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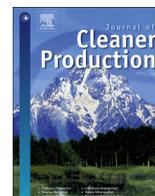
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Life cycle assessment of end-of-life options for two biodegradable packaging materials: sound application of the European waste hierarchy



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ABSTRACT

The purpose of this study is to assess whether for dry biodegradable packaging without food contamination, a detailed life cycle assessment supports the priorities suggested by the five-level hierarchy, as described by the European Waste Directive 2006/12/EC. Environmental impacts and water withdrawal were assessed using an extended version of IMPACT 2002+, accounting for the dynamic pattern of greenhouse gas releases for each scenario when determining Global Warming Potentials for a time horizon of 100 years (in this paper defined as dynamic assessment).

The present assessment shows that, for most impact categories, mechanical recycling is the most interesting option, followed by direct fuel substitution. Intermediate performances are obtained by anaerobic digestion and municipal incineration. Landfill and industrial composting of dry packaging generate the highest environmental impacts of the studied end-of-life options. Indeed, the composting of the studied materials does not substantially improve compost quality and does not enable energy recovery.

The hypothesis that composting is by default environmentally preferable over energy recovery because it is a form of recycling is not confirmed by the present study, thus underlining the importance of a sound and case-specific application of the EU waste hierarchy and the need to complete the hierarchy by product specific studies.

Though of limited effect on the present study, the dynamic assessment of greenhouse gas may moderately decrease the impacts effectively taking place over the 100-year horizon. More important is to consider the degradation patterns of biodegradable materials and present to the decision makers both the 100-year and the long term impacts of the end-of-life options.

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1. Introduction

The waste hierarchy concept as initially introduced by the European Commission in the European Waste Directive 2006/12/EC (The European Parliament and Council, 2006) included the following three parts: 1. Prevention and Reuse, 2. Recycling and Recovery, and 3. Disposal. The three-level hierarchy has been replaced by a five-level hierarchy including: 1. Prevention, 2. Reuse, 3. Recycling, 4. Other recovery, and 5. Disposal, maintained in broad lines in the revised waste framework Directive 2008/98/EC (The

European Parliament and Council, 2008). Recital 31 of this revised Directive¹ clearly supports the understanding that deviations from the hierarchy are acceptable in cases where other priorities would be more environmentally favourable (Manfredi et al., 2011), applying life cycle thinking as suggested by Lazarevic et al. (2012). Flexibility in applying the revised framework is in particular required because of variations in environmental conditions.

¹ 2008/98/EC, Recital 31: "The waste hierarchy generally lays down a priority order of what constitutes the best overall environmental option in waste legislation and policy, while departing from such hierarchy may be necessary for specific waste streams when justified for reasons of, inter alia, technical feasibility, economic viability and environmental protection."

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For new applications, it is therefore important to analyse in further detail which scenarios are preferable. Biodegradable plastics constitute an interesting case since they can be treated in all end-of-life options outlined in the hierarchy, including composting and anaerobic digestion. On the one hand, public perception towards composting and biodegradation has been particularly positive (Bidlingmaier et al., 2003). Biodegradation has frequently been claimed to be a sustainable solution for polymers (Narayan, 1994), mentioned as “the most relevant waste treatment technology for biodegradable plastics” (Ren, 2003) or used as a claim for environmental benefits in itself, without substantiation, as shown by Muse (2010). This founded or reinforced the idea that a biodegradable plastic is an “organic material” in the sense of the Directive 2008/98/EC, hence its composting could be called recycling,² leading to a narrow interpretation of the directive that composting is necessarily environmentally preferable over energy recovery. In addition, public and authority acceptance of energy recovery through incineration is limited in certain regions, and has resulted in local hierarchies of preferences within the “other recovery” category.

On the other hand, Finnveden et al. (2007) conclude that “composting has few advantages over biodigestion and incineration”. Several authors have studied the environmental consequences of a limited number of end-of-life treatments (e.g. Björklund and Finnveden, 2005; Finnveden et al., 2005; Detzel and Krüger, 2006; Eriksson et al., 2005; Hermann et al., 2011) showing that recycling is in general the best end-of-life option and composting the least interesting. A DEFRA (2011) report systematically places landfilling as the least preferable option and suggests that composting is not the preferred option for food, garden waste and lower grade wood, but does not mention biodegradable polymers in this list. Razza and Innocenti (2012) showed that biodegradable wet food packaging can contribute to an increase in the composting rate of food waste and the quality of composts. However the environmental performances of several disposal options need to be explored for the specific case of used dry packaging made of biodegradable plastics.

In addition, the release pattern and time of greenhouse gas emissions (GHG) has traditionally been neglected in LCA (Brandão et al., 2013). However, the dynamic of the CO₂ and CH₄ releases vary widely between biodegradable materials and end-of-life options. For example most of thermoplastic starch (TPS) is biodegraded within a 100 year time horizon in landfill, whereas only a very limited fraction of polylactic acid (PLA) is degraded. While ECJRC-IES (2010) suggests accounting for delayed releases using a fixed reduction of 0.01 year⁻¹, such a linear assumption does not accurately account for the pattern of greenhouse effect over time. Levasseur et al. (2013) show that such temporary storage does matter for biomass. Thus, there is a need to model and take into account the dynamic of greenhouse gas releases specific to each end-of-life treatment of biomaterials.

The objective of this study is to compare in detail the life cycle environmental impacts of six end-of-life options of two biodegradable materials, polylactic acid (PLA) and thermoplastic starch (TPS), used for dry packaging,³ while accounting for the dynamic pattern of greenhouse gas releases for each combination of material and end-of-life treatment. The purpose is to assess whether for

biodegradable dry packaging, a detailed life cycle assessment supports or not the priorities suggested by the five-level hierarchy. The focus of the study is on the end-of-life treatments of dry packaging (as opposed to wet packaging, contaminated with food residues). The focus is not on the packaging materials themselves which can only be studied in the context of a given application.

2. Methodology

2.1. Selected materials and end-of-life options

PLA and TPS were the selected packaging materials. The shape and characteristics of the dry packaging affect the compostability of these materials, hence it is considered that the resulting packaging meet the biodegradability criteria set in prevailing EU standards such as EN 13432 (CEN, 2000). Among the possible treatment alternatives, we selected six end-of-life options that cover the different levels of the EU waste treatment hierarchy. The following alternatives are either currently available for food packaging or may realistically become available on a large scale in the near future, provided that government policies favour such end-of-life options if they are shown to be environmentally preferable: mechanical recycling (MR), industrial composting (IC), anaerobic digestion (AD) (also called methanisation), direct fuel substitution in industrial facilities (DFS), incineration with heat recovery in municipal solid waste incinerators (MSWI) and landfilling (LF).

The **functional unit** is the end-of-life treatment of 1 kg of dry packaging material, as disposed of by a consumer.

2.2. System description

Fig. 1 shows the product system considered: it covers the full packaging life cycle, including primary material production and delivery, transformation into polymer resin as well as end-of-life treatment. The material production is included because the recycling credit considers avoiding primary material. The final product manufacturing, the distribution and the use stages are not considered within the scope of this paper. Hence, forming, labelling and printing are excluded from the study; however, recycling process includes purification of the recovered material. Material and energy recoveries are modelled using a system expansion, substituting background material and energy carriers, as described in the Life cycle inventory section. Europe has been chosen as the region of disposal. These are “average” scenarios which could be

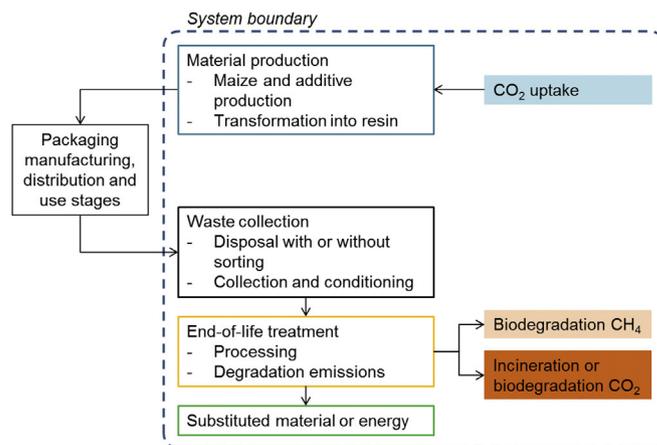


Fig. 1. PLA and TPS life cycle and boundary of the studied system. CO₂ uptake and CO₂ and CH₄ emissions occurring during the end-of-life treatment are highlighted in this system boundary diagram and in the results graphs.

² Directive 2008/98/EC, Art 3, al 17: “recycling” means any recovery operation by which waste materials are reprocessed into products, materials or substances whether for the original or other purposes. It includes the reprocessing of organic material but does not include energy recovery and the reprocessing into materials that are to be used as fuels or for backfilling operations.

³ For instance: packaging for non-food products; packaging for dry food such as crockets, cookies, pasta, chewing-gums; secondary packaging of any product.

representative for Europe but that could vary from region to region and case to case.

2.3. Life cycle inventory

PLA inventory is built from [Vink et al. \(2007, 2010\)](#) and TPS inventory is built from [BioApply \(2013\)](#) data and available literature, as described in [Supporting Information SI 1](#). We developed specific inventory models of PLA and TPS end-of-life treatments for industrial composting, anaerobic digestion as well as for landfilling and incinerations (MSWI and DFS). Greenhouse gas emissions are differentiated between the emissions in the first 100 years that are considered for the 100-year horizon impacts, and the subsequent emissions that are also taken into account in the long term global warming impacts. Mechanical recycling is modelled based on existing inventories for polyethylene (PE). Secondary data are obtained from the ecoinvent 2.2 database ([Frischknecht, 2005; Frischknecht et al., 2007](#); data representative of year 2000) and the full list of reference flows, models and data sources are given in sections [SI 1 to SI 3 of the Supporting Information](#). Since the aim of this paper is to study whether well-developed end-of-life options are environmentally desirable, we assume a) that PLA and TPS would reach the threshold justifying a separate EoL stream as is currently the case for polyethylene terephthalate (PET), b) that the required infrastructure is in place for all end-of-life options studied (including the “integrated framework” described by [Ren, 2003](#)); and c) that the studied materials are accepted by all treatment options.

The main model characteristics are detailed below:

2.3.1. End-of-life treatment processes

Primary data were collected by sending questionnaires or by interviews from three companies, namely Organic Waste Systems ([OWS, 2010](#)) in Belgium and [Valpak \(2010\)](#) in the UK for biodegradation data and [PET-Recycling Schweiz \(2011\)](#) in Switzerland for recycling logistic data.

- **Mechanical recycling (MR):** Mechanical recycling is considered as a possible option for TPS according to [Abbott et al. \(2012\)](#) as well as for PLA since both materials are thermoplastic and can be re-processed using melt processing. We use a typical mechanical recycling processing sequence, including debaling, grinding, washing, drying, extruding and pelletizing, as described by [Franklin Associates \(2010\)](#) for high-density polyethylene (HDPE). The material which is not recycled is incinerated in an MSWI. It is assumed that TPS is able to undergo the mechanical recycling process, but with higher quality loss than PLA, as described in [Table 1](#).

Applying substitution ratios commonly used for current recycled plastics ([Astrup et al., 2009](#)), 1 kg of collected PLA and TPS is considered to substitute 0.83 kg of primary PLA and 0.74 kg of primary TPS resin, respectively. Hygiene constraints related to

recycling into food-contact packaging are not considered, since non-food applications of the recycle are amply available.

- **Industrial composting (IC):** [Table 2](#) presents an overview of the models for industrial composting.

The biodegradation model for average industrial conditions is built based on the extensive literature review done by [Hermann et al. \(2011\)](#).

PLA and TPS do not contain nutrients in larger proportion than traces, but the biodegradation is made possible by the presence of nutrients contained in biowastes, with which they are composted. However, the resulting composted material does not replace N–P–K fertilisers, as these elements have not been brought by PLA or TPS. These materials also do not provide fibrous or porous texture; hence PLA and TPS do not provide soil structure improvement, nutrient retention, erosion and runoff reduction, or herbicide and water requirements reduction, as conventional inputs to composting facilities would ([Pladerer et al., 2008](#)). As a conservative estimate (i.e. favourable to composting), the non-degraded fraction of material is assumed to substitute growing media peat (as modelled in ecoinvent) in system expansion.

- **Anaerobic digestion (AD)** (also called methanisation): for PLA, thermophilic conditions are necessary to initiate biodegradation ([Shi and Palfery, 2010](#)) and dry thermophilic technology is expected to become a standard for energy crops ([De Baere, 2007](#)). The same process has been considered for TPS. [Table 3](#) therefore presents a degradation model for average dry thermophilic conditions, based on ecoinvent data and the extensive literature review by [Hermann et al. \(2011\)](#). As for composting, the biodegradation is made possible by the presence of nutrients contained in biowastes, with which they are digested.

Methane produced is considered to substitute natural gas supply and combustion for heat production. The combined production of electricity and heat is considered in a sensitivity analysis commented in [Table SI6 of the Supporting Information](#). The generated compost (from the digestate) is considered to substitute peat as per the composting scenario.

The digestate is commonly composted after methanisation. Methane emissions for the post-digestion composting strongly depends on how quickly aerobic conditions are restored and anaerobic bacteria are displaced by aerobic activity.

[Edelmann and Schleiss \(1999\)](#) measured rates between 5.1% and 13.5% of carbon emitted as CH₄ in various composting systems combining fresh material and digestate, with a rate of 7.8% for an open air industrial composting windrow. However, [Hermann et al. \(2011\)](#) consider that post-digestion composting operates as fresh material composting and therefore consider in their study only 0.11% of C degrades into CH₄.

Given the large range of potential values and the limited importance of digestate degradation, we have considered an upper range of 10% methane emissions for post-digestion composting. A

Table 1
Overview of the model for mechanical recycling (MR).

	PLA (dry)		TPS (dry)	
	Value	Source	Value	Source
Material loss during process	8%	Assumed similar to HDPE Franklin Associates, 2010	8%	Assumed similar to HDPE Franklin Associates, 2010
Quality loss leading to lower ability of substitution of primary material	10%	Astrup et al., 2009 . Medium loss assumption for future scenario	20%	Astrup et al., 2009 . Assumption for future scenario, considering larger loss than PLA because of solubility
Overall substitution ratio (kg _{primary} /kg _{secondary})	0.83	Calculated	0.74	Calculated

Table 2
Overview of the model for industrial composting (IC).

	PLA (dry)			TPS (dry)		
	Value	C value	Source	Value	C value	Source
Water added	1 kg/kg PLA		Based on Amlinger and Peyr (2003)	1 kg/kg TPS		Based on Amlinger and Peyr (2003)
Initial carbon content		500 g/kg PLA	Stoichiometric PLA		529 g/kg TPS	Stoichiometric starch + PCL
C degraded fraction in composting	80%		89.60% (Valpak, 2010), Between 55% and 95% according to Hermann et al. (2011) . The chosen value is low compared compostability certification but possible in actual operation.	80%		100% (ExcelPlas Australia et al., 2003 ; Klauss, 2004). MaterBi™: between 60% and >90%; starch: between 73% and 97% according to Hermann et al. (2011) . The chosen value is low compared compostability certification but possible in actual operation.
CH ₄ emissions in composting	1.03 g/kg PLA	0.77	0.19% of degraded C into CH ₄ (Hermann et al., 2011)	1.09 g/kg TPS	0.82	0.19% of degraded C into CH ₄ (Hermann et al., 2011)
CO ₂ emissions in composting	1464 g/kg PLA	399	Stoichiometric	1550 g/kg TPS	423	Stoichiometric
Amount of compost material generated	400 g/kg PLA	100	Considering 25% C in wet compost (OWS, 2010), 50% moisture	420 g/kg TPS	105	Considering 25% C in wet compost (OWS, 2010), 50% moisture
CO ₂ emissions from compost in soil over 100 first years	304 g/kg PLA	83	Smith et al., 2001 . This value is highly dependent on local conditions.	320 g/kg TPS	87	Smith et al., 2001 . This value is highly dependent on local conditions.
Long term CO ₂ emissions from compost	62 g/kg PLA	17	Remaining carbon quantity, degraded beyond 100 first years	66 g/kg TPS	18	Remaining carbon quantity, degraded beyond 100 first years

sensitivity study is carried out for a low value of 0.11% degradation as CH₄.

- *Direct fuel substitution in industrial facility (DFS)*: [Table 4](#) presents an overview of the models for municipal incineration and direct fuel substitution. The incineration model from [Doka \(2007\)](#) has been applied to PLA and TPS to calculate direct emissions. Thermal production is considered to substitute heavy fuel oil, as a conservative proxy of typical heavy fossil fuels used in the industry, in a system expansion according to energy content. It is assumed that the two combustion processes (for plastic and heavy fuel oil) have the same combustion and energy recovery efficiencies.
- *Incineration with heat recovery in municipal solid waste incinerator (MSWI)*: The same incineration model is used as for DFS above ([Table 4](#)). Electricity and thermal production are considered to substitute the Union for the Coordination of the Transmission of Electricity (UCTE) electricity mix and heat from natural gas, without importantly modifying the electricity offer in Europe. Net electric and thermal efficiencies for municipal waste-to-energy plants are 14.0% and 41.2% respectively ([CEWEP, 2009](#)). An increase in these yields to 17% and 50% respectively is considered in the sensitivity analysis.
- *Landfilling (LF)*: For landfill, the [Doka \(2007\)](#) model has been applied to PLA and TPS, using the following degraded fractions at year 100: 1% for PLA ([Kolstad et al., 2012](#); [Krüger et al., 2009](#)); 99% for TPS ([Cho et al., 2011](#); [Ishigaki et al., 2004](#)) ([Table 5](#)). To initiate biodegradation, PLA would require hydrolysis via high temperatures that are not present in landfill ([Shi and Palfery, 2010](#)). It is assumed here that the packaging is not exposed to high temperatures during its use stage. The biodegradation of TPS is assumed to be possible by the presence of water and nutrients contained in biowastes or dusts also present in the landfill.

According to [Couturier et al. \(2010\)](#), the landfill gas collection rate was 49% in Europe in 2008 (the rest is emitted to the atmosphere), of which 45% is recovered for energy, most of it with

gas engine to produce electricity. The rest is flared. Therefore, 22% of the generated landfill gas is recovered for energy. Based on Swiss data for the energy conversion efficiency of landfill facilities ([Doka, 2007](#)), 6.1% of the generated landfill gas energy is converted into electricity and 3% into useful heat. Electricity and heat are considered to replace the UCTE electricity mix and natural gas, respectively, in a system expansion. Landfill gas recovery is assumed to take place for 100 years and then stopped for economic reasons.

2.3.2. Dynamic of carbon releases and storage

For greenhouse gas releases, we modelled an exponentially decreasing degradation. In landfill, the degradation constant is determined based on the fraction of material remaining after 100 years. For compost on soil, following industrial composting or anaerobic digestion, the constant is based on the average delay emission (36.3 years according to [Smith et al. \(2001\)](#)). For other scenarios, the emissions are assumed to take place immediately at the time of the end-of-life treatment.

Carbon capture by plants is not dynamically accounted, as it is taking place in a relatively short period (less than a year) in typical crops used for bio-based plastics (e.g. potato, maize).

The credit from landfill gas recovery for the first 100 years is also dynamically assessed since, contrary to the other credits, it is spread over a long period of time.

2.3.3. End-of-life logistics and transport

Three types of logistics are considered, depending on the end-of-life option. They are detailed in [Supporting Information SI 2](#).

1. Dedicated waste collection for **recycling, industrial composting** and **anaerobic digestion**: the existing network used for polyethylene terephthalate (PET) recycling in Switzerland ([PET-Recycling Schweiz, 2011](#)) is used as a model for the edification of dedicated waste collection schemes, adapted to a European context. The Swiss network for PET collection is based on three parallel collection routes (% of collected quantity):
 - a) Collection through grocery stores, used by shoppers (68%),

Table 3
Overview of the model for anaerobic digestion (AD).

	PLA (dry)			TPS (dry)		
	Value	C value	Source	Value	C value	Source
Digestion technology	Dry thermophilic		Condition for optimal degradability (OWS, 2010)	Dry thermophilic		Considered for consistency with PLA
Initial carbon content		500 g/kg PLA	Stoichiometric PLA		529 g/kg TPS	Stoichiometric starch + PCL
C degraded fraction in digester	85.7%		Valpak, 2010 80% according to Hermann et al. (2011)	90%		Almost 100% for starch part (Russo et al., 2009) 70% according to Hermann et al. (2011)
Molar composition of output biogas	60% CH ₄ and 40% CO ₂		Valpak, 2010	60% CH ₄ and 40% CO ₂		Assumption, in line with most anaerobic digestion processes
Amount of CO ₂ produced	627 g/kg PLA	171	Stoichiometric	697 g/kg TPS	190	Stoichiometric
Amount of CH ₄ produced	344 g/kg PLA	258	Stoichiometric	380 g/kg TPS	285	Stoichiometric
CH ₄ emissions in digestion process (leakage)	3.4 g/kg PLA	2.55	Ooms and De Caemel, 2007: 0.025% leakage (p.46); 1% taken in this study as worst case	3.8 g/kg TPS	2.85	Ooms and De Caemel, 2007: 0.025% leakage (p.46); 1% taken in this study as worst case
Amount of energy recovered	17.2 MJ/kg PLA		Form the 99% not leaked. LHV CH ₄ : 50 MJ/kg	19.0 MJ/kg TPS		Form the 99% not leaked. LHV CH ₄ : 50 MJ/kg
Amount of digestate generated	397 g/kg PLA	71.5	Quantis model according to Fuchs et al. (2007) (digestate with 60% water and 45% C in dry mass)	297 g/kg TPS	53.5	Quantis model according to Fuchs et al. (2007) (digestate with 60% water and 45% C in dry mass)
Fraction of C degraded during post-digestion composting	30% of remaining C		OWS, 2010	30% of remaining C		OWS, 2010 and for consistency with PLA
CO ₂ emissions in post-digestion composting	70.8 g/kg PLA	19.3	Stoichiometric	53.0 g/kg TPS	14.5	Stoichiometric
CH ₄ emissions in post-digestion composting	2.86 g/kg PLA	2.15	10% of C degraded into CH ₄ during post-digestion. Upper range between Edelmann and Schleiss (1999) and Hermann et al. (2011)	2.14 g/kg TPS	1.61	10% of C degraded into CH ₄ during post-digestion. Upper range between Edelmann and Schleiss (1999) and Hermann et al. (2011)
Amount of compost material generated	200 g/kg PLA	50.1	Considering 25% C in wet compost (OWS, 2010)	150 g/kg TPS	37.5	Considering 25% C in wet compost (OWS, 2010)
CO ₂ emission from compost in soil over 100 first years	152 g/kg PLA	41.5	Smith et al., 2001	116 g/kg TPS	31.5	Smith et al., 2001
Long term CO ₂ emissions from compost	31 g/kg PLA	8.5	Remaining carbon quantity, degraded beyond 100 first years	24 g/kg TPS	6.5	Remaining carbon quantity, degraded beyond 100 first years

- b) Collection through municipal waste centres (20%), and
c) Local collection points reached by foot, served by small-scale networks using vans (12%).

After compaction, these parallel routes are connected to a larger-scale network of sorting sites, before transport to treatment sites. This architecture is used to model the collection network for recycling, industrial composting and anaerobic digestion up to the sorting site. The existing network used for green waste collection would not be used (de la Harpe, 2011). Transport distances from the

sorting site to the treatment site were evaluated at 200 km for recycling and 50 km for industrial composting and anaerobic digestion (the geographic density of IC and AD plants is higher than for recycling plants (de la Harpe, 2011)). The average European population density is lower than that of Switzerland for which a typical distance is 15–30 km for composting and anaerobic digestion sites.

2. Municipal waste collection for **landfilling** and **municipal incineration**: the ecoinvent municipal waste collection is

Table 4
Overview of the models for municipal incineration (MSWI) and direct fuel substitution (DFS).

	PLA (dry)		TPS (dry)	
	Value	Source	Value	Source
Lower heating value of material	19.5 MJ/kg PLA	Doka, 2007; Glasbrenner, 2009	18.6 MJ/kg TPS	Doka, 2007
Substituted electricity by MSWI production	0.78 kWh/kg PLA	CEWEP, 2009	0.744 kWh/kg TPS	CEWEP, 2009
Substituted heat by MSWI production	8.15 MJ/kg PLA		7.77 MJ/kg TPS	
Substituted heat by DFS production	19.5 MJ/kg PLA	Substitution based on energy content	18.6 MJ/kg TPS	Substitution based on energy content

Table 5
Overview of the model for landfilling (LF).

	PLA (dry)			TPS (dry)		
	Value	C value	Source	Value	C value	Source
Initial carbon content		500 g/kg PLA	Stoichiometric PLA		529 g/kg TPS	Stoichiometric starch + PCL
C degraded fraction in LF within 100 years	1%	5	Kolstad et al., 2012; Krüger et al., 2009.	99%	524	Cho et al., 2011; Ishigaki et al., 2004
CH ₄ generated inside LF over 100 first years (partly captured)	3.76 g/kg PLA	2.82	Couturier et al., 2010; Doka, 2007.	384 g/kg TPS	288	Couturier et al., 2010; Doka, 2007.
CO ₂ generated inside LF over 100 first years	7.98 g/kg PLA	2.18	PLA-specific emissions only	863 g/kg TPS	235	TPS-specific emissions only
CO ₂ emission from combustion of the captured fraction of CH ₄	5.07 g/kg PLA	1.38		518 g/kg TPS	141	
Final CH ₄ emission from LF over 100 first years	1.92 g/kg PLA	1.44		196 g/kg TPS	147	
Global warming score of emissions from LF over 100 first years, if accounted instantly	CO ₂ : 13.1 CH ₄ : 53.3 Total: 66.3 g CO ₂ -eq/kg PLA		Conventional score without dynamic accounting	CO ₂ : 1380 CH ₄ : 5440 Total: 6820 g CO ₂ -eq/kg TPS		Conventional score without dynamic accounting
Captured CH ₄ in LF (partly for energy production)	1.84 g/kg PLA	1.38	Couturier et al., 2010; Doka, 2007	188 g/kg TPS	141	Couturier et al., 2010; Doka, 2007
Substituted electricity by LF production	0.0032 kWh/kg PLA		45% of captured CH ₄ is used to produce energy (rest is flared)	0.33 kWh/kg TPS		45% of captured CH ₄ is used to produce energy (rest is flared)
Substituted heat by LF production	0.011 MJ/kg PLA			1.17 MJ/kg TPS		
Long term CH ₄ emissions (not captured)	373 g/kg PLA	280	Remaining carbon, degraded beyond 100 first years, split in proportion between CO ₂ and CH ₄ before CH ₄ capture, which is assumed to have ceased.	3.88 g/kg TPS	2.91	Remaining carbon, degraded beyond 100 first years, split in proportion between CO ₂ and CH ₄ before CH ₄ capture, which is assumed to have ceased.
Long term CO ₂ emissions	790 g/kg PLA	215		8.72 g/kg TPS	2.38	

applied for urban and suburban areas, corrected for typical distances in Europe (5 km for waste collection⁴). In rural areas, it is estimated that consumers drop off wastes at the municipal waste reception centre by car (4 km, 10% of the trip allocated to disposal of 1 kg of the considered material). Transportation to final disposal plant is made by truck; the distance is 25 km for urban and suburban areas and 50 km for rural areas.

3. Combined collection system for **direct fuel substitution**: this treatment is mainly adopted for refuses from dedicated networks and for some sorted municipal waste collection. It combines part of a dedicated network and simpler logistics. The model is built based on 50% municipal waste collection and 50% dedicated collection, with a distance to treatment site of 200 km.

2.3.4. Substituted materials and electricity

Production data for PLA are from Vink et al. (2007, 2010) and for TPS from BioApply (2013) and the ecoinvent 2.2 database (details in Supporting Information SI 1). The electricity consumed at end-of-life or substituted is taken as the UCTE mix from the ecoinvent 2.2 database.

2.4. Life cycle impact assessment (LCIA) and dynamic assessment of greenhouse gas releases

The IMPACT 2002+ LCIA method (Jolliet et al., 2003) has been chosen to assess the environmental impacts for different end-of-

life options, complemented by water withdrawal and turbined water indicators (Humbert et al., 2012).

For global warming impacts a dynamic 100-year time horizon has been considered, complemented by the assessment of long term impacts using IPCC 2007 GWP₅₀₀. Six different methods are proposed to account for the dynamic greenhouse gas releases in LCA (Brandão and Levasseur, 2011). Among these six methods, the dynamic LCA approach from Levasseur et al. (2010, 2011) is easy to apply to degradation releases, while enabling the differentiation between materials and their degradation rates when analysing the influence of temporary carbon storage. We therefore applied the DynCO₂ tool (Levasseur et al., 2013) to the degradation and end-of-life processes to determine how the greenhouse gas impacts for the 100-year time horizon are modified due to carbon storage for each relevant combination of end-of-life treatment and material (Table 6).

We also applied a conventional GWP₁₀₀ assessment for comparison. For a material such as PLA for which only 1% of the material degrades over 100 years in landfill conditions, the GWP₁₀₀ is reduced to only 0.0057 kg CO₂ dynamic eq/kg CO₂ from embedded carbon, most of the emissions occurring after 100 years. This value represents a reduction of 43% of the impact compared with a pulse initial emission of the amount of CO₂ that is emitted within the first 100 years.

The yearly emission rates for materials in landfill and compost in soil – and the corresponding GWP – are provided in Supporting Information, section SI 3.3.

2.4.1. Illustration of dynamic assessment

Fig. 2 illustrates how the dynamic assessment described by Levasseur et al. (2010, 2011) differs from the conventional

⁴ 3 km in urban areas and 10 km in suburban areas (Doka, 2007), weighted 2/3 and 1/3 respectively.

Table 6
Description of delayed GHG emission accounting: fraction of initial carbon left after 100 years, corresponding degradation rates, average delay or duration of carbon storage and dynamic Global Warming Potential equivalent compared to 1 kg pulse CO₂ emission.

		PLA in landfill	TPS in landfill	Compost in soil	Pulse emission GWP ₁₀₀
Description of the assumed exponentially decreasing degradation curve (e^{-k})					
Fraction left after 100 years	[–]	0.99 ^a	0.01 ^a	0.17	0
Degradation constant k	yr ⁻¹	0.000101	0.0461	0.0179	–
Average delay	yr	50.4	21.2	36.3 ^b	0
For a pool of 1 kg of potential gas produced from degradation of carbon embedded in material					
kg gas released within 100 years per kg potential gas from embedded carbon	kg emitted/kg embedded	0.01	0.99	0.83	1.0
CO ₂ GWP ₁₀₀ : Conventional impact of CO ₂ released within 100 years	kg CO ₂ eq/kg CO ₂ pulse	0.01	0.99	0.83	1
CO ₂ GWP ₁₀₀ : Impact of dynamic CO ₂ release within 100 years per kg CO ₂ from embedded carbon in material ^c	kg CO ₂ dyn eq/kg CO ₂ pulse	0.00569	0.827	0.584	1.0
Ratio of dynamic versus conventional calculation for CO ₂	[–]	56.9%	83.6%	70.1%	100%
CH ₄ GWP ₁₀₀ : Conventional impact of CH ₄ released within 100 years	kg CO ₂ eq/kg CH ₄ pulse	0.276	27.3	22.9	27.55
CH ₄ GWP ₁₀₀ : Impact of dynamic CH ₄ release within 100 years per kg CH ₄ from embedded carbon in material ^c	kg CO ₂ dyn eq/kg CH ₄ pulse	0.235	26.5	21.2	27.55
Ratio of dynamic versus conventional calculation for CH ₄	[–]	85.2%	97.2%	92.6%	100%
For 1 kg of material landfilled or put into soil					
CO ₂ emission per kg of material (released within 100 first years)	kg CO ₂ /kg material	0.0131	1.38	0.764 ^d	1
CH ₄ emission per kg of material (released within 100 first years)	kg CH ₄ /kg material	0.00192	0.196	0	0
CO ₂ GWP ₁₀₀ : Impact of CO ₂ released within 100 first years per kg of material, dynamically assessed ^c	kg CO ₂ dyn eq/kg material	0.0074	1.15	0.536 ^d	1.0
CH ₄ GWP ₁₀₀ : Impact of CH ₄ released within 100 first years per kg of material, dynamically assessed ^c	kg CO ₂ dyn eq/kg material	0.0450	5.25	0 ^d	0
Total global warming score of emissions over 100 first years, dynamically assessed	kg CO ₂ dyn eq/kg material	0.0525	6.41	0.536 ^d	1.0
Conventional total global warming score of delayed emissions over 100 first years	kg CO ₂ eq/kg material	0.0659	6.78	0.764 ^d	1.0
Difference between dynamic and conventional calculation (impact reduction due to delayed emission pushed out of the 100 years time frame)	kg CO ₂ -eq/kg material	–0.0135	–0.375	–0.228 ^d	0

Note: d expressed per kg of compost material (25% C). The amount of compost varies per kg of PLA and TPS and per treatment scenario.

Sources: a Valpak (2010); b Smith et al. (2001); c Levasseur et al. (2013), accounting for the oxidation of CH₄ into CO₂, thus a pulse GWP₁₀₀ of 27.55 kg CO₂ eq/kg CH₄ (often approximated by 25 + 2.75 = 27.75) since the CO₂ sequestration has been accounted for.

assessment, comparing the impact of a pulse emission of GHG at time zero (red shape – conventional assessment) to the impact of the linearly decreasing release of the same quantity of GHG over 100 years (blue shape – dynamic assessment). The red area between 0 and 100 years represents the integrated impact of the pulse emission according to GWP₁₀₀. The blue area between 0 and 100 years represents the dynamically assessed integrated impact of the slow emissions for a 100-year time frame. Within this restricted time frame, the dynamic assessment shows a lower area and thus a lower impact than the conventional assessment, due to the shift of the layers towards later years.

Taking into account a longer time frame (for instance over 500 years) reduces the difference between the two areas and thus between the two integrated impacts. In the long term, both methods tend to the same result.

3. Results

Since the carbon cycle is paramount for carbon-based materials and energy recovery is present in several scenarios, we first focus on global warming and resources depletion. These two categories are presented in details. The full set of IMPACT 2002+ damage indicators and results for GWP₅₀₀ are presented in further detail in the Supporting Information.

3.1. Global warming

This section presents the life cycle scores for global warming with two methods: dynamically assessed with a 100-year horizon

calculated accounting for the effective time of release and “conventionally” assessed with GWP₁₀₀, calculated as if all emissions occurring in the first 100 years were emitted as a pulse emission at first year. The long term global warming impacts, calculated considering emissions of the complete degradation, are presented in Supporting Information, section SI 4.

Production impacts of the resin (cradle-to-gate) are presented separately from the end-of-life impacts to facilitate comparison among end-of-life options and to identify the relative contribution of the end-of-life stage to the total life cycle impacts. CO₂ and CH₄ resulting directly from material degradation are also displayed separately. N₂O is not shown because the studied materials do not contain nitrogen in larger proportion than traces. Negative impacts arise from either carbon uptake during plant growth (material production) or avoided emissions due to end-of-life system expansion (e.g. recycling or energy recovery). The net total results, i.e. the sum of positive and negative emissions, are displayed as white diamonds.

3.1.1. Dynamic assessment of the emissions occurring in the 100 first years

Fig. 3 shows the global warming impacts dynamically assessed over a 100-year horizon. The net total results correspond to the impacts taking effectively place in the first 100 years, after deduction of the treatment credits.

For both materials, mechanical recycling leads to the lowest treatment impacts and a net benefit for the end-of-life. For composting, anaerobic digestion, direct fuel substitution and municipal incineration, the treatment impacts (including impacts

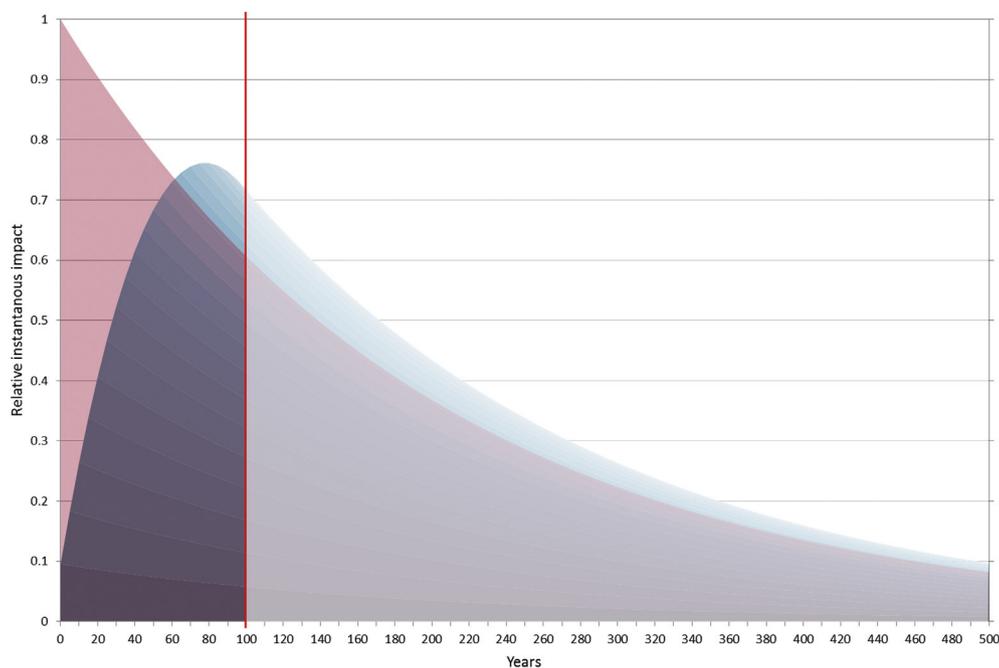


Fig. 2. Illustration of the dynamic assessment with an arbitrary limit of 100 years. Red area: integrated impact of a pulse GHG emission. Blue area: integrated impact of an identical amount of GHG continuously released with a linearly decreasing rate over 100 years. In the blue area, each layer represents the emissions of 5 years. This illustration is simplified on purpose. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

from the process and emissions from the decomposition of the polymeric material, excluding credits) are similar, due to the conversion of the embedded carbon into CO₂ releases. High credits for direct fuel substitution, anaerobic digestion and municipal incineration substantially reduce the net impacts of these options. The credit is however limited for composting due to the shortage of a valuable by-product in term of energy or useful material.

TPS degrades almost completely in a landfill over 100 years, emitting methane and causing the highest global warming net score despite partial recovery of methane to generate electricity and heat (Fig. 3b). Considering the first 100 years, **PLA** degradation in a landfill is limited, leading to a rather advantageous and close-to-zero global warming net total score, similar to direct fuel substitution (Fig. 3a). However, when considering the long term degradation in landfill, this end-of-life option also leads for **PLA** to the highest global warming impacts (section SI 4 of the Supporting Information). Therefore, a 100-year time frame will not fully capture the magnitude of the global warming impact given the behaviour of the material in the landfill because additional emissions can be expected beyond this horizon, as opposed to the **TPS** where nearly all material is degraded during the 100-year period.

The delay in CO₂ and CH₄ emissions also leads to a limited reduction of the impact when dynamically assessed compared to a conventional assessment, because a part of the impacts are postponed beyond the 100-year time frame of the method. As shown in the following section and quantified in the Supporting Information tables SI3 to SI5, the postponed impact is comparable in importance to the treatment credits of **PLA** and **TPS** composting and of **TPS** landfilling.

3.1.2. GWP₁₀₀ conventional assessment of the emissions occurring in the 100 first years

Fig. 4 shows the GWP₁₀₀ global warming impact of the emissions occurring in the 100 first years, hence the same amount of

emissions than in the dynamic assessment, but as if they were emitted as a pulse at first year.

The conventional results are slightly higher than dynamic results for industrial composting, anaerobic digestion and landfill, the three waste management systems that postpone emissions in time. The difference represents 5% of the dynamic impacts (not considering the credits) for IC, around 2% for AD and 13% for **PLA** in LF and 6% for **TPS** in LF.

In the case of these two biodegradable materials, the use of the dynamic assessment has only restricted influence on the final global warming impact at 100 years. It does not influence the observed ranking of the end-of-life scenarios. That ranking is identical for both materials and for all time horizons except for LF; it is the following: MR < DFS < AD < MSWI < IC. LF is similar to DFS for **PLA** over 100 years but the less advantageous scenario for **TPS** (all horizons) and for **PLA** when also accounting for long term emissions occurring after 100 years (Supporting Information, section SI 4).

3.2. Resources depletion

Fig. 5 presents the life cycle scores for resources depletion.

Resources depletion scores are dominated by energy credits for both materials. For the considered substitutions, recycling, direct fuel substitution, anaerobic digestion and municipal incineration lead to the lowest resources depletion scores, with the same ranking as global warming.

For **PLA**, both landfilling and composting show the highest net impacts, close to zero, with composting presenting slightly lower impacts. For **TPS**, landfilling is slightly more advantageous than composting because of landfill gas recovery.

3.3. Midpoint categories

Fig. 6 presents non-weighted scores of the material production and end-of-life for each midpoint category used by IMPACT 2002+.

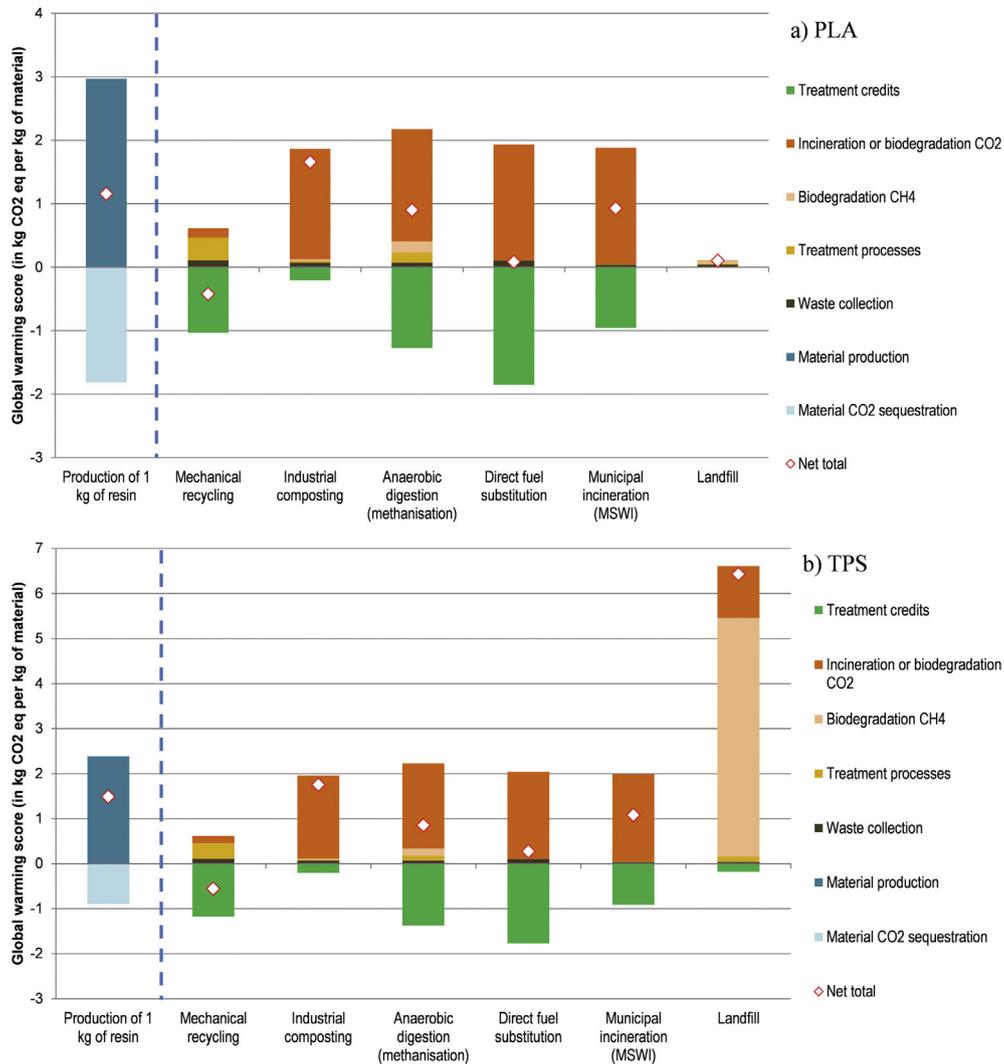


Fig. 3. Comparison of dynamically-assessed global warming impacts over 100 years associated with the six end-of-life treatments of PLA (a) and TPS (b). The bars on the left side present production impacts of the resin (cradle-to-gate) for comparison purposes.

For each category, impacts are scaled against the one with the highest score. Production is included to avoid large negative values scaled against a small positive value, which would make the figure impossible to read; this does not affect the ranking. Negative values appear in the centre of the chart, inside the black circle.

For both PLA and TPS, for most impact categories, results identify mechanical recycling as the least-burdening option, except for respiratory inorganics, ozone layer depletion and aquatic acidification, for which direct fuel substitution reduces the combustion-related impacts. Electricity-related ionizing radiation impacts are the smallest for the municipal incineration that enables substitution of nuclear power in the UCTE mix.

Because the chosen biodegradable materials are mostly agro-based, mechanical recycling of both materials gives a credit to midpoint indicators related to agriculture, particularly for aquatic eutrophication, land occupation, terrestrial ecotoxicity and aquatic ecotoxicity. Indeed, all other end-of-life options imply the loss of the material, while it has been assumed that recycling would avoid the production of the same material, produced the same way.

At the other end of the spectrum, industrial composting and landfilling are the least favourable options for the most impact categories, except for non-carcinogens and carcinogens for which

the incineration scenarios (MSWI and DFS) lead to the highest impacts.

Landfill performs slightly better for TPS than PLA for several categories, because of the energy recovery from landfill gas over a 100 year period.

The global warming results considering long term greenhouse gas emissions include the slow degradation of PLA in landfill. With this long term perspective, landfill becomes the least preferable option for PLA, in contrast with the dynamically assessed impacts over 100 years.

3.4. Other damage categories

Detailed LCA results for human health, ecosystem quality and water withdrawal are presented in the [Supporting Information, section SI 5](#). For these damage categories, the different end-of-life scenarios consistently show that mechanical recycling has the lowest or second lowest impacts because of its relatively important credits. DFS also shows large credits for human health, because heavy fuel combustion emissions, dominated by respiratory inorganics, are avoided in this scenario. Finally, where energy is recovered (DFS, MSWI and, in the case of TPS, LF), there are

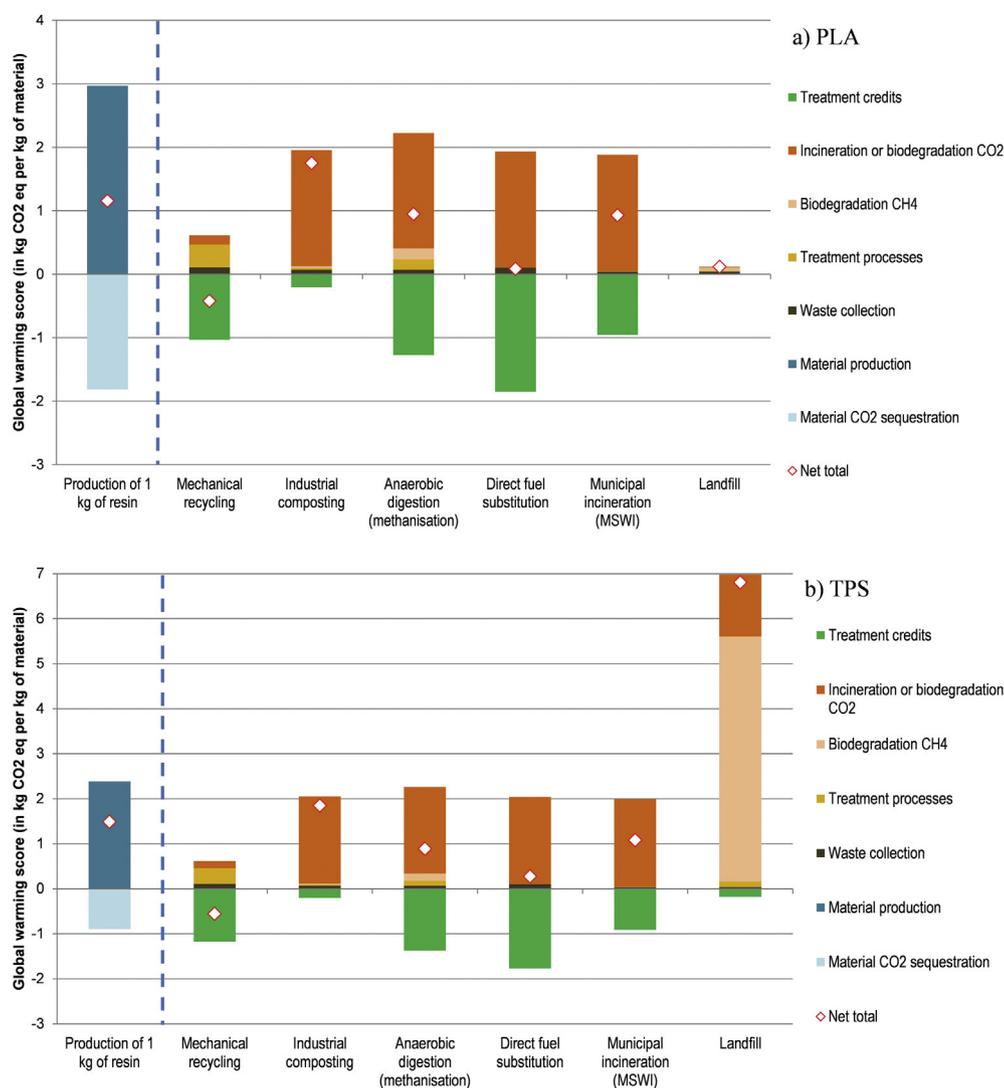


Fig. 4. Comparison of conventionally-assessed global warming impacts calculated with GWP_{100} for the emissions occurring during the first 100 years, associated with the six end-of-life treatments of PLA (a) and TPS (b). The bars on the left side present production impacts of the resin (cradle-to-gate) for comparison purposes.

moderate credits observed for water withdrawal. For other cases, results mostly show a net total impact close to zero.

The damage results confirm the observation at midpoint level: MR and DFS have the lowest impacts and IC and LF the highest, while the ranking is intermediary for AD and MSWI.

3.5. Uncertainty and sensitivity analysis

The results presented are dependent on model parameters, for each of which a qualitative assessment of uncertainty, variability and sensitivity has been performed. The main parameters of influence are:

- The substitution chosen for the recycled material: This choice determines the credits associated with recycling.
- The ability to prevent methane emissions during treatment: This is of particular importance in anaerobic digestion, where methane production is boosted. It depends on equipment as well as management qualities. Sub-optimal operations could have impacts counteracting the benefits of this option.
- The efficiency of energy recovery: The transformation of heat into a valuable form of energy is not always possible and

recovery efficiency may vary from case to case, depending on local heat consumption, equipment choice and conversion efficiency from heat into electricity).

The full list of the parameters assessed is shown in the [Supporting Information, section SI 6](#).

4. Discussion

4.1. Comparison with the EU waste hierarchy

For both materials, the ranking observed never matches the ranking suggested by a narrow interpretation of the EU waste hierarchy, mostly because industrial composting impacts are higher than those of material or energy recovery. For a proper waste hierarchy, the different forms of material recycling should be differentiated according to the value of the recycled product. Mechanical recycling leads to the highest benefits, whereas composting is one of the least favourable end-of-life options. These two options should not be aggregated into a common category of the waste hierarchy. For biodegradable plastics such as TPS and PLA, composting may be rather classified as “other recovery” or as “pre-

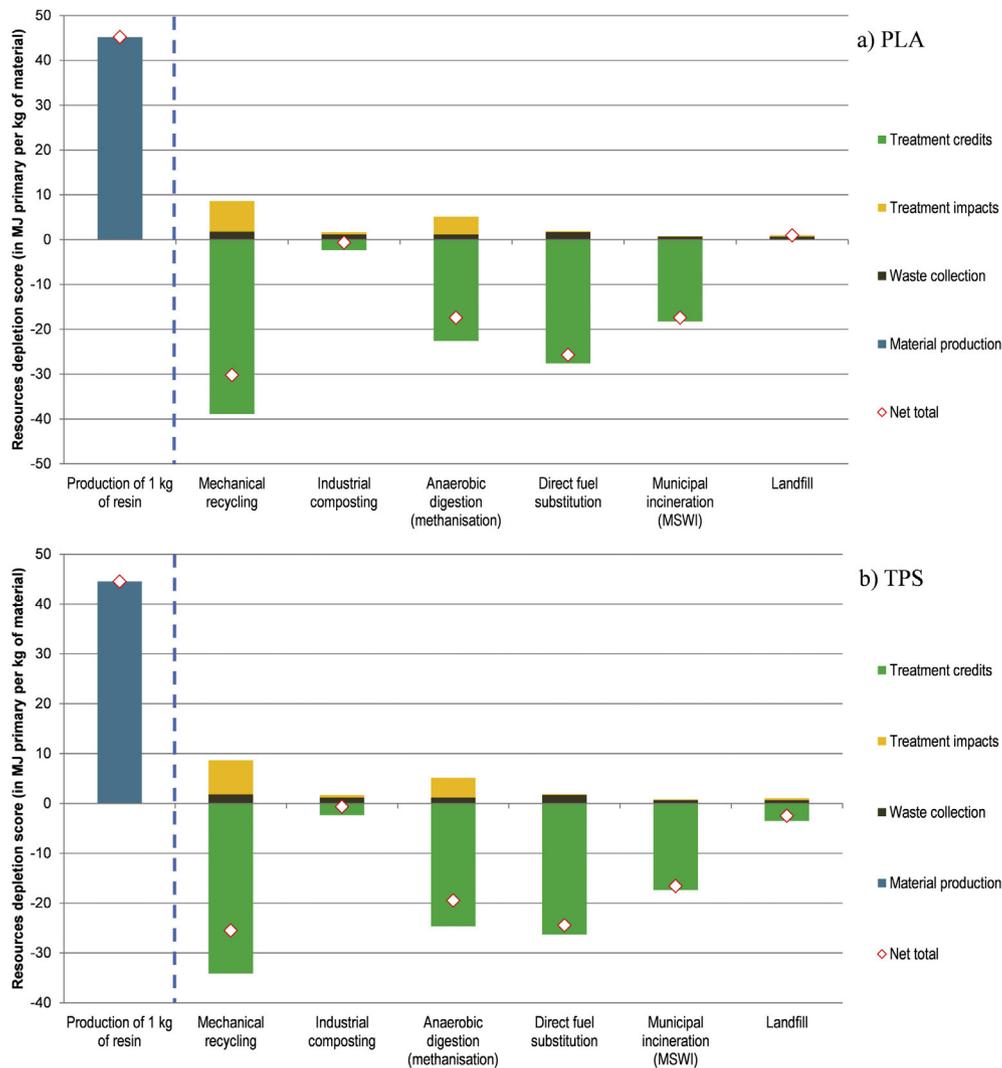


Fig. 5. Comparison of resources depletion impacts associated with the six end-of-life treatments of PLA (a) and TPS (b). The bars on the left side present production impacts of the resin (cradle-to-gate) for comparison purposes.

treatment before final disposal". This shows the importance of product-specific studies when interpreting the EU waste hierarchy. Other deviations from the hierarchy are less critical.

4.2. Dynamic assessment – importance of short term vs long term carbon storage

Compared to a pulse emission, accounting with dynamic calculation (Levasseur et al., 2013) for intermediary delays of e.g. 36.3 years for the compost scenario may reduce the impact at 100 years by 30%, and up to 43% for PLA in landfill. However, in the case of the two studied biodegradable materials, the dynamic treatment has only restricted influence on the final global warming impact at 100 years since either most of the material is degraded early within 100 years (TPS, short delay for most of the material) or only a very small fraction is degraded within 100 years (PLA, long delay for few of the material). Providing and comparing both 100 years and long term impacts (as presented in the Supporting Information) is useful and informative to demonstrate the magnitude of the sequestration occurring with PLA in LF scenario. The 100-year horizon is suitable for the upstream life cycle stages (i.e. manufacturing), but it only shows a fraction of the impacts for the end-of-life of slowly

degrading materials. Thus the importance to also use a time frame that includes long term emissions for end-of-life LCAs.

4.3. Comparison of end-of-life options

In many of the assessed impact categories, either mechanical recycling or direct fuel substitution have the lowest environmental impacts. This is related to substantial benefits (negative impacts counting as credits in Figs. 3–5) associated with the recovery of the original packaging material (in recycling) and the recovery of a large amount of thermal energy (DFS). This credit is conditional on the realized substitution of these materials on the market and depends on the biomass system considered for the production of plastic.

Mechanical recycling of TPS is not always possible with current technologies due to its solubility, depending on TPS grade.

Industrial composting has high environmental impacts in most categories, because the considered polymers degrade almost completely and have a very low N–P–K content, hence cannot replace fertilisers. A conservative approach of peat substitution was considered in this study to represent a maximum substitution potential, but there is little evidence that these materials contribute to

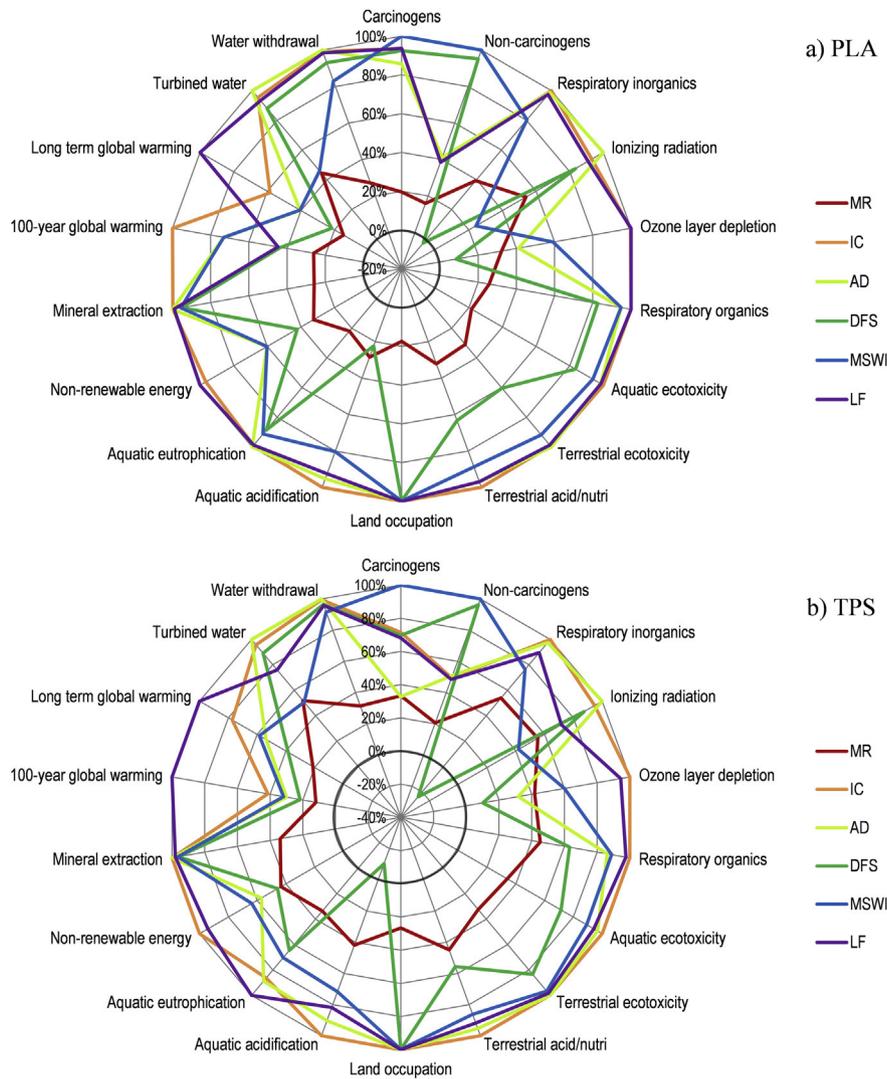


Fig. 6. Comparison of end-of-life options for PLA (a) and TPS (b) for each midpoint category.

porosity or moisture and nutrient retention when the final compost substrate is applied to soil.

Landfilling has relatively high impacts on resources depletion, given that very limited benefits are generated (related to low rate of methane capture to produce energy). Impacts of landfilling on global warming vary strongly between PLA and TPS: PLA is hardly degraded in a landfill over the 100-year time scale considered, which is equivalent to a biogenic carbon sink, while TPS degrades in landfill and results in significant methane emissions. Landfill emissions are one of the reasons why the European directive on the landfilling of waste calls for a phase-out of biodegradable materials in landfills (The European Parliament and Council, 1999). A complete phase-out is presently being discussed for future legislation.

If TPS degradation in a landfill were lower by a factor of 3 (only 30% degraded fraction after 100 years) because of drier conditions, a low microbial population, and/or lower degradability than expected, as suggested by European Bioplastics (2011), the methane emissions would still dominate the global warming impacts. The benefits related to energy recovery would also be smaller by a factor of 3 for all indicators, but this would not affect the conclusions.

Regarding landfilling, the two selected materials are at both extremes of the biodegradability range of materials currently on

the market and it can therefore be considered that other biodegradable materials will fall within these two cases.

Anaerobic digestion and municipal incineration present similar overall impacts because both recover energy with a limited yield. This contrasts with direct fuel substitution, which has a 100% yield but depends on the demand of other industries and hence can only be applied to a limited fraction of waste treatment. In general, anaerobic digestion has slightly lower impacts than municipal incineration. However, when considering that methane produced in anaerobic digestion substitutes UCTE electricity and heat – instead of heat only as in the base case – this option has a significantly lower impact than municipal incineration, despite the relatively high energy recovery yields considered for incineration plants in this study.

4.4. Availability of treatment facilities

Although composting and anaerobic digestion were considered as potential end-of-life options for the two materials studied, many sites currently available for composting and anaerobic digestion do not accept packaging materials. Operators are concerned that biodegradable plastics are not easily distinguishable from conventional plastics and that quality control of the substrate to be

treated will be difficult (de la Harpe, 2011) therefore, there are few existing facilities.

For all scenarios and particularly for mechanical recycling, the availability of infrastructures is of key importance for the practical implementation of the preferred treatment options. The different actors (consumers, producers, public/private systems for waste disposal) are not involved with the same level of implication in all scenarios.

4.5. Combination of end-of-life streams

The studied scenarios are modelled separately and show which option tends to be the best for the studied system. In reality, the environmental benefit of a given end-of-life treatment is likely to reach its maximum at a collection rate below 100%. The location of this maximum will vary with a wide number of parameters including packaging material and packaging type, degree of contamination, geographical spread, available volumes, and efficiency of identification and sorting. Beyond this maximum, a further increase of collection and sorting for a particular end-of-life process may become counterproductive due to the environmental impacts incurred by decontamination or increased transportation. In other words, the infrastructure needed to reach a 100% collection rate would be too heavy and sub-optimal. For example, in order to generate further environmental gains beyond the recycling optimum, it would be necessary to apply other end-of-life processes, such as incineration with heat recovery or anaerobic digestion to the remaining fraction. A combination of solutions is more likely to be the optimal solution for end-of-life treatment rather than a single solution.

4.6. Packaging containing or contaminated with food

The analysis presented in this paper does not consider packaging that cannot be easily separated from its contents. A study of end-of-life solutions for packaging heavily contaminated with wet food residues could lead to substantially different conclusions due to increased impacts related to decontamination during recycling operations, reduced calorific gain for incineration and anaerobic degradation of the food residues in landfills. In addition, the benefit of diverting biowaste from landfill or incineration to composting, for example, could be attributed to the biodegradable packaging, considerably modifying its final impact, as demonstrated by Razza and Innocenti (2012).

4.7. Presence of trace pollutants

The analysis presented in this paper does not consider trace pollutants possibly embedded in the bio-based material that could have non negligible toxic impacts, especially for the landfilling scenario. Future studies are required to assess the impacts of these trace pollutants, considering both the capture in the polymer production and their release during end-of-life.

5. Conclusions

The present study compares the environmental impacts of a representative set of end-of-life options for used dry packaging under the key assumptions that collection, sorting and reprocessing capacities are available and that materials are not heavily contaminated when entering treatment. In the case of dry packaging, industrial composting is the end-of-life option among those studied that generates the highest environmental impacts for the most categories. The hypothesis that composting should be considered more environmentally advantageous than energy

recovery because it is a form of recycling is therefore not confirmed, thus underlining the importance of an appropriate and case-specific application of the EU waste hierarchy.

Options that recover energy or materials are likely to contribute to a reduction in environmental impacts.

Biodegradability of dry packaging is best exploited by anaerobic digestion, but this characteristic leads nevertheless to limited environmental benefits, as demonstrated by the intermediary ranking of this end-of-life option.

Though of limited effect on the present study, considering dynamic impacts of greenhouse gases may moderately decrease the impacts effectively taking place over the 100-year horizon. More interesting is to present to decision makers both the 100-year and the long term impact of scenarios implying long term biodegradation.

The conclusions drawn from this study are only valid under the stated hypotheses. Multiple other factors not covered in this study may lead to different conclusions. In particular, packaging materials heavily contaminated with food may require cleaning steps that would make recycling unattractive from an environmental standpoint. Nevertheless, in such cases anaerobic digestion would deliver more environmental benefits than industrial aerobic composting. The results presented clearly support a case-to-case approach for a sound application of the European waste hierarchy.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at <http://dx.doi.org/10.1016/j.jclepro.2014.08.04>.

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