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Life Cycle Assessment of Waste Prevention Activities: Methodological Approach and Case Studies Focused on Packaging

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Abstract

This thesis deals with the evaluation of the environmental impacts of municipal waste prevention activities by means of life cycle thinking (LCT) and the associated quantitative tool, life cycle assessment (LCA). The script comprises an introductory section providing the background elements to understand the drivers and the basis of the research (Section 1), and the syntheses of four specific research activities.

First of all, the environmental and energy convenience of two packaging waste prevention activities based on product substitution is separately assessed. The first activity substitutes one-way or refillable bottled water by public network water withdrawn from the tap or public fountains (case study 1, Section 2). In the second activity, some categories of liquid detergents packaged in single-use plastic containers are substituted by those distributed loose through self-dispensing systems and refillable containers (case study 2, Section 3). In both the assessments, different baseline scenarios using the substituted product are compared with two waste prevention scenarios using the alternative product. Each baseline scenario accounts for the use of a particular type and/or size of packaging for the substituted product, while the waste prevention scenarios depict different ways of providing the citizens with the alternative less waste-generating product. The results of these case studies reveal that the ultimate environmental convenience of waste prevention activities based on product substitution often depends on the way the activity is actually implemented by the involved actors (citizens, institutions, producers etc.) and possible further variables associated with both the substituted and the alternative products.

The methodological choices preventing traditional waste management LCA from addressing waste prevention activities are then discussed, and the amendments or methods already proposed in the scientific literature to overcome this limitation are reviewed (Section 4). Based on these elements, and on further elaborations and research, two alternative methodological approaches (conceptual models) to incorporate waste prevention activities into LCA of integrated municipal waste management systems are identified, presented and discussed. By defining a proper functional unit and setting adequate system boundaries, it is thus possible to evaluate the environmental and energy performance of municipal waste management systems including different types of waste prevention activities (reduction in product or service consumption, product/service substitution, reuse and lifespan extension).

Finally, the effects of the two examined prevention activities on the overall environmental impacts of the municipal waste management system of Lombardia (Italy) are evaluated, by applying the proposed amendments (Section 5). A 2020 reference scenario is compared with different waste prevention scenarios, where the two activities are both separately and contemporarily implemented, by assuming a complete substitution of the traditional product(s). The results show that, when the implemented activity is actually beneficial, the overall environmental performance of the waste management system is improved, due mainly to the additional upstream benefits of waste prevention. The magnitude of these improvements obviously depends on the activity implemented (which does not always allows for appreciable benefits) and can vary significantly from one impact category to another.

Keywords: waste prevention; life cycle assessment; LCA; product substitution; municipal solid waste; MSW; waste management, bottled water, tap water; detergent distribution; self-dispensing; refillable containers; loose products.

Sommario

Il tema principale di questa tesi è la valutazione degli impatti ambientali delle azioni di prevenzione dei rifiuti urbani tramite l'approccio del life cycle thinking (LCT) e il suo strumento quantitativo, l'analisi del ciclo di vita (life cycle assessment; LCA). L'elaborato è composto da una parte introduttiva dove si riportano gli elementi utili a comprendere le motivazioni e le basi della ricerca (Capitolo 1), e dalla sintesi di quattro specifiche attività di ricerca.

Le prime due attività valutano separatamente la convenienza ambientale ed energetica di due azioni per la prevenzione dei rifiuti da imballaggio, che si basano sulla sostituzione con prodotti alternativi a minor produzione di rifiuti. La prima azione prevede la sostituzione dell'acqua confezionata in bottiglie monouso o riutilizzabili con quella di rete prelevata dal rubinetto o da fontanelli pubblici (caso di studio 1, Capitolo 2). La seconda azione considera invece la sostituzione di alcune tipologie di detersivi liquidi confezionati in flaconi monouso, con detersivi della stessa tipologia, ma distribuiti in modalità sfusa attraverso erogatori automatici e flaconi riutilizzabili (caso di studio 2, Capitolo 3). In entrambi i casi, si confrontano diversi scenari base in cui si utilizza il prodotto sostituito, con due scenari di prevenzione dei rifiuti, in cui si utilizza il prodotto alternativo. I risultati evidenziano che la convenienza ambientale delle azioni di prevenzione basate sulla sostituzione di prodotto dipende spesso dalle modalità di attuazione delle stesse da parte degli attori coinvolti (cittadini, istituzioni, produttori ecc.) e da eventuali altre variabili, relative sia al prodotto sostituito che a quello alternativo.

La tesi si focalizza in seguito sugli adeguamenti metodologici necessari all'integrazione delle azioni di prevenzione dei rifiuti all'interno delle tradizionali tecniche di analisi del ciclo di vita dei sistemi di gestione integrata dei rifiuti urbani (Capitolo 4). A partire da una rassegna degli adeguamenti proposti di recente nella letteratura scientifica e sulla base di ulteriori elaborazioni e ricerche, si identificano, presentano e discutono due approcci metodologici (modelli concettuali) di LCA, utilizzabili per condurre questa tipologia di valutazioni. Tali approcci dimostrano che, definendo l'unità funzionale e i confini del sistema in modo adeguato, è possibile valutare le prestazioni ambientali ed energetiche di sistemi di gestione integrata dei rifiuti urbani che comprendono diverse tipologie di azioni di prevenzione (riduzione del consumo di prodotti o servizi, sostituzione di prodotti o servizi con degli equivalenti a minor produzione di rifiuti, riutilizzo dei prodotti ed estensione della loro vita utile).

Infine, applicando gli adeguamenti metodologici proposti si valutano gli effetti delle due azioni di prevenzione già esaminate singolarmente sugli impatti ambientali complessivi dell'intero sistema di gestione dei rifiuti della Lombardia (Capitolo 5). A questo scopo, uno scenario di riferimento al

2020 viene confrontato con diversi scenari preventivi, in cui le due attività di prevenzione sono implementate, sia separatamente che contemporaneamente, ipotizzando una sostituzione completa del/dei prodotto/i tradizionale/i. I risultati evidenziano che, quando l'attività di prevenzione è effettivamente vantaggiosa, si consegue sempre un miglioramento delle prestazioni ambientali complessive del sistema. L'entità di tali miglioramenti dipende dall'azione implementata (che non sempre comporta dei benefici apprezzabili) e può variare in modo significativo da una categoria di impatto all'altra.

Parole chiave: prevenzione dei rifiuti; analisi del ciclo di vita; LCA; sostituzione di prodotto; rifiuti solidi urbani; RSU; gestione dei rifiuti; acqua in bottiglia; acqua di rete; distribuzione detersivi; flaconi riutilizzabili; prodotti sfusi.

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Introduction

Waste prevention has for years been the priority of the waste management strategy of Europe and most developed countries. However, in Europe, waste generation has always remained correlated with the economic trend, represented through GDP (in the case of total waste) or household consumption expenditure (in the case of municipal waste). For this reason, the latest waste framework Directive (2008/98/EC) has introduced particularly strengthened provisions on waste prevention, by requiring the development of national waste prevention programmes within the end of 2013. The programmes shall set out prevention objectives and identify the most suitable measures for their achievement. In Italy, the national regulations introduced also the additional requirement to prepare regional waste prevention programmes, identifying tangible actions to be implemented at the local level. According to both the European and Italian legislation, the aim of waste prevention objectives, measures and actions is not to simply reduce waste generation or to decouple it from the economic growth, but to break the link between this latter and the environmental impacts associated with the generation of waste.

The selection of waste prevention measures and actions to be included in national and regional programmes should thus be made only after a careful evaluation of the associated waste prevention potential and of their actual capability to reduce the overall impacts on the environment and human health. This is especially important when waste is not prevented through the 'simple' reduction in the consumption of a product or service, but through more complex mechanisms such as product substitution, reuse, or lifespan extension. In this case, in fact, additional waste and/or impacts are usually generated, which have to be carefully compared with those avoided to properly evaluate the actual environmental and energy convenience of the waste prevention activity. An example is the substitution of a given good or service by an alternative, less waste-generating one, providing the same function (e.g. one-way bottled water is substituted by public network water). This substitution does not imply only environmental and energy benefits (thanks to the avoided production, distribution, use and disposal of the substituted good or service), but also involves additional environmental and energy impacts, resulting from the consumption of the alternative good or service. Life cycle thinking (LCT) is the most suitable approach to evaluate the environmental and energy convenience of a waste prevention activity, since it takes into account the impacts of a product or service along its whole life cycle, to prevent unwanted "shifting of burdens" from one stage or location in the life cycle to another. In the case of waste prevention activities, this approach and the associated quantitative tool (life cycle assessment; LCA) can be used to avoid that a reduction in waste generation is followed by an unwanted increase in the overall environmental and energy impacts. Examples of prevention activities for which the application of LCA can prove to be useful are those addressing packaging waste, which often propose alternative systems for the distribution of large consumer goods such as liquid detergents and dry food products. Usually, these alternative systems compete with traditional supply chains that are highly optimised from the environmental standpoint (reduced amount of packaging material used per unit of product, optimised logistics etc.) and that may thus not be easily outperformed by new initiatives, potentially subject to important improvements.

Several life cycle assessments (LCAs) of municipal waste management systems are available in the scientific literature of the last two decades. However, they have rarely accounted for the effects of specific waste prevention activities. This is mostly because the traditional LCA of waste management adopts some methodological choices (the functional unit and the system boundaries being the most important) that prevent the comparison of scenarios where the total amount of waste is variable (as it is the case for waste prevention activities). A partial revision of the methodology is thus required to carry out this type of assessment. In addition, it is necessary to calculate more parameters (such as the waste prevention potential), to collect or estimate a larger quantity of data, as well as to broaden the knowledge of the supply chains affected by waste prevention activities. All these aspects have likely discouraged in the past the incorporation of waste prevention activities, especially if we consider that, in many European countries, the focus has until recently been on increasing material and energy recovery from waste.

Recently, some authors have individually proposed possible amendments to overcome the methodological limitations associated with traditional assessment techniques (Cleary, 2010, Gentil et al, 2011 and, partially, Matsuda et al., 2012). In some cases, similar approaches were proposed. A critical review of the proposed amendments, their reasoned reorganisation in one or more structured methodological approaches (conceptual models), as well as the individuation of possible alternative approaches is thus deemed to be of use.

Based on the proposed amendments, a few case studies for real or fictional geographical regions have been produced (Gentil et al., 2011; Matsuda et al., 2012, Cleary, 2014 and, partially, Slagstad and Brattebø, 2012). Most of them conclude recommending further research, by possibly focusing on other significant waste fractions and other prevention activities available.

Objectives of the research

The overall objective of the research is to investigate the environmental performance of municipal waste prevention activities, by means of life-cycle thinking and assessment. In particular, the research has the following specific objectives:

(1) To evaluate the environmental and energy convenience of some of the packaging waste prevention activities based on product substitution, which are identified by the most recent reviews of 'best practices' and most commonly considered during the preparation of national or regional waste prevention programmes;

(2) To identify and discuss alternative methodological LCA approaches (conceptual models) for the evaluation and the comparison of the potential environmental impacts of municipal solid waste management systems that include waste prevention activities; and

(3) To evaluate the effects of packaging waste prevention activities based on product substitution, on the environmental and energy performance of a modern waste management system, characterised by high levels of material and energy recovery.

Structure of the thesis

The contents of the thesis are organised as follows.

Section 1 provides an overview of the provisions on waste prevention and life cycle thinking laid down by the European and Italian waste legislation. A comprehensive review of municipal waste prevention activities is then presented and a possible classification is proposed. The environmental consequences of the different types of prevention activities reviewed are also briefly discussed. A general description of the LCA methodology is finally provided. Altogether, these elements provide the necessary background to understand the drivers and the basis of the research.

Sections 2 and **3** summarise separately the LCA studies of two specific packaging waste prevention activities: the substitution of one-way bottled water by public network water (case study 1, Section 2) and the substitution of liquid detergents packed in single-use containers by those distributed "loose" through self-dispensing systems and refillable containers (case study 2, Section 3). The assessment calculates the waste prevention potential of the two activities and evaluates whether, unconditionally or only under particular conditions, they actually reduce waste generation, the overall environmental and human health impacts and the total energy demand [objective 1].

Section 4 discusses the methodological choices of traditional waste management LCA that prevent it from addressing waste prevention activities, and provides a critical review of the amendments and methods recently proposed in the scientific literature in the attempt to overcome this limitation.

Two alternative methodological LCA approaches (conceptual models) for the evaluation of the environmental and energy performance of integrated municipal solid waste management systems incorporating different types of prevention activities are then presented and discussed. The two approaches are identified based on both the structured reorganisation of the amendments/methods reviewed and on further personal elaborations and research [objective 2].

Section 5 includes an assessment of the effects of the waste prevention activities analysed in Sections 3 and 4 on the impacts of the municipal waste management system of the Lombardia Region (Italy). The analysis is carried out by applying the methodological approach identified in Section 4 and compares a 2020 reference scenario with two waste prevention scenarios where the two activities are separately implemented. A comparison with a third scenario where both activities are included is also carried out. The study evaluates the effects on both the overall impacts of the systems and on those of its traditional components (i.e. collection, sorting and treatment operations). The effects of the variation of the substitution level of traditional products are also explored in a sensitivity analysis [objective 3].

Finally, **Section 6** presents the general conclusions of the research and provides recommendations and some suggestions for possible future research.

The research presented in this PhD thesis is partly summarised in three scientific papers:

Nessi S., Rigamonti L., Grosso M. (2012) LCA of waste prevention activities: a case study for drinking water in Italy. Journal of Environmental Management 108, 73-83.

Nessi S., Rigamonti L., Grosso M. (2013) Discussion on methods to include prevention activities in waste management LCA. The International Journal of Life Cycle Assessment 18(7), 1358-1373.

Nessi S., Rigamonti L., Grosso M. (2014) Waste prevention in liquid detergent distribution: a comparison based on life cycle assessment. Science of the Total Environment 499, 373-383.

In addition, during this PhD study, an internal report for Finlombarda SpA – Regione Lombardia was edited[1], and five contributions in the proceedings of international conferences were produced.

[1] The report is titled “Municipal waste prevention activities in Lombardia: environmental and energy evaluation by means of life cycle assessment” and is edited in Italian. It summarises the LCA studies of four different packaging waste prevention activities based on product susbtitution, two of which are reported in this thesis (Sections 2 and 3).

1 Waste prevention and life cycle assessment: an overview

This introductory section provides, first of all, an overview of the provisions on waste prevention and life cycle thinking (LCT) included in the European waste legislation. These are one of the key drivers for this research. A comprehensive review of the major types of municipal waste prevention activities and relating examples is then reported, representing the basis for the selection of the assessed activities and for the discussion of the methodological approaches described in Section 4. The environmental consequences of prevention activities are also briefly discussed, putting the rationale for the research. Finally, a background description of the life cycle assessment (LCA) methodology, which is used extensively throughout the thesis, is provided.

1.1 Waste prevention in the European legislation: requirements and definitions

Waste prevention has formally been part of the European legislation on waste since 1975 (year of adoption of the first Waste Framework Directive, 75/442/EEC; European Council, 1975). In this document, Member States are requested to "take appropriate steps to encourage the prevention, recycling and processing of waste, the extraction of raw materials and possibly of energy therefrom and any other process for the re-use of waste". The Directive was then amended over the years and replaced in 2008 by a revised version, currently in force, where waste prevention has become one of the core elements (Directive 2008/98/EC; European Parliament and Council, 2008). First of all, the so-called *waste hierarchy* is explicitly introduced as the priority order to be followed in waste management legislation, policies and practices (article 4; Figure 1.1). According to this hierarchy, which has been the core of the European waste management strategy since 1989 (Commission of the European Communities, 1989, 1996), waste prevention is given the highest priority, followed by preparing for re-use. Waste that cannot be prevented or reused should preferably be recycled or, alternatively, recovered in other forms (such as energy). Finally, disposal should be limited to that waste for which no recovery options are available. Thus, in principle, even before proceeding with an environmentally and economically sound management of the generated waste, all the efforts should be made to reduce its quantity (and hazardousness), by means of suitable waste prevention measures (see the official definition of waste prevention reported later). However, the directive clearly states that specific waste streams may move away from the hierarchy, if it can be demonstrated by life cycle thinking that a better overall environmental outcome is achieved (see Section 1.2 for further detail).

The innovative requirement for Member States to develop national *waste prevention programmes* is then introduced in the directive (article 29). The programmes shall set out waste prevention objectives and identify the most suitable prevention measures for their achievement. The aim of such objectives and measures is not to simply reduce waste generation, or dissociate it from economic indicators, but to break the link between economic growth and the environmental impacts associated with the generation of waste. Waste prevention measures shall thus allow for a reduction in both overall waste generation and environmental impacts. To help Member States with the development of the programmes, the Commission is required to create a system for sharing information on best waste prevention practices and to develop specific guidelines. Moreover, by the end of 2014, waste prevention and decoupling objectives for 2020 shall be set out by the Commission, based on best available practices (article 9).

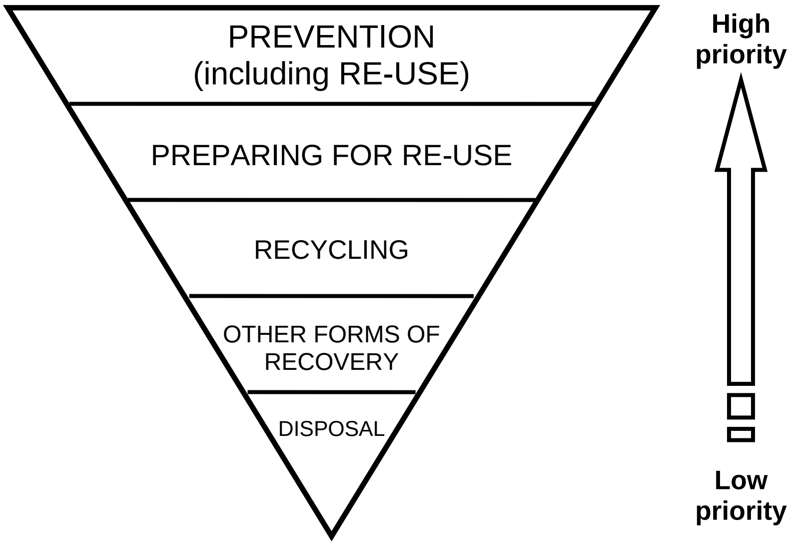


Figure 1.1: priority order to be applied in waste prevention and management legislation, policy and practice, according to the revised Waste Framework Directive 2008/98/EC. This priority order is usually referred to as waste hierarchy.

To prevent misleading interpretations and clarify the differences with the other components of the hierarchy, the Directive also provides the first legal definition of waste prevention at the Community level (article 3):

'prevention' means measures taken before a substance, material or product has become waste, that reduce:

(a) the <u>quantity of waste</u>, including through the re-use of products or the extension of the life span of products;

(b) the <u>adverse impacts of the generated waste</u> on the environment and human health; or

(c) the <u>content of harmful substances</u> in materials and products.

A first important aspect emerging from this definition is that waste prevention measures take strictly place only before a good has become waste. Therefore, unlike what is frequently thought, they do not include all those measures aimed at reducing waste disposal. Beyond waste prevention, these measures include also material and energy recovery, and are generally defined as *waste minimisation* (or *diversion*) measures (OECD, 2004)[1].

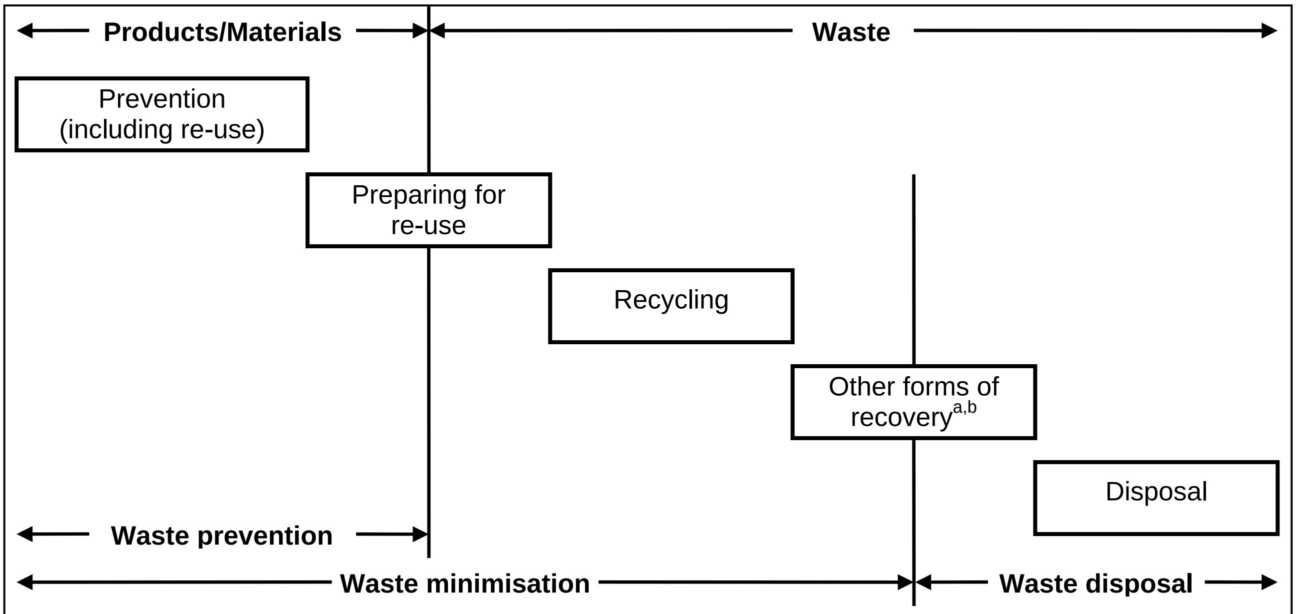
A distinction is then made between quantitative and qualitative waste prevention measures. The former are aimed at reducing the quantity of generated waste, while the latter aim at reducing the adverse impacts on the environment and human health from the management of the generated waste. Since these adverse impacts can also be caused by harmful substances in the waste, the reduction of their content is also included in the concept of qualitative waste prevention measures. Qualitative prevention, thus, does not exclude the other components of the waste hierarchy, while, by definition, they are excluded by quantitative prevention. The focus of this thesis is exclusively on quantitative waste prevention measures targeting municipal solid waste.

According to the definition, quantitative waste prevention measures include also reuse and lifespan extension, since they ultimately lead to a reduction in the amount of waste generated. Generally, such measures target goods that have already been manufactured, although lifespan extension may also be a design choice taken by producers for product that still have to be manufactured. Reuse is specifically defined as *any operation by which products or components <u>that are not waste</u> are used again for the same purpose for which they were conceived* (article 3) and is a form of waste prevention at two different levels. First of all, it delays the moment a product becomes waste, thus allowing for an immediate reduction in waste generation. Moreover, the overall quantity of a given product that is generated as waste by a given activity is reduced (European Commission DG Environment, 2010).

Formally, waste prevention measures exclude preparing for re-use, which is located at the second place of the waste hierarchy, immediately after prevention. This is because preparing for re-use acts on products that, from a legal standpoint, have already become waste. In fact, according to the definition provided by the Directive, preparing for re-use includes *checking, cleaning or repairing operations, by which products or components of products <u>that have become waste</u> are prepared so that they can be re-used without any other pre-processing* (article 3). However, some stakeholders argue that the distinction between reuse and preparing for reuse is merely a legal issue, depending on whether the targeted product is formally recognised as waste or not (European Commission DG

[1] A definition of waste minimisation is provided by OECD (2000): "preventing and/or reducing the generation of waste at the source, improving the quality of waste generated, such as reducing the hazard, and encouraging re-use, recycling and recovery".

Environment, 2010). Figure 1.2 attempts to clarify the relationship among the definition discussed up to now.



(a) Such as energy recovery.
(b) 'Other forms of recovery' represents a borderline option between waste minimisation and disposal since, according to the Waste Framework Directive 2008/98/EC, the incineration of municipal solid waste with energy recovery can be considered a recovery operation only if a given energy efficiency is achieved.

Figure 1.2: representation of the relationships among the different levels of the waste hierarchy (adapted with modifications from ACR+ (2010).

The key driver behind the strengthened provisions on waste prevention included in the revised waste framework directive is the Thematic Strategy on the prevention and recycling of waste (Commission of the European Communities, 2005). In this document, the failure to achieve previous Community and national waste reduction targets is recognised and additional policy measures to promote waste prevention are urged. These include the obligation for Member States do develop publicly available waste prevention programmes and the fostering of the use of the IPPC Directive[2], of the Integrated Product Policy and of other tools to encourage the spread of best waste prevention practice. The ultimate aim of the thematic strategy is to help Europe to become an economically and environmentally efficient 'recycling society', that seeks to avoid waste and uses waste as a resource.

The provisions of the revised Waste Framework Directive have been faithfully implemented in national legislation with the Legislative Decree 205/2010[3]. However, the additional requirement is introduced for the Italian Regions to develop a *regional waste prevention programme*, based on the

[2] IPPC: Integrated Pollution Prevention and Control.
[3] This decree amends and integrates most of the provisions included in the Legislative Decree 152/2006 (part IV), which regulates waste management practice at the national level.

national waste prevention programme (art. 199, par. 3, point r). The regional programme shall describe the existing waste prevention measures and establish further appropriate measures and prevention objectives. As for national programmes, the aim of such measures and objectives shall be to break the link between economic growth and the environmental impacts associated with the generation of waste. The regional programme shall include specific qualitative and quantitative benchmarks for waste prevention measures, in order to monitor and assess the achieved progress, even through the determination of indicators.

1.2 Life cycle thinking in the European waste legislation

Other than introducing strengthened provisions on waste prevention, the revised Waste Framework Directive introduces also life cycle thinking into waste management and prevention policy and practice. Life cycle thinking is an holistic approach that consists of taking into account the potential environmental and human health impacts of products or services throughout their whole life cycle, when decisions are to be made. The ultimate objective of the measures laid down by the directive, indeed, is not to merely promote waste prevention and increase energy and material recovery. Conversely, they are aimed at preventing or reducing the adverse impacts associated with the generation and management of waste, as well as contributing to reduce the overall impacts of resource use (article 1).

Deviating from the waste hierarchy is thus allowed, for specific waste streams, if it can be demonstrated by life cycle thinking that a better overall environmental outcome is achieved (article 4). Moreover, national waste prevention programmes shall ‘concentrate on the key environmental impacts and take into account the whole life cycle of products and materials’ (premises 40). Finally, waste prevention and decoupling objectives to be set out by the Commission by 2014, should be defined by ‘covering, as appropriate, the reduction of the adverse impacts of waste and of the amounts of waste generated’ (premises 40).

The quantitative tool supporting life cycle thinking is life cycle assessment, which is briefly described in Section 1.5. This methodology is extensively used in the present work, to evaluate the environmental and energy performance of waste prevention activities.

1.3 Review and classification of municipal waste prevention activities

An extensive review of viable municipal waste prevention activities was initially carried out. Moreover, a possible classification in different types and categories is proposed, based on the environmental consequences they are expected to generate (Table 1.1). This has provided the basis for the selection of the specific activities to be assessed (Sections 2 and 3) and the identification of the modelling approaches presented in Section 4. Several sources have been considered for the review, the most important of which are ACR+ (2010) and Federambiente (2010). Moreover, the following documents proved to be of use: European Commission (2012), European Commission DG Environment (2010), Cleary (2010), Salhofer et al. (2008), Cox et al. (2010) and Sharp et al.(2010). Finally, a number of regional waste prevention programmes/plans available at the beginning of the PhD research were taken into account, such as the ones of Lombardia (Regione Lombardia, 2009) and Piemonte (Regione Piemonte, 2009).

Four main categories of municipal waste prevention activities were identified (Table 1.1). The first includes those activities aiming at reducing waste generation thanks to a reduction in the consumption (or wastage) of a given good or service (types 1 and 2 activities). The activities included in the second category (types 3 to 6) are instead based on a more complex mechanism, which is the substitution of a given good or service by a less waste-generating equivalent one. The reuse of disposable or durable goods (types 7 and 8 activities), as well as lifespan extension of existing or new durable goods (types 9 and 10 activities) complete the framework.

Some municipal waste prevention activities can be undertaken directly by citizens (e.g. the reduction of food wastage or the direct reuse of a good). However, in most cases, even retailers, producers and public service providers need to be actively involved, especially when product or service substitution is to be implemented. In other cases, new businesses need to be started up (such as reuse and/or repair centres), so that further actors are to be engaged.

A number of instruments can be used to facilitate the implementation of waste prevention activities, such as:

- *Regulatory and legal instruments*, i.e. legal provisions and regulations introducing bans, authorisation requirements, product standards etc. For example, a voluntary ban on bottled water retailing could be imposed for a given area (as in the rural municipality of Boundanoon, Australia; ACR+, 2010).
- *Market-based or economic instruments* such as grants, subsidies, tax incentives and concessions, taxes and charges. A few examples are taxes and charges on disposable products, such as plastic bags and batteries; reduced tax on the sale of reused goods; differential charging for household waste, such as pay as you throw systems; subsidies to households for the purchase of less waste-generating goods (e.g. reusable nappies) and to retail establishments for the introduction of such goods among retailed products.
- *Suasive or communication instruments* such as public awareness campaigns, marketing of less waste-generating products and education of consumers to responsible purchases.

While regulatory-legal, market-based and economic instruments can be used exclusively by public administrations, suasive and communication instruments can also be exploited by other subjects such as retailers, producers or service providers.

Table 1.1: main types of municipal waste prevention activities reviewed and relating examples.

Type of waste prevention activities		Examples
Reduction in the consumption of goods or services	1) Reduction in the consumption of goods by citizens, companies or organisations (without reducing the consumption of the service originally provided by those goods)	- Reducing paper consumption through simple good practices such as double-sided printing and copying, using the back of unnecessary single-sided documents for notes and memos, printing less important documents with reduced margins and smaller characters etc. - Renting or borrowing/lending of goods instead of purchasing new ones (e.g. infrequently used clothes and textiles, office furniture, toys, books, home and garden tools, party/event decorations and supplies, paints etc.)
	2) Reduction in the wastage of goods (unnecessary to the consumer)	- Reducing household food waste (unconsumed or partially consumed food and leftovers) by planning food purchases, avoiding over-purchasing, taking into account the life time of products, storing food products in proper conditions, re-using leftovers etc. - Reducing food waste in the catering industry by allowing customers to take any leftover food away. - Reducing food waste from retailers by donating still edible food products, which are no longer sellable for market reasons, to social canteens, social supermarkets or other social welfare services intended for people in need (even unconsumed meals from public or private canteens can be intercepted and donated) - Reducing the delivery of unsolicited mail such as unaddressed advertising material by applying dissuasive stickers on the mailbox, subscribing to mail preference services etc.

Substitution of a product or service by a less waste-generating equivalent one	3) Reducing the amount of material used for the manufacturing or packaging of a good through a more efficient design (without reducing product performance)[a]	- Reducing the amount of steel used to manufacture a washing machine
		- Reduction in the amount of packaging material used per unit mass of packaged product, like:
		  - lightweighting of beverage bottles (without reducing their strength)
		  - increasing volume capacity of containers

	4) Substitution of an unpacked good for a packed one	- Drinking of (refined) public network water from the tap or public fountains/suppliers instead of bottled water

(a) e.g. the amount of packaged product damaged or lost is not increased.

Table 1.1 (continued)

Type of waste prevention activities		Examples
Substitution of a product or service by a less waste-generating equivalent one (continued)	5) Substitution of a reusable good or a good provided in a reusable packaging for a disposable good or a good provided in a disposable packaging	- Packaging of water or other beverages in refillable bottles rather than in one-way bottles - Distribution of liquid detergents through self-dispensing systems available at retail establishments, rather than packed in single-use containers (detergent withdrawal is made by refillable containers) - Distribution of ‘loose’ dry food products (e.g. pasta, rice and breakfast cereals) through gravity dispensers available at retail stores, rather than individually packaged (product withdrawal is made by disposable lightweight plastic or paper bags) - Distribution of raw milk through self-dispensing systems placed in public spaces by local farmers, as an alternative to packaged milk (milk withdrawal can be made with refillable bottles) - Delivery of local, unpacked, fruit and vegetable products to the households, by means of returnable crates - Shipment of goods by means of returnable cardboard boxes rather than disposable ones - Use of reusable transport packages rather than disposable ones (e.g. collapsible plastic crates for fruit and vegetable products) - Use of reusable shopping bags rather than disposable plastic or paper ones - Drying of hands by means of electric hand-dryers or cloth roll towels rather than paper bath-towels - Serving meals with reusable crockery rather than disposable ones (in the whole catering industry and during local or big events) - Swaddling babies in reusable nappies rather than disposable ones - Use of rechargeable batteries instead of disposable ones
	6) Substitution of a digital good for a disposable one	- Substitution of internet advertising brochures for printed ones by retailers - Reading of on-line newspapers instead of printed ones - Opting for internet billing and invoicing services instead of paper-based ones (by households, companies or organisations) - Digitalisation of documentation and bureaucratic procedures in companies, organisations and public administrations (communications, letters, invoices, orders, reports, forms, inventories, press releases etc.)

Table 1.1 (continued)

Type of waste prevention activities		**Examples**
Reuse of goods	7) Direct reuse of disposable goods or packages by the owner (private citizens or organisations) in substitution of disposable or durable goods or packages	- Reuse of a disposable shopping bag, of a disposable glass jar, of a one-way glass or plastic bottle etc.
	8) Reuse of durable goods through second-hand retailing/purchasing, donations and exchanges	- Selling/purchase in second-hand markets, donation to charities and people in need or exchange of durable goods such as clothes and textiles, furniture, electrical and electronic equipment, toys, books, bicycles, sport and fitness equipment, baby and nursery products and accessories, home and garden tools, party/event decorations and supplies etc.
Extension of the lifespan of durable goods	9) Extension of the lifespan of existing durable goods by citizens or repair centres	- Repairing of durable goods by citizens or repair centres (e.g. clothes and textiles, furniture, electrical and electronic equipment, bicycles, sport equipment, home and garden tools etc.) - Keeping appliances in a good working order by following manufacturers' recommendations for a proper operation and maintenance
	10) Extension of the useful life of durable goods by producers	- Extension of the useful life of domestic appliances through a more efficient design

1.4 Environmental consequences of waste prevention activities and the need for life cycle thinking

When waste is prevented through a reduction in the consumption of a given good or service (types 1 and 2 activities), an overall environmental benefit is generally achieved. The impacts associated with the whole life cycle of the targeted good or service are indeed avoided and no additional impacts are generally involved. Therefore, the environmental convenience of this type of activities needs not to be necessarily proven by life cycle thinking, although LCA may be used to quantify the potentially achievable benefits.

When a substitution by a less waste-generating equivalent good or service is made (types 3 to 6 activities), additional impacts are instead generated. They are associated with the whole life cycle of the substitutive good or service and can even be higher than avoided impacts. A careful assessment of the net life cycle impacts deriving from the performed substitution is thus needed to correctly evaluate its actual environmental convenience.

Sometimes, a replacement by alternative goods or services is argued also for activities based on a reduction in product/service consumption (Gentil et al., 2011; Cleary, 2010). An increased income is indeed available to the consumer, who may purchase alternative products or use additional services. However, the consequences of this 'rebound effect' are not easily predictable (which types of alternative goods or service are actually used by the consumer?) and large uncertainties may affect the assessment.

Reuse and lifespan extension of existing goods (types 7 to 9 activities) generate environmental impacts by increasing the use phase of reused goods, while avoiding the impacts from producing, using and disposing of one or more equivalent new goods (EC-JRC, 2011b). Even in this case, a careful comparison between avoided and additional impacts is needed. This especially when equivalent new goods can benefit of a less impacting use phase, thanks to technology improvements (e.g. reduced electricity consumptions for electric and electronic equipment). Finally, when actions to extend the useful life of a good are taken by producers at the design stage (types 10 activities), the impacts of the whole life cycle of shorter-lasting goods should be compared with those of the whole life cycle of longer-lasting goods, to prevent an unwanted increase in the overall environmental and human health impacts.

1.5 Life cycle assessment

Life cycle assessment is a methodology developed since the early 1970s, to evaluate the potential environmental and human health impacts of products or services. The assessment takes into account the whole life cycle of the specific product/service, from raw material extraction, through manufacture, distribution, use, possible reuse/recycling, up to final disposal. In other words, the life cycle is examined ‘from cradle to grave’ (or from cradle to cradle, if it ends with recycling).

This holistic approach to environmental assessment is useful to prevent unwanted “shifting of burdens”, i.e. that measures taken to reduce impacts at one stage in the life cycle, increase them at another stage. Similarly, unwanted shifting of burdens from one location in the life cycle to another and from one type of environmental impact to the other can be identified and prevented. The procedure to carry out an LCA study is currently regulated by two international standards, issued by ISO (International Organisation for Standardisation): the ISO 14040 (ISO, 2006a) and ISO 14044 (ISO, 2006b). Moreover, several national and international guides and handbooks have been developed over the years (e.g. Guinée et al., 2002 and Wenzel et al., 1997). One of the most recent and comprehensive is the International Reference Life Cycle Data System (ILCD) handbook by the European Commission’s Joint research Centre (EC-JRC, 2010a and 2010b). It provides practitioners with common technical guidelines to carry out consistent and quality assured LCA studies, with a special focus on the European context. While the handbook is based on and conforms to the ISO standards, it aims at providing further guidance so that consistent methodological choices are made even when no specific requirement are formulated by ISO standards, or when these allows alternative approaches to be chosen.

According to such standards, an LCA consists of four distinct phases, illustrated in Figure 1.3 and described briefly in Sections 1.5.1 to 1.5.4.

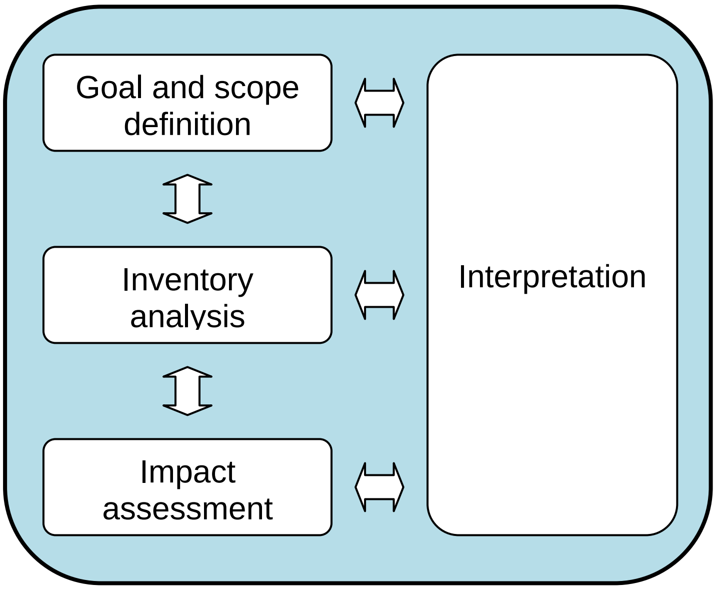


Figure 1.3: the four phases of a life cycle assessment (ISO, 2006a).

1.5.1 Goal and scope definition

In the goal definition phase, the reasons for carrying out the study shall unambiguously be declared. The intended application (how will the results be used) and intended audience (to whom the results will be shown) should also be specified. In the scoping phase, the product system[4] to be studied is described, and the relevant function(s) fulfilled by such a system is selected.

The flows of material and energy and the potential impacts of the system are indeed calculated with reference to a specific quantity of the pre-selected relevant function(s), the so-called functional unit, which is thus another fundamental element to be defined in the scoping phase of an LCA.

In the case of comparative assessments, the functional unit also ensures that the comparison among alternative product systems is fair, by always relating to the same unit. Only the comparison among product systems providing the same function is indeed allowed.

The system boundaries need to be also defined. This is made by specifying which unit processes of the investigated supply chain(s) are included in the studied product system(s) and which are not. The exclusion of relevant processes or life cycle stages needs to be adequately justified.

Other important issues to be addressed during the scope definition phase are the approaches that will be used to solve possible problems of multi-functionality; the environmental issues (impact categories) to be assessed; the methodology to evaluate the selected potential impacts (impact assessment methodology); as well as data quality requirements.

1.5.2 Life cycle inventory analysis (LCI)

The inventory analysis is the phase where, first of all, all inputs to and outputs from the unit processes within the boundaries of the studied product system(s) are identified and quantified with reference to the functional unit. Inputs include natural resources, raw materials, ancillary materials, energy flows and any intermediate products or co-products[5]. Outputs include the releases into the different environmental compartments (air, water and soil), raw materials, energy, intermediate products and co-products.

Based on the identified inputs and outputs and the magnitude of each unit process, the material and energy flows crossing the system boundaries are then calculated. The complete set of crossing flows per functional unit represents the ultimate outcome of the inventory analysis and is often referred to as *Inventory Table*. In principle, if the system is modelled correctly, the Inventory Table will include only elementary flows, i.e. natural resources or energy drawn from the environment without

[4] According to ISO, the collection of unit processes and associated elementary and product flows, which models the life cycle of a product or service is defined as a product system.

[5] Due to the increasing importance of land use and land transformation, they are also generally quantified, although they are not strictly inputs.

any previous human transformation (elementary inputs) and releases to air, water and soil (elementary outputs). Practically speaking, elementary flows are the inputs from and outputs to the environment and are most commonly referred to as *environmental burdens* or *interventions*.

1.5.3 Life cycle impact assessment (LCIA)

The third phase of an LCA is impact assessment. Here, an estimate is performed of the potential environmental and/or human health impacts associated with the results of the inventory analysis. Impact assessment consists of six steps, described below, the first three of which are mandatory, while the remainings are optional.

1) *Selection of impact categories, category indicators and characterisation models* (mandatory).
   Impact categories are represented by particular environmental issue of concern, to which the environmental burdens calculated in the inventory analysis may contribute. Their selection should be made in the attempt to cover all relevant environmental issues related to the product supply chain of interest, taking into account the specific goal of the study.
   Specific category indicators are used to quantitatively express the magnitude of the potential impacts involved by the environmental burdens contributing to the selected impact categories. A quantitative indicator must thus be chosen for each selected category. Finally, the models to be used to calculate category indicators starting from environmental burdens must be selected (the so-called *characterisation models*).

2) *Classification* (mandatory). It is the operation by which the environmental burdens qualified and quantified in the inventory analysis are assigned, on a purely qualitative basis, to the pre-selected impact categories to which they are expected to contribute.

3) *Characterisation* (mandatory). In this step, category indicator results are calculated for the selected impact categories. To this end, the environmental burdens assigned to a particular impact category in the classification step are firstly converted into a common unit for that category. This operation is made by means of characterisation factors, which are calculated for each relevant environmental burden (flow), by means of the characterisation model selected in the first step[6]. The impact indicator result is then calculated by aggregating the different

[6] The category indicator and the characterisation model selected for a given impact category, as well as the characterisation factors derived from the model, define the so-called *characterisation method*. Normally, predefined characterisation methods are available for the most common impact categories, so that a selection of the most suitable method needs to be simply made by the practitioner.

burdens. At the end of the characterisation step, each selected impact category is thus assigned a single numerical value, quantifying the overall potential impact of the studied product system(s) for that category.

4) *Normalisation* (optional). This operation consists in expressing category indicator results, calculated in the characterisation step, in terms of a common reference unit. This is made by dividing the indicator results by respective normalisation factors. Frequently, the total impacts involved in a geographical area (e.g., the world), or by one of its citizens, over a given time period (e.g., one year) are used as normalisation factors.
The main aim of normalisation is to better understand the relative importance and magnitude of the results for the studied product system(s). In other words, normalisation facilitates the identification of those impact categories where the examined product system(s) is more impacting and vice versa.

5) *Grouping* (optional). This is a step in which impact categories are aggregated into one or more homogenous groups, so that the normalised impact indicator results of all the categories included in a given group can be summed up. This operation facilitates the comparison among alternative product systems, when many impact categories are taken into account in the assessment.
Examples of grouping criteria are the spatial scale of the impact (global, regional and local), or the area of protection (human health, environment and resources).

6) *Weighting* (optional). In this last step, the normalised indicator results are assigned numerical factors, according to their relative importance, multiplied by these factors and then aggregated. A single weighted index is thus obtained, which is representative of the overall environmental performance of the studied product system(s).

As anticipated, according to ISO, the operations of normalisation, grouping and weighting are optional. This is because subjective (and thus debatable) choices are involved and result uncertainty is increased.

1.5.4 Life cycle interpretation

The last phase of an LCA is interpretation. Here, the results from the inventory analysis and impact assessment are presented and discussed with reference to the goal and scope of the study. If different product systems are compared, the best alternative is also possibly identified. Finally, conclusions are drawn and possible recommendations are provided to the intended audience.

During interpretation, results are also evaluated in terms of consistency, completeness and robustness. This operation may reveal that a sensitivity analysis is needed for certain parameters or assumptions, to check and/or improve the robustness of the results.

It is worth noticing that LCA is an iterative process, where the findings of a given phase may require a partial revision or modification of the preceding ones. For instance, as data are collected during the inventory analysis and more is learned about the system, new data requirements or limitations may be indentified that require revisions to the goal or scope of the study (e.g., system boundaries). Similarly, impact assessment may reveal that the objectives of the assessment cannot be met, thus needing to be modified (e.g., by excluding a particular impact category). Finally, the interpretation phase may involve the revising of the scope of the LCA, as well as of the nature and quality of the collected data in a way that is consistent with the defined goal.

1.5.5 Waste management LCA

The LCA methodology was originally developed to evaluate the potential environmental and human health impacts of products and processes. However, it has been developed further over the years, so it can be applied to all human activities interacting with the environment. Thus, LCA has been applied extensively not only to products, but also to services, including integrated solid waste management. A relatively recent review of selected waste management LCAs published in scientific journals in the last decade is provided by Cleary (2009).

There are a couple of important methodological differences between traditional product and waste management LCA, which need to be taken into account (Mc Dougall et al., 2001; Coleman et al., 2003). First of all, while in a product LCA the functional unit is generally defined with reference to the output of the studied system (i.e. the product), in a traditional LCA of waste management it is defined with reference to the input of the system, i.e., the waste. In fact, the function of an integrated waste management system is not to produce anything, but to deal with the waste of a given area. Therefore, a typical functional unit in an LCA of waste is “the management of the waste from a defined geographical area” (or one of its inhabitants). Alternatively, “the management of 1 tonne (or 1 kg) of waste", with a composition representative of the studied geographical area, is also frequently used as a functional unit.

A second key difference between product and waste LCA is represented by the system boundaries (Figure 1.4). Generally, in product LCAs the system boundaries include the whole life cycle of a particular product, from raw material extraction, through manufacture, distribution and use, to post-use waste management (cradle-to-grave approach). Indeed, a key aspect of LCA is that product systems should ideally be modelled in such a manner that inputs and outputs at their boundaries are elementary flows (ISO, 2006a; Finnveden, 1999).

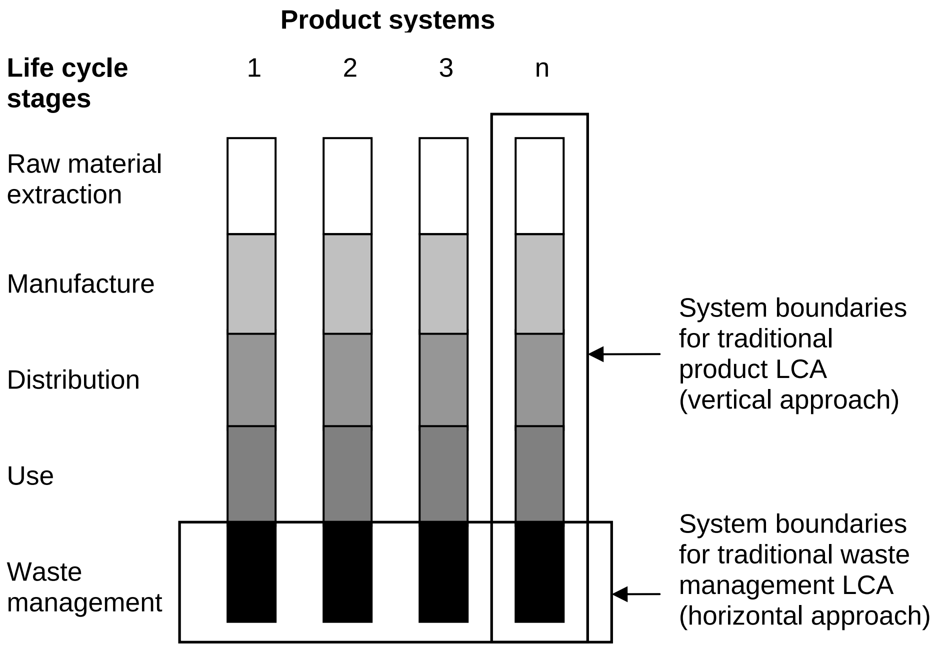


Figure 1.4: differences between the system boundaries of a product and a waste LCA. The former includes the whole life cycle of a particular product or packaging (vertical approach), while a waste LCA includes the waste management stage of every product or packaging discarded in a given area (horizontal approach). The two approaches overlap in the waste management stage of the studied product or packaging. The illustration is partly adapted from Coleman et al. (2003).

In a waste LCA, the system boundaries are instead traditionally included between the moment in which all products and packages used in a given geographical area become waste, by ceasing to have value, to that in which such a waste become an emission to air or water, an inert material in a landfill, or a useful product through a valorisation process (e.g., recycling, composting or incineration)[7]. All life cycle stages prior to the products becoming waste are thus generally excluded from the system boundaries, simplifying the assessment and allowing it to focus on waste management activities only (Cleary, 2010). This curtailment is usually referred to as *zero burden*

[7] If the assessment includes recycling or energy recovery, credits for the avoided environmental burdens of virgin material production or traditional energy generation are generally given. Such avoided production processes are thus generally included in addition in the system boundaries.

approach (or assumption), since the waste is not assigned the burdens and impacts from its previous life (Ekvall et al., 2007). Such an approach is still consistent with the LCA definition, if the same amount and composition of waste appears in all the management systems to be compared. In fact, in this case, all the activities occurring before waste is generated (upstream activities) can reasonably be assumed identical in all systems and, thus, disregarded without affecting their comparison. However, if one of the compared systems produces more or less waste than the others, the zero burden approach is no longer valid. Upstream system boundaries may thus have to be moved and upstream activities included, at least partially (Finnveden, 1999). This happens, for instance, when waste prevention activities are implemented in one of the compared systems. The adjustments needed in this situation will be widely discussed in Section 4, this being one of the main objectives of the PhD research.

2 Life cycle assessment of waste prevention in drinking water consumption

This section summarises the life cycle assessment (LCA) study of the packaging waste prevention activity based on the substitution of bottled water by public network water. This activity is likely one of the most meaningful for Italy, which has for years been the largest per capita consumer of bottled water in Europe, and one of the largest consumer globally (Bevitalia, 2014; Martinelli, 2010). In 2008, the apparent per capita consumption of bottled water amounted to about 190 litres (Bevitalia, 2014) and most of it (about 80%) was packed in one-way polyethylene terephthalate (PET) bottles (Table 2.1)[1]. This consumption is estimated to be responsible for the generation of more than 247,000 tonnes of waste per year (about 5 kg per inhabitant per year, including municipal and commercial waste). An immediate, less waste-generating alternative is however available and represented by public network water, the drinkability of which is a legal requirement (Decreto Legislativo no. 31, 2001).

The LCA was conducted following the general methodological framework outlined in Section 1.5, with the support of the SimaPro software (version 7.3.3). This tool facilitated the creation of a parametric model of eight alternative scenarios for drinking water consumption in Italy and the calculation of the respective potential impacts.

Table 2.1: estimated consumptions of packaged water observed in Italy during 2008 (Personal elaborations on market data available in Bevitalia, 2009).

Type of packaging	Internal (apparent) consumption		
	Million litres	**Litres per capita**	**%**
One-way PET[a] bottles - 2 litres	576.4	9.6	5
One-way PET[a] bottles - 1.5 litres	7,833.6	130.9	68
One-way PET[a] bottles - ≤ 0.5 litres	690	11.5	6
PET bottles - total	*9,100*	*152.1*	*79*
Glass bottles (one-way and refillable)	2,070	34.6	18
PLA[a] bottles, PC[a] and PET jars, brick	350	5.8	3
Total	*11,520*	*192.5*	*100*

(a) PET: polyethylene terephthalate; PC: polycarbonate; PLA: polylactic acid

[1] 2008 market data are reported since they are the most recent that were available at the time of the study and that were thus used for its realisation. However, similar figures are currently available for the 2012 packaged water market. In fact, it was characterised by an overall (apparent) internal consumption of 11.400 millions litres (192 litres per capita) and by the following subdivision among the different types of packages: 80% plastic bottles; 18% glass bottles (one-way and refillable); and 2% jars and bricks.

2.1 Goal and approach of the assessment

The objective of the assessment is to evaluate whether, unconditionally or only under particular conditions, the substitution of bottled water by public network water allows for an actual reduction of waste generation, of the overall impacts on the environment and human health and of the total energy demand. The environmental and energy performances of five baseline scenarios based on the consumption of one-way or refillable bottled water was thus assessed, and compared with those of two waste prevention scenarios, based on the consumption of public network water. However, since the increase of the use of refillable bottled water can be seen as an additional waste prevention activity for Italy[2], refillable bottled water baseline scenarios were also compared with one-way bottled water scenarios, to evaluate the convenience of this substitution. Finally, one-way scenarios were mutually compared in order to identify the possible opportunities to improve the environmental and energy profile of one-way bottled water. A detailed description of the analysed scenarios is provided in the following section.

2.2 Analysed scenarios

Table 2.2 presents a list of the analysed scenarios and of their major features. Four baseline scenarios foreseeing the use of one-way bottler were analysed first. The former (baseline scenario 1) uses virgin PET bottles and is representative of the current situation, since most of one-way water bottles available in the Italian market are made of this material. The other three scenarios foresee the consumption of water packed in 50% recycled PET bottles (baseline scenario2) and polylactic acid (PLA) bottles (baseline scenarios 3a and 3b) and represent two possible alternative scenarios for the Italian context. A recent decree of the Italian Health Ministry (Ministero della Salute, 2010) has indeed introduced the possibility to use up to 50% recycled PET for the manufacturing of mineral water bottles, some example of which are already available in the market. The use of PLA one-way bottles has instead been tested for some years, by one of the major Italian bottling company, to evaluate the suitability of this alternative renewable material (Parola and Aigotti, 2010). As shown in Table 2.2, two different scenarios were considered for the consumption of PLA bottled water, distinguished by the end of life of bottles: composting in one case (baseline scenario 3a) and incineration in the other (baseline scenario 3b). These two treatments are indeed the two currently feasible end-of-life options for post-consumer PLA bottles in Italy.

[2] As it can be inferred from Table 2.1, in Italy refillable (glass) bottled water covers only a small portion of the overall consumption (smaller than 18%). This portion is deemed to be mainly associated with the Ho.Re.Ca. (Hotel, Restaurant and Café) channel.

In this first group of one-way bottled water baseline scenarios, the overall consumption has been split between the most widespread packaging sizes according to 2008 market data reported in Bevitalia (2009), the most recent at the time of the analysis.

A second set of two baseline scenarios based on the use of refillable glass bottled water (baseline scenario 4) and refillable PET bottled water (baseline scenario 5) was then analysed. The use of refillable bottles of two different materials was thus investigated, although refillable PET bottles are not common in Italy (unlike, for instance, Germany). In both scenarios, 1 litre bottles were assumed to be representative of the domestic consumption. Moreover, as a base case, refillable glass bottles were assumed to be used for 10 cycles, while PET ones for 15 cycles.

Table 2.2: alternative scenarios for drinking water consumption analysed in the present study.

<table>
<tr><th colspan="3">Scenario</th><th>Drinking water consumption alternative</th><th>Other relevant features</th><th>Type of packaging</th></tr>
<tr><td rowspan="6">Baseline scenarios</td><td rowspan="4">One-way bottled water scenarios</td><td>Baseline scenario 1</td><td>Use of virgin PET one-way bottled water</td><td>-</td><td rowspan="4">Packaging mix composed of:
• 2 l bottles (6.3%)
• 1.5 l bottles (86.1%)
• 0.5 l bottles (7.6%)</td></tr>
<tr><td>Baseline scenario 2</td><td>Use of 50% recycled PET one-way bottled water</td><td>-</td></tr>
<tr><td>Baseline scenario 3a</td><td>Use of one-way PLA bottled water</td><td>Bottles are sent to composting</td></tr>
<tr><td>Baseline scenario 3b</td><td>Use of one-way PLA bottled water</td><td>Bottles are sent to incineration</td></tr>
<tr><td rowspan="2">Refillable bottled water scenarios</td><td>Waste prevention scenario 4</td><td>Use of refillable glass bottled water</td><td>-</td><td>1 litre glass bottles used 10 times</td></tr>
<tr><td>Waste prevention scenario 5</td><td>Use of refillable PET bottled water</td><td>-</td><td>1 litre PET bottles used 15 times</td></tr>
<tr><td colspan="2" rowspan="3">Waste prevention scenarios

(public network water scenarios)</td><td>Waste prevention scenario 1</td><td>Use of refined groundwater from the tap</td><td>-</td><td>1 reusable glass jug (1 year life span)</td></tr>
<tr><td>Waste prevention scenario 2 (no car)</td><td rowspan="2">Use of refined surface water from public fountains</td><td>No motorised vehicles are used for the roundtrip to the fountain</td><td rowspan="2">1.5 litre one-way PET bottles reused 5 times after the consumption of the water initially packed inside them</td></tr>
<tr><td>Waste prevention scenario 2 (car)</td><td>A private car is used for the roundtrip to the fountain</td></tr>
</table>

Acronyms: PET = polyethylene terephthalate; PLA = Polylactic acid.

The two sets of baseline scenarios were ultimately compared with two alternative waste prevention scenarios, where purified groundwater from the tap (refined at the domestic level - waste prevention scenario 1) and purified surface water from public fountains (refined at the municipal level, waste prevention scenario 2) is respectively used. Both scenarios assume that water is refined by means of proper devices just before withdrawal. In fact, if properly chosen and adequately maintained, they should allow for an improvement of the organoleptic properties of water, the unpleasant taste of which is the main reason why, according to a survey of some years ago (Temporelli and Cassinelli, 2005) Italian citizens tend to prefer bottled mineral water to public network water. In particular, waste prevention scenario 2 aims at the modelling of all those experiences of high quality water delivery from public fountains that are increasingly being implemented in a number of Italian municipalities (e.g. Publiacqua, 2014; CAP Holding, 2014). Two separate cases were considered for this latter scenario. In the former (referred to as “waste prevention scenario 2 - no car”) the roundtrip of the consumer to the public fountain is made without a motorised vehicle. In the latter (referred to as “waste prevention scenario 2 - car”) a private car is used.

2.3 Functional unit

The functional unit used as a reference for the assessment of waste generation and of the environmental and energy performance of the compared scenarios is “*the consumption of 152.1 litres of drinking water by a generic Italian citizen*”. Such an amount represents the estimated volume of one-way PET bottled water consumed in Italy during 2008 (Table 2.1), the most recent data available when carrying out the study.

2.4 System description

This section provides a short description of the two alternative drinking water delivery systems compared in this analysis. For bottled water, the description is mostly based on the evidence gathered during the field surveys at the bottling plant of two different companies located in northern Italy. For public network water, we referred partly to the supplying system of the city of Milan (that relies entirely on groundwater) and Florence (that relies entirely on surface water from the Arno river). Such systems were indeed assumed as a reference for the modelling of the two analysed waste prevention scenarios (see Section 2.7.2.1 for further details).

2.4.1 One-way and refillable bottled water delivery (baseline scenarios)

Bottled waters marketed in Italy are generally collected from natural springs or, less frequently, from the underground aquifer. After collection, water is conveyed to the bottling plant inside steel pipes and then stored inside big cylindrical steel tanks. These are directly connected with one or more automated bottling lines, where water is packed and palletised through a sequence of mechanical operations.

In the case of one-way bottled water, the first operation of the bottling line is bottle forming. It is carried out by stretch-blow moulding of preforms, the compact form of bottles (Figure 2.1). These semi finished products are similar to test tubes and consist of the upper threaded neck, which will not be altered by the forming process, and of a lower tubular part, which will be expanded during the moulding process to acquire the typical shape of a bottle. Preforms are generally manufactured in external plants by moulding of melted plastic granules within injection presses (the so-called injection moulding process). During the bottle forming process, preforms are firstly heated at about 100-120 °C, while revolving round a battery of infrared lamps. They are then transferred inside aluminium moulds where the blowing of compressed air and a stretching bar give them the final shape of bottles. These latter are then filled with water available in the tanks connected to the bottling line, capped with high-density polyethylene (HDPE) caps, provided with a tamper-evident band, and labelled with paper or plastic labels (low-density polyethylene or polypropylene). Filled bottles are then grouped six by six, wrapped round with a low-density polyethylene (LDPE) heat-shrink film and pushed through a heat-shrinking oven. Here, hot air is blown at the temperature of 220-240 °C, making the film shrinking and welding. Typical six-bottle bundles are thus formed, which represent the most popular retail units for one-way bottled water of any size. Bundles are then applied a handle consisting of a transparent adhesive strip coupled with a shorter printed cardboard strip in its central part.

Figure 2.1: example of 1.5 litre PET preform, the semi finished product from which water bottles are manufactured.

For transport to retailers, bundles are finally loaded on multi-layer reusable wooden pallets and wrapped around with linear-low-density polyethylene (LLDPE) stretch film, to keep the whole load stable. Layers are separated by cardboard interlayers, while a covering LDPE film is placed on the top layer, before the wrapping with heat-shrink film. Complete pallets are then stocked or directly loaded on lorries or articulated lorries for transport. At retailers, the stretch film, the top covering film and interlayers are removed, becoming commercial waste. Pallets are instead collected back during the following deliveries and transported again to the bottling plant, where they are reused to build new load units until damaged or broken. Finally, the heat-shrink film of the bundle, labelled bottles and the respective caps are discarded by the consumer as municipal waste and are managed accordingly.

When refillable bottles are used, the bottling line begins with the mechanical depalletisation of crates with returned empty bottles and the removal of possible remaining caps (which are discarded). Bottles are then removed from crates and sent to one or more industrial bottle washers, where a quite articulated sequence of washing, disinfection and rinsing stages takes place. Different hot solutions of water, caustic soda (sodium hydroxide), defoaming additives and descaling agents are used for washing, which also removes labels. Similarly, warm solutions of disinfectant and sequestering agents are used for disinfection. Rinsing is performed with fresh water only.

Even returned crates are mechanically washed in dedicated machines, where a warm solution of water and alkaline detergents is used. After the washing, broken or damaged bottles and crates are discarded, the rest being retained in the bottling line. Washed bottles are thus filled with water, capped with screw or crown aluminium caps (glass bottles) or HDPE screw caps (PET bottles) and finally labelled with paper or plastic labels. In particular, only paper labels are generally used for glass bottles, while both paper and plastic labels can be used for PET bottles (although plastic seems preferred).

For transport purposes, filled bottles are firstly packed inside plastic (HDPE) crates, containing 12 to 20 bottles, depending on size. Crates are then loaded on multi-layer reusable wooden pallets and fastened with a plastic strapping band. Complete pallets are thus transported by lorries to local distributors, where palletised empty bottles from previous deliveries are collected and transported back to the bottling plant, to be reused. The delivery of crates with filled bottles to the households is finally performed by means of small lorries (with a full-load mass generally lower than 7 tonnes). During the delivery trip, crates with empty bottles are also picked up at the households and transported back to local distributors.

2.4.2 Public network water delivery (waste prevention scenarios)

Public network water can be withdrawn from natural springs, from underground aquifers or from surface water, such as rivers and lakes. Generally, spring waters are not subject to any particular purification treatment, but only to sedimentation and fine sifting, carried out directly at the collection wells. Further treatments are seldom needed to reduce the natural concentration of particular substances (such as arsenic) to levels compatible with human consumption.

When water is withdrawn from the underground aquifer, the required treatments depend strongly on its characteristics. In some cases, the sole disinfection is sufficient. In other cases, this operation needs to be preceded by further treatments to remove specific contaminants, or to reduce the natural concentration of particular substances such as iron and manganese. In the case of Milan, groundwater collected through a network of about 550 wells is sent to 29 treatment and pumping stations before its delivery to users. The most popular treatment is activated carbon filtration, which is carried out in 23 out of the 29 stations in order to reduce the concentration of halogenated organic solvents (such as trichloroethylene and tetrachloroethylene) and of pesticides (such as atrazine and 2,6-dichlorobenzamide). In 5 stations, activated carbon filtration is followed by aeration in packed towers, which provides an additional contribution to the removal of chlorinated solvents (by air stripping). In the *Gorla* station, activated carbon filtration in instead followed by a reverse osmosis process, required to reduce nitrates and hexavalent chromium. In all stations, even those without any other particular treatments, water undergoes disinfection to remove possible microbiological contaminations and to prevent their diffusion into the distribution network. In most stations, disinfection is carried out with a solution of sodium hypochlorite. Exceptions are the stations of *Salemi* and *Feltre*, where ultraviolet (UV) irradiation is employed.
Generally, the chemico-physical and microbiological characteristics of surface waters are such that an intense purification process, based on the combination of both physical and chemical treatments, needs to be carried out in a centralised plant. An example is the *Anconella* drinking water treatment plant, which withdraws water from the Arno River and supplies it as purified water to the city of Florence and its suburban area. The treatment chain is quite articulated and includes the following stages: (a) pre-oxidation and pre-disinfection of the raw water with chlorine dioxide (ClO_2) to partially remove oxidable pollutants and reduce algal growing; (b) concurring feeding of powdered activated carbon (PAC) when a sudden increase in organic load occurs; (c) coagulation and flocculation of suspended solids by feeding of poly-aluminium chloride (PACl) and, when required, organic poly-electrolytes, into clarification basins; (d) intermediate oxidation and disinfection with sodium hypochlorite (NaClO) and chlorine dioxide; (e) filtration on quartziferous sand gravity filter to complete the removal of suspended flakes that formed during clarification, but did not settle; (f) ozonisation to achieve a significant reduction of the microbial load, a more complete oxidation of organic pollutants (such as surfactants) and micro pollutants (such as solvents and pesticides) and the improvement of the organoleptic quality of water (smell and taste); (g) filtration on granulated activated carbon (GAC) to remove residual organic compounds and any disinfection by products, as well as to further improve the organoleptic characteristics of water; and (h) post-disinfection with chlorine dioxide, which is fed into the three available compensation basins. Chemical sludge from

the clarification process are treated in a dedicated line. They are firstly homogenised in a unique basin, then thickened in two separate basins where anionic poly-electrolyte is fed, and finally dehydrated in two filter-presses, after the addition of cationic poly-electrolyte. Dehydrated sludge are mostly used for environmental restoration purposes, such as embankments, the rest being landfilled. The reuse rate is about 90%.

After purification through more or less complex processes, water is then introduced into the distribution network by pumping or, less frequently, by gravity. The users can be either private households or public fountains, if the other types of non-domestic users are excluded (e.g., industry, commercial premises, offices etc.). Purified water is thus withdrawn by means of a suitable container (generally jugs or bottles), possibly transported to the point of use, stored for a more or less longer period of time and then consumed. Between one withdrawal and the other, or less frequently, the container is rinsed with water or washed (manually or in a dishwasher).

Sometimes, public network water is characterised by unpleasant organoleptic properties (taste and smell). This is due, for instance, to the use of chlorine-based disinfectants during purification and the distribution within a network that is not always exempt from the penetration of extraneous substances. Particular devices, to be installed upstream the tap, can be used in the attempt to improve water quality. They can also act as an additional barrier against those contaminants that may have not been completely removed during the centralised purification process (e.g., disinfection by-products). Different types of such devices are available, which are based on the individual or combined use of: (a) ion exchange softeners; (b) reverse osmosis processes; (c) mechanical filters; (d) activated carbon filters; (e) composite filters; and (f) ultraviolet irradiation. However, the use of these systems has its own criticalities. The first risk is to apply excessive or, however, unnecessary treatments. For instance, the use of reverse osmosis systems can produce excessively demineralised water, with a saline content close to zero. Another critical aspect is the periodical maintenance of the components subject to wear, which needs to be properly carried out to avoid the risk to worsen water quality rather than improving it. For instance, it is extremely important to comply with the substitution frequency suggested for activated carbon filters, which can become sites for bacterial growth, due to their high porosity. Moreover, uncontrolled releases of adsorbed substances may occur when saturation is achieved.

Systems to improve water quality can be installed at both the domestic level (upstream the household tap) and the municipal level (upstream public fountains). A noteworthy example in this latter respect is the H_2O PLUS system, developed by Publiacqua, the company in charge of the integrated water service of the so-called “Optimal Territorial Area” n. 3 (Medio Valdarno), which includes the Provinces of Florence, Prato, Pistoia and Arezzo. The H_2O PLUS system is integrated in the nearly 60 public fountains that Publiacqua has installed in many pertaining municipalities since 1998, just to promote the use of public network water. Water quality improvement is performed through a sequence of devices, the first of which is an activated carbon cartridge. Possible residues of chlorine or related compounds are thus removed and a further improvement of the organoleptic properties is achieved. Water is then forced through a polyether sulphone (PES) spiral membrane with a molecular cut-off 20,000 Dalton. The membrane retains possible residual suspended substances, without altering the salt content (ionic species are not retained). A 5μm polypropylene filtering cartridge protects the membrane, allowing for the removal of possible coarse particulate substances that could occlude the pores of the membrane. The membrane is followed by an ultraviolet ray lamp, which performs disinfection while avoiding the use of chlorine-based products and related problems (possible formation of by-products and worsening of the organoleptic properties of water). A last passage through an absolute filter with a cut-off of 0.2 μm is finally carried out, to obtain a nearly complete removal of the microbial load. The filter acts indeed as a selective barrier against all those microorganisms that could have exceeded all the previous stages. If required, the incoming water is cooled down to 14 °C in order to limit bacterial growth inside the whole systems, especially the activated carbon filtering cartridge, and to provide more pleasant water to the consumer.

2.5 System boundaries

In baseline scenarios, the system boundaries include the life cycle of primary packages (bottles, caps and labels), of the secondary one (heat-shrink film) and of those used for transport purposes (pallets, stretch film and top-covering film for one-way bottles; pallets, crates and strap band for refillable bottles). The system includes also bottling plant operations, the transport of palletised water to retailers (for one-way bottled water) or local distributors (for refillable bottled water) and from these to the households by means of a private car (one-way bottled water) or small lorries (refillable bottled water).

In the waste prevention scenario using refined groundwater from the tap, the system includes: (a) collection, purification and delivery to users of groundwater; (b) water quality improvement at the domestic level by means of a device based on reverse osmosis[3]; as well as (c) the life cycle of a reusable glass jug used to withdraw water (including its dishwashing). Finally, besides water collection, purification and delivery, in the waste prevention scenario using refined surface water from public fountains, the system includes also: (a) water quality improvement with a dedicated

[3] This particular type of device is chosen since, among the different available options, it is believed to be the one with the highest consumptions of energy and water.

system, incorporated into the fountain; (b) the life cycle of PET bottles used five times to withdraw water at the fountain; and (c) the roundtrip carried out by the citizen with a private car to reach the fountain and transport filled bottles to the household. Simplified representations of the main processes included in the system boundaries in the two sets of baseline scenarios (one-way bottled water and refillable bottled water) and in the waste prevention scenarios, are provided in Figures 2.2 to 2.4.

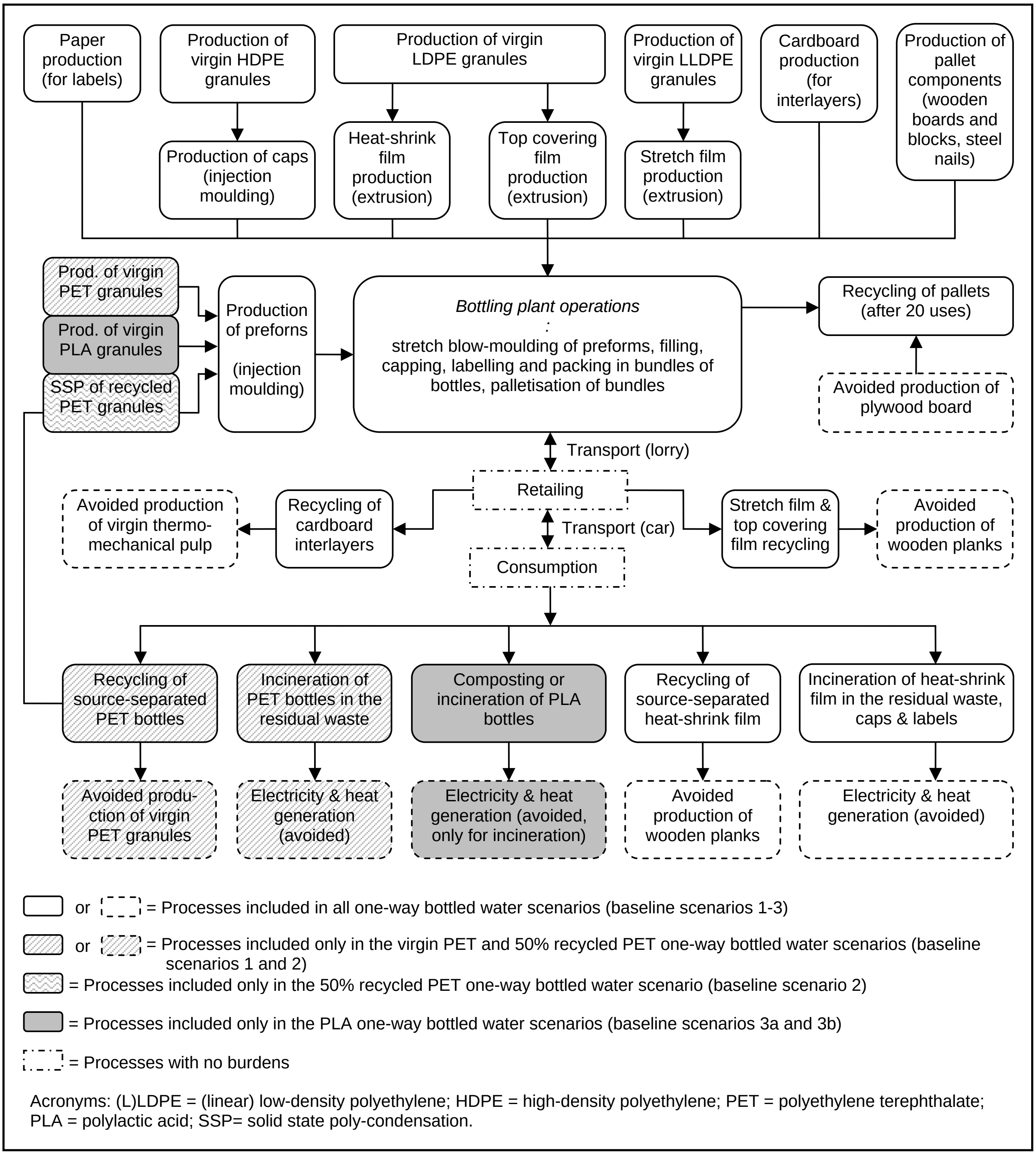


Figure 2.2: main processes included in the system boundaries in one-way bottled water scenarios (baseline scenarios 1 to 3).

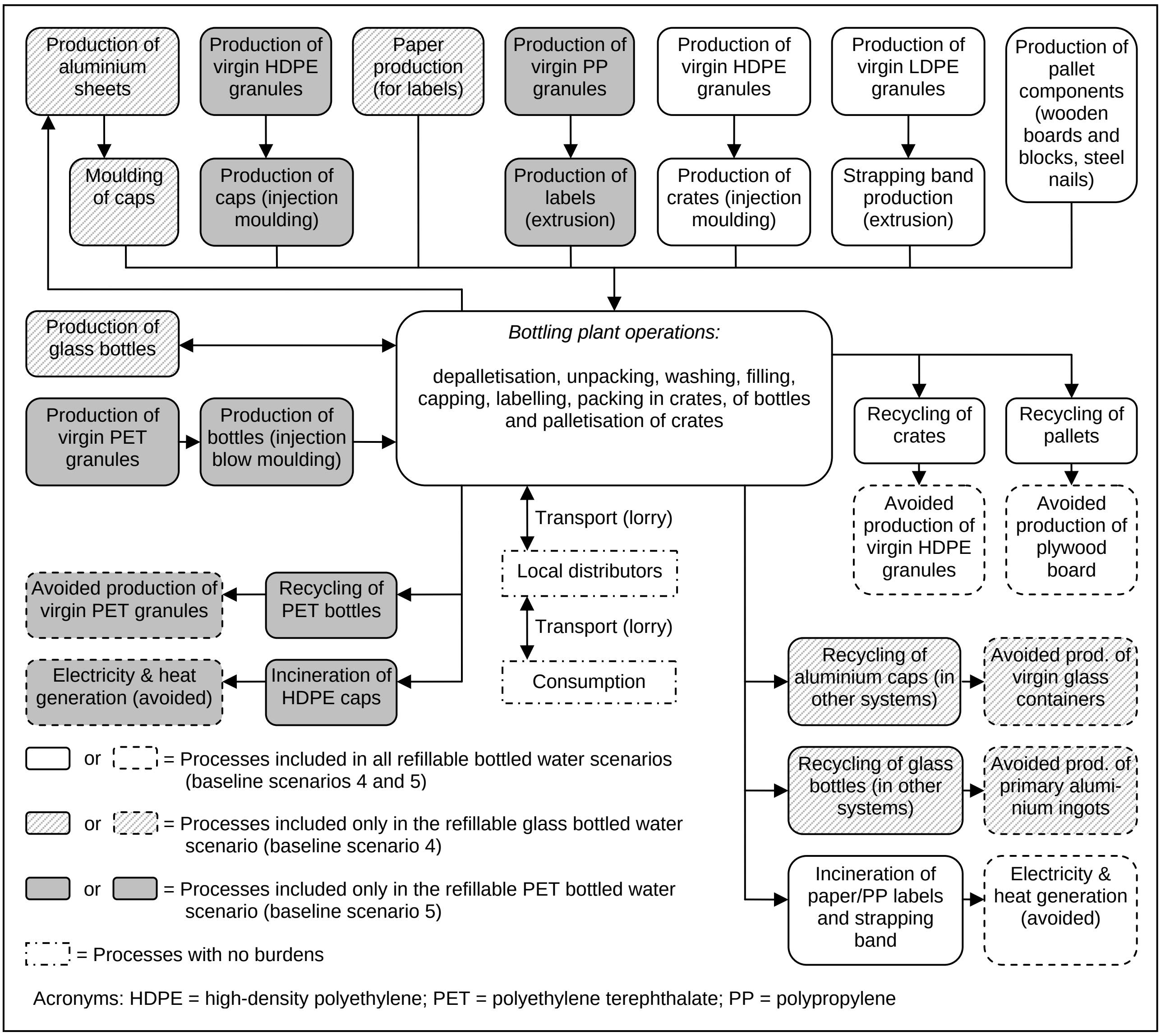


Figure 2.3: main processes included in the system boundaries in refillable bottled water scenarios (baseline scenarios 4 and 5).

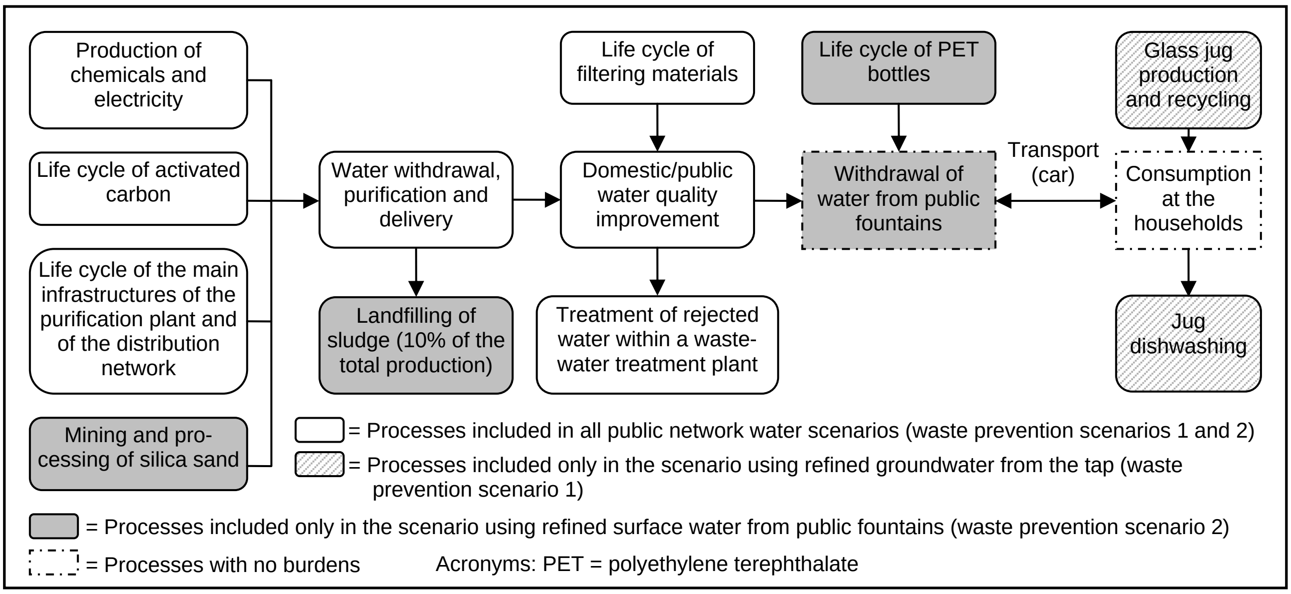


Figure 2.4: main processes included in the system boundaries in public network water scenarios (waste prevention scenarios 1 and 2).

2.6 Impact categories, indicators and characterisation models

According to the goal of the assessment, the generation of waste was calculated first. It includes the waste generated by the consumer at the point of use (municipal waste) and the transport packages discarded at retail establishments or producers (commercial or industrial waste). Manufacturing waste and other possible types of waste generated by retail establishments are instead excluded. Thirteen environmental and human health impact categories, evaluated at the midpoint level, were then considered:

- *climate change*;
- *ozone depletion*;
- *photochemical ozone formation, acidification*;
- *terrestrial eutrophication*;
- *freshwater eutrophication*;
- *marine eutrophication*;
- *freshwater ecotoxicity*;
- *human toxicity (cancer effects)*;
- *human toxicity (non-cancer effects)*;
- *particulate matter*;
- *water resource depletion*;
- *mineral and fossil resource depletion*;

These categories were selected out of those for which a recommended impact assessment model is identified by the European Commission’s Joint Research Centre (JRC) in the framework of the International Reference Life Cycle Data System (EC-JRC, 2011a). The selection was made in the attempt to cover all the potentially relevant environmental issues for the examined product systems. The two impact categories, *ionising radiation (human health)* and *land use* were thus excluded. No processes involving significant emissions of radioactive substances or important changes in land use are indeed included in the studied systems. A list of the midpoint level impact indicators and impact assessment models considered for the selected impact categories is provided in Table 2.3. The cumulative energy demand (CED) indicator was ultimately calculated, according to the method described in Hischier et al. (2010), in order to assess the energy performance of the compared scenarios.

Table 2.3: impact indicators and characterisation models considered for the selected environmental and human health impact categories (adapted, with modifications, from EC-JRC, 2011a).

Impact categories	**Midpoint level impact indicators**[a]	**Characterisation models**
Climate change	Increase of infrared radiative forcing [kg CO_2 equivalent]	Baseline model of the IPCC[b] defining the global warming potential of different greenhouse gases over a 100 year time horizon (IPCC, 2007)
Ozone depletion	Increase of stratospheric ozone breakdown [kg CFC-11[c] equivalent]	Model developed by the WMO[d] defining the ozone depletion potential for different ozone-depleting gases over an infinite time horizon (WMO, 1999)
Photochemical ozone formation	Tropospheric ozone concentration increase [kg NMVOC[e] equivalent]	LOTOS-EUROS model (Van Zelm et al., 2008) as applied in ReCiPe
Acidification	Accumulated exceedance of the critical load [mol H^+ equivalent]	Accumulated Exceedance model (Seppälä et al., 2006; Posch et al., 2008)
Terrestrial eutrophication	Accumulated exceedance of the critical load [mol N equivalent]	Accumulated Exceedance model (Seppälä et al., 2006; Posch et al., 2008)
Freshwater eutrophication	Fraction of nutrients reaching freshwater end compartment [kg P equivalent]	EUTREND model as implemented in ReCiPe (Struijs et al., 2009)
Marine eutrophication	Fraction of nutrients reaching marine end compartment [kg N equivalent]	EUTREND model as implemented in ReCiPe (Struijs et al., 2009)
Freshwater ecotoxicity	Comparative Toxic Unit for ecosystems [CTUe]	USEtox model (Rosenbaum et al., 2008)
Human toxicity (cancer effects)	Comparative Toxic Unit for humans [CTUh]	USEtox model (Rosenbaum et al., 2008)
Human toxicity (non-cancer effects)	Comparative Toxic Unit for humans [CTUh]	USEtox model (Rosenbaum et al., 2008)
Particulate matter	Intake fraction for fine particles [kg $PM_{2.5}$[f] equivalent]	RiskPoll model (Rabl and Spadaro, 2004; Greco et al., 2007)
Water resource depletion	Water use related to local scarcity of water [m^3 of water equivalent]	Ecological scarcity model (Frischknecht et al., 2009)
Mineral and fossil resource depletion	Resource extraction related to ultimate reserves and annual extraction rates[g] [kg Sb equivalent]	CML 2002 model (Guinée et al., 2002) as updated in Van Oers et al. (2002)

(a) The unit of measure of category indicator results is also reported in square brackets.
(b) IPCC: Intergovernmental Panel on Climate Change.
(c) CFC-11: trichlorofluoromethane, also called freon-11 or R-11, is a chlorofluorocarbon.
(d) WMO: World Meteorological Organization.
(e) NMVOC: Non-Methane Volatile Organic Compounds.
(f) $PM_{2.5}$: particulate matter with a diameter of 2,5 μm or less.
(g) Although the characterisation factors calculated as a function of the "reserve base" of resources are recommended by ILCD (EC-JRC, 2011a), those based on "ultimate reserves" are used in this assessment. The latter were indeed deemed more appropriate, since no uncertainties associated with considerations on technical and economical availability of resources are introduced in their estimate. Moreover, the baseline version of the recommended characterisation model (CML 2002) is just based on ultimate reserve.

2.7 Modelling of scenarios

This section summarises the approach used in the modelling of the different life cycle stages included in the system boundaries in the compared scenarios. In particular, input data are described and inventory data sources reported. Input data are the parameters and the assumptions used to calculate the quantity of each unit process included in the system (e.g. bottle mass or travelled distances). Inventory data define the exchanges of material and energy between a unit process and the environment or other unit processes. Based on these input and inventory data, a virtual model of each scenario was implemented in the SimaPro LCA software. Some input data were included into the models in the form of parameters, to allow a sensitivity analysis to be conducted on them (Section 2.8). The unit processes included into each model, the quantities required of these processes, and the respective source of inventory data are listed in Tables A.4 to A.10 of Appendix A.

2.7.1 Baseline scenarios (use of one-way or refillable bottled water)

2.7.1.1 Life cycle of primary, secondary and transport packages

The most important input data required to model the life cycle of the primary, secondary and transport packages used for the distribution of one-way and refillable bottled water are: (a) their average masses; (b) the number of items (e.g., filled bottles) included in each secondary packaging (e.g., bundles); (c) the average number of uses of reusable packages (e.g., crates and pallets); as well as (d) the average composition of pallets[4]. All these parameters were defined as described in Tables A.1 to A.3 of Appendix A, thus completing the information provided in Table 2.2.

Except for one-way PET bottles in baseline scenario 2, all primary, secondary and transport packages were assumed to be produced from virgin material, according to current practices surveyed. Regarding the end of life of primary packages, 77% of PET one-way bottles were assumed to be separately collected and mechanically recycled, the rest being incinerated in a waste to energy plant as residual waste. This assumption is based on the percentage of containers for liquids sent to recycling in Italy in 2009, estimated from the data available in Corepla (2010). PET bottle recycling allows for the production of secondary, amorphous, PET granules that in baseline scenario 2 are partly used in the system for bottle manufacturing, after being subject to a solid state poly-condensation (SSP) process[5]. As anticipated in Section 2.2, the two currently available end-of-

[4] A pallet composition indicates how many bundles or crates are included in each layer of the particular pallet and the total number of layers, so that the overall amount of drinking water charged on the pallet can be calculated.

[5] In the solid state poly-condensation process, recycled PET granules are heated at temperatures below the melting point, to increase the intrinsic viscosity of the material to levels compatible with the injection moulding process and to remove any residual organic contamination from preceding stages (Culbert and Christel, 2003).

life options for post-consumer PLA one-way bottles were separately considered: composting with source-separated organic waste (baseline scenario 3a) and incineration as residual waste (baseline scenario 3b). Refillable glass and PET bottles were instead assumed to be totally recycled, after being rejected at the bottling plant due to excessive damages or breakages. HDPE caps of one-way bottles and labels of all types of bottles are entirely incinerated, after being possibly rejected during sorting of separately collected plastic waste or during bottle recycling. Aluminium caps of refillable glass bottles and HDPE caps of refillable PET bottles were instead assumed to be totally recycled, once being removed from bottles at the packaging plant.

As for secondary packages, 33% of the heat-shrink film of bundles was assumed to be separately collected and recycled, the rest being incinerated in a waste to energy plant[6]. Most of transport packages (i.e., HDPE crates, pallets, interlayers, the stretch film and the top covering film) were instead assumed to be entirely recycled, becoming waste nearby commercial premises or at the bottling plant. The unique exception is the strapping band fastening reusable crates on pallets, which was assumed to be incinerated.

Inventory data on primary production processes of packaging materials and their subsequent conversion processes into finished products were derived from the *ecoinvent* database (v. 2.2.). For recycling processes, the data and recovery efficiencies suggested by Rigamonti and Grosso (2009) were used. An exception is the recovery efficiency of PET bottles, which was assumed equal to 80%, according to Li et al. (2010). The solid state poly-condensation process of recycled PET granules was instead modelled based on consumption data relating to the *viscoSTAR* technology, developed by the Austrian company *Starlinger*. For the incineration of packaging materials, specific inventories were developed based on the process carried out in a real waste to energy plant located in northern Italy (Turconi et al., 2011). However, waste specific features were taken into account to define airborne emissions, reagent consumptions, production of residues (bottom ashes) and the generated amount of electricity and heat.

Finally, regarding the industrial composting of PLA bottles, they were assumed to be mixed with organic waste in a ratio of 0.3:1, according to the experience of Ghorpade et al. (2001). Moreover, a complete degradation under aerobic conditions was reasonably assumed to be achieved, for bottles, at the end of the process. As a consequence, no compost is obtained from bottles and no methane emissions are generated from their degradation. Based on these assumptions, composting of PLA bottles was thus assigned the following burdens: (a) process specific consumptions of energy and water; (b) biogenic CO_2 emissions and leachate production (estimated based on the elemental composition of PLA, available from NatureWorks (2012b); and (c) the burdens associated with

[6] The assumption is based on the overall recycling rate of plastic packages achieved in Italy during 2009, estimated from data available in Corepla (2010).

composting of the added organic waste (2.33 kg per kg of bottles) and the benefits from the subsequent use of the obtained compost in substitution of peat and mineral fertilisers. Process specific burdens and those associated with organic waste composting were defined based on the process carried out in a real plant located in the north of Italy, so as reported in Punzi (2009).

2.7.1.2 Bottling plant operations

As explained in detail in Section 2.4.1, the operations carried out at the bottling plant include filling, packing and palletisation of bottles, as well as their washing (refillable bottles) and forming (one-way bottles). Such operations were modelled based on primary data relating to a medium sized bottling company located in northern Italy, which uses both one-way PET and refillable glass bottles for packaging purposes. The modelling included the following aspects: (a) consumptions of electricity and energy carriers (e.g. natural gas) for all the mentioned operations; (b) the life cycle of lubricating oil used for machinery maintenance; (c) consumptions of water, chemicals and detergents for the washing of bottles and the periodical cleaning of the bottling line; and (d) the treatment of wastewaters from these washing and cleaning operations.

Inventory data available in the *ecoinvent* database (v 2.2) were used for the production processes of the material and energy flows involved, for the end of life of lubricating oil and for wastewater treatment. However, for lubricating oil incineration, the avoided production of electricity and heat is added to the original dataset.

2.7.1.3 Transport to retailers or local distributors

Based on the distribution on the national territory of the bottling plants of the major brands of bottled water marketed in Italy, palletised bottles were assumed to be transported to retailers (one-way bottles) or local distributors (refillable bottles) along an overall average distance of 300 km. The return trip with empty pallets or pallets charged with empty bottles was also taken into account. Due to the great variability of the distance (from about 40 km to about 800 km), a sensitivity analysis was performed (Section 2.8).

Inventory data on the transport by lorries with a full load mass larger than 16 tonnes were derived from the *ecoinvent* database (v 2.2).

2.7.1.4 Transport to the point of use

As for one-way bottled water, a roundtrip distance of 10 km was assumed to be covered with a private car by the consumer to purchase a typical bundle containing six one-way bottles by 1.5 litres. Moreover, an overall purchase of 30 items was assumed. Therefore, only 1/30 of the overall burdens of the roundtrip were actually allocated to this life cycle stage. Being the number of items purchased contemporarily an arbitrary assumption, a sensitivity analysis was performed at the modelling level, to evaluate the effects on the overall impacts of one-way bottled water scenarios. In particular, the scenario using virgin PET one-way bottles was used as a reference, in the case in which water is transported to retailers along a distance of both 300 km and 40 km (Figure 2.5). For 300 km, no meaningful variations in the impacts are observed, neither for an overall purchase of 60 items (improved purchasing behaviour) nor for 15 items (worsened behaviour). An increase larger than 10% for at least one impact category is observed only for purchases lower than 11 items. For a distance of 40 km, an impact increase larger than 10% for at least one category takes place when less than 15 items are contemporarily purchased, although the overall picture is similar.

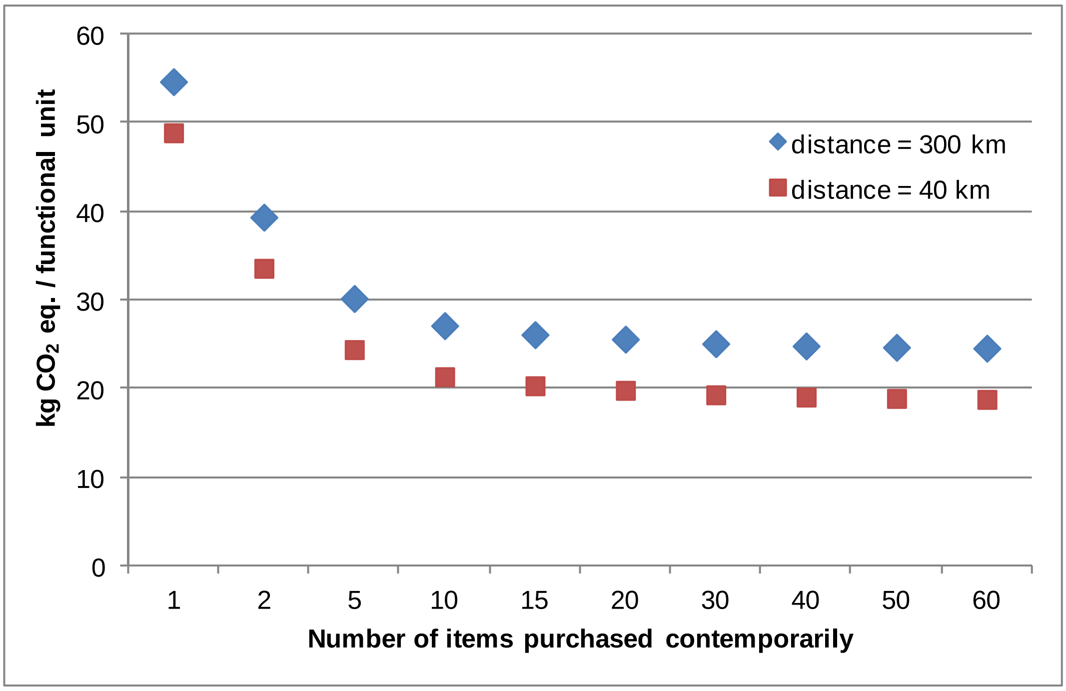


Figure 2.5: variation of the climate change impact indicator as a function of the number of items purchased contemporarily at the retail store, under the virgin PET one-way bottled water baseline scenario (baseline scenario 1).

In the refillable bottled water scenarios, returnable crates containing filled bottles were assumed to be transported along an overall distance of 20 km, by local distributors, for the delivery to the households. Here, crates containing empty bottles from previous deliveries are collected and transported back along the same distance during the return trip.

The inventories of the purchasing roundtrip by car and of the delivery trip by mean of lorries with a full load mass lower than 3.5 tonnes were compiled based on data available in the *ecoinvent* database (v 2.2).

2.7.2 Waste prevention scenarios (use of public network water)

2.7.2.1 Water withdrawal, purification and delivery to users

For waste prevention scenario 1, the sages of withdrawal, purification and delivery of groundwater were mainly modelled based on the features and primary data relating to the drinking water supply system of the city of Milan. It relies entirely on water withdrawn from the underground aquifer through a network of about 550 wells and then purified through diverse techniques such as activated carbon filtration, aeration, reverse osmosis and disinfection. Such treatments are carried out both individually (especially disinfection) or in a particular sequence. The modelling included: (a) water consumptions; (b) the production of electricity and chemicals; (c) the life cycle of granulated activated carbon; and (d) the life cycle of major infrastructures (activated carbon filters, aeration towers, pumping stations, reservoirs and the supply network).

For waste prevention scenario 2 (which uses surface water), the modelling was mainly made with reference to the features and the primary data relating to the drinking water supply system of the city of Florence and its suburban area. These municipalities are entirely supplied with water withdrawn from the Arno River and purified through an intense process in the Anconella plant, where both physical and chemical treatments are combined (see Section 2.4.2 for further details). Similarly to groundwater, the modelling included: (a) the consumptions of water; (b) the production of electricity, chemicals and silica sand; (c) the life cycle of activated carbon; and (d) the disposal into an inert material landfill of the small portion of sludge that is not used for any specific application. For comparability reasons, infrastructures were assumed to be identical to those considered for groundwater, since no specific estimations were carried out in this case.

Inventory data on the production of energy, chemicals, activated carbon and silica sand were derived from the *ecoinvent* database, the I-LCA database by ANPA[7] (activated carbon); the technical literature (e.g., for sodium chlorite) or directly from the operators of real manufacturing plants (e.g., for poly-aluminium chloride). For the landfilling of slag we referred once again to *ecoinvent*, while for the reactivation of activated carbon a new unit process was designed based on the data available in the environmental declaration of a real Italian plant (SICAV, 2009).

2.7.2.2 Water quality improvement (refining)

In waste prevention scenario 1, the stage of water quality improvement at the domestic level was modelled based on the average features of three devices based on activated carbon filtration and reverse osmosis. This technology is indeed deemed to be the most energy and water demanding

[7] ANPA (now ISPRA) was the National Agency for Environmental Protection, which in 2000 developed a publicly accessible database to support the realisation of LCA studies. For activated carbon manufacturing, a dataset depicting the production of carbon coke was specifically used as an approximation.

among the several options available to refine water quality. Other than electricity and water consumptions, the modelling included the life cycle of the annually replaced activated carbon filter and the treatment of water rejected by reverse osmosis, in a wastewater treatment plant[8]. The volume of water filtered over one year by the device was conservatively assumed to be the same as the one considered in the functional unit (152.1 litres).

The improvement of water quality carried out at public fountains (waste prevention scenario 2), was modelled based on the features of the H_2O plus system, described in Section 2.4.2. Even in this case, beyond water and electricity consumptions, we have also considered the life cycle of the activated carbon filter and of the polypropylene filtering cartridge. These components are indeed those with the highest frequencies of substitution (equal to about 2 and 6 months, respectively). The life cycle of the filter includes the manufacturing and the landfilling of the activated carbon. Conversely, the life cycle of the cartridge includes the manufacturing of a generic extruded polypropylene item and its incineration in a waste to energy plant. The treatment of the water rejected during the whole refining stage (considered as an unpolluted sewage) is finally taken into account.

Inventory data available in the *ecoinvent* database were used for most of the unit processes relating to the stage of water quality improvement. Exceptions are activated carbon manufacturing and polypropylene incineration. For the carbon, a dataset on carbon coke manufacturing from the I-LCA (Italian LCA) database by ANPA was used. For the incineration process, a new dataset was designed, as briefly explained in Section 2.7.1.1 for packaging materials.

2.7.2.3 Life cycle of containers

The 1 litre glass jug used to withdraw groundwater from the household tap in waste prevention scenario 1 was assigned a mass of 475 g, corresponding to the average mass of 1 litre refillable glass bottles used in baseline scenario 4. Moreover, a volume of water equal to the one considered in the functional (152.1 litres) was conservatively assumed to be withdrawn with the jug, during its whole life cycle, which ends with recycling (crushing and re-melting with virgin materials in a generic hollow glass container). Finally, as a base case, the jug was arbitrarily assumed to be washed in a dishwasher after every 4 uses as part of an overall load of 30 items. Since these parameters depend exclusively on the behaviour of the consumer, they were subject to sensitivity analysis (Section 2.8). The burdens of the washing stage were defined based on the average consumptions of water and electricity of more than 630 Energy Star qualified dishwashers (Energy Star, 2012). The consumption of detergent and the resulting waterborne emissions were instead not

[8] The life cycle of the filter includes the manufacturing of the activated carbon and its disposal into an inert material landfill. Moreover, for modelling purposes, rejected water is considered an unpolluted sewage.

quantified. However, the treatment of dishwashing waters (considered as an unpolluted sewage) in a wastewater treatment plant, was taken into account.

As for 1.5 litre PET bottles used to withdraw surface water from the public fountain (waste prevention scenario 2), an average mass equal to the one of one-way bottles used in baseline scenarios 1-3 was considered (32,5 grams). It is indeed likely that the withdrawal is made by means of one-way water bottles previously purchased by the consumer. However, for simplicity, no burdens from their preceding life cycle were shared with the portion spent in the present system. Bottles were thus only assumed to be used directly for 5 withdrawals and then mechanically recycled (production of secondary PET granules). Even HDPE caps were assigned the same mass as those of 1,5 litre one-way bottles used in baseline scenarios (2,06 grams). As in such scenarios, caps were assumed to be incinerated, after being rejected during sorting or recycling of separately collected bottles.

2.7.2.4 Transport to the point of use

When a private car is used to reach public fountains and transport withdrawn water to the point of use (waste prevention scenario 2 - car), the impacts associated with the roundtrip were taken into account. Based on the location of the public fountains available in the city of Florence, an overall distance of 5.5 km was thus initially assumed to be covered by citizens. A sensitivity analysis was then performed on this roundtrip distance (Section 2.8). As a base case, 9 litres of water were assumed to be withdrawn during each trip to the fountain (a volume corresponding to the filling of 6 bottles). Since this parameter depends heavily on the behaviour of the consumer, it was subject to a sensitivity analysis (Section 2.8).

2.7.3 Modelling of recycling

Product recycling allows for the production of secondary goods (raw materials or products), which are generally used in the studied product system or other systems in place of virgin raw materials or products (primary goods). In this study, product recycling was modelled according to the so-called *recyclability substitution approach* (EC-JRC, 2010a), which is more commonly known as *avoided burden approach*. Therefore, when a recycled good was used directly in the studied system, the net consumption of the corresponding primary good was decreased accordingly. When the use of a recycled good was made in other systems, the avoided burdens of the primary production of substituted goods were instead credited to the system. In particular, the “average primary production mix” was credited, according to the attributional approach (EC-JRC, 2010a). When the recycled good had a lower quality than the substituted primary good, the primary production of a lower amount was credited. Since the amount of product actually substituted was unknown, the substitution factors provided in Rigamonti and Grosso (2009) were adopted in the calculation. These factors take into account the difference in the market value of the recycled and the virgin products (for plastics), or in their inherent technical properties (for paper and wood).

2.8 Sensitivity analysis

A sensitivity analysis was singularly performed on those parameters affected by greater variability or uncertainty and that were expected to have a meaningful influence on the results. Moreover, the selected parameters were also combined in an attempt to define an upper and a lower boundary of the impacts of the analysed scenarios (i.e., a worst and a best case of such scenarios). Sensitivity parameters, their values and the way in which they were combined are described in Table 2.4.

Table 2.4: variations considered for the parameters subject to sensitivity analysis, in order to define a potential upper and lower boundary of the impacts of each analysed scenarios.

Scenario	Parameter	Best case	Base case	Worst case
Use of one-way and refillable bottled water (baseline scenarios 1-5)	Distance separating bottling plants from retailers or local distributors	40 km	300 km	800 km
Use of refillable bottled water (baseline scenarios 4-5)	Number of uses for refillable bottles	25[a]	Glass bottles: 10 PET bottles: 15	-
Use of refined groundwater from the tap (waste prevention scenario 1)	Frequency of washing of the jug used to withdraw water and number of items contemporarily washed in the dishwasher[b]	Washing after every 5 uses, as part of a load of 50 items	Washing after every 4 uses, as part of a load of 30 items	Washing after every use, as part of a load of 15 items
Use of refined surface water from public fountains (waste prevention scenario 2 - car)	Overall roundtrip distance covered by car to withdraw water at the fountain	2 km	5.5 km	10 km[c]
	Volume of water withdrawn at the fountain[d]	18 litres	9 litres	4.5 litres

Acronyms: PET = polyethylene terephthtalate.
(a) According to Co.Re.Ve (2013), nowadays refillable glass bottles are usually designed for a maximum of 25-30 uses. Similarly, the highest number of uses found in the literature for refillable PET bottles is equal to 25 (GDB, 2012).
(b) The impacts of the washing stage depend on these two parameters: the highest are the washing frequency and the number of items contemporarily washed, the lowest are the impacts. Since only arbitrary assumptions can be performed on these parameters, which depend exclusively on the behaviour of the consumer, they were subject to a sensitivity analysis.
(c) This worst case of the scenario aims at the modelling of the extreme situation where a citizen goes to the fountain available in a surrounding municipality to withdraw water.
(d) The impacts of the roundtrip performed by car to withdraw water at the fountain depend on the volume of transported water, which in turn depends on the behaviour of the consumer. The impacts of the scenario were thus recalculated both for the case where the consumer attempt to maximise the volume of withdrawn water (18 litres) and the one where only a limited volume of water is withdrawn (4.5 litres).

First of all, in baseline scenarios, the distance along which bottled water is transported to retailers or local distributors was varied, according to the minimum and maximum values estimated for the Italian context. Moreover, an increased number of uses was considered for bottles in refillable bottled water scenarios. Regarding public network water, the reusable glass jug used to withdraw refined groundwater from the tap was assumed to be washed under both more efficient criteria (after more uses and along with more items) and quite inefficient criteria (after less uses and along with fewer items). Finally, in waste prevention scenario 2, the roundtrip to the fountain by car was assumed to be performed both under improved conditions (a shorter distance is travelled to withdraw a higher volume of water) and under worsened conditions (a longer distance is travelled to withdraw a lower volume of water).

2.9 Results and discussion

2.9.1 Waste generation

Figure 2.6 shows the amount of waste generated, while Table 2.5 reports the differences between waste prevention and baseline scenarios.

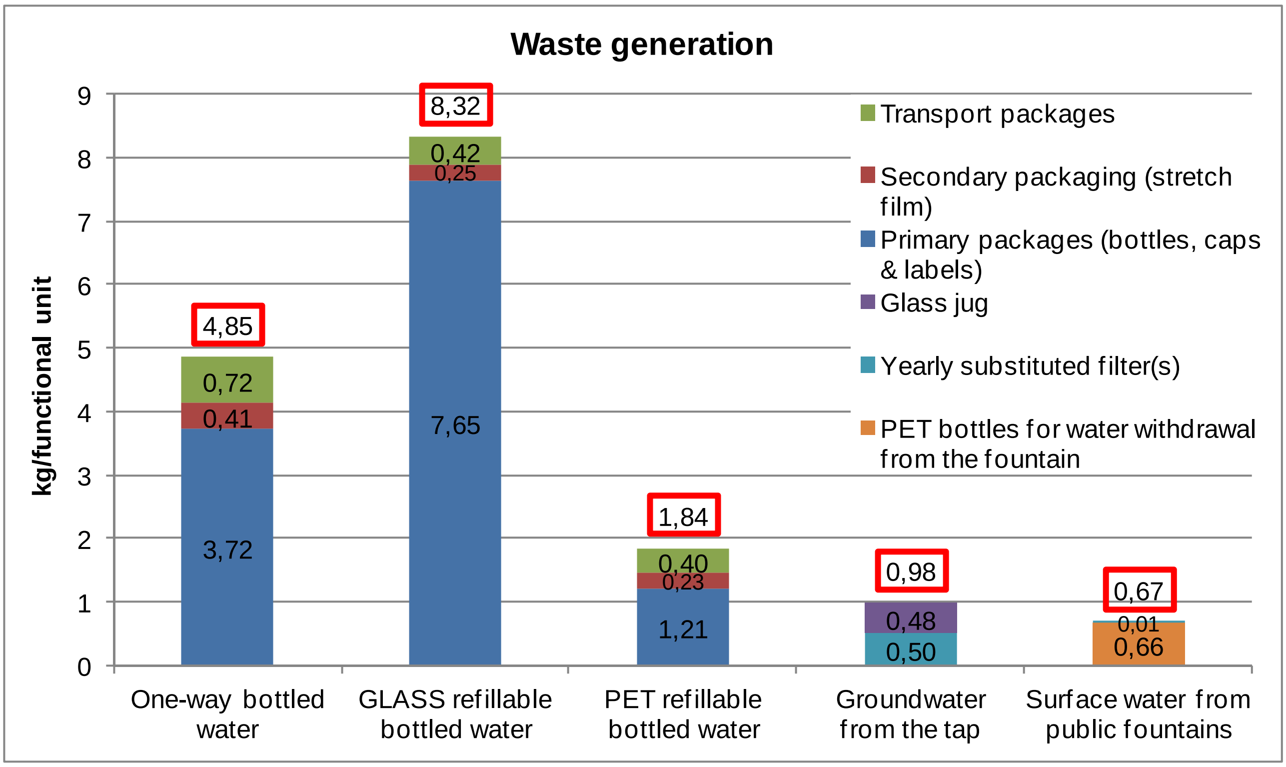


Figure 2.6: amount of waste generated in the analysed baseline and waste prevention scenarios.

The use of refined public network water from the tap or public fountains allows for an important reduction in waste generation compared to one-way bottled water (83% on average). A significant reduction (90% on average) is also achieved compared to glass refillable bottled water, the baseline alternative generating most waste. Compared to PET refillable bottled water, which is the least waste-generating baseline alternative, the reduction is instead lower (55% on average), but still significant.

The two waste prevention scenarios are almost comparable, with a slight advantage for the one where water is withdrawn from public fountains (waste prevention scenario 2). We must however remember that the estimate provided for the waste prevention scenario 1 (withdrawal from the tap) is quite conservative. It has indeed been made under the assumption that the glass jug is replaced annually. On the other hand, it is however clear that an inefficient use of the container could reduce its useful life and consequently increase waste generation up to values comparable with baseline scenarios (or even larger). This consideration is certainly valid also for waste prevention scenario 2.

Finally, the results show that a possible substitution of one-way bottles by refillable glass bottles would not allow for a reduction in the mass of generated waste, because of the higher specific mass of glass compared to PET or PLA. Only the number of items that eventually become waste is reduced with this consumption alternative. Conversely, the use of refillable PET bottles would allow for a quite important reduction in waste mass (about 62%), proving to be the less-waste generating option for packaged water distribution.

Table 2.5: difference between the amounts of waste generated in waste prevention and baseline scenarios.

Waste prevention scenario	**Reference baseline scenario**		
	One-way bottled water	GLASS refillable bottled water	PET refillable bottled water
Refined groundwater from the tap (waste prevention scenario 1)	-3.88 kg/fu[a] (-79.9%)	-7.35 kg/fu (-88.3%)	-0.87 kg/fu (-47.0%)
Refined surface water from public fountains (waste prevention scenario 2)	-4.18 kg/fu (-86.2%)	-7.65 kg/fu (-92.0%)	-1.17 kg/fu (-63.7%)

(a) functional unit.

2.9.2 Impact assessment results

The *climate change* impact indicator of all the analysed scenarios is shown in the upper part of Figure 2.7, as an example of the common profile characterising most indicators. An exception is the *water resource depletion* indicator, represented in the lower part of Figure 2.7. For this indicator, the contribution of water quality improvement to the overall potential impact of waste prevention scenario 1 is highlighted, to avoid misleading interpretations. The profiles of the remaining indicators are available in Figures A.1 to A.6 of Appendix A.

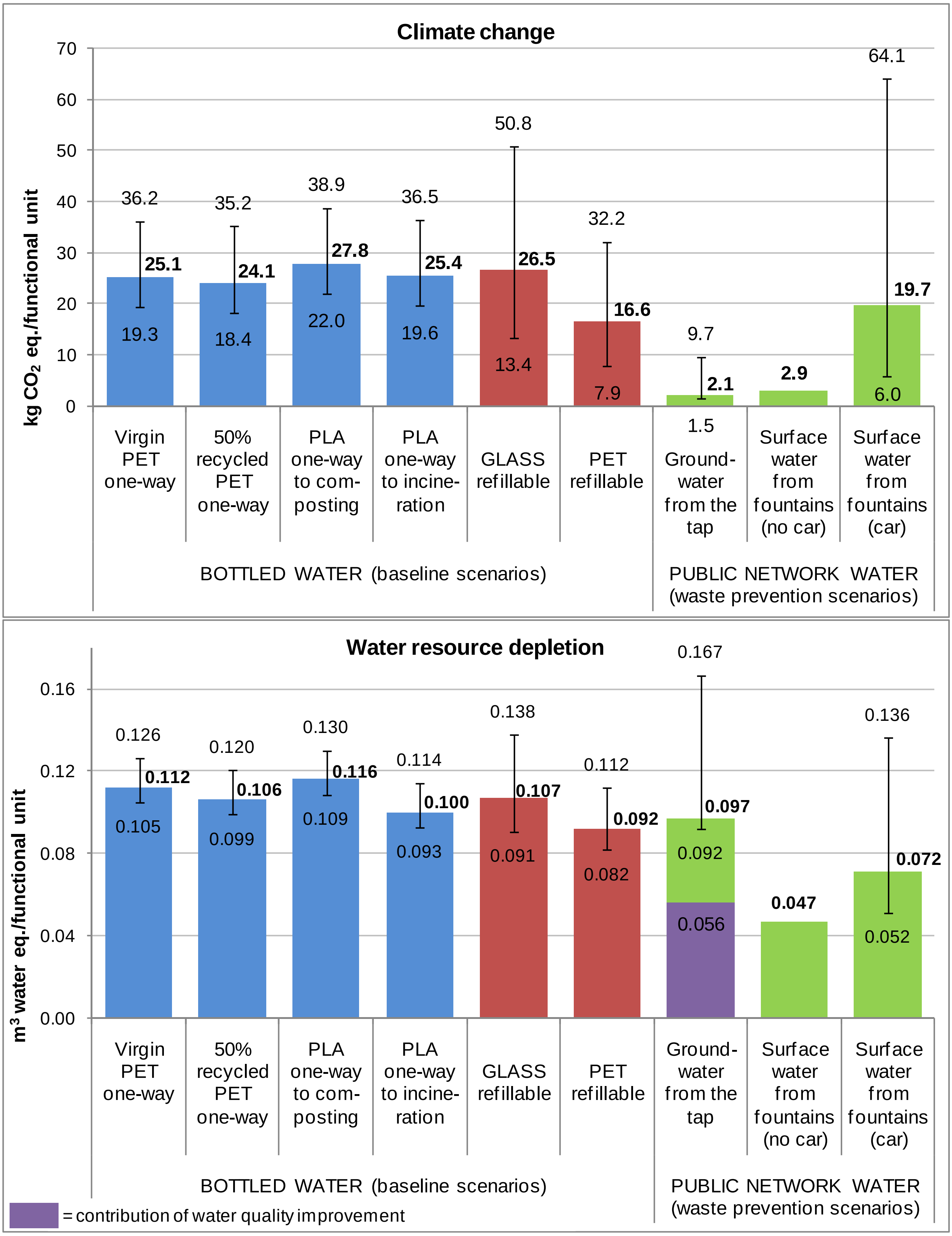


Figure 2.7: potential impacts of the analysed baseline and waste prevention scenarios, for the *climate change* and *water resource depletion* impact categories. For each scenario, the main bar represents the base case, while the error bar shows the upper and lower boundaries resulting from the variation of the sensitivity parameters described in Table 2.4.

According to the goal of the study (Section 2.1), the following sections discuss separately the results obtained for the three main groups of analysed scenarios (one-way bottled water, refillable bottled water and public network water). In particular, Section 2.9.2.1 compares waste prevention with baseline scenarios, while Section 2.9.2.2 compares refillable bottled water with one-way bottled water baseline scenarios. Finally, one-way bottled water scenarios are mutually compared in Section 2.9.2.3.

2.9.2.1 Public network water versus bottled water scenarios

For an easier understanding of the results, the following comparative considerations initially exclude the *water resource depletion* indicator, which will be discussed later in this section.

When the reusable glass jug is washed under average conditions (every four uses in a load of 30 items), drinking of refined groundwater from the household tap (waste prevention scenario 1) is definitely preferable to both one-way and refillable bottled water[9]. This even when this latter is transported to retailers or local distributors along a short distance (40 km) and refillable bottles are used 25 times (best cases of baseline scenarios). As it can be easily inferred from Figure 2.7, the same consideration would be valid also if surface water was supplied and withdrawn from the household tap. Thus, the origin of water does not significantly affect the comparison, although groundwater shows slightly better performance (less chemicals and activated carbon are used, compensating for the higher electricity consumption). Table 2.6 reports the lowest impact reductions achievable by substituting bottled water by refined tap (ground)-water, when average washing conditions are considered. Such reductions are observed when bottled water is transported along a short distance of 40 km, and refillable bottles are used 25 times. The highest reductions are instead achieved when a distance of 800 km is considered, and refillable bottles are used 10 or 15 times, as reported for completeness in Table A.11 of Appendix A.

[9] Due to uncertainties included in any LCA, a drinking water consumption alternative (scenario) was considered to be preferable to another one, only when an impact reduction larger than 10% takes place between the two scenarios, for a given impact category. For impact variations lower than ±10% the two scenarios were thus considered to be comparable, being the achieved variation insignificant (statistically speaking).

Table 2.6: impact reductions resulting from the substitution of refined groundwater from the tap (waste prevention scenario 1) for the different types of bottled water, when the reusable jug used to withdraw tap water is washed under average conditions (after every 4 uses in a load of 30 items) and bottles are transported to retailers or local distributors along a distance of 40 km (best case of baseline scenarios).

Impact categories	Reference baseline scenario					
	Virgin PET one-way bottles	**50% recy-cled PET one-way bottles**	**PLA one-way bottles to composting**	**PLA one-way bottles to incineration**	**Refillable glass bottles[a]**	**Refillable PET bottles[a]**
Climate change	-89.3%	-88.8%	-90.6%	-89.5%	-84.6%	-73.7%
Ozone depletion	-92.4%	-92.3%	-93.7%	-93.3%	-83.3%	-77.7%
Photochemical ozone formation	-85.6%	-84.8%	-88.8%	-87.7%	-86.9%	-78.9%
Acidification	-79.3%	-77.9%	-85.7%	-83.5%	-81.5%	-60.8%
Terrestrial eutrophication	-83.8%	-82.9%	-90.6%	-90.0%	-88.8%	-78.9%
Freshwater eutrophication	-91.2%	-90.3%	-94.8%	-94.4%	-83.8%	-75.9%
Marine eutrophication	-84.0%	-83.1%	-95.8%	-95.6%	-88.9%	-80.5%
Freshwater ecotoxicity	-87.4%	-85.6%	-91.4%	-91.4%	-71.6%	-57.4%
Human toxicity (cancer effects)	-79.9%	-77.5%	-81.4%	-80.7%	-63.7%	-47.5%
Human toxicity (non-cancer effects)	-82.8%	-80.6%	-89.8%	-89.1%	-89.5%	-63.8%
Particulate matter	-78.8%	-76.9%	-80.2%	-77.4%	-76.7%	-51.0%
Water resource depletion	-7.4%	-1.8%	-10.6%	4.7%	9.2%	24.5%
Mineral & fossil resource depletion	-83.3%	-82.0%	-84.7%	-82.7%	-72.9%	-59.0%
Cumulative energy demand	-86.5%	-85.4%	-89.8%	-88.9%	-78.4%	-68.2%
Minimum reduction[b]	*-78.8%*	*-76.9%*	*-80.2%*	*-77.4%*	*-63.7%*	*-47.5%*
Maximum reduction[b]	*-92.4%*	*-92.3%*	*-95.8%*	*-95.6%*	*-89.5%*	*-80.5%*

(a) Both glass and PET refillable bottles are used for 25 times.
(b) *Water resource depletion* is excluded from the calculation of the minimum and maximum reductions, because of its atypical behaviour. In fact, a reduction (or increase) smaller than ±10% is observed with respect to most bottled water scenarios, which are thus comparable to waste prevention scenario 1. A 24.5% increase is instead observed compared to refillable PET bottled water.

When very inefficient washing conditions, which are expected to be quite uncommon, are considered (washing after every use in a load of only 15 items), the impacts of waste prevention scenario 1 are increased by an average of 260%. For most indicators (9 out of 14), this scenario is thus outperformed by the best case of the refillable PET bottled water scenario, where bottles are transported to local distributors along a distance of only 40 km and are used 25 times. However, all one-way bottled water scenarios, and the refillable glass bottled water one, are still outperformed, even in the best case of a reduced transport distance to retailers or local distributors (40 km). The impact reductions achieved with respect to these best cases are the absolute lowest ones when the above inefficient washing conditions are considered for tap water (Table 2.7). The reductions achieved with respect to the worst case of baseline scenarios can instead be found in Table A.12 of Appendix A, and are the highest achievable under inefficient washing conditions.

Table 2.7: impact reductions resulting from the substitution of refined groundwater from the tap (waste prevention scenario 1) for the different types of bottled water, when the reusable jug used to withdraw tap water is washed under worsened conditions (after every use in a load of 15 items) and bottles are transported to retailers or local distributors along a distance of 40 km (best case of baseline scenarios). These reductions are the absolute lowest ones.

Impact categories	Reference baseline scenario					
	Virgin PET one-way bottles	50% recycled PET one-way bottles	PLA one-way bottles to composting	PLA one-way bottles to incineration	Refillable glass bottles[a]	Refillable PET bottles[a, b]
Climate change	-50.0%	-47.4%	-50.8%	-56.1%	-28.0%	23.1%
Ozone depletion	-75.3%	-75.0%	-78.3%	-79.5%	-45.8%	-27.7%
Photochemical ozone formation	-47.1%	-44.0%	-54.6%	-58.7%	-51.9%	-22.3%
Acidification	-18.1%	-12.7%	-34.6%	-43.4%	-26.9%	55.0%
Terrestrial eutrophication	-39.4%	-36.2%	-62.6%	-64.9%	-58.2%	-21.2%
Freshwater eutrophication	-53.4%	-48.9%	-70.6%	-72.3%	-14.4%	27.7%
Marine eutrophication	-39.5%	-36.1%	-83.2%	-83.9%	-58.0%	-26.0%
Freshwater ecotoxicity	-62.2%	-56.9%	-74.2%	-74.2%	-14.9%	27.7%
Human toxicity (cancer effects)	-46.8%	-40.4%	-48.8%	-50.8%	-3.9%	39.0%
Human toxicity (non-cancer effects)	-48.8%	-42.3%	-67.6%	-69.8%	-68.9%	7.6%
Particulate matter	-36.3%	-30.5%	-32.1%	-40.5%	-30.1%	47.4%
Water resource depletion	58.7%	68.3%	79.5%	53.2%	87.0%	113.3%
Mineral & fossil resource depletion	-48.8%	-44.5%	-46.8%	-52.8%	-16.7%	26.0%
Cumulative energy demand	-52.2%	-48.2%	-60.6%	-63.9%	-23.4%	13.0%
Minimum reduction[c]	*-18.1%*	*-12.7%*	*-32.1%*	*-40.5%*	*-3.9%*	*-21.2%*
Maximum reduction[c]	*-75.3%*	*-75.0%*	*-83.2%*	*-83.9%*	*-68.9%*	*-27.7%*

(a) Both glass and PET refillable bottles are used for 25 times.
(b) The variation of the impacts compared to the use of refillable PET bottled water is reported for completeness, although an increase is achieved, in this case, for most impact categories.
(c) *Water resource depletion* is excluded from the calculation of the minimum and maximum reductions, because of its atypical behaviour. An impact increase is indeed observed, for this indicator, compared to all bottled water scenarios.

Finally, when more efficient washing conditions are considered (washing after every 5 uses in a load of 50 items), the impacts of waste prevention scenario 1 are only marginally decreased (19% on average), meaning that the washing policy of the base case is already effective from an environmental and energy standpoint (at least for most indicators). Under these improved washing conditions, the highest impact reductions are achieved compared to bottled water. In particular, when this is transported along a distance of 800 km, the achieved reductions are the absolute highest ones (Table 2.8). The reductions achieved when a distance of 40 km is considered are instead lower, and are reported for completeness in Table A.13 of Appendix A.

As for *water resource depletion*, the outcomes of the comparison are a bit different, at least at a first glance. For average washing conditions (base case), refined tap water outperforms all types of one-way and refillable bottled water only when these are transported to retailers or local distributors along a distance of 800 km. For improved washing conditions (best case), the situation is similar. For inefficient conditions (worst case) the impact is drastically worsened, outweighing by far those of bottled water, even for a distance of 800 km. This latter worsening is due to the dramatic increase in the impact of the washing stage, because of the increased use of electricity and water.

However, these results must be read in view of the significant contribution provided by the stage of water quality improvement through a device based on reverse osmosis, which rejects 2 litres of water per each delivered litre. This additional consumption is responsible for an impact of about 0,056 m^3 eq. per functional unit, as represented in violet in Figure 2.7.

As it can be easily inferred, if water quality improvement was made with a device involving no additional water consumption, the outcome would be different, at least for average and improved washing conditions. In both cases, the impact would indeed be lower than all types of bottled water, even if this is transported along a distance of only 40 km. For inefficient washing conditions, the impact would instead be still greater than or comparable to bottled water, both for a distance of 300 km and 40 km. For 800 km, some bottled water scenarios are outperformed, the others being comparable.

Table 2.8: impact reductions resulting from the substitution of refined groundwater from the tap (waste prevention scenario 1) for the different types of bottled water, when the reusable jug used to withdraw tap water is washed under improved conditions (after every 5 uses in a load of 50 items) and bottles are transported to retailers or local distributors along a distance of 800 km (worst case of baseline scenarios). These reductions are the absolute highest ones.

Impact categories	**Reference baseline scenario**					
	Virgin PET one-way bottles	**50% recy-cled PET one-way bottles**	**PLA one-way bottles to composting**	**PLA one-way bottles to incineration**	**Refillable glass bottles[a]**	**Refillable PET bottles[a]**
Climate change	-95.9%	-95.7%	-95.9%	-96.1%	-97.1%	-95.4%
Ozone depletion	-96.3%	-96.3%	-96.6%	-96.7%	-96.9%	-95.3%
Photochemical ozone formation	-97.2%	-97.2%	-97.3%	-97.4%	-98.6%	-97.8%
Acidification	-94.1%	-94.0%	-94.6%	-95.0%	-96.9%	-94.7%
Terrestrial eutrophication	-97.4%	-97.3%	-97.7%	-97.7%	-98.7%	-97.9%
Freshwater eutrophication	-95.3%	-95.0%	-96.8%	-96.9%	-94.7%	-92.6%
Marine eutrophication	-97.3%	-97.3%	-98.3%	-98.3%	-98.7%	-97.9%
Freshwater ecotoxicity	-93.5%	-92.9%	-94.9%	-94.9%	-94.3%	-91.3%
Human toxicity (cancer effects)	-91.4%	-90.9%	-91.6%	-91.8%	-94.0%	-90.8%
Human toxicity (non-cancer effects)	-93.9%	-93.6%	-95.1%	-95.3%	-97.0%	-94.2%
Particulate matter	-91.8%	-91.5%	-91.6%	-92.1%	-96.2%	-91.7%
Water resource depletion	-27.1%	-23.5%	-19.4%	-29.2%	-33.1%	-17.7%
Mineral & fossil resource depletion	-92.1%	-91.8%	-92.0%	-92.5%	-94.2%	-90.9%
Cumulative energy demand	-93.8%	-93.6%	-94.5%	-94.8%	-95.5%	-93.0%
Minimum reduction[b]	*-27.1%*	*-23.5%*	*-19.4%*	*-29.2%*	*-33.1%*	*-17.7%*
Maximum reduction[b]	*-97.4%*	*-97.3%*	*-98.3%*	*-98.3%*	*-98.7%*	*-97.9%*

(a) Refillable glass bottles are used 10 times, while refillable PET bottles for 15 times.

(b) In this case, a significant reduction (>10%) is always achieved even for the *water resource depletion* indicator, which is thus included in the calculation of the minimum and maximum reductions.

If refined public network water is withdrawn from public fountains without using any motorised vehicle for the roundtrip (waste prevention scenario 2 - no car), a significant reduction of all impact indicators is achieved compared to all types of bottled water, irrespective of the distance along which this latter is transported to retailers or local distributors. In particular, when a distance of 800 km is covered, the highest reductions are achieved (Table 2.9). For a distance of only 40 km, the lowest reductions are instead achieved (Table A.14 of Appendix A).

Table 2.9: impact reductions resulting from the substitution of refined surface water withdrawn from public fountains for the different types of bottled water, when no motorised vehicles are used for the roundtrip to the fountain (waste prevention scenario 2 - no car) and bottles are transported to retailers or local distributors along a distance of 800 km (worst case of baseline scenarios).

Impact categories	Reference baseline scenario					
	Virgin PET one-way bottles	**50% recy-cled PET one-way bottles**	**PLA one-way bottles to composting**	**PLA one-way bottles to incineration**	**Refillable glass bottles[a]**	**Refillable PET bottles[a]**
Climate change	-91.9%	-91.7%	-92.5%	-92.0%	-94.3%	-90.9%
Ozone depletion	-96.8%	-96.8%	-97.1%	-97.0%	-97.3%	-95.9%
Photochemical ozone formation	-96.7%	-96.7%	-96.9%	-96.8%	-98.3%	-97.3%
Acidification	-93.3%	-93.2%	-94.3%	-93.9%	-96.5%	-93.9%
Terrestrial eutrophication	-97.0%	-97.0%	-97.4%	-97.4%	-98.6%	-97.7%
Freshwater eutrophication	-84.7%	-83.6%	-90.0%	-89.5%	-82.9%	-75.9%
Marine eutrophication	-96.8%	-96.7%	-98.0%	-97.9%	-98.4%	-97.5%
Freshwater ecotoxicity	-91.1%	-90.4%	-93.1%	-93.1%	-92.2%	-88.2%
Human toxicity (cancer effects)	-91.9%	-91.5%	-92.2%	-92.1%	-94.3%	-91.3%
Human toxicity (non-cancer effects)	-93.0%	-92.7%	-94.6%	-94.3%	-96.6%	-93.3%
Particulate matter	-91.8%	-91.5%	-92.1%	-91.6%	-96.2%	-91.7%
Water resource depletion	-62.7%	-60.9%	-63.8%	-58.8%	-65.8%	-57.9%
Mineral & fossil resource depletion	-91.7%	-91.3%	-92.1%	-91.5%	-93.9%	-90.4%
Cumulative energy demand	-90.7%	-90.3%	-92.2%	-91.7%	-93.2%	-89.4%
Minimum reduction	*-62.7%*	*-60.9%*	*-63.8%*	*-58.8%*	*-65.8%*	*-57.9%*
Maximum reduction	*-97.0%*	*-97.0%*	*-98.0%*	*-97.9%*	*-98.6%*	*-97.7%*

(a) Refillable glass bottles are used 10 times, while refillable PET bottles for 15 times.

The use of a private car for the roundtrip to the fountain significantly increases the potential impacts of this drinking water consumption alternative. In particular, for a roundtrip distance of 5.5 km and a withdrawal of 9 litres (base case of waste prevention scenario 2 - car) more than half of such impacts is larger than or comparable to those of the best cases of one-way PET bottled water scenarios and of the base case of the refillable PET bottled water scenario. However, in the best case of a reduced transport distance (40 km) and of an increased number of bottle uses (25), refillable PET bottled water is preferable to that withdrawn from fountains for all impact indicators except one, which is comparable.

If the roundtrip distance to the fountain is increased to 10 km and only 4.5 litres of water are withdrawn (worst case of waste prevention scenario 2 - car) all impacts are drastically worsened, with most of them exceeding by far those of the worst case of all one-way bottled water scenarios and of the one using refillable PET bottled water. If the use of a car is instead limited to an overall trip of 2 km, and 18 litres of water are withdrawn at the fountain (best case of waste prevention scenario 2 - car), all bottled water scenarios are outperformed with respect to all impact indicators, irrespective of the distance along which bottled water is transported (and the number of times refillable bottles are used). More specifically, we found that, for all impacts indicators to be lower than those of the best case of all bottled water scenarios, the maximum roundtrip distance to be covered with a car must not exceed 0.5 km for the withdrawal of 4.5 litres, 1 km for 9 litres, and 2 km for 18 litres.

2.9.2.2 Refillable versus one-way bottled water scenarios

Provided that refillable bottles are used for more than 10-15 times, the comparison between refillable and one-way bottled water mainly depends on the distance covered to reach retailers or local distributors. For an average distance of 300 km (base case of both scenarios), the use of refillable PET bottles prove to be the preferable option for packaged water delivery, with respect to most indicators. A first exception is *photochemical ozone formation*, where this alternative is comparable to the different types of one-way bottled water. For *terrestrial* and *marine eutrophication*, only PLA bottled water is instead outperformed (Table 2.10). For these three indicators, the decrease in the impacts of packaging life cycle is indeed compensated or exceeded by an increase in transport impacts. This increase is mostly due to the higher mass to be transported, compared to one-way bottled water, during both the delivery and the returning trip to retailers or local distributors. However, also the transport from these latter to the point of use shows an increased impact, especially for the three mentioned indicators.

For most indicators, the use of refillable glass bottled water has instead performance worse than or comparable to one-way bottled water, when an average distance of 300 km is considered (Table A.15 of Appendix A). This is again due to the increase in transport impacts, which is even larger than refillable PET bottled water, since refillable glass bottles are by far heavier.

Table 2.10: impact variations resulting from the substitution of PET refillable for one-way bottled water, when this is transported to retailers or local distributors along a distance of 300 km and refillable bottles are used 15 times (base case of baseline scenarios).

Impact categories	Reference baseline scenario			
	Virgin PET one-way bottles	**50% recycled PET one-way bottles**	**PLA one-way bottles to composting**	**PLA one-way bottles to incineration**
Climate change	-33.8%	-31.1%	-40.1%	-34.5%
Ozone depletion	-44.6%	-44.1%	-52.3%	-50.2%
Photochemical ozone formation	7.2%	10.1%	-5.5%	-0.7%
Acidification	-11.8%	-8.3%	-31.1%	-23.8%
Terrestrial eutrophication	14.7%	17.2%	-12.6%	-9.4%
Freshwater eutrophication	-49.2%	-44.8%	-68.8%	-66.9%
Marine eutrophication	16.1%	18.9%	-47.7%	-45.9%
Freshwater ecotoxicity	-47.0%	-41.0%	-61.7%	-61.6%
Human toxicity (cancer effects)	-31.3%	-25.4%	-35.2%	-33.2%
Human toxicity (non-cancer effects)	-18.7%	-12.0%	-44.7%	-41.5%
Particulate matter	-25.1%	-20.4%	-28.7%	-21.7%
Water resource depletion	-18.0%	-13.4%	-20.7%	-8.0%
Mineral & fossil resource depletion	-35.3%	-31.2%	-39.4%	-33.4%
Cumulative energy demand	-33.8%	-29.6%	-47.4%	-43.4%

Note: grey cells depict insignificant impact variations (lower than ±10%), while red cells depict the few situations in which the substitution involves an overall impact increase.

When the distance is reduced to 40 km (best case[10]), the use of refillable PET bottled water becomes preferable to all types of one-way bottled water, with respect to all indicators (Table 2.11). Even the use of refillable glass bottled water achieves an improved environmental and energy profile, but only for 8 out of 14 indicators all one-way bottled water scenarios are outperformed[11]. For the remaining indicators[12], virgin and 50% recycled PET one-way bottled water show instead better or comparable performance with respect to refillable glass bottled water (Table A.16 of Appendix A).

[10] In this case, an increase in the number of uses of refillable glass bottles is also considered. However, as explained later, this increase has only marginal effects on the overall impacts of refillable bottled water scenarios. Thus, the variation of impacts that is achieved compared to the base case is mostly due to the reduction in the travelled distance.

[11] These indicators include: *climate change*, *ozone depletion*, *human toxicity (cancer effects)*, *freshwater eutrophication*, *freshwater ecotoxicity*, *water re source depletion*, *mineral and fossil re source depletion* and the *cumulative energy demand*.

[12] These indicators include: *human toxicity (non-cancer effects)*, *particulate matter*, *photochemical ozone formation*, *acidification*, *terrestrial eutrophication* and *marine eutrophication*.

Table 2.11: impact variations resulting from the substitution of PET refillable for one-way bottled water, when this is transported to retailers or local distributors along a distance of 40 km and refillable bottles are used 25 times (best case of baseline scenarios).

Impact categories	Reference baseline scenario			
	Virgin PET one-way bottles	50% recycled PET one-way bottles	PLA one-way bottles to composting	PLA one-way bottles to incineration
Climate change	-59.4%	-57.3%	-64.3%	-60.0%
Ozone depletion	-65.9%	-65.5%	-71.6%	-70.1%
Photochemical ozone formation	-31.9%	-27.9%	-46.8%	-41.6%
Acidification	-47.2%	-43.7%	-63.5%	-57.8%
Terrestrial eutrophication	-23.1%	-19.0%	-55.5%	-52.6%
Freshwater eutrophication	-63.5%	-60.0%	-78.3%	-77.0%
Marine eutrophication	-18.2%	-13.7%	-78.3%	-77.3%
Freshwater ecotoxicity	-70.4%	-66.2%	-79.8%	-79.8%
Human toxicity (cancer effects)	-61.7%	-57.1%	-64.6%	-63.2%
Human toxicity (non-cancer effects)	-52.4%	-46.3%	-72.0%	-69.8%
Particulate matter	-56.8%	-52.8%	-59.6%	-53.9%
Water resource depletion	-25.6%	-21.1%	-28.2%	-15.8%
Mineral & fossil resource depletion	-59.3%	-56.0%	-62.6%	-57.8%
Cumulative energy demand	-57.7%	-54.2%	-68.1%	-65.2%

Finally, if the distance to be covered is increased to 800 km, most of the potential advantages resulting from the substitution of refillable PET bottled water for that packed in one-way bottles are vanished (Table 2.12), especially if the latter is transported along an average or short distance (300 or 40 km). In this case, in fact, the impacts of the refillable PET bottled water scenario are significantly worsened (average increase of 96,5%). A similar increase (about 89%) is observed also for refillable glass bottled water, which in this case becomes the worst alternative for packaged water delivery, for all indicators except for the *freshwater eutrophication* one (Table A.17 of Appendix A).

Increasing the number of times refillable bottles are used from 10 or 15 to 25 involves only a marginal reduction in the impacts. In particular, an average reduction by about 7% is achieved for glass bottles, while a 4% average reduction is observed for PET bottles when an average distance of 300 km is covered to reach local distributors and retailers[13]. This suggests that 10-15 uses are already a reasonable target, although certain environmental and energy benefits can be achieved by using bottles until this is technically feasible.

[13] For a shorter distance of 40 km, relative reductions are higher (9,5% for glass bottles and 7,5% for PET ones), due to the decreased importance of the transport stage in favour of the life cycle of bottles. As a consequence, for a larger distance of 800 km, achieved reductions are lower (equal to about 2% for both glass and PET bottles).

Table 2.12: impact variations resulting from the substitution of PET refillable for one-way bottled water, when this is transported to retailers or local distributors along a distance of 800 km and refillable bottles are used 15 times (worst case of baseline scenarios).

Impact categories	Reference baseline scenario			
	Virgin PET one-way bottles	**50% recycled PET one-way bottles**	**PLA one-way bottles to composting**	**PLA one-way bottles to incineration**
Climate change	-11.0%	-8.5%	-17.1%	-11.7%
Ozone depletion	-21.5%	-21.0%	-29.8%	-27.4%
Photochemical ozone formation	23.9%	25.5%	16.1%	19.1%
Acidification	10.2%	12.7%	-5.2%	0.9%
Terrestrial eutrophication	28.2%	29.6%	11.7%	13.8%
Freshwater eutrophication	-36.3%	-31.7%	-58.6%	-56.4%
Marine eutrophication	28.8%	30.2%	-18.9%	-17.0%
Freshwater ecotoxicity	-24.5%	-18.4%	-41.3%	-41.2%
Human toxicity (cancer effects)	-7.0%	-1.9%	-10.6%	-8.8%
Human toxicity (non-cancer effects)	4.2%	9.3%	-19.1%	-15.9%
Particulate matter	-1.5%	2.4%	-4.5%	1.3%
Water resource depletion	-11.5%	-7.1%	-14.1%	-2.1%
Mineral & fossil resource depletion	-13.3%	-9.4%	-17.3%	-11.5%
Cumulative energy demand	-12.5%	-8.6%	-26.0%	-21.9%

Note: grey cells depict insignificant impact variations (lower than ±10%), while red cells depict those situations in which the substitution involves an overall impact increase.

2.9.2.3 One-way bottled water scenarios: a brief comparison

Focusing on one-way bottled water scenarios, results show that the use of 50% recycled PET for bottle manufacturing (baseline scenario 2) only allows for a modest impact reduction compared to the use of virgin material only (baseline scenario 1). Savings ranging from about 1% to about 10% are indeed achieved by replacing virgin material, when water is transported to retailers along a distance of 300 km[14]. Therefore, despite the use of recycled material represents an initial appreciable effort to improve the environmental and energy profile of one-way bottled water, it seems that only minor environmental and energy benefits are involved. More meaningful improvements can instead be achieved by reducing the distance travelled to reach retailers beneath 50 km. An average 29-30% reduction in the impacts of both one-way PET bottled water scenarios is indeed observed when the distance is reduced from 300 to 40 km. Similarly, an average reduction by 23-24% is achieved for the two PLA-based one-way scenarios. On the other hand, a drastic increase by an average 56-59% for PET-based one-way scenarios and 44-46% for PLA-based ones takes place when the distance is raised to 800 km.

For the vast majority of indicators, the use of PLA bottles (baseline scenarios 3a and 3b) shows the worst performance, especially when composting is considered as end-of-life option. This is mostly

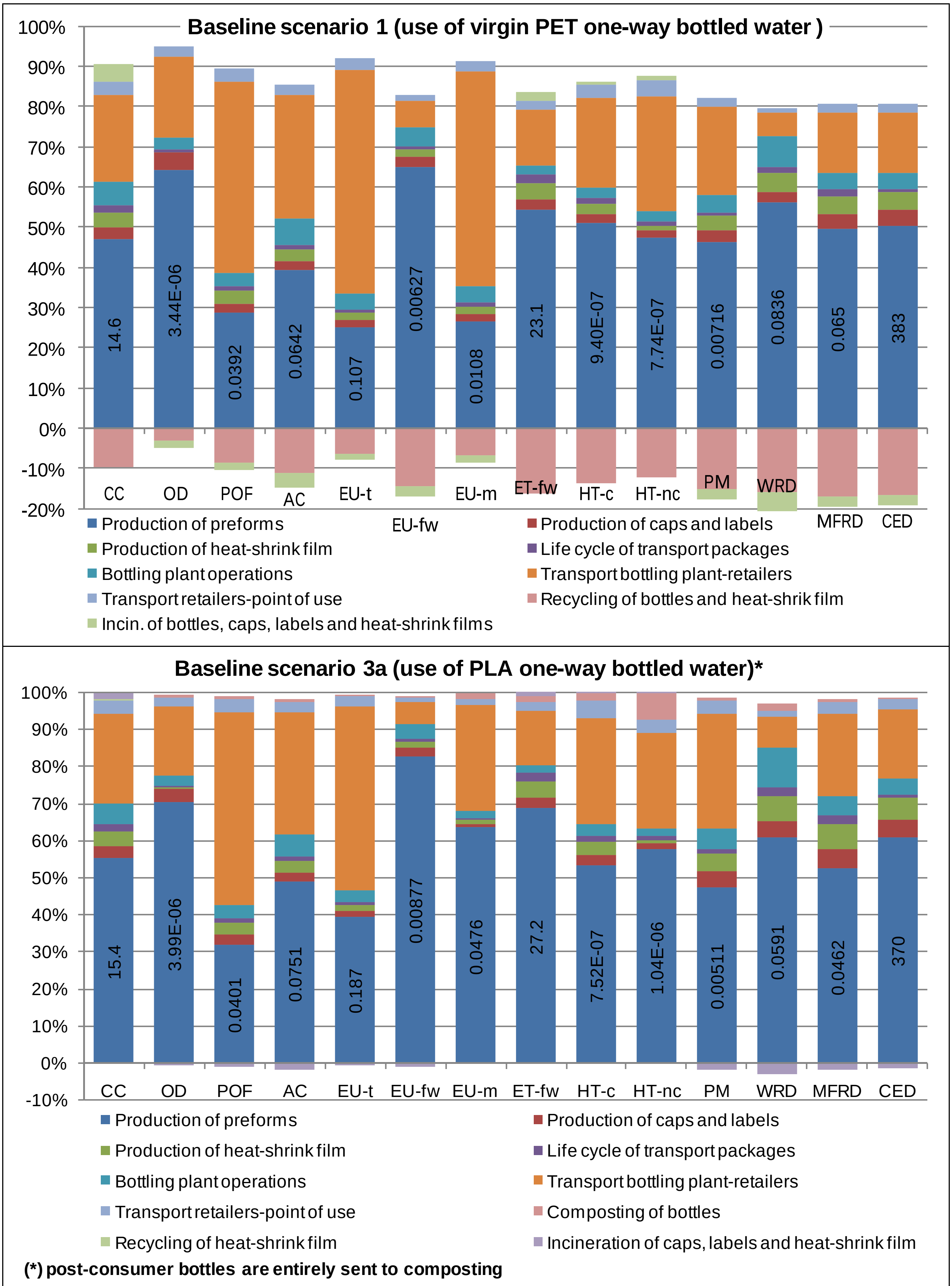
[14] For a distance of 40 km, the achieved reductions are slightly higher (from 1% to 12%), but always moderate. For 800 km, reductions are instead lower (from 0,5% to 7,5%).

due to the reduced benefits from composting or incineration of PLA bottles compared to mechanical recycling of PET bottles (Figure 2.8)[15]. Moreover, the use of fertilisers for maize cultivation significantly increases eutrophication impacts associated with bottle manufacturing compared to PET. Terrestrial and marine eutrophication are especially increased (by 75% and 342%, respectively), while a 40% increase is observed for freshwater eutrophication. Conversely, the substitution of PLA for PET allows for a 29% reduction of fossil resource depletion resulting from bottle production, but this reduction is again masked by the reduced benefits from recycling.

Therefore, composting or incineration do not prove the most sustainable end-of-life options for PLA bottles, and a possible large-scale substitution of PET by PLA bottles for water distribution seems unjustified under these conditions. It must, however, be noticed that an alternative end-of-life option for post-consumer PLA products could be their chemical or mechanical recycling. Through chemical recycling PLA products are de-polymerised via hydrolysis into lactic acid monomers, which can be used for the manufacturing of detergents or green solvents. Even new PLA resin could be produced by re-polymerisation of lactic acid, but the final quality of resin still needs to be improved (NatureWorks, 2012a; Loopla-Galacic, 2012). With mechanical recycling PLA products are instead only re-granulated, to be used for less-demanding, non food-grade applications. A proper collection or sorting system needs however to be implemented for both types of recycling process.

A possible substitution of composting or incineration of PLA bottles by their recycling may thus lead to completely different results from those obtained in this study. However, due to the experimental nature of the recycling process (especially the chemical one), no consolidated data were available to allow for a proper modelling. Moreover, a recycling scenario would be incompatible with collection schemes currently adopted in Italy for municipal waste.

[15] As it can be observed, for most indicators, a real adverse impact is involved by composting of bottles, rather than only a reduced benefit.



CC: climate change; OD: ozone depletion; HT-c: human toxicity (cancer effects); HT-nc: human toxicity (non-cancer effects); PM: particulate matter; POF: photochemical ozone formation; AC: acidification; EU-t: eutrophication (terrestrial); EU-fw: eutrophication (freshwater); EU-m: eutrophication (marine); ET-fw: ecotoxicity (freshwater); WRD: water resource depletion; MFRD: mineral and fossil resource depletion; CED: cumulative energy demand.

Figure 2.8: percentage contributions to the total impacts of baseline scenarios 1 and 3a in comparison.

2.10 Concluding remarks and recommendations

A life cycle assessment was carried out to evaluate whether the substitution of one-way bottled water by public network water is actually beneficial from an environmental and energy standpoint. However, the consequences of a possible substitution by refillable bottled water were also briefly investigated. Finally, the different types of one-way bottled water used as a reference in the assessment were mutually compared, in order to identify possible opportunities for the improvement of the respective environmental and energy performance.

The results reveal that the use of public network water withdrawn directly from the household tap (be it groundwater or surface water) allows for an actual reduction in waste generation compared to the use of water packaged in the most common types of one-way and refillable bottles (74-90%). If the automatic dishwashing of the container used to withdraw water is carried out according to minimum efficiency criteria (e.g., after 4-5 withdrawals in a load of at least 30 items), a significant reduction is achieved also for all the considered impact indicators. If the container is instead washed under quite inefficient conditions (e.g., after each use in a load of only 10-15 items) it is not possible to achieve, for the whole set of indicators, a (significant) reduction with respect to all types of bottled water when this is transported to retailers or local distributors along a short distance of 40 km. For *water resource depletion*, the same situation is observed even for a distance of 300 and 800 km. Finally, if a device based on reverse osmosis is used to improve water quality before withdrawal, the overall framework of results is mostly unchanged if the container is washed according to the minimum efficiency criteria mentioned above. Nevertheless, for *water resource depletion*, it is possible to achieve a significant impact reduction with respect to all types of bottled water, only when these are transported to retailers or local distributors along a distance of 800 km. For lower distances (300 km or 40 km), most or some baseline scenarios show instead a comparable or better performance. This is because the used device rejects 2 litres of water (as retentate) per each litre of delivered water.

Even the use of refined water withdrawn from public fountains allows for a significant reduction in waste generation compared to all types of bottled water (64-86%). An important reduction in the energy demand and potential impacts is also achieved, provided that a private car is not used for the roundtrip to the fountain and back. Conversely, to achieve a significant reduction even when bottled water is transported along a short distance (40-50 km), the overall journey by car has not to exceed 0.5 km for the transport of 4.5 litres, 1 km for 9 litres, and 2 km for 18 litres. If a system of public fountains is to be implemented in a given municipality, it should thus be designed so that the use of a private car by the consumer can be limited to short distances or, better, completely avoided. This could be achieved, for instance, by developing a capillary network of fountains and by limiting their use only to the citizens belonging to the municipality. On the other hand, if a car is used, the consumer should withdraw at least 9-10 litres of water and, in case, should carry out the journey also for further purposes (e.g. going or coming back to the workplace). Finally, all the efforts should be made by the consumer to extend as much as possible the useful life of bottles. In particular, traditional one-way PET bottles should be used for at least 4-5 times after their first use for the distribution of packaged water. By following these recommendations, the potential impacts associated with the practice of drinking water delivered from public fountains are minimised and, very likely, lower than those associated with the use of any type of bottled water.

The substitution of one-way by refillable PET bottled water allows for a quite important reduction in waste generation, although it is moderately lower than the one achievable by substituting public network water. The substitution would be beneficial also with respect to all the considered impact indicators if water is transported to retailers or local distributors along a distance not exceeding 40 km for 15 uses of bottles and 50 km for 25 uses. For larger distances, an increasing number of indicators becomes comparable or favourable to the best types of one-way bottled water, due to the increased transport impacts of refillable bottled water. However, only the *marine* and *terrestrial eutrphication*, as well as the *photochemical ozone formation* are affected by this worsening if the distance is lower than 250 km (for 15 uses) or 300 km (for 25 uses).

The substitution by refillable glass bottles would result in an increased mass of generated waste, due to the higher density (specific mass) of glass compared to one-way PET or PLA bottles. Moreover, a number of impact indicators remain favourable to the best types of one-way bottled water (50% recycled or virgin PET bottles) even when the transport takes place along a short distance (40-50 km).

As for one-way bottled water, the use of 50% recycled PET for the manufacturing of bottles allows for a modest improvement in the overall performance (5% on average for the base case distance of 300 km). Much more meaningful improvements can however be obtained by limiting the distance along which water is transported to retailers to, e.g., 50-100 km, and by performing an efficient car trip for water purchasing (e.g. by purchasing at least 30 items, or by carrying out the journey for multiple purposes). Retailers hold a key role in distance shortening, and should pursue this by opting into the sale of water bottled at the local or regional level (almost all Italian Regions host one or more bottling companies). Finally, the use of one-way PLA bottles showed the worst performance, because of the lower benefits resulting from their composting and incineration compared to mechanical recycling of PET bottles.

3 Life cycle assessment of waste prevention in liquid detergent distribution

In Italy, liquid detergents are traditionally distributed with single-use plastic containers. However, an alternative distribution method has recently been introduced by some producers, with the declared aim of reducing waste generation and the adverse impacts on the environment. Retail establishments are equipped with an automatic self-dispensing system, where different types of detergent are offered ‘loose’. The consumer can thus withdraw the product by means of refillable plastic containers available at the store and reuse them for several times. Due to the estimated waste prevention potential, this practice has been included in the set of measures identified by the national waste prevention programme adopted in 2013 (Ministero dell’Ambiente e della Tutela del Territorio e del Mare, 2013) and by some regional programmes (e.g. Regione Lombardia, 2009).

This section summarises a life cycle assessment (LCA) study which compares the distribution of liquid detergents through self-dispensing systems, with the one based on single-use plastic containers. The LCA technique was applied in all its four basic stages (goal and scope definition, inventory analysis, impact assessment and interpretation), with the support of the SimaPro 7.3.3 software. This tool facilitated the development of a parametric model of the two compared distribution systems and the calculation of the respective potential impacts.

3.1 Brief review of available LCAs for detergent distribution

Different formulations for powder and liquid laundry detergents and relative distribution methods have been compared in a number of LCAs carried out by researchers of Procter and Gamble (Saouter et al., 2002; Van Hoof et al., 2003a, 2003b; Dewaele et al., 2006). These studies found that packages are generally responsible for only a small portion of the overall impact, although their end of life is excluded. Most impact categories are indeed dominated by the washing stage (heating of the water or waterborne emissions) or by the production of detergent ingredients. However, packages contribute approximately to 7–15% of total solid waste generation.

Two potentially less waste-generating methods for liquid detergent distribution have recently been compared with traditional methods in two separate LCAs (Bolzonella and Gittoi, 2011 and CURA, 2012). Both are screening assessments and focus on climate change only. In the first study (Bolzonella and Gittoi, 2011), detergents are delivered to retailers by means of 20 litre, reusable, high-density polyethylene (HDPE) tanks. From these tanks, the consumer can then withdraw directly the product with 1 litre refillable HDPE containers. Under the assumption that tanks are used 115 times overall and containers 20 times, this alternative method allows approximately for a

44% reduction of the impact on climate change compared to distribution with 1 litre single-use HDPE containers. In the second study (CURA, 2012), detergent distribution is made by means of 20 litre disposable “bag-in-box” containers (plastic pouches included into corrugated cardboard boxes). These packages are used to refill a simple self-dispensing system, from which the product can be withdrawn manually by means of dedicated 1 litre refillable HDPE containers. Compared to traditional distribution with 1 litre single-use HDPE containers, this alternative method is approximately responsible for a 78% lower impact on climate change if containers are used 30 times overall.

To the authors' knowledge, only one comparative LCA study (carried out on behalf of Assocasa[1]) focused on an alternative distribution system similar to the one examined in this assessment. Nevertheless, only a brief summary of the results was disclosed to the public. The major conclusion of the study is that, for most indicators, a minimum of 5-10 uses of the refillable container are needed for the alternative system to be advantageous compared to traditional distribution. However, for some indicators, the best “traditional” scenarios proved to be comparable or preferable to the alternative system even beyond 10 uses.

3.2 Goal of the assessment

The first objective of the study is to evaluate whether, and under which conditions, distribution of liquid detergents through self-dispensing systems allows to reduce waste generation, the overall potential impacts on the environment and on human health and the total energy demand, compared to the distribution with single-use containers. If this is the case, the second objective is to quantify achievable waste prevention and impact reduction potentials.

Out of the five categories of detergents that are currently distributed loose, the study focuses on those which presumably have the highest market shares, i.e., laundry detergents, fabric softeners and hand dishwashing detergents (Table 3.1). Liquid detergents intended for washing of delicate garments and floor cleaning were thus excluded.

Table 3.1: value sales of some (liquid) detergent categories in Italy for 2011 (data from Euromonitor International).

Detergent Category	Retail Value (EUR million)
Automatic Laundry Detergents (liquid)	602.9
Hand Wash Laundry Detergents	25.9
Hand Dishwashing Detergents (liquid)	307.7
Fabric Softeners (liquid)	287.8
Floor cleaners (total)	81.5

[1] Assocasa is the association of Italian companies dealing with cleaning, maintenance and hygiene products.

3.3 Analysed scenarios

For each selected detergent category, a comparison was made between a first set of scenarios where the detergent is distributed by means of single-use containers (baseline scenarios), with two scenarios where it is distributed 'loose' (waste prevention scenarios). A list of the scenarios defined for each category is provided in Table 3.2.

Table 3.2: alternative scenarios for liquid detergent distribution analysed in the present LCA study.

CATEGORY OF DETERGENT	BASELINE SCENARIOS		WASTE PREVENTION SCENARIOS
	Distribution with single-use HDPE[a] containers with a size (in ml) of[b]	Distribution with single-use PET[a] containers with a size (in ml) of[b]	Distribution through self-dispensing systems with the provision to the consumer of a refillable virgin HDPE container with a size (in ml) of[b]
Laundry detergents	750; 1000; (1500-1518); (1820-2100); (2409-2625); (3000-3066); (3900-4000); 5000	750; 924; 1848	1000 (scenario 1)[c]
			3000 (scenario 2)
Fabric softeners	750; 1000; (1500-1560); (2000-2015); 2460; (2990-3000); 4000	750; 1000; 1500; 2000	1000 (scenario 1)[c]
			2000 (scenario 2)
Hand dishwashing detergents	750; (1000-1110); 1250; 1500; 2000; 3000; 4000; 5000	(500-650); 750; 1000; 1250; 1500	1000 (62 g; scenario 1)[c]
			1000 (71.5 g; scenario 2)

Acronyms: HDPE = high-density polyethylene; PET = polyethylene terephthalate.
(a) As a base case, containers were assumed to be entirely produced from virgin material. However, the use of 100% recycled material was also explored in a sensitivity analysis (Section 3.8).
(b) Each size or size class coincides with a specific scenario. In the case of size classes, scenarios foresee the distribution by means of single-use containers with a size that can ideally range from the lower bound to the upper bound of the respective class.
(c) The first waste prevention scenario is identical for all detergent categories. This is because in the real experience of distribution through self-dispensing systems assumed as a reference in that scenario, the same container is provided to the consumer for the withdrawal of all types of detergents offered loose.

Baseline scenarios differ in the material with which the disposable container is made and in its size. They were defined based on an extensive survey of the types of containers used in 2013 for the distribution of the major brands of the three considered categories of detergents in Italy. Such brands covered more than 80% of the market for laundry detergents, 35% for fabric softeners and 50% for hand dishwashing detergents. An alternative approach could have been the modelling of a unique baseline scenario, where each type of single-use container is used in proportion to the respective market share. Doing so, the actual mix of substituted packages is taken into account. Unfortunately, no publicly accessible data were available on the popularity of each type of container and, therefore, we could not define an average unique scenario.

The two waste prevention scenarios were defined with reference to the pilot experiences of distribution through self-dispensing systems currently implemented by two Italian producers of detergents. The two scenarios differ primarily in the size of the refillable container (or in its mass in the case of hand dishwashing detergents). Moreover, in waste prevention scenario 1, reusable caps are transported to retail outlets with dedicated packages. In waste prevention scenario 2, caps are instead screwed directly on the refillable containers at the packaging plant and, consequently, transported together with containers inside the same packages.

3.4 Functional unit

The function of the compared systems is the delivery of liquid detergent to a generic Italian consumer who makes his own purchasing activities nearby the large-scale retail trade. The functional unit used in the study is thus "*the distribution of 1000 litres of detergent nearby a retail outlet of the large-scale retail trade in Italy*". This represents the unit used as a reference for the calculation of waste generation, of the potential impacts and of the energy demand of the compared scenarios.

3.5 System description

This section describes briefly the two alternative distribution systems compared in this study. The description is mostly based on the evidence gathered during a field survey at the manufacturing plant of an Italian producer of liquid detergents. Information retrieved from direct contacts with other producers was also taken into account.

3.5.1 Distribution with single-use plastic containers (baseline scenarios)

At the manufacturing plant, detergents are firstly packed into HDPE or PET single-use containers. These are subsequently capped with polypropylene (PP) capsules and labelled with paper or plastic labels. For transpor purposes, filled containers are placed inside disposable corrugated cardboard boxes. Each box normally includes 4 to 20 containers, depending on the size. Boxes are then loaded on reusable wooden pallets and wrapped with a disposable linear low-density polyethylene (LLDPE) stretch film, which assures the stability of the whole load. Complete load units are then stocked until they will be transported to the distribution platforms of the different supermarket chains and, subsequently, to the single retail outlets. During the return trip, empty pallets from previous deliveries are transported back to the packaging plant, where they are reused to build new load units. At retail outlets, film and boxes are removed and become commercial wastes. Empty containers and respective caps, which are discarded by the consumer at the household, are instead collected as municipal solid wastes and are managed accordingly.

3.5.2 Distribution with self-dispensing systems (waste prevention scenarios)

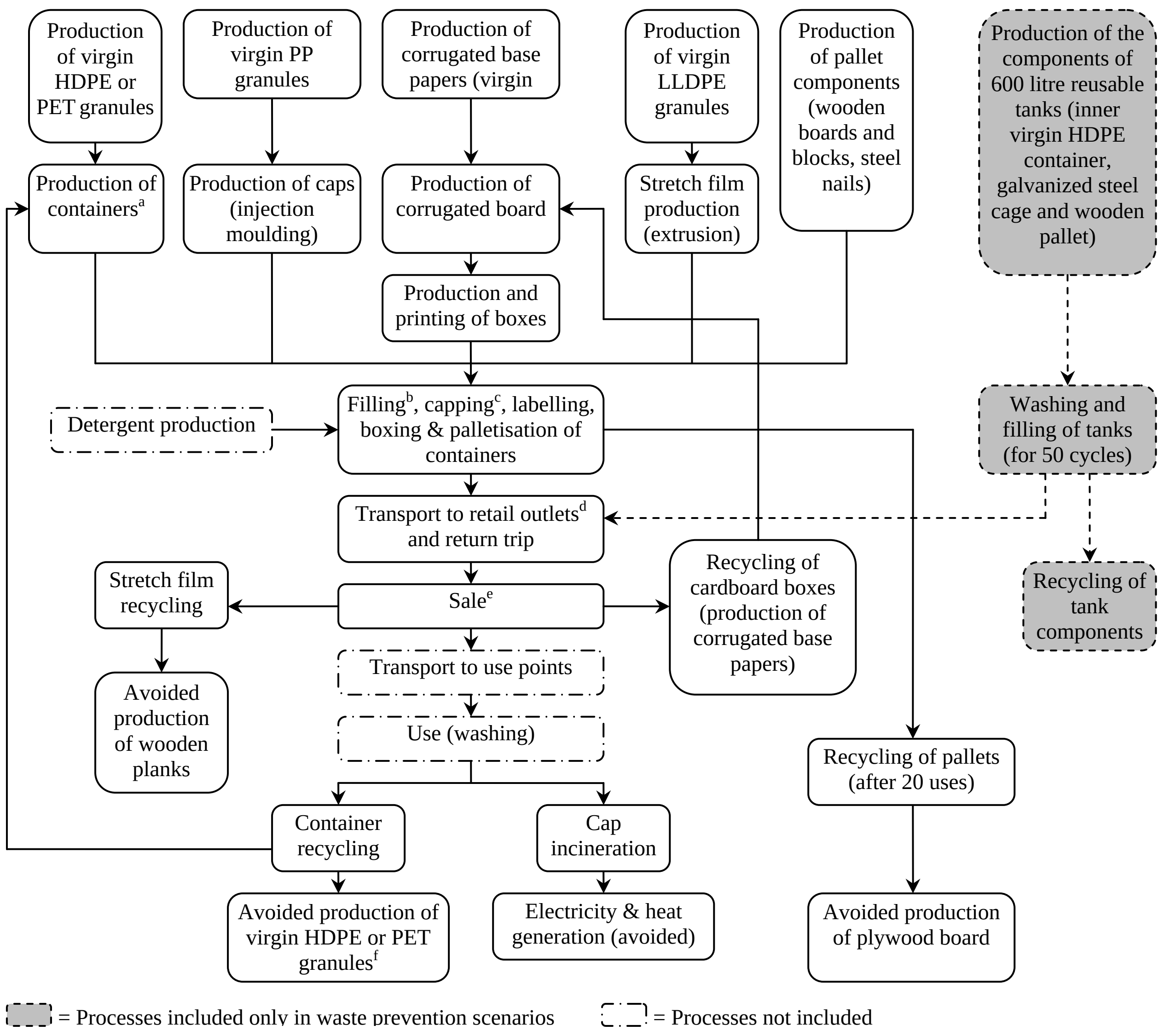
At the manufacturing plant, the detergent is filled inside 600 or 1000 litre reusable tanks. These consist of an inner virgin HDPE container, an external cage made of galvanized tubular steel and a wooden pallet on which the cage is fixed (Figure B.1 of Appendix B). Filled tanks are then transported to the distribution platforms of the different supermarket chains first, and to the respective retail outlets afterwards. Empty refillable containers and their caps are transported as well to retail outlets, following the same pathway. To this purpose, empty refillable containers are placed, like single-use ones, inside disposable corrugated cardboard boxes, which are subsequently loaded on reusable wooden pallets and wrapped with a disposable LLDPE stretch film. However, since empty containers are lighter than full ones, boxes are bigger and can include many more containers (up to 100 in the examined experiences). A lower amount of boxes is thus needed, overall, per load unit.

At retail outlets, the detergent contained in reusable tanks is used to refill an automatic self-dispensing system (Figure B.2 of Appendix B). This system is equipped with smaller tanks, each of which can hold a given type of detergent. Generally, four tanks with a volume of 80 litres each are available. The detergents stored in the system can be withdrawn by the consumer with the provided refillable containers, which will be filled completely.

Disposable packages used for the transport of containers and caps (boxes and stretch film) become commercial wastes at retail outlets. Wooden pallets used for the same purpose and reusable tanks are instead collected during the subsequent deliveries and transported back to the packaging plant. Here, reusable tanks are washed with network water and refilled, while pallets are reused to build new load units. Like single-use containers and caps, end-of-life refillable containers and their caps are discarded by the consumer as municipal wastes and are managed as such.

3.6 System boundaries

The major processes included in the system boundaries in both baseline and waste prevention scenarios are represented in Figure 3.1. In both cases, system boundaries include the life cycle of containers (single-use or refillable), of their caps and of their transport packages; all the operations carried out at the packaging plant; the transport of palletised containers to retail outlets; and the return trip with empty reusable pallets from previous deliveries. In addition, waste prevention scenarios include also the life cycle of reusable tanks, their transport to retail outlets, the return trip with empty tanks from previous deliveries, product purchase from the self-dispensing system, its refilling and the life cycle of its main components. Detergent production is instead always



(a) HDPE containers are produced at the packaging plant by extrusion blow moulding of virgin or recycled HDPE granules. PET containers are instead produced by stretch-blow moulding of PET preforms, which in turn are produced in an external facility by means of injection moulding of loose PET granules. Note also that the input of recycled granules is included only when a 100% recycled content is assumed for single-use containers (sensitivity analysis; Section 3.8).

(b) Filling of containers is carried out only in baseline scenarios.

(c) In waste prevention scenario 1, reusable caps are transported separately from refillable containers, which are therefore not capped. In this case, the life cycle of the packages used for cap transport is included as well in the system boundaries. These packages are identical to those used for containers, although a disposable LDPE bag is used in addition. Loose caps are firstly placed in this bag and then packed in cardboard boxes. Packing and palletisation operations are also included in the system. These operations are directly carried out by the cap producer (i.e. they are not made at the detergent packaging plant). In this scenario also refillable containers are directly packed and palletised by the respective producer.

(d) Generally, palletised items are firstly transported to distribution centres of single supermarket chains. Here, according to the specific needs of single retail outlets, new load units consisting of different packed products are built and subsequently transported to the intended destinations. Due to the extreme variability of this stage, palletised items were assumed to be directly transported to retail outlets.

(e) The major burdens of sale and purchase activities are generally those associated with the operation of retail establishments (lighting, conditioning, etc.) and with the use of fork-lift trucks for the handling of palletised products. However, these burdens are likely very similar in both baseline and waste prevention scenarios. No specific burdens are thus attributed to sale and purchase activities in baseline scenarios. Conversely, waste prevention scenarios include the additional burdens associated with the operation of the self-dispensing system and with the life cycle of its main components.

(f) When single-use containers are assumed to be entirely produced from recycled material (sensitivity analysis; Section 3.8), the

Figure 3.1: main processes included in (and excluded from) the system boundaries in the baseline and waste prevention scenarios compared in the present LCA study.

excluded since it is assumed to be the same in all compared scenarios (all types of liquid detergents can be distributed loose). For the same reason, the purchasing roundtrip possibly performed with a private car by the consumer and the washing stage at the household are excluded as well. Finally, the whole life cycle of labels applied to containers is excluded since the amount of material used per functional unit is small (approximately 1-3 g per litre of detergent) and the contribution to the total impacts is deemed to be negligible.

3.7 Impact categories, indicators and characterisation models

To evaluate the effectiveness of the examined prevention activity in reducing waste and estimate the achievable waste prevention potential, the amount of waste generated in all the compared scenarios was calculated first. The indicator accounts for the packages discarded by the consumer at the point of use and the transport packages discarded at retail establishments or at the packaging plant. Other possible types of waste generated during retailing or manufacturing were instead excluded, as well as the waste generated in previous life cycle stages.

For coherence with the assessment described in Section 2, the same thirteen impact categories were selected, along with the same category indicators and characterisation models (Table 2.3). Similarly, the cumulative energy demand indicator was calculated lastly.

3.8 Sensitivity analysis

Single-use containers were initially assumed to be produced from virgin material (extruded plastic granules). Nevertheless, up to 100% of recycled polymers can also be used for this purpose, especially for dull coloured containers. The potential impacts of baseline scenarios were thus recalculated also under the assumption that single-use containers are entirely made from recycled HDPE or PET granules. The impacts of waste prevention scenarios were instead calculated as a function of the number of uses of the refillable container. The aim was to evaluate how the behaviour of the consumer affects the ultimate performance of such scenarios and of the prevention activity as a whole. The calculation of the impacts was thus repeated for a number of utilisation cycles ranging from 1 to 50.

3.9 Modelling of scenarios

A parametric model of the two alternative distribution systems investigated was created in the SimaPro software. A relatively easy transition from one scenario to the other and from one category of detergent to the other could thus be performed by adjusting a set of parameters. The main parameters are the average masses of containers, caps, cardboard boxes and stretch film needed per functional unit (baseline scenarios) or per litre of detergent (waste prevention scenarios), the average number of pallets needed (all scenarios), as well as the number of uses considered for the refillable container (waste prevention scenarios). Other parameters are the average detergent density and one indicating the waste prevention scenario to be assessed (no. 1 or 2).

The following sections describe briefly how input and inventory data were defined for the different life cycle stages included in the virtual model of the compared distribution systems. Further details on the modelling approach are available in Appendix B (Section B.2).

3.9.1 Life cycle of primary and transport packages

The average masses of single-use containers, caps and cardboard boxes needed per functional unit in each baseline scenario were estimated experimentally. To this purpose, 219 single-use containers and respective caps were weighed, along with 133 cardboard boxes. Similarly, the average number of pallets needed per functional unit in the baseline scenarios was estimated based on a sample of real pallet compositions[2], acquired from detergent producers or from retailers. A brief description of the estimation procedure used for each packaging and the obtained results are available in Appendix B (Section B.2.1.1). The amount of stretch film needed per functional unit was estimated based on annual consumption and production data acquired from an Italian manufacturer of liquid detergents. The same estimate (0.62 g of stretch film per litre of detergent) was then assumed in all baseline scenarios. No product-specific data were indeed available.

The masses of the different types of refillable containers and of the respective caps were also measured experimentally. Conversely, the masses of the packages used for the transport of such containers and caps were acquired from their producers, along with the composition of the respective pallet. The average masses of the different components of reusable tanks were directly provided by their producer, as well. All the mentioned data are available in Appendix B (Tables B.10 to B.13).

Based on collected evidence, we assumed that all packages are produced from virgin material except the disposable cardboard boxes, which were assumed entirely produced from recycled fibres. As for the reusable tank, according to Classen et al. (2009), 37% of the steel cage is produced from post-consumer iron scraps. The remaining components (inner HDPE container and pallet) are instead produced from virgin material.

[2] A pallet composition indicates the number of cardboard boxes loaded on that pallet, so that the overall volume of detergent transported can ultimately be calculated.

Regarding the end of life, we assumed that all packages are recycled except the caps, which are incinerated in a waste to energy plant after being sorted as residues from plastic wastes. Even the different components of the reusable tank are recycled at the end of their useful life, which was assumed equal to 50 cycles of transport. Further details on the type of recycling process considered for each packaging and for tank components are available in Appendix B (Section B.2.1.2). For a short description of the common approach used in the modelling of end-of-life recycling activities and recycled content (the *avoided burden* approach), the reader is instead referred to Section 2.7.3.

Inventory data on the unit processes characterising the life cycle of primary and transport packages were derived from the *ecoinvent* database (directly or with some adaptations and updates), from elaborations on literature data (e.g., for plastic recycling) or from equipment manufacturers (e.g., for HDPE tank recycling). See Section B.2.1.3 of Appendix B for further details.

3.9.2 Packing operations

The operations carried out at the manufacturing plant for detergent packing were modelled based on primary data related to a medium sized plant located in central Italy. In this plant, different types of liquid detergents are formulated and packed in single-use containers or in reusable tanks for their subsequent distribution through self-dispensing systems. The burdens associated with packing and palletisation of refillable containers and respective caps were estimated based on data relating to the same plant. Specific consumptions attributed to all packing operations are reported in detail in Appendix B (Section B.2.2), along with the sources of inventory data for such inputs.

3.9.3 Transport to retail outlets

In both baseline and waste prevention scenarios, packed detergents were assumed to be transported to retail outlets along an overall average distance of 340 km. This was estimated based on the location of the plants where the major brands of laundry detergents marketed in Italy are produced. The same distance was assumed also for fabric softeners and hand dishwashing detergents.

In order to estimate the average mass of detergent transported per functional unit, an average density was measured experimentally for each of the three categories of detergent. A brief description of the procedure and the obtained results can be found in Appendix B (Section B.2.3). Inventory data on the transport stage with a truck were derived from the *ecoinvent* database.

3.9.4 Detergent sale and purchase

Both refilling of the automatic self-dispensing system and withdrawal by the consumer require electricity. An overall consumption of about 0.0037 kWh per litre of delivered detergent was estimated, based on the technical features of the equipment used in one of the examined experiences. The masses of the main components of the self-dispensing system were also estimated based on the same data (Table B.16 of Appendix B). The estimate focused on major steel components (frame and other steel parts), on HDPE tanks and on expanded polyvinylchloride (PVC) covering panels. All these components were assumed to be produced from virgin material except for the steel, which is partly produced from sorted iron scraps. At the end of their useful life, all the components are recycled, except for PVC panels, which are incinerated in a waste to energy plant. A useful life of 10 years was specifically assumed for the self-dispensing system, along with an annual supply of about 75,000 litres.

The source of inventory data for the unit processes pertaining to the sale stage is mainly the *ecoinvent* database, but data from the literature and from equipment manufacturers were also used (further details are available in Section B.2.4 of Appendix B).

3.10 Results and discussion

3.10.1 Waste generation

In baseline scenarios, the generation of waste includes primary packages (containers and caps) and the respective transport packages (cardboard boxes, stretch film and pallets). In waste prevention scenarios, reusable tanks were also included, as well as the packages possibly used for reusable cap transport. In this case, the calculation was carried out for a number of uses of the refillable container ranging from 1 to 50. The results for the laundry detergents are represented in Figure 3.2, while those for fabric softeners and hand dishwashing detergents are shown in Figure B.3 of Appendix B. Table 3.3 reports, for the laundry detergents, the difference between the best waste prevention scenario and the two extreme baseline scenarios (i.e., those in which the lowest and the highest amount of waste is generated). The results obtained for the other two categories of detergents are reported in Tables B.17 and B.18 of Appendix B.

For laundry detergents and fabric softeners, the best waste prevention scenario is the one with a bigger container (i.e., waste prevention scenario 2). On the contrary, for hand dishwashing detergents, it is the one with a lighter container (i.e., waste prevention scenario 1). The comparison with baseline scenarios was thus made by focusing directly on these less waste-generating waste prevention scenarios.

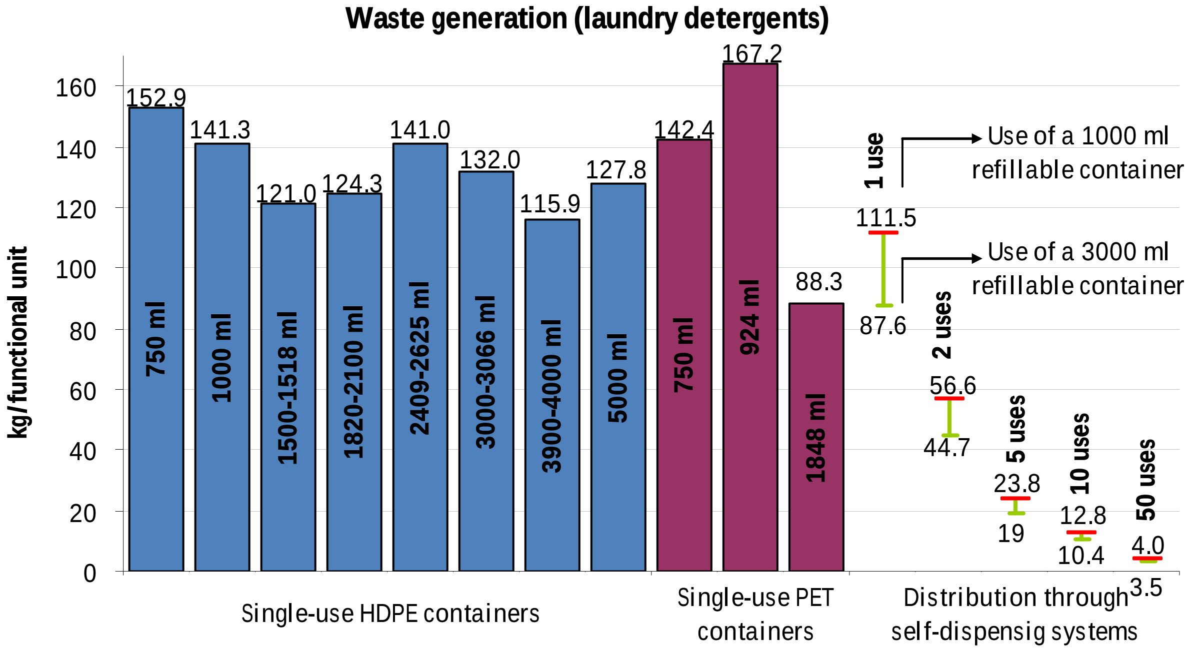


Figure 3.2: waste generated in laundry detergent distribution. Bars are the baseline scenarios, while horizontal dashes are the two waste prevention scenarios for different number of uses of the refillable container.

If the refillable container is used just once, the distribution of laundry detergents and fabric softeners through self-dispensing systems does not significantly reduce waste generation compared to the best baseline scenario (Tables 3.3 and B.17). For hand dishwashing detergents, waste generation will even increase by about 24% (21.5 kg/functional unit, Table B.18). Compared to the worst baseline scenario, a reduction can instead be observed: 48% for laundry detergents (80 kg/functional unit), 32% for fabric softeners (49 kg/functional unit) and 24% for hand dishwashing detergents (45 kg/functional unit). This is because in the waste prevention scenario a bigger container is used and many more empty containers are transported in each cardboard box. The amount of primary and transport packages wasted per functional unit is thus lower, even if the container is used only once.

A much more important reduction in waste generation is obviously obtained with the increase in the uses of the refillable container. For 50 uses, a maximum reduction in the range of 97–98% is obtained compared to the worst baseline scenario. A similar percentage reduction (about 96% for all detergent categories) is observed also when the comparison is made with the best baseline scenario. However, the decrease per functional unit is lower (85–103 kg versus 150–164 kg).

As expected, container reuse is the main driver for achieving such significant reductions in waste generation. Reuse allows a larger volume of detergent to be delivered by each container over its whole life cycle, with a consequent lower amount of required primary and transport packages. A lower amount of waste is thus generated overall in waste prevention scenarios, even if reusable tanks are used in addition to containers, caps and their transport packages. The additional contribution provided by tanks is indeed limited (about 1.8 kg per functional unit), since a very large volume of detergent is delivered over their whole life cycle (30,000 litres in this study).

Finally, it is worth noting that, starting from 5 uses of the refillable container, the difference between the two waste prevention scenarios is decreasing and tending to zero. Provided that this minimum target is achieved, and hopefully exceeded, the effectiveness of the distribution through self-dispensing systems is then not significantly affected by the size (or the mass) chosen for refillable containers.

Table 3.3: difference between the amount of waste generated in the scenario where laundry detergents are distributed loose with a 3000 ml refillable container (waste prevention scenario generating less waste) and that generated in the two respective baseline scenarios with the lowest and the highest generation of waste.

Reference baseline scenario	Number of uses of the 3000 ml refillable container				
	1	2	5	10	50
Distribution with a 1848 ml PET container (baseline scenario generating less waste)	-0.69 kg/fu[a,b] (-0.78 %)	-43.6 kg/fu (-49.4 %)	-69.3 kg/fu (-78.5 %)	-77.9 kg/fu (-88.2 %)	-84.7 kg/fu (-96.0 %)
Distribution with a 924 ml PET container (baseline scenario generating most waste)	-79.6 kg/fu (-47.6 %)	-122.5 kg/fu (-73.3 %)	-148.2 kg/fu (88.6 %)	-156.8 kg/fu (-93.8 %)	-163.6 kg/fu (-97.9 %)

(a) fu = functional unit.

(b) Negative variations per functional unit represent the waste prevention potentials achievable with distribution of laundry detergents through self-dispensing systems. They are expressed as the amount of waste prevented per 1000 litres of detergent distributed loose rather than packed in a single-use container of the type considered in the baseline scenario of reference.

3.10.2 Impact assessment results

For all the three categories of detergent, most of calculated impact indicators show a profile similar to the one of *climate change*, represented in the upper part of Figure 3.3 for laundry detergents. The *human toxicity, cancer effects* indicator is instead characterised by a slightly different profile, as shown in the lower part of Figure 3.3, always for laundry detergents. The profile of the same indicators calculated for fabric softeners and hand dishwashing detergents, as well as that of the remaining indicators for laundry detergents, is available in Section B.3.2 of Appendix B. An overview of the impacts of all baseline scenarios and of the two waste prevention scenarios for increasing uses of the refillable container can be found in Tables 3.4 and 3.5 for laundry detergents. For fabric softeners and hand dishwashing detergents, an overview is provided, respectively in Tables B.20- B.21, and B.24-B.25 of Appendix B.

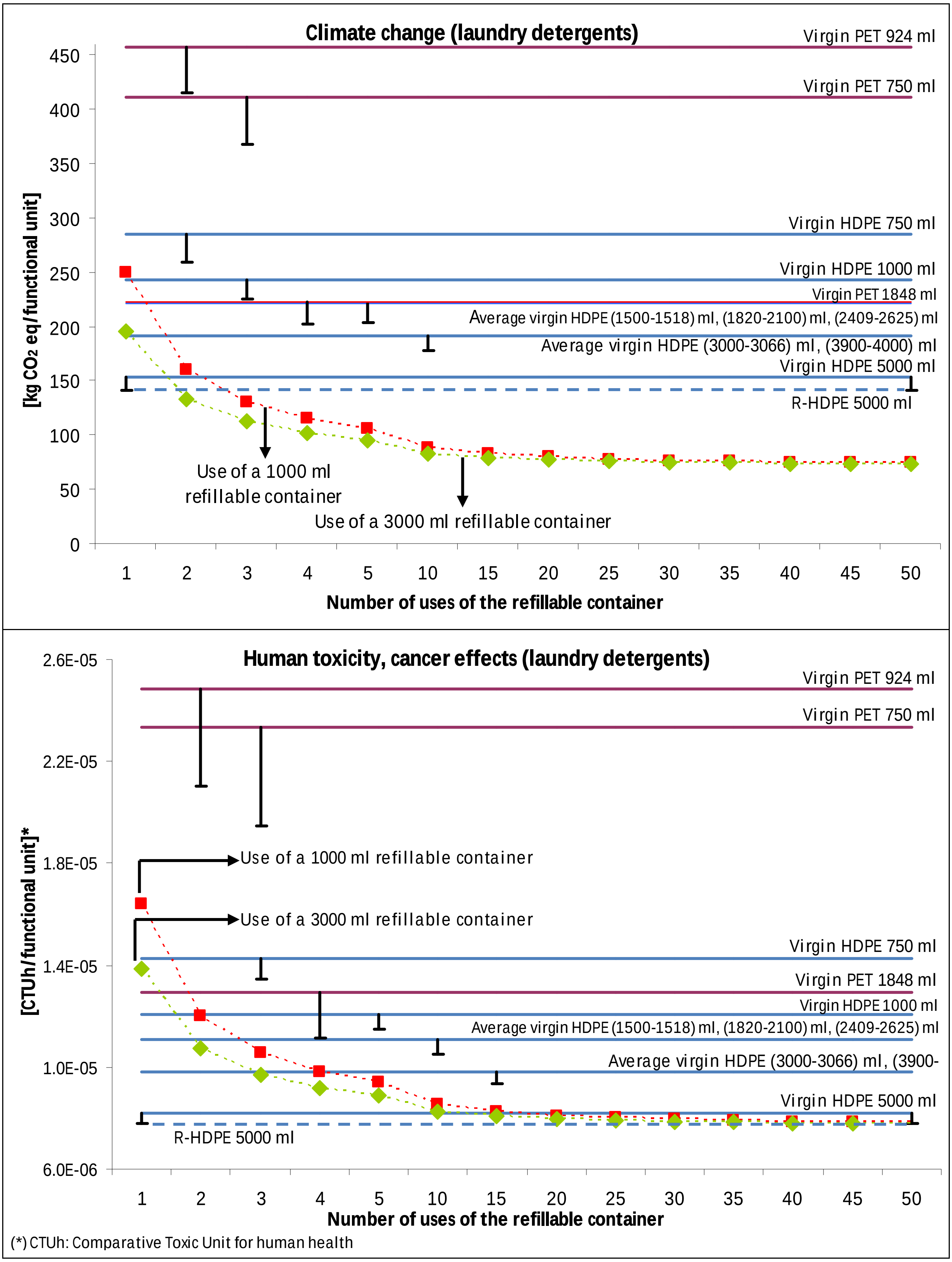


Figure 3.3: *climate change* and *human toxicity, cancer effects* impact indicators for laundry detergents. Horizontal lines represent the impacts of baseline scenarios, while squares and rhombuses the impacts of the two waste prevention scenarios as a function of the number of uses of the refillable container. Error bars represent the variation of the impacts when single-use containers are produced entirely from recycled material.

Table 3.4: potential impacts of baseline scenarios for laundry detergents. Values in parentheses refer to containers being produced entirely from recycled material (as considered in the sensitivity analysis).

Impact category	**Unit of measure**	**Scenario**										
		Distribution with single-use HDPE containers with a size of:								**Distribution with single-use PET containers with a size of:**		
		750 ml	**1000 ml**	**1500-1518 ml**	**1820-2100 ml**	**2409-2625 ml**	**3000-3066 ml**	**3900-4000 ml**	**5000 ml**	**750 ml**	**924 ml**	**1848 ml**
Climate change	kg CO_2 eq.	285 (259)	244 (225)	221 (205)	223 (205)	218 (202)	193 (179)	189 (177)	153 (141)	411 (368)	457 (415)	222 (202)
Ozone depletion	kg CFC-11 eq.	2.77E-5 (2.77E-5)	2.50E-5 (2.50E-5)	2.32E-5 (2.32E-5)	2.28E-5 (2.28E-5)	2.24E-05 (2.24E-05)	2.01E-5 (2.00E-5)	1.97E-5 (1.97E-5)	1.62E-5 (1.62E-5)	8.53E-5 (8.31E-5)	9.11E-5 (8.89E-5)	4.36E-5 (4.25E-5)
Photochemical ozone formation	kg NMVOC eq.	1.20 (1.08)	1.08 (0.992)	0.998 (0.927)	1.01 (0.934)	1.02 (0.944)	0.942 (0.879)	0.927 (0.873)	0.834 (0.781)	1.40 (1.27)	1.52 (1.39)	0.941 (0.882)
Acidification	mol H^+ eq.	1.43 (1.33)	1.23 (1.16)	1.12 (1.06)	1.15 (1.08)	1.14 (1.08)	1.04 (0.979)	1.02 (0.968)	0.881 (0.834)	1.95 (1.75)	2.08 (1.88)	1.17 (1.07)
Terrestrial eutrophication	mol N eq.	3.91 (3.72)	3.56 (3.43)	3.32 (3.20)	3.37 (3.24)	3.43 (3.31)	3.19 (3.08)	3.19 (3.11)	2.83 (2.75)	4.66 (4.32)	5.02 (4.68)	3.28 (3.12)
Freshwater eutrophication	kg P eq.	0.103 (0.103)	0.0827 (0.0824)	0.0723 (0.0721)	0.0757 (0.0755)	0.0741 (0.0739)	0.0638 (0.0636)	0.0629 (0.0627)	0.0483 (0.0481)	0.192 (0.172)	0.202 (0.182)	0.0974 (0.088)
Marine eutrophication	kg N eq.	0.403 (0.386)	0.367 (0.355)	0.340 (0.329)	0.344 (0.332)	0.354 (0.344)	0.324 (0.315)	0.334 (0.326)	0.277 (0.269)	0.485 (0.451)	0.531 (0.497)	0.338 (0.322)
Freshwater ecotoxicity	CTU_e	340 (305)	295 (270)	265 (244)	266 (242)	271 (249)	232 (213)	245 (229)	167 (151)	520 (415)	589 (484)	285 (235)
Human toxicity (cancer effects)	CTU_h	1.43E-5 (1.34E-5)	1.21E-5 (1.15E-5)	1.09E-05 (1.04E-05)	1.12E-5 (1.06E-5)	1.11E-5 (1.06E-5)	9.91E-6 (9.45E-6)	9.70E-6 (9.30E-6)	8.17E-6 (7.79E-6)	2.34E-5 (1.95E-5)	2.49E-5 (2.10E-5)	1.30E-5 (1.11E-5)
Human toxicity (non-cancer effects)	CTU_h	1.96E-5 (1.95E-5)	1.81E-5 (1.80E-5)	1.63E-05 (1.62E-05)	1.62E-5 (1.61E-5)	1.79E-5 (1.78E-5)	1.50E-5 (1.49E-5)	1.80E-5 (1.80E-5)	9.89E-6 (9.84E-6)	2.70E-5 (2.36E-5)	3.15E-5 (2.81E-5)	1.79E-5 (1.62E-5)
Particulate matter	kg $PM_{2.5}$ eq.	0.137 (0.125)	0.116 (0.107)	0.105 (0.0969)	0.107 (0.0988)	0.107 (0.0989)	0.0945 (0.0876)	0.0946 (0.0886)	0.0757 (0.0699)	0.178 (0.154)	0.194 (0.170)	0.103 (0.091)
Water resource depletion	m^3 water eq.	1.22 (1.15)	0.985 (0.934)	0.869 (0.826)	0.906 (0.858)	0.877 (0.833)	0.764 (0.726)	0.738 (0.705)	0.597 (0.566)	2.99 (2.73)	3.10 (2.85)	1.49 (1.37)
Mineral and fossil resource depletion	kg Sb eq.	0.965 (0.784)	0.810 (0.679)	0.729 (0.618)	0.744 (0.621)	0.723 (0.610)	0.639 (0.540)	0.617 (0.532)	0.512 (0.430)	1.32 (1.13)	1.46 (1.27)	0.707 (0.616)
Cumulative energy demand	MJ eq.	6412 (5378)	5354 (4602)	4794 (4157)	4910 (4204)	4796 (4148)	4216 (3649)	4098 (3610)	3339 (2869)	8507 (7385)	9383 (8272)	4557 (4027)

Table 3.5: potential impacts of the two waste prevention scenarios for laundry detergents as a function of the number of uses of the refillable container.

Impact category	Unit of measure	Waste prevention scenario 1					Waste prevention scenario 2				
		Number of uses of the 1000 ml refillable container					Number of uses of the 3000 ml refillable container				
		1	**2**	**5**	**10**	**50**	**1**	**2**	**5**	**10**	**50**
Climate change	kg CO_2 eq.	249	160	106	88.4	74.1	195	133	95.5	83	73.0
Ozone depletion	kg CFC-11 eq.	2.51E-5	1.76E-5	1.31E-05	1.17E-5	1.05E-05	2.03E-5	1.52E-05	1.22E-5	1.12E-5	1.04E-5
Photochemical ozone formation	kg NMVOC eq.	1.13	0.852	0.686	0.631	0.587	0.976	0.776	0.656	0.616	0.584
Acidification	mol H^+ eq.	1.30	0.903	0.663	0.583	0.518	1.08	0.79	0.62	0.56	0.51
Terrestrial eutrophication	mol N eq.	3.66	2.87	2.39	2.23	2.11	3.3	2.7	2.3	2.2	2.1
Freshwater eutrophication	kg P eq.	0.0875	0.0496	0.0269	0.0193	0.0132	0.0658	0.0388	0.0225	0.0171	0.0128
Marine eutrophication	kg N eq.	0.358	0.275	0.224	0.207	0.194	0.328	0.259	0.218	0.204	0.193
Freshwater ecotoxicity	CTU_e	307	204	142	121	105	262	181	133	117	104
Human toxicity (cancer effects)	CTU_h	1.64E-5	1.20E-5	9.40E-6	8.52E-6	7.82E-6	1.39E-5	1.08E-5	8.89E-6	8.27E-6	7.77E-6
Human toxicity (non-cancer effects)	CTU_h	1.69E-5	1.27E-5	1.02E-5	9.30E-6	8.62E-6	1.77E-5	1.31E-5	1.03E-5	9.38E-6	8.64E-6
Particulate matter	kg $PM_{2,5}$ eq.	0.118	0.0762	0.0510	0.0426	0.0359	0.0956	0.0649	0.0465	0.0404	0.0355
Water resource depletion	m^3 water eq.	1.05	0.614	0.350	0.262	0.191	0.789	0.481	0.297	0.235	0.186
Mineral and fossil resource depletion	kg Sb eq.	0.844	0.526	0.335	0.271	0.221	0.645	0.426	0.295	0.252	0.217
Cumulative energy demand	MJ eq.	5521	3370	2078	1648	1304	4219	2719	1818	1518	1278

3.10.2.1 Laundry detergents

Out of the two waste prevention scenarios, the one based on 1000 ml refillable containers has the highest impact in most categories (i.e. all except for the *human toxicity, non-cancer effects* one; Table 3.5). If such a container is used at least 10 times, distribution through self-dispensing systems is preferable for all impact categories except for those related to human toxicity[3] (Figure 3.3, Figures B.4 to B.9 of Appendix B and Table B.19). In the prevention scenario based on 3000 ml refillable containers, only 5 uses are needed as the minimum threshold for an improved environmental performance (Table 3.6).

For the *human toxicity, non-cancer effects* impact category, in both waste prevention scenarios, the distribution through self-dispensing systems outperforms the single-use based one only starting from 25 uses (Figure B.7 of Appendix B, Table B.19 and Table 3.6). For the category *human toxicity, cancer effects*, waste prevention scenarios are instead preferable to the vast majority of baseline scenarios starting from 10 uses, but they remain comparable to the best baseline scenario even up to 50 uses (Figure 3.3, Table B.19 of Appendix B and Table 3.6). The toxicity indicators are the most uncertain, since a complex mechanism relates emissions of toxic substances to their ultimate effects. One has thus to be aware that the use of a different impact assessment method could lead to different results for these indicators.

The variation of the impacts between the waste prevention scenario based on 3000 ml refillable containers and the best baseline scenario is reported in Table 3.6. With the exclusion of the human toxicity-related categories, a 12–53% reduction of the total impact for 5 uses and 24–73% for 50 uses is observed. A 54–90% decrease for 5 uses and 58–94% for 50 uses is instead observed compared to the worst baseline scenario (Table 3.7).

As expected, this overall impact reduction is mainly a result of the decrease in the impact of the life cycle of primary packages (containers and caps), which can reach 100%. On average, the life cycle of primary packages contributes to about 50% of the total impacts of baseline scenarios if human toxicity categories are excluded. The observed percentage reductions in impact are thus significant also per functional unit.

A significant percentage reduction in the impact of the life cycle of transport packages is also observed in waste prevention scenarios (up to 98%). However, for most impact categories, the contribution provided by the life cycle of such packages to the total impact of baseline scenarios is modest (about 17% on average if human toxicity categories are excluded). Therefore, impact reductions per functional unit are limited, too.

[3] Due to uncertainties included in the analysis, only differences (positive or negative) between scenario impacts larger than 10%were considered significant in this study. Therefore, distribution through self-dispensing systems was considered preferable to that based on single-use containers only when the impact of the respective waste prevention scenario was lower than the impact of the best baseline scenario for at least 10%.

The impact of the transport stage, which on average contributes to about 29% of the total impacts of the baseline scenarios, is instead comparable in both examined distribution systems. The same is valid also for the impact provided by the remaining life cycle stages, which altogether are on average responsible for less than 3% of the total impacts of baseline scenarios and for less than 16% of those of waste prevention scenarios.

Table 3.6: percentage variation between the impacts of the 3000 ml-based prevention scenario for laundry detergents and those of the respective best baseline scenario for each category (i.e. the one based on 5000 ml single-use HDPE containers made from recycled material).

Impact category	Number of uses of the 3000 ml refillable container													
	1	**2**	**3**	**4**	**5**	**10**	**15**	**20**	**25**	**30**	**35**	**40**	**45**	**50**
Climate change	37.9	-6.1	-20.8	-28.1	-32.5	-41.3	-44.3	-45.7	-46.6	-47.2	-47.6	-47.9	-48.2	-48.4
Ozone depletion	24.9	-6.3	-16.7	-21.9	-25.0	-31.2	-33.3	-34.3	-35.0	-35.4	-35.7	-35.9	-36.1	-36.2
Photochemical ozone formation	24.9	-0.7	-9.2	-13.5	-16.1	-21.2	-22.9	-23.8	-24.3	-24.6	-24.9	-25.0	-25.2	-25.3
Acidification	29.1	-5.3	-16.8	-22.5	-26.0	-32.9	-35.2	-36.3	-37.0	-37.5	-37.8	-38.0	-38.2	-38.4
Terrestrial eutrophication	20.0	-2.2	-9.7	-13.4	-15.6	-20.0	-21.5	-22.3	-22.7	-23.0	-23.2	-23.4	-23.5	-23.6
Freshwater eutrophication	36.8	-19.5	-38.2	-47.6	-53.2	-64.4	-68.2	-70.0	-71.2	-71.9	-72.5	-72.9	-73.2	-73.4
Marine eutrophication	22.0	-3.5	-12.1	-16.3	-18.9	-24.0	-25.7	-26.5	-27.1	-27.4	-27.6	-27.8	-28.0	-28.1
Freshwater ecotoxicity	73.1	19.8	2.0	-6.9	-12.2	-22.9	-26.4	-28.2	-29.3	-30.0	-30.5	-30.9	-31.2	-31.4
Human toxicity (cancer effects)	77.9	38.0	24.7	18.1	14.1	6.1	3.5	2.1	1.3	0.8	0.4	0.1	-0.1	-0.3
Human toxicity (non-cancer effects)	79.8	32.8	17.2	9.4	4.7	-4.7	-7.8	-9.4	-10.3	-11.0	-11.4	-11.7	-12.0	-12.2
Particulate matter	36.8	-7.1	-21.7	-29.1	-33.5	-42.2	-45.2	-46.6	-47.5	-48.1	-48.5	-48.8	-49.1	-49.3
Water resource depletion	39.5	-14.9	-33.0	-42.1	-47.6	-58.4	-62.1	-63.9	-65.0	-65.7	-66.2	-66.6	-66.9	-67.1
Mineral and fossil resource depletion	50.1	-0.8	-17.7	-26.2	-31.3	-41.5	-44.9	-46.6	-47.6	-48.3	-48.8	-49.1	-49.4	-49.6
Cumulative energy demand	47.1	-5.2	-22.7	-31.4	-36.6	-47.1	-50.6	-52.3	-53.4	-54.1	-54.6	-54.9	-55.2	-55.5

Even the lower benefits achievable for human toxicity impact categories can be explained by looking at the variation of the impacts of the most important life cycle stages. For these categories, a reduction in the impact of the life cycle of primary packages up to 99% is still observed. However, this is partially or totally compensated by an increase in the impact of the life cycle of packages used for detergent transport. Such an increase can be as high as 300% for the *human toxicity, cancer effects* and 41% for the *human toxicity, non-cancer effects*. Responsibility for this increase is in charge to the tanks made of a galvanized steel component (Figure B.1 of Appendix B). In fact, for carcinogenic effects, about 73% of the human health impact associated with the life cycle of transport packages used in waste prevention scenarios is caused by the steel cage of the tanks. In particular, waterborne emissions of chromium from the landfilling of slag generated during steel production and recycling provide the largest contribution. Direct airborne emissions of zinc resulting from its primary production and from its subsequent use for the coating of the cage are instead responsible for about 80% of the overall human health impact in the case of non-carcinogenic effects.

Table 3.7: percentage variation between the impacts of the 3000 ml-based prevention scenario for laundry detergents and those of the respective worst baseline scenario for each category (i.e. the one based on 924 ml single-use PET containers made from virgin material).

Impact category	**Number of uses of the 3000 ml refillable container**													
	1	**2**	**3**	**4**	**5**	**10**	**15**	**20**	**25**	**30**	**35**	**40**	**45**	**50**
Climate change	-57.3	-70.9	-75.5	-77.7	-79.1	-81.8	-82.7	-83.2	-83.5	-83.6	-83.8	-83.9	-83.9	-84.0
Ozone depletion	-77.8	-83.3	-85.2	-86.1	-86.6	-87.8	-88.1	-88.3	-88.4	-88.5	-88.5	-88.6	-88.6	-88.6
Photochemical ozone formation	-35.6	-48.8	-53.2	-55.4	-56.7	-59.4	-60.3	-60.7	-61.0	-61.1	-61.3	-61.4	-61.4	-61.5
Acidification	-48.3	-62.1	-66.6	-68.9	-70.3	-73.1	-74.0	-74.5	-74.7	-74.9	-75.1	-75.2	-75.2	-75.3
Terrestrial eutrophication	-34.4	-46.5	-50.6	-52.6	-53.8	-56.3	-57.1	-57.5	-57.7	-57.9	-58.0	-58.1	-58.2	-58.2
Freshwater eutrophication	-67.3	-80.8	-85.2	-87.5	-88.8	-91.5	-92.4	-92.8	-93.1	-93.3	-93.4	-93.5	-93.6	-93.7
Marine eutrophication	-38.1	-51.1	-55.4	-57.6	-58.9	-61.5	-62.3	-62.8	-63.0	-63.2	-63.3	-63.4	-63.5	-63.6
Freshwater ecotoxicity	-55.5	-69.2	-73.8	-76.0	-77.4	-80.2	-81.1	-81.5	-81.8	-82.0	-82.1	-82.2	-82.3	-82.4
Human toxicity (cancer effects)	-44.3	-56.7	-60.9	-63.0	-64.2	-66.7	-67.6	-68.0	-68.2	-68.4	-68.5	-68.6	-68.7	-68.7
Human toxicity (non-cancer effects)	-43.8	-58.5	-63.4	-65.8	-67.3	-70.2	-71.2	-71.7	-72.0	-72.2	-72.3	-72.4	-72.5	-72.6
Particulate matter	-50.7	-66.5	-71.8	-74.4	-76.0	-79.2	-80.2	-80.8	-81.1	-81.3	-81.4	-81.6	-81.6	-81.7
Water resource depletion	-74.6	-84.5	-87.8	-89.4	-90.4	-92.4	-93.1	-93.4	-93.6	-93.7	-93.8	-93.9	-94.0	-94.0
Mineral and fossil resource depletion	-55.7	-70.7	-75.7	-78.2	-79.7	-82.7	-83.7	-84.2	-84.5	-84.7	-84.9	-85.0	-85.1	-85.1
Cumulative energy demand	-55.0	-71.0	-76.4	-79.0	-80.6	-83.8	-84.9	-85.4	-85.7	-86.0	-86.1	-86.2	-86.3	-86.4

3.10.2.2 Hand dishwashing detergents

Most of the comparative considerations between the two alternative distribution methods made for laundry detergents can be extended also to hand dishwashing detergents, although a few differences are observed. First of all, starting from the same minimum number of uses, the distribution with refillable containers is preferable to the single-use based one with respect to all impact categories, except for the only *human toxicity, cancer effects*. For laundry detergents, also the *human toxicity, non-cancer effects* impact category was excluded. Moreover, with the exclusion of these human toxicity categories, impact reductions achieved in waste prevention scenarios are moderately lower than laundry detergents (Tables B.26 and B.27 of Appendix B). This is because the impacts of waste prevention scenarios are higher compared to laundry detergents, since smaller (or lighter) refillable containers are used. Moreover, the impacts of the two extreme baseline scenarios are moderately lower than laundry detergents.

Finally, for the *human toxicity, cancer effects* impact category, distribution through self-dispensing systems starts being comparable with the best baseline scenario from a greater number of uses of the refillable container than laundry detergents. In the best waste prevention scenario[4] this happens starting from 15 uses, while in the worst[5] 20 uses are needed. For laundry detergents, such a threshold was 10 uses, for both waste prevention scenarios.

3.10.2.3 Fabric softeners

Out of the two waste prevention scenarios, the one based on 2000 ml refillable containers shows the lowest impact for all categories except for *human toxicity, non-cancer effects* and *marine eutrophication*. However, starting from 4 uses, in both waste prevention scenarios, the distribution through self-dispensing systems is preferable to that based on single-use containers with respect to all impact categories except for the *human toxicity, cancer effects* one. For this category, the two waste prevention scenarios start to be comparable with the best baseline scenario from 10 uses of the container, similarly to laundry detergents.

With the exclusion of human toxicity categories, impact reductions achieved in the best waste prevention scenario are similar to laundry and hand dishwashing detergents when the comparison is made with the best baseline scenario (Table B.22 of Appendix B). Compared to the worst baseline scenario, achieved reductions are instead lower than laundry detergents, but comparable to hand dishwashing detergents (Table B.23 of Appendix B). This is mainly because the impacts of the reference baseline scenario are lower than laundry detergents.

3.10.2.4 General remarks

Focusing on the impacts of waste prevention scenarios, a reduction is observed by increasing the number of uses of the refillable containers. Most of this reduction takes place between 2 and 5-10 uses, depending on the impact category. After this threshold, such impacts tend to stabilize over an asymptotic value and increasingly smaller and negligible differences are observed between the impacts of the two alternative waste prevention scenarios. Conversely, if the container is used for less than 10 times, differences are more pronounced and waste prevention scenarios where a bigger or a lighter refillable container is used are preferable (at least for most impact categories).

[4] The waste prevention scenario where a 1000 ml refillable container weighing 62 g is provided to the consumer shows the lowest impact for all the considered impact categories.

[5] The waste prevention scenario where a 1000 ml refillable container weighing 71.5 g is provided to the consumer shows the highest impact for all the considered impact categories.

3.11 Concluding remarks and recommendations

Life cycle assessment was used to evaluate whether detergent distribution through self-dispensing systems actually allows to achieve the expected reduction in waste generation and environmental impacts compared to the distribution with single-use containers. Laundry detergents, fabric softeners and hand dishwashing detergents were analysed, by defining a set of baseline single-use scenarios and two alternative waste prevention scenarios.

The results showed that if the refillable container is used at least 5 times, the distribution through self-dispensing systems allows for an actual reduction of municipal waste generation compared to the distribution with the main types of single-use plastic containers available in the Italian market. Depending on the category of detergent and on the reference baseline scenario, a 74–89% reduction for 5 uses of the container and 95.5–98% for 50 uses is achieved. When referred to the functional unit, the reduction ranges from 66 kg to 148 kg for 5 uses and from 85 kg to 164 kg for 50 uses.

Distribution through self-dispensing systems allows also for a progressive reduction of the energy demand and of most of the potential impacts, starting from a minimum number of uses of the refillable container. For laundry and hand dishwashing detergents, at least 5–10 uses are needed, depending on the scenario. For fabric softeners, 4 uses are enough in both waste prevention scenarios. The potential impact on human health due to total life cycle emissions of toxic substances with non-carcinogenic effects is reduced as well. This happens starting from 4 uses of the refillable container for fabric softeners and from 5 uses for hand dishwashing detergents, but at least 25 uses are needed for laundry detergents. When total emissions of carcinogenic substances are considered, distribution through self-dispensing systems involves instead a potential impact on human health comparable to the distribution with big-sized single-use HDPE containers[6] made from recycled material even for 50 uses of the refillable container. The results obtained for human toxicity impact categories are however characterized by greater uncertainty than other categories and may vary depending on the impact assessment model used for their calculation. Moreover, these results do not take into account that the different types and sizes of single-use containers are actually used in different proportions by the consumers. If the actual mix of substituted packages was taken into account, the substitution may prove more beneficial even for human toxicity categories, as big-sized single-use containers are likely used to a limited extent.

If distribution through self-dispensing systems is to be implemented as a waste prevention measure, the consumer shall be adequately made aware that the number of uses of the refillable container plays a key role on the ultimate environmental and energy performance. As a general rule, at least

[6] The size of the container is 5000ml for laundry and hand dishwashing detergents and 4000 ml for fabric softeners.

10–15 uses of the refillable container should be encouraged. However, all the efforts should be made to use the container as far as this is technically feasible.

An improvement of the benefits on human health impacts could be obtained by targeting the packaging used for detergent transport (reusable tanks). For instance, an alternative material could be employed for the production of the cage surrounding the tank. Moreover, all the efforts should be made by both detergent producers and retailers to extend as much as possible the useful life of tanks (which in this study was conservatively assumed equal to 50 transport cycles). Finally, also a reduction of the distance from packaging plants to retailers could be very beneficial. In fact, detergent transport is one of the two major contributors to the total impact of waste prevention scenarios in the human toxicity categories, along with the life cycle of reusable tanks. The travelled distance depends on the actual location of packaging plants and cannot be easily changed. However, retailers should be encouraged to prefer the distribution of detergents produced or packed as nearest as possible to the respective retail outlets. Distance reduction would obviously be advantageous also for many other impact categories, where the detergent transport stage is responsible for most of the overall impact.

4 Discussion on methods to include prevention activities in waste management LCA

Over the last two decades, the life cycle assessment (LCA) methodology has widely been used to evaluate the environmental and energy performance of real or fictional integrated municipal solid waste (MSW) management systems (e.g. among the most recent, Antonopoulos et al., 2013; Blengini et al., 2012; Giugliano et al., 2011; Pires et al., 2011; Bovea et al., 2010; De Feo and Malvano, 2009; Liamsanguan and Gheewala, 2008; Buttol et al., 2007). Nevertheless, as also recently reported by Saner et al. (2012), waste prevention has rarely been included in such evaluations, despite it is universally indicated as the preferable waste management option (e.g., at the European level, by the Waste Framework Directive 2008/98/EC; European Parliament and Council, 2008).

This exclusion is mainly a consequence of the fact that traditional waste management LCA has some inherent methodological characteristics (the functional unit and the system boundaries) that prevent the comparison of scenarios generating different overall amounts of waste (Ekvall et al., 2007). Indeed, the methodology has originally been developed to compare the environmental performance of different systems for the collection, treatment and disposal of the MSW arising, in a given amount and composition, from a given area (McDougall et al., 2001). Secondly, accounting for waste prevention can make the assessment more complicated as some processes upstream waste collection are likely affected (thus needing to be taken into account) and additional parameters need to be estimated (such as the waste prevention potential). A last, but not less important aspect is that attention so far was mainly oriented on finding environmentally sound strategies for the treatment and disposal of waste, but not for its reduction (Wilson et al., 2010). Conversely, now that high recycling rates are generally achieved and incineration with energy recovery is generally well established in most developed countries, increasing attention is devoted to waste prevention at policy and regulatory level.

In order to 'facilitate the comparison of MSW management scenarios incorporating waste prevention and the various methods of waste treatment', Cleary (2010) has recently proposed a conceptual model (the Waste Management and Prevention LCA model), which he later applied to a case study for the city of Toronto (Cleary, 2014). A similar method has also more recently been described by Gentil et al. (2011), who used it to compare different municipal waste management scenarios for a hypothetical European municipality. Conversely, Matsuda et al. (2012) adopted a partially different approach to calculate greenhouse gas emissions from alternative household waste management scenarios for the city of Kyoto.

Based on the structured reorganisation of the common methodological amendments proposed so far in the scientific literature and on further personal elaborations and research, two alternative LCA approaches (conceptual models) were identified, to evaluate the environmental and energy performance of integrated MSW management systems that, beyond conventional treatments, include the effects of waste prevention activities. This section presents and discusses the two identified approaches, with reference to the classification of municipal waste prevention activities proposed in Section 1.3. A presentation of the methodological aspects that prevent traditional waste management LCA from addressing waste prevention activities and a review of the amendments and of the approaches proposed so far in the scientific literature in the attempt to overcome this limitation are also initially reported, this being the basis upon which the two approaches were developed.

Considering the increased strategic and policy relevance of waste prevention worldwide, the availability of alternative LCA tools capable of evaluating all the options of the waste hierarchy is deemed to be of use. Moreover, in a 2009 editorial of the *International Journal of Life Cycle Assessment*, the inability of traditional waste management LCA to account for the effects of waste prevention activities is identified as one of the limitations to the applicability of the methodology as a decision support tool in waste management planning and policy making, to be addressed in further researches (Gheewala, 2009).

4.1 Methodological limitations of waste management LCA in addressing waste prevention activities

There are two interrelated methodological aspects that prevent traditional waste management LCA from accounting for the effects of waste prevention activities: the choice of the functional unit and the resulting definition of the system boundaries. The functional unit is often defined as the management (collection and treatment) of a given amount of waste with a given composition, representative of the real or fictional geographical area under study (e.g. the management of 1 tonne of waste from a given municipality). Examples in this sense are provided by Gunamantha, 2012; Menikpura et al., 2012; Abduli et al., 2011; Koci and Trecakova, 2011; Kaplan et al., 2009; Rigamonti et al., 2009; Chaya and Gheewala, 2007; Bovea and Powell, 2006; Hong et al., 2006 and Weitz et al., 1999. In other cases (e.g. Koroneos and Nanaki, 2012; Zhao et al., 2011; Zhao et al., 2009; Buttol et al., 2007; Muñoz et al., 2004; Beccali et al., 2001), the functional unit is more generally defined as the management of the waste generated in a given geographical area over a given time period (e.g. one specific year). The corresponding amount is then considered as the constant input waste to all of the possibly compared scenarios. Therefore, in principle, the use of a functional unit based on a given amount of waste to be managed does not allow for the comparison of scenarios where different overall amounts of waste are generated and have to be managed, as in the case where some of them include waste prevention activities (Ekvall et al., 2007).

Being the amount of generated waste identical in all of the analysed scenarios, comparative assessments are then generally simplified by defining the system boundaries according to the so-called 'zero burden' approach or assumption (Ekvall et al., 2007). This means that all the processes and activities occurring before the moment in which products become waste (upstream processes/activities) are usually excluded from the system boundaries, since they are common to all scenarios. However, when waste prevention activities are included in the assessment, both different amounts of waste are generated in the compared scenarios and the magnitude or the typology of some of the upstream processes/activities is likely affected. The 'zero burden' assumption is, in general, no longer valid and at least those parts of upstream processes/activities which differ among scenarios should be included in the system boundaries (Finnveden, 1999). Conversely, the impacts of a scenario that produces less waste are overestimated compared to the others (Finnveden, 1999) when waste prevention is actually achieved without increasing the overall upstream impacts. Even worse, when the 'avoided burden method' (Finnveden et al., 2009) is employed to solve the multifunctionality of material and energy recovery processes, negative values of the LCA impact indicators can be obtained (environmental benefits). In this situation, a reduction in waste generation would be paradoxically associated with a reduction in the downstream benefits, as these are proportional to the amount of waste recovered. Thus, the most is the waste, the best.

4.2 Review of proposed amendments to traditional waste management LCA

In order to overcome the limitation imposed by a functional unit based on a given quantity of waste to be managed, Ekvall et al. (2007) suggest to define a functional unit such as "the annual quantity of waste generated in a geographical area", without specifying any amount. Doing so, it would be possible to compare scenarios where different amounts of waste are generated, but, as observed in Section 4.1, the 'zero burden' assumption is, in general, no longer valid. Therefore, the authors conclude reasonably wondering whether the environmental burdens associated with the production of all the products that eventually become waste should be taken into account.

Trying to answer this question, Cleary (2010) has more recently claimed that a complete abandonment of the 'zero burden' assumption may not be necessary, and proposed a conceptual model, referred to as *WasteMAP* LCA (Waste Management and Prevention LCA), to compare MSW management scenarios including waste prevention activities and the various methods of waste treatment. According to this approach, only the upstream components of the product systems affected by waste prevention activities need to be included within the system boundaries (in those scenarios accounting for waste prevention activities). The components of 'the product systems that are subtracted from the total MSW subject to treatment' (i.e. the 'targeted product systems') will be included as avoided processes. Conversely, those of 'the product systems that may need to be added to the total MSW' (i.e. the 'alternate product systems') will be included as additional processes (Figure 4.1). A targeted product system may be represented, for instance, by bottled water delivery, while a respective alternate product system by public network water delivery (if the prevention activity is based on the substitution of bottled water by public network water).

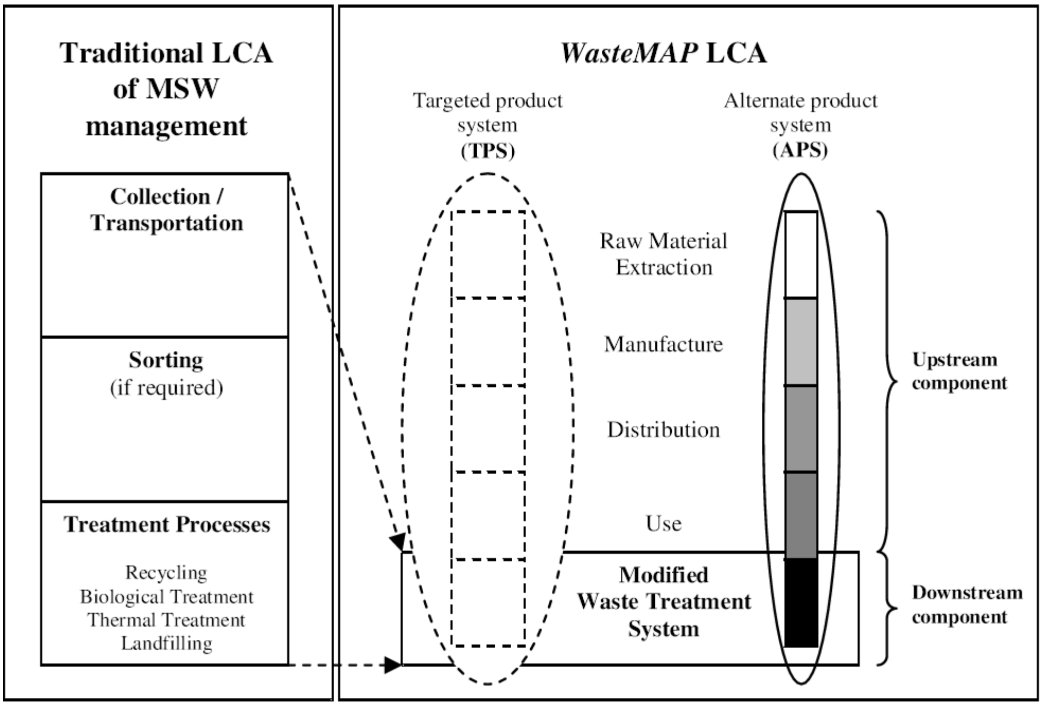


Figure 4.1: system boundaries of the *WasteMAP* (Waste Management and Prevention) LCA model, for a municipal waste management system including a prevention activity based on product or service substitution (adapted from Cleary, 2010).

The *WasteMAP* LCA model adopts a *primary functional unit* depicting 'the amount (mass or volume) of material addressed by the MSW management system on an annual basis'. This amount is identical for all the compared scenarios and is equal to the sum of the amount of waste prevented and of that collected and treated under each scenario. Moreover, the definition of a *secondary functional units* for each considered prevention activity is recommended, in order to 'ensure that MSW management scenarios subject to comparison will supply functionally equivalent product services to the residents of the municipality' under study. Secondary functional units should measure the product services of all the targeted product systems removed and ensure that these removed services are equivalent to those provided by the replacing alternate product systems. For example, when the substitution of bottled water by public network water is considered, the secondary functional unit can be the supply of a given amount of drinking water to the residents of a municipality over a particular year. The amount supplied through prevented water bottles shall then be identical to that supplied through the public network.

The *WasteMAP* LCA model is thus mainly conceived to deal with waste prevention activities based on dematerialisation, i.e. activities that do not affect the amount of product services supplied to the citizens of the studied area. Scenarios including waste prevention activities based on a reduction in product or service consumption 'without product service substitution' cannot be compared with a reference scenario on the basis of a complete functional equivalence, since 'there would be no replacement product service provided' (and no secondary functional units can be defined). However, if the product service provided by the targeted product or service is deemed unwanted by the population, no secondary functional unit is required to ensure the functional equivalence of product services (and of scenarios). This happens, for instance, when unwanted unaddressed advertising is prevented.

An approach somehow similar to that proposed by Cleary (2010) is the one adopted by Gentil et al. (2011). In order to compare a waste management scenario including different waste prevention activities with a baseline scenario without waste prevention, the authors define an apparently traditional functional unit, i.e. 'the service of managing 100,000 tonnes of average MSW from a fictional European municipality'. This seems to be in contrast with the introduction of waste prevention, which involves a reduction in the amount of waste generated and to be managed. Actually, in the waste prevention scenario, prevented waste fractions are considered as virtual waste flows, which are not subject to any transformation within the traditional waste management system and, hence, do not involve any downstream environmental burden. Therefore, similarly to the approach of Cleary (2010), the functional unit is effectively composed of the amount of waste actually generated and of that prevented ('virtual' waste), the sum of which is identical in both the compared scenarios.

Regarding the system boundaries, the authors argue that the upstream processes in the life cycle of all the waste fractions could still be excluded, since the same amount and composition of waste (real or virtual) enter the management system in the compared scenarios (in which constant consumption levels are assumed). Nevertheless, extraction, manufacture, distribution and use of prevented waste fractions are likely avoided and shall be included in the system boundaries. Therefore, in the waste prevention scenario, the virtual flows of prevented waste are included in the mass balance of the system and routed to a fictional burden free waste management technology, which is credited with the avoided production (extraction, manufacture, distribution and use) of the prevented waste fractions themselves (Figure 4.2).

No explicit mention is made by the authors about the need of also accounting for the additional upstream burdens possibly involved by the implementation of waste prevention activities due, for instance, to the consumption of alternative products or services. Nevertheless, it seems that these additional burdens have been taken into account in the case study described in their paper.

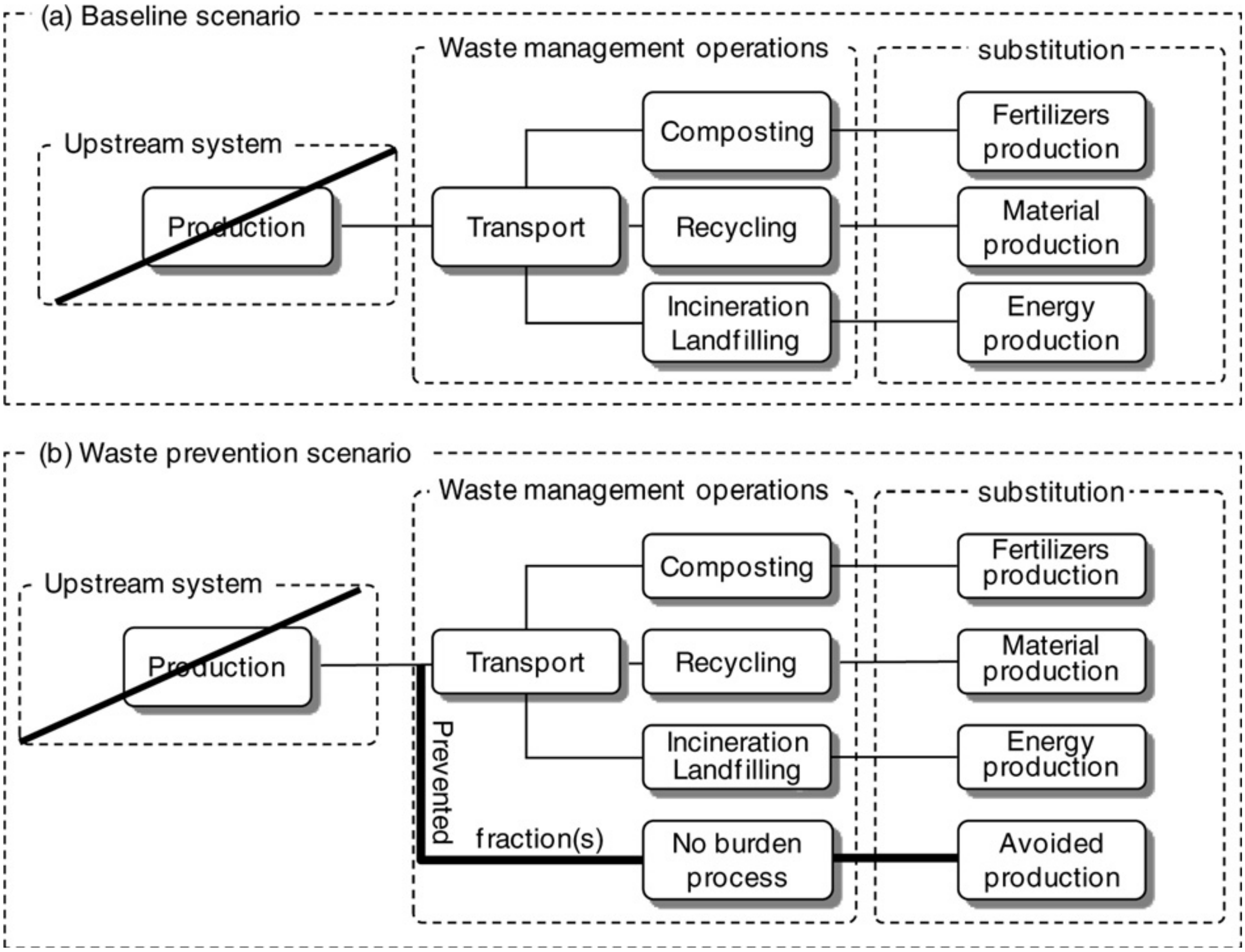


Figure 4.2: boundaries of a municipal waste management system without (a) and with (b) waste prevention, according to the approach adopted by Gentil et al. (2011).

In the study by Matsuda et al. (2012), different scenarios are compared for the management of the household combustible waste of Kyoto, Japan. One of these scenario accounts for a partial reduction in the quantity of edible food waste (food losses), as a consequence of the introduction of separate collection of the food waste. While the authors define a primary and a secondary functional unit in keeping with those proposed by Cleary (2010)[1], a different choice is made for the system

[1] The primary functional unit is defined as “the annual management of household combustible waste in Kyoto, Japan”. The “annual food ingestion (mass and composition) by the residents of Kyoto”, is instead used as the secondary functional unit.

boundaries. The whole upstream life cycle of the waste fraction targeted for prevention (food losses) is indeed included in all the compared scenarios, as the magnitude of these upstream processes is affected by the waste prevention activity. This means that each scenario includes, in addition to the collection and treatment of all the considered waste streams, also the production, distribution and cooking of the whole amount of food losses actually generated as waste.

Finally, some recommendations about the system boundaries needed to evaluate the environmental performance of waste prevention activities are provided by the European Commission's Joint Research Centre (EC-JRC, 2011b). All the processes upstream of collection and treatment should be included in the system boundaries whenever they are modified by waste prevention activities. Such modified processes may be excluded only if changes are negligible and do not significantly affect the results.

4.3 Modelling approaches

Based on the amendments discussed in Section 4.2 and on further personal research, two alternative modelling approaches (*Approach 1* and *Approach 2*) were identified for the LCA-based comparison of integrated MSW management scenarios implementing one or more waste prevention activities (in the following *waste prevention scenarios*) with scenarios in which no waste prevention activities are undertaken and the same amount of waste is generated (in the following *baseline scenarios*). The approaches are characterised in terms of the perspective from which they look at waste prevention activities, on the resulting functional unit and system boundaries, and on the procedure for the calculation of the potential impacts. All these methodological aspects are discussed in the following sections (4.3.1 to 4.3.4).

4.3.1 Perspective on waste prevention activities

According to the first approach (*Approach 1*), waste prevention activities are considered to be an actual waste management method. The amount of waste managed through conventional treatments[2] after its public collection, through domestic or on-site treatments[3] and through waste prevention activities, is therefore the same in all compared scenarios. In fact, in each scenario, the sum of the amount of waste generated and of that prevented is identical. In particular, it is defined as the amount of waste potentially produced in the studied area (or system), as better explained in Section 4.3.2.

[2] i.e. possible sorting, mechanical and biological treatments, recycling, thermal treatments and landfilling.
[3] e.g. home or community composting.

In the second approach (*Approach 2*) waste prevention activities are not considered as a real waste management method. Therefore, the amount of waste to be managed (only through conventional or domestic/on-site treatments) varies between baseline and waste prevention scenarios, as a consequence of the implementation of waste prevention activities.

4.3.2 Functional unit

According to *Approach 1*, the functional unit is defined as:

"*the integrated management of the waste potentially produced over a given period in a given geographical area in which waste prevention activities will be undertaken (or by one of its inhabitants)*", without specifying any amount. An example is "the integrated management of the waste potentially produced during one year by one inhabitant of the Lombardia Region, Italy, in which an activity will be implemented to reduce the amount of one-way water bottles generated as waste".

The amount of waste potentially produced in the studied area is identical in all the compared scenarios. It is defined with reference to a baseline scenario, where it corresponds to the amount of waste actually generated and managed through conventional treatments (after public collection), or through domestic and on-site treatments (e.g. home or community composting). In waste prevention scenarios, a portion of the amount of the potentially produced waste is instead prevented, being managed through one or more waste prevention activities.

A functional unit defined as above may be more suitable for the assessment of waste management scenarios relating to a specific geographical context, whether real or hypothetical. However, it is less suitable for the evaluation of the consequences of introducing waste prevention in a generic waste management system. In this case, more similarly to a traditional waste management LCA, the functional unit can be alternatively defined as "*the integrated management of a given (numerical) amount of waste*", such as "the management of 1 tonne of waste", without referring to any specific geographical context. Similarly to the approaches proposed by Gentil et al. (2011) and Cleary (2010), such an amount will then be composed of the amount of waste actually generated and of that prevented and will be the same in all compared scenarios. As explained above, in baseline scenarios the whole amount of waste specified by the functional unit is managed through conventional or domestic/on-site waste treatments, while, in waste prevention scenarios, a portion of that amount represent the prevented waste, being managed through one or more waste prevention activities.

In order to allow the comparisons of scenarios where the total amount of waste to be managed[4] is variable, in *Approach 2* the functional unit is instead defined as:

"*the management of the waste produced over a given period in a given geographical area (or by one of its in-habitants)*", without specifying any amount.

In this case, contrary to *Approach 1*, it is not possible to define an alternative functional unit based on a numerical amount of waste to be managed, since waste prevention is not considered as a possible waste management method and, as a consequence, the amount of waste to be managed effectively differs among compared scenarios. It is therefore necessary to relate the analysed waste management system(s) to a real or hypothetical geographical context.

4.3.3 System boundaries

Figure 4.3 illustrates the main processes that, according to both *Approach 1* and *2*, need to be included in the boundaries of the waste management system in a baseline scenario and in a waste prevention one. To be comprehensive, the latter scenario was assumed to include all the types of waste prevention activities reviewed at the beginning of the research and described in Table 1.1.

As usual, both approaches include, in both baseline and waste prevention scenarios, all the operations carried out on all the waste streams effectively generated in the studied area or system under such scenarios. These operations can include collection, sorting of source-separated material, mechanical and biological pre-treatment of the residual waste, as well as the subsequent treatment through, recycling, biological and thermal treatments and/or landfilling. As an alternative, domestic or on-site treatments, such as home or community composting, can also be carried out, without the need of collecting the waste. The part of the waste management system that includes all the mentioned operations (downstream processes) is defined as *waste treatment system* (WTS, see Figure 4.3). In particular, according to *Approach 1*, in a baseline scenario the waste treatment system will coincide with the waste management system.

To evaluate the downstream consequences[5] of the considered waste prevention activities, the waste treatment system of a baseline scenario will have to include at least the treatment processes of all the waste streams (goods or packages) that will be prevented in the waste prevention scenario(s). In contrast, the waste treatment system of a waste prevention scenario will have to include at least the treatment processes of all the substitutive goods or packages that can be generated as additional waste to be managed, as a consequence of the implementation of the considered waste prevention

[4] Only through conventional or domestic/on-site treatments.

[5] Downstream consequences are the consequences on the impacts involved by the processes taking place after the waste has been generated (i.e. the processes of the waste treatment system).

activities. Examples of prevented and substitutive waste goods and packages are reported in Table C.1 of Appendix C, for the different types of waste prevention activities identified in Table 1.1.

It is worth noticing that when the implemented waste prevention activities are based on reuse of disposable goods in substitution of durable goods (type 7 activities of Table 1.1), or on reuse and lifespan extension of existing durable goods (types 8 and 9 activities), in the last resort the prevented waste is not represented by the reused goods or by the goods the lifespan of which has been extended. Conversely, the prevented waste is constituted by those equivalent new durable goods that would be used (and wasted at the end of their useful life) if reuse or lifespan extension activities were not undertaken[6] (EC-JRC, 2011b). Therefore, one should take into account that equivalent new goods may be composed of different materials compared to more outdated reused ones (e.g. furniture using more environmentally friendly adhesives and resins or electric and electronic equipment using parts made up of different metals).

According to *Approach 1*, in waste prevention scenarios, system boundaries of traditional waste management LCA are then expanded upstream to include all the upstream processes avoided thanks to the implemented waste prevention activities and all the additional upstream processes that take place as a consequence of the implementation of such activities. In fact, the management of the prevented waste (goods or packages) through one or more prevention activities (which in *Approach 1* are considered to be an actual waste management method), involves the following upstream consequences. First of all, the processes belonging to the whole upstream life cycle of the prevented waste (raw material extraction, material production, product manufacture, distribution and use) are generally avoided. Moreover, additional upstream processes take place when the implemented waste prevention activities are based on product/service substitution, reuse or lifespan extension (types 3 to 10 activities). In the case of product/service substitution (types 3 to 6 activities), such additional processes are those associated with the whole upstream life cycle of the less waste-generating substitutive goods or services that will be used, i.e. goods or packages using less material (type 3 activities), unpacked goods (type 4 activities), reusable goods or goods provided in reusable packages (type 5 activities) and digital goods (type 6 activities). When disposable goods or packages are reused (type 7 activities), a cleaning phase may be performed before each subsequent use. However, the impacts associated with cleaning and reuse(s) are likely limited and, thus,

[6] Actually, an equivalent new good is prevented in its entirety only if its average lifespan is identical to the duration of the second life of the reused good, or of the additional life of the good the lifespan of which has been extended. If this second or additional life is shorter or longer than the lifespan of the equivalent new good, only a fraction (smaller or greater than 1, respectively) of this equivalent new good is actually prevented. Such fraction is equal to the ratio between the duration of the second life of the reused good (or the duration of the additional life of the good the lifespan of which has been extended) and the average lifespan of the equivalent new good. This is not very close to the reality, where waste is defined by integer quantities, but it is a possible modelling approach also reported in the guide prepared by the Joint Research Centre of the European Commission (EC-JRC, 2011b).

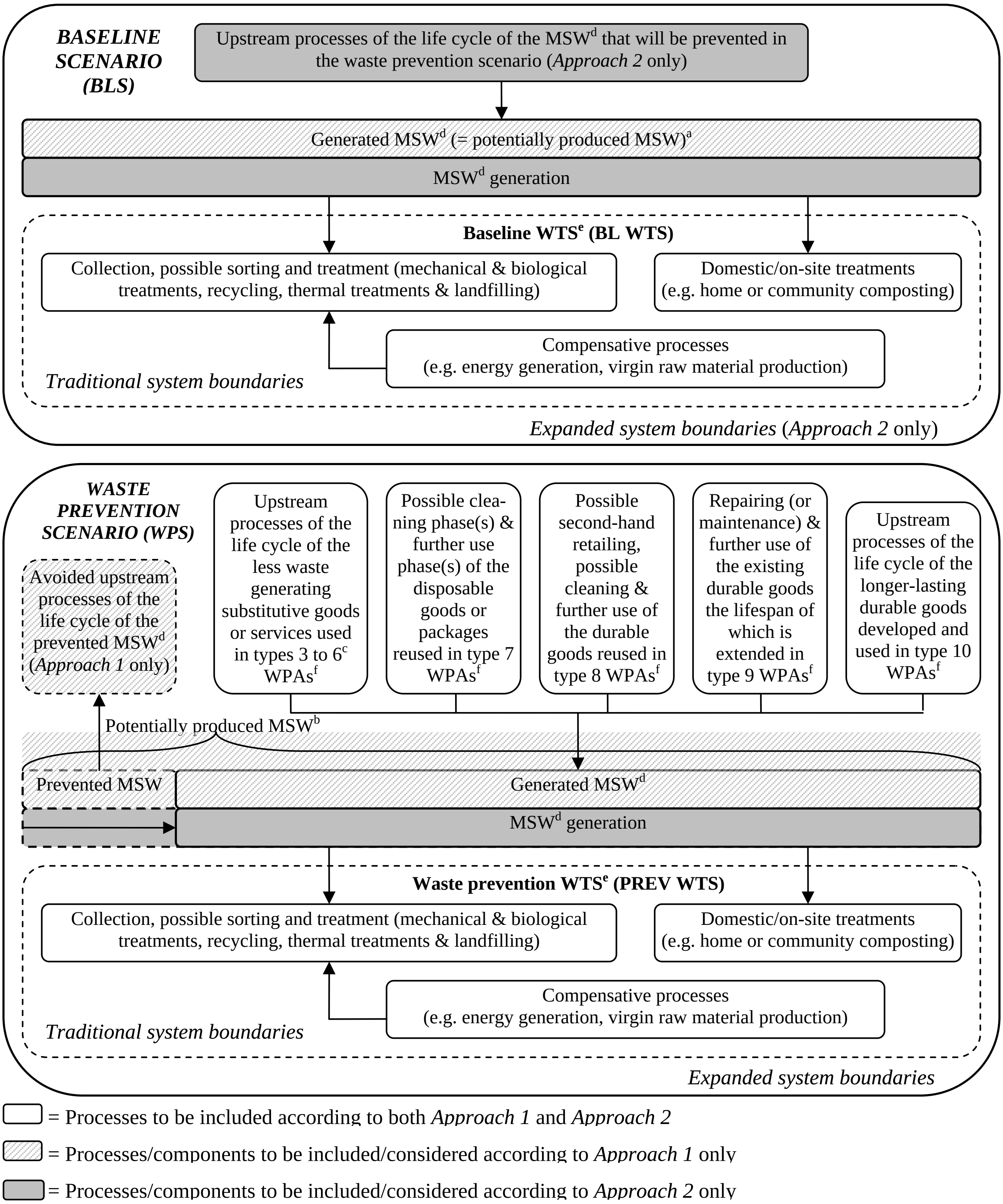
negligible. If durable goods are reused (type 8 activities), additional processes that can take place are second-hand retailing and cleaning operations. Moreover, the reuse phase shall be accounted for, if relevant impacts are involved. When the lifespan of existing durable goods is extended (type 9 activities), possible repairing or maintenance operations are carried out, along with the phase of further use. Finally, when longer-lasting durable goods are used (type 10 activities), their whole upstream life cycle is involved. All the mentioned upstream processes and any other upstream process that may be affected by the implemented waste prevention activities shall therefore be included in the system boundaries in waste prevention scenarios, unless the effects on the resulting impacts is negligible.

According to *Approach 2*, traditional system boundaries are expanded upstream in both baseline and waste prevention scenarios. Those upstream processes are included, which differ among the compared scenarios for their type or magnitude, because of the implementation of waste prevention activities. In a baseline scenario, the included processes are those belonging to the whole upstream life cycle of the waste (goods or packages) that will be prevented in waste prevention scenario(s) and that is effectively generated in the baseline scenario. In a waste prevention scenario, the included upstream processes are those that take place as a consequence of the implementation of waste prevention activities based on product/service substitution, reuse or lifespan extension (see above in this section for a more detailed description of these processes).

Therefore, according to *Approach 2* a baseline scenario will implicitly include the whole life cycle of the waste (goods or packages) that will be prevented in waste prevention scenario(s). Conversely, depending on the type of prevention activity, a waste prevention scenario will implicitly include: the whole life cycle of the less waste-generating substitutive goods or services used (types 3 to 6 activities); the part of the life cycle after the first use of reused disposable goods or packages (type 7 activities); the second life of the disposable goods or packages reused in place of durable goods (type 7 activities); the second life of reused durable goods (type 8 activities), the additional life of durable goods the lifespan of which has been extended (type 9 activities); and/or the whole life cycle of longer-lasting durable goods (type 10 activities)[7].

In principle, the whole upstream life cycle of all waste streams should be included in the system boundaries in all of the compared scenarios. However, in a comparative analysis, identical parts can be omitted. The upstream processes that are not affected by waste prevention activities in their type or magnitude can thus, once again, be excluded.

[7] This is valid also for Approach 1, even if not explicitly stated during the description of the system boundaries adopted in a waste prevention scenario according to this approach.



[a] According to *Approach 1*, in baseline scenarios the whole amount of waste potentially produced in the studied area/system is actually generated as waste and, therefore, the two amounts are identical.

[b] According to *Approach 1*, in waste prevention scenarios a portion of the amount of waste potentially produced in the studied area/system is prevented, so that the respective amount is greater than the amount of waste actually generated.

[c] Goods or packages using less material (type 3 activities), unpacked goods (type 4 activities), reusable goods or goods provided in reusable packages (type 5 activities) and digital goods (type 6 activities).

[d] MSW: Municipal solid waste; [e] WTS: Waste treatment system; [f] WPA: Waste prevention activity

Figure 4.3: boundaries of the waste management system in a waste prevention scenario and in a respective baseline scenario, according to the two presented approaches (*Approach 1* and *Approach 2*). The waste prevention scenario includes all the types of prevention activities reviewed at the beginning of the research and described in Table 1.1.

An approach similar to *Approach 2* is that used by Matsuda et al. (2012), who evaluated the consequences of reducing edible food waste (food loss) on green-house gas emissions associated with the management of household combustible waste in Kyoto, Japan. To this aim, the authors compared two scenarios, with and without the partial prevention of food losses. Both of them included, besides collection and treatment of all the waste streams generated in such scenarios, also the processes of production, distribution and cooking of the whole amount of food losses generated in the same scenarios. In a small variant, and following *Approach 2*, only the boundaries of the baseline scenario are expanded to include the processes belonging to the upstream life cycle of the food losses that will be prevented, while no upstream processes are included in the waste prevention scenario. In fact, since the upstream life cycle of not prevented food losses is identical in both waste prevention and baseline scenarios, it can be omitted without affecting their comparison. This alternative approach is especially useful when only the amount of waste which can be potentially prevented through a given activity is known, but not the overall amount of the waste fraction targeted for prevention (i.e. only an estimate of the amount and of the composition of preventable food losses is known, but not the overall amount of food losses generated in the baseline case).

In conclusion, in *Approach 1*, all the upstream processes affected by waste prevention activities are included, as avoided and additional processes, in waste prevention scenarios (as they are the processes avoided or involved by managing the prevented waste through one or more prevention activities). Conversely, in *Approach 2*, the affected upstream processes are included, as additional processes, in the scenario in which they actually occur (since they are the processes which differ among compared scenarios because of waste prevention activities). Therefore, from the practical point of view, the two approaches essentially differ in the following. According to *Approach 1* the upstream processes of the life cycle of the prevented waste are included, as avoided processes, in waste prevention scenarios, while in *Approach 2* they are included as additional processes in the baseline scenario in which they take place (i.e. where the waste that will be prevented is generated).

4.3.4 Impact calculation

The procedure to be used, according to both *Approach 1* and *Approach 2*, for the calculation of the environmental impacts of a baseline and a waste prevention scenario reflects the boundaries considered for the waste management system in such scenarios.

According to *Approach 1* the impacts of a baseline scenario (BLS) coincide with the impacts of the respective baseline waste treatment system (BL_WTS; Equation 1). This is the unique component of the baseline waste management system when *Approach 1* is adopted, and includes all the operations applied to the many waste streams generated in the assessed baseline scenario. The impacts of the baseline waste treatment system can be calculated by means of a traditional waste management LCA.

$$\mathrm{BLS} = \mathrm{BL\_WTS} \tag{1}$$

According to *Approach 2*, the impacts of a baseline scenario of a system in which *n* waste prevention activities will be implemented, are instead calculated as follows (Equation 2):

$$\mathrm{BLS} = \mathrm{BL\_WTS} + \sum_{i=1}^{n} \mathrm{UpP\_PW}_i \tag{2}$$

where the terms BLS and BL_WTS have the same meaning as in Equation 1, while $\mathrm{UpP\_PW}_i$ represents the impacts of the upstream processes (raw material extraction, material production, product manufacture, distribution and use) of the life cycle of the waste that will be prevented in waste prevention scenario(s) thanks to the implementation of the i-th waste prevention activity.
According to *Approach 1*, the impacts of a waste prevention scenario are calculated with the following equation (Equation 3):

$$\mathrm{WPS} = \mathrm{PREV\_WTS} - \sum_{i=1}^{n} \mathrm{UpP\_PW}_i + \sum_{j=1}^{m} \mathrm{Add\_UpP}_j \tag{3}$$

where:

- WPS = particular impact of a waste prevention scenario including *n* waste prevention activities, of which *m* are based on product or service substitution, reuse or lifespan extension (types 3 to 10 activities);
- PREV_WTS = particular impact of the waste prevention waste treatment system, i.e. the part of the waste management system that includes all the operations applied to the many waste streams generated in the waste prevention scenario under analysis. Such an impact can be calculated with a traditional waste management LCA of the waste prevention waste treatment system itself;
- $\mathrm{UpP\_PW}_i$ = particular impact of the (avoided) upstream processes (raw material extraction, material production, product manufacture, distribution and use) of the life cycle of the waste prevented thanks to the implementation of the i-th waste prevention activity (as in Equation 2);
- $\mathrm{Add\_UpP}_j$ = particular impact of the additional upstream processes that take place as a consequence of the implementation of the j-th waste prevention activity of types 3 to 10.

According to *Approach 2*, the impacts of a waste prevention scenario that includes *n* waste prevention activities, of which *m* of types 3 to 10, are instead calculated as follows (Equation 4):

$$\mathrm{WPS} = \mathrm{PREV\_WTS} + \sum_{j=1}^{m} \mathrm{Add\_UpP}_j \qquad (4)$$

where all the terms have the same meaning as in Equation 3 (*Approach 1*) but, compared to this, excludes the avoided impacts of the upstream processes of the life cycle of the waste prevented thanks to all the *n* implemented waste prevention activities ($\sum_{i=1}^{n} \mathrm{UpP\_PW}_i$). In fact, according to the system boundaries considered by *Approach 2*, the impacts of such upstream processes have already been included, with a positive sign, in Equation 2, which is used to calculate the impacts of a baseline scenario when such an approach is used.

If out of the *m* waste prevention activities of types 3 to 10 included in a waste prevention scenario, *a* is of types 3 to 6, *b* of type 7, *c* of type 8, *d* of type 9 and *e* of type 10 (with *a*+*b*+*c*+*d*+*e*=*m*), the last term of Equations 3 and 4 can be better expressed as follows (Equation 5):

$$\sum_{j=1}^{m} \mathrm{Add\_UpP}_j = \sum_{k=1}^{a} \mathrm{Add\_UpP}_k + \sum_{l=1}^{b} \mathrm{Add\_UpP}_l + \sum_{p=1}^{c} \mathrm{Add\_UpP}_p + \sum_{q=1}^{d} \mathrm{Add\_UpP}_q + \sum_{r=1}^{e} \mathrm{Add\_UpP}_r \qquad (5)$$

where:

$\mathrm{Add\_UpP}_k$ = impact of the upstream processes of the life cycle of the less waste-generating substitutive goods or services used in the k-th waste prevention activity of types 3 to 6 (goods or packages using less material, unpacked goods, reusable goods or goods provided in reusable packages, digital goods);

$\mathrm{Add\_UpP}_l$ = impact of the possible cleaning phase(s) and of the subsequent use phase(s) of the disposable goods or packages reused in the l-th waste prevention activity of type 7;

$\mathrm{Add\_UpP}_p$ = impact of the possible second-hand retailing, possible cleaning and of the further use phase of the durable goods reused in the p-th waste prevention activity of type 8 (generally, the impact of the use phase is relevant only if the reused goods use consumables);

$\mathrm{Add\_UpP}_q$ = impact of the possible repairing or maintenance phases and of the further use phase of the existing durable goods the lifespan of which is extended in the q-th waste prevention activity of type 9 (generally, the impact of the use phase is relevant only if the goods the lifespan of which is extended use consumables);

$\mathrm{Add\_UpP}_r$ = impact of the upstream processes of the whole life cycle of the longer-lasting durable goods developed and used in the r-th waste prevention activity of type 10.

To calculate the net impacts associated with the implementation of the considered waste prevention activities, and with any possible change in the management of the remaining MSW (e.g. a variation of separated collection efficiencies), the impacts of the baseline scenario (Equation 1 or 2, depending on the used approach) shall be subtracted from those of the waste prevention scenario, calculated with Equation 3 or 4. The results of this operation is independent on the approach used, as the equations adopted by the two approaches include the same terms overall and those included in different equations have an opposite sign. In particular, while in *Approach 1* the impact of the upstream processes of the life cycle of the prevented waste is included with a negative sign in the equation used to calculate the impact of a waste prevention scenario (Equation 3), in *Approach 2* it is included with a positive sign in the equation used to calculate the impacts of a baseline scenario (Equation 2). Therefore, both the approaches are equivalent in terms of obtained solution, even if their application is more suitable in different situations, as discussed more extensively in Section 4.4.

4.3.5 Example

A tangible example of the system boundaries adopted by the two presented approaches is provided, considering that the waste prevention activity based on the substitution of one-way bottled water by public network water is undertaken by a number of citizens of a given municipality.

According to *Approach 1*, in a baseline scenario without waste prevention, the system boundaries shall include at least the end of life (collection, possible sorting, treatment and the possibly associated compensative processes) of the one-way water bottles, caps and labels that will be prevented by drinking tap water.

In a waste prevention scenario, system boundaries will instead include the end of life of any goods generated as waste as a consequence of the consumption of public network water (e.g. reusable jugs or bottles) and of any other waste stream possibly considered in the baseline scenario. Moreover, the boundaries will also be expanded to include the upstream processes that are avoided and that take place as a consequence of the performed substitution. Avoided upstream processes are those belonging to the whole upstream life cycle of the substituted one-way bottled water and include water withdrawal, packing, palletisation, transport, distribution and use[8]. Additional upstream processes to be included as well in the system boundaries in a waste prevention scenario are instead

[8] The end of life of all the goods used in these operations shall also be included, if not already included in the waste management system.

those belonging to the whole upstream life cycle of the replacing amount of public network water, i.e. its collection, purification, distribution and use.

If *Approach 2* is used rather than *Approach 1*, the processes of the whole upstream life cycle of substituted one-way bottled water will not be included as avoided processes in the waste prevention scenario, but as additional processes in the baseline scenario (which include also the end of life of the municipal waste generated by the consumption of one-way bottled water). In *Approach 2*, in fact, the system boundaries are expanded in both the baseline and the waste prevention scenario, in order to include the upstream processes that differ from one to the other because public network water is substituted for one-way bottled water.

4.4 Discussion

Two different LCA approaches (*Approach 1* and *Approach 2*) are described to evaluate the environmental and energy performance of MSW management scenarios that include waste prevention activities. They differ in the perspective from which they look at waste prevention activities (Section 4.3.1) and in the functional unit consequently adopted (Section 4.3.2). However, on the basis of different premises and in different scenarios, both of them foresee the expansion of traditional system boundaries of waste management LCA to include the upstream processes affected by the implemented waste prevention activities (Section 4.3.3) and the associated impacts (Section 4.3.4). In particular, while in *Approach 1* the upstream processes of the life cycle of the prevented waste and associated impacts are included as avoided processes/impacts in waste prevention scenarios, in *Approach 2* they are included as additional processes/impacts in the baseline scenario where they actually occur.

For this reason, both the approaches provide the same result in terms of difference between the impacts of a waste prevention scenario and of a baseline one, which represent the net impacts associated with the implementation of the considered waste prevention activities and with any possible change in the management of not prevented MSW. They lead therefore to the same overall conclusion when the performance of a waste prevention and a baseline scenario are compared. Nevertheless, because of the partially different upstream system boundaries, the LCA results of single scenarios calculated with *Approach 1* are different from those calculated with *Approach 2*. The interpretation of these results thus needs to be carried out differently, in particular for baseline scenarios. The evaluation of the upstream consequences of implementing the considered waste prevention activities requires a different procedure, as well. Finally, the use of the two approaches will be more suitable in different situations and in studies with different purposes. These three issues will be briefly discussed in the following, starting from the assessment of the consequences of waste prevention activities on downstream and upstream impacts.

Both approaches include in the system boundaries the same downstream processes in both baseline and waste prevention scenarios (i.e. the management operations of the many MSW streams generated in such scenarios). Therefore, with both the approaches, in order to evaluate the net downstream consequences of the considered waste prevention activities, it is enough to compare the downstream impacts of the waste prevention scenario with those of the baseline scenario. A comparison among the overall downstream impacts is appropriate only if no changes in the management of the not prevented MSW take place in the waste prevention scenario, compared to the baseline. Conversely, in order to distinguish the downstream consequences of waste prevention activities from those of any changes in the management of the not prevented MSW, the impacts from the management of the prevented waste (in baseline scenarios) and of the possible replacing waste (in waste prevention scenarios) should be isolated from the impacts associated with the management of the other waste streams.

As early discussed, in *Approach 1* all the upstream processes affected by waste prevention activities and the associated impacts are included in waste prevention scenarios as avoided and additional processes/impacts. By using this approach, it is thus possible to identify directly in the LCA results of a waste prevention scenario the upstream impacts (loads) and the avoided upstream impacts (benefits) of the considered waste prevention activities, and subsequently evaluate their net consequences on upstream impacts. According to *Approach 2*, the upstream processes affected by waste prevention activities and their impacts are instead included as additional processes/impacts in the scenario in which they actually occur. In this case, the net upstream consequences of the considered waste prevention activities are evaluated by comparing the upstream impacts of baseline scenarios with those of waste prevention scenarios.

Another issue that is worth discussing is the interpretation of the LCA results of baseline scenarios. When *Approach 1* is adopted, the impacts of a baseline scenario represent, as usual in waste management LCA, the impacts of managing the different waste streams according to a given treatment scheme. No upstream processes are indeed included in the system boundaries in baseline scenarios. The interpretation can thus be carried out according to usual practices, such as the comparison between the possible benefits and the adverse impacts of the waste management scheme adopted.

If *Approach 2* is used, the impacts of baseline scenarios are instead deviated by the positive contribution provided by the upstream processes of the life cycle of the prevented waste. Apparently, these processes have no direct connection with the management system, as it excludes waste prevention activities, which affects upstream supply chains. Therefore, the possible interpretation of the LCA results of a baseline scenario in isolation from those of the associated waste prevention scenario(s) requires more caution, especially when the intended audience is waste managers and policy makers. To facilitate this operation, the impacts of upstream processes may temporarily be excluded from the results.

Focusing on the applicability of the two approaches, in general, *Approach 1* is preferable if: (a) there is an interest in accounting for the upstream consequences/impacts of waste prevention activities only in waste prevention scenarios; (b) a method based on the concept of avoided and additional impacts is preferred; and (c) the LCA results of baseline scenario(s) need to be also interpreted singularly, other than in comparison with those of the respective waste prevention scenario(s). More specifically, *Approach 1* is preferable when more (baseline) scenarios without waste prevention activities, distinguished only by variations in the management method of the same waste streams, need to be contemporarily compared with one or more waste prevention scenarios. In this situation (in contrast to *Approach 2*), no upstream processes that would remain unchanged from one baseline scenario to the other would be included in the system boundaries in such scenarios, thus facilitating the interpretation of the LCA results of baseline scenarios and their possible comparison. An example is a study to evaluate which of the two following measures would be more effective to reduce the impacts from the landfilling of specific waste fractions: the partial substitution by high-efficiency incineration or the partial prevention through specific activities? In this case, a baseline scenario mainly relying on landfill would be compared with both an alternative baseline scenario where landfilling is partially replaced by incineration, and a waste prevention scenario where a partial prevention of the landfilled waste is introduced.

Finally, the use of *Approach 1* is also preferable when the impacts of individually implementing different waste prevention activities in a given waste management system need to be mutually compared. With *Approach 1*, only one baseline scenario would have to be modelled and then compared individually with each of the waste prevention scenarios where the different waste prevention activities to be assessed are singularly implemented. If *Approach 2* was used, a baseline scenario would have also to be modelled for each considered waste prevention activity. This scenario would include the upstream processes of the life cycle of the waste that will be prevented thanks to that particular activity.

Approach 2 is more suitable in the following situations: (a) when there is an interest in accounting for the impacts of the upstream processes affected by waste prevention activities in the scenario where such processes actually occur; (b) when an independent assessment is required of the impacts of the many activities that can be implemented to prevent a given type of waste; and (c) when the practitioner intends to compare the consequences of entirely preventing different types of waste (e.g. bottles made of different materials) through a given activity. For instance, with regard to this latter situation, one may be interested in evaluating the benefits of substituting tap water for that packaged in bottles of different materials, such as virgin polyethylene terephthalate (PET), partly recycled PET and polylactic acid (PLA). With *Approach 2*, three baseline scenarios would be modelled first (one for each type of material). They would include, among the other downstream processes, the end of life of water bottles that will be prevented thanks to the use of tap water, and the associated upstream processes (from manufacture to use by citizens). These baseline scenarios would then be singularly compared with a unique waste prevention scenario, where post-consumer water bottles are completely removed and the end of life of the goods possibly generated as additional waste to be managed is included, along with all the upstream processes in the life cycle of replacing tap water (from purification to use by citizens). If *Approach 1* was instead used, three waste prevention scenarios would have to be modelled. Each scenario would include, besides the already mentioned downstream and upstream processes, the avoided upstream processes in the life cycle of water packaged in bottles made out of one particular material. Each waste prevention scenario would then be singularly compared with the respective baseline scenario. However, if only a fraction of post-consumer water bottles is prevented, even *Approach 2* requires the modelling of three different waste prevention scenarios. In fact, each time, not prevented water bottles would be of a different material and their end of life would likely be different.

Sometimes, different activities can be implemented to target a particular waste stream (point b above). For instance, one-way water bottles can be prevented by using either tap water or refillable bottled water. Similarly, paper hand towels can be replaced by either electric hand-dryers or cloth roll towels. The independent evaluation of the effects of the different viable activities by means of *Approach 1* would require including the same avoided upstream processes in all of the waste prevention scenarios to be modelled (one for each considered activity), as the type of prevented waste is the same. With *Approach 2* it is instead possible to include such upstream processes only once, as additional processes, in a baseline scenario, thus simplifying the modelling of the different waste prevention scenarios.

Evaluating the performance of a system including waste prevention activities with the presented approaches can be more complex than performing a traditional waste management LCA. First of all, additional upstream processes need to be modelled beyond traditional waste management operations. For many of these processes (e.g. packing operations, product assembly etc.) an inventory is not available in the most widespread databases and, in some cases, it can be difficult to gather reliable life cycle data for its development. This requires dealing with further subjects beyond the operators of the waste management sector, thus making the data collection stage more time consuming than in traditional waste management LCA. Moreover, some upstream processes take place in foreign countries (e.g. product manufacture or assembly in the Far East) or in more countries, so that their proper modelling can be a complex task. Conversely, traditional waste management systems are generally “local” systems, since most waste treatment operations often take place within a district or at a regional level. It is therefore less challenging to carry out the data collection stage, which usually allows one to acquire primary data, actually representative of the system to be modelled. Another drawback of including upstream processes in the boundaries of the analysed systems is the potential introduction of greater uncertainty in the LCA results, as more parameters and assumptions are needed.

A second additional operation that can complicate the application of the presented approaches is the evaluation of the waste prevention potential of the assessed activities (i.e. the overall quantity of waste that can be potentially prevented thanks to such activities). Estimates of this parameter (generally expressed in kg of waste prevented per inhabitant per year or as percentage reduction of the targeted waste fraction) are available for different waste prevention activities. Such estimates are based on the results from the monitoring of real experiences (Salhofer et al., 2008; ACR+, 2010) or are derived from elaborations on market and waste statistics related to particular regions (Salhofer et al., 2008). However, the applicability of these estimates to the studied area should be checked on the basis of the pertaining regional statistics and the expected participation rates of the actors (citizens, organisations, retailers, producers etc.) who are requested to undertake the considered prevention activities (e.g. by changing their consumption behaviour). A survey may be carried out in the studied area to estimate such participation rates. In particular, a survey would be of help when more activities can be implemented to prevent a given type of waste, i.e., generally, when different less waste-generating goods can be used in substitution of the one generating the prevented waste. In this case, a survey would allow to estimate the percentage of actors that will prefer to undertake one activity rather than the other. Moreover, it would allow to calculate the quantity of the targeted good which is substituted by a less waste-generating good rather than the other and, therefore, the quantity of the associated upstream and downstream processes to be included in the system boundaries.

A last critical aspect associated with the application of the presented approaches is just that, in some cases, it is not immediate to identify the quantity in which the processes of the life cycle of a less waste-generating good are to be included in the system boundaries in a waste prevention scenario. This frequently happens when substitutive goods are digital goods. For instance, if a super-market chain completely replaces printed advertising brochures with digital ones, it is not straightforward to determine the time spent overall by consumers at reading on-line brochures. This parameter is however needed to establish the quantity in which the upstream process associated with the activity ‘use of a personal computer’ have to be included in a waste prevention scenario.

4.5 Concluding remarks

The environmental and energy performance of MSW management scenarios that include waste prevention activities can be evaluated in a life cycle perspective with the two approaches presented and discussed in this section. The choice of a proper functional unit and the expansion of system boundaries allow waste management LCA to contemporarily account for all the options of the waste hierarchy.

The two presented approaches can be used for many specific purposes, such as:

(a) quantifying the overall environmental and energy consequences of implementing one or more waste prevention activities in a system where waste is managed according to a given treatment scheme (*Approach 1* or *2*);

(b) comparing the consequences of individually implementing different waste prevention activities, or of preventing different types of waste, in a given waste management system (*Approach 1*);

(c) assessing the consequences of implementing the same waste prevention activity(ies) (or of preventing the same type(s) of waste) in different waste management systems (*Approach 1*, if also the baseline scenarios of the different waste management systems are to be compared, otherwise, even *Approach 2* is suitable);

(d) comparing the effects of introducing waste prevention in an existing waste management system, with those of other viable measures to improve the performance of that particular system (e.g. increasing material and energy recovery, reducing distances etc.; *Approach 1*);

(e) evaluating the consequences of implementing one or more waste prevention activities, on the optimal management option for a constant amount and composition of waste (*Approach 1*);

(f) evaluating the consequences of preventing the same type of waste through different activities (*Approach 2*); and

(g) evaluating the consequences of preventing different types of waste (e.g. bottles of different materials) through a given prevention activity (*Approach 2*, if a complete removal of the targeted waste stream is considered, otherwise, even *Approach 1* is suitable).

5 Life cycle assessment of municipal waste prevention and management at the regional level: the case of Lombardia

In this section, the two waste prevention activities assessed in Sections 2 and 3 are implemented in a real waste management system to evaluate, by means of life cycle assessment (LCA), the potential effects on its overall environmental and energy performance.

The LCAs of Sections 2and 3 were carried out at the scale of the different products potentially involved in the prevention activity, by evaluating the relative impact variations achievable for a given level of consumption of such products. However, they provided no information on the potential of the substitutions to affect the overall environmental and energy performance of waste management in a given geographical region. Here, this potential is evaluated, by taking into account the actual levels of consumption of the products subject to substitution at the regional level.

The region selected for the assessment is Lombardia, Italy, where municipal waste is managed according to a quite advanced scheme, comprising high levels of material and energy recovery. This system has already been characterised in a recent LCA study (Grosso et al., 2012; Rigamonti et al., 2013), aimed at supporting the update of the regional waste management programme. In this study, a 2009 reference scenario is compared with different 2020 perspective scenarios, to identify viable options for the improvement of the overall environmental performance of the system.

According to the national legislation, the new regional waste management programme (officially adopted in 2014; Regione Lombardia, 2014) includes a waste prevention programme setting reduction targets for 2020. To facilitate their achievement, a package of specific waste prevention actions is proposed, which also comprises those examined in this thesis (Sections 3 and 4). In this framework, the waste management system of Lombardia is an ideal case study for an evaluation of the potential life cycle effects of such prevention activities.

The assessment summarised in this section is carried out by following the methodological approach discussed in Section 4, with the support of the SimaPro software (version 7.3.3). With this tool, a parametric model of the municipal waste management system of Lombardia was created and the respective potential impacts calculated.

5.1 Analysed waste management scenarios

Five scenarios for municipal waste management in Lombardia were analysed: a baseline scenario and four waste prevention scenarios implementing specific prevention activities (Table 5.1). The baseline scenario is a 2020 perspective scenario and is used as a reference. It was defined based on forecasted increases in population and per-capita waste generation compared to 2009 (Grosso et al., 2012; Rigamonti et al, 2013). The quantity of waste estimated for 2020 amounts to 4,838,297 tonnes, 66% of which are source separated for material recovery (3.194.431 t), while the remaining 34% (1.643.866 t) are collected as residual waste. Source separated packaging materials are sent to recycling, after being sorted. The organic fraction is entirely routed to anaerobic digestion, while green waste is sent to composting. Finally, most of the residual waste (73.7%) is directly routed to energy recovery in dedicated incineration plants, the rest being subject to mechanical-biological treatments, producing refuse derived fuel (RDF), bio-dried material or, however, an improved material for incineration. The RDF is partly incinerated in waste-to-energy plants and partly used to displace pet coke in cement kilns, while the bio-dried material is totally incinerated.

The baseline scenario is compared with a first waste prevention scenario (WPS1), where bottled water consumed domestically is entirely substituted by public network water withdrawn from the tap. Two waste prevention scenarios implementing the same activity are then compared with the baseline (WPS2a and WPS2b). In these scenarios, different types of liquid detergents packaged in single-use plastic containers are completely substituted by those distributed loose through self-dispensing systems and refillable containers. The substitution was applied to liquid laundry detergents (automatic and hand wash), fabric softeners and hand-dishwashing detergents sold through all traditional retail channels. However, in WPS2a the replacing detergent is assumed to have the same average washing performance (i.e. number of washings per litre) as substituted ones. In WPS2b, the replacing detergent is assigned a specific washing performance, according to the experiences implemented so far in Lombardia. As this performance is worse than the average one of substituted detergents (most of which is more concentrated), a greater volume of detergent is needed to perform the same average number of washings as the baseline.

Finally, a third waste prevention scenario (WPS3) is compared, where both waste prevention activities are implemented, again considering a complete substitution of the traditional products. In this case, the replacing detergent has the same average washing performance as substituted traditional ones.

Each waste prevention scenario is individually compared with the baseline scenario, to evaluate the effects of the waste prevention activity(ies) on the overall environmental and energy performance of the waste management system as a whole.

Table 5.1: municipal waste management scenarios compared in this study and respective quantities of waste generated and prevented.

Scenario	2020 baseline scenario	Waste prevention scenario 1	Waste prevention scenario 2a	Waste prevention scenario 2b	Waste prevention scenario 3
Waste prevention activity implemented	None	Substitution of bottled water consumed domestically by public network water from the tap	Substitution of single-use packaged liquid detergents by those distributed loose through self-dispensing systems and refillable containers[a]		Both product substitutions of waste prevention scenarios 1 and 2a
Washing performance of loose detergents[b]	-	-	Same as substituted detergents	Worse than substituted detergents[c]	Same as substituted detergents
Total waste [t] of which:	4,838,297	4,813,172	4,831,370	4,832,281	4,806,245
Source separated waste (mono-material collection)	2,883,429	2,861,283	2,877,402	2,878,195	2,855,256
Source separated waste (multi-material collection)	311,002	308,023	310,101	310,220	307,123
Residual waste	1,643,866				
Prevented waste [t]	-	25,125 (0.52%)[d]	6,927 (0.14%)[d]	6,016 (0.12%)[d]	32,052 (0.66%)[d]

(a) The types of detergent subject to substitution are laundry detergents (automatic and hand wash), fabric softeners and hand dishwashing detergents.
(b) i.e. average number of washings per litre of detergent.
(c) For laundry detergents and fabric softeners, a number of 10 washings per litre was assumed, while 51 washings per litre were assumed for hand dishwashing detergents (corresponding to a dosage of 20 g of detergent per 5 litres of water). The assumptions are based on the washing performance of the detergents used in the real experiences of distribution through self-dispensing systems recently implemented in Lombardia.
(d) Of the total waste.

5.2 Functional units

The functional unit is defined as "*the integrated management of the waste potentially generated in Lombardia in 2020*". Depending on the scenario, the potential waste includes the waste actually collected and managed through conventional treatment operations, as well as the waste possibly prevented thanks to the waste prevention activity(ies) implemented in that scenario, by which it is managed.

Following the suggestion by Cleary (2010), one or more secondary functional units were also defined for each waste prevention scenario, depending on the type and number of waste prevention activities implemented. Secondary functional units are used to ensure that the amount of product service provided to the citizens of Lombardia by the product systems affected by the considered waste prevention activity(ies) is equivalent in both baseline and waste prevention scenarios. For the bottled water substitution, the secondary functional unit depicts the delivery to the citizens of the volume of drinking water subject to the substitution (1,188 million litres). For the liquid detergent substitution, the secondary functional unit quantifies the overall number of washings performed by the citizens with each type of detergent involved in the substitution. This number was estimated with reference to the average washing performance of the detergent packaged in each type and size of single-use containers subject to substitution, and is reported in Table 5.2 for each type of substituted detergent. Table 5.2 also shows the overall volume of detergent needed to carry out these washings under each waste prevention scenario.

Table 5.2: overall number of washings performed yearly with the different types of liquid detergents subject to product substitution and corresponding volume of detergent needed under each compared waste management scenario.

Type of detergent	Number of washings	Volume of detergent (litres)		
		Baseline scenario	Waste prevention scenarios 2a and 3	Waste prevention scenario 2b
Automatic laundry detergents	904,172,560	59,868,546	59,868,546	90,417,256
Hand wash laundry detergents	14,366,458	833,157	833,157	1,436,646
Fabric softeners	1,152,848,153	41,197,621	41,197,621	115,284,815
Hand dishwashing detergents	4,481,623,610	44,332,239	44,332,239	87,874,973

5.3 System Boundaries

Figure 5.1 provides a simplified representation of the boundaries of the waste management system in the baseline and waste prevention scenarios. As usual, in every scenario the system includes all the operations applied to the different waste streams effectively generated. In particular, the system accounts for all the operations comprised from the moment the waste is collected to that in which it becomes an emission to air and water, an inert material in a landfill, a secondary raw material or an energy flow. Moreover, according to the commonly applied ‘avoided burden method’ (Finnveden et al., 2009) the system is expanded to include avoided primary production processes of the materials and energy recovered from waste.

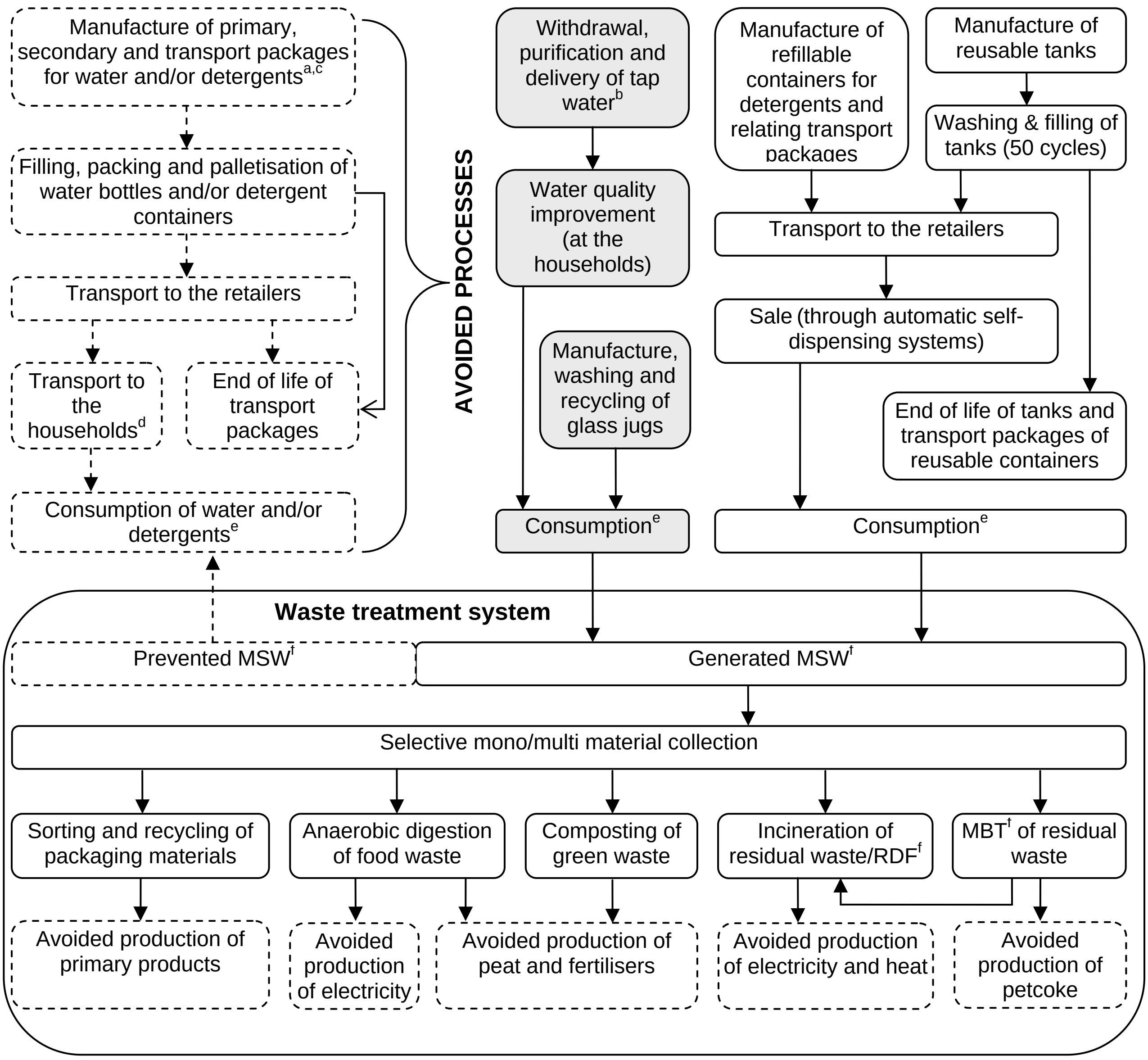
In waste prevention scenarios, the system boundaries are further expanded to include the processes upstream waste collection (avoided and additional) affected by the implemented waste prevention activity(ies). The avoided processes are those belonging to the whole upstream life cycle of the substituted goods. The additional processes are those belonging to the whole upstream life cycle of the substitutive less waste-generating goods. Thus, in WPS1 (bottled water substitution), the system comprises the avoided withdrawal, packing, palletisation, and transport to the retailers and to the point of use of the substituted volume of bottled water. Moreover, it includes the additional processes of withdrawal, purification and delivery to users of an equivalent volume of public network water, as well as water quality improvement at the domestic level.

Similarly, in WPS2a and WPS2b (substitution of single-use packaged liquid detergents), upstream system boundaries are expanded to include the avoided packing, palletisation and transport to the retailers of the substituted volume of single-use packaged detergent, as well as the additional processes of packing in reusable tanks and transport to the retailers of an equivalent amount of “loose” detergent. Packing and palletisation of refillable containers and their transport to the retailers are also included as additional upstream processes, as well as the operation of the self-dispensing systems (refilling and withdrawal) and the life cycle of its main components. Finally, when a lower washing performance is considered for the replacing loose detergent (due to a lower concentration level), the production of the additional volume of demineralised water used for dilution is included, as well as its transport to the retailers and subsequent withdrawal from self-dispensing systems by means of refillable containers.

In WPS 3, the avoided and additional upstream processes included in both WPS1 and WPS2a are taken into account.

5.4 Impact categories and characterisation models

The impact categories selected for the assessment are the same as the LCA studies summarised in Sections 2 and 3, for coherence. Similarly, the same category indicators and characterisation models were selected. The complete list of the selected impact categories, and respective category indicators and characterisation models is available in Table 2.3.



(a) Avoided upstream processes related to the life cycle of bottled water are included in waste prevention scenarios 1 and 3, while those related to the life cycle of single-use packaged detergents are included in waste prevention scenarios 2a, 2b and 3.

(b) Additional upstream processes related to the life cycle of tap water (grey boxes) are included in waste prevention scenarios 1 and 3, while those related to the life cycle of loose detergents (white boxes) are included in waste prevention scenarios 2a, 2b and 3. Waste prevention scenario 2b also includes the additional production of demineralised water used to further dilute the detergents (compared to those substituted), the transport of this water to the retailers and its withdrawal from self-dispensing systems by means of refillable containers.

(c) Primary packages include bottles/containers, caps and labels (these latter only for water bottles). Secondary packages are used only for bottled water (heat-shrink film of the bundles containing bottles). Transport packages include, cardboard boxes (for detergents only), pallets, stretch film, and cardboard interlayers (for water only).

(d) Avoided transport to the households is included only for bottled water, as for detergents it is identical to that taking place after the substitution.

(e) Consumption involves no impacts (note: the use of detergents for washing is excluded, as it is identical for both the substituted and the alternative ones)

(f) MBT: mechanical-biological treatment; MSW: municipal solid waste; RDF: refuse derived fuel

Figure 5.1: boundaries of the waste management system in the compared scenarios (upstream components are included only in waste prevention scenarios).

5.5 Estimate of the waste prevention potential

For each selected waste prevention activity, the quantities of waste removed from and added to the waste management system were estimated. The balance between avoided and additional waste (i.e. the waste prevention potential of the activities), was also calculated (Table 5.3).

Available statistics on municipal waste generation (e.g. the annual reports by ARPA[1] or national packaging consortia) were inadequate to estimate the quantity of waste avoided and the corresponding amount of product (water or detergent) subject to the substitution. This is because such statistics provide a too little detailed picture of collected waste fractions (e.g. plastic, glass etc.). It was thus necessary to adopt a reverse procedure, which on the basis of the amount of product undergoing substitution, estimates the amounts of avoided and additional waste. For this purpose, 2013 data on the Italian market of substituted products were acquired, from market databases or market research institutes. For bottled water, data on volume sales by type and size of bottle were available. For single-use packaged detergents, only the total volume retailed in Italy was instead available. An empirical subdivision of the total sales by type and size of container was thus performed. This was based on an average packaging composition, estimated by observing the frequencies with which each type and size of packaging was available in some retail stores of Lombardia.

The consumption of bottled water and single-use packaged liquid detergents by type and size of packaging was then estimated for Lombardia, based on national sales and on the ratio between the regional population expected for 2020 (10,557,381 inhabitants) and the national population in 2013. Water packaged in one-way polyethylene terephthalate (PET) bottles with a size lower than 1000 ml was excluded from the substitution, as it is mainly connected to outdoor consumption. All types of glass bottled water were also excluded, as they are manly characterised by particular properties (e.g. thermal waters), which would hardly be replaced with tap water. However, glass bottled water covers only 4% of the overall consumption. Finally, 5000 ml one-way PET bottled water and that packaged in 500 or 1000 ml bricks was excluded, as it represents an insignificant proportion of the total consumption (0.04% and 0.14%, respectively). For liquid detergents, the whole consumption was instead assumed to be suitable for the replacement with loose detergents, as there are no specific restrictions.

The total amount of waste avoided was thus calculated (Table 5.3), based on the previously estimated consumptions of the substituted products by type and size of packaging and on the average masses of the packages that would have been generated as waste as a consequence of such

[1] ARPA is the acronym for the Regional Environmental Protection Agency (Agenzia Regionale per la Protoezione dell'Abiente).

consumptions. For the bottled water substitution, the avoided waste includes bottles, caps and labels, as well as the heat-shrink film of the bundles containing bottles. For these items, the average masses reported in Table A.1 of Appendix A were considered, which in turn refer to the estimates reported in Federambiente (2010). In addition, experimental estimates were produced for 1 litre one-way PET bottles. For the liquid detergent substitution, the avoided waste is represented by single-use containers and respective caps. Their average specific masses were estimated experimentally, according to the procedure described in Appendix B (Section B.2.1.1), which also reports the considered values. The whole procedure used for the calculation of the quantities of waste avoided is summarised in Tables D.1 and D.3-D.6 of Appendix D, depending on the product subject to substitution (bottled water and the different types of liquid detergents).

Table 5.3: types and quantities of waste added to and removed from the waste management system by the examined waste prevention activities based on product substitution.

Waste prevention activity	Type of waste	Quantity	
		[tonnes]	[% of total waste for 2020]
Substitution of bottled water by tap water	Avoided waste	30,769	0.64
	Bottles (PET)	*25,581*	*0.53*
	Caps (HDPE)	*1,583*	*0.03*
	Labels (paper)	*226*	*0.01*
	Labels (plastic)[a]	*499*	*0.01*
	Heat-shrink film (LDPE)	*2,880*	*0.06*
	Additional waste	5,644	0.12
	Glass jugs	*5,644*	*0.12*
	Prevention potential	25,125	0.52
Substitution of single-use packaged liquid detergents by loose detergents	Avoided waste	7,786	0.16
	Single-use containers (HDPE)	*4,388*	*0.09*
	Single-use containers (PET)	*2,454*	*0.05*
	Caps (PP)	*944*	*0.02*
	Additional waste	859 (1,770)[b]	0.018 (0.037)
	Refillable containers (HDPE)	*772 (1,589)*	*0.016 (0.033)*
	Reusable caps (PP)	*87 (181)*	*0.002 (0.004)*
	Prevention potential	6,927 (6,016)	0.14 (0.12)

Acronyms: HDPE = high-density polyethylene; LDPE = linear low-density polyethylene; PET = polyethylene terephthalate; PP = polypropylene.
(a) In this study, plastic labels were assumed to be made out of polypropylene.
(b) Values in parenthesis refer to the case in which the washing performances of the replacing loose detergents are worse than the average ones of substituted detergents.

The amount of waste added to the systems was ultimately calculated (Table 5.3), based on the amount of substituted product and on the average masses of those goods that are generated as waste after the substitution. Additional waste generated by the bottled water substitution includes 1 litre glass jugs used to withdraw network water from the tap. By assuming an average mass of 475 g and the use for 100 times, they contribute 5.644 tonnes of additional waste to the system. For the liquid detergent substitution, the additional waste instead comprises refillable plastic containers and the respective caps, which were assigned the sizes and the experimental masses reported in Table B.10 of Appendix B (scenario 2). According to the recommendations drawn from the LCA summarised in Section 3, refillable containers were assumed to be used for 10 times overall by the citizens undertaking the product substitution. The calculation procedure of the quantities of additional waste is illustrated, fro each product subject to substitution, in Tables D.2 and D.7 of Appendix D.

For both the considered substitution, the waste added to the system is lower than that removed (avoided), so that a net reduction in the overall amount of waste to be collected is observed (Table 5.3). Specifically, the bottled water substitution allows for the avoidance of 25.125 tonnes of waste (2.4 kg per capita), corresponding to about 0.52% of the total waste. A net prevention of 6.927 tonnes (0.14 % of total waste) is instead allowed by the liquid detergent substitution, in the case of an equivalent washing performance. If a worse washing performance is considered, the net reduction is lower, being equal to 6,016 tonnes (0.12% of total waste).

5.6 Waste flows

The flows of waste characterising the compared scenarios are illustrated in Figure 5.2, which also quantifies the flows that are unaffected by waste prevention activities. Table 5.4 instead quantifies the flows that are subject to change from one scenario to another, because of prevention activities. For the 2020 baseline scenario, the waste flows identified in Grosso et al. (2012) were taken into account. These flows are estimated based on 2009 flows, by assuming an 8% increase in the regional population and a 5% increase in the per-capita waste generation, while keeping the composition of the gross waste constant. In turn, 2009 waste flows are defined based on data reported in the annual report on waste by the Regional Environmental Protection Agency (ARPA Lombardia, 2009) and in the database by the Regional Waste Observatory (ORSO: Osservatorio Rifiuti Sovraregionale), which both include figures on regional waste generation and management. In addition, an extensive survey of the treatment plants receiving most of the different waste fractions was carried out in the mentioned study, to quantify missing flows (e.g. the quantities of residues from the different applied treatments).

In waste prevention scenarios, waste flows were calculated based on the estimated quantities of waste removed and added to the system due to the implemented waste prevention activity(ies) (Section 5.5). In this study, the avoided and additional waste flows were assumed to only affect the amount of separately collected fractions, excluding the residual waste. Indeed, generally, the products removed from or added to the waste stream (Section 5.5) are very easily recognised by the citizens as items to be source-separated once they have become waste. Therefore, prevented water PET bottles (with the respective caps and labels) and the heat-shrink film wrapping bottles around, affect the quantity of source-separated plastic and multi material fraction. The same happens for single-use (prevented) and refillable (additional) plastic containers for detergent and the respective caps. Reusable glass jugs used for water withdrawal instead affect the quantity of source-separated glass and multi-material fraction.

As these flows of avoided and additional waste are substantially uncontaminated, it is assumed that the overall amount of residues produced during sorting operation of source-separated fractions is not affected , compared to the baseline scenario. Conversely, reprocessing efficiencies of recycling operations are applied also to these waste streams, except for caps and labels. These items were assumed to be entirely removed during recycling and then rejected, thus affecting the amount of residues coming from the plastic recycling process (PET and HDPE).

Table 5.4: mass of the yearly waste flows affected by the waste prevention activities, for each analysed scenario.

Waste flow	**Mass of waste [t]**				
	2020 baseline scenario	Waste prevention scenario 1	Waste prevention scenario 2a	Waste prevention scenario 2b	Waste prevention scenario 3
Avoided MSW	-	-30,769	-7,786	-7,786	-38,555
Additional MSW	-	5,644	859	1,770	6,503
Total MSW for collection	4,838,297	4,813,172	4,831,370	4,832,281	4,806,245
Source-separated packaging materials to sorting					
Aluminium	1,111				
Paper and cardboard	694,200				
Wood	222,144				
Ferrous metals	83,304				
Plastics	188,822	162,053	182,796	183,588	156,027
Glass	460,949	465,572	460,949	460,949	465,572
Multi-material fraction	311,002	308,023	310,101	310,220	307,123
Sorted packaging materials to recycling					
Plastics (total)	171,887	141,118	164,960	165,871	134,191
PET	*92,475*	*64,586*	*89,641*	*89,641*	*61,752*
HDPE	*23,548*	*23,548*	*19,455*	*20,367*	*19,455*
Mix of polyolefins	*55,863*	*52,983*	*55,863*	*55,863*	*52,983*
Glass	505,939	511,583	505,939	505,939	511,583
Recycled materials					
Plastics (total)	124,530	103,489	119,423	120,159	98,381
PET	*69,819*	*50,505*	*67,966*	*67,966*	*48,652*
HDPE	*21,194*	*21,194*	*17,939*	*18,675*	*17,939*
Mix of polyolefins	*33,518*	*31,790*	*33,518*	*33,518*	*31,790*
Glass	505,939	511,583	505,939	505,939	511,583
Residues from recycling	152,966	143,239	151,146	151,322	141,419

Acronyms: MSW = municipal solid waste; HDPE = high-density polyethylene PET = polyethylene terephthalate;.

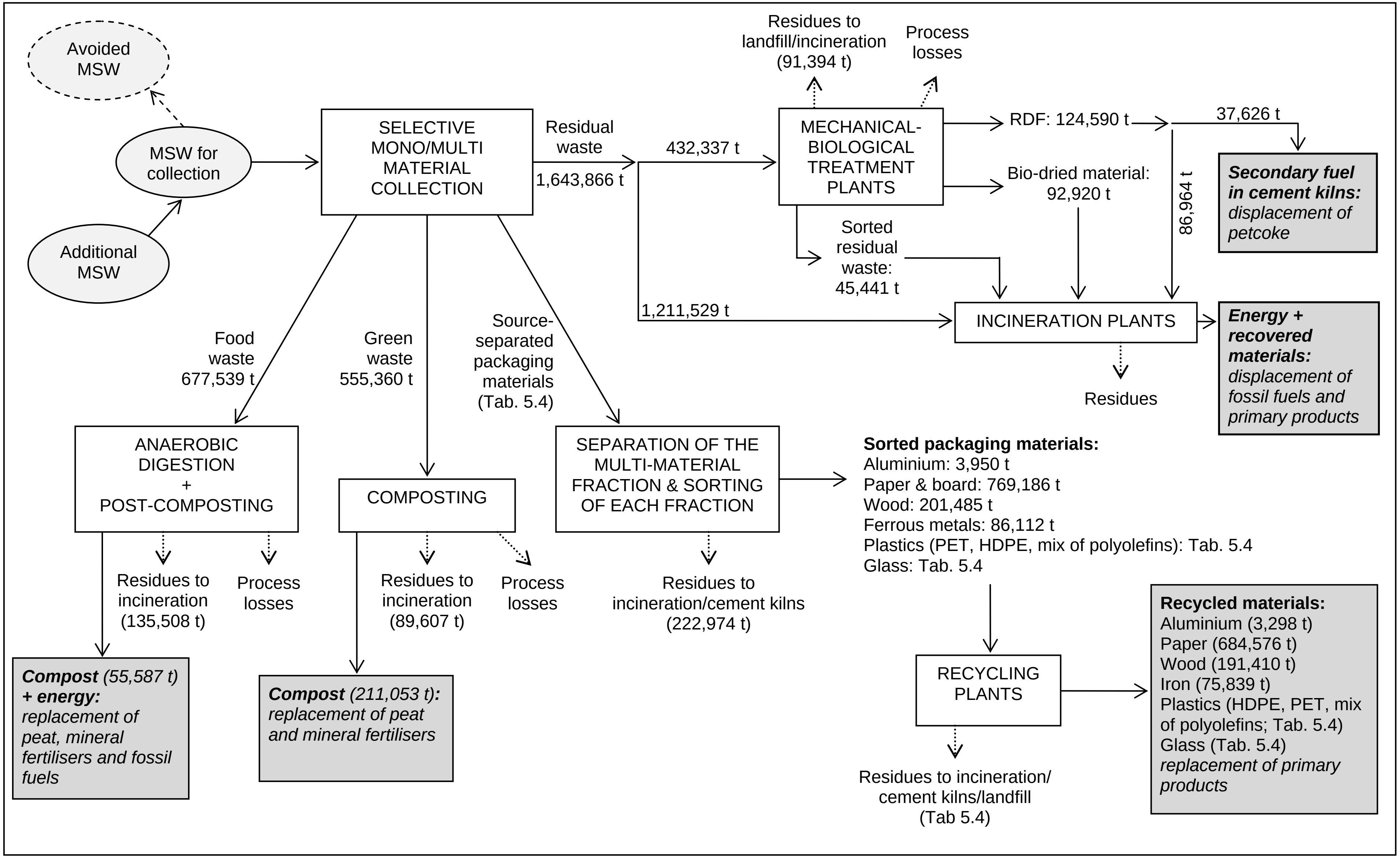


Figure 5.2: yearly flows of waste of the analysed scenarios for the waste management system of the Lombardia Region. The magnitudes of the waste flows affected by waste prevention activities are reported in Table 5.4.

5.7 Waste management modelling

The modelling of the process units depicting traditional waste management operations (collection, transport, treatments etc.) was carried out according to the approach described in detail in Grosso et al. (2012) and briefly summarised in Rigamonti et al. (2013). The most important assumptions include the types of primary products substituted by those obtained from material and energy recovery processes, and on the respective substitution ratio (Table 5.5). This parameter takes into account the possible difference between the quality (inherent technical properties) of the secondary and the primary products. When the quality of the secondary product is worse than the corresponding primary product, a substitution ratio lower than 1 is considered, so that a lower amount is actually substituted. Producing recycled products with a lower quality can have different consequences. For instance, the recycled product could be used only in certain application, or there could be a limit to the number of recycling operations that this product can go through. Thus, the use of substitution factors aims at properly modelling these situations.

For many process units (e.g. paper recycling, anaerobic digestion, composting and incineration of the residual waste) data on the type and magnitude of inputs and outputs are primary, i.e. directly acquired from the operators of real plants. For other processes such as collection, transport, some recycling processes and primary production processes, inventory data from the ecoinvent v 2.2 database were used. However, they were frequently adapted and/or updated with more recent data from reference documents on best available techniques (BREFs) or other sources. Finally, data available from the technical and scientific literature were used for the remainder of the processes, such as mechanical-biological treatments of the residual waste and plastic recycling.

Table 5.5: products obtained from material and energy recovery operations and corresponding primary products assumed to be substituted (the substitution ratio is also reported).

Waste fraction	Secondary product	Substituted primary product	Substitution ratio
Ferrous metals	Generic steel product from continuous casting of melted iron scraps	Generic steel product from continuous casting of melted pig iron	1:1
Aluminium	Aluminium ingots from aluminium scraps	Aluminium ingots from bauxite	1:1
Glass	Generic green glass container from cullet (83.5%) and virgin raw materials (16.5%)	Generic glass container from virgin raw materials only	1:1
Wood	Particleboard from recovered wood	Plywood board from virgin wood	1:0.6[a]
Paper	Pulp from wastepaper (non-deinked)	Virgin thermo-mechanical pulp	1:0.8[b]
Plastics	Granules of recycled PET	Granules of virgin PET (amorphous)	1:0.8[c]
	Granules of recycled HDPE	Granules of virgin HDPE	1:0.8[c]
	Profiled bars from a mix of polyolefins	Wooden planks (50%) No substitution (50%)	1:1
Food waste	Compost	Peat and mineral fertilisers	-
	Electricity	Electricity from gas-fired combined cycle power plants[d]	
Green waste	Compost	Peat and mineral fertilisers	-
Residual waste and RDF (sent to incineration)	Electricity	Electricity from gas-fired combined cycle power plants[d]	1:1
	Heat	Heat from domestic gas-fired boilers[d]	1:1
RDF (sent to cement kilns)	RDF (secondary fuel)	Petcoke	1:1

Acronyms: HDPE = high-density polyethylene; PET = polyethylene terephthalate; RDF = refuse-derived fuel.
(a) Based on the technical properties (mechanical strength) of particle board and plywood one.
(b) Based on the maximum number of recycling operations that paper fibres can go through (5 in this study).
(c) Based on the market value of recycled and virgin granules.
(d) Based on the situation in Lombardia as of 2009, where natural gas represents more than 90% of the primary energy from fossil fuels used in both the electric and thermal sectors.

5.8 Modelling of upstream processes

5.8.1 Bottled water substitution

For the bottled water waste prevention activity, avoided and additional upstream processes depict, respectively, the whole upstream life cycle of the substituted bottled water and that of the replacing public network water. These processes were modelled according to the general approach (input data, inventory data etc.) described in Section 2.7. However, some parameters and assumptions were specifically adapted for this case study. The most important are summarised below.

Regarding bottled water, one-way PET bottles were assumed to be entirely manufactured from virgin raw materials, as recycled raw materials are currently used only to a limited extent. HDPE caps, plastic (PP) labels, heat-shrink wraps (LDPE) and most transport packages (wooden pallets, stretch film and top covering film) are manufactured from virgin raw materials as well. Paper labels and cardboard interlayers are instead partly (labels) or mostly (interlayers) manufactured from waste paper. The features of each packaging in terms of average mass, capacity and number of uses are summarised in Table A.1 of Appendix A and were defined based on experimental estimates available in the literature or data relating to a real bottling company located in northern Italy. Regarding the end of life of transport packages, they were all assumed to be entirely recycled as, generally, they are separately collected within commercial premises or bottling plants and then entrusted to private operators for recycling. These recycling processes are thus not part of the municipal waste management systems but of the avoided upstream processes.

An average distance of 275 km was then assumed to be covered by lorry to transport palletised water from bottling plants to retailers. This updated estimate takes into account the location of the facilities where the major brands of bottled water retailed in Lombardia are packaged. Finally, an overall distance of 10 km was assumed to be covered with a private car by the citizens, during each roundtrip to the retail outlets to purchase bottled water. Each roundtrip was assumed to be carried out to purchase 30 items overall, comprising a typical bundle containing 6 shrink-wrapped water bottles. Thus, each roundtrip was assigned only 1/30 of its overall potential impacts (see Section 2.7.1.4 for further details on the impacts of different assumptions on the number of items purchased contemporarily).

Regarding public network water, 94% of the total consumption was assumed to be groundwater withdrawn from natural springs and wells, the remaining 6% being surface water from lakes and mountain torrents (Regione Lombardia, 2008). Based on elaborations on data reported in the same source, 80% of groundwater was then assumed to undergo the sole disinfection by sodium hypochlorite (NaClO), while the remaining 20% is also subject to aeration and activated carbon filtration. Surface water is instead entirely subject to a more intense purification process carried out in a centralised plant and based on a sequence of both chemical and physical treatments. In Lombardia, network losses (adduction and distribution) amount to 20% on average (Regione Lombardia, 2008). In this study, all these losses were conservatively assumed to take place during distribution, so that 20% of purified water leaving treatment plants is lost. At the households, purified network water is further refined by means of a device based on activated carbon filtration and reverse osmosis (the most energy and water demanding technology available). The yearly volume of water filtered by a single device was conservatively assumed to be equal to 100 litres.

Refined water is finally withdrawn by means of 1 litre refillable glass jugs with an average mass of 475 g. These jugs were assumed to be used 100 times overall and washed in a dishwasher after every 5 uses as part of an overall load of 30 items (see the results reported in Section 2.9.2.1 for an overview of the effects of assuming different washing conditions).

5.8.2 Liquid detergent substitution

When single-use packaged liquid detergents are substituted, the processes belonging to their whole upstream life cycle are avoided. Conversely, the upstream processes in the life cycle of the replacing loose detergents are involved in addition. The modelling of these processes was carried out according to the data and the assumptions outlined in Section 3.9. Most input data for unit processes depicting the upstream life cycle of primary and transport packages were determined experimentally (e.g. the average masses of single-use and refillable containers and of their caps) or based on technical information directly acquired from packaging producers, retailers and/or technical data sheets available online (e.g. average masses of stretch film and pallet compositions). Moreover, a number of assumptions were performed about the origin of the packaging materials (virgin or recycled), the number of uses of refillable/reusable packages and the end of life of transport packages. In particular, substituted single-use containers were assumed to be entirely made out of virgin raw materials, as this is currently the most common practice. Refillable containers are exclusively manufactured from virgin raw materials, as well, to ensure their durability over time. Moreover, according to the recommendations drawn from the assessment summarised in Section 3, such containers were then assumed to be used for 10 times overall before being discarded by the citizens. Virgin raw materials are used also for the production of caps and most transport packages (i.e. the pallets, the stretch-film and the inner container of reusable tanks), while cardboard boxes are entirely produced from recycled fibres. Similarly, the steel cage of reusable tanks is partly produced from post-consumer iron scraps.

Similarly to the assumptions performed for bottled water (Section 5.8.1), all transport packages were assumed to be recycled at the end of their useful life, including the different components of the reusable tanks used for the transport of loose detergents. However, these tanks were assumed to be used for 50 cycles of transport before being discarded at the packaging plant and sent to recycling.

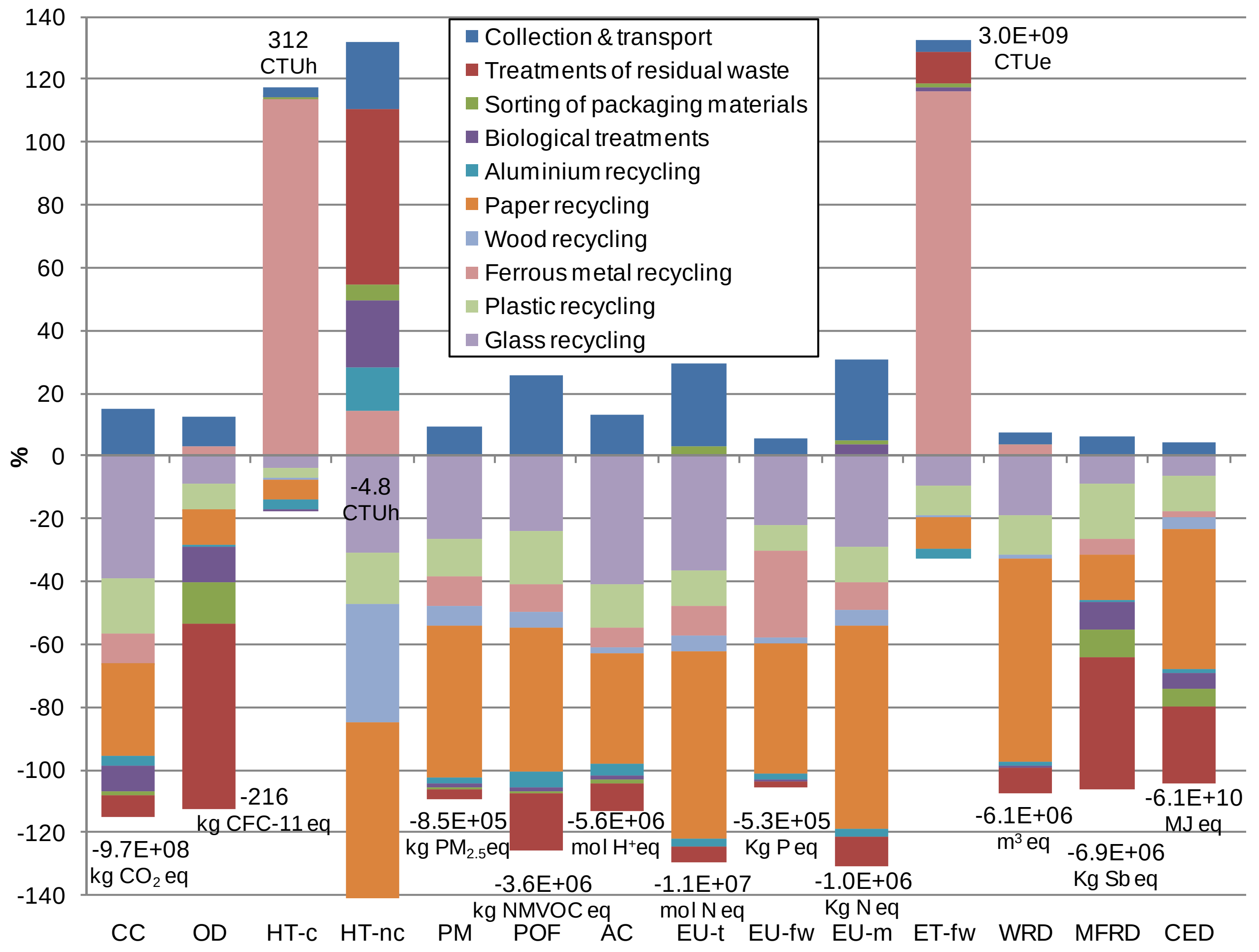
For the transport phase, an average distance of 340 km was estimated to be covered by lorries to deliver the detergent and the associated packages to retailers. The same distance was assumed for both single-use packaged and loose detergents.

Finally, when loose detergents are used, the burdens of the sale and purchase phases were also estimated. First, the modelling included the consumptions of electricity for the refilling of the self-dispensing system and for the withdrawal of the detergent by the consumers. Moreover, the life cycles of the main components of the systems were taken into account. For this purpose, a useful life of 10 years was specifically assumed for the self-dispensing system, along with an annual supply of about 75,000 litres.

5.9 Results and discussion

5.9.1 Environmental performance of the baseline scenario

As usual in LCA of advanced waste management systems, most impact indicators show a negative value (Figure 5.3). This means that the overall benefits from recycling and energy recovery operations compensate for the adverse impacts (loads) from the collection, transport and processing of the many waste streams (Figure 5.3). Exceptions are the *human toxicity, cancer effects* and the *freshwater ecotoxicity* indicators, which are both positive. This is because of the huge adverse impact of the recycling of ferrous metals, which, together with the other minor positive contributions, by far exceed the limited benefits associated with the recycling of the other source-separated materials (Figure 5.3). For *human toxicity, non-cancer effects* the impact is close to zero, as loads and benefits are balanced. In this category, loads are not only associated with collection, transport and sorting of the source-separated fractions. Conversely, also the processing of the residual waste, the biological treatment of the organic fractions and the recycling of ferrous metals and aluminium show an overall adverse impacts. Recycling of glass, paper, plastic and wood still involves an overall benefit.

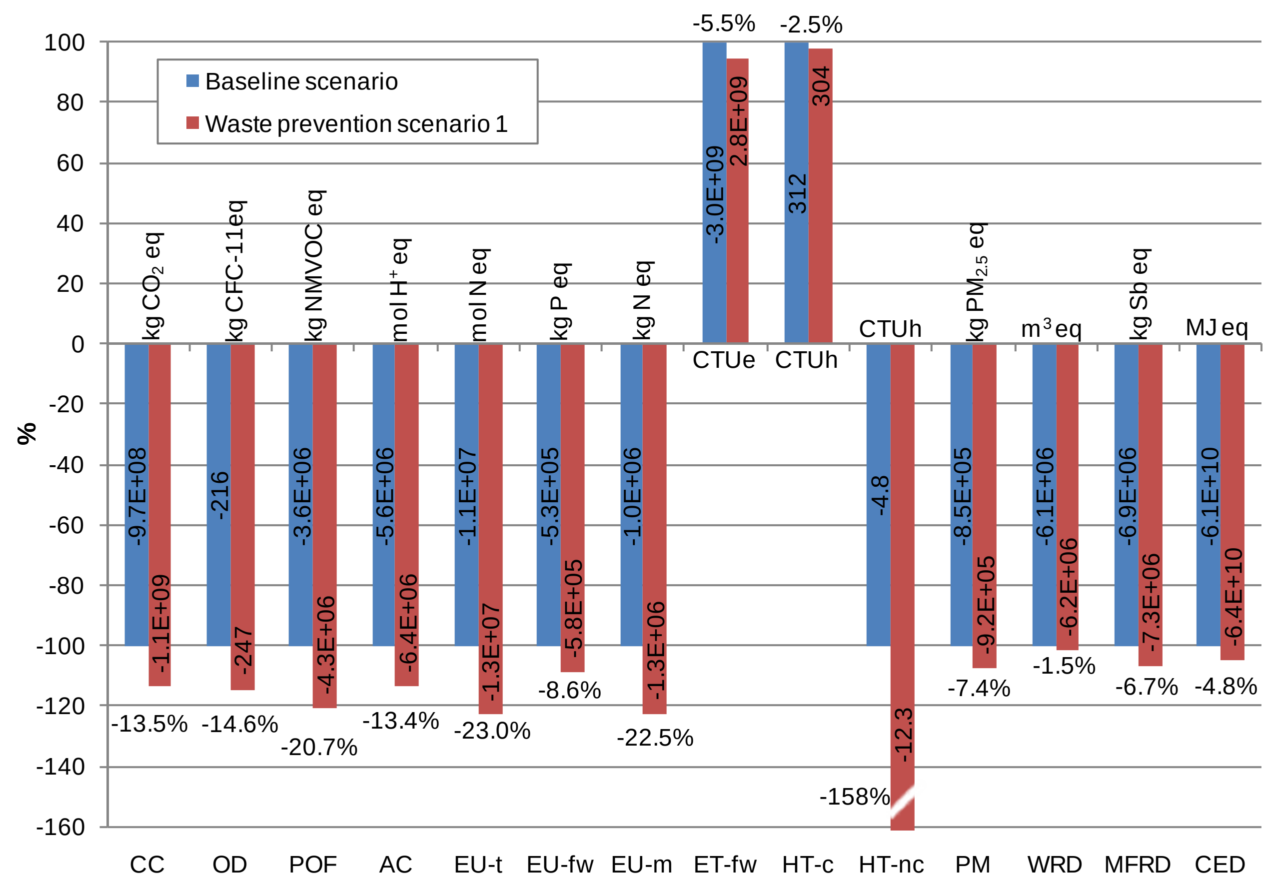


CC: climate change; OD: ozone depletion; HT-c: human toxicity (cancer effects); HT-nc: human toxicity (non-cancer effects); PM: particulate matter; POF: photochemical ozone formation; AC: acidification; EU-t: eutrophication (terrestrial); EU-fw: eutrophication (freshwater); EU-m: eutrophication (marine); ET-fw: ecotoxicity (freshwater); WRD: water resource depletion; MFRD: mineral and fossil resource depletion; CED: cumulative energy demand.

Figure 5.3: potential impacts of the baseline scenario and percentage contribution of the main sub-processes to the total impacts.

5.9.2 Impact of the bottled water substitution

When a complete substitution of bottled water by public network water is introduced in the system as a waste prevention activity (WPS1), the overall environmental profile is further improved compared to the baseline scenario, although to a different extent among the selected impact categories (Figure 5.4). For half of them, an increase in benefits larger than 10% is observed: *climate change* (13.5%), *ozone depletion* (14.5%), *human toxicity* (*non-cancer effects*), 158%, *photochemical ozone formation* (21%), *acidification* (13.5%), *terrestrial eutrophication* (23%) and *marine eutrophication* (22.5%). The remaining categories show a reduced improvement (increase in benefits or reduction in impacts), which is however lower than 5% only for *the human toxicity, cancer effects* impact category (2.5%) and the *water resource depletion* one (1.5%).



CC: climate change; OD: ozone depletion; HT-c: human toxicity (cancer effects); HT-nc: human toxicity (non-cancer effects); PM: particulate matter; POF: photochemical ozone formation; AC: acidification; EU-t: eutrophication (terrestrial); EU-fw: eutrophication (freshwater); EU-m: eutrophication (marine); ET-fw: ecotoxicity (freshwater); WRD: water resource depletion; MFRD: mineral and fossil resource depletion; CED: cumulative energy demand.

Figure 5.4: comparison between the potential impacts of the baseline scenario and of the waste prevention scenario substituting bottled water by tap water (percentage impact variations between the latter and the former are also reported at the end of each couple of bars).

The observed improvements are mostly due to the additional upstream benefits introduced in the system by the implemented waste prevention activity (Figure 5.5). These benefits are the balance between the savings from the avoided production, transport and purchase of the substituted bottled water and the additional impacts from the production, refining and consumption of the replacing public network water. Since additional upstream impacts are always lower than those avoided, a net upstream benefit is achieved, overall, for all impact categories (Table 5.6). Conversely, waste prevention involves only marginal effects on the impacts of the traditional components of the waste management systems (downstream components) affected by the prevention activity (Figure 5.5). This is likely because the waste prevented is only a small and relatively harmless fraction of the total waste (0.52%). In particular, the impacts of waste collection and transport decrease by 0.8% on average, while those of sorting of source-separated packaging materials by 8% (Table D.8 of Appendix D). Benefits (or impacts) from recycling activities are instead reduced by an average

3.5%. The result is an overall increase in net downstream impacts (be it positive or negative) as the decrease in recycling benefits exceeds the reduction in the impacts of collection, transport, and sorting of the recyclable materials affected by prevention. Even this overall increase is limited (lower than 4% for most categories), so that it is always compensated by the upstream benefits from waste prevention.

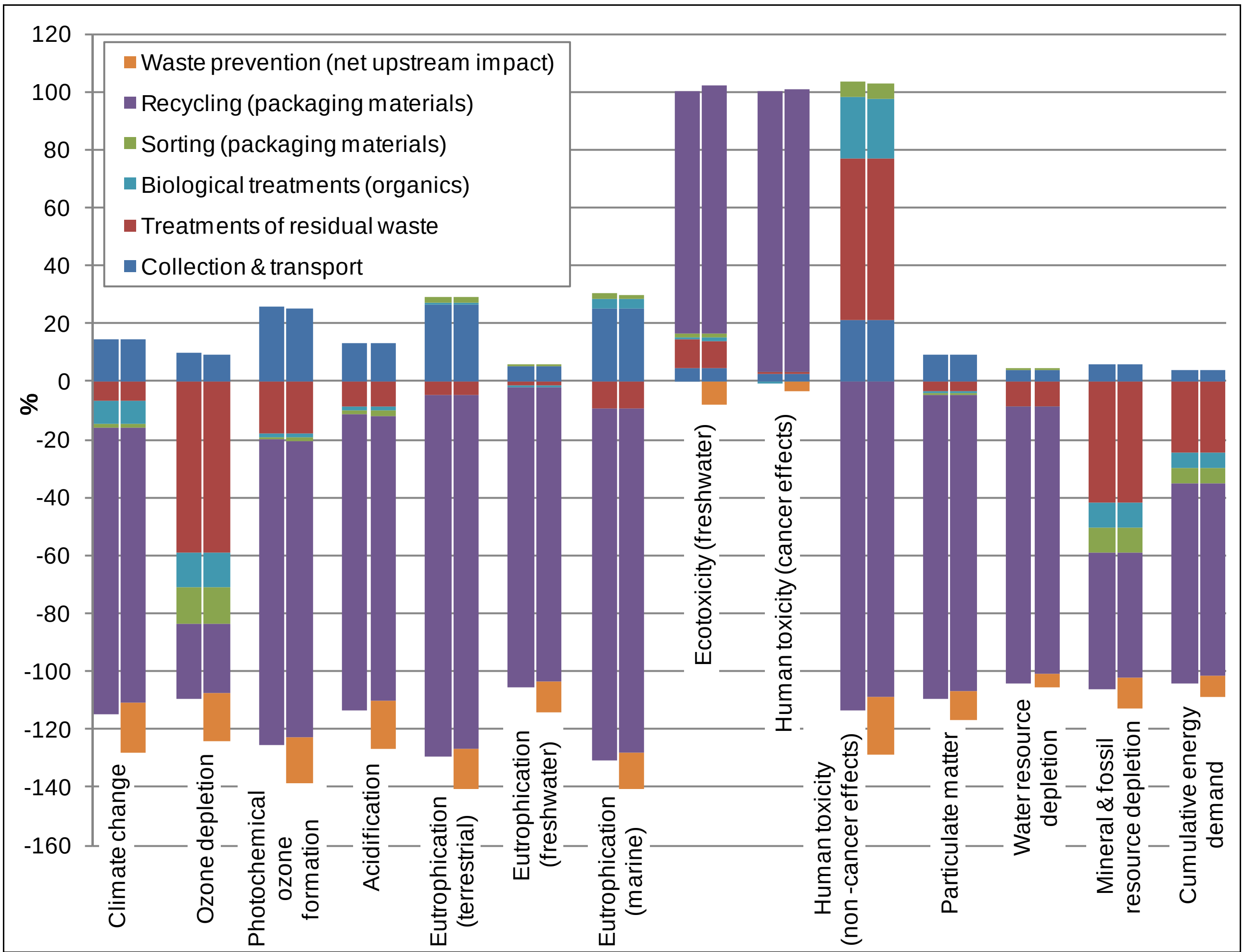


Figure 5.5: contribution of the main sub-processes to the total impacts of the baseline scenario (left bar of each couple) and of the waste prevention scenario substitution bottled water by tap water (right bar). Contributions are expressed as the percentage of the total impact of the baseline scenario.

Thus, it is not surprising that the most significant improvements in the overall performance of the system generally occur for those impact categories where the contribution of these additional upstream benefits to the total impact of the system is more important (and vice versa). However, some results require a specific interpretation. For instance, the nearly 160% increase in the overall benefits observed for *human toxicity, non-cancer effects* is a consequence of the fact that the total impact of traditional waste management operations is close to zero[2] (-2.8 CTU_h per functional unit)

[2] Because the overall loads and benefits are balanced out (Figure 5.5).

and the additional upstream benefit from waste prevention is three times greater than such an impact (-9.5 CTU_h per functional unit).

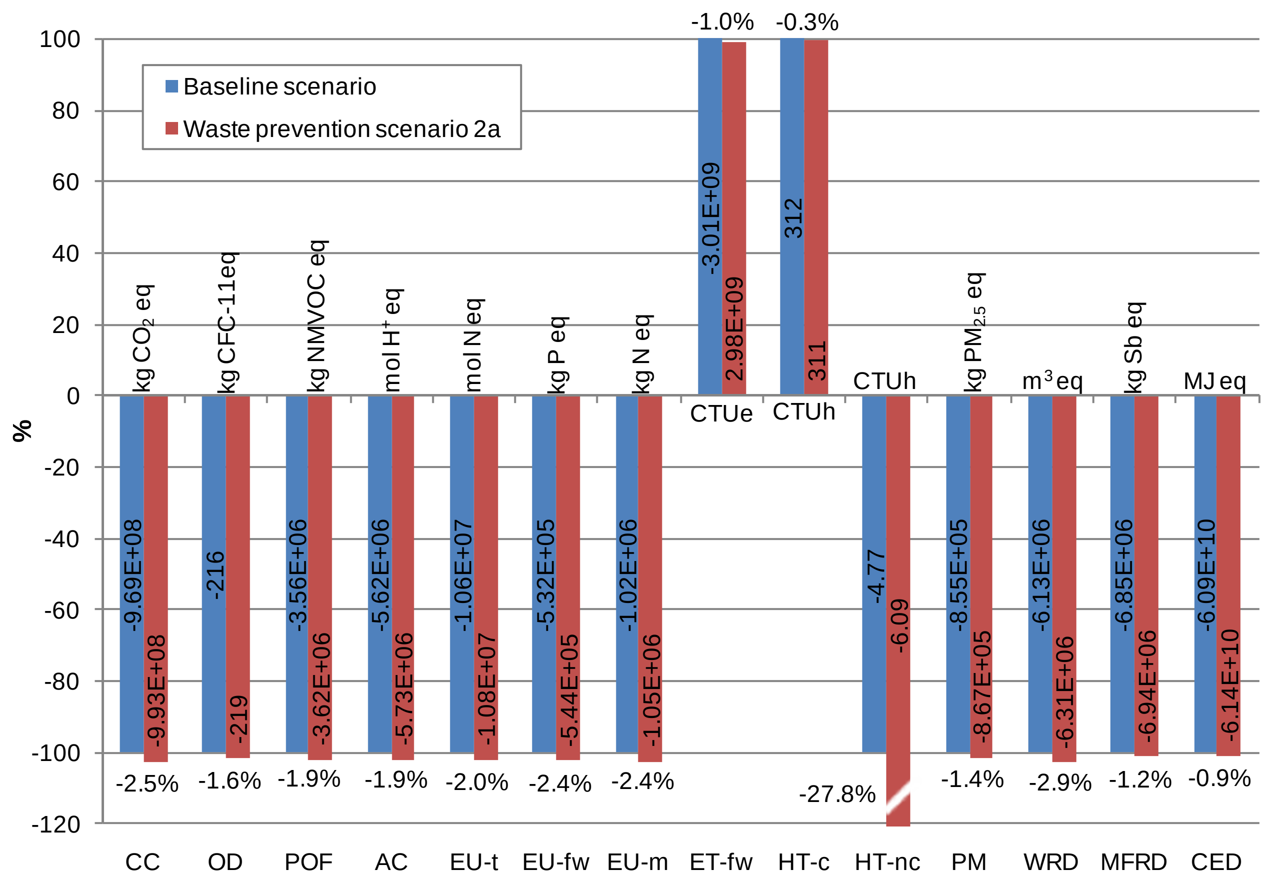
Table 5.6: upstream impacts of the waste prevention activity substituting bottled water by tap water within waste prevention scenario 1.

Impact category	Unit	Avoided upstream impact	Additional upstream impact	Net upstream impact	% of scenario total impact
Climate change	kg CO_2 eq.	$1.99x10^8$	$3.22x10^7$	$-1.67x10^8$	-15.2
Ozone depletion	kg CFC-11 eq.	37.5	2.2	-35.3	-14.3
Photochemical ozone formation	kg NMVOC eq.	$9.05x10^5$	$6.26x10^4$	$-8.43x10^5$	-19.6
Acidification	mol H+ eq.	$1.04x10^6$	$1.26x10^5$	$-9.18x10^5$	-14.4
Terrestrial eutrophication	mol N eq.	$2.87x10^6$	$2.15x10^5$	$-2.66x10^6$	-20.4
Freshwater eutrophication	kg P eq.	$6.06x10^4$	$3.67x10^3$	$-5.69x10^4$	-9.9
Marine eutrophication	kg N eq.	$2.72x10^5$	$2.02x10^4$	$-2.52x10^5$	-20.1
Freshwater ecotoxicity	CTUe	$2.58x10^8$	$2.49x10^7$	$-2.33x10^8$	-8.2
Human toxicity (cancer effects)	CTUh	11.8	1.6	-10.2	-3.4
Human toxicity (non-cancer effects)	CTUh	10.5	1.1	-9.4	-76.9
Particulate matter	kg $PM_{2.5}$ eq.	$9.63x10^4$	$1.22x10^4$	$-8.41x10^4$	-9.2
Water resource depletion	m^3 water eq.	$1.10x10^6$	$8.16x10^5$	$-2.81x10^5$	-4.5
Mineral and fossil resource depletion	kg Sb eq.	$7.89x10^5$	$8.72x10^4$	$-7.02x10^5$	-9.6
Cumulative energy demand	MJ eq.	$4.68x10^9$	$3.81x10^8$	$-4.29x10^9$	-6.7

5.9.3 Impact of the liquid detergent substitution

Figures 5.6 and 5.7 show the results obtained when liquid detergents packaged in single-use containers are entirely substituted by those distributed loose through self-dispensing systems and refillable containers (WPS2a and WPS2b). In the comparison depicted in Figure 5.6, both types of detergents are assumed to have the same average washing performance, so that they are used in the same quantity. In Figure 5.7, "loose" detergents are instead more diluted than many substituted detergents. A higher volume of detergent is thus needed in the waste prevention scenario, to perform the same overall number of washings as the baseline[3].

[3] The additional volume needed was assumed to be demineralised water used to dilute the detergents.



CC: climate change; OD: ozone depletion; HT-c: human toxicity (cancer effects); HT-nc: human toxicity (non-cancer effects); PM: particulate matter; POF: photochemical ozone formation; AC: acidification; EU-t: eutrophication (terrestrial); EU-fw: eutrophication (freshwater); EU-m: eutrophication (marine); ET-fw: ecotoxicity (freshwater); WRD: water resource depletion; MFRD: mineral and fossil resource depletion; CED: cumulative energy demand.

Figure 5.6: comparison between the potential impacts of the baseline scenario and of the waste prevention scenario substituting single-use packaged liquid detergents by loose detergents with the same average washing performance (percentage impact variations between the waste prevention scenario and the baseline are also reported at the end of each couple of bars).

When no difference is considered between the two types of detergents (Figure 5.6), the performance of the waste management system is improved only to a minor extent (1 to 3%) for most impact categories. For *human toxicity, cancer effects*, insignificant changes are involved (0.3%), while for non-cancer effects the achieved improvements are higher, reaching 28% (but again due to a minor change applied to very low absolute value).

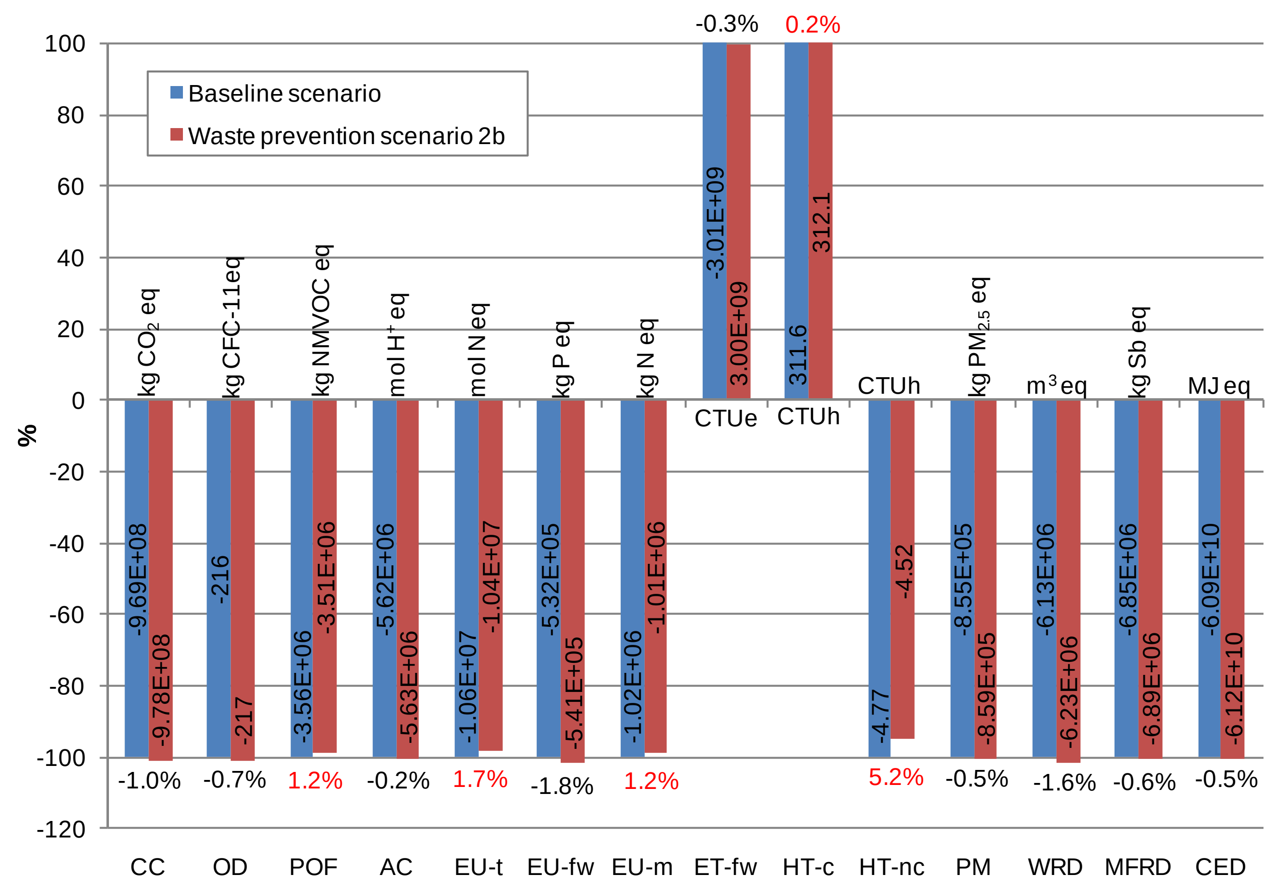
Arguably, the reduced waste prevention potential (0.14 % of the total waste) is the main reason for achieving these limited improvements. As only a small (and relatively harmless) portion of the total waste is removed from the system, the effects of waste prevention on downstream impacts are again insignificant: 0.2% mean reduction for collection and transport, 2% for sorting of source-separated packaging materials, and decrease in recycling benefits by an average 0.7%. As a result, the net downstream impacts are increased by only 0.75% on average (Table D.9 of Appendix D).

Even the net upstream benefits produced by waste prevention are generally limited, reaching between 0.4 and 3% of the absolute impacts of the waste prevention scenario (if the *human toxicity, non-cancer effects* impact category is excluded; Table 5.7). As shown in Table 5.7, this is not because the additional impacts of the upstream life cycle of the replacing loose detergents tend to balance the avoided impacts of the upstream life cycle of the substituted, single-use, packaged detergents. Conversely, it is likely that the upstream impacts are limited compared to those of the system as a whole, just because the quantity of material removed from and added to the system is small relatively to the total waste. However, it is noteworthy that the relative improvements in the overall performance of the system (1-28%) are always proportionally greater than the net proportion of prevented waste (0.14%).

Table 5.7: upstream impacts of the waste prevention activity substituting single-use packaged liquid detergents by loose detergents within waste prevention scenario 2a (both types of detergents have the same average washing performance).

Impact category	Unit	Avoided upstream impact	Additional upstream impact	Net upstream impact	% of scenario total impact
Climate change	kg CO_2 eq.	3.87×10^7	6.70×10^6	-3.20×10^7	-3.2
Ozone depletion	kg CFC-11 eq.	4.58	0.57	-4.01	-1.8
Photochemical ozone formation	kg NMVOC eq.	1.30×10^5	3.06×10^4	-9.93×10^4	-2.7
Acidification	mol H+ eq.	1.79×10^5	3.69×10^4	-1.42×10^5	-2.5
Terrestrial eutrophication	mol N eq.	3.51×10^5	9.35×10^4	-2.57×10^5	-2.4
Freshwater eutrophication	kg P eq.	1.56×10^4	2.05×10^3	-1.35×10^4	-2.5
Marine eutrophication	kg N eq.	3.82×10^4	9.14×10^3	-2.90×10^4	-2.8
Freshwater ecotoxicity	CTUe	5.56×10^7	1.31×10^7	-4.25×10^7	-1.4
Human toxicity (cancer effects)	CTUh	2.09	0.86	-1.23	-0.4
Human toxicity (non-cancer effects)	CTUh	2.44	0.92	-1.52	-24.9
Particulate matter	kg $PM_{2.5}$ eq.	1.94×10^4	3.25×10^3	-1.62×10^4	-1.9
Water resource depletion	m^3 water eq.	2.34×10^5	3.04×10^4	-2.04×10^5	-3.2
Mineral and fossil resource depletion	kg Sb eq.	1.73×10^5	2.66×10^4	-1.46×10^5	-2.1
Cumulative energy demand	MJ eq.	1.09×10^9	1.65×10^8	-9.22×10^8	-1.5

When a worse washing performance is assumed for the replacing loose detergents, no significant change is involved in the overall performance of the system for most impact categories (impact variation not exceeding 1% compared to the baseline; Figure 5.7). Moreover, for some categories, an overall worsening is observed: human toxicity, non-cancer effects (5.2%); terrestrial eutrophication (1.7%) photochemical ozone formation (1.2%) and marine eutrophication (1.2%). For these categories, the additional upstream impacts from the waste prevention activity exceed the avoided upstream impacts (Table 5.8), mostly due to an increased impact of transport. An additional amount of detergent (dilution water) needs indeed to be transported to retailers when the detergent is distributed loose. Moreover, for *human toxicity, non-cancer effects* (and for toxicity-related impact categories in general), an important additional upstream impact is provided by the life cycle of the reusable tanks used for the transport of the detergent (for the reasons explained in Section 3). For most of the remaining categories, more than 50% of the avoided upstream impacts are compensated by the additional upstream impacts, always because of the increase in transport impacts. This explains the negligible improvements achieved in the overall performance of the waste management system for these categories.



CC: climate change; OD: ozone depletion; HT-c: human toxicity (cancer effects); HT-nc: human toxicity (non-cancer effects); PM: particulate matter; POF: photochemical ozone formation; AC: acidification; EU-t: eutrophication (terrestrial); EU-fw: eutrophication (freshwater); EU-m: eutrophication (marine); ET-fw: ecotoxicity (freshwater); WRD: water resource depletion; MFRD: mineral and fossil resource depletion; CED: cumulative energy demand.

Figure 5.7: comparison between the potential impacts of the baseline scenario and of the waste prevention scenario substituting single-use packaged liquid detergents by loose detergents with a worse washing performance (percentage impact variations between the waste prevention scenario and the baseline are also reported at the end of each couple of bars).

Table 5.8: upstream impacts of the waste prevention activity substituting single-use packaged liquid detergents by loose detergents within waste prevention scenario 2b (loose detergents are more diluted than substituted ones).

Impact category	Unit	Avoided upstream impact	Additional upstream impact	Net upstream impact	% of scenario total impact
Climate change	kg CO_2 eq.	3.87×10^7	2.23×10^7	-1.64×10^7	-1.7
Ozone depletion	kg CFC-11 eq.	4.58	2.60	-1.98	-0.9
Photochemical ozone formation	kg NMVOC eq.	1.30×10^5	1.45×10^5	1.54×10^4	0.4
Acidification	mol H+ eq.	1.79×10^5	1.40×10^5	-3.91×10^4	-0.7
Terrestrial eutrophication	mol N eq.	3.51×10^5	4.91×10^5	1.41×10^5	1.3
Freshwater eutrophication	kg P eq.	1.56×10^4	5.11×10^3	-1.04×10^4	-1.9
Marine eutrophication	kg N eq.	3.82×10^4	4.61×10^4	7.96×10^3	0.8
Freshwater ecotoxicity	CTUe	5.56×10^7	3.43×10^7	-2.13×10^7	-0.7
Human toxicity (cancer effects)	CTUh	2.09	2.27	0.18	0.1
Human toxicity (non-cancer effects)	CTUh	2.44	2.49	0.06	1.3
Particulate matter	kg $PM_{2.5}$ eq.	1.94×10^4	1.10×10^4	-8.45×10^3	-1.0
Water resource depletion	m^3 water eq.	2.34×10^5	1.12×10^5	-1.22×10^5	-2.0
Mineral and fossil resource depletion	kg Sb eq.	1.73×10^5	7.98×10^4	-9.32×10^4	-1.4
Cumulative energy demand	MJ eq.	1.09×10^9	4.84×10^8	-6.03×10^8	-1.0

5.9.4 Impact of the combined substitution

When both the considered product substitutions are implemented in the system as waste prevention activities (WPS 3), the comparison with the baseline scenario provides the results reported in Table 5.9. As expected, the relative improvements in the overall performance of the system are simply the sum of those obtained by singularly implementing the two waste prevention activities (Figures 5.4 and 5.6). Specifically, an improvement in the range of 15-25% is achieved for half of the impact categories. Most of the remaining categories are improved by between 6% and 8%, while the improvements achieved for *human toxicity (cancer effects)* and *water resource depletion* are still limited (2.8% and 4%, respectively). With the current level of consumptions of liquid detergents in Italy, introducing an alternative distribution method based on self-dispensing systems and refillable containers can thus contribute to increase the benefits of a structured set of prevention activities, although it proved to be little effective as a stand-alone activity.

Table 5.9: comparison between the potential impacts of the baseline scenario and of the waste prevention scenario implementing the substitution of both bottled water and liquid detergents (assuming an identical washing performance).

Impact category	Unit	Baseline scenario (BLS)	Waste prevention scenario 3 (WPS3)	Variation between BLS and WPS3 (%)
Climate change	kg CO_2 eq.	$-9.69x10^8$	$-1.12x10^9$	-16.0
Ozone depletion	kg CFC-11 eq.	-216	-251	-16.2
Photochemical ozone formation	kg NMVOC eq.	$-3.56x10^6$	$-4.36x10^6$	-22.6
Acidification	mol H+ eq.	$-5.62x10^6$	$-6.49x10^6$	-15.4
Terrestrial eutrophication	mol N eq.	$-1.06x10^7$	$-1.33x10^7$	-25.0
Freshwater eutrophication	kg P eq.	$-5.32x10^5$	$-5.90x10^5$	-11.0
Marine eutrophication	kg N eq.	$-1.02x10^6$	$-1.28x10^6$	-24.9
Freshwater ecotoxicity	CTUe	$3.01x10^9$	$2.82x10^9$	-6.5
Human toxicity (cancer effects)	CTUh	312	303	-2.8
Human toxicity (non-cancer effects)	CTUh	-4.77	-13.6	-186
Particulate matter	kg $PM_{2.5}$ eq.	$-8.55x10^5$	$-9.30x10^5$	-8.8
Water resource depletion	m^3 water eq.	$-6.13x10^6$	$-6.40x10^6$	-4.4
Mineral and fossil resource depletion	kg Sb eq.	$-6.85x10^6$	$-7.40x10^6$	-7.9
Cumulative energy demand	MJ eq.	$-6.09x10^{10}$	$-6.44x10^{10}$	-5.7

5.9.5 Results of the sensitivity analysis

A sensitivity analysis was finally undertaken to evaluate the effects of varying the proportion of traditional product subject to substitution on the achieved improvements. Specifically, the analysis focused on the bottled water substitution, by assuming than only 50% of the overall volume of bottled water suitable for substitution is actually replaced with public network water by the citizens of Lombardia. The results of the analysis are shown in Table 5.10, which compares the variations in the overall impacts of the system, achieved with both a complete (100%) and a partial (50%) replacement of the volume of bottled water suitable for substitution. As it can be observed, a 50% decrease in the volume of substituted bottled water, reduces the achievable improvements in the same proportion. Thus, as expected, the benefits achievable by implementing a waste prevention activity based on product substitution depend linearly on the percentage of traditional product actually replaced wit the alternative, less waste-generating one.

Table 5.10: percentage impact variation between waste prevention scenario 1 and the baseline scenario, for two different levels of substitution of bottled water by public network water.

Impact category	Complete (100%) substitution	Partial (50%) substitution
Climate change	-13.5%	-6.8%
Ozone depletion	-14.6%	-7.3%
Photochemical ozone formation	-20.7%	-10.4%
Acidification	-13.4%	-6.7%
Terrestrial eutrophication	-23.0%	-11.5%
Freshwater eutrophication	-8.6%	-4.3%
Marine eutrophication	-22.5%	-11.3%
Freshwater ecotoxicity	-5.5%	-2.8%
Human toxicity (cancer effects)	-2.5%	-1.3%
Human toxicity (non-cancer effects)	-158%	-78.9%
Particulate matter	-7.4%	-3.7%
Water resource depletion	-1.5%	-0.7%
Mineral and fossil resource depletion	-6.7%	-3.3%
Cumulative energy demand	-4.8%	-2.4%

5.10 Concluding remarks

Two packaging waste prevention activities were separately and then contemporarily implemented in a 2020 municipal solid waste management scenario for Lombardia, Italy. The first activity implements the complete substitution of the domestic consumption of one-way bottled water by that of purified water from the public network (tap). In the second activity, the consumption of three categories of liquid detergents packaged in single-use containers is entirely replaced with that of the same type of detergents distributed “loose” through self-dispensing systems and refillable containers.

The results revealed that, when the substitution is beneficial[4], the overall environmental performance of the waste management system is improved to a greater proportion than that of waste prevented, independently of the substitution performed. For instance, the percentage increase in the overall benefits of the system is proportionally greater than the net percentage decrease in the total waste mass resulting from a particular prevention activity.

In keeping with the results of some recent studies, these overall improvements were found to be mostly enabled by the upstream benefits introduced in the system by waste prevention, rather than by a reduction in the net impacts (loads or benefits) of the downstream components of the system itself. In fact, an overall increase in net downstream impacts is generally observed, because the decrease in recycling benefits is larger than the decrease in the adverse impacts of collection,

[4] i.e. when the net impacts avoided thanks to the substitution are lower than the net additional impacts.

transport and sorting of recyclable materials affected by waste prevention. However, this overall increase in downstream impacts is generally limited and widely compensated by the additional upstream benefits from waste prevention. These benefits are a result of the balance between the benefits from the avoided production and use of the substituted good and the additional impacts from the production and use of the replacing less waste-generating goods, which are always lower than the former.

Despite the relative improvements in the overall performance of the system are always proportionally greater than the relative reduction in the tonnage of waste collected, they are appreciable only for the activity replacing bottled water with public network water. In this case, a 0.5% reduction of the total waste mass allows for an improvement (increase in benefits or reduction in impacts), which for most categories ranges between 5 and 23%. As this prevention activity is relatively easy to undertake by citizens and does not require important structural changes in upstream supply chains, its implementation is encouraged to further improve the performance of waste management at the regional level. The observed improvements are achieved for a complete substitution of bottled water, which would hardly take place in reality. However, a lower substitution at the domestic level may be at least partially compensated by expanding the activity also to the catering industry.

When the implemented activity replaces single-use packaged liquid detergents with an identical amount of loose detergents (i.e. both types of detergents have the same washing performance) the improvements in the overall performance of the system are definitely lower than those achieved when bottled water is replaced. Excluding the *human toxicity, non-cancer effects* impact category (where a 28% increase in the overall benefits is achieved) such improvements never exceed 3%. This is mostly because the substituted volume of detergent is smaller compared to bottled water (88% less), as the estimated consumption is lower (14 vs 113 litres per inhabitant) and, as a consequence, the fraction of prevented waste is limited (0.14 % of total waste mass), although a complete substitution was assumed.

The implementation of this prevention activity can thus contribute to moderately increase the benefits of a structured set of municipal waste prevention measures, but it is little effective as a stand-alone measure. However, when implementing the activity, it is fundamental that the replacing detergents have equivalent (or better) washing performance to that of substituted detergents (i.e., generally, similar concentrations). In this condition, an identical amount of detergent will be approximately used before and after the substitution, so that no additional impacts will be involved by the life cycle of the added detergent. Otherwise, the modelling demonstrated that most of the poor improvements in the overall performance of the system are vanished and a worsening even takes place for some impact categories.

6 Conclusions

6.1 Concluding remarks

Waste prevention has become one of the pillars of the European waste management policy in the last decade. To help Europe to become society that seeks to avoid waste and use them as resources, the member states are now required to develop national waste prevention programmes, setting quantitative targets and appropriate measures for their achievement (Waste Framework Directive 2008/98/EC). Such programmes should focus on the key environmental impacts associated with the whole life cycle of products and materials becoming waste, and pursue the dissociation of these impacts from the economic growth. In Italy, each Region is also required to prepare a specific waste prevention programme that defines further objectives and tangible actions to be implemented locally. Both the waste framework directive and the Italian legislation require that waste management (and thus prevention) options are chosen so that an overall positive environmental outcome is achieved when the impacts associated with the whole life cycle of products are taken into account. Thus, selected waste prevention measures and actions shall not only allow for a reduction in the quantity or hazardousness of waste, but also in the overall environmental impacts.

6.1.1 Review and classification of municipal waste prevention activities

A comprehensive review of viable measures and actions for the prevention of municipal waste (Section 1) has shown that these can be based on four main mechanisms, which generate different environmental consequences. Such mechanisms include (a) the reduction in the consumption of products or services; (b) the substitution of products or services with less waste-generating equivalent ones; (c) the reuse of disposable or durable goods; and (d) the extension of the lifespan of durable goods. While a reduction in product or service consumption is expected to generate only environmental and energy benefits (provided that possible “rebound effects” due to the increased income available to the consumers are not taking place), the other types of mechanisms also involve additional impacts (due, e.g., to the consumption of alternative products or services). The balance between avoided and additional impacts needs thus to be carefully evaluated, in a life cycle perspective, to properly assess the environmental and energy convenience of those waste prevention activities that are not based on the “simple” reduction in product or service consumption. The results of the two life cycle assessment (LCA) case studies reported in the first part of this thesis (Sections 2 and 3) definitely support this claim.

6.1.2 Life cycle assessment of packaging waste prevention activities based on product substitution

The environmental and energy convenience of two municipal waste prevention activities based on product substitution was initially evaluated, by means of life cycle assessment. The first activity (case study 1, Section 2) aims at reducing the amount of waste generated from the consumption of drinking water by substituting one-way or refillable bottled water by public network water withdrawn from the tap or public fountains. It was considered one of the most meaningful activities for Italy, which is among the largest per-capita consumers of bottled water worldwide. Moreover, this activity is frequently included in the most recent reviews of best practices and in many regional waste prevention programmes. In the second examined activity (case study 2, Section 3), three categories of liquid detergents packaged in single-use plastic containers are instead substituted by those that can be withdrawn from self-dispensing systems by means of refillable containers (the so-called loose detergents). This alternative type of distribution has recently been tested by some Italian producers, in an attempt to reduce waste generation and the environmental impacts associated with the delivery of liquid detergents. For both activities, the approach was to compare different baseline scenarios using the substituted product (e.g. bottled water) with two waste prevention scenarios using the alternative product (e.g. tap water), and then to evaluate the net impact variation. Each baseline scenario considered the use of a particular type of potentially substituted product, such as drinking water packaged in bottles of a given material and size. Similarly, waste prevention scenarios depicted different ways of providing the citizens with the alternative, less waste-generating product (e.g. refined drinking water from the household tap or from public fountains).

The results of the two LCAs revealed that the ultimate environmental and energy convenience of a waste prevention activity based on product substitution often depends on a number of variables, which frequently depict the way the activity is actually implemented by the actors involved (citizens, institutions, producers etc.). For instance, when bottled water is replaced by network water withdrawn from public fountains (case study 1, Section 2), the convenience of the substitution primarily depends on whether a car is used for the roundtrip to the fountain (i.e. citizen's behaviour). If no motorised vehicles are used, the substitution is beneficial for all impact categories. Otherwise, the convenience depends on further variables. These include the travelled distance (way of implementation of the activity by municipalities), the volume of water withdrawn and transported to the household (citizens' behaviour) and the distance along which the replaced bottled water is transported to retailers or local distributors (geographic variable). Similarly, if the container used to withdraw tap water at the household is washed under inefficient conditions, the replacement of refillable PET[1] bottled water is beneficial only for few impact categories, when the latter is transported to local distributors along a short distance (e.g. 40 km). In this case, other than the behaviour of the consumer and the transport distance, even the type of packaging used for the substituted traditional product is an important variable.

When loose liquid detergents are substituted for those packaged in single-use containers (case study 2, Section 3), the behaviour of the consumers plays once again a key role. Indeed, refillable containers should be used at least 5-10 times (depending on the waste prevention scenario) for all the examined product substitutions to be advantageous in most impact categories. Moreover, for a particular category (human toxicity, non cancer effects), in case of substituting big sized single-use HDPE[1] containers, no significant benefits are achieved even for 50 uses of the refillable container.

6.1.3 Discussion on methods to include prevention activities in waste management LCA

Two alternative methodological approaches to incorporate waste prevention activities into LCAs of integrated municipal waste management systems were identified, presented and discussed (Section 4). The identification was based on both a structured reorganisation of the amendments and approaches already available in the scientific literature and on further personal elaborations and research. The two approaches are conceived to compare municipal waste management scenarios implementing specific prevention activities with one or more baseline scenarios in which no waste prevention activities are explicitly undertaken, and the same amount of waste is generated. In particular, the approaches were characterised in terms of their perspective on waste prevention activities, functional unit, system boundaries, and the resulting procedure for the calculation of the potential impacts. Both approaches provide the same results when the difference between the impacts of a waste prevention scenario and a baseline one is evaluated. As this difference represents the net impacts of the considered waste prevention activities (and of any possible change in the management of the remaining waste), both approaches can indifferently be used to evaluate the effects of implementing specific waste prevention activities on the overall environmental performance of a real or fictional waste management system. Nevertheless, due to the partially different upstream system boundaries, the results of single scenarios are different when calculated with one approach rather than the other, so that the interpretation of these results needs to be carried out differently. For the same reason, the application of the two approaches will be more suitable in studies with different specific purposes.

[1] PET: Polyethylene terephthalate; HDPE: high-density polyethylene.

By defining a proper functional unit and setting adequate system boundaries, it is thus possible to evaluate and compare the environmental and energy performance of integrated municipal waste management systems which include all the different types of waste prevention activities reviewed (Section 1). As waste prevention is now one of the core elements of waste management policy and practice in Europe and other countries, the availability of a unique LCA tool capable to contemporarily take into account all the management options of the waste hierarchy may prove to be very useful for waste managers and planners at any level (national, regional etc.). The presented approaches can be used for many purposes, such as, evaluating the consequences of implementing waste prevention activities in a system where waste is managed according to a given treatment scheme; complementing waste reduction indicators with LCA-based indicators; providing the basis for decoupling evaluations; and produce quantitative evidence of the potential benefits of waste prevention, to support its strategic and policy relevance at the European and national level.

6.1.4 Life cycle assessment of municipal waste prevention and management in Lombardia

As a final step, the consequences of introducing the two examined waste prevention activities in a real waste management system were analysed, by means of the discussed methodology (Section 5). Specifically, the municipal waste management system of the Lombardia Region, Italy, was selected for the assessment, as it has recently been the object of a recent LCA study. Moreover, Lombardia has set specific waste reduction targets for 2020, to be achieved through a set of waste prevention activities including those previously assessed (substitution of bottled water by public network water and substitution of single-use packaged liquid detergents by those withdrawn from self-dispensing systems by means of refillable containers).

A 2020 reference scenario was thus compared with different waste prevention scenarios, where the two activities are both separately and contemporarily implemented, by assuming a complete substitution of the traditional product(s). The substitution accounted for the actual levels of consumption of bottled water and of three categories of liquid detergents by size and type of packaging in Lombardia.

The results showed that, if the implemented substitution is actually beneficial, a modest reduction in waste generation (0.14 to 0.52%) allowed for disproportionate improvements in the overall performance of the waste management system (0.3 to 28% and 1.5 to 158%, respectively). Most of these improvements are due to the additional upstream benefits of the avoided production and use of the substituted products. These benefits always compensate for the additional upstream impacts from the production and use of the alternative product and for the moderate reductions in downstream benefits (due to lower recycling). The impacts of downstream components affected by waste prevention (i.e. collection, transport, sorting and recycling) are indeed only marginally altered, as the net quantity of waste removed is insignificant compared to the total waste.

Although this limited reduction in waste generation produced proportionally greater improvements in the overall performance of the waste management system, they were not always appreciable. For the bottled water substitution, half of the impact categories showed an improvement larger 10%, with a maximum of 148%. Four categories were improved between 5 and 10%, while the remaining three categories for less than 5%. Conversely, the improvements achieved by substituting single-use packaged liquid detergents were lower than 3% for nearly all impact categories. This product substitution is thus little effective when it is implemented as a stand-alone activity, but it can provide an additional contribution to the potential benefits of a structured set of prevention activities targeting packaging waste or also other relevant waste fractions. For instance, if it was implemented in combination with the bottled water substitution, the overall performance of the system would be improved by 6 to 25% for most impact categories.

6.2 Recommendations and future research

This research clearly demonstrated that a preventive evaluation of the environmental and energy convenience of municipal waste prevention activities, by means of life cycle thinking and assessment, is essential. This is particularly important when the activities are based on product or service substitution, reuse or lifespan extension. In this case, additional environmental and energy impacts are involved, which need to be carefully evaluated in comparison with avoided impacts, to determine the actual convenience of the activity. For product substitution, the research specifically revealed that the ultimate convenience often depends on the way the activity is actually implemented by citizens, institutions, producers or other possibly involved actors.

By means of life cycle thinking and assessment, it is thus possible to identify any critical point of an activity, possible improvement strategies and the way it can be best implemented by the involved actors to actually achieve the expected benefits. LCA allows to go beyond the simple reduction of the generated waste which, alone, does not automatically imply a reduction in the overall environmental and energy impacts.

The application of life cycle thinking and assessment as decision support tools is thus strongly recommended during the preparation of national or regional waste prevention programmes or, however, whenever a set of waste prevention measures or activities is to be selected for a given country, region or municipality. This will avoid the selection of measures/activities that could potentially increase the overall impacts, and facilitate the choice of those measures and activities providing the greatest environmental benefits.

This thesis focused on two particular waste prevention activities, and some studies are available in the literature for other activities (e.g. Cleary, 2013). However, there are further relevant examples that could be the object of future studies by the scientific community. For activities based on product substitution, the assessment can be carried out either at a product level or at the level of a specific geographical region (country, region or municipality). At the product level, the LCA can individually evaluate the effects of substituting the alternative less waste-generating product(s) for the different types of traditional products available (e.g. different types and sizes of packages for a product). It is thus possible to evaluate whether the examined substitution is beneficial with respect to all types of traditional products. At the regional level, the actual levels of consumption of the different types of potentially substituted products in a given region are taken into account. With this type of assessment, the net impacts resulting from the implementation of the considered waste prevention activity (substitution) in the examined region can be evaluated. Moreover, if estimates are available for the overall impacts (or emissions) generated in that region, it is possible to evaluate the contribution provided by the examined activity to the reduction of such overall impacts/emissions. On these bases, different waste prevention activities could also be compared in terms of their effectiveness in reducing the overall impacts or emissions in a given region. If feasible, both product level and regional level assessments should be carried out for a comprehensive picture. In this thesis, only product level assessments were performed for the examined activities, so that evaluations at the regional scale can be the object of further research.

The research has also revealed that, by performing appropriate methodological choices (functional unit and system boundaries), it is possible to evaluate the effects of waste prevention activities on the overall impacts of the integrated waste management system in which such activities are to be implemented. Other than product and regional scale evaluations, also evaluations at the level of waste management systems are encouraged, especially when a comparison of the performance of different activities is needed, during the drafting of waste management plans or programmes.

Although the methodological approaches presented in this thesis provided important general guidelines to incorporate waste prevention activities into LCAs of municipal waste management systems, further research and discussion on the modelling approaches applicable to specific situations may be useful to practitioners. For instance, it is generally acknowledged that, in the last instance, the waste prevented thanks to the reuse or lifespan extension of durable goods is represented by those equivalent new goods that would be used (and then wasted) if reuse or lifespan extension were not undertaken. However, in the short term, the goods actually reused (or subject to lifespan extension) are removed from the waste management system. A discussion on the implications associated with using one approach rather than the other and on the situations in which they can be more suitable may thus be worth.

Another situation that may need further clarification is the one in which the prevented waste include a certain recycled content. Which approaches are available to account for the avoided recycling of the secondary raw materials included in the waste and which are their consequences?

Finally, some problems may arise when the prevented waste is sent to incineration. How should one model the removal of a specific material from a multi-material incineration process? This may not be a problem if waste-specific burdens are taken into account in the modelling (e.g. by means of dedicated transfer coefficients provided by a particular waste LCA model). In this case, both waste-specific and process specific burdens would be reduced accordingly to the quantity of waste removed form the process. However, municipal waste incineration is also frequently modelled by considering all burdens as being process-specific (e.g. when the process carried out in a real plant of the examined region is modelled). Which solutions, if any, could be adopted in this situation?

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Appendix A

This appendix provides additional information on the life cycle assessment (LCA) study summarised in Section 2, which evaluates the environmental and energy convenience of the substitution of bottled water for public network water.

A.1 Features of the packaging systems

Table A.1 presents the most important features considered for the packaging system by which one-way bottled water is delivered to the consumer in baseline scenarios 1 to 3. For baseline scenarios 4 (glass refillable bottled water) and 5 (PET refillable bottled water), the same type of data is reported in Tables A.2 and A.3, respectively. The reported parameters were used as input data to define the quantities of the unit processes depicting the life cycle of primary, secondary and transport packages.

Table A.1: main features considered for the packaging system delivering one-way bottled water to the consumer in baseline scenarios 1 to 3.

Parameter	Value			Data source
Bottle volume [l]	2	1.5	0.5	-
Market share [%]	6.3	86.1	7.6	Bevitalia (2009)
Primary packages				
Bottle mass (PET, R-PET, PLA) [g]	33.42	32.55	18.06	Experimental estimates from Federambiente (2010)
Cap mass (HDPE) [g]	1.72	2.06	2.45	
Label mass (paper)[a] [g]	0.52	0.57	0.4	
Secondary packaging				
Bundle heat-shrink film mass (LDPE) [g]	26	21.8	10.5	Experimental estimates from Federambiente (2010)
Bottles per bundle [-]	6	6	6	Typical composition of bundles
Transport packages				
Wooden pallet mass [kg]	22	22	22	Bottling company located in northern Italy[b]
Cardboard interlayer mass [g]	600	600	600	
LLDPE stretch-film mass [g]	245	245	245	
LDPE top covering film mass [g]	175	175	175	
Pallet composition				
Layers per pallet [-]	4	4	7	Based on load practices of a bottling company located in northern Italy
Bundles per layer [-]	19	21	36	
Bundles per pallet [-]	76	84	252	
Bottles per pallet [-]	456	504	1512	
Water volume per pallet [l]	912	756	756	
Number of uses				
Pallets	20	20	20	Creazza and Dallari (2007)

(a) One-way bottles were assumed to be applied paper labels, which were used more frequently than plastic (LDPE or PP) ones, at the time of the analysis.
(b) The masses of transport packages are identical for all the three sizes of bottles.

Table A.2: main features considered for the packaging system delivering glass refillable bottled water to the consumer in baseline scenario 4.

Parameter	Value	Data source
Bottle volume [l]	1	Assumed to be representative of the domestic consumption
Market share [%]	100	
Primary packages		
Bottle mass (glass) [g]	475	Average mass of the bottles used by the major brands of glass bottled water retailed in Italy
Cap mass (Aluminium) [g]	1.75	Bottling company located in northern Italy
Label mass (Paper) [g]	1.06	Bottling company located in northern Italy
Transport packages		
Crate mass (HDPE) [kg]	2	Bottling company located in northern Italy
Bottles per crate [-]	12	
Wooden pallet mass [kg]	26	
Strapping band mass (LDPE) [g]	21	
Pallet composition		
Layers per pallet [-]	5	Based on load practices of a bottling company located in northern Italy
Crates per layer [-]	9	
Crates per pallet [-]	45	
Bottles per pallet [-]	540	
Water volume per pallet [l]	540	
Number of uses		
Bottles[a]	10	Some Italian bottling companies
Crates	100	Bottling company located in northern Italy
Pallets	20	Creazza and Dallari (2007)

(a) The reported number of uses refers to the base case of the scenario, but a sensitivity analysis has been performed on such a parameter, as described in Section 2.8.

Table A.3: main features considered for the packaging system delivering PET refillable bottled water to the consumer in baseline scenario 5[a].

Parameter	Value	Data source
Bottle volume [l]	1	Assumed to be representative of the domestic consumption
Market share [%]	100	
Primary packages		
Bottle mass [g]	62	
Cap mass (HDPE) [g]	3.2	IFEU (2008; 2010)
Label mass (PP) [g]	0.6	
Transport packages		
Crate mass (HDPE) [kg]	1.85	
Bottles per crate [-]	12	IFEU (2008; 2010)
Wooden pallet mass [kg]	22	
Strapping band mass (LDPE) [g]	18	
Pallet composition		
Layers per pallet [-]	5	
Crates per layer [-]	8	
Crates per pallet [-]	40	IFEU (2008; 2010)
Bottles per pallet [-]	480	
Water volume per pallet [l]	480	
Number of uses		
Bottles[b]	15	IFEU (2008; 2010)
Crates	100	IFEU (2008; 2010)
Pallets	20	Creazza and Dallari (2007)

(a) For this scenario, the features of the packaging system were mainly defined based on literature data concerning the German market, because refillable PET bottles are not used in Italy.
(b) The reported number of uses refers to the base case of the scenario, but a sensitivity analysis has been performed on such a parameter, as described in Section 2.8.

A.2 Selected unit processes and respective quantities

Tables A.4 to A.11 list, for each scenario, the major unit processes included, the quantity required of each process and the source of inventory data respectively considered. For most processes, inventory datasets available in commercial databases were used. However, a number of datasets were developed on purpose by the author.

Table A.4: major processes included in the baseline scenario 1, quantities required of these processes and respective sources of inventory data considered.

Processes / inputs & outputs	Amount per functional unit		Data source for the modelling
Life cycle of primary packages			
Manufacturing of PET preforms[a]			
Production of virgin bottle grade PET granules Injection moulding of preforms from granules	kg	3.44	ecoinvent
Manufacturing of HDPE caps[a]			
Production of virgin HDPE granules Injection moulding of caps from granules	kg	0.246	ecoinvent
Manufacturing of paper labels			
Production of wood-containing mechanical paper	kg	0.0615	ecoinvent
End of life			
Sorting & recycling of bottles (regranulation)	kg	2.63	Rigamonti & Grosso (2009)
Production of virgin PET granules (avoided)	kg	-1.71	ecoinvent
Incineration of bottles	kg	0.790	Dataset developed on purpose[b]
Incineration of caps	kg	0.245	
Incineration of labels	kg	0.0615	
Life cycle of secondary packages			
Manufacturing of LDPE heat shrink film for bundles[a]			
Production of virgin LDPE granules Extrusion of the film from granules	kg	0.387	ecoinvent
Manufacturing of PP adhesive tape for the handle of the bundles[a]			
Production of virgin PP granules Extrusion of the tape from granules	g	8.37	ecoinvent
Manufacturing of cardboard strips for the bundles			
Production of cardboard (white lined chipboard)	g	22.2	ecoinvent
End of life			
Sorting & recycling of the heat-shrink film (production of profiled bars)	kg	0.125	Rigamonti & Grosso (2009)
Production of wooden planks (avoided)	kg	-0.0749	ecoinvent
Incineration of the heat-shrink film	kg	0.253	Dataset developed on purpose[b]
Incineration of the adhesive tape of the handles	g	8.21	
Incineration of the cardboard strips of the handles	g	22.2	

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Table A.4 (continued)

Life cycle of transport packages			
Production of 80×120 cm EUR-EPAL pallets (only materials: wood & nails)	units	$9.95{\times}10^{-3}$	ecoinvent
Manufacturing of cardboard interlayers			
Production of cardboard (white lined chipboard)	kg	0.386	ecoinvent
Manufacturing of LLDPE stretch film[a]			
Production of virgin LLDPE granules Extrusion of the film from granules	kg	0.050	ecoinvent
Manufacturing of LDPE top covering film[a]			
Production of virgin LDPE granules Extrusion of the film from granules	kg	0.0357	ecoinvent
End of life			
Recycling of pallets (particle board production)	kg	0.245	Rigamonti & Grosso (2009)
Plywood board production (avoided)	m^3	$-1.30{\times}10^{-4}$	ecoinvent
Recycling of the steel nails of pallets (re-melting in electric arc furnaces)	kg	$1.95{\times}10^{-3}$	Rigamonti & Grosso (2009)
Primary steel production in basic oxygen furnaces (avoided)	kg	$-1.76{\times}10^{-3}$	ecoinvent[c]
Recycling of cardboard interlayers (secondary pulp production)	kg	0.386	Rigamonti & Grosso (2009)
Thermo-mechanical pulp production (avoided)	kg	-0.286	ecoinvent[c]
Recycling of stretch and top covering film (production of profiled bars)	kg	0.0837	Rigamonti & Grosso (2009)
Production of wooden planks (avoided)	kg	-0.0502	ecoinvent
Bottling plant operations			
Electricity for bottling plant operations (including stretch blow moulding of preforms) -Italian production mix-	kWh	3.15	ecoinvent
Lubricating oil production (for the maintenance of machineries)	kg	$2.37{\times}10^{-4}$	ecoinvent
Lubricating oil incineration	kg	$2.37{\times}10^{-4}$	ecoinvent[d]
Washing of the filler machine			
Water, unspecified origin (natural resource)	litres	5.90	-
Alkaline detergent (daily washing)	kg	$5.70{\times}10^{-4}$	ecoinvent
Acid detergent (daily washing)	kg	$3.80{\times}10^{-4}$	ecoinvent
Foaming disinfectant (daily washing)	kg	$2.86{\times}10^{-4}$	ecoinvent
Caustic detergent (weekly washing)	kg	$2.86{\times}10^{-4}$	ecoinvent
Non-foaming disinfectant (weekly washing)	kg	$9.96{\times}10^{-4}$	ecoinvent
COD waterborne emissions	g	$1.54{\times}10^{-1}$	-
Nitrogen (N) waterborne emissions	g	$2.48{\times}10^{-3}$	-
Phosphorus (P) waterborne emissions	g	$3.36{\times}10^{-2}$	-
Treatment of washing waters (unpolluted sewage) in wastewater treatment plants	litres	5.90	ecoinvent
Transports			
Transport of palletised water from bottling plants to retailers for 300 km (and return trip with empty pallets) by lorry > 16 t (European fleet average)	t×km	50	ecoinvent
Water purchasing roundtrip (retailers-consumers' houses) by car (10 km)	km	5.63	ecoinvent

(a) The transport of granules for 100 km by lorry and 200 km by rail from the manufacturing to the conversion plant was also considered.
(b) The dataset was defined based on the process carried out in an existing waste to energy plant located in northern Italy, but taking into account waste-specific burdens, so as better detailed in Section 2.7.1.1.
(c) The *ecoinvent* dataset was modified according to the adjustments reported by Rigamonti and Grosso (2009).
(d) The avoided production of electricity and heat was also included in addition in the selected *ecoinvent* dataset.

Table A.5: major processes included in the baseline scenario 2 which differ from baseline scenario 1, quantities required of these processes and respective sources of inventory data considered.

Processes / inputs & outputs	Amount per functional unit		Data source for the modelling
Life cycle of primary packages			
Manufacturing of PET preforms[a]			
Production of virgin, bottle grade, PET granules	kg	1.72	ecoinvent
Sorting and recycling of post consumer bottles (regranulation)	kg	2.15	Rigamonti & Grosso (2009)
Solid state poly-condensation (SSP) of recycled PET granules	kg	1.72	Dataset developed on purpose[b]
Injection moulding of preforms from granules	kg	3.44	ecoinvent
End of life			
Sorting and recycling of post consumer bottles (regranulation)	kg	0.48	Rigamonti & Grosso (2009)
Production of virgin PET granules (avoided)	kg	-0.31	ecoinvent
Incineration of bottles	kg	0.79	Dataset developed on purpose[c]

(a) The transport of granules for 100 km by lorry and 200 km by rail from the manufacturing to the conversion plant was also considered.
(b) The dataset was defined based on the data reported in Starlinger (2010).
(c) The dataset was defined based on the process carried out in an existing waste to energy plant located in northern Italy, but taking into account waste-specific burdens, so as better detailed in Section 2.7.1.1.

Table A.6: major processes included in baseline scenarios 3a and 3b which differ from baseline scenario 1, quantities required of these processes and respective sources of inventory data considered.

Processes / inputs & outputs	Amount per functional unit		Data source for the modelling
Life cycle of primary packages			
Manufacturing of PLA preforms[a]			
Production of virgin PLA granules Injection moulding of preforms from granules	kg	3.44	ecoinvent
End of life			
Composting (or incineration) of PLA bottles	kg	3.42	Dataset developed on purpose[b]

(a) The transport of granules for 100 km by lorry and 200 km by rail from the manufacturing to the conversion plant was also considered.
(b) For industrial composting, a dataset was defined based on the process carried out in an existing composting plant located in northern Italy, and by taking into account PLA-specific burdens, so as better detailed in Section 2.7.1.1. For incineration, the development of the dataset was based on the process carried out in an existing waste to energy plant located in northern Italy, but taking into account waste-specific burdens, so as better detailed in Section 2.7.1.1.

Table A.7: major processes included in the baseline scenario 4, quantities required of these processes and respective sources of inventory data considered.

Processes / inputs & outputs	Amount per functional unit		Data source for the modelling
Life cycle of primary packages			
Manufacturing of glass bottles			
Production of white glass bottles (sorting and melting of cullet with 39.5% of virgin raw materials)	kg	3.61	ecoinvent
Production of green glass bottles (sorting and melting of cullet with 16.5% of virgin raw materials)	kg	3.61	ecoinvent
Manufacturing of aluminium caps			
production of aluminium ingots (European mix) Production of aluminium sheets from ingots (hot and cold rolling) Moulding of caps from sheets (approximated with the process of cold impact extrusion)	kg	0.266	ecoinvent
Manufacturing of paper labels			
Production of wood-containing mechanical paper	kg	0.161	ecoinvent
End of life			
Recycling of bottles (sorting and re-melting with 16.5% of virgin raw materials)	kg	2.02	Rigamonti & Grosso (2009)
Production of generic virgin glass containers (avoided)	kg	-2.42	ecoinvent[a]
Recycling of caps (re-melting of aluminium scraps into ingots)	kg	0.237	Rigamonti & Grosso (2009)
Production of aluminium ingots from virgin raw materials (avoided)	kg	-0.198	ecoinvent[a]
Incineration of labels	kg	0.161	Dataset developed on purpose[b]
Life cycle of transport packages			
Manufacturing of HDPE crates[c]			
Production of virgin HDPE granules Injection moulding of crates from granules	kg	0.256	ecoinvent
Production of 95×120 cm pallets (only materials: wood & nails)	units	0.0141	ecoinvent
Manufacturing of LDPE strapping band[c]			
Production of virgin LDPE granules Extrusion of the strapping band from granules	g	6.08	ecoinvent
End of life			
Recycling of crates (regranulation)	kg	0.254	Rigamonti & Grosso (2009)
Production of virgin HDPE granules (avoided)	kg	-0.185	ecoinvent
Recycling of pallets (particle board production)	kg	0.411	Rigamonti & Grosso (2009)
Plywood board production (avoided)	m^3	-2.18×10^{-4}	ecoinvent
Recycling of the steel nails of pallets (re-melting in electric arc furnaces)	kg	3.25×10^{-3}	Rigamonti & Grosso (2009)
Primary steel production in basic oxygen furnaces (avoided)	kg	-2.94×10^{-3}	ecoinvent[a]
Incineration of the strapping band	g	5.93	Dataset developed on purpose[b]

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Table A.7 (continued)

Bottling plant operations			
Electricity for bottling plant operations (including bottle washing) - Italian production mix-	kWh	2.04	ecoinvent
Lubricating oil production (for the maintenance of machineries)	kg	2.37×10^{-4}	ecoinvent
Lubricating oil incineration	kg	2.37×10^{-4}	ecoinvent[d]
Washing of the filler machine			
modelled as in one-way bottled water scenarios (Table A.4)			
Bottle washing			
Water, unspecified origin (natural resource)	litres	152.1	ecoinvent
Natural gas (burned in industrial furnace)	MJ	33.9	ecoinvent
Caustic soda (NaOH), pure substance	kg	0.121	ecoinvent
Descaling agent	kg	0.0380	ecoinvent
Defoaming agent	kg	0.0333	ecoinvent
Sequestering agent	kg	0.0143	ecoinvent
Non-foaming disinfectant	kg	8.08×10^{-3}	ecoinvent
COD waterborne emissions	g	24.8	-
Nitrogen (N) waterborne emissions	g	0.914	-
Phosphorus (P) waterborne emissions	g	0.208	-
Treatment of washing waters (unpolluted sewage) in wastewater treatment plants	litres	152.1	ecoinvent
Transports			
Transport of palletised water from bottling plants to local distributors for 300 km (and return trip with empty palletised bottles) by lorry > 16 t (European fleet average)	t×km	109.5	ecoinvent
Transport of crates with filled bottles from local distributors to the households for 20 km (and return trip with empty bottles) by lorry 3.5-16 t (European fleet average)	t×km	6.94	ecoinvent

(a) The *ecoinvent* dataset was modified according to the adjustments reported by Rigamonti and Grosso (2009).
(b) The dataset was defined based on the process carried out in an existing waste to energy plant located in northern Italy, but taking into account waste-specific burdens, so as better detailed in Section 2.7.1.1.
(c) The transport of granules for 100 km by lorry and 200 km by rail from the manufacturing to the conversion plant was also considered.
(d) The avoided production of electricity and heat was also included in addition in the selected *ecoinvent* dataset.

Table A.8: major processes included in the baseline scenario 5, quantities required of these processes and respective sources of inventory data considered.

Processes / inputs & outputs	Amount per functional unit		Data source for the modelling
Life cycle of primary packages			
Manufacturing of PET bottles[a]			
Production of virgin, bottle grade, PET granules	kg	0.629	ecoinvent
Injection stretch blow moulding of bottles from granules	kg	0.629	ecoinvent[b]
Manufacturing of HDPE caps[a]			
Production of virgin HDPE granules Injection moulding of caps from granules	kg	0.490	ecoinvent
Manufacturing of PP labels[a]			
Production of virgin PP granules Extrusion of labels (film) from granules	kg	0.0935	ecoinvent
End of life			
Sorting & recycling of PET bottles (regranulation)	kg	0.629	Rigamonti & Grosso (2009)
Production of virgin PET granules (avoided)	kg	-0.407	ecoinvent
Recycling of HDPE caps (regranulation)	kg	0.487	Rigamonti & Grosso (2009)
Production of virgin HDPE granules (avoided)	kg	-0.355	ecoinvent
Incineration of labels	kg	0.091	Dataset developed on purpose[c]
Life cycle of transport packages			
Manufacturing of HDPE crates[a]			
Production of virgin HDPE granules Injection moulding of crates from granules	kg	0.236	ecoinvent
Production of 80×120 cm EUR-EPAL pallets (only materials: wood & nails)	units	0.0158	ecoinvent
Manufacturing of LDPE strapping band[a]			
Production of virgin LDPE granules Extrusion of the strapping band from granules	g	5.84	ecoinvent
End of life			
Recycling of crates (regranulation)	kg	0.234	Rigamonti & Grosso (2009)
Production of virgin HDPE granules (avoided)	kg	-0.170	ecoinvent
Recycling of pallets (particle board production)	kg	0.389	Rigamonti & Grosso (2009)
Plywood board production (avoided)	m^3	-2.06×10^{-4}	ecoinvent
Recycling of the steel nails of pallets (remelting in electric arc furnaces)	kg	3.09×10^{-3}	Rigamonti & Grosso (2009)
Primary steel production in basic oxygen furnace (avoided)	kg	-2.80×10^{-3}	ecoinvent[d]
Incineration of the strapping band	g	5.70	Dataset developed on purpose[c]
Bottling plant operations			
modelled as in the glass refillable bottled water scenario (Table A.7) except for natural gas consumption of bottle washing:			
Natural gas (burned in industrial furnace)	MJ	13.5	ecoinvent

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Table A.8 (continued)

Transports			
Transport of palletised water from bottling plants to local distributors for 300 km (and return trip with empty palletised bottles) by lorry > 16 t (European fleet average)	t×km	70.3	ecoinvent
Transport of crates with filled bottles from local distributors to the households for 20 km (and return trip with empty bottles) by lorry 3.5-16 t (European fleet average)	t×km	4.4	ecoinvent

(a) The transport of granules for 100 km by lorry and 200 km by rail from the manufacturing to the conversion plant was also considered.
(b) The *ecoinvent* dataset *Stretch blow moulding/RER* has been updated on purpose with the data reported in the latest eco-profile by Plastics*Europe* for this type of process (TNO, 2010).
(c) The dataset was defined based on the process carried out in an existing waste to energy plant located in northern Italy, but taking into account waste-specific burdens, so as better detailed in Section 2.7.1.1.
(d) The *ecoinvent* dataset was modified according to the adjustments reported by Rigamonti and Grosso (2009).

Table A.9: major processes included in the waste prevention scenario 1, quantities required of these processes and respective sources of inventory data considered.

Processes / inputs & outputs	Amount per functional unit		Data source for the modelling
Water withdrawal, purification and delivery			
Groundwater (natural resource)	litres	170.4	-
Electricity (Italian production mix)	kWh	0.074	ecoinvent
Production of virgin activated carbon (modelled as carbon coke)	kg	9.31×10^{-5}	I-LCA (ANPA, 2000)
Reactivation of exhausted activated carbon	kg	1.86×10^{-3}	Dataset developed on purpose[a]
Production of sodium hypochlorite (NaClO - pure substance)	kg	1.57×10^{-5}	ecoinvent
Life cycle of the main components of the water supply network			
Manufacturing of carbon steel hot rolled sheets	kg	2.68×10^{-4}	ecoinvent
Drawing of seamless pipes from hot rolled steel sheets	kg	2.68×10^{-4}	ecoinvent
Manufacturing of cast iron ingots	kg	1.81×10^{-3}	ecoinvent
Hot rolling of sheets from cast iron ingots	kg	1.81×10^{-3}	ecoinvent
Drawing of seamless pipes from hot rolled cast iron sheets (approximation of the real process)	kg	1.81×10^{-3}	ecoinvent
Production of cement mortar for the coating of cast iron pipes (internal surface)	kg	4.55×10^{-5}	ecoinvent
Production of zinc for the coating of cast iron pipes (external surface)	kg	1.25×10^{-5}	ecoinvent
Production of virgin HDPE granules Extrusion of pipes from HDPE granules	kg	3.67×10^{-6}	ecoinvent
Recycling of steel and cast iron pipes (remelting in electric arc furnaces)	kg	2.08×10^{-3}	Rigamonti & Grosso (2009)
Primary steel production in basic oxygen furnaces (avoided)	kg	-1.88×10^{-3}	ecoinvent[b]
Recycling of HDPE pipes (regranulation)	kg	3.65×10^{-6}	Rigamonti & Grosso (2009)
Production of virgin HDPE granules (avoided)	kg	-2.66×10^{-6}	ecoinvent
Excavation with hydraulic diggers for laying of pipes (from ecoinvent)	m^3	2.08×10^{-4}	ecoinvent
Building machine life cycle (for laying of pipes, from ecoinvent, as MJ of consumed diesel)	MJ	1.12×10^{-3}	ecoinvent

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Table A.9 (continued)

Life cycle of the materials of GAC filters and aeration towers			
Manufacturing of stainless steel hot rolled sheets	kg	9.44×10^{-6}	ecoinvent
Manufacturing of carbon steel hot rolled sheets	kg	7.82×10^{-6}	ecoinvent
Cold rolling of steel sheets for the manufacturing of filters and towers	kg	1.72×10^{-5}	ecoinvent
Production of virgin PP granules Injection moulding of towers from granules (approximation of the real process)	kg	4.27×10^{-8}	ecoinvent
Recycling of steel filters and towers (remelting in electric arc furnaces)	kg	1.72×10^{-5}	Rigamonti & Grosso (2009)
Primary steel production in basic oxygen furnaces (avoided)	kg	-1.47×10^{-5}	ecoinvent[b]
Recycling of PP towers (regranulation)	kg	4.24×10^{-8}	BUWAL 250
Production of virgin PP granules (avoided)	kg	-3.82×10^{-8}	BUWAL 250
Pumping stations: life cycle of the materials of pumps and other infrastructures (from ecoinvent)	units	3.03×10^{-10}	ecoinvent
Water reservoirs: life cycle of the materials (from ecoinvent)	units	8.09×10^{-10}	ecoinvent
Domestic water quality improvement			
Purified groundwater from the tap (module described above in this table)	litres	456.3	This study
Electricity (Italian production mix)	kWh	0.400	ecoinvent
Production of activated carbon for the filter (modelled as carbon coke)	kg	0.50	I-LCA (ANPA, 2000)
Disposal of activated carbon into an inert material landfill	kg	0.50	ecoinvent
Treatment of rejected water (unpolluted sewage) in wastewater treatment plants	litres	304.2	ecoinvent
Life cycle of glass jugs			
Production of generic white glass containers (sorting and melting of cullet with 39.5% of virgin raw materials)	g	474.6	ecoinvent
Recycling (sorting and remelting with 16.5% of virgin raw materials), only the amount not employed for the production of the jugs	g	187.1	Rigamonti & Grosso (2009)
Production of generic virgin glass containers (avoided)	g	-224.1	ecoinvent[b]
Dishwashing of jugs			
Electricity (Italian production mix)	kWh	1.7	ecoinvent
Purified tap water	litres	19.9	ecoinvent
Treatment of washing waters (unpolluted sewage) in wastewater treatment plants	litres	19.9	ecoinvent

(a) The dataset was developed based on the data reported in the environmental declaration of a real Italian company (SICAV S.r.l.), which deals with the reactivation of exhausted activated carbons (SICAV, 2009).

(b) The *ecoinvent* dataset was modified according to the adjustments reported in Rigamonti and Grosso (2009).

Table A.10: major processes included in the waste prevention scenario 2, quantities required of these processes and respective sources of inventory data considered.

Processes / inputs & outputs	Amount per functional unit		Data source for the modelling
Water withdrawal, purification and delivery			
River water (natural resource)	litres	179.5	-
Electricity (Italian production mix)	kWh	0.058	ecoinvent
Production of hydrochloric acid (HCl) - pure substance	kg	9.63×10^{-4}	ecoinvent
Production of sodium chlorite ($NaClO_2$) - 25% m/m solution	kg	2.56×10^{-3}	Dataset developed on purpose[a]
Production of poly-aluminium chloride (PACl) - 10% as Al_2O_3 m/m sol	kg	1.10×10^{-2}	Dataset developed on purpose[b]
Production of sodium hypochlorite (NaClO) - pure substance	kg	2.22×10^{-4}	ecoinvent
Production of acrylonitrile (modelling PWG[c] polyelectrolyte)	kg	4.90×10^{-5}	ecoinvent
Production of quartziferous sand	kg	1.79×10^{-3}	ecoinvent
Production of virgin activated carbon (modelled as carbon coke)	kg	1.96×10^{-4}	I-LCA (ANPA, 2000)
Reactivation of exhausted activated carbon	kg	1.64×10^{-3}	Dataset developed on purpose[d]
Disposal of sludge into an inert material landfill	kg	2.65×10^{-3}	ecoinvent
Life cycle of the main components of the water supply network (as groundwater, Table A.9, for coherence)			
Life cycle of the materials of GAC filters and aeration towers (as groundwater, Table A.9, for coherence)			
Pumping stations: life cycle of the materials of pumps and other infrastructures (from ecoinvent, as groundwater, Table A.9)	units	3.03×10^{-10}	ecoinvent
Water reservoirs: life cycle of the materials (from ecoinvent, as groundwater, Table A.9)	units	8.09×10^{-10}	ecoinvent
Public water quality improvement			
Purified surface water from the network (module described above in this table)	litres	164.3	This study
Electricity (Italian production mix)	kWh	1.5	ecoinvent
Production of virgin PP granules for the production of pre-filters Extrusion of pre-filters from PP granules	kg	2.34×10^{-3}	ecoinvent
Production of activated carbon for the filters (modelled as carbon coke)	kg	6.34×10^{-3}	I-LCA (ANPA, 2000)
Incineration of PP pre-filters	kg	2.28×10^{-3}	Dataset developed on purpose[e]
Disposal of activated carbon into an inert material landfill	kg	6.34×10^{-3}	ecoinvent
Treatment of rejected water (unpolluted sewage) in wastewater treatment plants	litres	12.2	ecoinvent
Life cycle of glass bottles			
Production of a generic green glass container (sorting and melting of cullet with 16.5% of virgin raw materials)	kg	2.14	ecoinvent
Production of a generic white glass container (sorting and melting of cullet with 39.5% of virgin raw materials)	kg	2.14	ecoinvent
Bottle recycling (sorting and re-melting with 16.5% of virgin raw materials), only the amount not employed for bottle production	kg	1.20	Rigamonti & Grosso (2009)
Production of generic virgin glass containers (avoided)	kg	-1.44	ecoinvent[f]

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Table A.10 (continued)

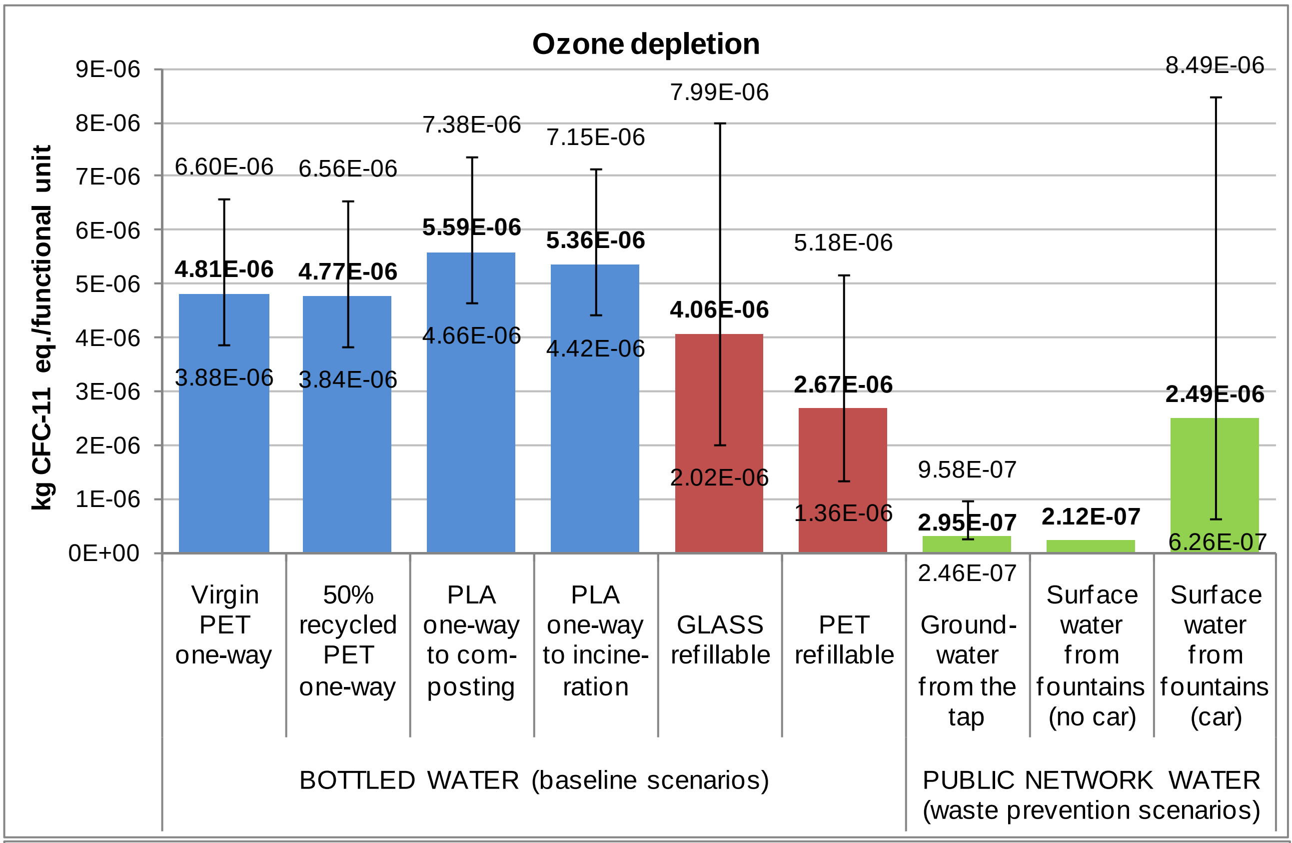
Water transport			
Roundtrip transport of empty/filled bottles from public fountains to the households by car for 5.5 km	km	92.9	ecoinvent

(a) The dataset was defined based on the reaction stechiometry and the information reported in ATSDR (2004), IARC (1991), Kaczur and Cawlfield (2000), Madduri (2007) and Vogt et al. (2000).
(b) The dataset was defined based on the data and the information gathered from a real producer of poly-aluminium chloride, Consito (2010) and Solvay Solexis (2005).
(c) PWG: Potable Water Grade.
(d) The dataset was defined based on the data reported in the environmental declaration of a real Italian company (SICAV S.r.l.), which deals with the reactivation of exhausted activated carbons (SICAV, 2009).
(e) The dataset was defined based on the process carried out in an existing waste to energy plant located in northern Italy, but taking into account waste-specific burdens, so as better detailed in Section 2.7.1.1.
(f) The *ecoinvent* dataset was modified according to the adjustments reported by Rigamonti and Grosso (2009).

A.3 Further results

This section completes the framework of results of the impact assessment phase (Figures A.1 to A.6) and presents additional results in terms of impact difference between scenarios. In particular, Tables A.11 to A.14 compare waste prevention with baseline scenarios, while Tables A.15 to A.17, compare refillable bottled water scenarios with those based on one-way bottled water.

A.3.1 Impact assessment results



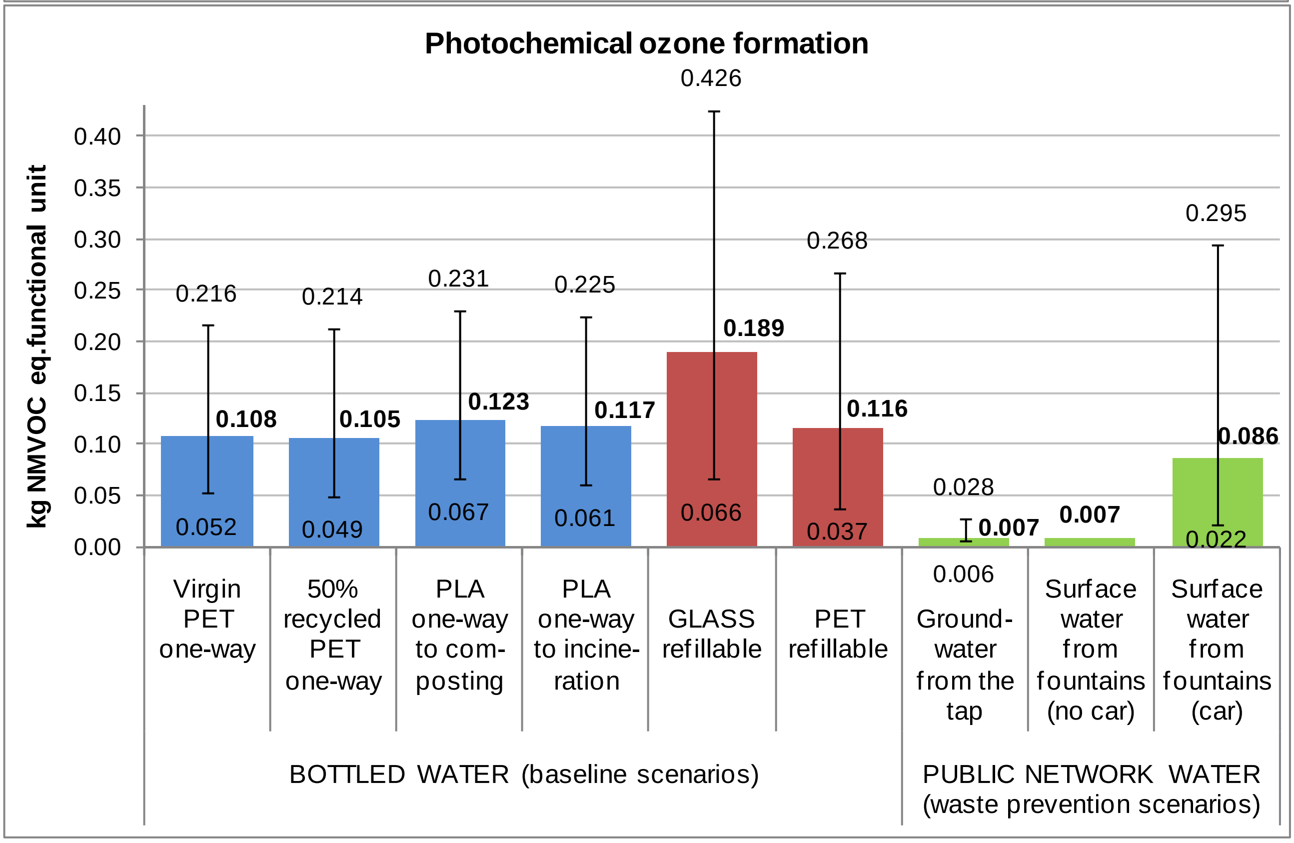


Figure A.1: potential impacts of the analysed baseline and waste prevention scenarios, for the *ozone depletion* and *photochemical ozone formation* impact categories. For each scenario, the main bar represents the base case, while the error bar shows the upper and lower boundaries resulting from the variation of the sensitivity parameters described in Table 2.4.

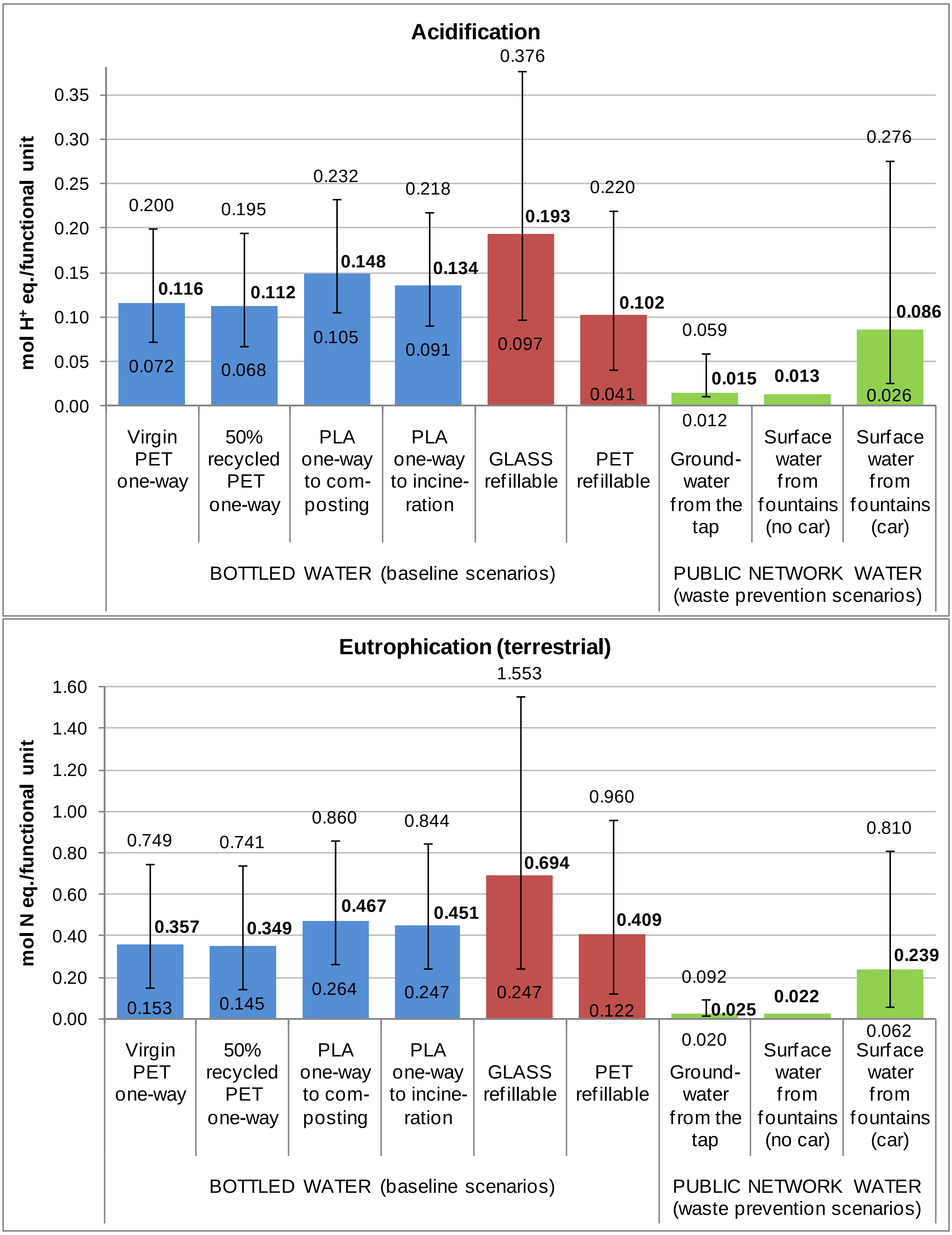


Figure A.2: potential impacts of the analysed baseline and waste prevention scenarios, for the *acidification* and *terrestrial eutrophication* impact categories. For each scenario, the main bar represents the base case, while the error bar shows the upper and lower boundaries resulting from the variation of the sensitivity parameters described in Table 2.4.

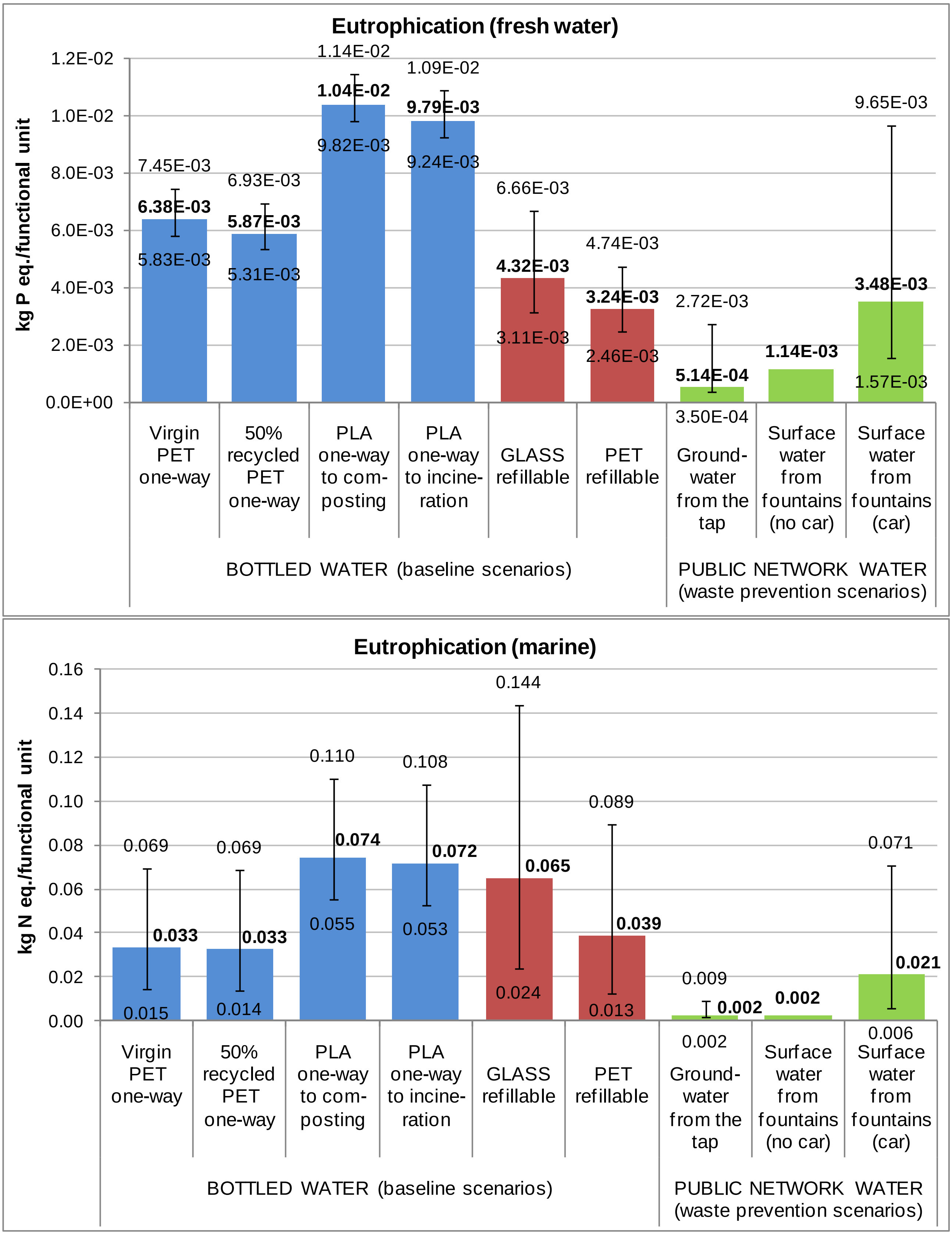


Figure A.3: potential impacts of the analysed baseline and waste prevention scenarios, for the *freshwater* and *marine eutrophication* impact categories. For each scenario, the main bar represents the base case, while the error bar shows the upper and lower boundaries resulting from the variation of the sensitivity parameters described in Table 2.4.

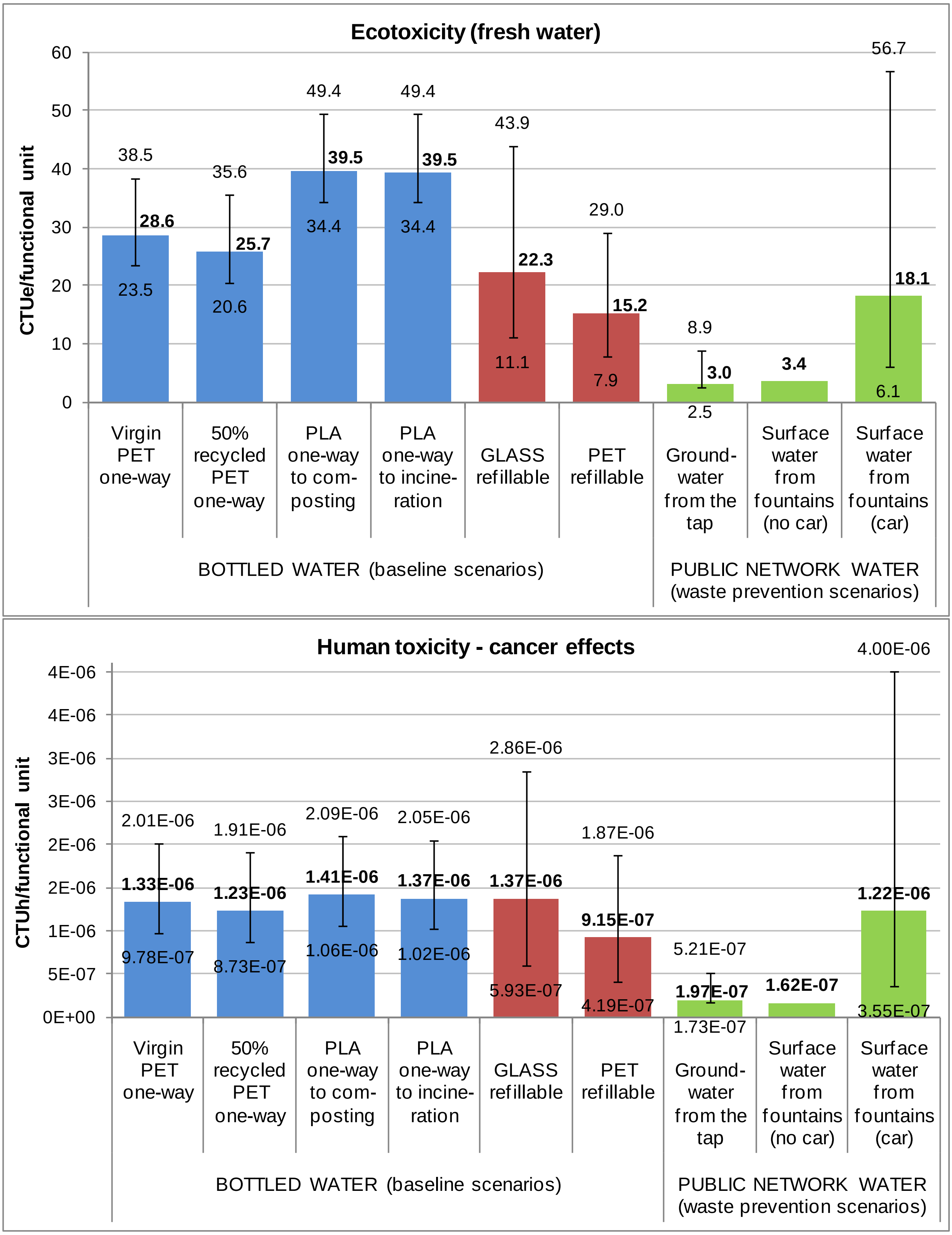


Figure A.4: potential impacts of the analysed baseline and waste prevention scenarios, for the *freshwater ecotoxicity* and *human toxicity (cancer effects)* impact categories. For each scenario, the main bar represents the base case, while the error bar shows the upper and lower boundaries resulting from the variation of the sensitivity parameters described in Table 2.4.

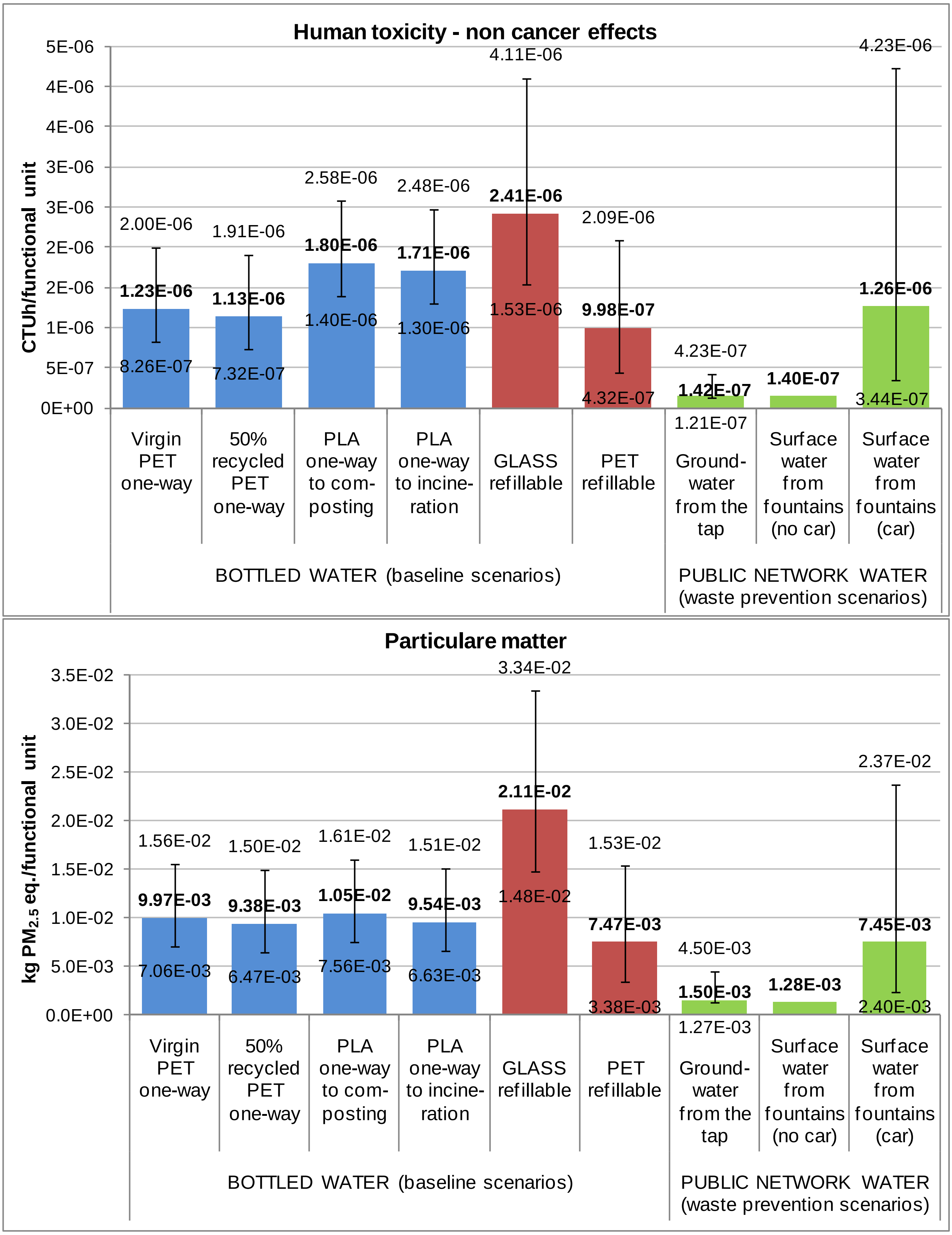


Figure A.5: potential impacts of the analysed baseline and waste prevention scenarios, for the *human toxicity (non-cancer effects)* and *particulate matter* impact categories. For each scenario, the main bar represents the base case, while the error bar shows the upper and lower boundaries resulting from the variation of the sensitivity parameters described in Table 2.4.

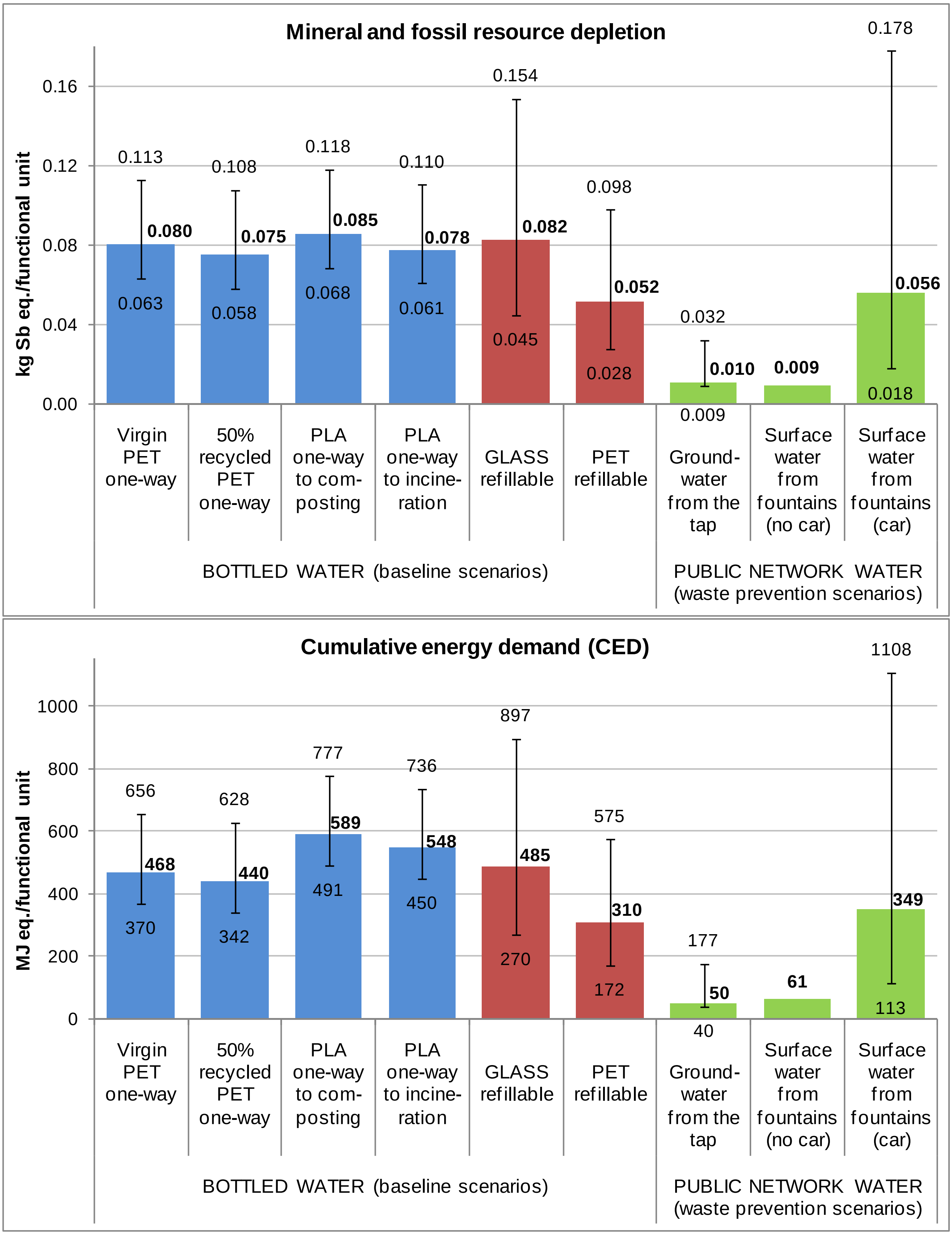


Figure A.6: potential impacts of the analysed baseline and waste prevention scenarios, for the *mineral and fossil resource depletion* and *cumulative energy demand* impact categories. For each scenario, the main bar represents the base case, while the error bar shows the upper and lower boundaries resulting from the variation of the sensitivity parameters described in Table 2.4.

A.3.2 Scenario comparison (impact variations)

Table A.11: impact reductions resulting from the substitution of refined groundwater from the tap (waste prevention scenario 1) for the different types of bottled water, when the reusable jug used to withdraw tap water is washed under average conditions (after every 4 uses in a load of 30 items) and bottles are transported to retailers or local distributors along a distance of 800 km (worst case of baseline scenarios).

Impact categories	Reference baseline scenario					
	Virgin PET one-way bottles	**50% recycled PET one-way bottles**	**PLA one-way bottles to composting**	**PLA one-way bottles to incineration**	**Refillable glass bottles[a]**	**Refillable PET bottles[a]**
Climate change	-94.3%	-94.1%	-94.7%	-94.3%	-95.9%	-93.6%
Ozone depletion	-95.5%	-95.5%	-96.0%	-95.9%	-96.3%	-94.3%
Photochemical ozone formation	-96.5%	-96.5%	-96.8%	-96.7%	-98.2%	-97.2%
Acidification	-92.5%	-92.3%	-93.5%	-93.1%	-96.0%	-93.2%
Terrestrial eutrophication	-96.7%	-96.7%	-97.1%	-97.1%	-98.4%	-97.4%
Freshwater eutrophication	-93.1%	-92.6%	-95.5%	-95.3%	-92.3%	-89.2%
Marine eutrophication	-96.6%	-96.6%	-97.9%	-97.8%	-98.4%	-97.4%
Freshwater ecotoxicity	-92.3%	-91.7%	-94.0%	-94.0%	-93.3%	-89.8%
Human toxicity (cancer effects)	-90.2%	-89.7%	-90.6%	-90.4%	-93.1%	-89.5%
Human toxicity (non-cancer effects)	-92.9%	-92.5%	-94.5%	-94.3%	-96.5%	-93.2%
Particulate matter	-90.4%	-90.0%	-90.7%	-90.1%	-95.5%	-90.2%
Water resource depletion	-23.0%	-19.2%	-25.3%	-14.8%	-29.4%	-13.1%
Mineral & fossil resource depletion	-90.7%	-90.3%	-91.1%	-90.5%	-93.2%	-89.3%
Cumulative energy demand	-92.4%	-92.1%	-93.6%	-93.2%	-94.4%	-91.3%
Minimum reduction[b]	*-23.0%*	*-19.2%*	*-25.3%*	*-14.8%*	*-29.4%*	*-13.1%*
Maximum reduction[b]	*-96.7%*	*-96.7%*	*-97.9%*	*-97.8%*	*-98.4%*	*-97.4%*

(a) Refillable glass bottles are used 10 times, while refillable PET bottles for 15 times.
(b) The *water resource depletion* indicator is also included in the calculation of the minimum and maximum reductions, as a significant reduction (>10%) is achieved, compared to all bottled water scenarios, also for this indicator. Therefore, these values cannot be directly compared with those reported in Table 2.6.

Table A.12: impact reductions resulting from the substitution of refined groundwater from the tap (waste prevention scenario 1) for the different types of bottled water, when the reusable jug used to withdraw tap water is washed under worsened conditions (after every use in a load of 15 items) and bottles are transported to retailers or local distributors along a distance of 800 km (worst case of baseline scenarios).

Impact categories	Reference baseline scenario					
	Virgin PET one-way bottles	**50% recycled PET one-way bottles**	**PLA one-way bottles to composting**	**PLA one-way bottles to incineration**	**Refillable glass bottles[a]**	**Refillable PET bottles[a]**
Climate change	-73.3%	-72.6%	-73.5%	-75.1%	-81.0%	-70.0%
Ozone depletion	-85.5%	-85.4%	-86.6%	-87.0%	-88.0%	-81.5%
Photochemical ozone formation	-87.3%	-87.1%	-87.8%	-88.1%	-93.5%	-89.7%
Acidification	-70.3%	-69.6%	-72.8%	-74.4%	-84.2%	-73.0%
Terrestrial eutrophication	-87.7%	-87.5%	-89.0%	-89.3%	-94.1%	-90.4%
Freshwater eutrophication	-63.5%	-60.8%	-75.0%	-76.3%	-59.2%	-42.7%
Marine eutrophication	-87.1%	-87.0%	-91.7%	-91.9%	-93.8%	-90.0%
Freshwater ecotoxicity	-77.0%	-75.1%	-82.0%	-82.1%	-79.8%	-69.5%
Human toxicity (cancer effects)	-74.1%	-72.7%	-74.6%	-75.1%	-81.8%	-72.1%
Human toxicity (non-cancer effects)	-78.9%	-77.9%	-83.0%	-83.6%	-89.7%	-79.7%
Particulate matter	-71.1%	-69.9%	-70.3%	-72.0%	-86.5%	-70.6%
Water resource depletion	31.9%	38.4%	45.9%	28.0%	21.0%	49.0%
Mineral & fossil resource depletion	-71.4%	-70.1%	-70.8%	-72.7%	-79.0%	-67.0%
Cumulative energy demand	-73.0%	-71.8%	-75.9%	-77.2%	-80.3%	-69.2%
Minimum reduction[b]	*-63.5%*	*-60.8%*	*-70.3%*	*-72.0%*	*-59.2%*	*-42.7%*
Maximum reduction[b]	*-87.7%*	*-87.5%*	*-91.7%*	*-91.9%*	*-94.1%*	*-90.4%*

(a) Refillable glass bottles are used 10 times, while refillable PET bottles for 15 times.
(b) *Water resource depletion* is excluded from the calculation of the minimum and maximum reductions, because of its atypical behaviour. An impact increase is indeed observed, for this indicator, compared to all bottled water scenarios.

Table A.13: impact reductions resulting from the substitution of refined groundwater from the tap (waste prevention scenario 1) for the different types of bottled water, when the reusable jug used to withdraw tap water is washed under improved conditions (after every 5 uses in a load of 50 items) and bottles are transported to retailers or local distributors along a distance of 40 km (best case of baseline scenarios).

Impact categories	Reference baseline scenario					
	Virgin PET one-way bottles	**50% recy-cled PET one-way bottles**	**PLA one-way bottles to composting**	**PLA one-way bottles to incineration**	**Refillable glass bottles[a]**	**Refillable PET bottles[a]**
Climate change	-92.3%	-91.9%	-92.4%	-93.2%	-88.8%	-80.9%
Ozone depletion	-93.7%	-93.6%	-94.4%	-94.7%	-86.1%	-81.4%
Photochemical ozone formation	-88.5%	-87.8%	-90.1%	-91.0%	-89.5%	-83.1%
Acidification	-83.8%	-82.8%	-87.1%	-88.8%	-85.6%	-69.4%
Terrestrial eutrophication	-87.1%	-86.4%	-92.0%	-92.5%	-91.1%	-83.2%
Freshwater eutrophication	-94.0%	-93.4%	-96.2%	-96.4%	-89.0%	-83.5%
Marine eutrophication	-87.3%	-86.6%	-96.5%	-96.6%	-91.2%	-84.5%
Freshwater ecotoxicity	-89.3%	-87.8%	-92.7%	-92.7%	-75.8%	-63.7%
Human toxicity (cancer effects)	-82.4%	-80.2%	-83.0%	-83.7%	-68.2%	-54.0%
Human toxicity (non-cancer effects)	-85.3%	-83.4%	-90.7%	-91.3%	-91.1%	-69.1%
Particulate matter	-82.0%	-80.3%	-80.8%	-83.1%	-80.2%	-58.3%
Water resource depletion	-12.3%	-7.0%	-0.8%	-15.4%	3.4%	17.9%
Mineral & fossil resource depletion	-85.9%	-84.7%	-85.4%	-87.0%	-77.1%	-65.3%
Cumulative energy demand	-89.1%	-88.2%	-91.0%	-91.8%	-82.5%	-74.2%
Minimum reduction[b]	*-82.0%*	*-80.2%*	*-80.8%*	*-83.1%*	*-68.2%*	*-54.0%*
Maximum reduction[b]	*-94.0%*	*-93.6%*	*-96.5%*	*-96.6%*	*-91.2%*	*-84.5%*

(a) Both glass and PET refillable bottles are used for 25 times.
(b) *Water resource depletion* is excluded from the calculation of the minimum and maximum reductions, because of its atypical behaviour. A reduction (or increase) smaller than ±10% is indeed observed with respect to most bottled water scenarios, which are thus comparable to waste prevention scenario 1. A 17.9% increase is instead observed compared to refillable PET bottled water. Therefore, these values cannot be directly compared with those reported in Table 2.8.

Table A.14: impact reductions resulting from the substitution of refined surface water withdrawn from public fountains for the different types of bottled water, when no motorised vehicles are used for the roundtrip to the fountain (waste prevention scenario 2 - no car) and bottles are transported to retailers or local distributors along a distance of 40 km (best case of baseline scenarios).

Impact categories	**Reference baseline scenario**					
	Virgin PET one-way bottles	**50% recycled PET one-way bottles**	**PLA one-way bottles to composting**	**PLA one-way bottles to incineration**	**Refillable glass bottles[a]**	**Refillable PET bottles[a]**
Climate change	-84.9%	-84.1%	-86.7%	-85.1%	-78.2%	-62.8%
Ozone depletion	-94.5%	-94.5%	-95.5%	-95.2%	-88.0%	-84.0%
Photochemical ozone formation	-86.3%	-85.5%	-89.3%	-88.2%	-87.5%	-79.8%
Acidification	-81.6%	-80.4%	-87.3%	-85.3%	-83.6%	-65.2%
Terrestrial eutrophication	-85.5%	-84.7%	-91.6%	-91.1%	-90.0%	-81.1%
Freshwater eutrophication	-80.4%	-78.5%	-88.4%	-87.7%	-64.1%	-46.4%
Marine eutrophication	-84.8%	-84.0%	-96.0%	-95.8%	-89.5%	-81.4%
Freshwater ecotoxicity	-85.4%	-83.3%	-90.0%	-90.0%	-67.1%	-50.6%
Human toxicity (cancer effects)	-83.4%	-81.4%	-84.7%	-84.1%	-70.1%	-56.7%
Human toxicity (non-cancer effects)	-83.0%	-80.9%	-90.0%	-89.3%	-89.7%	-64.3%
Particulate matter	-81.9%	-80.3%	-83.1%	-80.8%	-80.2%	-58.2%
Water resource depletion	-55.1%	-52.4%	-56.7%	-49.3%	-47.1%	-39.7%
Mineral & fossil resource depletion	-85.1%	-83.9%	-86.3%	-84.6%	-75.8%	-63.5%
Cumulative energy demand	-83.6%	-82.2%	-87.6%	-86.5%	-73.7%	-61.2%
Minimum reduction	*-55.1%*	*-52.4%*	*-56.7%*	*-49.3%*	*-47.1%*	*-39.7%*
Maximum reduction	*-94.5%*	*-94.5%*	*-96.0%*	*-95.8%*	*-90.0%*	*-84.0%*

(a) Both glass and PET refillable bottles are used for 25 times.

Table A.15: impact variations resulting from the substitution of glass refillable for one-way bottled water, when this is transported to retailers or local distributors along a distance of 300 km and refillable bottles are used 10 times (base case of baseline scenarios).

Impact categories	Reference baseline scenario			
	Virgin PET one-way bottles	**50% recycled PET one-way bottles**	**PLA one-way bottles to composting**	**PLA one-way bottles to incineration**
Climate change	5.7%	10.0%	-4.4%	4.6%
Ozone depletion	-15.6%	-14.8%	-27.4%	-24.1%
Photochemical ozone formation	74.8%	79.6%	54.1%	61.9%
Acidification	66.2%	72.8%	29.8%	43.5%
Terrestrial eutrophication	94.5%	98.8%	48.4%	53.7%
Freshwater eutrophication	-32.2%	-26.3%	-58.3%	-55.9%
Marine eutrophication	94.7%	99.3%	-12.3%	-9.2%
Freshwater ecotoxicity	-22.0%	-13.2%	-43.6%	-43.5%
Human toxicity (cancer effects)	2.7%	11.4%	-3.2%	-0.3%
Human toxicity (non-cancer effects)	96.6%	112.7%	33.8%	41.5%
Particulate matter	112.0%	125.4%	102.0%	121.7%
Water resource depletion	-4.8%	0.5%	-8.0%	6.7%
Mineral & fossil resource depletion	2.7%	9.3%	-3.8%	5.8%
Cumulative energy demand	3.5%	10.1%	-17.7%	-11.5%

Note: grey cells depict insignificant impact variations (lower than ±10%), while green cells depict the few situations in which a significant impact reduction (>10%) is achieved.

Table A.16: impact variations resulting from the substitution of glass refillable for one-way bottled water, when this is transported to retailers or local distributors along a distance of 40 km and refillable bottles are used 25 times (best case of baseline scenarios).

Impact categories	Reference baseline scenario			
	Virgin PET one-way bottles	**50% recycled PET one-way bottles**	**PLA one-way bottles to composting**	**PLA one-way bottles to incineration**
Climate change	-30.6%	-26.9%	-39.0%	-31.6%
Ozone depletion	-54.4%	-53.9%	-62.1%	-60.0%
Photochemical ozone formation	9.9%	16.4%	-14.1%	-5.7%
Acidification	12.1%	19.4%	-22.6%	-10.6%
Terrestrial eutrophication	44.9%	52.6%	-16.1%	-10.6%
Freshwater eutrophication	-45.6%	-40.3%	-67.7%	-65.7%
Marine eutrophication	44.2%	52.2%	-61.7%	-59.9%
Freshwater ecotoxicity	-55.6%	-49.3%	-69.7%	-69.7%
Human toxicity (cancer effects)	-44.6%	-37.9%	-48.8%	-46.7%
Human toxicity (non-cancer effects)	64.4%	85.4%	-3.1%	4.2%
Particulate matter	-8.8%	-0.5%	-14.8%	-2.9%
Water resource depletion	-15.1%	-10.0%	-18.1%	-4.0%
Mineral & fossil resource depletion	-38.5%	-33.4%	-43.4%	-36.2%
Cumulative energy demand	-37.5%	-32.4%	-52.9%	-48.6%

Note: grey cells depict insignificant impact variations (lower than ±10%), while red cells depict the few situations in which the substitution involves an overall impact increase.

Table A.17: impact variations resulting from the substitution of glass refillable for one-way bottled water, when this is transported to retailers or local distributors along a distance of 800 km and refillable bottles are used 10 times (worst case of baseline scenarios).

Impact categories	Reference baseline scenario			
	Virgin PET one-way bottles	**50% recycled PET one-way bottles**	**PLA one-way bottles to composting**	**PLA one-way bottles to incineration**
Climate change	40.5%	44.3%	30.8%	39.4%
Ozone depletion	20.9%	21.7%	8.1%	11.8%
Photochemical ozone formation	96.9%	99.6%	84.5%	89.4%
Acidification	88.4%	92.7%	62.0%	72.5%
Terrestrial eutrophication	107.4%	109.6%	80.6%	84.1%
Freshwater eutrophication	-10.6%	-4.0%	-41.8%	-38.7%
Marine eutrophication	107.3%	109.7%	30.6%	33.6%
Freshwater ecotoxicity	14.2%	23.5%	-11.1%	-11.0%
Human toxicity (cancer effects)	42.0%	49.8%	36.5%	39.3%
Human toxicity (non-cancer effects)	105.3%	115.3%	59.4%	65.7%
Particulate matter	114.6%	123.1%	107.9%	120.7%
Water resource depletion	9.0%	14.4%	5.8%	20.5%
Mineral & fossil resource depletion	36.5%	42.6%	30.2%	39.3%
Cumulative energy demand	36.7%	42.8%	15.5%	21.9%

Note: grey cells depict insignificant impact variations (lower than ±10%), while green cells depict the few situations in which a significant impact reduction (>10%) is achieved.

Appendix B

This appendix provides additional information on the life cycle assessment (LCA) study summarised in Section 3, which evaluates the environmental and energy convenience of the substitution of liquid detergents packed in single-use containers for those distributed loose by means of self-dispensing systems and refillable containers. In particular, Section B.1 provides some graphical details on this alternative distribution method, while Section B.2 includes further details on the modelling of the compared scenarios. Finally, Section B.3 presents additional results relating to waste generation, impact indicators and respective variations between waste prevention and baseline scenarios.

B.1 Graphical details on the distribution of liquid detergents through self-dispensing systems

Figure B.1: example of a 600 litre reusable tank (intermediate bulk container) used for the transport of liquid detergents to retail establishments when the product is to be distributed loose through self-dispensing systems. The image was taken during a survey of the manufacturing and packaging plant of a medium-sized company located in central Italy.

Figure B.2: example of an automatic self-dispensing system for liquid detergent distribution by means of refillable containers. The image was taken nearby a retail establishment located in northern Italy.

B.2 Modelling of scenarios: further details

This section provides additional information on the approach, input data and inventory data used for the modelling of the different life cycle stages included in the system boundaries under the compared scenarios.

B.2.1 Life cycle of primary and transport packages

B.2.1.1 Input data

To estimate the average mass of containers and caps needed per functional unit under the baseline scenarios, the following procedure was adopted. For each baseline scenario, a sample of filled single-use containers was acquired first. Containers used for the marketing of the major brands of the category of detergent of concern were included in each sample, along with those used by some minor brands (private labels). However, for some scenarios, only minor brands were included, since they were the unique brands using the type of container of interest. The acquired containers were then emptied and weighed, along with the respective caps. Overall, 219 containers and caps were weighed: 90 for laundry detergents, 68 for fabric softeners and 61 for hand dishwashing detergents. The mass of packaging needed per litre of detergent was then calculated for each item of the sample, by dividing the measured mass by the size of the respective container. Finally, single values were averaged and the result converted to the functional unit. The number of items acquired for each sample (i.e. for each scenario) and the calculated average masses are reported in Table B.1 for laundry detergents, Table B.2 for fabric softeners and Table B.3 for hand dishwashing detergents.

Table B.1: number of single-use containers and caps weighed for each baseline scenario pertaining to laundry detergents and estimated average masses.

Scenario	Material of the container	Size of the container (ml)	Number of weighed items	Container average mass (grams)	Average mass of containers needed per functional unit (kg)	Cap average mass (grams)	Average mass of caps needed per functional unit (kg)
1	**HDPE**	**750**	6	52.8	70.4	7.1	9.4
2		**1000**	11	51.2	51.2	8.2	8.2
3		**1500-1518**	8	65.2	43.4	11.8	7.8
4		**1820-2100**	23	92.0	48.1	13.0	6.8
5		**2409-2625**	15	110.3	44.1	13.1	5.2
6		**3000-3066**	14	116.0	38.6	12.7	4.2
7		**3900-4000**	4	131.8	33.2	10.9	2.7
8		**5000**	2	160.0	32.0	11.5	2.3
9	**PET**	**750**	5	45.1	60.1	8.4	11.2
10		**924**	1	55.0	59.5	16.0	17.3
11		**1848**	1	52.5	28.4	2.5	1.4
Total			*90*	-	-	-	-

Table B.2: number of single-use containers and caps weighed for each baseline scenario pertaining to fabric softeners and estimated average masses.

Scenario	Material of the container	Size of the container (ml)	Number of weighed items	Container average mass (grams)	Average mass of containers needed per functional unit (kg)	Cap average mass (grams)	Average mass of caps needed per functional unit (kg)
1	**HDPE**	**750**	4	48.4	64.5	8.6	11.5
2		**1000**	1	46.5	46.5	13.5	13.5
3		**1500-1560**	9	70.6	46.6	12.1	8.0
4		**2000-2015**	11	78.3	39.1	12.4	6.2
5		**2460**	1	101	41.1	13	5.3
6		**2990-3000**	9	126.2	42.1	12.7	4.2
7		**4000**	9	131.5	32.9	12.9	3.2
8	**PET**	**750**	13	39.6	52.8	7.8	10.4
9		**1000**	3	44.2	44.2	10.3	10.3
10		**1500**	5	61.3	40.9	9.9	6.6
11		**2000**	3	67.8	33.9	8	4.0
Total			*68*	-	-	-	-

Table B.3: number of single-use containers and caps weighed for each baseline scenario pertaining to hand dishwashing detergents and estimated average masses.

Scenario	Material of the container	Size of the container (ml)	Number of weighed items	Container average mass (grams)	Average mass of containers needed per functional unit (kg)	Cap average mass (grams)	Average mass of caps needed per functional unit (kg)
1	**HDPE**	**750**	4	43.1	57.4	4.1	5.4
2		**1000-1100**	3	58.7	55.2	3.7	3.5
3		**1250**	7	60.3	48.2	4.1	3.3
4		**1500**	2	65.8	43.8	4.5	3.0
5		**2000**	2	78.8	39.4	5.5	2.8
6		**3000**	3	108.8	36.3	8.9	3.0
7		**4000**	3	132.3	33.1	10.4	2.6
8		**5000**	2	137.5	27.5	12.0	2.4
9	**PET**	**500-650**	13	30.3	57.6	4.4	8.4
10		**750**	9	39.8	53.0	4.9	6.5
11		**1000**	9	45.8	45.8	5.5	5.5
12		**1250**	3	52.2	41.7	4.5	3.6
13		**1500**	1	49.5	33.0	4.5	3
Total			*61*	-	-	-	-

The estimate of the average mass of corrugated cardboard boxes needed per functional unit under each baseline scenario was carried out by following a sampling and calculation procedure similar to that described above for single-use containers and caps. A total of 133 cardboard boxes were thus acquired and weighted: 52 for laundry detergents, 43 for fabric softeners and 37 for hand dishwashing detergents. In this case, also the number of single-use containers included into each of the weighed boxes had to be acquired. The number of boxes acquired for each scenario (sample) and calculated average masses are reported in Tables B.4, B.5 and B.6 for laundry detergents, fabric softeners and hand dishwashing detergents, respectively.

Table B.4: number of cardboard boxes weighed for each baseline scenario pertaining to laundry detergents and estimated average masses.

Scenario	Material of the container	Size of the container (ml)	Number of weighed items	Box average mass (grams)	Average mass of cardboard boxes needed per functional unit (kg)
1	**HDPE**	**750**	5	277	32.6
2		**1000**	6	408	33.5
3		**1500-1518**	3	268	29.7
4		**1820-2100**	16	368	28
5		**2409-2625**	7	398	35.8
6		**3000-3066**	9	625	27.3
7		**3900-4000**	2	643	40.6
8		**5000**	2	1030[a]	11.4
9	**PET**	**750**	-	-	32.6[b]
10		**924**	1	218	47.2
11		**1848**	1	260	28.1
Total			*52*	-	-

(a) 5000 ml containers are transported within exhibitor boxes
(b) Due to missing data, an amount per functional unit equal to that calculated for 750 ml HDPE containers was assumed.

Table B.5: number of cardboard boxes weighed for each baseline scenario pertaining to fabric softeners and estimated average masses.

Scenario	Material of the container	Size of the container (ml)	Number of weighed items	Box average mass (grams)	Average mass of cardboard boxes needed per functional unit (kg)
1	**HDPE**	**750**	4	355.0	33.8
2		**1000**	1	294.0	36.8
3		**1500-1560**	6	332.4	23.9
4		**2000-2015**	4	403.4	27.3
5		**2460**	1	320.0	21.7
6		**2990-3000**	8	411.9	29.6
7		**4000**	5	631.2	26.9
8	**PET**	**750**	8	302.8	33.0
9		**1000**	1	302.9	25.2
10		**1500**	4	458.0	26.4
11		**2000**	1	444.0	24.7
Total			*43*	-	-

Table B.6: number of cardboard boxes weighed for each baseline scenario pertaining to hand dishwashing detergents and estimated average masses.

Scenario	Material of the container	Size of the container (ml)	Number of weighed items	Box average mass (grams)	Average mass of cardboard boxes needed per functional unit (kg)
1	**HDPE**	**750**	3	355.8	28.0
2		**1000-1100**	-	-	31.2[a]
3		**1250**	3	380.1	25.3
4		**1500**	2	358.5	25.0
5		**2000**	1	387.0	32.3
6		**3000**	2	312.0	22.0
7		**4000**	2	368.0	23.0
8		**5000**	2	382.0	19.1
9	**PET**	**500-650**	8	289.5	34.5
10		**750**	5	264.9	24.9
11		**1000**	6	356.4	31.2
12		**1250**	1	303.0	24.1
13		**1500**	1	492.0	27.3
Total			*37*	-	-

(a) An amount per functional unit equal to that calculated for 1000 ml PET containers was assumed, as no samples were available for this size.

For each baseline scenario, the number of pallets needed per functional unit was estimated by acquiring a sample of pallet compositions. A pallet composition indicates the number of cardboard boxes loaded overall on a single pallet and allows one to calculate the overall volume of detergent transported on that pallet (provided the number of containers included in each box is known). For each composition retrieved for a given scenario, the number of pallets needed per litre of transported detergent was calculated. The calculated values were then averaged and the result converted to the functional unit. To this purpose, the assumption is made that pallets are used for 20 transport cycles overall before breaking and being discarded (Creazza and Dallari, 2007). The whole procedure was repeated for each baseline scenario. The number of pallet compositions acquired for each scenario and calculated average values are separately reported for the three considered categories of detergent in Tables B.7, B.8 and B.9.

Table B.7: number of pallet compositions acquired for each baseline scenario pertaining to laundry detergents and estimated average number of pallets needed per functional unit.

Scenario	Material of the container	Size of the container (ml)	Number of pallet compositions acquired	Pallet mass (kg)	Average number of pallets needed per functional unit (-)
1	**HDPE**	**750**	7	22	0.0903
2		**1000**	9		0.108
3		**1500-1518**	6		0.0899
4		**1820-2100**	27		0.0927
5		**2409-2625**	9		0,125
6		**3000-3066**	10		0.139
7		**3900-4000**	4		0.0880
8		**5000**	2		0.185
9	**PET**	**750**	2		0.0860
10		**924**	1		0.0966
11		**1848**	1		0.0676
Total			*78*	-	-

Table B.8: number of pallet compositions acquired for each baseline scenario pertaining to fabric softeners and estimated average number of pallets needed per functional unit.

Scenario	Material of the container	Size of the container (ml)	Number of pallet compositions acquired	Pallet mass (kg)	Average number of pallets needed per functional unit (-)
1	**HDPE**	**750**	4	22	0.0992
2		**1000**	1		0.0781
3		**1500-1560**	7		0.0954
4		**2000-2015**	9		0.104
5		**2460**	1		0.106
6		**2990-3000**	10		0.0984
7		**4000**	9		0.109
8	**PET**	**750**	10		0.0974
9		**1000**	1		0.0694
10		**1500**	5		0.0870
11		**2000**	1		0.174
Total			*58*	-	-

Table B.9: number of pallet compositions acquired for each baseline scenario pertaining to hand dishwashing detergents and estimated average number of pallets needed per functional unit.

Scenario	Material of the container	Size of the container (ml)	Number of pallet compositions acquired	Pallet mass (kg)	Average number of pallets needed per functional unit (-)
1	**HDPE**	**750**	3	22	0.0802
2		**1000-1100**	1		0.140
3		**1250**	4		0.104
4		**1500**	2		0.191
5		**2000**	1		0.0868
6		**3000**	2		0.153
7		**4000**	5		0.0856
8		**5000**	3		0.0916
9	**PET**	**500-650**	9		0.105
10		**750**	6		0.0911
11		**1000**	8		0.0889
12		**1250**	2		0.151
13		**1500**	1		0.0694
Total			*47*	-	-

Table B.10 reports, for each waste prevention scenario, the masses of the refillable container provided to the consumer and of the respective cap. These masses were estimated by weighing one sample for each scenario.

Table B.10: estimated masses of the refillable containers provided to the consumer in waste prevention scenarios and of the respective caps.

Scenario	Detergent category	Material of the container	Size of the container (ml)	Refillable container mass (grams)[a]	Cap mass (grams)[a]
1	All categories	HDPE	1000	62	9
2	Laundry detergents	HDPE	3000	120	12
	Fabric softeners		2000	103	12
	Hand dishwashing detergents		1000	71.5	8.5

(a) The mass of containers and caps needed per functional unit is not reported since it depends on the number of times the refillable container is used.

The average masses of the packages used for the transport of the refillable containers under the waste prevention scenarios are reported in Table B.11, along with the composition of the respective pallet. For reusable caps employed in waste prevention scenario 1 (which are transported separately from containers) the same type of data is instead provided in Table B.12. For both containers and caps, these data were acquired from the respective producer.

Table B.11: average masses of the packages used for the transport of empty refillable containers to retail outlets and composition of the respective pallet.

Scenario	Detergent category	Cardboard box mass (grams)[a]	Number of containers per box	LLDPE stretch film mass (grams)[a]	Pallet mass (kg)[a]	Number of boxes per pallet
1	All categories	650	100	50	22	8
2	Laundry detergents	2355	41	250	22	8
	Fabric softeners		56			
	Hand dishwashing detergents		102			

(a) The mass of cardboard boxes and stretch film needed per functional unit, as well as the number of pallets needed, are not reported since they depend on the number of times the refillable container is used.

Table B.12: average masses of the packages used for the transport of reusable caps to retail outlets (waste prevention scenario 1) and composition of the respective pallet.

Scenario	Detergent category	LDPE bag mass (grams)[a]	Number of caps per bag	Cardboard box mass (grams)[a]	Number of bags per box	LLDPE stretch film mass (grams)[a]	Pallet mass (kg)[a]	Number of boxes per pallet
1	All categories	10	300	475	1	50	22	25

(a) The mass of bags, cardboard boxes and stretch film needed per functional unit, as well as the number of pallets needed, are not reported since they depend on the number of times the refillable container is used.

Finally, table B.13 shows the average masses considered for the different components of the 600 litre reusable tank used in waste prevention scenarios for detergent transport to retail outlets. These masses were provided directly by the producer of the tanks and are reported in the mentioned table also with reference to the functional unit. These latter were calculated assuming that tanks are used for 50 transport cycles.

Table B.13: average masses of the different components of the 600 litre reusable tanks used in waste prevention scenarios for detergent delivery to retail outlets and mass of the material of each component needed per functional unit.

Component	**Mass (kg)**	**Mass of material needed per functional unit (kg)**
Inner HDPE container (including screw cap and outlet valve)	13	0.433
Outer tubular steel cage (galvanized)	20	0.667
Wooden pallet	22	3.33×10^{-5} (a)

(a) Number of pallets needed per functional unit.

B.2.1.2 Packaging end of life

Both single-use and refillable containers were assumed to be separately collected together with other plastic wastes. They are then sorted (i.e. subdivided by type of polymer) and mechanically recycled for the production of secondary HDPE or PET granules. These substitute virgin granules in other products systems or in the studied system (when a 100% recycled content is assumed for single-use containers in the sensitivity analysis). Also PP caps were assumed to be separately collected along with other plastic wastes, but subsequently rejected during sorting operations or during container recycling. They are finally incinerated in a waste to energy plant producing both electricity and heat.

All transport packages and the different components of reusable tanks were assumed to be recycled, as well. In particular, cardboard boxes are pressed to bales and used in the production of corrugated board base papers (recycled medium and testliner). These papers are then entirely used in the studied system for the manufacturing of boxes, which have a 100% recycled content. The stretch film is flaked along with other polyolefins and used for the manufacturing of profiled bars, which substitute wooden planks. Pallets are instead grinded and used in the production of particle board, which substitutes plywood board. Finally, the HDPE container of tanks is shredded, grinded and granulated, substituting virgin HDPE granules. Conversely, the steel cage of tanks is pressed to blocks and used in the manufacturing of a generic semi-finished product from continuous casting of liquid steel in electric arc furnaces. This is partly used in the system for the manufacturing of the cage and partly in other systems in substitution of primary steel semi-finished products.

B.2.1.3 Inventory data for unit processes

Inventory data on the primary production of packaging materials, on their subsequent conversion into finished products and on the possible transport between the two stages were derived from the ecoinvent database (version 2.2). The dataset related to primary steel production was however updated with data reported in Remus et al. (2013). Data from the ecoinvent database were used also for the recycling of cardboard boxes, of the steel cage of tanks and of wooden pallets. However, data provided for steel recycling were updated with those reported in Remus et al. (2013), while data for wood recycling were adjusted according to the procedure reported in Rigamonti and Grosso (2009). Mechanical recycling of single-use and refillable containers, and of the LLDPE stretch film, was modelled based on inventory data provided in Rigamonti and Grosso (2009), as well. The recycling of the inner HDPE container of the reusable tanks was instead modelled based on data directly provided by a manufacturer of lines indented for their shredding, grinding and granulation. Finally, a dataset from the ecoinvent database was used to model the incineration of polypropylene caps in a waste to energy plant, but the avoided burdens associated with the substitution of electricity and heat from traditional sources were included in addition.

B.2.2 Packing operations

Table B.14 summarises the consumptions ascribed to the following operations: (1) detergent packing in single-use containers and subsequent boxing and palletisation of containers (baseline scenarios); (2) detergent packing in reusable tanks (waste prevention scenarios); and (3) packing and palletisation of refillable containers (waste prevention scenarios). The estimates reported were produced based on annual consumption and production data related to a detergent manufacturing and packaging plant located in central Italy.

Table B.14: type and magnitude of the consumptions ascribed to packing operations.

Operation	Scenario	Type of consumption	Magnitude of the consumption
Filling, capping, labelling, boxing and palletisation of single-use containers	Baseline scenarios	Electricity (medium voltage)	0.0147 kWh/litre of packed detergent
Washing and filling of tanks	Waste prevention scenarios	Public network water	0.0333 litres/litre of packed detergent
		Compressed air (6 bar)	0.0175 m^3/litre of packed detergent
Boxing and palletisation of refillable containers and respective caps[a]	Waste prevention scenario 1	Electricity (medium voltage)	0.0147 kWh/litre of packed detergen[c]
Capping, labelling, boxing and palletisation of refillable containers[b]	Waste prevention scenario 2	Electricity (medium voltage)	0.0147 kWh/litre of packed detergent[c]

(a) This operation is carried out directly by the producer of the refillable containers and of the reusable caps.
(b) This operation is carried out at the detergent manufacturing plant
(c) Due to the lack of specific data, the same consumption estimated for baseline scenarios (first row) was attributed also to this operation.

Inventory data on medium voltage electricity supply (Italian energy mix), public network water delivery and compressed air generation were derived from the ecoinvent database (v. 2.2).

B.2.3 Transport to retail outlets

To calculate the average amount of detergent transported per functional unit, an average density was estimated for all the three considered categories of detergent. To this end, the density of the detergent packed in each of the containers weighed for the purposes described in Section B.2.1.1 was calculated. The estimate was based on the difference between the mass of the filled and of the empty container. Single estimates were then averaged and the results converted to the functional unit, providing the values reported in Table B.15.

Table B.15: estimated average densities for the three categories of detergent and mass of detergent transported to retail outlets per functional unit.

Category of detergent	Estimated density (g/litre)	Mass of detergent transported per functional unit (kg)
Laundry detergents	1025	1025
Fabric softeners	997	997
Hand dishwashing detergents	1020	1020

B.2.4 Detergent sale and purchase

Table B.16 provides the masses of the main components of the self-dispensing system used in waste prevention scenarios. In particular, the masses of the HDPE tanks and of the PVC covering panels were estimated based on the technical features of the self-dispensing system used in the real experience depicted in waste prevention scenario 1. For steel parts, a rough estimate was instead directly provided by the producer of the device. For each component, the amount of material needed per functional unit is also reported in Table B.16. It was estimated by assuming that the average lifespan of the self-dispensing system is equal to 10 years and that an overall volume of 74.880 litres is delivered by the system over one year. This volume corresponds to a daily depletion of the content of the three tanks incorporated into the system and to a use of the system for 312 days per year (6 days per week and 52 weeks per year).

Table B.16: estimated masses of the main components of the self-dispensing system used for detergent distribution in waste prevention scenarios and mass of the material of each component needed per functional unit.

Component	Mass (kg)	Mass of material needed per functional unit (kg)
Steel parts (frame, plates, etc.)	150	0.2
HDPE tanks (3 per each device)	6.4	8.53×10^{-3}
Expanded PVC covering panels	20.2	0.0269

The ecoinvent database (v. 2.2) is the main source of inventory data on primary or secondary production of the material of each component, on the processes of conversion into semi-finished products and on the possible transport between the two stages. For steel manufacturing and recycling, an update was however performed according to Remus et al. (2013). Data from ecoinvent were used also for the incineration of PVC covering panels in a waste to energy plant, but the avoided burdens associated with electricity and heath generation were included in addition. Finally, the recycling of HDPE tanks was modelled based on data directly provided by a manufacturer of lines indented for shredding, grinding and granulation of the inner container.

B.3 Further results

The following sections complete the framework of results produced for the analysed scenarios. In particular, Section B.3.1 provides additional results for waste generation, while Sections B.3.2 and B.3.3 focus, respectively, on the potential impacts and on their variation between waste prevention and baseline scenarios.

B.3.1 Waste generation

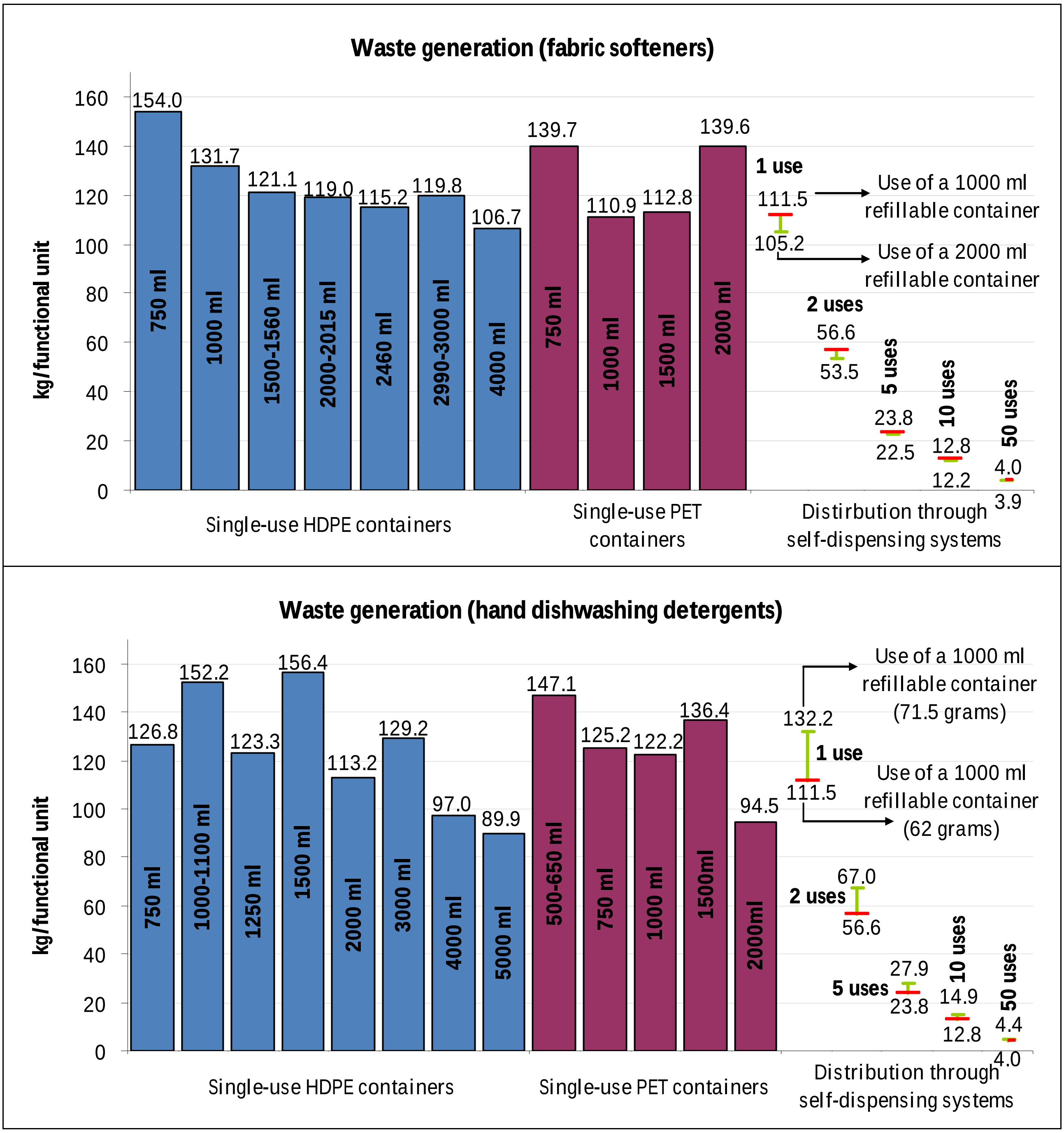


Figure B.3: waste generated in fabric softener and hand dishwashing detergent distribution. Bars are the baseline scenarios, while horizontal dashes are the two waste prevention scenarios for different number of uses of the refillable container.

Table B.17: difference between the amount of waste generated in the scenario where fabric softeners are distributed loose with a 2000 ml refillable container (waste prevention scenario generating less waste) and that generated in the two respective baseline scenarios with the lowest and the highest generation of waste.

Reference baseline scenario	Number of uses of the 2000 ml refillable container				
	1	2	5	10	50
Distribution with a 4000 ml HDPE container (baseline scenario generating less waste)	-1.5 kg/fu[a,b] (-1.4 %)	-53.2 kg/fu (-49.9 %)	-84.2 kg/fu (-78.9 %)	-94.6 kg/fu (-88.6 %)	-102.8 kg/fu (-96.3 %)
Distribution with a 750 ml HDPE container (baseline scenario generating most waste)	-48.8 kg/fu (-31.7 %)	-100.5 kg/fu (-65.3 %)	-131.5 kg/fu (-85.4 %)	-141.9 kg/fu (-92.1 %)	-150.1 kg/fu (-97.5 %)

(a) fu = functional unit.
(b) Negative variations per functional unit represent the waste prevention potentials achievable with the distribution of fabric softeners through self-dispensing systems. They are expressed as the amount of waste prevented per 1000 litres of detergent distributed loose rather than packed in a single-use container of the type considered in the baseline scenario of reference.

Table B.18: difference between the amount of waste generated in the scenario where hand dishwashing detergents are distributed loose with a 1000 ml refillable container weighting 62 grams (waste prevention scenario generating less waste) and that generated in the two respective baseline scenarios with the lowest and the highest generation of waste.

Baseline scenario of comparison	Number of uses of the 1000 ml refillable container (62 grams)				
	1	2	5	10	50
Distribution with a 5000 ml HDPE container (baseline scenario generating less waste)	21.5 kg/fu[a] (23.9 %)	-33.3 kg/fu (-37.0 %)	-66.2 kg/fu (-73.6 %)	-77.1 kg/fu (-85.8 %)	-85.9 kg/fu (-95.5 %)
Distribution with a 1500 ml HDPE container (baseline scenario generating most waste)	-45.0 kg/fu (-24.2 %)	-99.8 kg/fu (-61.5 %)	-132.7 kg/fu (-83.8 %)	-143.6 kg/fu (-91.3 %)	-152.4 kg/fu (-97.3 %)

(a) See footnotes to Table B.17.

B.3.2 Impact assessment results

B.3.2.1 Laundry detergents

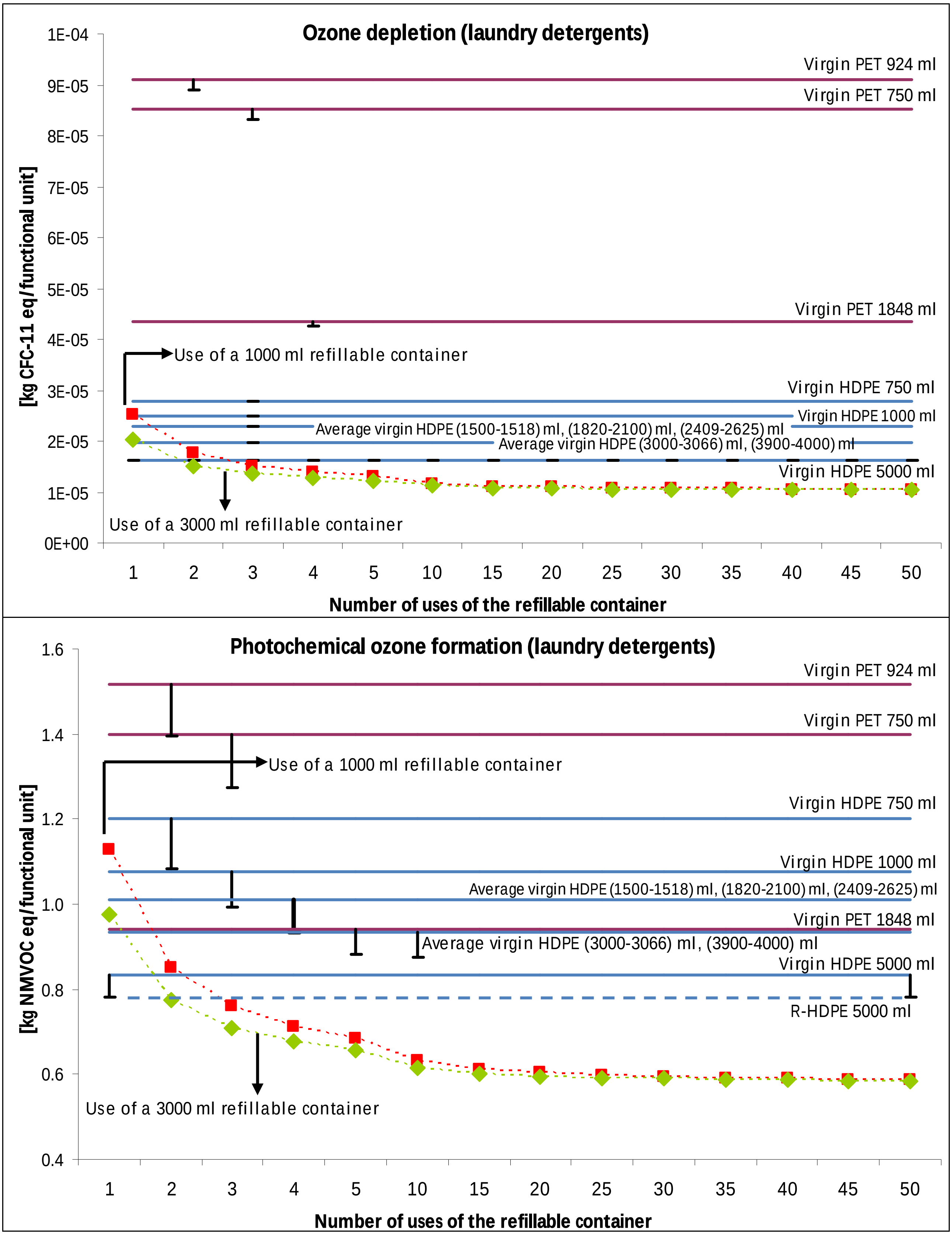


Figure B.4: *ozone depletion* and *photochemical ozone formation* impact indicators for laundry detergents. Horizontal lines represent the impacts of baseline scenarios, while squares and rhombuses the impacts of the two waste prevention scenarios as a function of the number of uses of the refillable container. Error bars represent the variation of the impacts when single-use containers are produced entirely from recycled material.

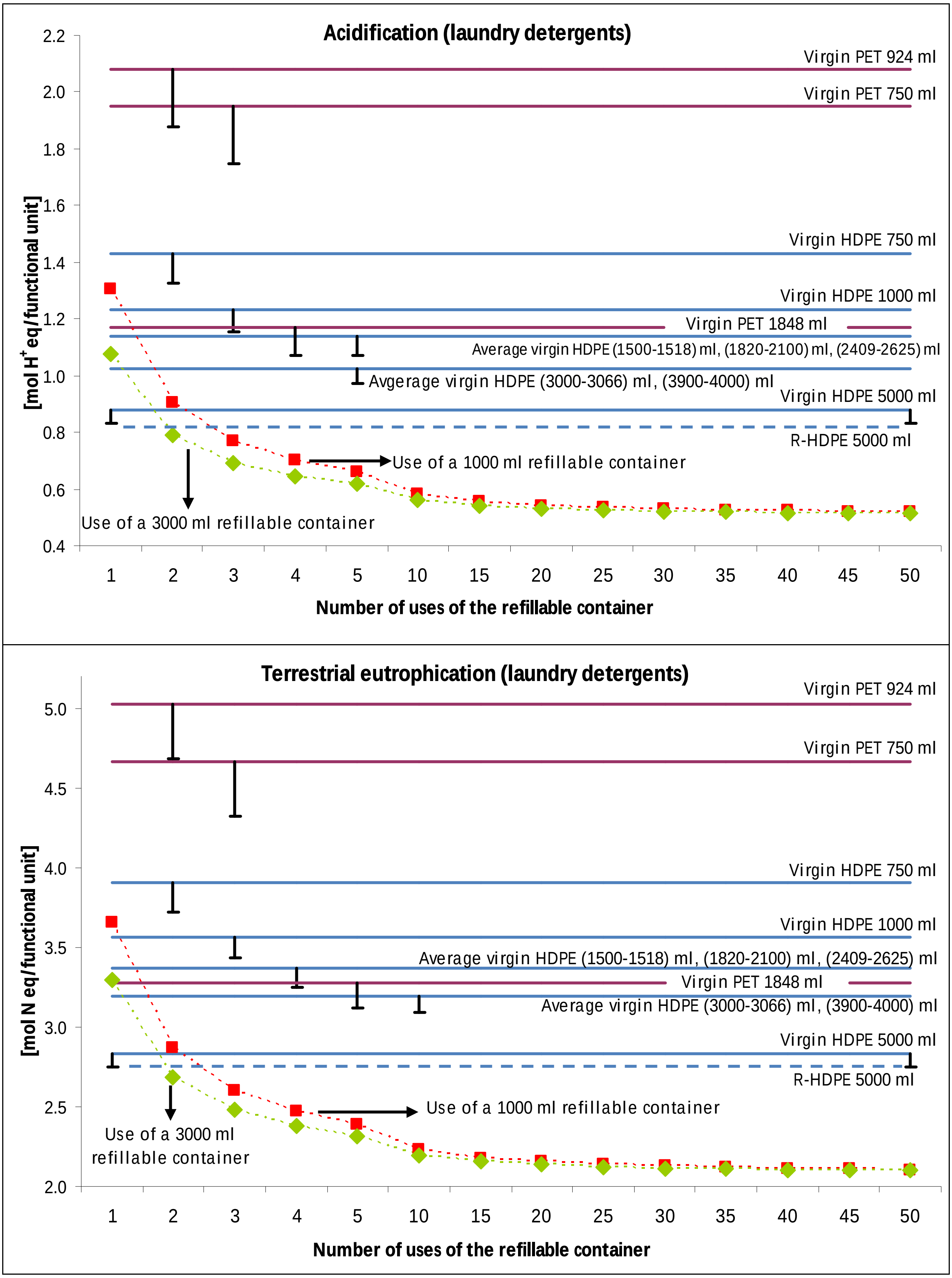


Figure B.5: *acidification* and *terrestrial eutrophication* impact indicators for laundry detergents. Horizontal lines represent the impacts of baseline scenarios, while squares and rhombuses the impacts of the two waste prevention scenarios as a function of the number of uses of the refillable container. Error bars represent the variation of the impacts when single-use containers are produced entirely from recycled material.

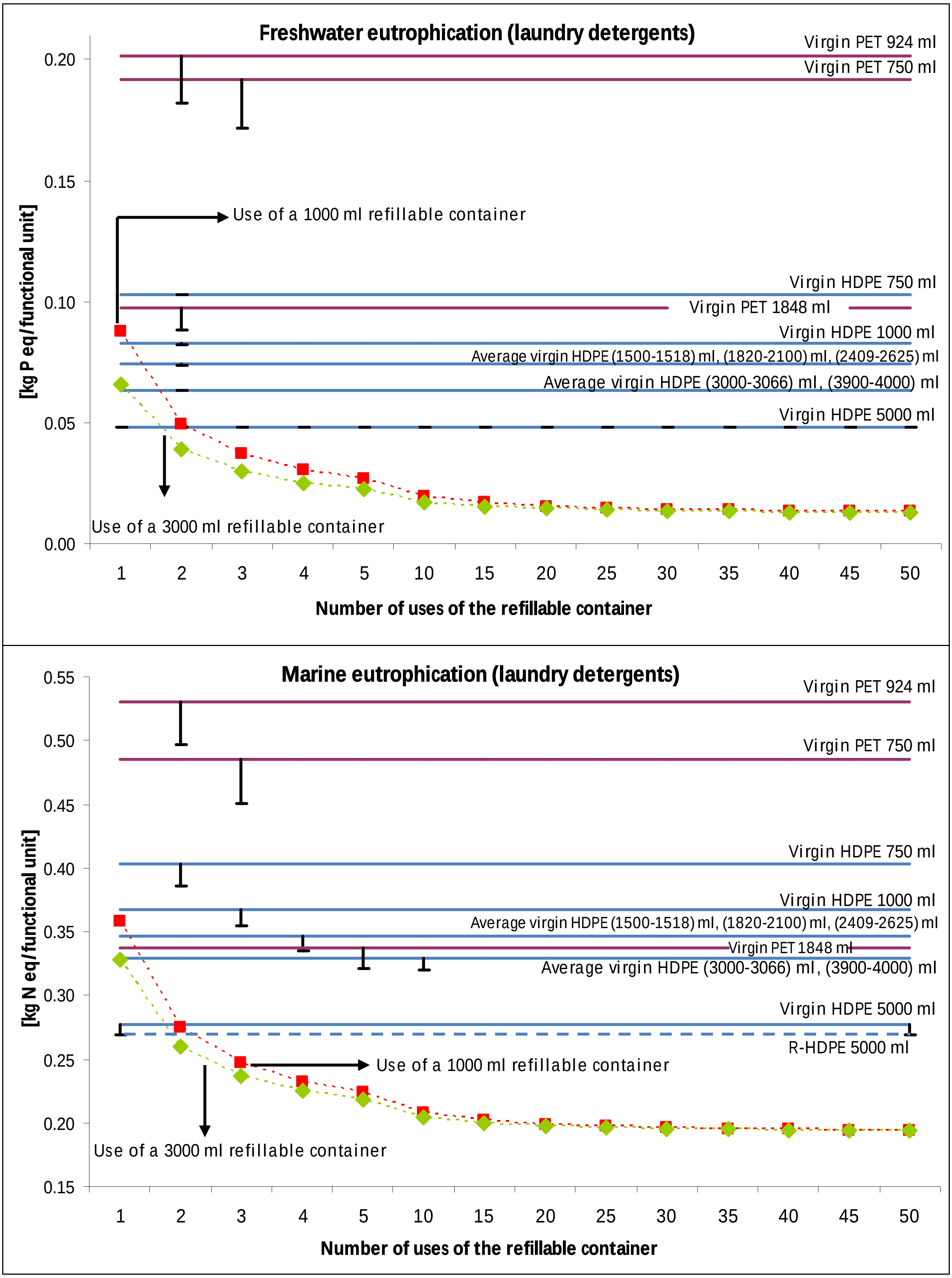


Figure B.6: *freshwater eutrophication* and *marine eutrophication* impact indicators for laundry detergents. Horizontal lines represent the impacts of baseline scenarios, while squares and rhombuses the impacts of the two waste prevention scenarios as a function of the number of uses of the refillable container. Error bars represent the variation of the impacts when single-use containers are produced entirely from recycled material.

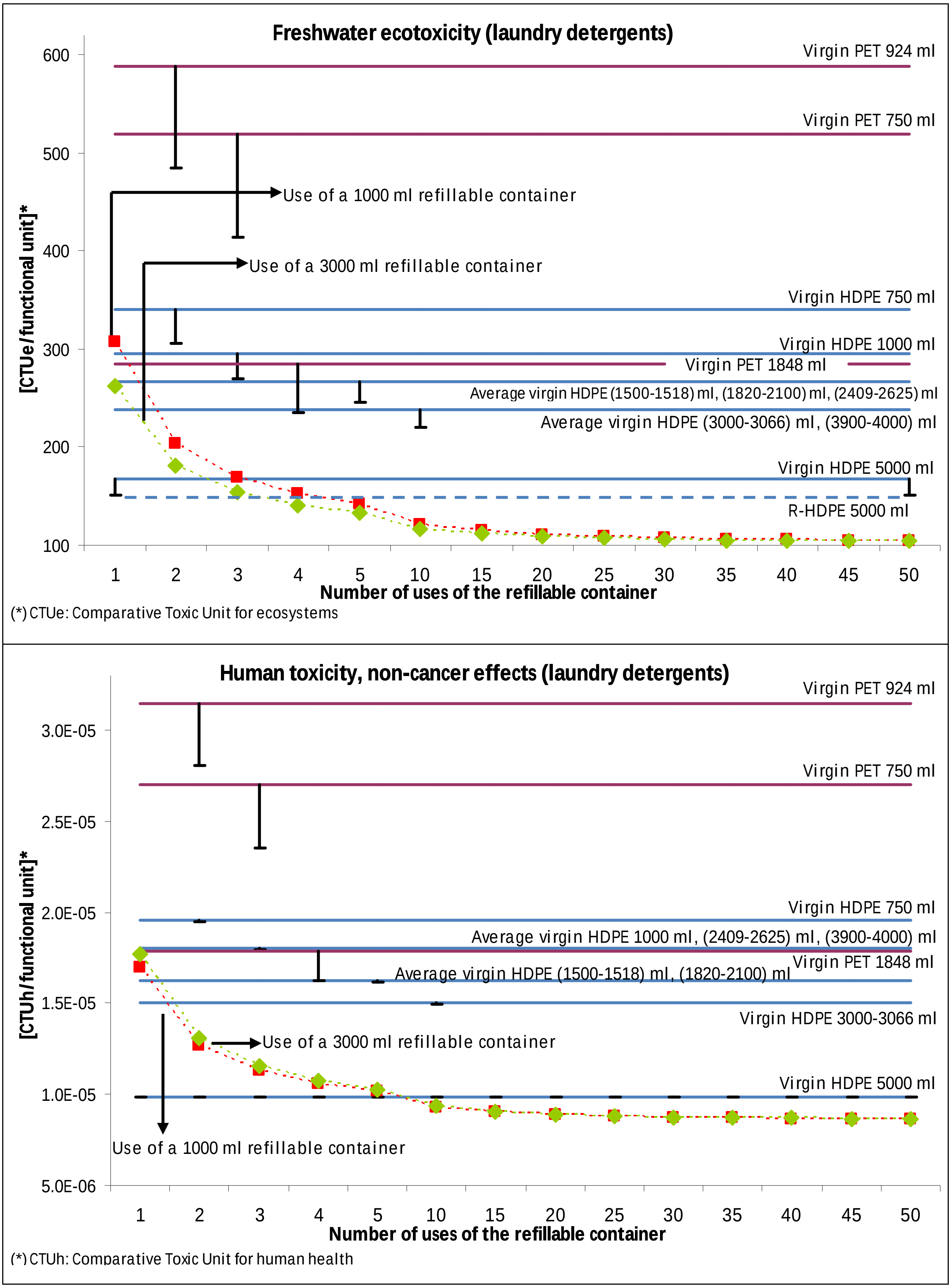


Figure B.7: *freshwater ecotoxicity* and *human toxicity, non-cancer effects* impact indicators for laundry detergents. Horizontal lines represent the impacts of baseline scenarios, while squares and rhombuses the impacts of the two waste prevention scenarios as a function of the number of uses of the refillable container. Error bars represent the variation of the impacts when single-use containers are produced entirely from recycled material.

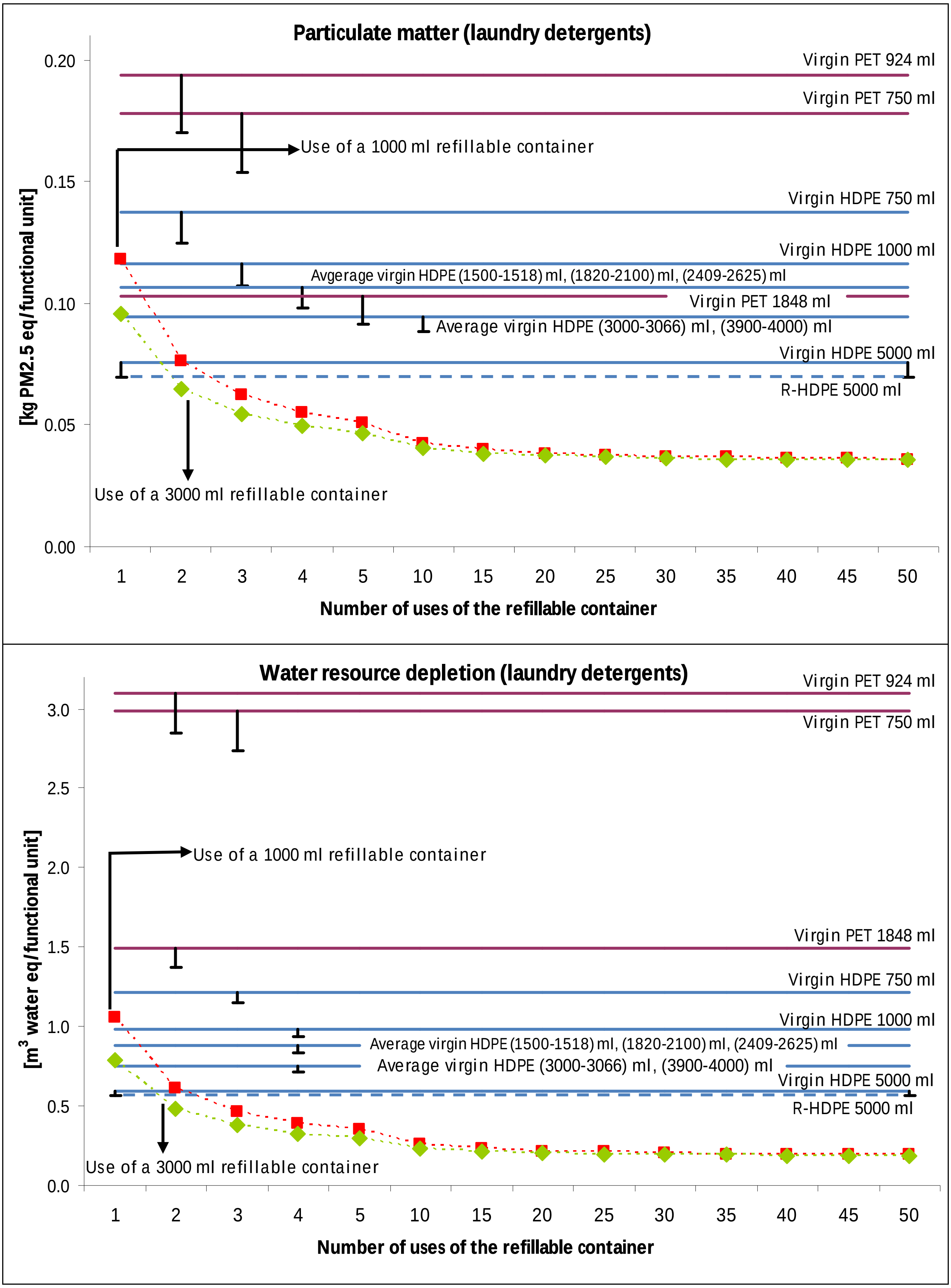


Figure B.8: *particulate matter* and *water resource depletion* impact indicators for laundry detergents. Horizontal lines represent the impacts of baseline scenarios, while squares and rhombuses the impacts of the two waste prevention scenarios as a function of the number of uses of the refillable container. Error bars represent the variation of the impacts when single-use containers are produced entirely from recycled material.

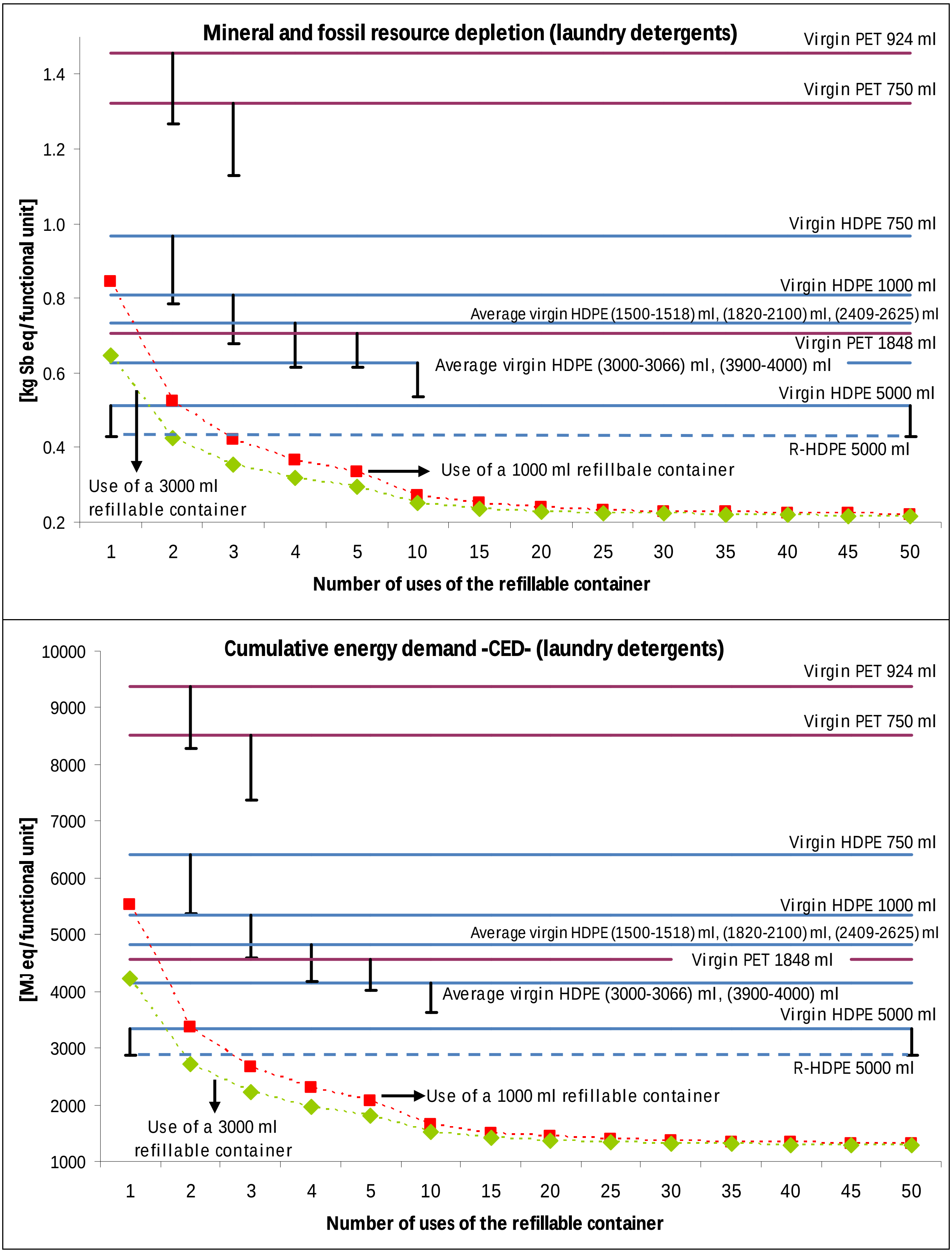


Figure B.9: *mineral and fossil resource depletion* and *cumulative energy demand* impact indicators for laundry detergents. Horizontal lines represent the impacts of baseline scenarios, while squares and rhombuses the impacts of the two waste prevention scenarios as a function of the number of uses of the refillable container. Error bars represent the variation of the impacts when single-use containers are produced entirely from recycled material.

Table B.19: percentage variation between the impacts of the 1000 ml-based prevention scenario for laundry detergents and those of the respective best baseline scenario for each category (i.e. the one based on 5000 ml single-use HDPE containers made from recycled material).

Impact category	**Number of uses of the 1000 ml refillable container**													
	1	**2**	**3**	**4**	**5**	**10**	**15**	**20**	**25**	**30**	**35**	**40**	**45**	**50**
Climate change	76.2	13.1	-8.0	-18.5	-24.9	-37.5	-41.7	-43.8	-45.1	-45.9	-46.5	-47.0	-47.3	-47.6
Ozone depletion	54.8	8.7	-6.7	-14.4	-19.0	-28.2	-31.3	-32.8	-33.8	-34.4	-34.8	-35.1	-35.4	-35.6
Photochemical ozone formation	44.3	9.0	-2.8	-8.7	-12.2	-19.2	-21.6	-22.8	-23.5	-24.0	-24.3	-24.5	-24.7	-24.9
Acidification	56.3	8.3	-7.7	-15.7	-20.5	-30.1	-33.3	-34.9	-35.9	-36.5	-37.0	-37.3	-37.6	-37.8
Terrestrial eutrophication	33.3	4.4	-5.2	-10.1	-12.9	-18.7	-20.6	-21.6	-22.2	-22.6	-22.8	-23.0	-23.2	-23.3
Freshwater eutrophication	81.8	3.1	-23.2	-36.3	-44.2	-59.9	-65.2	-67.8	-69.4	-70.4	-71.2	-71.7	-72.2	-72.5
Marine eutrophication	33.3	2.1	-8.3	-13.5	-16.6	-22.9	-24.9	-26.0	-26.6	-27.0	-27.3	-27.5	-27.7	-27.9
Freshwater ecotoxicity	102.9	34.7	11.9	0.6	-6.3	-19.9	-24.4	-26.7	-28.1	-29.0	-29.6	-30.1	-30.5	-30.8
Human toxicity (cancer effects)	110.4	54.3	35.6	26.2	20.6	9.4	5.6	3.8	2.6	1.9	1.4	1.0	0.6	0.4
Human toxicity (non-cancer effects)	72.2	29.1	14.7	7.5	3.2	-5.5	-8.3	-9.8	-10.6	-11.2	-11.6	-11.9	-12.2	-12.4
Particulate matter	69.1	9.0	-11.0	-21.0	-27.0	-39.0	-43.0	-45.0	-46.2	-47.0	-47.6	-48.0	-48.4	-48.6
Water resource depletion	86.4	8.5	-17.4	-30.4	-38.2	-53.7	-58.9	-61.5	-63.1	-64.1	-64.9	-65.4	-65.8	-66.2
Mineral and fossil resource depletion	96.4	22.4	-2.3	-14.7	-22.1	-36.9	-41.8	-44.3	-45.7	-46.7	-47.4	-48.0	-48.4	-48.7
Cumulative energy demand	92.5	17.5	-7.5	-20.0	-27.5	-42.5	-47.5	-50.0	-51.5	-52.5	-53.3	-53.8	-54.2	-54.5

B.3.2.2 Fabric softeners

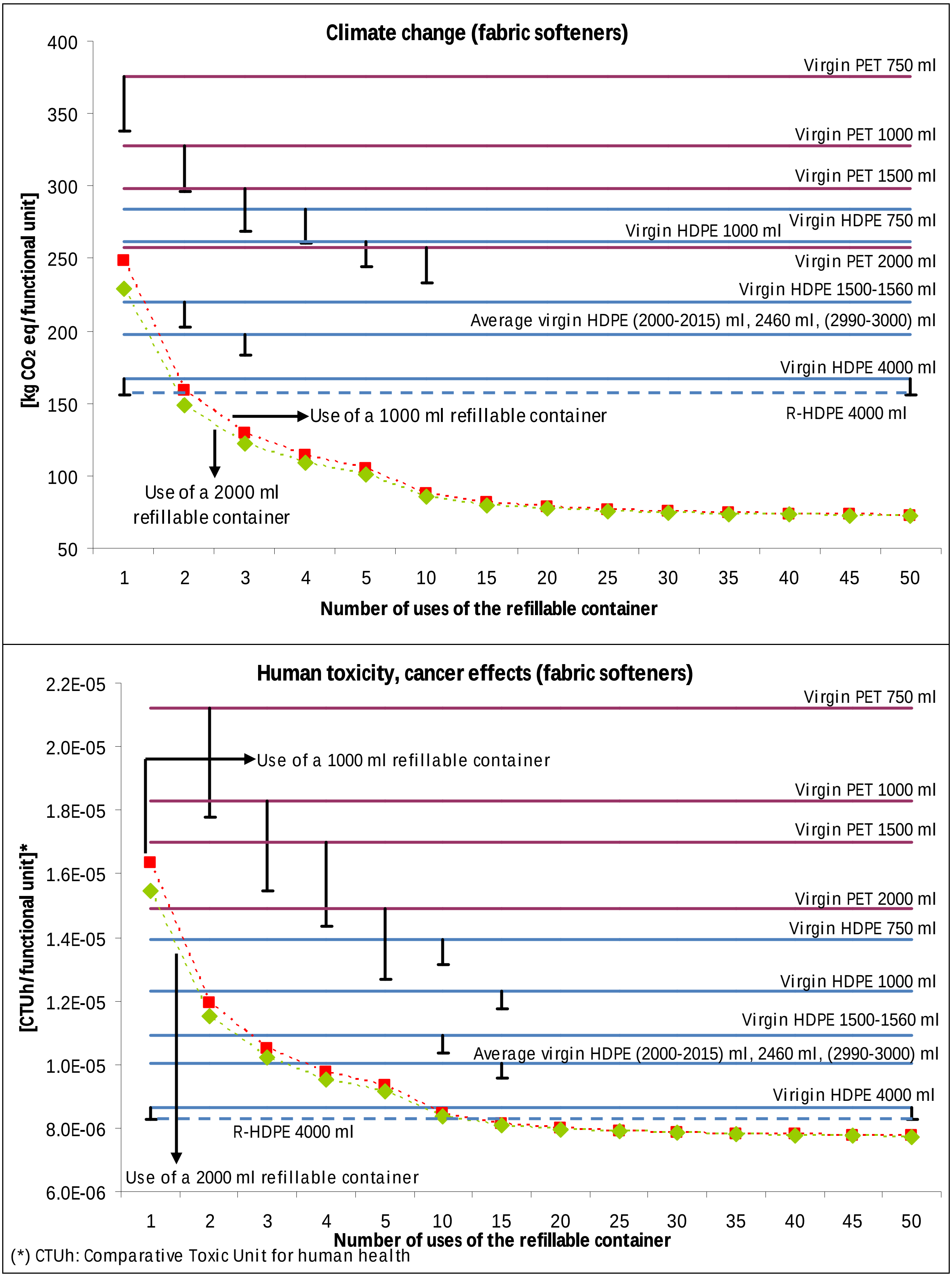


Figure B.10: *climate change* and *human toxicity, cancer effects* impact indicators for fabric softeners. Horizontal lines represent the impacts of baseline scenarios, while squares and rhombuses the impacts of the two waste prevention scenarios as a function of the number of uses of the refillable container. Error bars represent the variation of the impacts when single-use containers are produced entirely from recycled material.

Table B.20: potential impacts of baseline scenarios for fabric softeners. Values in parentheses refer to containers being produced entirely from recycled material (as considered in the sensitivity analysis).

Impact category	**Unit of measure**	**Scenario**										
		Distribution with single-use HDPE containers with a size of:							**Distribution with single-use PET containers with a size of:**			
		750 ml	**1000 ml**	**1500-1560 ml**	**2000-2015 ml**	**2990-3000 ml**	**2460 ml**	**4000 ml**	**750 ml**	**1000 ml**	**1500 ml**	**2000 ml**
Climate change	kg CO_2 eq.	284 (261)	262 (245)	220 (203)	201 (187)	200 (184)	193 (178)	168 (155)	376 (338)	328 (297)	298 (269)	257 (233)
Ozone depletion	kg CFC-11 eq.	2.87E-5 (2.87E-5)	2.85E-5 (2.85E-5)	2.28E-5 (2.28E-5)	2.12E-5 (2.12E-5)	2.02E-5 (2.02E-5)	2.00E-5 (2.00E-5)	1.76E-5 (1.76E-5)	7.68E-5 (7.49E-5)	6.65E-5 (6.48E-5)	6.04E-5 (5.89E-5)	5.15E-5 (5.02E-5)
Photochemical ozone formation	kg NMVOC eq.	1.18 (1.07)	1.09 (1.01)	0.986 (0.910)	0.938 (0.874)	0.945 (0.876)	0.919 (0.852)	0.848 (0.794)	1.31 (1.20)	1.17 (1.08)	1.11 (1.03)	1.04 (0.972)
Acidification	mol H^+ eq.	1.39 (1.30)	1.23 (1.17)	1.12 (1.05)	1.04 (0.984)	1.06 (1.00)	1.02 (0.963)	0.919 (0.871)	1.79 (1.61)	1.56 (1.41)	1.47 (1.33)	1.32 (1.20)
Terrestrial eutrophication	mol N eq.	3.83 (3.66)	3.55 (3.43)	3.25 (3.13)	3.14 (3.04)	3.19 (3.07)	3.07 (2.96)	2.88 (2.79)	4.37 (4.07)	3.92 (3.66)	3.77 (3.53)	3.58 (3.38)
Freshwater eutrophication	kg P eq.	0.0990 (0.0987)	0.0829 (0.0827)	0.0730 (0.0728)	0.0655 (0.0653)	0.0682 (0.0680)	0.0643 (0.0641)	0.0542 (0.0541)	0.172 (0.154)	0.145 (0.130)	0.134 (0.120)	0.113 (0.102)
Marine eutrophication	kg N eq.	0.396 (0.380)	0.369 (0.358)	0.330 (0.319)	0.321 (0.311)	0.327 (0.317)	0.310 (0.300)	0.290 (0.282)	0.455 (0.425)	0.403 (0.378)	0.388 (0.365)	0.365 (0.346)
Freshwater ecotoxicity	CTU_e	341 (309)	320 (297)	257 (234)	241 (222)	242 (222)	225 (205)	197 (181)	477 (384)	408 (331)	375 (303)	323 (264)
Human toxicity (cancer effects)	CTU_h	1.39E-5 (1.32E-5)	1.23E-5 (1.18E-5)	1.09E-5 (1.04E-5)	1.00E-5 (9.59E-6)	1.02E-5 (9.73E-6)	9.84E-6 (9.35E-6)	8.67E-6 (8.28E-6)	2.12E-5 (1.78E-5)	1.83E-5 (1.54E-5)	1.70E-5 (1.44E-5)	1.49E-5 (1.27E-5)
Human toxicity (non-cancer effects)	CTU_h	1.95E-5 (1.94E-5)	1.89E-5 (1.89E-5)	1.49E-5 (1.48E-5)	1.51E-5 (1.50E-5)	1.58E-5 (1.57E-5)	1.36E-5 (1.35E-5)	1.27E-5 (1.27E-5)	2.53E-5 (2.23E-5)	2.12E-5 (1.87E-5)	2.05E-5 (1.82E-5)	1.84E-5 (1.65E-5)
Particulate matter	kg $PM_{2,5}$ eq.	0.134 (0.122)	0.119 (0.111)	0.104 (0.0957)	0.0963 (0.0892)	0.0983 (0.0907)	0.0936 (0.0862)	0.0825 (0.0765)	0.163 (0.142)	0.141 (0.123)	0.131 (0.115)	0.115 (0.102)
Water resource depletion	m^3 water eq.	1.18 (1.11)	1.00 (0.955)	0.883 (0.836)	0.789 (0.750)	0.811 (0.769)	0.777 (0.736)	0.657 (0.625)	2.67 (2.44)	2.26 (2.07)	2.08 (1.91)	1.75 (1.61)
Mineral and fossil resource depletion	kg Sb eq.	0.954 (0.788)	0.857 (0.737)	0.733 (0.613)	0.664 (0.563)	0.664 (0.556)	0.645 (0.540)	0.554 (0.470)	1.21 (1.04)	1.05 (0.908)	0.955 (0.824)	0.824 (0.715)
Cumulative energy demand	MJ eq.	6312 (5365)	5613 (4930)	4810 (4126)	4361 (3787)	4394 (3776)	4239 (3636)	3637 (3154)	7756 (6771)	6717 (5892)	6130 (5367)	5299 (4666)

Table B.21: potential impacts of the two waste prevention scenarios for fabric softeners as a function of the number of uses of the refillable container.

Impact category	Unit of measure	Waste prevention scenario 1					Waste prevention scenario 2				
		Number of uses of the 1000 ml refillable container					Number of uses of the 2000 ml refillable container				
		1	**2**	**5**	**10**	**50**	**1**	**2**	**5**	**10**	**50**
Climate change	kg CO_2 eq.	248	159	105	87.2	72.9	229	149	101	85.2	72.5
Ozone depletion	kg CFC-11 eq.	2.49E-5	1.74E-5	1.29E-5	1.14E-5	1.02E-5	2.31E-5	1.65E-5	1.26E-5	1.13E-5	1.02E-5
Photochemical ozone formation	kg NMVOC eq.	1.11	0.839	0.674	0.619	0.575	1.07	0.816	0.664	0.614	0.573
Acidification	mol H^+ eq.	1.29	0.893	0.653	0.573	0.509	1.22	0.856	0.638	0.565	0.507
Terrestrial eutrophication	mol N eq.	3.62	2.82	2.35	2.19	2.06	3.55	2.79	2.33	2.18	2.06
Freshwater eutrophication	kg P eq.	0.0874	0.0495	0.0267	0.0192	0.0131	0.0802	0.0459	0.0253	0.0184	0.0130
Marine eutrophication	kg N eq.	0.354	0.270	0.220	0.203	0.190	0.356	0.271	0.220	0.204	0.190
Freshwater ecotoxicity	CTU_e	306	203	141	120	104	302	201	140	120	104
Human toxicity (cancer effects)	CTU_h	1.63E-5	1.19E-5	9.32E-6	8.45E-6	7.75E-6	1.55E-5	1.15E-5	9.15E-6	8.36E-6	7.73E-6
Human toxicity (non-cancer effects)	CTU_h	1.69E-5	1.26E-5	1.01E-5	9.21E-6	8.53E-6	1.93E-5	1.38E-5	1.06E-5	9.46E-6	8.58E-6
Particulate matter	kg $PM_{2,5}$ eq.	0.118	0.0755	0.0504	0.0420	0.0353	0.111	0.0723	0.0491	0.0413	0.0351
Water resource depletion	m^3 water eq.	1.05	0.612	0.348	0.260	0.190	0.956	0.564	0.329	0.250	0.188
Mineral and fossil resource depletion	kg Sb eq.	0.840	0.522	0.331	0.268	0.217	0.765	0.484	0.316	0.260	0.215
Cumulative energy demand	MJ eq.	5500	3348	2057	1627	1282	5028	3112	1963	1579	1273

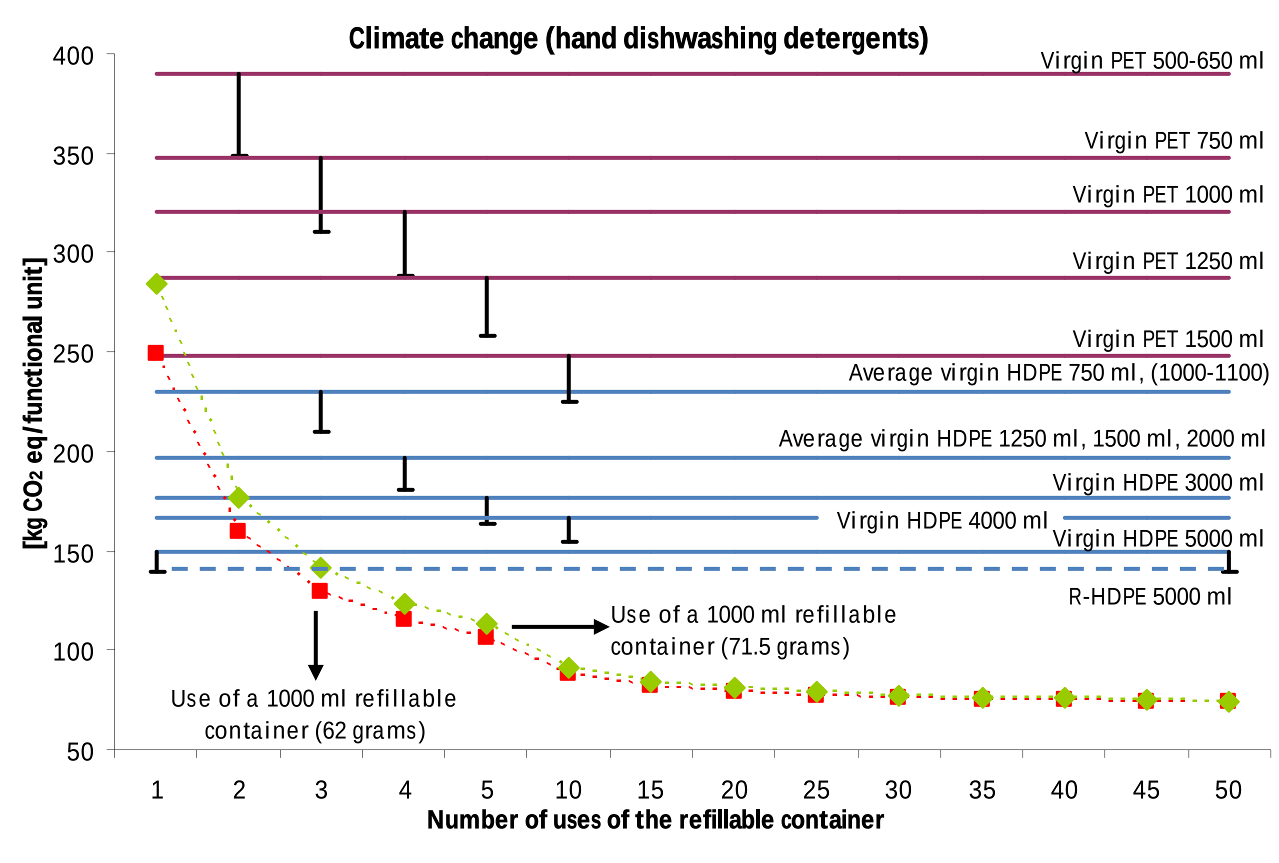
Table B.22: percentage variation between the impacts of the 2000 ml-based prevention scenario for fabric softeners and those of the respective best baseline scenario for each category (i.e. the one based on 4000 ml single-use HDPE containers made from recycled material).

Impact category	Number of uses of the 2000 ml refillable container													
	1	2	3	4	5	10	15	20	25	30	35	40	45	50
Climate change	47.2	-4.1	-21.2	-29.8	-34.9	-45.2	-48.6	-50.3	-51.3	-52.0	-52.5	-52.9	-53.1	-53.4
Ozone depletion	31.4	-6.0	-18.5	-24.7	-28.4	-35.9	-38.4	-39.7	-40.4	-40.9	-41.3	-41.5	-41.7	-41.9
Photochemical ozone formation	34.5	2.7	-7.9	-13.2	-16.3	-22.7	-24.8	-25.9	-26.5	-26.9	-27.2	-27.5	-27.6	-27.8
Acidification	39.9	-1.8	-15.6	-22.6	-26.7	-35.1	-37.9	-39.2	-40.1	-40.6	-41.0	-41.3	-41.6	-41.7
Terrestrial eutrophication	27.0	-0.2	-9.3	-13.8	-16.5	-21.9	-23.8	-24.7	-25.2	-25.6	-25.8	-26.0	-26.2	-26.3
Freshwater eutrophication	48.3	-15.2	-36.3	-46.9	-53.2	-65.9	-70.1	-72.2	-73.5	-74.3	-74.9	-75.4	-75.7	-76.0
Marine eutrophication	26.1	-3.9	-13.9	-18.9	-21.9	-27.9	-29.9	-30.9	-31.5	-31.9	-32.2	-32.4	-32.6	-32.7
Freshwater ecotoxicity	66.9	11.0	-7.7	-17.0	-22.6	-33.8	-37.5	-39.4	-40.5	-41.2	-41.8	-42.2	-42.5	-42.7
Human toxicity (cancer effects)	86.7	39.1	23.2	15.3	10.5	1.0	-2.2	-3.8	-4.7	-5.4	-5.8	-6.2	-6.4	-6.6
Human toxicity (non-cancer effects)	52.0	9.0	-5.4	-12.6	-16.9	-25.5	-28.4	-29.8	-30.7	-31.2	-31.6	-32.0	-32.2	-32.4
Particulate matter	45.1	-5.5	-22.4	-30.8	-35.9	-46.0	-49.4	-51.1	-52.1	-52.7	-53.2	-53.6	-53.9	-54.1
Water resource depletion	53.0	-9.7	-30.6	-41.1	-47.4	-59.9	-64.1	-66.2	-67.4	-68.3	-68.9	-69.3	-69.7	-70.0
Mineral and fossil resource depletion)	62.8	3.1	-16.8	-26.7	-32.7	-44.6	-48.6	-50.6	-51.8	-52.6	-53.2	-53.6	-53.9	-54.2
Cumulative energy demand	59.4	-1.3	-21.6	-31.7	-37.8	-49.9	-54.0	-56.0	-57.2	-58.0	-58.6	-59.0	-59.4	-59.6

Table B.23: percentage variation between the impacts of the 2000 ml-based prevention scenario for fabric softeners and those of the respective worst baseline scenario for each category (i.e. the one based on 750 ml single-use PET containers made from virgin material).

Impact category	Number of uses of the 2000 ml refillable container													
	1	2	3	4	5	10	15	20	25	30	35	40	45	50
Climate change	-39.1	-60.3	-67.4	-70.9	-73.1	-77.3	-78.7	-79.4	-79.9	-80.1	-80.3	-80.5	-80.6	-80.7
Ozone depletion	-70.0	-78.5	-81.4	-82.8	-83.6	-85.3	-85.9	-86.2	-86.4	-86.5	-86.6	-86.6	-86.7	-86.7
Photochemical ozone formation	-18.2	-37.6	-44.0	-47.2	-49.1	-53.0	-54.3	-54.9	-55.3	-55.6	-55.8	-55.9	-56.0	-56.1
Acidification	-31.9	-52.1	-58.9	-62.3	-64.3	-68.4	-69.7	-70.4	-70.8	-71.1	-71.3	-71.4	-71.5	-71.6
Terrestrial eutrophication	-18.9	-36.2	-42.0	-44.9	-46.6	-50.1	-51.3	-51.9	-52.2	-52.4	-52.6	-52.7	-52.8	-52.9
Freshwater eutrophication	-53.3	-73.3	-79.9	-83.3	-85.3	-89.3	-90.6	-91.3	-91.7	-91.9	-92.1	-92.3	-92.4	-92.5
Marine eutrophication	-21.8	-40.4	-46.6	-49.7	-51.6	-55.3	-56.5	-57.2	-57.5	-57.8	-58.0	-58.1	-58.2	-58.3
Freshwater ecotoxicity	-36.7	-57.9	-65.0	-68.5	-70.6	-74.9	-76.3	-77.0	-77.4	-77.7	-77.9	-78.1	-78.2	-78.3
Human toxicity (cancer effects)	-27.1	-45.7	-51.9	-55.0	-56.9	-60.6	-61.8	-62.4	-62.8	-63.1	-63.2	-63.4	-63.5	-63.6
Human toxicity (non-cancer effects)	-23.9	-45.4	-52.6	-56.2	-58.4	-62.7	-64.1	-64.8	-65.3	-65.6	-65.8	-65.9	-66.0	-66.1
Particulate matter	-31.7	-55.6	-63.5	-67.5	-69.8	-74.6	-76.2	-77.0	-77.5	-77.8	-78.0	-78.2	-78.3	-78.4
Water resource depletion	-64.2	-78.9	-83.8	-86.2	-87.7	-90.6	-91.6	-92.1	-92.4	-92.6	-92.7	-92.8	-92.9	-93.0
Mineral and fossil resource depletion	-36.6	-59.8	-67.6	-71.4	-73.8	-78.4	-80.0	-80.7	-81.2	-81.5	-81.7	-81.9	-82.0	-82.1
Cumulative energy demand	-35.2	-59.9	-68.1	-72.2	-74.7	-79.6	-81.3	-82.1	-82.6	-82.9	-83.2	-83.3	-83.5	-83.6

B.3.2.3 Hand dishwashing detergents



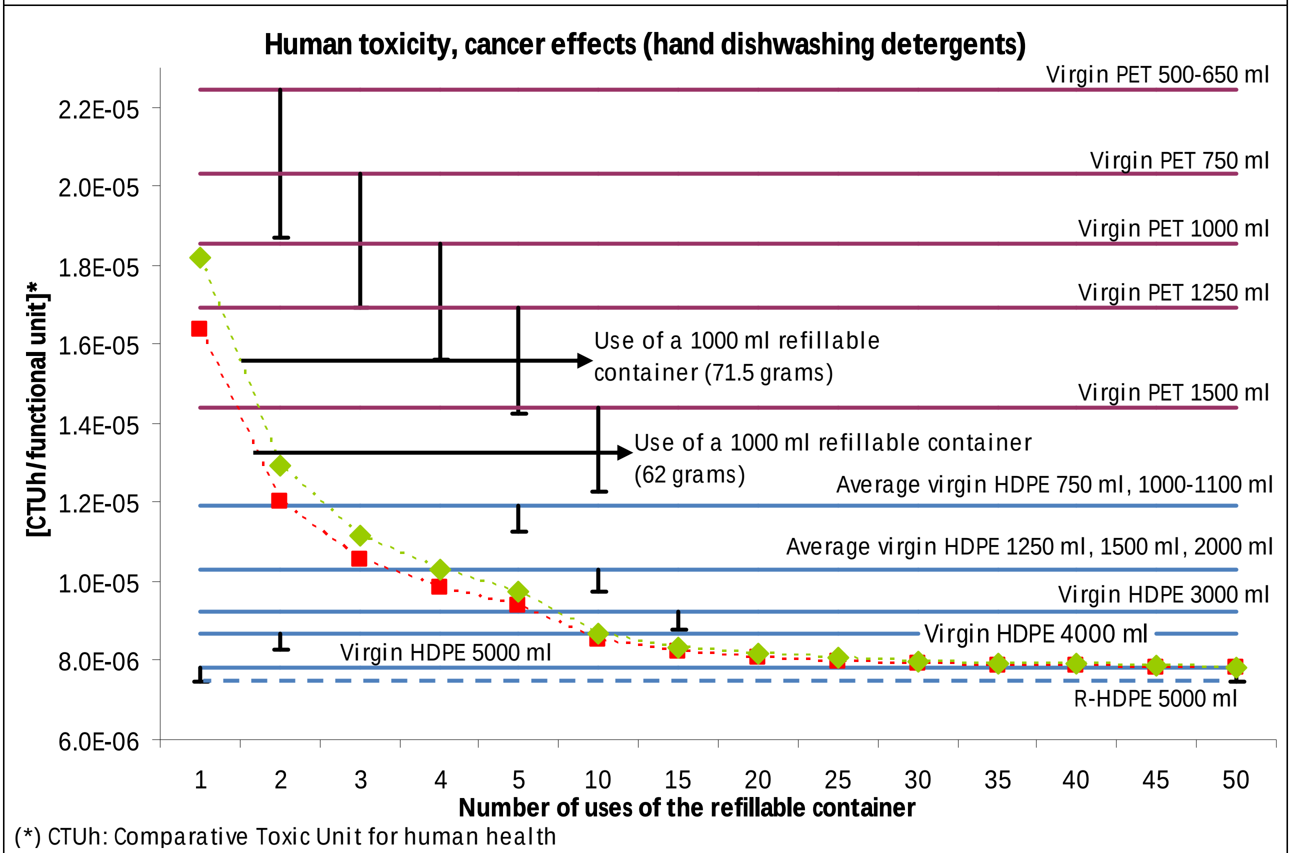


Figure B.11: *climate change* and *human toxicity, cancer effects* impact indicators for hand dishwashing detergents. Horizontal lines represent the impacts of baseline scenarios, while squares and rhombuses the impacts of the two waste prevention scenarios as a function of the number of uses of the refillable container. Error bars represent the variation of the impacts when single-use containers are produced entirely from recycled material.

Table B.24: potential impacts of baseline scenarios for hand dishwashing detergents. Values in parentheses refer to containers being produced entirely from recycled material (as considered in the sensitivity analysis).

Impact category	**Unit of measure**	**Scenario**												
		Distribution with single-use HDPE containers with a size of:								**Distribution with single-use PET containers with a size of:**				
		750 ml	**1000-1100 ml**	**1250 ml**	**1500 ml**	**2000 ml**	**3000 ml**	**4000 ml**	**5000 ml**	**500-650 ml**	**750 ml**	**1000 ml**	**1250 ml**	**1500 ml**
Climate change	kg CO_2 eq.	234 (212)	227 (206)	203 (185)	197 (180)	191 (177)	177 (163)	166 (154)	150 (140)	389 (349)	348 (311)	320 (288)	288 (258)	248 (225)
Ozone depletion	kg CFC-11 eq.	2.26E-5 (2.26E-5)	2.18E-5 (2.18E-5)	1.99E-5 (1.99E-5)	1.97E-5 (1.97E-5)	1.93E-5 (1.93E-5)	1.83E-5 (1.83E-5)	1.73E-5 (1.73E-5)	1.61E-5 (1.61E-5)	8.09E-5 (7.87E-5)	7.32E-5 (7.13E-5)	6.56E-5 (6.39E-5)	5.94E-5 (5.78E-5)	4.95E-5 (4.83E-5)
Photochemical ozone formation	kg NMVOC eq.	1.05 (0.957)	1.06 (0.966)	0.970 (0.891)	0.973 (0.901)	0.929 (0.865)	0.898 (0.838)	0.848 (0.794)	0.796 (0.751)	1.36 (1.24)	1.24 (1.14)	1.18 (1.09)	1.12 (1.03)	1.00 (0.932)
Acidification	mol H^+ eq.	1.23 (1.14)	1.22 (1.14)	1.10 (1.03)	1.08 (1.02)	1.04 (0.978)	0.976 (0.922)	0.921 (0.872)	0.842 (0.802)	1.88 (1.69)	1.72 (1.54)	1.59 (1.43)	1.47 (1.33)	1.28 (1.16)
Terrestrial eutrophication	mol N eq.	3.48 (3.33)	3.54 (3.39)	3.26 (3.13)	3.29 (3.17)	3.16 (3.06)	3.05 (2.95)	2.89 (2.80)	2.73 (2.65)	4.57 (4.24)	4.19 (3.89)	4.03 (3.77)	3.82 (3.58)	3.45 (3.26)
Freshwater eutrophication	kg P eq.	0.0843 (0.0840)	0.0824 (0.0821)	0.0720 (0.0718)	0.0675 (0.0673)	0.0656 (0.0654)	0.0581 (0.0579)	0.0545 (0.0543)	0.0467 (0.0466)	0.184 (0.165)	0.166 (0.148)	0.149 (0.133)	0.133 (0.119)	0.111 (0.0996)
Marine eutrophication	kg N eq.	0.356 (0.342)	0.363 (0.350)	0.331 (0.319)	0.332 (0.322)	0.326 (0.317)	0.306 (0.297)	0.292 (0.284)	0.272 (0.266)	0.477 (0.444)	0.432 (0.402)	0.418 (0.392)	0.391 (0.367)	0.355 (0.336)
Freshwater ecotoxicity	CTU_e	278 (249)	274 (247)	240 (217)	233 (211)	237 (217)	207 (189)	197 (181)	174 (160)	497 (397)	436 (343)	409 (329)	361 (288)	316 (258)
Human toxicity (cancer effects)	CTU_h	1.20E-5 (1.13E-5)	1.19E-5 (1.12E-5)	1.06E-5 (1.00E-5)	1.03E-5 (9.82E-6)	9.90E-6 (9.44E-6)	9.23E-6 (8.80E-6)	8.66E-6 (8.27E-6)	7.81E-6 (7.49E-6)	2.24E-5 (1.87E-5)	2.03E-5 (1.69E-5)	1.86E-5 (1.56E-5)	1.69E-5 (1.43E-5)	1.44E-5 (1.23E-5)
Human toxicity (non-cancer effects)	CTU_h	1.68E-5 (1.67E-5)	1.75E-5 (1.74E-5)	1.51E-5 (1.51E-5)	1.48E-5 (1.47E-5)	1.62E-5 (1.62E-5)	1.32E-5 (1.32E-5)	1.31E-5 (1.30E-5)	1.15E-5 (1.15E-5)	2.68E-5 (2.35E-5)	2.29E-5 (1.98E-5)	2.29E-5 (2.03E-5)	2.00E-5 (1.76E-5)	1.88E-5 (1.69E-5)
Particulate matter	kg $PM_{2,5}$ eq.	0.115 (0.105)	0.114 (0.104)	0.102 (0.0928)	0.0979 (0.0900)	0.0957 (0.0886)	0.0873 (0.0807)	0.0827 (0.0767)	0.0741 (0.0691)	0.171 (0.148)	0.153 (0.132)	0.143 (0.124)	0.129 (0.113)	0.113 (0.0995)
Water resource depletion	m^3 water eq.	0.998 (0.941)	0.969 (0.914)	0.855 (0.807)	0.805 (0.761)	0.775 (0.735)	0.701 (0.664)	0.657 (0.624)	0.572 (0.544)	2.86 (2.61)	2.60 (2.37)	2.30 (2.11)	2.08 (1.90)	1.71 (1.56)
Mineral and fossil resource depletion	kg Sb eq.	0.790 (0.643)	0.766 (0.625)	0.685 (0.561)	0.659 (0.546)	0.634 (0.533)	0.587 (0.494)	0.551 (0.466)	0.493 (0.422)	1.25 (1.07)	1.12 (0.955)	1.03 (0.883)	0.926 (0.793)	0.793 (0.687)
Cumulative energy demand	MJ eq.	5249 (4406)	5113 (4303)	4542 (3834)	4369 (3725)	4203 (3624)	3865 (3331)	3616 (3130)	3211 (2807)	8093 (7018)	7230 (6241)	6637 (5782)	5965 (5186)	5102 (4486)

Table B.25: potential impacts of the two waste prevention scenarios for hand dishwashing detergents as a function of the number of uses of the refillable container.

Impact category	Unit of measure	Waste prevention scenario 1					Waste prevention scenario 2				
		Number of uses of the 1000 ml refillable container (62 grams)					Number of uses of the 1000 ml refillable container (71,5 grams)				
		1	**2**	**5**	**10**	**50**	**1**	**2**	**5**	**10**	**50**
Climate change	kg CO_2 eq.	249	160	106	88.2	73.9	284	177	113	91.7	74.6
Ozone depletion	kg CFC-11 eq.	2.51E-5	1.76E-5	1.31E-5	1.16E-5	1.04E-5	2.76E-5	1.89E-5	1.36E-5	1.19E-5	1.05E-5
Photochemical ozone formation	kg NMVOC eq.	1.13	0.849	0.684	0.629	0.585	1.24	0.908	0.707	0.640	0.587
Acidification	mol H^+ eq.	1.30	0.901	0.661	0.581	0.517	1.47	0.987	0.695	0.598	0.520
Terrestrial eutrophication	mol N eq.	3.65	2.86	2.38	2.22	2.10	4.06	3.06	2.46	2.26	2.11
Freshwater eutrophication	kg P eq.	0.0875	0.0496	0.0268	0.0193	0.0132	0.104	0.0579	0.0302	0.0209	0.0135
Marine eutrophication	kg N eq.	0.358	0.274	0.224	0.207	0.193	0.409	0.300	0.234	0.212	0.194
Freshwater ecotoxicity	CTU_e	307	204	142	121	105	365	233	153	127	106
Human toxicity (cancer effects)	CTU_h	1.64E-5	1.20E-5	9.39E-6	8.51E-6	7.81E-6	1.82E-5	1.29E-5	9.75E-6	8.69E-6	7.85E-6
Human toxicity (non-cancer effects)	CTU_h	1.69E-5	1.27E-5	1.01E-5	9.29E-6	8.61E-6	2.19E-5	1.52E-5	1.11E-5	9.78E-6	8.71E-6
Particulate matter	kg $PM_{2.5}$ eq.	0.118	0.0761	0.0509	0.0425	0.0358	0.138	0.0858	0.0548	0.0445	0.0362
Water resource depletion	m^3 water eq.	1.05	0.614	0.349	0.261	0.191	1.23	0.703	0.385	0.279	0.194
Mineral and fossil resource depletion	kg Sb eq.	0.843	0.525	0.334	0.271	0.220	0.963	0.585	0.358	0.283	0.222
Cumulative energy demand	MJ eq.	5518	3366	2075	1644	1300	6368	3791	2245	1729	1317

Table B.26: percentage variation between the impacts of the best prevention scenario[1] for hand dishwashing detergents and those of the respective baseline scenario with lowest impacts (i.e. the one based on 5000 ml single-use HDPE containers made from recycled material).

Impact category	Number of uses of the 1000 ml refillable container (62 grams)													
	1	**2**	**3**	**4**	**5**	**10**	**15**	**20**	**25**	**30**	**35**	**40**	**45**	**50**
Climate change	78.3	14.3	-7.0	-17.7	-24.1	-36.9	-41.1	-43.3	-44.5	-45.4	-46.0	-46.5	-46.8	-47.1
Ozone depletion	56.0	9.5	-6.0	-13.8	-18.5	-27.8	-30.9	-32.4	-33.4	-34.0	-34.4	-34.8	-35.0	-35.2
Photochemical ozone formation	49.9	13.1	0.9	-5.2	-8.9	-16.2	-18.7	-19.9	-20.7	-21.1	-21.5	-21.8	-22.0	-22.1
Acidification	62.3	12.4	-4.3	-12.6	-17.6	-27.6	-30.9	-32.6	-33.6	-34.2	-34.7	-35.1	-35.3	-35.6
Terrestrial eutrophication	37.6	7.7	-2.2	-7.2	-10.2	-16.2	-18.2	-19.2	-19.8	-20.2	-20.4	-20.7	-20.8	-21.0
Freshwater eutrophication	87.8	6.5	-20.6	-34.2	-42.3	-58.6	-64.0	-66.8	-68.4	-69.5	-70.2	-70.8	-71.3	-71.6
Marine eutrophication	34.6	3.0	-7.5	-12.7	-15.9	-22.2	-24.3	-25.4	-26.0	-26.4	-26.7	-26.9	-27.1	-27.3
Freshwater ecotoxicity	91.5	27.1	5.6	-5.1	-11.6	-24.5	-28.8	-30.9	-32.2	-33.1	-33.7	-34.1	-34.5	-34.8
Human toxicity (cancer effects)	118.9	60.5	41.0	31.2	25.4	13.7	9.8	7.8	6.7	5.9	5.3	4.9	4.6	4.3
Human toxicity (non-cancer effects)	47.8	10.7	-1.6	-7.8	-11.5	-18.9	-21.4	-22.6	-23.4	-23.9	-24.2	-24.5	-24.7	-24.9
Particulate matter	70.8	10.1	-10.1	-20.3	-26.3	-38.5	-42.5	-44.5	-45.8	-46.6	-47.1	-47.6	-47.9	-48.2
Water resource depletion	93.6	12.7	-14.2	-27.7	-35.8	-52.0	-57.4	-60.1	-61.7	-62.8	-63.5	-64.1	-64.6	-64.9
Mineral and fossil resource depletion	99.8	24.4	-0.7	-13.2	-20.8	-35.9	-40.9	-43.4	-44.9	-45.9	-46.6	-47.2	-47.6	-47.9
Cumulative energy demand	96.5	19.9	-5.7	-18.4	-26.1	-41.4	-46.5	-49.1	-50.6	-51.7	-52.4	-52.9	-53.4	-53.7

Table B.27: percentage variation between the impacts of the best prevention scenario[1] for hand dishwashing detergents and those of the respective baseline scenario with highest impacts (i.e. the one based on 500-650 ml single-use PET containers made from virgin material).

Impact category	Number of uses of the 1000 ml refillable container (62 grams)													
	1	**2**	**3**	**4**	**5**	**10**	**15**	**20**	**25**	**30**	**35**	**40**	**45**	**50**
Climate change	-36.0	-59.0	-66.6	-70.5	-72.8	-77.3	-78.9	-79.6	-80.1	-80.4	-80.6	-80.8	-80.9	-81.0
Ozone depletion	-69.0	-78.2	-81.3	-82.9	-83.8	-85.6	-86.3	-86.6	-86.7	-86.9	-87.0	-87.0	-87.1	-87.1
Photochemical ozone formation	-17.2	-37.5	-44.2	-47.6	-49.6	-53.7	-55.1	-55.7	-56.1	-56.4	-56.6	-56.7	-56.9	-56.9
Acidification	-30.9	-52.2	-59.3	-62.8	-64.9	-69.2	-70.6	-71.3	-71.7	-72.0	-72.2	-72.4	-72.5	-72.6
Terrestrial eutrophication	-20.1	-37.4	-43.2	-46.1	-47.8	-51.3	-52.5	-53.0	-53.4	-53.6	-53.8	-53.9	-54.0	-54.1
Freshwater eutrophication	-52.5	-73.1	-79.9	-83.3	-85.4	-89.5	-90.9	-91.6	-92.0	-92.3	-92.5	-92.6	-92.7	-92.8
Marine eutrophication	-24.9	-42.5	-48.4	-51.3	-53.1	-56.6	-57.8	-58.4	-58.7	-59.0	-59.1	-59.3	-59.4	-59.4
Freshwater ecotoxicity	-38.2	-59.0	-65.9	-69.4	-71.5	-75.6	-77.0	-77.7	-78.1	-78.4	-78.6	-78.8	-78.9	-79.0
Human toxicity (cancer effects)	-27.0	-46.5	-53.0	-56.2	-58.2	-62.1	-63.4	-64.0	-64.4	-64.7	-64.9	-65.0	-65.1	-65.2
Human toxicity (non-cancer effects)	-36.7	-52.6	-57.9	-60.5	-62.1	-65.3	-66.4	-66.9	-67.2	-67.4	-67.6	-67.7	-67.8	-67.8
Particulate matter	-31.0	-55.5	-63.7	-67.8	-70.2	-75.1	-76.8	-77.6	-78.1	-78.4	-78.6	-78.8	-79.0	-79.1
Water resource depletion	-63.2	-78.5	-83.7	-86.2	-87.8	-90.9	-91.9	-92.4	-92.7	-92.9	-93.1	-93.2	-93.3	-93.3
Mineral and fossil resource depletion	-32.8	-58.1	-66.6	-70.8	-73.3	-78.4	-80.1	-81.0	-81.5	-81.8	-82.0	-82.2	-82.4	-82.5
Cumulative energy demand	-31.8	-58.4	-67.3	-71.7	-74.4	-79.7	-81.5	-82.3	-82.9	-83.2	-83.5	-83.7	-83.8	-83.9

[1] The waste prevention scenario where a 1000 ml refillable container weighing 62 grams is provided to the consumer shows the lowest impact for all the considered impact categories.

Appendix C

This appendix supports the discussion of the methodological approaches presented in Section 4 for the incorporation of waste prevention activities into life cycle assessment of municipal solid waste management systems. Specifically, it provides numerous examples of waste streams removed from and added to the municipal waste management system by the most common types of waste prevention activities (reduction in product or service consumption, product/service substitution, reuse and lifespan extension; see Section 1.3 for details). The examples are listed in Table C.1, which is based on the comprehensive review of municipal waste prevention activities reported in Table 1.1.

Table C.1: Examples of waste streams removed from and added to the municipal waste management system by different types of waste prevention activities.

Type of waste prevention activities		Examples	Prevented MSW[a] (prevented waste goods or packages)	Substitutive goods or packages generated as additional MSW[a] to be managed
Reduction in the consumption of goods or services	1) Reduction in the consumption of goods by citizens, companies or organisations (without reducing the consumption of the service originally provided by those goods)	- Reducing paper consumption through double-sided printing and copying (and other good practices) - Renting or borrowing/lending of goods instead of purchasing new ones (e.g. infrequently used clothes and textiles, office furniture, toys, books, home and garden tools, party/event decorations and supplies, paints etc.)	- White graphical paper (in an mount depending on the number of double-sided printed documents) - New equivalent finished products (in an amount depending on the number of rent or borrowed/lent products)	- None - None
	2) Reduction in the wastage of goods (unnecessary to the consumer)	- Reducing household food waste (unconsumed or partially consumed food and leftovers) by improving one's own purchasing and storage behaviour, avoiding leftovers etc. - Reducing retail food waste by donating still edible, but no longer sellable food, to social canteens, social supermarkets or other social welfare services intended for people in need - Reducing the delivery of unsolicited mail such as unaddressed advertising material by applying dissuasive stickers on mailboxes, subscribing to mail preference services etc.	- Food waste (in an amount depending on the number of people that will change their behaviour and the specific generation of food waste) - Food waste (in an amount depending on the quantity of donated food actually consumed) - Printed paper and/or brochures delivered by post to households (in an amount depending on the number of households participating in the initiative and the specific generation of unsolicited mail)	- None - None - None

Substitution of a product or service by a less waste-generating equivalent one	3) Reducing the amount of material used for the manufacturing or packaging of a good through a more efficient design (without reducing product performance)[b]	- Reducing the amount of steel used to manufacture a washing machine
		- Reduction in the amount of packaging material used per unit mass of packaged product, like:
		  - lightweighting of beverage bottles (without reducing their strength)
		  - increasing volume capacity of containers
			- One or more heavier washing machines depending on the number of lighter ones that will be used
			- Heavier beverage bottles (in an amount depending on the volume of beverage packaged in lighter bottles that will be consumed)
			- Smaller containers (in an amount depending on the amount of product packaged in bigger containers that will be consumed)
				- One or more lighter washing machines
				- Lighter beverage bottles
				- Bigger containers

	4) Substitution of an unpacked good for a packed one	- Drinking of (refined) public network water from the tap or public fountains/suppliers instead of bottled water	- Plastic or glass water bottles (in an amount proportional to the volume of public network water that will be consumed for drinking purposes)	- One or more reusable jugs or bottles

Table 1.1 (continued)

Type of waste prevention activities		Examples	Prevented MSW[a] (prevented waste goods or packages)	Substitutive goods or packages generated as additional MSW[a]
Substitution of a product or service by a less waste-generating equivalent one (continued)	5) Substitution of a reusable good or a good provided in a reusable packaging for a disposable good or a good provided in a disposable packaging	- Packaging of water or other beverages in refillable bottles rather than in one-way bottles	- Plastic or glass one-way bottles (in an amount proportional to the volume of beverage packed in refillable bottles)	- Refillable glass or plastic bottles
		- Distribution of liquid detergents through self-dispensing systems rather than packed in single-use containers	- Single-use plastic containers (in an amount proportional to the volume of detergent withdrawn from self-dispensing systems)	- Refillable plastic containers
		- Distribution of ‘loose’ dry food products through gravity dispensers rather than individually packaged	- Disposable packages (in an amount proportional to the quantity of dry food products purchased ‘loose’)	- Lightweight (reusable) plastic or paper bags
		- Delivery of local, unpacked, fruit and vegetable products to the households, by means of returnable crates	- Disposable packages (in an amount proportional to the quantity of products delivered to the households)	- Returnable crates
		- Shipment of goods by means of returnable cardboard boxes rather than disposable ones	- Disposable cardboard boxes (in an amount proportional to the number of shipments performed with the same reusable boxes)	- Reusable cardboard boxes
		- Use of reusable shopping bags rather than disposable plastic or paper ones	- Disposable shopping bags (in an amount proportional to the number of purchasing activities performed with the same reusable bag)	- Reusable shopping bags
		- Drying of hands by means of electric hand-dryers rather than paper bath-towels	- Paper bath-towels (in an amount proportional to the number of hand pairs dried with electric hand-dryers)	- One or more electric hand-dryers
		- Serving meals with reusable crockery rather than disposable ones	- Disposable crockery (in an amount proportional to the number of meals served with reusable crockery)	- Reusable crockery
		- Swaddling babies in reusable nappies rather than disposable ones	- Disposable nappies (in an amount proportional to the number of children swaddled in reusable nappies)	- Reusable nappies
	6) Substitution of a digital good for a disposable one	- Substitution of internet advertising brochures for printed ones by retailers	- Printed brochures (in an amount depending on the number of brochures that would traditionally be delivered to the households or made available at retail establishments)	- None[c]
		- Reading of on-line newspapers instead of printed ones	- Printed newspapers (in an amount depending on the number of newspapers that will be read on the internet)	- None[c]
		- Digitalisation of documentation and bureaucratic procedures in companies, organisations and public administrations	- Printed documents (in an amount depending on the number of documents that will be digitalised)	- None[c]

Table 1.1 (continued)

Type of waste prevention activities		**Examples**	**Prevented MSW[a] (prevented waste goods or packages)**	**Substitutive goods or packages generated as additional MSW[a]**
Reuse of goods	7) Direct reuse of disposable goods or packages by the owner (private citizens or organisations) in substitution of disposable or durable goods or packages	- Reuse of a disposable shopping bag, of a disposable glass jar, of a one-way glass or plastic bottle etc.	- Identical new disposable goods (e.g. shopping bags) in an amount depending on the number of reused disposable goods and the number of times they are reused, or equivalent new durable goods (e.g. jars) in an amount depending on the number of reused disposable goods and the ratio between the duration of their second life and the average lifespan of equivalent new goods	- None
	8) Reuse of durable goods through second-hand retailing/purchasing, donations and exchanges	- Selling/purchase in second-hand markets, donation to charities and people in need or exchange of durable goods such as clothes and textiles, furniture, electrical and electronic equipment, toys, books, bicycles, sport and fitness equipment, baby and nursery products and accessories, home and garden tools, party/event decorations and supplies etc.	- Equivalent new durable goods (in an amount depending on the number of reused goods and the ratio between the duration of their second life and the average lifespan of equivalent new goods)	- None
Extension of the lifespan of durable goods	9) Extension of the lifespan of existing durable goods by citizens or repair centres	- Repairing of durable goods by citizens or repair centres (e.g. clothes and textiles, furniture, electrical and electronic equipment, bicycles, sport equipment, home and garden tools etc.) - Keeping appliances in a good working order by following manufacturers' recommendations for a proper operation and maintenance	- Equivalent new durable goods (in an amount depending on the number of repaired goods and the ratio between the duration of their second life and the average lifespan of equivalent new goods) - Equivalent new durable goods (in an amount depending on the number of goods the lifespan of which has been extended and the ratio between the duration of their additional life and the average lifespan of equivalent new goods)	- None - None
	10) Extension of the useful life of durable goods by producers	- Extension of the useful life of domestic appliances through a more efficient design	- Shorter-lasting domestic appliances (in an amount depending on the number of longer-lasting domestic appliances used and on the ratio between the duration of their lifespan and that of substituted shorter-lasting domestic appliances)	- Longer-lasting domestic appliances actually used

(a) MSW: municipal solid waste.
(b) e.g. the amount of packaged product damaged or lost is not increased.
(c) Any additional waste is not generated in place of substituted disposable good(s) only under the assumption that the electronic devices required to use the substitutive digital goods are already owned by the citizens or organizations participating in the prevention activities. Nevertheless, even in this case, a portion of the life cycle of such devices and of the associated MSW generation may be allocated to the use of substitutive digital goods.

Appendix D

This appendix provides further details on the life cycle assessment (LCA) case study summarised in Section 5, dealing with the prevention and management of municipal solid waste in Lombardia, Italy. In particular, the procedure used for the calculation of the quantity of waste removed from and added to the waste management system by the examined waste prevention activities is described (Section D.1) and additional results are provided (Section D.2).

D.1 Estimate of the avoided and additional waste flows

The following tables describe the procedure used for the calculation of the quantities of waste removed from and added to the waste management system by the examined waste prevention activities. In particular Tables D.1 and D.2 refer to the bottled water substitution, while Tables D.3 to D.7 to the substitution of single-use packaged liquid detergents. In this case, the estimate of the avoided waste is separately reported for each of the four categories of detergent involved in the substitution (automatic and hand wash laundry detergents, fabric softeners and hand dishwashing detergents).

Table D.1: estimate of the mass of waste avoided (removed from the waste management system) with the substitution of bottled water by public network water.

Avoided consumption of (one-way PET) bottled water by bottle size[a]			Types and quantities of products removed from the waste stream		Average masses of the products removed from the waste stream[b] [grams]					Avoided waste flows [tonnes]				
Bottle size [litres]	**Consumption [litres]**	**%**	**Bottles/ caps/ labels[c]**	**Heat-shrink wraps**	**Bottles**	**Caps**	**Plastic labels[d]**	**Paper labels[d]**	**Heat-shrink wraps**	**Bottles (PET)**	**Caps (HDPE)**	**Plastic labels (PP)**	**Paper labels**	**Heat-shrink wraps (LDPE)**
1	70 $\times 10^6$	5.9	70$\times 10^6$	12 $\times 10^6$	28.0	1.62	0.62	1.38	17.5	1,960	113	22	48	204
1.5	989 $\times 10^6$	83.2	659 $\times 10^6$	110 $\times 10^6$	32.6	2.06	0.57	1.25	21.8	21,460	1,358	188	413	2,395
2	129 $\times 10^6$	10.9	65 $\times 10^6$	11 $\times 10^6$	33.4	1.72	0.52	1.14	26.0	2,162	111	17	37	280
Total	1,188 $\times 10^6$	100	794 $\times 10^6$	132 $\times 10^6$	-	-	-	-	-	25,581	1,583	226	499	2,880
Total avoided waste: 30,769 tonnes														

(a) Estimated based on 2013 volume sales of bottled water by type and size of bottle in Italy (data acquired from the Passport database by Euromonitor International).
(b) Based on the experimental estimates reported in Federambiente (2010) (1.5 and 2 litre bottles) and on experimental estimates by the author (1 litre bottles).
(c) The same number of bottles, caps and labels is avoided.
(d) 50% of avoided labels were assumed to be made out of plastic (polypropylene) and the other 50% from paper.

Table D.2: estimate of the mass of additional waste resulting form the substitution of bottled water by public network water.

Additional consumption of public network water [litres]	Types of products added to the waste stream	Quantities of the products added to the waste stream	Average masses of the products added to the waste stream [grams]	Additional waste flows [tonnes]
1,188 $\times 10^6$	Reusable glass jugs (1 litre)	11,882,955[a]	475[b]	5,644[a]

(a) The number (and the mass) of glass jugs generated as additional waste to be managed is calculated by conservatively assuming that a single jug is used for 100 times overall.
(b) Assumption based on the average mass of 1 litre refillable glass bottles used for bottled water distribution in Italy (estimated experimentally).

Table D.3: estimate of the mass of waste avoided (removed from the waste management system) with the substitution of automatic liquid laundry detergents packaged in single-use containers by those distributed loose through self-dispensing systems and refillable containers.

Avoided consumption of automatic liquid laundry detergents by type and size of container[a]				Average washing performance		Average specific masses of the products removed from the waste stream[d] [g/litre of detergent]		Avoided waste flows [tonnes]		
Container material	**Container size [ml]**	**Consumption [litres]**	**%**	**Washings per litre**	**Washings per year[b]**	**Containers**	**Caps**	**Containers (HDPE)**	**Containers (PET)**	**Caps (PP)**
HDPE	625	299,343	0.5	40.0	11,973,709	67.2	12.0	20	-	4
	750	3,592,113	6	36.2	130,000,271	70.4	9.4	253	-	34
	1000	2,993,427	5	13.4	39,993,818	51.7	7.8	155	-	23
	1314	598,685	1	13.7	8,201,171	57.1	11.4	34	-	7
	1500-1518[c]	8,082,254	13.5	14.5	117,008,991	44.0	7.8	356	-	63
	1820-2100[c]	18,858,592	31.5	14.6	274,810,558	48.1	6.8	908	-	128
	2409-2625[c]	7,483,568	12.5	13.4	99,931,045	44.1	5.2	330	-	39
	3000-3066[c]	10,177,653	17	11.4	115,543,058	38.6	4.2	393	-	43
	3900-4000[c]	1,197,371	2	12.2	14,604,297	33.2	2.7	40	-	3
	5000	3,592,113	6	8.2	29,455,325	32.0	2.3	115	-	8
PET	750	598,685	1	28.9	17,295,358	59.6	10.4	-	36	6
	924	598,685	1	30.3	18,141,984	59.5	17.3	-	36	10
	1848	1,796,056	3	15.2	27,212,975	28.4	1.4	-	51	2
Total		59,868,546	100	-	904,172,560	-	-	2,603	122	371
Total avoided waste: 3,096 tonnes										

(a) Estimated based on the volume of automatic liquid laundry detergents sold in Italy during 2013 (acquired from IRi) and by considering an average packaging composition estimated empirically.
(b) This parameter is not directly involved in the calculation of the mass of avoided waste, but is needed to calculate the consumption of the replacing loose detergent and the resulting flows of additional waste (Table D.7).
(c) Similar sizes were grouped in a single class, to simplify the calculation procedure and the modelling as a whole.
(d) Average specific masses of containers and caps were estimated experimentally, as described in Section B.2.1.1 of Appendix B. Note that the simple average mass of containers and caps is useless as, in the case of size classes, it would not be associated to any particular container size.

Table D.4: estimate of the mass of waste avoided (removed from the waste management system) with the substitution of hand wash liquid laundry detergents packaged in single-use containers by those distributed loose through self-dispensing systems and refillable containers.

Avoided consumption of hand wash liquid laundry detergents by type and size of container[a]				**Average washing performance**		**Average specific masses of the products removed from the waste stream[c] [g/litre of detergent]**		**Avoided waste flows [tonnes]**		
Container material	**Container size [ml]**	**Consumption [litres]**	**%**	**Washings per litre**	**Washings per year[b]**	**Containers**	**Caps**	**Containers (HDPE)**	**Containers (PET)**	**Caps (PP)**
HDPE	750	249,947	30	22.7	5,680,617	66.7	9.3	17	-	2
	1000	333,263	40	14.6	4,860,083	50.4	9.0	17	-	3
	1500	104,145	12.5	17.3	1,800,723	39.0	8.0	4	-	1
PET	750	145,803	17.5	13.9	2,025,035	61.0	12.3	-	9	2
Total		833,157	100	-	14,366,458	-	-	38	9	8
Total avoided waste: 54										

(a) Estimated based on the volume of hand wash liquid laundry detergents sold in Italy during 2013 (acquired from IRi) and by considering an average packaging composition estimated empirically.
(b) This parameter is not directly involved in the calculation of the mass of avoided waste, but is needed to calculate the consumption of the replacing loose detergent and the resulting flows of additional waste (Table D.7).
(c) Average specific masses of containers and caps were estimated experimentally, as described in Section B.2.1.1 of Appendix B.

Table D.5: estimate of the mass of waste avoided (removed from the waste management system) with the substitution of liquid fabric softeners packaged in single-use containers by those distributed loose through self-dispensing systems and refillable containers.

Avoided consumption of liquid fabric softeners by type and size of container[a]				Average washing performance		Average specific masses of the products removed from the waste stream[d] [g/litre of detergent]		Avoided waste flows [tonnes]		
Container material	**Container size [ml]**	**Consumption [litres]**	**%**	**Washings per litre**	**Washings per year[b]**	**Containers**	**Caps**	**Containers (HDPE)**	**Containers (PET)**	**Caps (PP)**
HDPE	625	411,976	1	40.0	16,479,048	67.2	12.0	28	-	5
	750	3,295,810	8	38.2	125,973,171	64.5	11.5	213	-	38
	1000	1,647,905	4	37.0	60,972,479	46.5	13.5	77	-	22
	1500-1560[c]	2,471,857	6	15.7	38,908,864	46.6	8.0	115	-	20
	2000-2015[c]	3,913,774	9.5	12.8	49,948,514	39.1	6.2	153	-	24
	2460	205,988	0.5	16.7	3,433,135	41.1	5.3	8	-	1
	2990-3000[c]	4,737,726	11.5	14.1	66,611,624	42.1	4.2	199	-	20
	4000	5,561,679	13.5	11.5	63,856,313	32.9	3.2	183	-	18
PET	750	10,711,382	26	37.5	401,478,448	52.8	10.4	-	566	111
	1000	3,295,810	8	40.0	131,832,388	44.2	10.3	-	146	34
	1500	3,295,810	8	38.7	127,437,975	40.9	6.6	-	135	22
	2000	1,647,905	4	40.0	65,916,194	33.9	4.0	-	56	7
Total		41,197,621	100	-	1,152,848,153	-	-	976	902	322
Total avoided waste: 2,200										

(a) Estimated based on the volume of liquid fabric softeners sold in Italy during 2013 (acquired from IRi) and by considering an average packaging composition estimated empirically.
(b) This parameter is not directly involved in the calculation of the mass of avoided waste, but is needed to calculate the consumption of the replacing loose detergent and the resulting flows of additional waste (Table D.7).
(c) Similar sizes were grouped in a single class, to simplify the calculation procedure and the modelling as a whole.
(d) Average specific masses of containers and caps were estimated experimentally, as described in Section B.2.1.1 of Appendix B. Note that the simple average mass of containers and caps is useless as, in the case of size classes, it would not be associated to any particular container size.

Table D.6: estimate of the mass of waste avoided (removed from the waste management system) with the substitution of liquid hand dishwashing detergents packaged in single-use containers by those distributed loose through self-dispensing systems and refillable containers.

Avoided consumption of liquid hand dishwashing detergents by type and size of container[a]				**Average washing performance**		**Average specific masses of the products removed from the waste stream**[d] **[g/litre of detergent]**		**Avoided waste flows [tonnes]**		
Container material	**Container size [ml]**	**Consumption [litres]**	**%**	**Washings per litre**	**Washings per year**[b]	**Containers**	**Caps**	**Containers (HDPE)**	**Containers (PET)**	**Caps (PP)**
HDPE	750	2,216,612	5	67.5	149,695,195	57.4	5.4	127	-	12
	1000-1100[c]	2,216,612	5	69.4	153,758,984	55.2	3.5	122	-	8
	1250	3,546,579	8	62.7	222,366,573	48.2	3.3	171	-	12
	1500	3,103,257	7	67.4	209,159,506	43.8	3.0	136	-	9
	2000	1,551,628	3.5	51.4	79,676,117	39.4	2.8	61	-	4
	3000	3,103,257	7	62.6	194,160,431	36.3	3.0	113	-	9
	4000	886,645	2	50.2	44,478,366	33.1	2.6	29	-	2
	5000	443,322	1	34.2	15,146,848	27.5	2.4	12	-	1
PET	500	7,979,803	18	190.0	1,515,949,795	58.4	8.8	-	466	70
	600-650[c]	4,876,546	11	118.1	575,722,739	54.8	7.3	-	267	36
	750	5,098,208	11.5	105.8	539,567,164	53.0	6.5	-	270	33
	1000	7,093,158	16	89.0	631,261,535	45.8	5.5	-	325	39
	1250	2,216,612	5	68.0	150,680,356	41.7	3.6	-	93	8
Total		44,332,239	100	-	4,481,623,610	-	-	772	1,421	243
Total avoided waste: 2,435										

(a) Estimated based on the volume of liquid hand dishwashing detergents sold in Italy during 2013 (acquired from IRi) and by considering an average packaging composition estimated empirically.
(b) This parameter is not directly involved in the calculation of the mass of avoided waste, but is needed to calculate the consumption of the replacing loose detergent and the resulting flows of additional waste (Table D.7).
(c) Similar sizes were grouped in a single class, to simplify the calculation procedure and the modelling as a whole.
(d) Average specific masses of containers and caps were estimated experimentally, as described in Section B.2.1.1 of Appendix B. Note that the simple average mass of containers and caps is useless as, in the case of size classes, it would not be associated to any particular container size.

Table D.7: estimate of the mass of additional waste resulting form the substitution of single-use packaged liquid detergents (all targeted categories) by those distributed loose through self-dispensing systems and refillable containers.

Detergent category	Washings to be performed per year[a]	Average washing performance of loose liquid detergent [washings per litre]	Additional consumption loose liquid detergents [litres]	Types of products added to the waste streams	Average masses of the products added to the waste stream[d] [grams]		Number of uses of the refillable containers[e]	Avoided waste flows [tonnes]	
					Containers	Caps		Containers (HDPE)	Caps (PP)
Automatic laundry detergents	904,172,560	15.1[b]	59,868,546	3000 ml refillable containers and respective caps	120	12	10	239	24
		10[c]	90,417,256					362	36
Hand wash laundry detergents	14,366,458	17.2[b]	833,157	3000 ml refillable containers and respective caps	120	12	10	3.3	0.3
		10[c]	1,436,646					5.7	0.6
Fabric softeners	1,152,848,153	28[b]	41,197,621	2000 ml refillable containers and respective caps	103	12	10	212	25
		10[c]	115,284,815					594	69
Hand dishwashing detergents	4,481,623,610	101[b]	44,332,239	1000 ml refillable containers and respective caps	71.5	8.5	10	317	38
		51[c]	87,874,973					628	75
Total			146,231,564 (295,013,690)[f]	-	-	-	-	772 (1,589)[f]	87 (181)[f]
Total additional waste: 859 (1,770)[f]									

(a) See Tables D.3 to D.6 of this Appendix.
(b) Same as the overall average washing performance of substituted single-use packaged detergents (estimated based on data reported in Table D.3 to D.6 of this Appendix).
(c) Specific washing performance defined based on the real experiences of detergent distribution through self-dispensing systems recently implemented in Lombardia.
(d) Estimated experimentally, by weighting the refillable containers (and respective caps) used in the real experiences of detergent distribution through self-dispensing systems recently implemented in Lombardia.
(e) According to the recommendation drawn from the LCA study summarised in Section 3.
(f) For the case in which a worsened washing performance (i.e. a lower number of washings per litre) is assumed for loose detergents.

D.2 Additional results

This section provides further results in terms of variations of the impacts of the downstream components of the waste management systems, between waste prevention scenarios 1 and 2a and the baseline one (Tables D.8 and D.9). The downstream components of the waste management systems affected by waste prevention are collection and transport, sorting of source-separated packaging materials and the recycling of these materials. The process chains relating to the management of the residual waste and of the organic waste are instead unaffected by prevention.

Table D.8: percentage variation in the impacts of the downstream components of the waste management system, involved by the bottled water substitution (waste prevention scenario 1).

Impact category	**Collection and transport**	**Sorting of source separated packaging materials**	**Recycling**	**Total downstream impact**
Climate change	-0.82%	-7.2%	4.0%	3.8%
Ozone depletion	-0.82%	-0.4%	7.2%	1.7%
Photochemical ozone formation	-0.83%	-15.9%	3.2%	3.0%
Acidification	-0.83%	-6.6%	3.1%	2.9%
Terrestrial eutrophication	-0.83%	-8.6%	2.0%	2.0%
Freshwater eutrophication	-0.84%	-8.8%	2.1%	2.1%
Marine eutrophication	-0.83%	-10.9%	2.1%	2.1%
Freshwater ecotoxicity	-0.82%	-1.2%	2.8%	2.2%
Human toxicity (cancer effects)	-0.82%	-5.1%	0.8%	0.8%
Human toxicity (non-cancer effects)	-0.82%	-1.1%	3.8%	40.4%
Particulate matter	-0.84%	-9.2%	2.4%	2.4%
Water resource depletion	-0.83%	-43.8%	3.4%	3.1%
Abiotic depletion	-0.82%	-0.4%	7.8%	3.6%
Cumulative energy demand	-0.83%	-0.5%	3.4%	2.3%
Minimum variation	*-0.82%*	*-0.4%*	*0.8%*	*0.8%*
Maximum variation	*-0.84%*	*-43.8%*	*7.8%*	*40.4%*
Average variation	*-0.83%*	*-8.6%*	*3.4%*	*5.2%*

Table D.9: percentage variation in the impacts of the downstream components of the waste management system, involved by the substitution of single-use packaged liquid detergents (waste prevention scenario 2a).

Impact category	Collection and transport	Sorting of source separated packaging materials	Recycling	Total downstream impact
Climate change	-0.19%	-1.68%	0.88%	0.82%
Ozone depletion	-0.19%	-0.08%	1.01%	0.23%
Photochemical ozone formation	-0.19%	-1.21%	0.13%	0.12%
Acidification	-0.19%	-0.27%	0.40%	4.00%
Terrestrial eutrophication	-0.20%	-2.12%	0.51%	0.50%
Freshwater eutrophication	-0.19%	-3.68%	0.90%	0.86%
Marine eutrophication	-0.19%	-1.54%	0.63%	0.59%
Freshwater ecotoxicity	-0.19%	-1.99%	0.44%	0.45%
Human toxicity (cancer effects)	-0.20%	-2.07%	0.19%	0.18%
Human toxicity (non-cancer effects)	-0.19%	-2.53%	0.44%	0.44%
Particulate matter	-0.19%	-0.28%	0.53%	0.43%
Water resource depletion	-0.19%	-10.3%	0.47%	0.42%
Abiotic depletion	-0.19%	-0.10%	1.94%	0.89%
Cumulative energy demand	-0.19%	-0.11%	0.84%	0.57%
Minimum variation	*-0.19%*	*-0.08%*	*0.13%*	*0.12%*
Maximum variation	*-0.20%*	*-10.3%*	*1.94%*	*4.00%*
Average variation	*-0.19%*	*-2.00%*	*0.66%*	*0.75%*