth of pathogens and causing off-flavours and tissue damage (Allende et al., 2004; Owoyemi, Porat & Rodov, 2021).

Further considerations beyond packaging attributes alone may contribute significantly towards shelf-life extension. Temperature management remains an effective way – if not the most – to delay produce deterioration and preserve quality of fresh-cut fruit and vegetables (Kou et al., 2014). Mineral and flavonoid content as well as antioxidant activity were found highest in spinach samples stored at low temperatures (0-4°C), while weight loss was minimised with decreasing storage temperature (Kou et al., 2014; Mudau et al., 2018). Since colour represents a key factor in quality assessment and purchase decision for consumers (Rizzo & Muratore, 2009), and given that 'freshness' is predominantly driven by the rate of water loss and chlorophyll (an antioxidant) breakdown, ensuring storage of fresh produce at low temperatures directly translates into its commercial value (Batziakas et al., 2020).

However, optimal storage conditions are not always feasible; the use of MAP at non-optimal postharvest temperatures has been shown to limit weight loss, off-odours and quality deterioration (Batziakas et al., 2020; Mudau et al., 2018). Thus, while a combination of MAP and low storage temperatures would maximise shelf-life (Mudau et al., 2018), MAP represents a valuable and cost-effective strategy for extending shelf-life in circumstances where access to refrigerated storage is limited (Batziakas et al., 2020).

Yet, it is important to acknowledge that the impacts of plastic packaging on shelf-life extension and therefore on FW are complex and, in some cases, may even contribute to rather than minimise FW (Schweitzer et al., 2018). For example, while NE and NK treatments resulted in lower weight loss than MP and NVS treatments, pronounced condensation inside the packaging caused challenges in emptying the bags, which may result in higher rates of FW generation at the consumer level (Williams et al., 2020). In addition, the size of packaging might play a significant role in FW generation; displaying information about optimal product safety and storage may provide higher environmental and economic benefits than those associated with the use of MAP (Williams et al., 2020). Research on the causes and mechanisms of household FW generation is limited and efforts should be dedicated to uncovering the relationships between emerging packaging technology and consumer attitudes and food saving practices (Brennan et al., 2021).

In addition, a non-trivial amount of FW generated both at harvest and at the consumer level is set by aesthetic standards prior to packaging (e.g. discarding 'wonky' fruit and vegetables) (Schweitzer et al., 2018). Here again, the contribution of novel packaging design, including MAP and BBPs, may not be as relevant to the food sustainability debate and may limit the implementation of more systemic changes across the food supply chain (Schweitzer et al., 2018). Thus, the advantages offered by any packaging technology should not override the priority of eliminating unnecessary (bio)plastic packaging and minimising FW generation.

5. Conclusions & Future work

This chapter aimed to frame the question of the biocircularity of BBP food packaging within the wider food supply chain these novel materials are ultimately a part of. It investigated the shelf-life performance of both conventional and BBP film packaging for fresh-cut leafy green vegetables, based on spinach as model produce, as well as the role of micro-perforations. Shelf-life performance was assessed on visual quality (overall visual quality, decay, discolouration and wilting), weight loss, water exchange, relative humidity and relative O_2 and CO_2 concentrations, resulting in the following ranking: (1) micro-perforated conventional film, macro-perforated BBP films with low (2) and moderate (3) WVTRs, (4) macro-perforated conventional film and (5) macro-perforated BBP film.

Study findings showed that film micro-perforations enabled the creation of modified atmosphere, which is essential for reducing ambient plant respiration rate and resulting quality deterioration, in turn extending the shelf-life of the packaged spinach produce. Due to supply constraints, this study did not include micro-perforated BBP films. The findings would benefit from additional trials with micro-perforated BBP films to enable a fair side-by-side comparison with micro-perforated conventional films. Since NE and NK treatments displayed moderate shelf-life performance and surpassed that of the MP treatment in a number of variables assessed, micro-perforating BBP films may enable the formation of a modified atmosphere and elevate their performance to that of the miP treatment displayed in this study.

The shelf-life performance of BBP films was most likely dictated by their WVTR, which affected water exchange rate across the film and associated weight loss and visual quality deterioration. The elevated weight loss and severe wilting made NVS unsuitable as packaging option for fresh-cut spinach, while NE and NK displayed moderate weight loss but were undermined by high levels of condensation on the inside surface of the film. Further research should be dedicated to optimising the WVTR of BBP films to retain moisture while avoiding condensation, as well as their impact on consumer perception and waste generation practices.

At the same time, in line with the overarching theme of this research, these enhanced BBP films will also need to be designed to degrade according to industrial AD practices. This represents a challenge from a material science perspective, given the WVTR of BBPs is intrinsically linked to their hydrophilicity, which in turn affects their biodegradability. The direct opposition between shelf-life extension (functionality design) and biodegradability (EoL design) properties of BBP food packaging will require a careful and interdisciplinary consideration to avoid burden-shifting and/or unintended consequences across the food supply chain.

The lack of microbial characterisation[1] and its implications for food safety and nutritional profile represents a limitation of this study. The formation of a distinct atmosphere in the miP treatment and the resulting changes in respiration and other metabolic rates are likely to have led to a unique microbial and nutritional profiles. Nutritionally relevant elements (e.g. calcium, magnesium, zinc, potassium and iron) as well as vitamins (beta-carotene (pre-vitamin A), folic acid (vitamin B9), ascorbic acid (vitamin C), phylloquinone (vitamin K1)) and their antioxidant activity could be further investigated through e.g. inductively coupled plasma mass spectroscopy (ICP-MS) and fluorescence recovery after photobleaching (FRAP) in order to provide further insights into the impact of BBP films on food quality when compared to conventional packaging.

[1] Microbial DNA extraction from leaves for day 10 and day 15 samples across both trials was performed. Due to budgetary and timing constraints, microbial sequencing was not conducted within the timeframe of the PhD but is likely to be conducted in the post-doctoral stage to provide further insights in the context of this study.

Chapter 9 – Conclusions & Outlook

1. Thesis summary

This thesis has addressed the suitability of BBPs in the current organic waste management infrastructure through co-AD with municipal FW. The primary aim of this research was to further the understanding of the compatibility of BBPs with the proposition of a circular bioeconomy framework and identify the challenges and opportunities arising from it.

In **Chapter 4**, co-AD batch trials of both conventional and certified compostable plastic fragments with SFW were conducted to investigate the impacts of introducing BBPs into FW AD on biogas and CH_4 yields, as well as on microbial communities. Results showed that at the relatively low concentrations used (0.2-5% w/w), neither conventional plastics nor BBPs affected the performance of AD, with the biggest changes in microbial community structure likely due to the introduction of easily biodegradable SFW, rather than plastic fragments. Nonetheless, given that microscopic characterisation of plastic fragments and digestate quality assessment were not undertaken, caution must be taken when extrapolating these results and applying them to full-scale, commercial practices.

This word of caution is indeed reflected by the outcomes of the stakeholder study conducted as part of **Chapter 5**. Content analysis of semi-structured interviews with stakeholders related to BBP waste management in AD in the UK revealed that concerns over the ultimate biodegradability of BBPs and its impact on digestate quality (and hence market viability) remain at the centre of the debate. Unless a suitable standard for anaerobic biodegradability and complementary regulations are introduced, application of digestate onto agricultural land could lead to the introduction of partially degraded (but not fully assimilated) BBP fragments and, eventually, microplastics. This would exacerbate the already poorly reversible plastic pollution (MacLeod et al., 2021). The scepticism shown by some stakeholders regarding academic research in this field reinforces the need for increased knowledge exchange between industry, academia and legislation, as well as the development of realistic and consistent experimental designs reflective of commercial practices discussed in **Chapter 4**.

Interview content analysis also highlighted the role of consumers in ensuring a clean feedstock stream for FW AD. The difficulty to distinguish between biodegradable and non-biodegradable plastics, as well as a lack of harmonised FW collections across the UK were reported as major barriers for enabling circularity in the organic waste management sector. Building on these findings, a novel systems framework was developed and presented in **Chapter 6** in order to identify and structure systemic factors (system elements) that influence how consumers interact with BBP packaging, ultimately enabling or hindering the flow of BBPs across stages of acquisition, consumption and disposal.

The framework developed in **Chapter 6** led the identification and characterisation of 18 system elements (further split into 35 system sub-elements) across 6 broad categories. Following from the previous chapter, in **Chapter 7** a survey was designed to gain insights into how these system elements interact and influence consumer behaviour, focusing on circular disposal behavioural intention (which, in the context of this thesis, was defined as the intention to dispose of BBP food packaging waste alongside FW/organic waste bins) and disposal behaviour more broadly.

The survey was conducted at two academic institutions, ICL in the UK and UCD in California (US) and formed the basis of a comparative case study and network analysis, in which the contextual setting of each case study site was also investigated. Both state and institutional policies on FW management contributed towards a more developed and consistent organic waste management infrastructure (including for food and food packaging waste) at UCD. This correlated with a higher proportion of the surveyed participants having access to a separate FW collection scheme and adopting circular disposal behavioural intentions, when compared to ICL's population.

The network analysis was also used to investigate which system elements were most likely to influence circular disposal behavioural intentions and how these system elements related to each other in the network. Results suggested that the presence and consistency of the relevant waste management infrastructure and knowledge of BBP terminology and of disposal routes played a key role in enhancing circular disposal intentions (and, hypothetically, circular disposal) among the surveyed populations. Clarifying the role BBP play in enhancing sustainability of the broader plastics value chain was identified as a potential strategy to shape the entire network, due to the highly connected nature of its corresponding node.

Finally, **Chapter 8** extended the debate on the compatibility of BBPs with circular bioeconomy frameworks to the wider food supply chain and food system that BBP food packaging are ultimately embedded within, focusing on BBP shelf-life extension performance. Informed by earlier research conducted in the first part of this 1 + 3 PhD studentship (MSc + PhD), which highlighted the significant GHG contributions of FW relative to those of food packaging regardless of packaging type, a shelf-life study on fresh baby spinach stored under different plastic packaging types was conducted. The results showed that BBP film packaging with reduced permeability to water vapour offers the potential to compete with conventional plastic packaging, although its intrinsic hydrophilicity remains an issue and its applicability to fresh produce packaging is currently limited. These findings acted as a reminder of the dependencies within the food-energy-waste nexus and the importance of adopting a systems view when addressing complex issues in the wider plastic sustainability arena.

Altogether, this thesis showed that BBP food packaging face a number of design limitations, both in terms of EoL (their ability to biodegrade in the current AD waste management infrastructure) and functionality (their ability to maximise shelf-life, thereby minimising FW). Most stakeholders in the waste management and agricultural sectors (including practitioners, legislators and regulators) remain sceptical towards BBPs, despite – or because of – the recognition of the urgency in addressing plastic and microplastic pollution in agricultural soils. The promotion of BBPs continues to be centred around material substitution, with little consideration for the role of consumers as key players in ensuring circularity. BBPs cannot effectively address plastic pollution without a systemic redesign of consumption systems and a comprehensive mapping of the key actors in the system and, crucially, of their behaviours.

2. Policy implications

The proposition of a circular bioeconomy framework as a means of moving from a fossil-based to bio-based economy, with an emphasis on natural capital and waste valorisation and circularisation, has benefited the BBP manufacturing industry, particularly in food packaging applications. Conceptually, BBPs contribute towards the 'circularisation' of the plastics value chain and anchor it within the food-energy-waste nexus. The commercialisation of BBPs has tended to focus on research and development of alternatives to conventional plastics with novel polymer properties, as well as a shift away from fossil-based materials, with a growing interest in waste-derived feedstocks, in line with a circular bioeconomy framework.

As exposed in **Chapter 5**, in the UK, BBPs were initially promoted by BEIS under the Bioeconomy Strategy, now withdrawn and supplanted by the Innovation Strategy, published in 2021 (HM Government, 2021). Whilst the former put emphasis on tackling microplastic pollution through the design of alternatives to conventional plastics, the Innovation Strategy does not mention the word 'microplastic(s)' once. The term 'plastic-free' packaging is used once, linking it to innovations in synthetic biology and bioengineering. On the other hand, little to no attention is given to BBP EoL, and the single mention of the possible need for and implications of a standard for bio-based and biodegradable plastics in the Bioeconomy Strategy no longer appears in its updated document.

The fact that these policy documents fail to consider the wider context in which BBPs exist may not be surprising, and could to some extent be considered as acceptable, given that waste management, including for plastic packaging, is outside BEIS' policy scope and is instead overseen by DEFRA and its devolved administrations (the Scottish Executive Environment and Rural Affairs Department (SEERAD) in Scotland and the Department of Agriculture, Environment and Rural Affairs (DAERA) in Northern Ireland). Nevertheless, there is a clear need for crosstalk between governmental departments to avoid unintended consequences of material substitution, as highlighted by the findings in **Chapters 4 & 8**.

As suggested by semi-structured interviews in **Chapter 5** and the network analysis conducted in **Chapter 7**, both access to and uniformity of designated waste collections seem pivotal in enabling circular disposal behaviour to take place. However, in many countries, separate organic waste collections from households and public spaces remain less common than residual waste or recycling bins (Taufik et al., 2020). As highlighted in **Chapter 3** (**Section 3.1**), only about a quarter of all organic waste generated in the EU is currently collected separately (EEA, 2020). The remaining 75% of this waste finds its way into the general waste stream and is either landfilled or incinerated, severely undermining the circularity of the food (and associated FW) supply chain and hindering the recovery of valuable nutrients. In addition, what can and cannot be put into separately collected bins (e.g. food scraps, food-contaminated cardboard/plastic packaging) may vary considerably from one municipality to another. In the UK, 39 different plastic collection schemes were identified among 391 local authorities, leading to significant confusion amongst consumers (Burgess et al., 2021).

The lack of uniformity across collection schemes can have detrimental consequences on convenience of disposal. The mandate for separately collected FW from households and businesses from 2023 across the EU will contribute towards a more accessible, and hopefully more uniform waste collection framework. The fact that the UK delayed the implementation of its scheme to 2025 is alarming, given the growth of the BBP market in food packaging, horticultural and agricultural applications (European Bioplastics, 2021), and, perhaps more importantly, the increased recognition of FW as an important source of global GHG emissions (Crippa et al., 2021).

While expanding the organic waste management infrastructure will play a key role towards the realisation of a circular bioeconomy, it is important to remember that concerns of BBP biodegradability will remain and need to be addressed for BBPs to deliver on their theoretical benefits. Taking inspiration from the composting sector, Vegware, a compostable food packaging manufacturer, launched a partnership with Paper Round, a waste management company, to ensure their packaging is captured and effectively treated in the relevant composting infrastructure (Ellen MacArthur Foundation, 2020). The partnership oversees both the initial setting-up stage of the waste collection as well as the transport of the collected packaging materials to a designated composting facility (Ellen MacArthur Foundation, 2020). The service rollout was informed by a 12-month pilot programme across a dozen sites in London. The company’s commitments to due diligence and continuous collaboration with the waste industry has positioned Vegware as a sustainability leader and has led to their products being accepted in a range of FW collection schemes across the UK and Ireland. In the near future, similar partnerships may be conceivable at a wider scale and would involve BBP manufacturers, local authorities, waste collectors and AD plant operators.

The comparative network analysis presented in **Chapter 7** provided further evidence towards the roles played by the ability to distinguish between conventional and biodegradable plastics and knowledge of (relevant) disposal routes for BBP waste in enabling circular disposal intentions, aligning with emerging literature on consumer disposal behaviour (Dilkes-Hoffman et al., 2019a; Herbes, Beuthner & Rammer, 2018; Mehta et al., 2020; Sijtsema et al., 2016; Taufik et al., 2020). Policies centred around consumer protection and addressing greenwashing issues related to a currently leaky BBP terminology could have positive knock-on effects on knowledge of disposal by limiting misleading claims on biodegradability and simplifying the BBP landscape. At the same time, they could help regain trust among the general public, as consumers put increasing responsibility on governments to address plastic pollution through tighter environmental legislation (Dilkes-Hoffman et al., 2019b).

Finally, while drastic action, such as a suggested ban on the use of conventional plastics in food packaging, similar to California's Senate Bill SB 54 requiring all state-owned food catering facilities to only serve food in reusable, recyclable or compostable packaging by 2032, may represent a steppingstone towards achieving a circular plastics value chain, **Chapter 8** acts as a reminder that FW prevention must remain at the heart of the debate. Given the contribution of FW towards GHG at both product (i.e. LCA) and sector (i.e. food/agriculture) scales (Crippa et al., 2021; Dilkes-Hoffman et al., 2018), it is important to recognise that the environmental benefits associated with packaging material substitution remain limited and do not represent a panacea to address the ever-growing plastics market, as highlighted below.

3. Limitations & Future research

Limitations and future research related to each study objective(s) were addressed in the corresponding chapters. However, the recognition that substituting conventional plastic food packaging with BBP alternatives does not represent a silver-bullet solution to addressing plastic pollution – and, in some cases, may even exacerbate it – points at a major overall limitation of both BBPs and the perspective adopted in this thesis. Indeed, material substitution alone provides only a limited picture of a sustainable plastics value chain and fails to address a fundamental flaw in the system: its linearity. The majority of BBPs are manufactured with single-use food packaging applications in mind, perpetuating an old adage of the linear economy based on a take-make-use-dispose approach.

Nonetheless, the fact that BBPs are compatible – at least conceptually – with organic waste streams and waste management strategies represents an opportunity to contribute towards closed-loop organics recycling systems and minimise the risk of cross-contamination in both conventional plastic and organic waste streams. This thesis aimed to address the challenges and opportunities within that proposition.

The focus on AD – as opposed to composting, including industrial and home composting – was intentional, based on the advantageous position of AD at the interface between waste management, renewable energy and sustainable agriculture identified in the literature review and the emphasis on AD as preferred waste management strategy across the UK and the EU (**Chapter 3 Section 3.2**). Nonetheless, while composting is to some extent a more mature waste treatment strategy for the treatment BBP waste – several standards for both industrial and home composting exist –, both AD and composting face similar issues, notably around biodegradation and consumer behaviour. Thus, study findings are at least partly relevant to the composting sector (especially industrial composting).

Building on the findings and limitations of this thesis, several directions for future research were identified, including: (1) undertake further co-AD biodegradation studies with relevant study parameters, which would contribute towards a better characterisation of the biodegradation mechanisms of commercialised BBPs; (2) enhance shelf-life extension properties of BBP food packaging without compromising on their biodegradation properties; (3) design field experiments controlling for the key factors that were suggested to influence appropriate disposal of BBPs most systematically; (4) expand the survey to the wider population and additional countries to ensure the findings of the comparative case study analysis can be applied more broadly; and (5) assess organic waste collection schemes at both local and national scales to identify and characterise the most successful schemes, in order to help design effective food and food packaging waste collection and treatment strategies.

In addition, the following key action points were suggested: (1) develop a biodegradation standard for BBP packaging suitable for AD through trilateral collaboration involving the waste management industry, regulators and farmers/land managers and (2) a corresponding disposal logo, which would involve social scientist, designers and consumers. At a local scale, and with immediate effect: (3) introduce, at the very least, separate FW bins at the most concentrated points of consumption (e.g. at on-site institutional catering sites).

Ultimately, this thesis has aimed to uncover the synergies and conflicts that emerge from the use of BBPs in food packaging and their integration in the organic waste stream, as a contribution towards the design and implementation of a circular bioeconomy framework.

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Appendix

1. ANOVA tables for anaerobic co-digestion trials [Chapter 4]

Tables 27-29 present the results from one-way ANOVA analyses for co-digestion trials (**Chapter 4**).

Table 27 | One-way ANOVA table for treatment effects on total biogas production.

	Df	Sum of Squares (SS)	Mean Square (MS)	F-value	p-value	Significance
Treatment	14	161221	11516	1.120	0.378	ns
Error	27	275153	10191			

Table 28 | One-way ANOVA table for treatment effects on total biogas yield.

	Df	Sum of Squares (SS)	Mean Square (MS)	F-value	p-value	Significance
Treatment	14	2515.3	179.66	1.600	0.143	ns
Error	27	3032.2	112.30			

Table 29 | One-way ANOVA table for treatment effects on total methane yield.

	Df	Sum of Squares (SS)	Mean Square (MS)	F-value	p-value	Significance
Treatment	14	1316.8	94.053	1.281	0.281	ns
Error	27	1982.3	73.420			

2. Topic guides for semi-structured interviews [Chapter 5]

This section provides the relevant topic guides that were used to support the interviews with stakeholders conducted as part of **Chapter 5**. The topic guide for civil servants consisted of the following questions:

1. With the planned collection of household food waste, what do you plan to do with it?
2. Are any other industries, such as anaerobic digestion, receiving support of any kind linked to food collection? If so, what kind of support?
3. How are biodegradable plastics considered within the organic waste management debate?
4. The Department for Business, Energy and Industrial Strategy (BEIS) has supported the development of alternative materials to plastics. Do you believe the promotion of biodegradable plastics within the bio-economy and industrial strategy is based on scientific evidence?
5. To what extent do you take industry and academic evidence into account?
6. Where do you believe research efforts should be targeted?

7. In the plastic packaging sector, where does Government prioritise in terms of end-of-life? How do you reconcile conventional plastic recycling and bioplastic innovations?
8. How does the promotion of biodegradable plastics fit with the current revision of the PAS 110 fertiliser regulation to lower threshold for physical contaminants (including plastics)?
9. How do biodegradable plastics fit within the upcoming plastics regulations?

The topic guide for all other stakeholders consisted of the following questions:

1. What is the current proportion of plastic packaging (of any type and bioplastic in particular) in the anaerobic digestion stream? *(where applicable)*
2. What is the major source of food waste that you deal with in your plant? *(where applicable)*
3. What are your thoughts on the promotion of biodegradable plastics for the food waste anaerobic digestion industry sound given the current waste management infrastructure?
4. What are the main barriers to their efficient implementation in food waste anaerobic digestion?
5. Are you aware of any biodegradable plastics that fully biodegrade in anaerobic digestion?
6. If biodegradable plastics were to effectively enhance biogas production, do you believe the anaerobic digestion industry would be more willing to accept them in their process?
7. According to you, are there any areas where biodegradable plastics can play a role and what are they?
8. Where do you believe research efforts should be targeted?
9. What is the impact of biodegradable plastics on digestate quality?
10. With the current revision of the PAS 110 fertiliser regulation and plans to lower the threshold for physical contaminants (including plastics), how do you foresee biodegradable plastics uptake by plant operators and the agricultural sector?

3. Survey questionnaire [Chapter 7]

To gain quantitative insights from relationships between the system elements (**Table 18**), a questionnaire was designed. The questionnaire included constructs from different behavioural/design theories, which a GGM approach is particularly suited for (Bhushan et al., 2019). 16 individual variables reflecting the system elements of the framework were measured by 38 items. Questionnaire items related to effort and knowledge were informed by Wang et al. (2021) and Tonglet, Phillips & Read (2004). Items related to attitudes were adapted from Zwicker et al. (2020) and Dilkes-Hoffman et al. (2019); social and personal norms, from Bhushan et al. (2019), Tonglet, Phillips & Read (2004) and van der Werff, Steg & Keizer (2013); habit by Knussen & Yule (2008). Items related to sensorial and technical features were based on terminology adopted by Brockhaus, Petersen & Kersten (2016) and Karana (2012); those related to value (cost consideration, broader value chain), by Lynch, Klassen & Broerse (2017). Items related to data (marketing, labelling, signposting and consumer education) and infrastructure (treatment and collection systems) categories were newly created to reflect the purpose of this study and were informed by the focus groups. Finally, behavioural intention items were guided by Francis et al. (2004).

The following sections correspond to the questionnaire verbatim, with minor differences between surveys to reflect nuances between British and American English added where relevant.

- **Section I: Introduction & Consent**

You are being invited to take part in an online survey conducted by researchers at Imperial College London/UC Davis. This survey aims to explore how a range of factors interact and influence the disposal behaviour of biodegradable plastic food packaging. As a member of Imperial College London/UC Davis, your participation would contribute towards implementing strategic system design to enable members of the community to adopt more environmentally friendly, circular behaviours.

The survey will take approximately 5-7 minutes to complete, and you will be able to enter a draw to win one of ten GiftPay vouchers worth £/$10, £/$20, £$50 or £/$100 if you enter by Tuesday 21st of December 2021/Wednesday 15th of June 2022.

Any information concerning you will be kept confidential, and only accessible by the research team conducting this survey. By proceeding with the survey, you are giving your consent for your responses to be recorded. You are free to leave the survey at any time, and if you would like more information about this research study, please contact the research team or refer to the Survey Participation Info Sheet below.

Survey Participant Info Sheet

Q0: To proceed, please tick the following box:

- ☑ I consent to take part in this survey

- **Section II: BBP Use & Disposal**

Q1: You will find below examples of **biodegradable plastic food packaging**[1]. To the best of your knowledge, which ones do you **most commonly use/interact with** in your daily life? (Tick as many as apply)

- ☑ None
- ☑ Take-away (e.g. lunch box/bowl, coffee cup, cutlery)
- ☑ Food caddy liner
- ☑ Soft packaging (e.g. salad film, pouch, sandwich wrapper, tea bag)
- ☑ Other (please specify) ___________________________________
- ☑ I am not sure

Q2: Thinking about **biodegradable plastic food packaging** in the previous question, how likely are you to **dispose them in the following way**?

	Extremely unlikely	Somewhat unlikely	Neither likely nor unlikely	Somewhat likely	Extremely likely
Home compost/ Allotment/ Community garden	•	•	•	•	•
General/residual waste	•	•	•	•	•
Dry mixed recycling	•	•	•	•	•
Food waste (brown) recycling	•	•	•	•	•
Open environment	•	•	•	•	•

[1] Images are not shown here but were provided to facilitate understanding through visual support.

- **Section III: System Elements**

Q3: You will now be presented with a number of **statements related to biodegradable plastic packaging and/or their disposal**, split into 5 sections. Please indicate to what extent you agree or disagree with each of the statements. (1/5)

Note**: The numbers in the first column of Q3-Q7 below indicate the corresponding constructs in **Figure 41**

	Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
The texture of the packaging helps me determine if it is biodegradable (1)	•	•	•	•	•
The visual appearance of the packaging material (not the label) helps me determine if it is biodegradable (2)	•	•	•	•	•
The smell of the packaging helps me determine if it is biodegradable (3)	•	•	•	•	•
The durability of plastic packaging helps me determine if it is biodegradable (4)	•	•	•	•	•
Functionality of biodegradable plastic packaging is a key consideration for me (5)	•	•	•	•	•

Q4 (Continued): You will now be presented with a number of **statements on biodegradable plastic packaging and/or their disposal**. Please indicate to what extent you agree or disagree with each of the statements. (2/5)

	Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
Simplicity is important for me when disposing packaging (6)	•	•	•	•	•
Convenience is important for me when disposing of packaging (7)	•	•	•	•	•
Time-efficiency is important for me when disposing of packaging (8)	•	•	•	•	•
I understand the difference between conventional and biodegradable plastics (9)	•	•	•	•	•
I know how to dispose biodegradable plastic packaging appropriately (10)	•	•	•	•	•
I am unsure whether biodegradable plastics are actually better than conventional plastics (11)	•	•	•	•	•
I find the benefits of biodegradable plastic packaging ambiguous (12)	•	•	•	•	•
I am scared of contaminating the waste stream unintentionally (13)	•	•	•	•	•
I am worried I might get a penalty/told off for not disposing biodegradable plastic packaging appropriately (14)	•	•	•	•	•
I trust brands' claims of biodegradability on packaging (15)	•	•	•	•	•

Q5 (Continued): You will now be presented with a number of **statements on biodegradable plastic packaging and/or their disposal.** Please indicate to what extent you agree or disagree with each of the statements. (3/5)

	Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
My peers find it important to be conscious about plastic sustainability (16)	•	•	•	•	•
The opinion of my peers matters to me (17)	•	•	•	•	•
Thinking about plastic sustainability is an important part of who I am (18)	•	•	•	•	•
I see myself as person who thinks about plastic sustainability (19)	•	•	•	•	•
I feel responsible to play my part in addressing plastic pollution (20)	•	•	•	•	•
It would be wrong of me not to recycle my waste (21)	•	•	•	•	•
Recycling/sorting my food and packaging waste is a habit for me (22)	•	•	•	•	•

Q6 (Continued): You will now be presented with a number of **statements on biodegradable plastic packaging and/or their disposal**. Please indicate to what extent you agree or disagree with each of the statements. (4/5)

	Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
It is important for me that brands communicate the role biodegradable plastics play in addressing plastic pollution (23)	•	•	•	•	•
It is important for me that brands communicate how biodegradable plastics are sourced (24)	•	•	•	•	•
I find the written content of labelling useful when disposing packaging (25)	•	•	•	•	•
I find the visual content of labelling useful when disposing packaging (26)	•	•	•	•	•
I find having a clear association between packaging and the relevant bin helpful when disposing packaging (27)	•	•	•	•	•
I find recycling leaflets/posters helpful when disposing packaging (28)	•	•	•	•	•
I find verbal/oral guidance helpful when disposing packaging (29)	•	•	•	•	•
I find recycling websites helpful when disposing packaging (30)	•	•	•	•	•
Awareness campaigns would help consumers make better disposal choices (31)	•	•	•	•	•
Educational programmes at school would help consumers make better disposal choices (32)	•	•	•	•	•

Q7 (Continued): You will now be presented with a number of **statements on biodegradable plastic packaging and/or their disposal**. Please indicate to what extent you agree or disagree with each of the statements. (5/5)

	Strongly disagree	Somewhat disagree	Neither agree nor disagree	Somewhat agree	Strongly agree
I would be willing to pay a little bit extra for biodegradable plastic packaging (33)	•	•	•	•	•
Price is an important consideration for me (34)*	•	•	•	•	•
I see the value of biodegradable plastic packaging because of the sustainability issues they aim to address (e.g. plastic pollution, food waste recycling) (35)	•	•	•	•	•
The presence of the relevant waste management facility makes me more confident that plastic packaging will be treated appropriately (36)	•	•	•	•	•
Currently, I have access to the appropriate waste collection infrastructure for me to dispose of biodegradable plastic packaging (37)	•	•	•	•	•
Over the past two years, I have had access to a uniform waste collection for different waste types (38)	•	•	•	•	•

* Reverse-coded

- **Section IV: Demographics**

Q8: Do you have access to a home garden/community garden/allotment?

☑ No
☑ Yes

Q9: Do you have access to a food waste bin/collection service?

☑ No
☑ Yes

Q10: Are you currently...?

☑ Undergraduate student
☑ Postgraduate student
☑ Academic/Faculty staff member
☑ Research staff member
☑ Teaching staff member
☑ Technical/scientific support staff member
☑ Professional/administrative staff member
☑ Operational staff member

Q11: What is your gender?

☑ Male
☑ Female
☑ Non-binary / third gender
☑ Prefer not to say

Q12: What is your age?

☑ 18-24 years old
☑ 25-34 years old
☑ 35-44 years old
☑ 45-54 years old
☑ 55-64 years old
☑ 65-74 years old
☑ 75 years or older

- **Section V: Optional Raffle Entry**

Q00: If you would like to enter the draw for a chance to win any of £/$10, £/$20, £/$50 and £/$100 GiftPaid vouchers, please provide your institutional email address (or personal, if you don't have one). Your details will not be included in the data analysis.

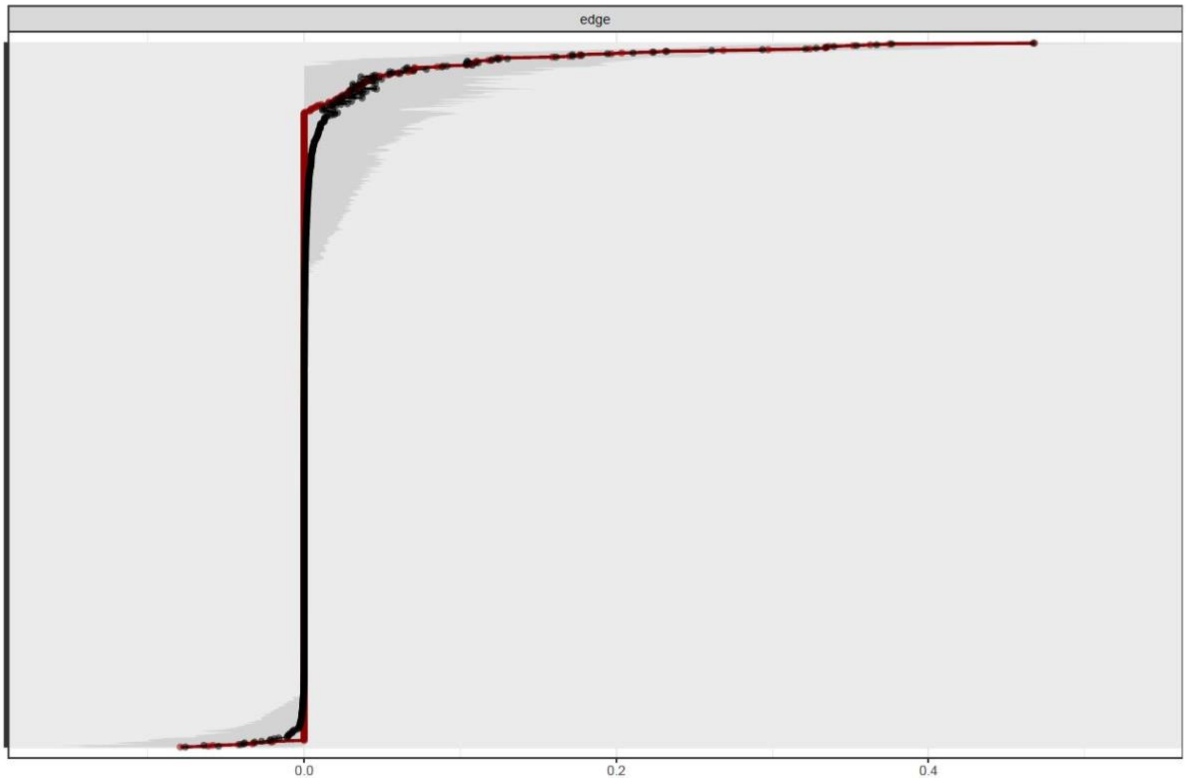
4. Network stability analysis [Chapter 7]

A bootstrapping method was applied to assess to overall robustness of the network, using the `bootnet` package in R. As a random sampling method, bootstrapping estimates the precision of a statistic of interest by using random subsets of the original dataset or resampling the dataset with replacement to create new plausible datasets and can be used to appraise the validity of psychological networks (Epskamp, Borsboom & Fried, 2018).

First, edge-weight accuracy was estimated. 95% confidence intervals (CIs) were used to assess the variability of edge-weights, such that in 95% of cases a CI will contain the true value of the parameter. **Figure 48** shows the bootstrapped CIs for the estimated edge-weights from the bootnet function. The bootstrapped intervals of the strongest relationships in the network do not overlap the confidence intervals of the weakest edges. This indicates that the key relationships displayed in the graph are estimated reliably (Epskamp, Borsboom & Fried, 2018).

Constructing CIs for centrality indices is more challenging than for edge-weights (Epskamp, Borsboom & Fried, 2018). For this reason, the stability of centrality indices was investigated instead, based on subsets of the data (called case-dropping, specified by type = “case”) in the bootnet function. Stability can be interpreted as the consistency in the order of centrality indices after dropping cases, where a case corresponds to a single survey participant’s response (i.e. a single row in the dataset). **Figure 49** shows the resulting plots. The closeness index was not displayed due to lack of variance. The stability of betweenness drops more steeply than that of the strength index, and both appear less stable in the UCD dataset than the ICL dataset. This stability can be quantified using the CS-coefficient, which quantifies the maximum proportion of cases that can be dropped to retain, with 95% certainty, a correlation with the original centrality of higher than (by default) 0.7. The CS-coefficient indicates that in both networks, betweenness (CS_{ICL} = 0.05 and CS_{UCD} = 0.042) is not stable under case-dropping bootstrapping. Node strength performs better (CS_{ICL} = 0.672 and CS_{UCD} = 0.595), reaching the cut-off of 0.5 required for a metric to be considered stable. Therefore, we conclude that the order of node strength is interpretable with some care, while the orders of betweenness and closeness are not.

A



B

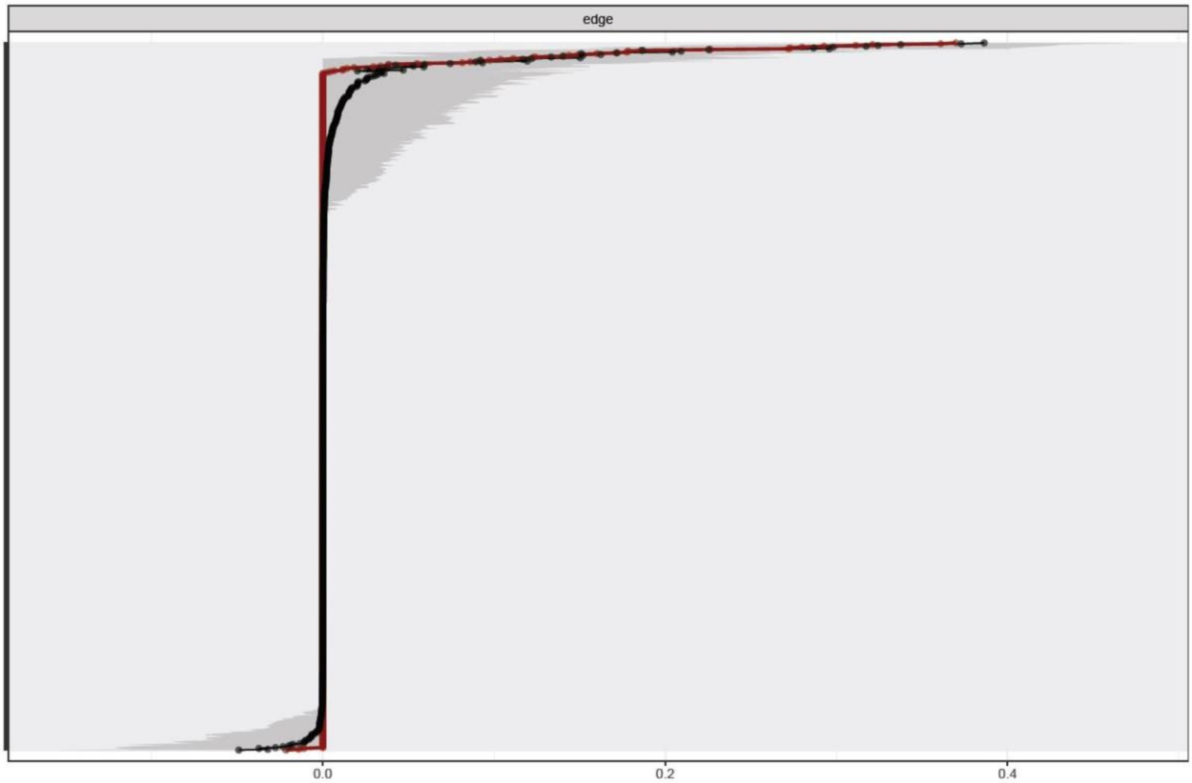
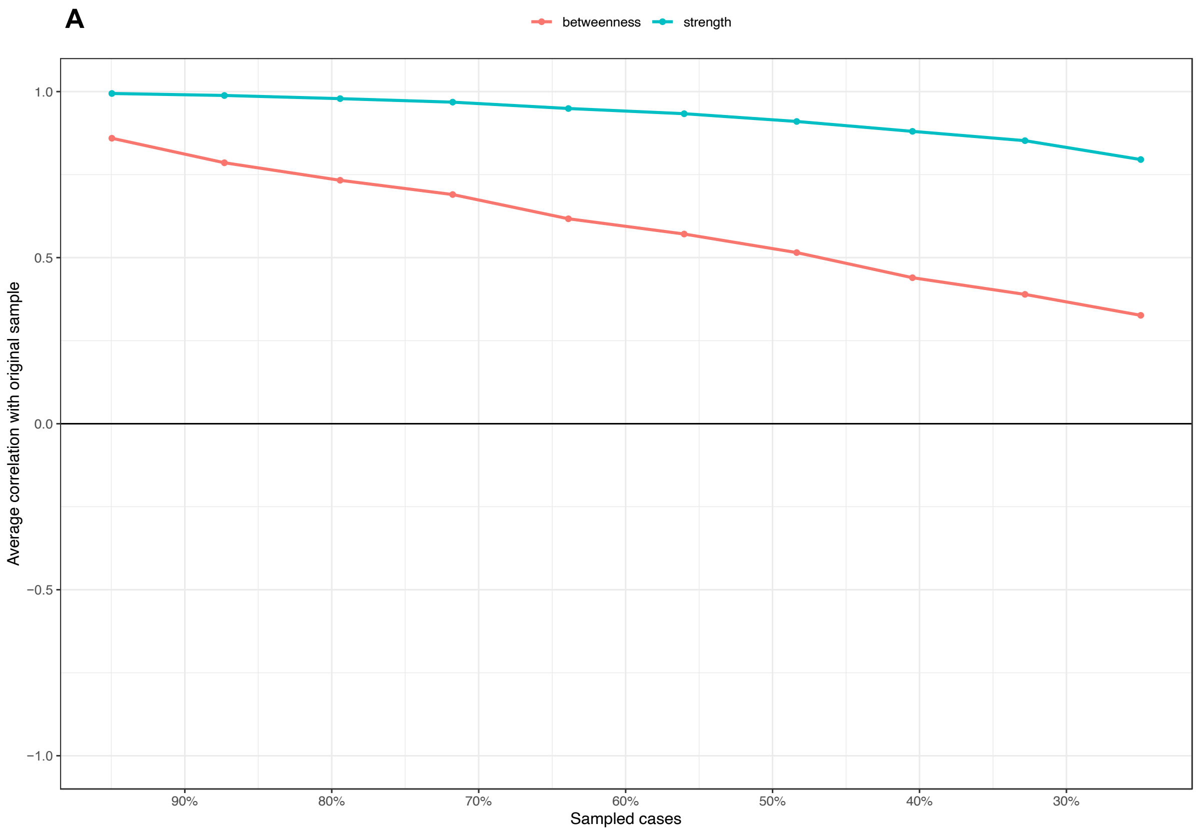


Figure 48 | Bootstrapped confidence intervals (CIs) of estimated edge-weights for the estimated ICL (A) and UCD (B) networks. The red dots indicate the sample estimates, while the black dots represent the bootstrap mean (i.e. the mean of all bootstrap simulations obtained from resampling). The grey area the bootstrapped 95% Cis around the strength of a particular edge. Each horizontal line represents one edge of the network, ordered from the edge with the highest edge-weight to the edge with the lowest edge-weight. The y-axis labels, which indicate the edges in the Gaussian graphical model, have been removed to avoid cluttering. Plot based on a non-parametric bootstrap with 2,500 samples.



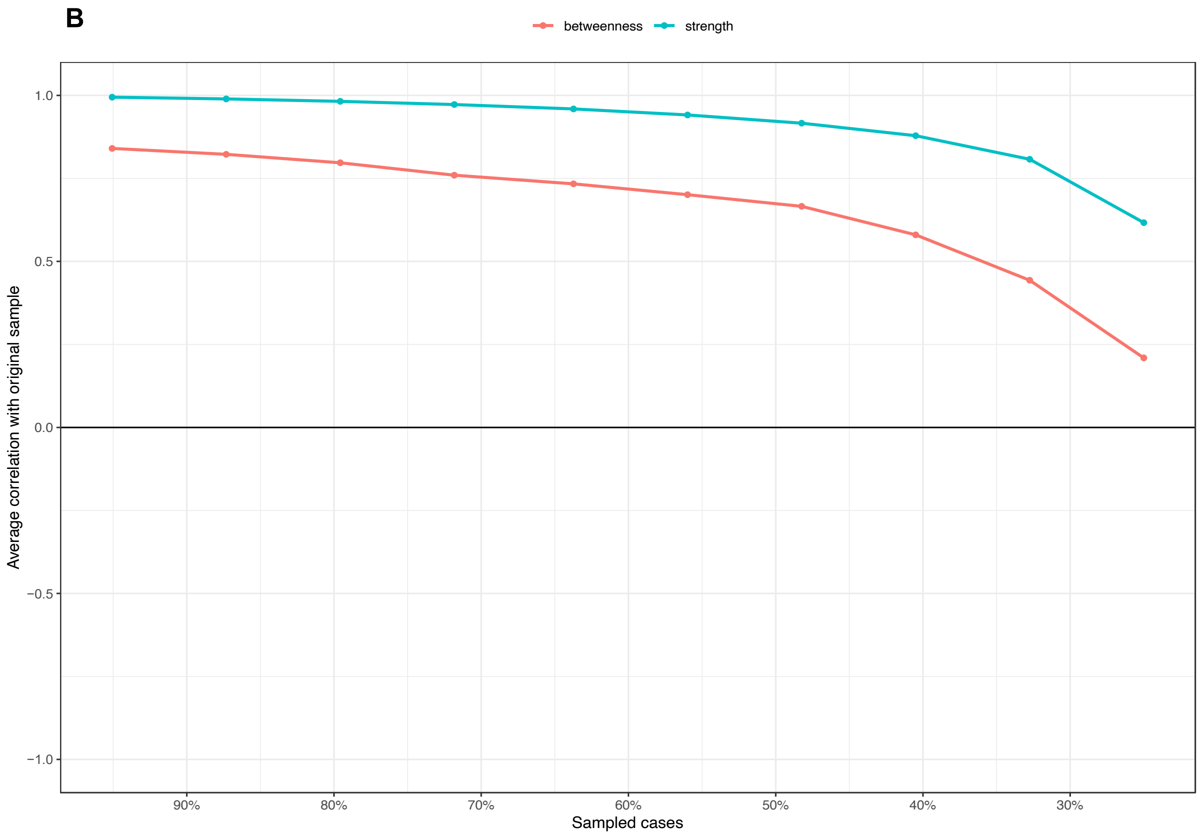


Figure 49 | Average correlations between centrality indices of networks sampled with number of cases dropped and the original ICL (A) and UCD (B) datasets. Lines indicate the means and areas indicate the range from the 2.5th quantile to the 97.5th quantile. Red line: betweenness; blue line: strength. Plots based on a non-parametric bootstrap with 2,500 samples.

5. ANOVA tables [Chapter 8]

Tables 30-49 present the results from two-way ANOVA analyses for the shelf-life study (**Chapter 8**).

Table 30 | Two-way ANOVA results for treatment and day interaction for June and September trials. Df: degrees of freedom (interaction, error/residuals); ns: non-significant. Values below 0.001 are reported as < 0.001.

TREATMENT	TRIAL	Df	F-VALUE	p-VALUE	SIGNIFICANCE
Overall	June	4, 50	17.323	< 0.001	***
	September	7, 80	2.337	0.032	*
Colour	June	4, 50	3.908	0.008	**
	September	7, 80	1.285	0.268	ns
Decay	June	4, 50	2.444	0.059	ns
	September	7, 80	1.074	0.388	ns
Wilting	June	4, 50	21.934	< 0.001	***
	September	7, 80	14.282	< 0.001	***
Weight loss	June	4, 50	65.628	< 0.001	***
	September	7, 80	49.533	< 0.001	***
Water exchange	June	4, 50	16.630	< 0.001	***
	September	7, 80	1.663	0.130	ns
Relative humidity	June	4, 50	2.059	0.100	ns
	September	7, 80	0.596	0.757	ns
O_2	June	4, 50	0.119	0.975	ns
	September	7, 80	1.947	0.073	ns
CO_2	June	4, 50	0.292	0.552	ns
	September	7, 80	1.311	0.256	ns

Table 31 | Two-way ANOVA table for treatment and time effects on overall visual quality (June).

	Df	Sum of Squares (SS)	Mean Square (MS)	F-value	p-value	Significance
Day	1	232.320	232.320	667.586	< 0.001	***
Treatment	4	54.900	13.725	39.440	< 0.001	***
Day*Treatment	4	24.113	6.028	17.323	< 0.001	***
Error	50	17.400	0.348			

Table 32 | Two-way ANOVA table for treatment and time effects on overall visual quality (September).

	Df	Sum of Squares (SS)	Mean Square (MS)	F-value	p-value	Significance
Day	1	271.502	271.502	415.830	< 0.001	***
Treatment	7	88.990	12.713	19.471	< 0.001	***
Day*Treatment	7	10.681	1.526	2.337	0.032	*
Error	80	52.233	0.653			

Table 33 | Two-way ANOVA table for treatment and time effects on discolouration (June).

	Df	Sum of Squares (SS)	Mean Square (MS)	F-value	p-value	Significance
Day	1	14.963	14.963	62.871	< 0.001	***
Treatment	4	1.600	0.400	1.681	0.169	ns
Day*Treatment	4	3.720	0.930	3.908	0.008	**
Error	50	17.400	0.238			

Table 34 | Two-way ANOVA table for treatment and time effects on discolouration (September).

	Df	Sum of Squares (SS)	Mean Square (MS)	F-value	p-value	Significance
Day	1	18.408	18.408	111.01	< 0.001	***
Treatment	7	3.458	0.494	2.979	0.008	**
Day*Treatment	7	1.492	0.213	1.285	0.268	ns
Error	80	13.267	0.166			

Table 35 | Two-way ANOVA table for treatment and time effects on decay (June).

	Df	Sum of Squares (SS)	Mean Square (MS)	F-value	p-value	Significance
Day	1	13.653	13.653	56.889	< 0.001	***
Treatment	4	1.333	0.333	1.389	0.251	ns
Day*Treatment	4	2.347	0.587	2.444	0.059	ns
Error	50	12.000	0.240			

Table 36 | Two-way ANOVA table for treatment and time effects on decay (September).

	Df	Sum of Squares (SS)	Mean Square (MS)	F-value	p-value	Significance
Day	1	4.602	4.602	11.738	< 0.001	***
Treatment	7	16.490	2.356	6.008	< 0.001	***
Day*Treatment	7	2.948	0.421	1.074	0.388	ns
Error	80	31.367	0.392			

Table 37 | Two-way ANOVA table for treatment and time effects on wilting (June).

	Df	Sum of Squares (SS)	Mean Square (MS)	F-value	p-value	Significance
Day	1	49.613	49.613	218.882	< 0.001	***
Treatment	4	29.567	7.392	32.610	< 0.001	***
Day*Treatment	4	19.887	4.972	21.934	< 0.001	***
Error	50	11.333	0.227			

Table 38 | Two-way ANOVA table for treatment and time effects on wilting (September).

	Df	Sum of Squares (SS)	Mean Square (MS)	F-value	p-value	Significance
Day	1	69.008	69.008	217.921	< 0.001	***
Treatment	7	113.833	16.262	51.353	< 0.001	***
Day*Treatment	7	31.658	4.523	14.282	< 0.001	***
Error	80	25.333	0.317			

Table 39 | Two-way ANOVA table for treatment and time effects on weight loss (June).

	Df	Sum of Squares (SS)	Mean Square (MS)	F-value	p-value	Significance
Day	1	3246.600	3246.600	422.413	< 0.001	***
Treatment	4	4170.900	1042.700	135.670	< 0.001	***
Day*Treatment	4	2017.600	504.400	65.628	< 0.001	***
Error	50	384.300	7.700			

Table 40 | Two-way ANOVA table for treatment and time effects on weight loss (September).

	Df	Sum of Squares (SS)	Mean Square (MS)	F-value	p-value	Significance
Day	1	5958.900	5958.900	697.215	< 0.001	***
Treatment	7	7772.200	1110.300	129.912	< 0.001	***
Day*Treatment	7	2963.400	423.300	49.533	< 0.001	***
Error	80	683.700	8.500			

Table 41 | Two-way ANOVA table for treatment and time effects on water exchange (June).

	Df	Sum of Squares (SS)	Mean Square (MS)	F-value	p-value	Significance
Day	1	1.296	1.296	27.439	< 0.001	***
Treatment	4	18.471	4.618	97.748	< 0.001	***
Day*Treatment	4	3.143	0.786	16.630	< 0.001	***
Error	50	2.362	0.047			

Table 42 | Two-way ANOVA table for treatment and time effects on water exchange (September).

	Df	Sum of Squares (SS)	Mean Square (MS)	F-value	p-value	Significance
Day	1	394.060	394.060	38.563	< 0.001	***
Treatment	7	1064.190	152.03	14.878	< 0.001	***
Day*Treatment	7	118.950	16.990	1.663	0.130	ns
Error	80	817.470	10.220			

Table 43 | Two-way ANOVA table for treatment and time effects on relative humidity (June).

	Df	Sum of Squares (SS)	Mean Square (MS)	F-value	p-value	Significance
Day	1	117.944	117.944	38.027	< 0.001	***
Treatment	4	161.400	40.350	13.010	< 0.001	***
Day*Treatment	4	25.544	6.385	2.059	0.100	ns
Error	50	155.079	3.102			

Table 44 | Two-way ANOVA table for treatment and time effects on relative humidity (September).

	Df	Sum of Squares (SS)	Mean Square (MS)	F-value	p-value	Significance
Day	1	170.270	170.270	155.110	< 0.001	***
Treatment	7	127.850	18.264	16.638	< 0.001	***
Day*Treatment	7	4.580	0.654	0.596	0.757	ns
Error	80	87.819	1.098			

Table 45 | Two-way ANOVA table for treatment and time effects relative O_2 concentration (June).

	Df	Sum of Squares (SS)	Mean Square (MS)	F-value	p-value	Significance
Day	1	0.056	0.056	0.222	0.640	ns
Treatment	4	16.756	4.189	16.516	< 0.001	***
Day*Treatment	4	0.121	0.030	0.119	0.975	ns
Error	50	12.681	0.254			

Table 46 | Two-way ANOVA table for treatment and time effects relative O_2 concentration (September).

	Df	Sum of Squares (SS)	Mean Square (MS)	F-value	p-value	Significance
Day	1	0.090	0.090	0.395	0.532	ns
Treatment	7	23.684	3.383	14.806	< 0.001	***
Day*Treatment	7	3.115	0.445	1.947	0.073	ns
Error	80	18.282	0.229			

Table 47 | Two-way ANOVA table for treatment and time effects on relative CO_2 concentration (June).

	Df	Sum of Squares (SS)	Mean Square (MS)	F-value	p-value	Significance
Day	1	0.155	0.155	0.526	0.472	ns
Treatment	4	18.163	4.541	15.438	< 0.001	***
Day*Treatment	4	0.343	0.086	0.292	0.552	ns
Error	50	14.706	0.294			

Table 48 | Two-way ANOVA table for treatment and time effects on relative CO_2 concentration (September).

	Df	Sum of Squares (SS)	Mean Square (MS)	F-value	p-value	Significance
Day	1	0.000	0.003	0.001	0.974	ns
Treatment	7	39.208	5.601	18.387	< 0.001	***
Day*Treatment	7	2.796	0.399	1.311	0.256	ns
Error	80	24.371	0.305			

Table 49 | Two-way ANOVA results for treatment and trial interaction. Df: degrees of freedom (interaction, error/residuals). Tests for overall visual quality, decay, discolouration, wilting and weight loss were performed on day 15 samples only; for the remaining variables, all time points were used. Two-way ANOVAs were also conducted for day 15 only on those variables and resulted in the same significance levels.

Treatment	Df	F-value	p-value	Significance
Visual overall	3, 16	22.667	< 0.001	***
Decay	3, 16	3.451	0.042	*
Discolouration	3, 16	18.133	< 0.001	***
Wilting	3, 16	11.458	< 0.001	***
Weight loss	3, 16	5.375	0.009	**
Water exchange	3, 88	8.931	< 0.001	***
Relative humidity	3, 88	5.417	0.002	**
O_2	3, 88	0.193	0.901	ns
CO_2	3, 88	0.915	0.437	ns

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