Received: 13 June 2023 | Revised: 8 November 2023 | Accepted: 15 November 2023

DOI: 10.1002/sd.2839

RESEARCH ARTICLE

Sustainable Development WILEY

Adopting a socio-material perspective on life cycle assessment: Environmental impacts of circular tableware systems in Italy's bioplastics context

Anna M. Walker[1,2] | Alberto Simboli[1] | Walter J. V. Vermeulen[2] | Andrea Raggi[1]

[1]Department of Economic Studies, University "G. d'Annunzio", Pescara, Italy

[2]Copernicus Institute of Sustainable Development, Utrecht University, Utrecht, the Netherlands

Correspondence
Anna M. Walker, Department of Economic Studies, University "G. d'Annunzio", Viale Pindaro 42, 65127 Pescara, Italy.
Email: anna.walker@unich.it

Funding information
Horizon 2020 Framework Programme, Grant/Award Number: 765198

Abstract

Life cycle assessment (LCA) is amongst the most frequently used methodologies to identify and evaluate the environmental impacts of the circular supply chain configurations. This article aims to showcase how a socio-material perspective can increase the embeddedness of such an assessment within inter-firm networks. A comparative LCA of two circular product systems is conducted, namely single-use and reusable tableware within a canteen located in northern Italy. To contextualise the LCA, the Actor Network Theory (ANT) is used as an epistemological lens to identify the framing and problematisation of bioplastics in Italy, as well as the power constellation of actors affecting the bioplastics life cycle. The ANT complements the LCA in three ways: firstly, it informs the end-of-life modelling of the product systems away from public narratives to the actual waste management practices and secondly, it contextualises the interpretation of the LCA results—in favour of the reuse system—with the socio-technical factors in Italy. Finally, the socio-material perspective allows for a discussion on the performative role of LCA in the light of its increasing popularity in the public and private sectors and on its potential to guide more sustainable production and consumption patterns.

KEYWORDS

actor network theory, circular economy, compostable plastics, reusable tableware, single-use plastics, sustainability assessment

1 | INTRODUCTION

As part of the first Circular Economy Action plan, the European Single Use Plastics Directive 2019/904 has now banned certain single-use plastics (SUP) products from being sold within the European single market (Camilleri, 2020). Up to 50% of plastic waste in the EU comes from single-use packaging and products (Elliott et al., 2020) and with this directive, the EU aims to reduce greenhouse gas emissions by 2.63 million tonnes per year and the amount of marine pollution from SUP by 4850 tonnes per year (European Commission, 2018). Some of the 10 product categories considered in the Directive are just to be reduced, or subject to eco-design requirements. However, cotton bud sticks, cutlery, plates, straws and stirrers, balloons and sticks for balloons from any type of SUP, as well as food containers from expanded polystyrene (EPS) are prohibited from being produced and sold. This ban of SUP products has large implications especially for the catering sector, where the use of single-use tableware is common (ANGEM, 2020; Fieschi & Pretato, 2018). The COVID-19 pandemic has further increased the tendency to opt for single-use products for hygienic reasons (European Environment Agency, 2021; Leal Filho et al., 2021). Herberz et al. (2020) have found that any type of single-use products, even if it is not SUP, but made of paper or other bio-based materials, generally has a more detrimental effect on the environment than opting for a reusable product system. In line with these findings, several meta-studies comparing single-use and reuse systems with life cycle assessments (LCAs) have concluded that the reuse options are often either comparable or preferable to single-use systems in terms of environmental impact (Coelho, Corona, & Worrell, 2020; Lewis et al., 2021a, 2021b; Miliutenko et al., 2020; Walker, 2022; Watson, 2021). LCA is currently one of the most used and internationally standardised environmental assessment methods, frequently also applied in an EU policy context (Bishop et al., 2021a; European Commission, 2018; Sala et al., 2021). Given its scientific rigour and coverage of a broad set of impact categories, it is expected to also take an important role in the upcoming EU legislation on green claims (European Commission, 2023a) and the recently implemented Corporate Sustainability Reporting Directive (European Commission, 2023b). Findings from these LCA studies are relevant in the context of a Europe's transition towards a circular economy (CE) (Camilleri, 2020), as they confirm that the higher value retention option (Reike et al., 2018), reuse, tends to have lower environmental impacts than lower value retention options: recycling or recovery. With regards to sustainable development, LCA is particularly relevant for evaluating the environmental sustainability of products, thus supporting the sustainable development goal (SDG) 12 concerned with sustainable production and consumption. Moreover, it can also address SDG6 on water reuse and efficiency, SDG7 on energy efficiency, SDG8 on resource decoupling, SDG9 on sustainable industrialisation and SDG13 on climate action (Sanyé-Mengual & Sala, 2022).

EU member countries transposing the SUP directive into national law have chosen different magnitudes of stringency with regards to banning SUP in the short and long run (Copello et al., 2021). In Italy, public, private and academic actors argue (Fieschi & Pretato, 2018; Pellicanò, 2021; Rosenboom et al., 2022) that using bioplastics[1] is actually a CE strategy and thus in line with the overall Circular Economy Action Plan under the Green Deal of the EU. However, it was shown by Rossi et al. (2015) and Utilitalia (2020), the Italian Confederation of Multi-utility Companies, that bioplastics are actually mostly not composted, but incinerated, because they are not visually distinguishable from fossil-based plastic impurities in the organic waste fraction. To get a better overview of the respective context and its relevance for potential environmental assessments, Niero et al. (2021) as well as Baumann and Lindkvist (2022) propose the use of socio-material perspectives. This epistemological lens on the interaction between society and technology helps to obtain a deeper understanding of the main actors, their power constellations, and the processes that connect them along the value chain of the systems (Babri et al., 2018).

Therefore, this paper showcases the application of a socio-material analysis, the Actor Network Theory (ANT), in the Italian context of bioplastics to support the scoping of a LCA and the interpretations of its results. The hypothetical example is situated in northern Italy, where the environmental impact between a single-use system and a reuse system in a company canteen is analysed through a comparative LCA. Tableware was chosen as a suitable sub-category of the banned products under the SUP, given the Italian Bioplastics Association has recorded sales growth rates of bio-based tableware of 123% in 2019 and 116% in 2020, making it the fastest growing bioplastics category (Arcelli, 2020, 2021). Combining the LCA with the descriptive ANT perspective can capture how these actors active in the bioplastics sector are related and how they use their agency to push certain narratives. This information is meant to improve the basis of decision-making and to identify the role which the LCA itself, used to determine the potential environmental impact, plays in this context (Law, 2008).

The article is structured as follows: Section 2 will outline the results of previous studies comparing single-use and reuse systems, the effect of LCA positionality, previous attempts to systematically embed LCA in its local context and the proposition of using ANT for this endeavour. Thereafter, the method for the case study will be presented, including elaborations on the Italian context and insights from the theoretical background for the sensitivity analysis of the LCA (Section 3). Section 4 discusses and contextualises the LCA results of the case study with the recent developments in the Italian bioplastics sector and presents an analytical model of the socio-material relations. Finally, the conclusion presents the limitations of the study and future research avenues.

2 | THEORETICAL BACKGROUND

Walker (2022) provides a brief review of 30 publications which compare the environmental impacts of single-use tableware covered by the SUP directive with those of reusable tableware by means of a LCA. As anticipated by previous studies, it was found that most impact categories were in favour of the reuse systems over the single-use systems, if reuses were above a certain number, and transport distances were short (Walker, 2022). The single-use systems made of bio-based plastics were found to have higher impacts in acidification, ionising radiation, ozone depletion and freshwater eutrophication compared to other bio-based single-use options (Fieschi & Pretato, 2018; Moretti et al., 2021). These other options consisted, for example, of wood, paper, cardboard or leaves, instead of bio-based polymers. If opting for single-use alternatives, Herberz et al. (2020) underline the importance of establishing the sustainable sourcing of virgin material, by means of product certification. While several LCA studies exist comparing single-use bioplastics with fossil-based plastics options and reuse options for tableware in the take-away sector (Fieschi & Pretato, 2018; Genovesi et al., 2022; Ramboll, 2021; Razza et al., 2009), LCAs specifically concerning the catering sector for stationary settings are scarcer. Genovesi et al. (2022) have conducted a similar comparison with different material choices, but they did not analyse the context of the LCA setting in more detail. Furthermore, this LCA is focussing on Italy, while another similar study (with a fossil-based single-use scenario) was conducted in the United States (Antony & Gensch, 2017). Before starting with modelling the system, it is essential to reflect the use of the LCA itself as well as the social-material context, where the assessment is taking place.

2.1 | Positionality affecting LCA

LCA is considered as one of the most scientifically robust assessment methodologies for evaluating the environmental impact of products throughout their life cycle, also shedding light on trade-offs between different impact categories (Helling, 2017; Sala et al., 2013). These characteristics have made it an essential tool for eliciting trust in the private sector, through, for example, the Environmental Product Declaration Label, and in the public sector, where it is part of, for example, the Better Regulation Toolbox for sound EU policy making (Sala et al., 2021). However, it was also shown that the results of LCA reports and studies were often in line with the perspectives of the organisations commissioning the LCA which has been pointed out as an issue by Pryshlakivsky and Searcy (2021) and Freidberg (2018). This was the case, for example, in the study by Ramboll (2021), commissioned by the European Paper Packaging Alliance, showing favourable results of single-use paper tableware over any other option in the take-away sector, and the study commissioned by a dishwasher manufacturer (Antony & Gensch, 2017), presenting superior results for reusable systems. It needs to be underlined that the Ramboll (2021) study was the only one clearly favouring paper single-use tableware over several types of reusable tableware, which can be partly explained by the low estimated number of reuses (e.g., 200 for a ceramic plate) and a modelling approach with a 30% increased energy use due to drying of the polypropylene dishes. In contrast, the report by Pro.mo (2015) documented favourable results for the reuse-system, while being commissioned by the Italian packaging consortium. There were also cases where scholars (Woods & Bakshi, 2014) have re-analysed older studies as, for example, the one by Ligthart and Ansems (2007) and contextualised it with updated data, reaching a different result (in favour of reuse). These findings underline the need for close scrutiny of LCA studies in terms of their modelling assumptions, especially in contexts that use these LCAs' results to inform decision-making or to make favourable environmental claims about products to consumers (Baitz et al., 2013; Freidberg, 2018).

2.2 | Embedding LCA in its social context

Along this line of reasoning, Niero et al. (2021) pointed out that a LCA might not suffice to predict the environmental impact of a reuse system, if the social system into which the product system is embedded is not analysed priorly or concurrently. Therefore, Niero (2023) proposes a decision-making framework combining ANT and lifecycle based assessment methodologies to facilitate the implementation to the upcoming EU packaging regulation. Baumann and Lindkvist (2022) also underlined that including the socio-material dimension of the analysis is essential to identify potential actions that induce more sustainable outcomes. To date, Niero (2023), Niero et al. (2021) and Baumann and Lindkvist (2022) are amongst the few scholars actively applying and advocating for the use of socio-material perspectives in connection to LCAs, which is why this study aims to advance knowledge in this field. Of the reviewed papers only Baumann et al. (2018) included the potential social uptake of reuse systems in their study, favouring a semi-centralised distribution of reusable food containers. They reasoned that mere reliance on consumers to push for reuse schemes was not realistic, requiring a certain degree of centralisation as well as financial incentives to launch successful reuse systems (Coelho, Corona, ten Klooster, & Worrell, 2020). Dorn and Stöckli (2018) have also found that if customers witness others making use of reusable take-away boxes, they are only slightly, but statistically significantly, more likely to also opt for the reusable option. At the same time, a negative effect on the propensity to reuse has been observed when reusable take-away containers exhibited signs of use (Collis et al., 2023). Taking into account such contextual factors when modelling a system that depends strongly on the collaboration of consumers and other actors can increase the validity of the LCA results (Niero, 2023; Niero et al., 2021).

2.3 | An ANT perspective on LCA

Before starting a LCA, it is essential to understand the socio-material aspects of the local context into which this assessment is embedded, because this can increase the adequacy of scenarios, especially for the end-of-life (EoL) options (Bishop et al., 2021a; Cristóbal et al., 2023). To capture the Italian bioplastics context, the socio-material perspective called ANT (Latour, 1996; Law, 1992), used to describe the interaction of human and non-human actors, such as materials, discourses, organisations, people, technologies, and different calculative devices, is briefly explained. The goal of adopting this perspective is to better characterise the network of actors and their relations as to understand their role in bioplastics management along the life cycle. Rather than a methodology, ANT can be described as an epistemological lens with which actor networks can be analysed (Latour, 2005). It uses the principles of translation and punctualisation, amongst others, to describe the organisation and stability of systems composed of heterogeneous actors (Steen, 2010). Translation can elicit how one actor with agency can initiate the connection of, for example, people, statements, machines, and laws which produce a temporary stability through relationally organised interactions. Actors have agency, if they can influence the behaviour of other actors (Sayes, 2014). In this process, some actors are more durable than others, as for instance oral communication is less durable than capturing the content of this speech in a piece of text (Law, 1992). This stability (both temporal or spatial) is an outcome of the relations within the respective network and not inherent in the actors which make up this network (Steen, 2010). Translation can also lead to the punctualisation of actors to the point that a network of connections underlying the punctualised actor are no longer visible, or necessary to be understood (Babri et al., 2018). An example would be LCA results, which can be viewed as given, as long as they are not challenged by other actors, requiring a deconstruction of this punctualised unit into the network of its underlying assumptions, LCIA methods, data and people involved in the establishment of these results. Analysing the LCA process and its results in this light can help to produce meaningful and actionable results (Baumann & Lindkvist, 2022).

3 | METHODS AND MATERIALS

The method used in this paper is a comparative LCA, the modelling and result interpretation of which is supported by the ANT perspective. The comparative LCA is in line with the ISO 14040 and ISO 14044 (ISO, 2006a, 2006b) and compares tableware in a business as usual (single-use) system with the reuse system for a stationary canteen setting. This section continues with a description of the wider context of bioplastics issue in Italy, relevant for the LCA modelling and interpretation of its results, followed by design of the LCA study.

3.1 | Italian context of bioplastics

The Italian regulation allows the continuous production and sales of bio-based single-use plastics (Decreto Legislativo n. 196/2021), while most other EU countries have transposed the SUP directive into laws that explicitly favour reuse and include bioplastics in the materials banned (Azzurro, 2021). Notwithstanding, the Guidelines by the European Commission on the implementation of the SUP directive, drafted in June 2021 as an answer to an initial Italian legislative proposal, left little doubt that bio-based plastics was to be included in the banned plastics materials for single-use products (European Commission, 2021a). As a reason for the Italian exception, the country's particular situation, namely its investment in research and development in this sector, employment of 2780 full time employees, 815 million € in sales, production of 110 million tonnes in the bioplastics sector alone in 2020, was cited (Arcelli, 2021). Furthermore, 13,000 workers are partially involved in the bioplastics production and companies which also produce bioplastics besides fossil-based plastics have sales of over 5.3 billion € (Arcelli, 2021). Most of the production is concentrated in Northern Italy, with several large companies leading the market (Vinci et al., 2021). Bioplastics is thus also advertised as sustainable, because it is part of the bioeconomy and considered as plastics produced locally, providing employment (though the raw materials are coming from abroad) (Biber-Freudenberger et al., 2020). But more importantly, the government underlined Italy's virtuous organic waste management, a leading example in Europe, which is supposed to feed back the regenerative feedstock of the bioplastics into the soil through composting (Vinci et al., 2021). Somewhat undermining this argument, in their position paper on 'Managing and recovering', Utilitalia (2020), the Italian consortium of multi-utility companies, stated that they had no suitable equipment to sort and manage the EoL phase of bioplastics in their industrial composting plants, at the quantities that are being released to the market. It is confirmed by literature (Rosenboom et al., 2022) that the sorting machinery needed for purer composting streams are still not available at scale. Even if the bioplastics is labelled as compostable, under the EN 13432, requiring it to disintegrate after 12 weeks and completely biodegrade after 6 months, this is not necessarily achievable under current industrial composting practices in Italy (Utilitalia, 2020). This is also connected to the development period of multi-utility plants, often taking over 10 years; they were planned at a time when bioplastics did not yet exist on an industrial scale. Therefore, bioplastics is often treated as normal plastics (also due to its visual indistinguishability) and sent to incineration after sorting it out (Rossi et al., 2015). Utilitalia (2020) also underline, similar to the findings in Rosenboom et al. (2022) and WRAP (2021), that consumers often confuse bioplastics with petrochemical plastics and hence do not properly dispose of them in the organic waste fraction. Therefore, despite being considered as circular by the Italian packaging consortium CONAI, the bioplastics Extended Producer Responsibility (ERP) consortium Biorepack, and the Association Bioplastics, products from bioplastics in Italy are still mostly incinerated or landfilled, depending on the region (Utilitalia, 2020). Incidentally, several scholars found that incineration and landfilling might potentially be the preferable EoL management options when compared to industrial composting from an environmental point of view, given the low emissions in landfills and the energy recovery during incineration (Cosate de Andrade et al., 2016; Hottle et al., 2017; Rossi et al., 2015). Only mechanical recycling of bioplastics was considered superior, as it avoided the agricultural cultivation of the biomass for the production of bioplastics pellets, being the most impactful stage of the bioplastics life cycle (Cristóbal et al., 2023; Maga et al., 2019). However, this option is again not yet available at industrial scale (Moretti et al., 2021; Rosenboom et al., 2022).

3.2 | Goal and scope definition

Within this setting, the study aims to compare the environmental impacts of the current single-use system of a canteen using single-use tableware with a respective hypothetical reuse system. The canteen of a large company, located in Northern Italy, provides about 540 meals a day during lunchtime, 5 days a week. The canteen currently does not have a dishwasher and the company has contracted a catering service to deliver and cook the food daily.

The functional unit is serving 540 daily lunch meals 1000 times. For every meal, a new set of disposable tableware is produced, whereas reusable tableware can be used as long as it is not broken, given that its life span is expected to be at least 1000 uses (Antony & Gensch, 2017). Because in this canteen setting it is required that more than 540 reusable tableware sets are available to account for fluctuating numbers of canteen guests, the reuse scenario includes the production of 125% of 540 tableware sets, that is, 675, of which, however, only 540 are assumed to be washed 1000 times. The risk of breakage is included in the conservative average number of reuses estimated, as some canteens have reported use cycles of up to 1700 uses for their tableware (Antony & Gensch, 2017).

Regarding the system boundary, this LCA is modelled from cradle to grave and accounts for the environmental impacts of material extraction, manufacturing, transport, use and EoL scenarios of tableware used for providing the meal service. The amount of packaging material was only included for the single-use system, where in the study of Antony and Gensch (2017) it made up between 5% and 7% of the total impacts throughout the life cycle. Due to the high number of reuses in the reuse system, the impact of packaging of reusable tableware was omitted. Napkins were excluded because paper napkins were presumed to be used in both systems. Furthermore, the meal preparation and food waste were not considered either, because they were assumed not to differ between the two systems. The production of the dishwasher, as well as its EoL were not included, because the share of the relevant impacts that would be allocated to the functional unit is supposed to be minimal, given the machine's life span of 10–20 years (Rüdenauer et al., 2011).

The authors used the Ecoinvent 3.5 database (Wernet et al., 2016) and SimaPro 9.0 by Pré-Consultants and applied the Environmental Footprint 2.0 impact assessment method, adapted to the SimaPro software. This method is in line with the ambitions of the EU to standardise the impact assessments within the EU, and considers a rich set of impact categories (Bishop et al., 2021a). It needs to be underlined that this study does not follow the product environmental footprint guidelines in a strict sense, given there are no Product Environmental Footprint Category Rules for the sector analysed (European Commission, 2021b). Moreover, except for the number of daily meals and the location, all data were taken from literature and modelled through Ecoinvent processes. Since the overarching aim of this study was to show how a LCA can be contextualised and not the accuracy of the product-related material flows themselves, reverting to secondary literature for data collection was deemed appropriate. The modelling of the processes can be accessed in the Supplementary Material.

3.3 | Life cycle inventory

In this subsection, the reference flows of the different life cycle phases depicted in Figure 1 are described in more detail. The tables in the following sub-sections contain the necessary information to model the material and energy flows in the LCA software.

3.3.1 | Production of tableware systems

The disposable tableware is assumed to be produced in the Emilia-Romagna region, Italy, whereas the ceramic plates, cutlery and glasses are assumed to be produced in Shenzen, China (Martin et al., 2018). Therefore, transport on a transatlantic ship (14,618 km) to the port of Genoa and road transport to the canteen location via truck (295 km) were calculated through Searoutes (2021) and included in the life cycle inventory. Based on data from Fieschi and Pretato (2018), the materials of the single-use plates, cutlery and cups are modelled as compostable bioplastics (PLA and Mater-Bi®) and virgin cardboard with a Mater-Bi® film, making up 5% of the mass (Vercalsteren et al., 2010), for hot beverages, given that these materials are currently the most commonly used in Italy. The single-use tableware set, characterised in more detail in Table 1, consists of two plates, a cup for cold drinks, a cup for hot drinks, a coffee stirrer, a fork, a spoon, and a knife. It also includes packaging from PLA-film and corrugated board, based on an estimate by Antony and Gensch (2017).

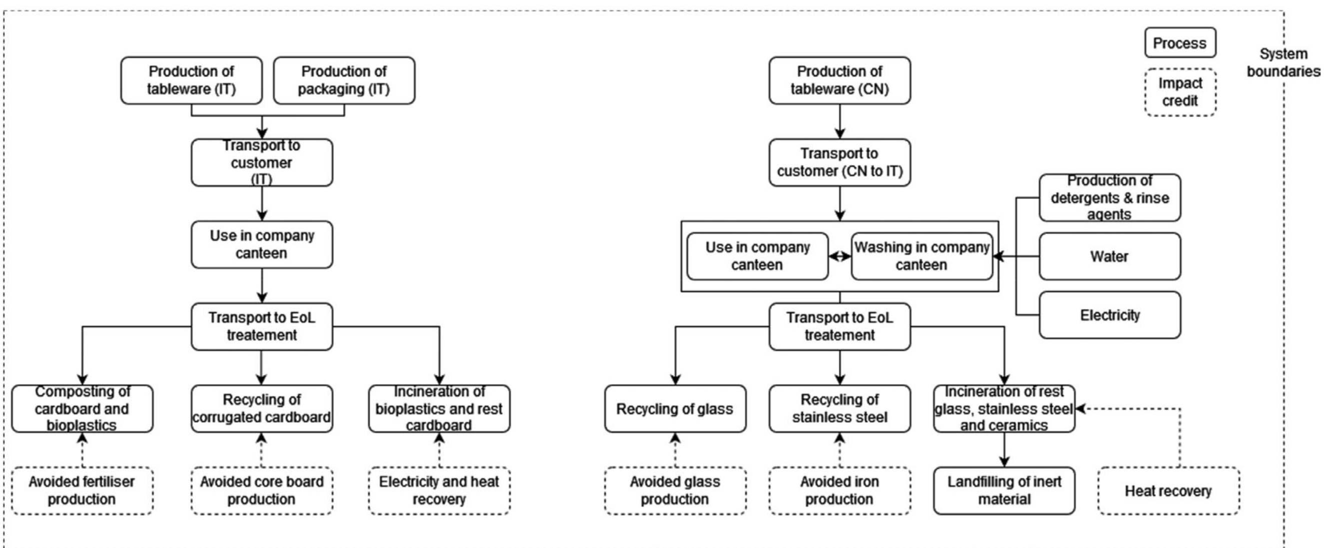


FIGURE 1 Flowchart of single-use system (left) and reuse system (right).

TABLE 1 Material composition of single-use system according to Fieschi and Pretato (2018).

Tableware	Material	Unit mass (g)	Quantity per place setting	Total mass (g)
Plate	PLA	12.9	2	25.8
Knife/fork/spoon	Mater-Bi®	3.8	3	11.4
Tableware envelope	PLA	1.4	1	1.4
Hot beverage cup (100 cL)	Mater-Bi® (5%) + Solid bleached cardboard (SBB)	3	1	3
Coffee stirrer	Mater-Bi®	1	1	1
Cold beverage cup (250 cL)	PLA	4.4	1	4.4
			Total PLA	31.6
			Total Mater-Bi	12.6
			Total SBB	2.8
			Total mass	47.0
Packaging film	PLA	0.5	1	0.5
Cardboard box	Corrugated board	7.0	1	7.0
		Total packaging mass		7.5

TABLE 2 Material composition of reuse-system according to Antony and Gensch (2017).

Tableware	Material	Unit mass (g)	Quantity per place setting	Total mass (g)
Plate	Ceramics	255	2	510
Fork	Stainless steel	22.1	1	22.1
Spoon	Stainless steel	26.3	1	26.3
Knife	Stainless steel	44.4	1	44.4
Espresso cup and saucer	Ceramics	180	1	180
Teaspoon	Stainless steel	13.3	1	13.3
Drinking glass	Glass	240.8	1	240.8
	Total ceramics			690
	Total stainless steel			106.1
	Total glass			240.8
	Total mass			1036.9

Meanwhile, the reusable tableware is assumed to be made from ceramics, glass, and stainless steel, corresponding to the reusable items proposed in the MEIKO study (Antony & Gensch, 2017). One reusable tableware set, presented in Table 2, includes two plates, a glass, an espresso cup with a saucer, a teaspoon, a fork, spoon, and a knife. Given the long lifespan of the reusable tableware, the packaging was not included. Though it is evident that the single-use and the reusable tableware systems have different material properties, these constitute the most frequently used materials used in Italian canteens, while their functional properties are comparable.

3.3.2 | Use phase of tableware systems

The use and EoL phase of the tableware in both systems are located in the Emilia-Romagna region. Whereas the single-use system does not entail additional resource flows during the use phase, the use phase in the reuse system necessitates energy and material flows related to the washing process. The amount of energy, water, detergent, and rinse agents required by the dishwasher (hood machine), representing current best available technology (BAT), are based on studies from Rüdenauer et al. (2011) and Antony and Gensch (2017) and depicted in Table 3. Furthermore, it is assumed that the same volume of water input is then managed as water output through a wastewater treatment process.

3.3.3 | EoL of tableware systems

For the disposable system described in Table 4, it is assumed that of the paper (from the dirty cups), 62% is sent to industrial composting, thus obtaining compost as co-product, and 38% is incinerated, according to the regional environmental agency (ARPAE Emilia-Romagna, 2021). For the composting, the avoided benefits of fertiliser production are included and taken from Bishop et al. (2021a) who analysed bioplastics packaging contaminated with food waste, while carbon

TABLE 3 Inputs for washing tableware in reuse system, according to Rüdenauer et al. (2011) and Antony and Gensch (2017).

Item	Amount per place setting	Unit	Composition	Share (%)
Electricity	40.33	Wh	Italian electricity mix (low voltage)	100
Water	320	mL	Tap water	100
Standard detergent	0.61	g	Potassium tripolyphosphate solution, 50% (mass fraction)	20
			Potassium hydroxide, 50% (mass fraction)	36
			Sodium silicate	23
			Oxidising agent (e.g., sodium perborate)	2
			Deionised water	Rest
Rinse agent	0.06	mL	Acetic acid	5.5
			Alcohol	5.5
			Ethoxylated alcohol	2
			Deionised water	Rest

TABLE 4 EoL scenario for single-use system according to ARPAE Emilia-Romagna (2021) and Moretti et al. (2021).

Material	Share (%)	Waste treatment
Bioplastics	15	Industrial composting
Bioplastics	85	Incineration
Paper	62	Industrial composting
Paper	38	Incineration
Cardboard	66	Recycling
Cardboard	34	Incineration

TABLE 5 EoL scenario for reuse system according to ARPAE Emilia-Romagna (2021).

Material	Share (%)	Waste treatment
Ceramics	100	Incineration
Glass	84	Recycling
Glass	16	Incineration
Stainless steel	63	Recycling
Stainless steel	37	Incineration

sink is assumed to be GWP neutral as per Bishop et al. (2021b). Given that the majority of compost is used as fertilising substitute (Fieschi & Pretato, 2018), other uses such as substitution of peat soil are not considered. For the incineration, the generated heat and electricity are also included in the analysis as avoided products, namely heat created by natural gas and the Italian electricity mix (medium voltage) in line with Razza et al. (2009). In contrast, the packaging cardboard is mostly recycled (66%) and the rest incinerated (34%) (ARPAE Emilia-Romagna, 2021). According to the Ecoinvent processes, it is assumed that the recycled corrugated cardboard replaces core board production in the EU. The items made from PLA and Mater-Bi® are almost all (85%) incinerated even though they are labelled as biodegradable and compostable according to EN 13432. As anticipated in Section 3.1, biopolymers are generally sorted out in composting facilities because they take too long to decompose (Moretti et al., 2021; Utilitalia, 2020). Therefore, it was assumed (in line with Moretti et al., 2021) that only 15% of bioplastics is actually composted.

In the case of the reusable system (Table 5), the ceramic plates are also assumed to be incinerated, and the inert waste disposed in a landfill (Martin et al., 2018). The other reusable items are recycled (glass 84%, metals 63%) or incinerated (glass 16%, metals 37%) (ARPAE Emilia-Romagna, 2021). For both systems, the transport distance of 50 km to the place of disposal, the average determined by the Italian Composting Consortium, is assumed to be the same (Consorzio Italiano Compostatori, 2021).

3.4 | Multi-functionality issues

A system expansion has been applied with regard to the EoL treatment of both systems. Taking into account credits for energy generated through incineration or the avoided production of fertilisers, for example, can be of relevance with regard to the overall results of the comparison. This is particularly true for the case of the single-use system. Within the course of this study, the cut-off plus credit approach (Ekvall et al., 2020) has been employed for all scenarios. It implies that through a system expansion, the avoided impacts of the production of compost, glass, core board, pig iron and the generation of energy are included. This approach was also the most frequently used one in studies comparing petrol-based and bio-based plastics (Bishop et al., 2021a).

3.5 | Sensitivity analyses

To inquire whether certain data and modelling assumptions have significant impacts on the results, several sensitivity analyses were conducted. Each scenario's single score of the EF method was then compared with the basic single-use scenario (S_BAU). The scenarios are described in Table 6.

TABLE 6 Scenarios for primary and sensitivity analysis.

S_BAU	R-1000	S_comp	S_paper	S_paper_comp	R_renew	R-500	R-125
Single-use tableware basic scenario	Reusable tableware reused 1000 times	Single-use tableware with higher share of composting at EoL	Single-use tableware with paper plates and cups	Single-use tableware with paper plates and cups with higher share of composting at EoL	Reusable tableware with renewable energy mix	Reusable tableware only reused 500 times	Reusable tableware only reused 125 times

TABLE 7 Different composition of single-use tableware.

Tableware	Material	Unit mass (g)	Quantity per place setting	Total mass (g)
Paper plates	Mater-Bi® (5%) + SBB	12.9	2	25.8
Paper cold beverage cup (250 cL)	Mater-Bi® (5%) + SBB	7.4	1	7.4
Rest	Same as in Table 2			16.8
			Total mass	50.0
Packaging film	PLA	0.5	1	0.5
Cardboard box	Corrugated board	7.5	1	7.5
		Total packaging mass		8.0

3.5.1 | Different composition of tableware

In the single-use scenario, an analysis with a higher share of paper-based tableware, namely the plates and cold beverage cups, was employed to identify whether this would reduce the impacts. Paper was chosen, as it is amongst the most commonly used material substitutes for plastics (Herberz et al., 2020). The material composition is depicted in Table 7.

3.5.2 | Different EoL scenario for single-use tableware

Several publications set the composting rate of compostable plastics higher than the current assumption in 3.3 (Bishop et al., 2021b; Fieschi & Pretato, 2018; Moretti et al., 2021; Razza & Innocenti, 2012). Therefore, an additional sensitivity analysis was conducted with a higher percentage of composting, namely 50%, as proposed by Moretti et al. (2021) for future scenarios. Given mechanical recycling is not yet established at scale (Maga et al., 2019), this EoL scenario was omitted.

3.5.3 | Lower number of reuses for reusable tableware

It is expected that a high percentage of the reusable system's impact are going to be related to the electricity consumption of the dishwasher, rendering the impacts related to the raw material extraction and processing, as well as the production phase of the tableware less prominent. Therefore, the number of reuses was decreased to evaluate the scenarios of 500 and 125 reuses in comparison with the single-use system.

3.5.4 | Renewable energy for reusable system

In case the company would use renewable energy to power the dishwasher instead of the national energy mix, the environmental impacts are expected to drop even further. Yet, this might involve additional costs for either installing autonomous energy generators (e.g., photovoltaic panels) or purchasing more costly renewable electricity from energy providers. To reduce complexity, this sensitivity analysis was modelled with the certified energy mix from Switzerland, as a similar Italian energy mix was not available.

3.6 | Assumptions and limitations

It is acknowledged that the impact categories used for impact assessment in LCA cannot display all environmental effects caused by the analysed systems (e.g., effects on biodiversity are left out). This is relevant given the potential of (bio)plastic leakage in the terrestrial and marine environments, where they do not biodegrade as quickly as under controlled conditions and can remain especially in marine environments for several years (Chamas et al., 2020; Emadian et al., 2017). The implications of this leakage and slow biodegradation are not well researched to date (Pinto da Costa et al., 2020) and have hence not yet been included in LCIAs.

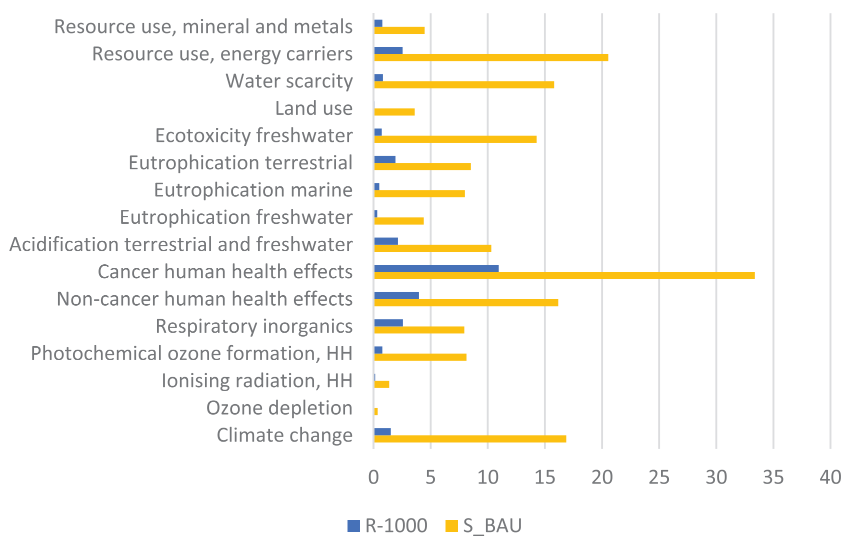
4 | RESULTS

The LCIA characterisation results of the two baseline scenarios (S_BAU and R-1000) reported in Table 8 show that the single-use system has higher environmental impacts across all impact categories.

TABLE 8 Characterisation of LCA results with EF2.0 (adapted) LCIA method.

Impact category	Unit	S_BAU	R-1000
Climate change	kg CO_2 eq	130845.34	11774.81
Ozone depletion	kg CFC11 eq	0.01	0.00
Ionising radiation, HH	kBq U-235 eq	5819.22	606.23
Photochemical ozone formation, HH	kg NMVOC eq	330.26	31.18
Respiratory inorganics	disease inc.	0.01	0.00
Non-cancer human health effects	CTUh	0.01	0.00
Cancer human health effects	CTUh	0.00	0.00
Acidification terrestrial and freshwater	mol H^+ eq	572.53	118.63
Eutrophication freshwater	kg P eq	11.24	0.84
Eutrophication marine	kg N eq	226.39	14.41
Eutrophication terrestrial	mol N eq	1508.86	339.97
Ecotoxicity freshwater	CTUe	168714.71	8509.45
Land use	Pt	4818625.08	125938.01
Water scarcity	m^3 depriv.	181411.45	9443.85
Resource use, energy carriers	MJ	1341533.64	166731.38
Resource use, mineral and metals	kg Sb eq	0.26	0.04

FIGURE 2 Normalised LCIA results of single-use and reuse scenarios.



When normalised (excluding long-term emissions) according to global resource and emission factors from 2010 (Sala et al., 2017), it is again the single-use system with a significantly higher impact across impact categories, as visible in Figure 2. Merely in one of the three toxicity categories, that is, human (cancerous and non-cancerous) toxicity and ecotoxicity of the freshwater, which are not yet to be considered as decisive due to the low robustness of the impact model (Crenna et al., 2019), the impact of the reusable system is noteworthy. This is mainly due to the production of the ceramic plates and the steel cutlery, as can be deduced from the process contribution to the impacts, available in the Supplementary Materials.

The largest process contribution of the single-use system, available in the Supplementary Material, is the production of the PLA pellets with 57% of all impacts, while the Mater-Bi® production accounts for 15% of the impacts, the paper cup production for only about 4% of the impacts and, interestingly, the packaging almost 13% of the impacts. The disposal stage makes up a negligible share of impacts of the full life cycle, but this share is composed of the negative impacts of the waste treatment (15%) and the impact credits of the avoided products and energy generation (−16%).

In contrast, the most impactful life cycle phase for the reuse scenario is the use phase, with 67% of the impacts stemming from electricity and 10% from the production and use of the detergents. Only about 23% concern the raw material extraction and tableware production. This goes into the same direction, but is slightly lower than the values reported in the previous literature review (Walker, 2022), where the washing process accounted for 90% of impacts. Especially the impacts of the detergent are lower than assumed in other studies. The disposal stage also results in a net positive impact, given the recovery of glass and steel through recycling, and of the ferrous ashes

TABLE 9 Percentage difference to single-use scenario (superior performance is indicated in green, inferior in red).

Impact category	Unit	S_BAU	R-1000 (%)	S_comp (%)	S_paper (%)	S_paper_comp (%)	R-renew (%)
Climate change	kg CO_2 eq	130845.34	−91	−9	−40	−43	−98
Ozone depletion	kg CFC11 eq	0.01	−85	12	6	10	−97
Ionising radiation, HH	kBq U-235 eq	5819.22	−90	4	−3	−2	−98
Photochemical ozone formation, HH	kg NMVOC eq	330.26	−91	2	−18	−17	−97
Respiratory inorganics	disease inc.	0.01	−68	6	24	26	−77
Non-cancer human health effects	CTUh	0.01	−75	−2	109	108	−83
Cancer human health effects	CTUh	0.00	−67	−3	−14	−15	−69
Acidification terrestrial and freshwater	mol H^+ eq	572.53	−79	11	−20	−16	−95
Eutrophication freshwater	kg P eq	11.24	−93	1	−49	−49	−95
Eutrophication marine	kg N eq	226.39	−94	1	−41	−41	−97
Eutrophication terrestrial	mol N eq	1508.86	−77	14	−22	−18	−95
Ecotoxicity freshwater	CTUe	168714.71	−95	−9	−47	−50	−96
Land use	Pt	4818625.08	−97	1	117	117	−99
Water scarcity	m^3 depriv.	181411.45	−95	1	−62	−61	−98
Resource use, energy carriers	MJ	1341533.64	−88	9	−13	−11	−97
Resource use, mineral and metals	kg Sb eq	0.26	−83	0	−31	−30	−85

from incineration. Only ceramics cannot not be recycled, which is why this material only has a few, negative environmental impacts.

4.1 | Sensitivity analyses

Table 9 provides an overview of the percentage differences to the single-use baseline scenario (S_BAU) with all sensitivity analyses, except for the reduced number of reuses. The latter can be accessed in the Supplementary Materials.

When changing the EoL of the single-use scenario (S_BAU) to include more composting (S_comp), the differences in impact are minor, but have a positive tendency. This implies that from an environmental point of view, a higher composting rate of 50% might not necessarily be favourable. Especially the categories of ozone depletion, acidification, terrestrial eutrophication and resource use of energy carriers were negatively affected, stemming from the higher volume of biowaste undergoing the composting process.

In the case of the paper-based single-use scenario (S_paper), it is noteworthy that the items replaced with cardboard (i.e., two plates and a cup for cold drinks) now only make up 48% of the total impact, while the packaging accounts for 18% and the cutlery (still in Mater-Bi®) for 31% of the impacts. More information on these relative impacts are visible in Figure 3 and the Supplementary Materials. Another difference is that in this scenario, the disposal stage has an impact of −4%, given the higher amount of waste composted, the energy recovered from paper incinerated, and lower emissions of plastic waste incinerated. When compared to the basic single-use scenario (S_BAU), a significant increase in the non-cancer human health effect and land use can be observed. These are due to the production of the solid bleached board, also affecting the somewhat lower increase visible in the respiratory inorganics category. However, considering all impact categories, the paper-based scenario (S_paper) has lower impacts than the baseline scenario (S_BAU).

A notable impact category diverging from the overall pattern is water scarcity (about −60% between S_BAU and S_paper), where it was found that the production of the PLA pellets has a higher impact than the production of pulp. However, Pauer et al. (2020) point out that Ecoinvent in SimaPro does not employ regionalised datasets with regards to water scarcity, leading to a potential overestimation of this indicator, compared to other databases. Further impact categories in favour of the paper-based single-use scenario (S_paper) are freshwater and marine eutrophication, related to the cultivation of biomass in the basic single-use scenario (S_BAU) (Maga et al., 2019). Importantly, the performance in the climate change category is also in favour of the paper-based scenario. Increasing the share of composting in the paper-based scenario (S_paper_comp) only has minor impacts on the impact categories, following the same pattern as the single-use scenario (S_comp) with a higher share of composting.

When comparing the basic single-use scenario (S_BAU) to reuse scenarios with the number of reuses of 500 and 125, respectively, the absolute impact decreases, while the impact of production increases relatively to the impact of electricity and detergent use. However, the overall impacts of the reuse systems (R-500, R-125) remain below the impact of all single-use scenarios. It is only in the R-125 scenario that the reuse system is less favourable than the single-use systems

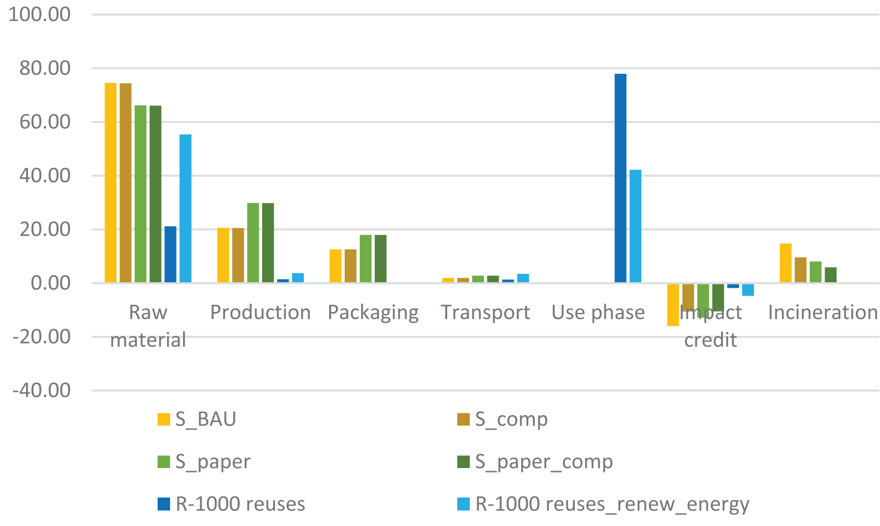


FIGURE 3 Relative contribution (in %) of comprehensive single score impact across scenarios' life cycle phases.

for the impact categories respiratory inorganics and cancer human health effects, related to the production of ceramics.

Finally, for the reuse scenario with renewable energy (R_renew) the impacts decrease even further across all impact categories. When compared to the baseline reuse scenario (R-1000), the relative impacts across the life cycle shift from the use phase to the production phase, as visible in Figure 3.

5 | DISCUSSION

In this case study, it was found that, though both the single-use scenario and the reuse scenario are considered as circular, one of them was overall environmentally favourable. This result is in line with the waste hierarchy, namely that the higher value retention option, reuse, is preferable to the lower one, recovery or composting (Reike et al., 2018). At the same time, it was also observed that the putatively more circular and sustainable EoL option, composting with nutrient recovery, had slightly higher environmental impacts than incineration with energy recovery, which opposes the waste hierarchy. This has also been found by Rossi et al. (2015) and Maga et al. (2019). Yet, if Italy were to have a cleaner energy mix in the future, this evaluation might develop in favour of the composting scenario. To investigate this option further, it is recommended to conduct a consequential LCA employing marginal power generation technologies as performed by Bishop et al. (2021a). It was additionally shown that using paper-based tableware improves the environmental profile of the single-use scenario, though the impact is still higher than the reuse scenario, as anticipated by Herberz et al. (2020). However, it is essential to mention that the results depend on the efficiency of the dishwasher as well as its context. In the case of a non-stationary setting, for instance, in a large event with limited amount of tableware reuses, single-use tableware might be preferable (Fieschi & Pretato, 2018). The same would hold for take-away restaurants which do not have enough space for a dishwasher or do not intend to operate for a long period (e.g., pop-up restaurants). Nevertheless, if applied in coordination within a certain region, reusable food containers as proposed by Baumann et al. (2018) and Bouchet and Boucher (2021) could also be used.

Despite the superior performance of reuse systems in this analysis, the Italian National Association for Catering and various Services (ANGEM) confirms that disposable tableware is still common in Italian canteens and its disposal in the organic waste fraction is recommended (ANGEM, 2020). During the COVID-19 pandemic, Italian school canteens were even encouraged in a safety protocol by the Ministry of Education to use only single-use tableware (preferably compostable) (Ministero dell'Istruzione, 2020), though this recommendation was then retracted in the document updated for 2021 (Ministero dell'Istruzione, 2021).

Contextualising the abovementioned situation in Italy, the following two subsections line out how ANT benefits the embeddedness of LCA. It is applied first, on the level of LCA modelling, informing the EoL scenarios, second on the level of LCA result interpretation and third on the level of the role of LCA itself. A visualisation of the socio-material analysis including its actors is presented in Figure 4.

5.1 | LCA modelling in a socio-material context

The bioplastics sector is of strategic importance to the Italian economy and is said to produce local, compostable and thus circular single-use products. This narrative is put forward by the bioplastics producers, associations, consortia as well as lawmakers, and has been translated into legislation from January 2022 onwards, moving away from the European Guidelines of the SUP. Within this legislation, products made from bioplastics are excluded from the ban under certain conditions, if they are certified as biodegradable and compostable according to the UNI EN 13432 or UNI EN 14995 standards, with minimum percentages of renewable raw materials (Decreto Legislativo n. 196/2021). Both the legislative decree and the EN composting standards can be considered as non-human actors, sending the producers and consumers the sign that they are acting environmentally

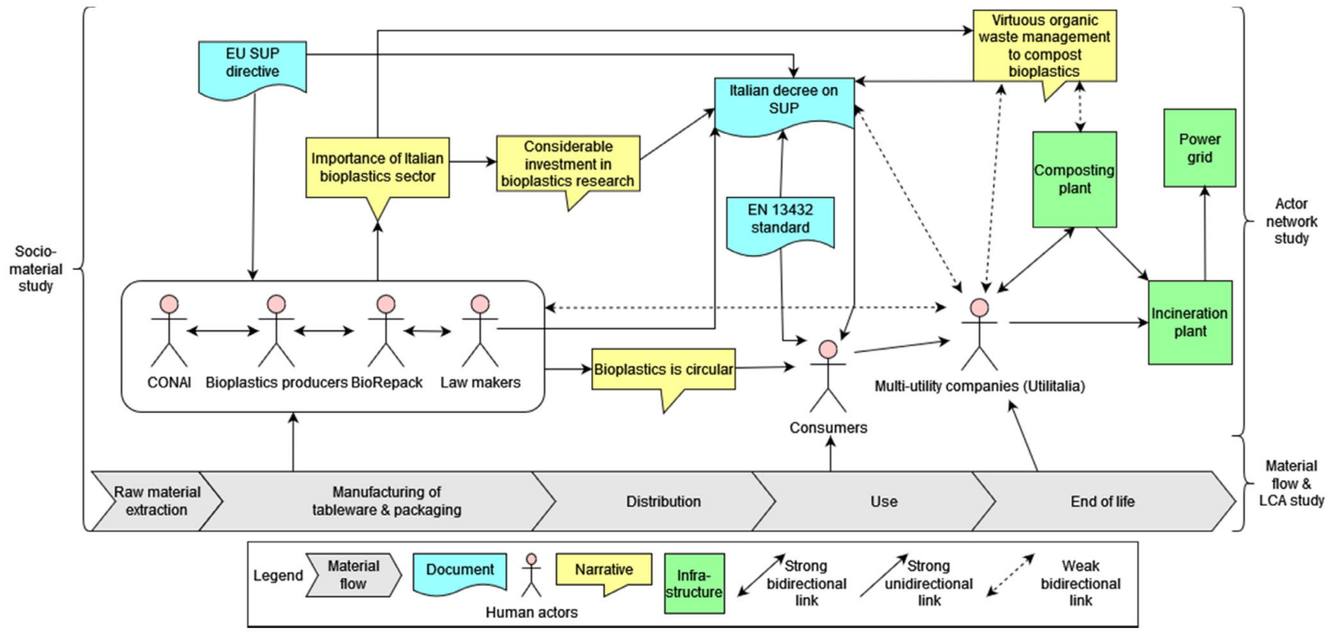


FIGURE 4 Socio-material context of bioplastics in Italy, actors are distinguished into material flows, documents, narratives, human actors, and infrastructure (inspired by Niero et al. (2021) and Baumann and Lindkvist (2022)).

beneficially, when using and composting bioplastics. It thus becomes apparent that composting itself is understood by the bioplastics proponents as a punctualised unit (Law, 1992), meaning that it does not need to be disaggregated into all its internal processes and actors, as long as it fulfils its function (the degradation of bioplastics and recovery of nutrients) adequately. While there are some papers that assume composting rates of up to 100% (Fieschi & Pretato, 2018; Genovesi et al., 2022; Razza et al., 2009; Razza & Innocenti, 2012), these do not seem plausible under current waste management practices (Cristóbal et al., 2023; Kakadellis & Harris, 2020; Rossi et al., 2015). It also needs to be noted that a number of publications assuming high shares of composing were funded by a major Italian bioplastic producer. This information is essential given the role LCAs can take as a 'calculative tool' for advocating for CE solutions, with the power to convince other actors of the superiority of a potential strategy based on its assessed impact (Niero et al., 2021, p. 6). A rigorously conducted LCA induces trust in the results and in the process of translation from one actor to the next, a potentially political and performative nature is conferred upon it (Law, 2008). Utilitalia (2020), representing the companies actually composting the biopolymers, have pointed out that there are problems within this punctualised process: first, the inability of current machinery to distinguish between fossil-based and compostable plastics, and second, the longer degradation periods of bioplastics compared to the rest of the organic waste fraction. Whereas composting might be a more viable option of EoL management in the future (Moretti et al., 2021), current waste management practices and infrastructure are not designed to compost bioplastics, which is why it is mainly incinerated. Therefore, in practice, the supply chain of bioplastics has a lower level of circularity than is generally communicated, namely, renewable inputs and the energy recovery from incineration, instead of nutrient recovery from composting. Furthermore, even if future improvements in collection and sorting efficiency might optimise composting rates up to 50%, as analysed in two scenarios, the comparative LCA of this case study has shown that the reuse system still remains favourable from an environmental point of view.

Moreover, research has found that mechanical recycling of bioplastics would be environmentally favourable to composting and incineration, as it can help replace virgin material production (Cristóbal et al., 2023; Hottle et al., 2017; Maga et al., 2019). However, taking a system perspective as advised by Niero et al. (2021) and Baumann and Lindkvist (2022), this would imply an additional recycling stream to be collected and sorted, potentially further confusing consumers (Rosenboom et al., 2022). Already now, the incorrect disposal of compostable and fossil-based plastics impedes the recycling and composting processes, contaminating both material streams (Utilitalia, 2020; WRAP, 2021). Therefore, mechanical recycling, without an intensive informational campaign to consumers, might not actually improve the environmental impact of bio-based plastics when implemented (Cristóbal et al., 2023). This exemplifies the findings by Baumann and Lindkvist (2022) that the optimal scenario is not necessarily the least environmentally impactful one from a technological possibility perspective, but rather the least impactful one within the social context it is embedded in.

5.2 | LCA result interpretation in a socio-material context

These considerations highlighted through an ANT lens show that making a legal exception for the production of single-use bioplastics, based on the argument of a virtuous collection of organic waste is not necessarily valid. After all, bioplastics products are more frequently incinerated than composted, meaning they do not recover nutrients, as advertised by producers, but energy. Furthermore, incineration (with the current Italian energy mix) has in several instances shown to be the preferable EoL option from an environmental point of view and hence it is questionable whether composting should actually be aimed for, as is proposed by bioplastics proponents.

In contrast to the argument of environmental benefits, the socio-economic argument for continuing the production of bioplastics banned in other EU countries might hold more merit. Taking into consideration the larger context of the connected industries, trade-offs between the environmental and social issues, as well as economic prosperity should be made more transparent (Babri et al., 2018; Hahn et al., 2010). From an environmental perspective, the LCA of this hypothetical case study has shown that for canteens in Northern Italy using a similar BAT dishwasher, reusable tableware would be the preferable choice. However, the social-economic dimension of this choice is yet to be analysed and might bear the answers on why the reusable tableware is not yet set as an industry standard (ANGEM, 2020). Indeed, the Italian exception is potentially better founded on the argument of local employment, and long-term investment in this sector. Therefore, it is important to understand how the current employees in this sector would be affected, if the SUP directive were to be translated as it was intended from a EU perspective, favouring reuse systems (Copello et al., 2021). Not naming this issue and putting forward claims of environmental superiority prevents effective decision-making for consumers and policy makers. It is still up to these stakeholder groups to weigh the socio-economic aspects higher than the environmental ones, but, in the name of transparency, this choice should be made on a basis of sound scenario assumptions.

5.3 | Contribution to theory and practice

The design and interpretation of this case study were substantially informed by a preceding analysis of the social context that it is embedded in. Analysing this context with ANT helped to better understand how issues are framed by actors, which can be both human and non-human (Niero et al., 2021). While the considerations that this socio-material perspective elicits could be considered as implicit, it is essential to be explicit about the contextual assumptions. This is particularly relevant, given the multitude of studies still modelling their systems based on technical feasibility rather than the social realities in which the technologies are embedded. With ANT, it was possible to increase the accuracy of the EoL modelling (Bishop et al., 2021a; Niero et al., 2021) by analysing the diverging messages communicated by the bioplastics proponents on the one hand, and the Italian multi-utility companies on the other. In addition to ANT, Niero (2023) proposes the application of Practice Theory, centred on how a product is being used, which could provide additional insights in terms of actions preceding recycling. To create a mapping of actor relations, stakeholder involvement plays an essential role and has been advocated for by many scholars (Roos Lindgreen et al., 2022; Sala et al., 2013; Troullaki et al., 2021). This study has provided a proposition of how such an integrated mapping could be visualised (Figure 4). By better understanding the power relations between the actors and the current waste management practices, it is therefore possible to scope LCAs closer to reality and to obtain information that can be more effectively used to support sustainable production and consumption. Moreover, this case study also showed that the wider argument in favour of bioplastics was framed as an environmental issue, to be answered with a calculative device such as the LCA, while it should also be addressed from a socio-economic perspective. Input–output analyses could be used to determine the sectoral shifts in terms of labour and value added, in case the single-use of bio-based plastic tableware were to be banned. Naturally, it is also possible to use other mass balance or life cycle based assessment approaches, such as a material flow analysis (Niero, 2023), life cycle costing, social LCA or their combination, life cycle sustainability assessment, in connection with a socio-material perspective.

It is not expected that a company, looking for the most sustainable option to serve its meals, conduct a full ANT analysis of its context. However, practitioners should try to identify, if the CE strategy they are choosing is suitable within their respective environment and should refrain from simply applying a CE strategy because it is circular and thus potentially sustainable (Walker et al., 2021). Hence, a sustainability assessment of the proposed CE practices should be conducted, including the stakeholders along the life cycle of the product (system) in question to verify the modelling assumptions (Roos Lindgreen et al., 2022). The bioeconomy, and with it the replacement of fossil fuels with bio-based resources, has now become more widespread and is actively encouraged by EU policies (Camilleri, 2020) in, for example, the energy, packaging and textile sectors (Biber-Freudenberger et al., 2020; Friedrich, 2021). However, practitioners are invited to aim beyond incremental changes, that is, merely exchanging a material from fossil-based to bio-based plastics. Instead, they should actively inquire about possibilities to employ a higher value retention option, namely the reuse of tableware from other materials, even though that might entail changes in the user behaviour. In the past, several types of reuse systems had been well established, before single-use plastics became a prominent choice (Geyer, 2020). It would be interesting to use ANT to analyse how this change had taken place, but this is left to future research. Finally, it is essential to understand the limitations of LCA as a calculative device and decision-making tool, given it can only assess parts (i.e., environmental aspects, excluding biodiversity and marine plastic pollution) of the systems within its scope (Niero et al., 2021). Besides the limited range of LCA results, they will be employed by actors that use both their rationality but also intuition and organisational preferences to take a decision (Pryshlakivsky & Searcy, 2021). Thus, it needs to be acknowledged that LCAs and their results can be of political and performative nature, even though this normative perspective is not inherent in the academic field LCA originated from (Boons & Roome, 2000). In line therewith, both academics and practitioners are encouraged to view LCA not necessarily as a neutral methodology, but as an assessment approach that, at least in part, mirrors the world view of LCA practitioners in the choices of how the LCA is modelled (Freidberg, 2018; Gasparatos, 2010). As the LCA community is getting into the spotlight given the increasing demand for LCAs in the public and private sector, the transparency on data, modelling assumptions, but also affiliations and funding is becoming more important to ensure the continuous integrity of LCA in practice. In this way, the LCA can fulfil its function as a non-human actor inducing trust in its results and thus constituting a sound contribution to decision-making for more sustainable product systems and consumption choices. As a potential avenue of future research, it would be interesting to further analyse the current use of this calculative device by conducting a discourse analysis of LCA in academic and grey CE literature, as was done for the plastics sector by Calisto Friant et al. (2022).

6 | CONCLUSION

When applied transparently and conscientiously, LCA is one of the most effective environmental assessment approaches for products, applied by the public and private sector alike. This article analyses a hypothetical case study of a stationary canteen in Northern Italy through a comparative LCA of single-use bio-plastics and reusable tableware. The LCA was complemented with a socio-material perspective, in order to make implicit contextual factors potentially relevant to the LCA modelling and result interpretation more explicit. Applying an ANT lens, it was found that, though the general discourse made it seem as if composting were the practiced and preferred EoL option for bioplastics, incineration with energy recovery was actually more common and environmentally favourable. The LCA results also showed that the reuse scenario was the environmentally superior choice, even when substantially reducing the assumed number of reuses, as emerged from the sensitivity analyses. The most impactful life cycle stage in the single-use scenario was the cultivation of the biomass, while in the reuse scenario it was the use phase, mainly driven by energy use. Should it not be possible to install a dishwasher, using paper-based tableware would be a preferable option for the single-use scenario. Given the findings of the contextualised LCA, the argument that bioplastics is a circular, environmentally sustainable option might fall short to substantiate the Italian legislative exception of the SUP ban for bioplastics. However, it may be possible for bioplastic proponents to put forward socio-economic arguments in favour of their products, though the impacts of the ban on socio-economic aspects have not yet been assessed. In this analysis, ANT has therefore helped to improve the LCA modelling, but also supported the contextualisation of the LCA's results in the framing of the bioplastics discourse. Furthermore, it has shown the role of the LCA as a calculative tool, the results of which can potentially be used by actors to advocate for their preferred CE solution, thus rendering the assessment performative.

The authors acknowledge that the results of this hypothetical case study are limited in their generalisability beyond Northern Italy and might depend on additional factors such as personnel costs for dishwashing services and limited space in the kitchen. Furthermore, the data for this study were mostly taken from literature and might thus benefit from improvement in terms of detail, which could be achieved through interviews with the respective actors and data collection on site. It is also not the intention to refute the Italian bioplastics exception from the SUP directive based on a single case study. However, this article has shown that for companies that would like to apply a CE practice in their canteen that yields high environmental performance, single-use tableware from bioplastics is not necessarily the preferable choice. Moreover, this study advocates for companies to look beyond sustainability claims of powerful actors which have entrenched their discourse in legislation and standards, and to apply LCAs with an awareness of the methodology's limitations. This is especially relevant given the increasing importance of LCA as a calculative tool proposed by EU policies to verify sustainability claims of companies. LCA has rightly earned its place amongst the most rigorous methodologies available to evaluate environmental sustainability aspects in a CE context. Yet, to ensure the continued transparency of underlying assumptions, validity of results, and thus effectiveness in furthering sustainable development, it is crucial to also pay attention to how it is applied and talked about by practitioners.

ACKNOWLEDGMENTS

This article is part of the research project CRESTING (Circular Economy: Sustainability implications and guiding progress), funded by the European Union's Horizon 2020 research and innovation Marie Skłodowska-Curie grant agreement number 765198. The authors would like to thank Benedetta Bellotti, Roberto Buonamici, Gioia Garavini, Alessandra Zamagni and Paolo Masoni for their market data insights and support, as well as Ioannis Arzoumanidis for his helpful comments on methodological considerations regarding the LCA modelling.

ORCID

Anna M. Walker https://orcid.org/0000-0001-7420-4864
Alberto Simboli https://orcid.org/0000-0001-8753-3974
Walter J. V. Vermeulen https://orcid.org/0000-0002-5947-8688

ENDNOTE

[1] For the remaining article, the term bioplastics is intended to cover only plastics that is both bio-based and compostable as per the EN 13432 standard. The authors are aware that bioplastics could be perceived as too generic of a term, since not all types of bio-based plastics are necessarily compostable or biodegradable, nor is biodegradability limited to bio-based plastics (Rosenboom et al., 2022; Vinci et al., 2021).

REFERENCES

ANGEM. (2020). *14a Commissione Politiche dell'Unione Europea, Senato della Repubblica Legge di delegazione europea 2019 n. 1721*. Osservazioni del settore della Ristorazione Collettiva Senato della Repubblica. https://www.senato.it/leg/18/BGT/Schede/Ddliter/documenti/52774_documenti.htm

Antony, F., & Gensch, C.-O. (2017). *Life cycle comparison of reusable and non-reusable crockery for mass catering in the USA (Commissioned by MEIKO Maschinenbau GmbH & Co. KG)*. Oeko-Institut e.V.

Arcelli, P. (2020). La filiera dei polimeri compostabili Dati 2019 e prospettive. Plastic consult.

Arcelli, P. (2021). La filiera dei polimeri compostabili Dati 2020 e prospettive. Plastic consult.

Arpae Emilia-Romagna. (2021). *La gestione dei rifiuti in Emilia-Romagna REPORT 2020* (p. 182). Regione Emilia-Romagna.

Attuazione della direttiva (UE). (2019/904). del Parlamento europeo e del Consiglio del 5 giugno 2019 sulla riduzione dell'incidenza di determinati prodotti di plastica sull'ambiente, Gazzetta Ufficiale della Repubblica Italiana, no. 285, pp. 6–36, (GU n.285 del 30-11-2021-Suppl. Ordinario n. 41), President (2021).

Azzurro, P. (2021). Dalla riduzione del monouso in plastica alla riduzione del monouso: Indicazioni per il recepimento della direttiva SUP in Italia (p. 56). Greenpeace. https://www.greenpeace.org/italy/rapporto/13523/dalla-riduzione-del-monouso-in-plastica-alla-riduzione-del-monouso-indicazioni-per-il-recepimento-della-direttiva-sup-in-italia/

Babri, M., Corvellec, H., & Stål, H. (2018). Power in the development of circular business models: An actor network theory approach. Corporate Responsibility Research Conference 2018, 10–12 September, 2018, Devonshire Hall, Leeds, UK. http://urn.kb.se/resolve?urn=urn:nbn:se:umu:diva-151918

Baitz, M., Albrecht, S., Brauner, E., Broadbent, C., Castellan, G., Conrath, P., Fava, J., Finkbeiner, M., Fischer, M., Fullana, I., Palmer, P., Krinke, S., Leroy, C., Loebel, O., McKeown, P., Mersiowsky, I., Möginger, B., Pfaadt, M., Rebitzer, G., … Tikana, L. (2013). LCA's theory and practice: Like ebony and ivory living in perfect harmony? *The International Journal of Life Cycle Assessment*, *18*(1), 5–13. https://doi.org/10.1007/s11367-012-0476-x

Baumann, C., Behrisch, J., Brennan, T., Giurco, D., & Moy, C. (2018). *FEASIBILITY STUDY: Reusable food containers for takeaway food in the Sydney CBD*. Institute for Sustainable Futures, University of Technology Sydney.

Baumann, H., & Lindkvist, M. (2021). A sociomaterial conceptualization of flows in industrial ecology. *Journal of Industrial Ecology*, *26*, 655–666. https://doi.org/10.1111/jiec.13212

Biber-Freudenberger, L., Ergeneman, C., Förster, J. J., Dietz, T., & Börner, J. (2020). Bioeconomy futures: Expectation patterns of scientists and practitioners on the sustainability of bio-based transformation. *Sustainable Development*, *28*(5), 1220–1235. https://doi.org/10.1002/sd.2072

Bishop, G., Styles, D., & Lens, P. N. L. (2021a). Environmental performance comparison of bioplastics and petrochemical plastics: A review of life cycle assessment (LCA) methodological decisions. *Resources, Conservation and Recycling*, *168*, 105451. https://doi.org/10.1016/j.resconrec.2021.105451

Bishop, G., Styles, D., & Lens, P. N. L. (2021b). Environmental performance of bioplastic packaging on fresh food produce: A consequential life cycle assessment. *Journal of Cleaner Production*, *317*, 128377. https://doi.org/10.1016/j.jclepro.2021.128377

Boons, F., & Roome, N. (2000). Industrial ecology as a cultural phenomenon: On objectivity as a normative position. *Journal of Industrial Ecology*, *4*(2), 49–54. https://doi.org/10.1162/108819800569799

Bouchet, A., & Boucher, J. (2021). ReCIRCLE Lebenszyklus und Kreislaufanalyse. e.a-earth.

Calisto Friant, M., Lakerveld, D., Vermeulen, W. J. V., & Salomone, R. (2022). Transition to a sustainable circular plastics economy in The Netherlands: Discourse and policy analysis. *Sustainability*, *14*, 190. https://doi.org/10.3390/su14010190

Camilleri, M. A. (2020). European environment policy for the circular economy: Implications for business and industry stakeholders. *Sustainable Development*, *28*(6), 1804–1812. https://doi.org/10.1002/sd.2113

Chamas, A., Moon, H., Zheng, J., Qiu, Y., Tabassum, T., Jang, J. H., Abu-Omar, M., Scott, S. L., & Suh, S. (2020). Degradation rates of plastics in the environment. *ACS Sustainable Chemistry & Engineering*, *8*(9), 3494–3511. https://doi.org/10.1021/acssuschemeng.9b06635

Coelho, P. M., Corona, B., ten Klooster, R., & Worrell, E. (2020). Sustainability of reusable packaging–Current situation and trends. *Resources, Conservation & Recycling: X*, *6*, 100037. https://doi.org/10.1016/j.rcrx.2020.100037

Coelho, P. M., Corona, B., & Worrell, E. (2020). Reusable vs. Single-Use Packaging. Reloop & Zero Waste Europe.

Collis, B., Baxter, W., Baird, H. M., Meade, K., & Webb, T. L. (2023). Signs of use present a barrier to reusable packaging systems for takeaway food. *Sustainability*, *15*(11), 8857. https://doi.org/10.3390/su15118857

Consorzio Italiano Compostatori. (2021). CO2mpost v. 1.0—Tool online per la valutazione della CO2 prodotta ed evitata per le attività di compostaggio e digestione anaerobica. Manuale utente. Consorzio Italiano Compostatori.

Copello, L., Haut, G., Maillot, J., & Mongodin, F. (2021). Moving on from single-use plastics: How is Europe doing? Rethink Plastic Alliance.

Cosate de Andrade, M. F., Souza, P. M. S., Cavalett, O., & Morales, A. R. (2016). Life cycle assessment of poly(lactic acid) (PLA): Comparison between chemical recycling, mechanical recycling and composting. *Journal of Polymers and the Environment*, *24*(4), 372–384. https://doi.org/10.1007/s10924-016-0787-2

Crenna, E., Secchi, M., Benini, L., & Sala, S. (2019). Global environmental impacts: Data sources and methodological choices for calculating normalization factors for LCA. *The International Journal of Life Cycle Assessment*, *24*(10), 1851–1877. https://doi.org/10.1007/s11367-019-01604-y

Cristóbal, J., Federica Albizzati, P., Giavini, M., Caro, D., Manfredi, S., & Tonini, D. (2023). Management practices for compostable plastic packaging waste: Impacts, challenges and recommendations. *Waste Management*, *170*, 166–176. https://doi.org/10.1016/j.wasman.2023.08.010

Dorn, M., & Stöckli, S. (2018). Social influence fosters the use of a reusable takeaway box. *Waste Management*, *79*, 296–301. https://doi.org/10.1016/j.wasman.2018.07.027

Ekvall, T., Björklund, A., Sandin, G., & Lage, J. (2020). Modeling recycling in life cycle assessment (C551; RE:SOURCE, p. 138). Swedish Environmental Research Institute. https://www.ivl.se/english/ivl/publications/publications/modeling-recycling-in-life-cycle-assessment.html

Elliott, T., Gillie, H., & Thomson, A. (2020). Chapter 24—European Union's plastic strategy and an impact assessment of the proposed directive on tackling single-use plastics items. In T. M. Letcher (Ed.), *Plastic waste and recycling* (pp. 601–633). Academic Press. https://doi.org/10.1016/B978-0-12-817880-5.00024-4

Emadian, S. M., Onay, T. T., & Demirel, B. (2017). Biodegradation of bioplastics in natural environments. *Waste Management*, *59*, 526–536. https://doi.org/10.1016/j.wasman.2016.10.006

European Commission. (2018). Impact assessment—Reducing marine litter: Action on single use plastics and fishing gear. DG ENV. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A52018SC0254

European Commission. (2021a). Commission guidelines on single-use plastic products in accordance with Directive (EU) 2019/904 of the European Parliament and of the Council of 5 June 2019 on the reduction of the impact of certain plastic products on the environment (Commission Notice C(2021) 3762 final; p. 55). European Union.

European Commission. (2021b). Single market for green products—The product environmental footprint pilots—Environment. https://ec.europa.eu/environment/eussd/smgp/PEFCR_OEFSR_en.htm

European Commission. (2023a). Proposal for a Directive of the European Parliament and of the Council on substantiation and communication of explicit environmental claims (Green Claims Directive), 2023/0085, European Commission (2023). https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:52023PC0166

European Commission. (2023b). Directive (EU) 2022/2464 of the European Parliament and of the Council of 14 December 2022 amending Regulation (EU) No 537/2014, Directive 2004/109/EC, Directive 2006/43/EC and Directive 2013/34/EU, as regards corporate sustainability reporting, 2022/2464, European Commission (2023). https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32022L2464

European Environment Agency. (2021). Impacts of COVID-19 on single-use plastic in Europe's environment—European Environment Agency

[Briefing]. European Environment Agency. https://www.eea.europa.eu/publications/impacts-of-covid-19-on

Fieschi, M., & Pretato, U. (2018). Role of compostable tableware in food service and waste management. A life cycle assessment study. *Waste Management*, *73*, 14–25. https://doi.org/10.1016/j.wasman.2017.11.036

Freidberg, S. (2018). From behind the curtain: Talking about values in LCA. *The International Journal of Life Cycle Assessment*, *23*(7), 1410–1414. https://doi.org/10.1007/s11367-015-0879-6

Friedrich, D. (2021). Benefits from sustainable development using bioplastics: A comparison between the food and fashion industries. *Sustainable Development*, *29*(5), 915–929. https://doi.org/10.1002/sd.2184

Gasparatos, A. (2010). Embedded value systems in sustainability assessment tools and their implications. *Journal of Environmental Management*, *91*(8), 1613–1622. https://doi.org/10.1016/j.jenvman.2010.03.014

Genovesi, A., Aversa, C., Barletta, M., Cappiello, G., & Gisario, A. (2022). Comparative life cycle analysis of disposable and reusable tableware: The role of bioplastics. *Cleaner Engineering and Technology*, *6*, 100419. https://doi.org/10.1016/j.clet.2022.100419

Geyer, R. (2020). A brief history of plastics. In M. Streit-Bianchi, M. Cimadevila, & W. Trettnak (Eds.), *Mare Plasticum—The Plastic Sea: Combatting plastic pollution through science and art* (pp. 31–47). Springer International Publishing. https://doi.org/10.1007/978-3-030-38945-1_2

Hahn, T., Figge, F., Pinkse, J., & Preuss, L. (2010). Trade-offs in corporate sustainability: You can't have your cake and eat it. *Business Strategy and the Environment*, *19*(4), 217–229. https://doi.org/10.1002/bse.674

Helling, R. K. (2017). The role of LCA in sustainable development. In *Encyclopedia of sustainable technologies* (pp. 237–242). Elsevier. https://doi.org/10.1016/B978-0-12-409548-9.10063-6

Herberz, T., Barlow, C. Y., & Finkbeiner, M. (2020). Sustainability assessment of a single-use plastics ban. *Sustainability*, *12*(9), 9. https://doi.org/10.3390/su12093746

Hottle, T. A., Bilec, M. M., & Landis, A. E. (2017). Biopolymer production and end of life comparisons using life cycle assessment. *Resources, Conservation and Recycling*, *122*, 295–306. https://doi.org/10.1016/j.resconrec.2017.03.002

ISO. (2006a). ISO 14040:2006 Environmental management—Life cycle assessment—Principles and framework. ISO. https://www.iso.org/cms/render/live/en/sites/isoorg/contents/data/standard/03/74/37456.html

ISO. (2006b). ISO 14044:2006 Environmental management—Life cycle assessment—Requirements and guidelines. ISO. https://www.iso.org/cms/render/live/en/sites/isoorg/contents/data/standard/03/84/38498.html

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. https://doi.org/10.1016/j.jclepro.2020.122831

Latour, B. (1996). On actor-network theory: A few clarifications. *Soziale Welt*, *47*(4), 369–381.

Latour, B. (2005). *Reassembling the social: An introduction to actor-network-theory*. Oxford University Press.

Law, J. (1992). Notes on the theory of the actor-network: Ordering, strategy, and heterogeneity. *Systems Practice*, *5*(4), 379–393. https://doi.org/10.1007/BF01059830

Law, J. (2008). Actor network theory and material semiotics. In B.S. Turner (Ed.), *The new Blackwell companion to social theory* (pp. 141–158). John Wiley & Sons, Ltd. https://doi.org/10.1002/9781444304992.ch7

Leal Filho, W., Salvia, A. L., Minhas, A., Paço, A., & Dias-Ferreira, C. (2021). The COVID-19 pandemic and single-use plastic waste in households: A preliminary study. *Science of the Total Environment*, *793*, 148571. https://doi.org/10.1016/j.scitotenv.2021.148571

Lewis, Y., Gower, A., & Notten, P. (2021a). Single-use beverage cups and their alternatives: Recommendations from Life Cycle Assessments (Life Cycle Initiative). United Nations Environment Programme (UNEP). https://www.lifecycleinitiative.org/library/single-use-beverage-cups-and-their-alternatives-lca/

Lewis, Y., Gower, A., & Notten, P. (2021b). Single-use plastic tableware and its alternatives: Recommendations from Life Cycle Assessments (Life Cycle Initiative). United Nations Environment Programme (UNEP). https://www.lifecycleinitiative.org/library/single-use-plastic-tableware-and-its-alternatives-recommendations-from-life-cycle-assessments-2/

Ligthart, T. N., & Ansems, A. M. M. (2007). Single use cups or reusable (coffee) drinking systems: An environmental comparison. TNO.

Maga, D., Hiebel, M., & Thonemann, N. (2019). Life cycle assessment of recycling options for polylactic acid. *Resources, Conservation and Recycling*, *149*, 86–96. https://doi.org/10.1016/j.resconrec.2019.05.018

Martin, S., Bunsen, J., & Ciroth, A. (2018). Case Study Ceramic cup vs. Paper cup (OpenLCA). Green Delta.

Miliutenko, S., Sandin, G., & Liptow, C. (2020). Single-use plastic take-away food packaging and its alternatives—Recommendations from Life Cycle Assessments (Life Cycle Initiative). United Nations Environment Programme (UNEP). https://www.lifecycleinitiative.org/library/single-use-plastic-take-away-food-packaging-and-its-alternatives/

Ministero dell'Istruzione. (2020). Protocollo d'intesa per garantire l'avvio dell'anno scolastico nel rispetto delle regole di sicurezza per il contenimento della diffusione di COVID 19 (anno scolastico 2020/2021), 13 (2020). https://www.miur.gov.it/documents/20182/2467413/Protocollo_sicurezza.pdf/

Ministero dell'Istruzione. (2021). Protocollo d'intesa per garantire l'avvio dell'anno scolastico nel rispetto delle regole di sicurezza per il contenimento della diffusione di COVID 19 (anno scolastico 2021/2022), 25 (2021). https://www.miur.gov.it/documents/20182/0/m_pi.AOOGABMI.ATTI+del+Ministro%28R%29.0000021.14-08-2021.pdf/f2defe64-6a94-2676-f652-7c1ba87d304d?version=1.0&t=1629311928566

Moretti, C., Hamelin, L., Jakobsen, L. G., Junginger, M. H., Steingrimsdottir, M. M., Høibye, L., & Shen, L. (2021). Cradle-to-grave life cycle assessment of single-use cups made from PLA, PP and PET. *Resources, Conservation and Recycling*, *169*, 105508. https://doi.org/10.1016/j.resconrec.2021.105508

Niero, M. (2023). Implementation of the European Union's packaging and packaging waste regulation: A decision support framework combining quantitative environmental sustainability assessment methods and socio-technical approaches. *Cleaner Waste Systems*, *6*, 100112. https://doi.org/10.1016/j.clwas.2023.100112

Niero, M., Jensen, C. L., Fratini, C. F., Dorland, J., Jørgensen, M. S., & Georg, S. (2021). Is life cycle assessment enough to address unintended side effects from circular economy initiatives? *Journal of Industrial Ecology*, *25*(5), 1111-1120. https://doi.org/10.1111/jiec.13134

Pauer, E., Wohner, B., & Tacker, M. (2020). The influence of database selection on environmental impact results. Life cycle assessment of packaging using GaBi, Ecoinvent 3.6, and the environmental footprint database. *Sustainability*, *12*(23), 9948. https://doi.org/10.3390/su12239948

Pellicanò, G. (2021). Bianco: "Intesa firmata con Biorepack tassello importante verso economia circolare e sostenibilità". www.anci.it. https://www.anci.it/bianco-intesa-firmata-con-biorepack-tassello-importante-verso-economia-circolare-e-sostenibilita/

Pinto da Costa, J., Rocha-Santos, T., & Duarte, A. C. (2020). The environmental impacts of plastics and micro-plastics use, waste and pollution: EU and national measures (PE 658.279; p. 76). Policy Department for Citizens' Rights and Constitutional Affairs. https://www.europarl.europa.eu/thinktank/en/document.html?reference=IPOL_STU(2020)658279

Pro.mo. 2015. Comparative Life Cycle Assessment (LCA) study of tableware for alimentary use. Unionplast.

Pryshlakivsky, J., & Searcy, C. (2021). Life cycle assessment as a decision-making tool: Practitioner and managerial considerations. *Journal of Cleaner Production*, *309*, 127344. https://doi.org/10.1016/j.jclepro.2021.127344
Ramboll. (2021). *Single-use vs multiple-use using science to challenge the misconceptions*. EPPA—European Paper Packaging Alliance.
Razza, F., Fieschi, M., Innocenti, F. D., & Bastioli, C. (2009). Compostable cutlery and waste management: An LCA approach. *Waste Management*, *29*(4), 1424–1433. https://doi.org/10.1016/j.wasman.2008.08.021
Razza, F., & Innocenti, F. D. (2012). Bioplastics from renewable resources: The benefits of biodegradability. *Asia-Pacific Journal of Chemical Engineering*, *7*(S3), S301–S309. https://doi.org/10.1002/apj.1648
Reike, D., Vermeulen, W. J. V., & Witjes, S. (2018). The circular economy: New or refurbished as CE 3.0?—Exploring controversies in the conceptualization of the circular economy through a focus on history and resource value retention options. *Resources, Conservation and Recycling*, *135*, 246–264. https://doi.org/10.1016/j.resconrec.2017.08.027
Roos Lindgreen, E., Opferkuch, K., Walker, A. M., Salomone, R., Reyes, T., Raggi, A., Simboli, A., Vermeulen, W. J. V., & Caeiro, S. (2022). Exploring assessment practices of companies actively engaged with circular economy. *Business Strategy and the Environment*, *31*(4), 1414-1438. https://doi.org/10.1002/bse.2962
Rosenboom, J.-G., Langer, R., & Traverso, G. (2022). Bioplastics for a circular economy. *Nature Reviews Materials*, *1–21*, 117–137. https://doi.org/10.1038/s41578-021-00407-8
Rossi, V., Cleeve-Edwards, N., Lundquist, L., Schenker, U., Dubois, C., Humbert, S., & Jolliet, O. (2015). Life cycle assessment of end-of-life options for two biodegradable packaging materials: Sound application of the European waste hierarchy. *Journal of Cleaner Production*, *86*, 132–145. https://doi.org/10.1016/j.jclepro.2014.08.049
Rüdenauer, I., Blepp, M., Brommer, E., Gensch, C.-O., Graulich, K., Mudgal, S., Cervantes, R., Faninger, T., Lyons, L., & Seifried, D. (2011). *Preparatory studies for eco-design requirements of energy-using products lot 24: Professional washing machines, dryers and dishwashers [part: Dishwashers task 4: Technical analysis existing products]*. Öko-Institut e.V. Institute for Applied Ecology.
Sala, S., Amadei, A. M., Beylot, A., & Ardente, F. (2021). The evolution of life cycle assessment in European policies over three decades. *The International Journal of Life Cycle Assessment*, *26*(12), 2295–2314. https://doi.org/10.1007/s11367-021-01893-2
Sala, S., Crenna, E., Secchi, M., & Pant, R. (2017).). Global normalisation factors for the environmental footprint and Life Cycle Assessment. Publications Office of the European Union. https://doi.org/10.2760/88930
Sala, S., Farioli, F., & Zamagni, A. (2013). Progress in sustainability science: Lessons learnt from current methodologies for sustainability assessment: Part 1. *International Journal of Life Cycle Assessment*, *18*(9), 1653 – 1672. https://doi.org/10.1007/s11367-012-0508-6
Sanyé-Mengual, E., & Sala, S. (2022). Life cycle assessment support to environmental ambitions of EU policies and the sustainable development goals. *Integrated Environmental Assessment and Management*, *18*, 1221–1232. https://doi.org/10.1002/ieam.4586
Sayes, E. (2014). Actor–Network Theory and methodology: Just what does it mean to say that nonhumans have agency? *Social Studies of Science*, *44*(1), 134–149. https://doi.org/10.1177/0306312713511867
Searoutes. (2021). Sea routing. https://classic.searoutes.com
Steen, J. (2010). Actor-network theory and the dilemma of the resource concept in strategic management. *Scandinavian Journal of Management*, *26*(3), 324–331. https://doi.org/10.1016/j.scaman.2010.05.003
Troullaki, K., Rozakis, S., & Kostakis, V. (2021). Bridging barriers in sustainability research: A review from sustainability science to life cycle sustainability assessment. *Ecological Economics*, *184*, 107007. https://doi.org/10.1016/j.ecolecon.2021.107007
Utilitalia. (2020). Managing and recovering bioplastics [position paper]. Utilitalia. https://www.utilitalia.it/area_ambiente/studi_e_ricerche?92608d3f-3aa0-40c6-832c-ab03badbc05f
Vercalsteren, A., Spirinckx, C., & Geerken, T. (2010). Life cycle assessment and eco-efficiency analysis of drinking cups used at public events. *The International Journal of Life Cycle Assessment*, *15*(2), 221 – 230. https://doi.org/10.1007/s11367-009-0143-z
Vinci, G., Ruggieri, R., Billi, A., Pagnozzi, C., Di Loreto, M. V., & Ruggeri, M. (2021). Sustainable management of organic waste and recycling for bioplastics: A LCA approach for the Italian case study. *Sustainability*, *13*(11), 6385. https://doi.org/10.3390/su13116385
Walker, A. M. (2022). Sustainability assessment in circular inter-firm networks [Doctoral Thesis]. G. d'Annunzio University Pescara.
Walker, A. M., Opferkuch, K., Roos Lindgreen, E., Raggi, A., Simboli, A., Vermeulen, W. J. V., Caeiro, S., & Salomone, R. (2021). What is the relation between circular economy and sustainability? Answers from Frontrunner Companies Engaged with Circular Economy Practices. Circular Economy and Sustainability. https://doi.org/10.1007/s43615-021-00064-7
Watson, A. (2021). Addressing single-use plastic products pollution using a life cycle approach (Life Cycle Initiative). United Nations Environment Programme (UNEP). https://www.lifecycleinitiative.org/library/addressing-single-use-plastic-products-pollution-using-a-life-cycle-approach/
Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., & Weidema, B. (2016). The ecoinvent database version 3 (part I): Overview and methodology. *The International Journal of Life Cycle Assessment*, *21*(9), 1218–1230. https://doi.org/10.1007/s11367-016-1087-8
Woods, L., & Bakshi, B. R. (2014). Reusable vs. disposable cups revisited: Guidance in life cycle comparisons addressing scenario, model, and parameter uncertainties for the US consumer. *The International Journal of Life Cycle Assessment*, *19*(4), 931–940. https://doi.org/10.1007/s11367-013-0697-7
WRAP. (2021). *A guide to compostable plastic packaging (The UK Plastics Pact)*. WRAP.

SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

How to cite this article: Walker, A. M., Simboli, A., Vermeulen, W. J. V., & Raggi, A. (2023). Adopting a socio-material perspective on life cycle assessment: Environmental impacts of circular tableware systems in Italy's bioplastics context. *Sustainable Development*, 1–17. https://doi.org/10.1002/sd.2839