NORSUS
Norwegian Institute for Sustainability Research

Life cycle assessment of plastic bags and other carrying solutions for groceries in Norway

AUTHOR(S)
CECILIA ASKHAM, ANNA FURBERG, JOHN BAXTER

REPORT NO.
OR.45.21

YEAR
2021

ISBN NO.
978-82-7520-877-2

ISSN NO.
2703-8610

REPORT TYPE
Commissioned

CONFIDENTIALITY
Open

NORSUS

PROJECT TITLE

I pose og sekk

PROJECT NUMBER

3163

COMMISSIONED BY

Handelens Miljøfond

REFERENCE

Sjur Kvifte Nesheim

INTERNAL QUALITY CONTROL

Gaylord Booto

NUMBER OF PAGES

162

KEY WORDS

LCA, plastic bags, Norway, shopping

PHOTO FRONTPAGE

Stockbilder

Summary

There are several previous studies on single use plastic bags and their alternatives for shopping. The need for a study for Norwegian conditions arose as the bags used in Norway were perceived by industry as of high quality and multifunctional (used as bin liners by Norwegian consumers). The seemingly poor performance of multiuse alternatives in other Scandinavian studies motivated NORSUS to want to perform such a study. Thus, the actors involved had different preconceptions, but a common interest in performing a robust study. This study will be made public and used to inform Norwegian consumers, thus a critical panel has been involved during the whole study.

The functional unit used in this study has been chosen to reflect the quantity of goods purchased by Norwegian consumers as well as the capacity of carrying solutions and how that capacity is utilised. In addition, relevant aspects of consumer behaviour have been investigated from literature and surveys.

The functional unit selected for this study was:

The carrying solutions needed to transport one month's grocery shopping for a Norwegian consumer and provide an equal amount of bin liners as would be provided from utilising plastic (LDPE or HDPE) bags for the shopping.

The carrying solutions modelled in this study were: LDPE and HDPE plastic bags (both with 80% recycled content), paper bag (no recycled content), nylon multiuse bag (no recycled content), PET multiuse bag (100% recycled content), organic cotton multiuse bag (no recycled content), cardboard box (46% recycled content) and a rucksack (no recycled content).

Two main modelling approaches were used for material recycling and waste incineration with energy recovery: cut-off and system expansion (using the net scrap approach). The choice of modelling approach has a significant effect on the results. A base case of 50% volume utilisation was defined for the carrying solutions, and lower and upper ranges defined for "small shopping" (1.1 litres) and large shopping (80% of carrier volume) for sensitivity analysis. Further sensitivity analyses were also performed on the proportion of recycled materials used to make the carrier, number of uses, carrier volume utilisation, transport distances, cotton bag volume, and changing electricity mixes for carrier production electricity from location specific mixes to the average European electricity production mix. These were parameters perceived to be of importance to the results and relative rankings of the carrying solutions.

This study compares carrier solutions available and suitable for the Norwegian market. The goal, scope and data used are relevant for Norway, Norwegian retailers and consumers. This means that the results and conclusions are relevant for this and not necessarily applicable to other regions.

For both the cut-off and net scrap modelling approaches, the ***multiuse carriers are preferable*** to the single use carriers, due to the multiuse carrier impacts for production processes being shared across the number of uses for the given carrier. Consumers should be encouraged to use a carrier that they already own instead of buying a new one. However, in situations where a multiuse carrier (already owned) is not available the retailer wishes to know which single use carrier is preferable. For cut-off modelling the endpoint results show the following:

- For potential human health damage: the paper bag, but the plastic single use bag made in Scandinavia is not likely to be significantly worse.
- For potential ecosystem damage (excl. litter impacts): the single use plastic carriers.

- For resources: the paper bag.

For net scrap modelling the endpoint results show the following:

- For potential human health damage: the plastic single use bag made in Scandinavia, but if made far away, then the paper bag is better.
- For potential ecosystem damage (excl. litter impacts): the single use plastic carrier made in Scandinavia.
- For resources: the paper bag, but the single use plastic carrier made in Scandinavia is not likely to be significantly different.

Where ***littering*** is an aspect of concern, the bio-based carrier alternatives (paper, cotton and cardboard) are preferable, as well as the rucksack, which is extremely unlikely to become litter. If the littering problem is weighted strongly for the decision-maker, the single use option that is environmentally preferable is the paper bag.

It would be environmentally beneficial for purchasers of carriers to influence their suppliers to use cleaner energy for production processes and transport, as well as use recycled raw materials.

Norwegian consumers use their single use plastic shopping bags as bin liners. This is an important part of their function for Norwegian consumers included in this study. Whichever carrier is chosen, re-use as many times as possible before it reaches waste treatment is important. If it can be used as a bin liner for its last use, this maximises the function of the product for the same resource input. The perception that the use of single use bags as bin liners is environmentally beneficial is however only partially correct. If the alternative is to dispose of the bags in the waste management system without achieving this second function, bin liners would have to be purchased by the consumer. The inclusion of bin liners in the functional unit has a significant impact on the results for all of the multiuse carriers, independent of the modelling approach. The bin liner is light weight, lighter than the single use plastic carriers (LDPE and HDPE). This means that a plastic carrier bag is over-designed (has too much material, or is of too good quality) to be used as a bin liner. If a multiuse carrier is successfully used by the consumer (remembered when they go to the shop), it is environmentally preferable that they purchase lightweight, adequately sized bin liners for their household waste bin, rather than purchase carrier bags because they have run out of bin liners.

Norwegian consumers each buy about 12 new plastic bags per month, which amounts to 782 million plastic bags per year. The results in this report can be used to calculate the environmental benefits of reducing consumer use of single use plastic (SUP) bags. It is unlikely that one could change the shopping habits of every consumer in Norway, but ambitious reduction targets would be environmentally beneficial.

Sammendrag

Det finnes flere tidligere studier på engangsplastposter og deres alternativer ved dagligvarehandel. Behovet for en studie av norske forhold oppstod ettersom handleposene som brukes i Norge anses av industrien som av høy kvalitet og de har flere mulige bruksområder (de brukes som avfallsposer av norske forbrukere). Det at de alternative løsningene med flere bruksområder har kommet dårlig ut i andre skandinaviske studier motiverte NORSUS til å ville gjennomføre en slik studie for norske forhold. De involverte aktørene hadde ulike meninger om hvordan utfallet ville bli, men de satt med samme interesse for å gjennomføre en robust studie. Denne studien vil bli offentliggjort og brukt til å informere norske forbrukere, og derfor har et kritisk gjennomgangspanel vært involvert i hele studien.

Den funksjonelle enheten brukt I denne studien ble valgt for å reflektere mengden varer kjøpt av norske forbrukere, samt bæreløsningers kapasitet og hvordan denne kapasiteten utnyttes. I tillegg ble relevante aspekter knyttet til forbrukeratferd undersøkt i litteratur og spørreundersøkelser.

Den funksjonelle enheten valgt for denne studien var som følger:

Bæreløsningene som trengs for å frakte en måneds dagligvarehandel for en norsk forbruker, som også gir en lik mengde avfallsposer som ved å bruke plastposer (LDPE eller HDPE) til handelen.

Bæreløsningene som ble modellert i studien var: LDPE- og HDPE-plastposer (begge med 80 % resirkulert materiale), papirpose (ikke noe resirkulert materiale), flerbrukspose av nylon (ikke noe resirkulert materiale), flerbukspose av PET (100 % resirkulert materiale), flerbuksposer av økologisk bomull (ikke noe resirkulert materiale), pappeske (46 % resirkulert materiale) og en ryggsekk (ikke noe resirkulert materiale).

To modelleringsmetoder ble brukt for materialgjenvinning og energigjenvinning: cut-off og systemutvidelse (ved bruk av net scrap-metoden). Valg av modelleringsmetode har betydelig innvirkning på resultatene. En grunncase med 50 % volumutnyttelse ble definert for bæreløsningene, og øvre og nedre grenser ble definert for «småhandling» (1,1 liter) og storhandel (80 % av volumet i bæreløsningen) til bruk i sensitivitetsanalyse. Sensitivitetsanalyser ble også utført for andel resirkulert materiale brukt til å lage bæreløsningen, antall bruk, bæreløsningens volumutnyttelse, transportavstander, volumet av bomullsposer, og endrede strømmikser for produksjon av bæreløsninger - fra stedsspesifikke mikser til gjennomsnittlig europeisk strømmiks.

Denne studien sammenligner bæreløsninger som er tilgjengelige og passende for det norske markedet. Målet, omfanget og data er relevante for Norge, norske dagligvareaktører og forbrukere. Dette betyr at resultater og konklusjoner er relevant for dette formålet, og er ikke nødvendigvis anvendelig i andre regioner.

For både cut-off- og net scrap-modelleringsmetodene, er flerbruksbæreløsningene å foretrekke framfor engangsbæreløsningene. Dette er fordi påvirkningen fra produksjon av flerbruksløsningene fordeles på antall ganger den gitte bæreløsningen brukes. Forbrukerne bør oppmuntres til å bruke en bæreløsning som de allerede eier heller enn å kjøpe en ny. Men, i situasjoner der en (allerede eid) flerbruksløsning ikke er tilgjengelig, vil dagligvareaktørene vite hvilken engangsløsning som er foretrukket.

For cut-off-modelleringen viser sluttresultatene følgende:

- For potensielle menneskelige helseskader: papirposen, men det er ikke sannsynlig at engangsplastposen produsert i Skandinavia er atskillig dårligere.
- For potensiell økosystemskade: engangsplastpose
- For ressursbruk: papirpose

For net scrap-modelleringen viser sluttresultatene følgende:

- For potensielle menneskelige helseskader: engangsplastposer produsert i Skandinavia, men dersom laget langt unna er papirposen bedre.
- For potensiell økosystemskade: engangsplastpose produsert i Skandinavia.
- For ressursbruk: papirposen, men det er ikke sannsynlig at engangsplastposen produsert i Skandinavia er atskillig dårligere.

I tilfeller der forsøpling er viktig, vil de biobaserte bæreløsningene (papir, bomull og papp) foretrekkes, i tillegg til ryggsekken, som er svært lite sannsynlig at ender opp som forsøpling. Dersom forsøplingsproblemet vektes tungt av beslutningstakeren, er papirposen det miljømessig foretrukne engangsalternativet.

Det vil være miljømessig gunstig at innkjøpere av bæreløsninger påvirker sine tilbydere i retning av å bruke renere energi i produksjonsprosessen og transport, i tillegg til å bruke resirkulerte råmaterialer.

Contents

Summary iii
Sammendrag v
1 Introduction 1
2 Goal of the study 2
3 Scope of the study 3
3.1 Functional unit 3
3.2 The carrier solutions included in this study 4
3.3 Number of carriers per functional unit 5
3.4 System boundaries 9
3.4.1 Cut-off criteria 11
3.5 Modelling recycling in LCA 12
3.5.1 The allocation problem 12
3.5.2 Cut-off/Recycled content 13
3.5.3 Net scrap approach 13
3.6 Data Requirements and Data Quality 14
3.7 Life Cycle Impact Assessment 16
3.7.1 Littering 19
3.7.1 Endpoint damage categories 20
3.8 Sensitivity Analysis 21
3.9 Reporting and Critical Review 21
4 Comparison of goal and scope for other carrier studies 23
5 Systems studied 30
5.1 Low-density polyethylene (LDPE) plastic bag 30
5.2 High-density polyethylene (HDPE) plastic bag 32
5.3 Bin liner 33
5.4 Paper bag 34
5.5 Nylon bag 35
5.6 Polyethylene terephthalate (PET) bag 37
5.7 Cotton bag 39
5.8 Cardboard box 40
5.9 Rucksack 42
6 Results 46
6.1 Endpoint results - Cut-off 46
6.1.1 Human health 46
6.1.2 Ecosystems 47
6.1.3 Resources 47

6.1.4 Number of times a given carrier must be used – Endpoint cut-off 48
6.2 Midpoint results - Cut-off 49
6.2.1 Results for all impact categories 51
6.2.2 Global warming potential 54
6.2.3 Marine eutrophication potential 55
6.2.4 Water deprivation potential 56
6.2.5 Fine particulate matter formation potential 57
6.2.6 Ionizing radiation 58
6.2.7 Littering 59
6.2.8 Number of times a given carrier must be used - Midpoint cut-off 60
6.3 Endpoint results – Net scrap approach 61
6.3.1 Human health 61
6.3.2 Ecosystems 62
6.3.3 Resources 62
6.3.4 Number of times a given carrier must be used – Endpoint net scrap 63
6.4 Midpoint results - Net scrap approach (EoL substitution) 64
6.4.1 Global warming potential 69
6.4.2 Marine eutrophication potential 70
6.4.3 Water deprivation potential 71
6.4.4 Fine particulate matter formation potential 72
6.4.5 Ionizing radiation potential 73
6.4.6 Land use potential 74
6.4.7 Number of times a given carrier must be used - Midpoint net scrap 74
6.5 Sensitivity analysis 76
6.5.1 Recycled content 77
6.5.2 Number of uses 81
6.5.3 Volume utilisation 84
6.5.4 Transport distances 87
6.5.5 Cotton bag volume 90
6.5.6 Electricity mixes 92
7 Discussion 99
8 Conclusion 106
9 Critical Review Summary 107
9.1 Rebuttal 107
10 References 114
Appendix 1: Shopping habits investigation and reference flows 119
Appendix 2: Use of rucksacks for grocery transport 126
Appendix 3: Use of textile bags (“bags-for-life”) 128
Appendix 4: Background system data for the studied systems 129

Appendix 5: Textile waste Management in Norway 146
Amounts of Norwegian textile waste incinerated and recycled 146
Recycling of textiles 147
Data for collection and sorting of textile waste 148
Washing and drying of textiles for reuse and recycling 149
Energy recovery from incineration of fibres 149
The choice of recycling process and replaced material 149
Appendix 6: LCIA calculation procedures 154
Appendix 7: Critical review report 156
A7.1 The study 156
A7.2 The review 156
A7.3 Comments 157
A7.3.1 General 157
A7.3.2 Background of study 157
A7.3.3 LCI modelling approaches 158
A7.3.4 Transparency 158
A7.3.5 Utilization of carrier capacity 158
A7.3.6 The number of uses 159
A7.3.7 Cotton production 159
A7.3.8 Life cycle impact assessment 160
A7.3.9 Littering 160
A7.3.10 Calculations and results 161
A7.3.11 Conclusions in the report 162
A7.3.12 Recommendations in the report 162

1 Introduction

There are several previous studies on single use plastic bags and their alternatives for shopping (UNEP 2020). The need for an additional study, but for Norwegian conditions, arose as the bags used in Norway were perceived by industry as of better quality and multifunctional (they are widely used as bin liners by Norwegian consumers). The life cycle assessment (LCA) experts from NORSUS were sceptical to the seemingly bad performances of multiuse alternatives in other Scandinavian studies. Thus, the actors involved had different preconceptions, but a common interest in performing a robust study. As the authors acknowledge the above potential bias and the study should be public and used to inform Norwegian consumers in public information campaigns, a critical panel was essential. Involving the critical panel has made this LCA study truly iterative.

The function studied (and therefore functional unit used) reflects the quantity of goods purchased by Norwegian consumers as well as the capacity of carrying solutions and how that capacity is utilised. In addition, relevant aspects of consumer behaviour have been investigated from literature and surveys. In Norway, 80% of lightweight plastic carrier bags bought in stores are used as bin liners (SIFO, 2019).

This study focuses on the assessment of commonly used carrying solutions available in shops in Norway currently, both single use and reusable solutions. In addition to these commonly used carrying solutions rucksacks and cardboard boxes are also included as alternatives. It should be noted that biodegradable shopping bags are not included in this study, as Norwegian retail actors do not perceive these as a viable option for the Norwegian market in the near term (up to 5 years).

The capacity of shopping carriers and how they are utilised is a complicated issue. The functional unit and number of carriers influencing material reference flow considerations are extensive and detailed in this study. NORSUS has also performed a limited observational study to include Norwegian consumer behaviour. The data gathering stage and consumer study were performed during restrictions due to the corona pandemic, which imposed limitations on the approaches that could be used.

This study has been conducted in accordance with ISO 14044 (ISO 2006b and ISO 2020). The critical panel noted that of the two modelling approaches used, the cut-off approach is not consistent with the new Annex D in ISO 14044:2006/A2:2020. Also, that it is probably not consistent with ISO/TR 14049. They also note that Annex D is not normative but descriptive, and 14049 is not a standard but a technical report. Each step in the allocation procedure is also a recommendation rather than a requirement (indicated by the use of “should” instead of “shall”). This, leaves room for flexibility while remaining formally consistent with ISO 14044. Divergence from Annex D and 14049 is therefore mentioned here to ensure clarity.

2 Goal of the study

The goal of this study is to assess the environmental impact of various carrying solutions currently available for transporting groceries between the store and the consumer's home in Norway, and to investigate under what circumstances these solutions become more or less environmentally preferable. Thus, a comparison of the environmental impacts of using plastic bags and other relevant carrying solutions, such as those made from cotton or paper, will be conducted with the focus to identify which aspects make solutions more or less suitable from an environmental perspective. A carrying solution is here defined as a means of carrying groceries from the store to the consumer's home in Norway.

The main reason for conducting this study is to provide recommendations for the circumstances in which various carrying solutions are environmentally preferable within the Norwegian context. The intended application of the study is to guide public authorities in the development of policies to reduce the environmental impacts of these carrying solutions. The study results are also intended to guide supermarkets and retailers in the procurement of carrying solutions and to enable consumers to make informed choices regarding carrying solutions from an environmental perspective. The intended audience are public authorities, such as the Norwegian Environment Agency, and other interested parties including supermarkets, retailers, environmental organisations, and consumers. The Norwegian Retailer's Environment Fund is also an interested party, having provided the funding for this work. The results from the study are intended to be used in comparative assertions to be disclosed to the public, thus a critical review will be conducted in accordance with ISO (2006 a, b).

3 Scope of the study

3.1 Functional unit

The main function of a carrying solution, such as a plastic bag, is to carry groceries from the store to the consumer's home. How well this function is fulfilled depends on the carrying capacity of the solution but also on the character of the groceries (weight and volume) to be transported. A larger carrier bag does not necessarily fulfil the function of carrying goods to a higher degree than a smaller bag, this depends on the weight and volume of the groceries to be transported (UNEP, 2020). The carrying solution should provide a convenient, practicable option for the consumer to carry the goods. For example, the consumer needs to be able to bear the weight of the carrying solution including groceries, thus a solution with a very high carrying capacity might never be fully utilised.

The functional unit should be carefully selected to enable a fair comparison. In this study, the functional unit initially considered was carrying one month's grocery shopping from the grocery store to the consumer's home in Norway in 2019. This functional unit needs further specification by using data on the average weight and volume of groceries bought by a person for one month's shopping in Norway. A consumption study by the Norwegian Directorate of Health (2021) indicates food consumption per inhabitant of around 616 kg per annum in 2019. This 616 kg does not include the food packaging, or non-food products, such as cleaning products, etc., so is incomplete, but allows us to establish a value for one month's food shopping to be one-twelfth of that total or 51.4 kg. The amount of shopping including packaging and non-food groceries would be larger, which would increase the scale of the results shown in Chapter 6, but the relative results would not change. The number of carriers of a particular type that are required to fulfil this function is a central parameter used to calculate the reference flow of materials. In principle the functional unit and reference flow should reflect the actual quantity of goods purchased by the customer, in mass and/or in volume terms, also the capacity of carrying solutions and how that capacity is utilised (in mass or in volume as appropriate).

In addition to this carrying function, in Norway 80% of lightweight plastic carrier bags brought in stores are used as bin liners (SIFO, 2019). This implies that plastic bags, in addition to providing the function of carrying groceries, also provide a secondary function as they can be reused as bin liners. The number of bin liners provided for the plastic bags was derived from the number of light-weight plastic carrier bags that would be needed to transport the 51.4 kg of shopping and of which a certain share (80%) become reused as bin liners in the home. For example, in the baseline case 19 low density polyethylene (LDPE) plastic carrier bags are needed to carry 51.4 kg groceries (see

Table 3) and if 80% of these become reused, then the plastic bags also provide the function of 15.2 bin liners. The plastic bags not reused as bin liners become disposed of as plastic waste (see Chapter 5.1). The carrying solutions that are not single use plastic (SUP) bags do not become reused as bin liners, thus a comparable number of bin liners must be added to the life cycle of these carriers. The bin liner function has been incorporated into the functional unit in this study using system expansion (using the net scrap approach) and the functional unit selected for this study was:

The carrying solutions needed to transport one month's grocery shopping for a Norwegian consumer and provide an equal amount of bin liners as would be provided from utilising single use plastic (LDPE or HDPE) bags for the shopping.

3.2 The carrier solutions included in this study

Currently, the plastic bag is the most available as well as the most affordable carrying solution offered in Norwegian stores (SIFO, 2019). At the same time other carrying solutions, including paper bags and reusable options made from textiles or plastic, are becoming increasingly available (SIFO 2019). This study will focus on the assessment of these commonly used carrying solutions. Both single use and reusable solutions as well as their typical material compositions and recycled content on the Norwegian market will be considered. In addition to these commonly used carrying solutions, some other promising alternatives were included, as they were considered likely to be used in Norway in the near future, e.g., within a five-year timeframe. Thus, rucksacks and cardboard boxes are also included as alternatives in this study. It should be noted that biodegradable shopping bags are not included in this study, as Norwegian retail actors do not perceive these as a viable option for the Norwegian market in the near term (up to 5 years). There are two main reasons for this; the biodegradable plastic on the market today can only be degraded in industrial composting facilities and thus not in Norwegian natural conditions. This type of plastic would therefore not solve the issue of littered plastic being an environmental problem, that creates microplastic during the degradation process and both macro- and microplastic that can end up in the ocean. Plastic bags made from biodegradable material are also not recyclable, if they enter the recycling waste stream they will degrade the quality of the recycled plastic product (Lopez et al. 2012).

The industry representatives in the review panel (see Chapter 3.9and NORSUS collaborated to define relevant carrying solutions to include in the study. The choice of carrying solutions was also discussed with the whole review panel and observer. The carrying solutions included in this study are:

- LDPE (low density polyethylene) single use plastic bags purchased in store - 20% virgin LDPE, 80% recycled LDPE
- HDPE (high density polyethylene) single use plastic bags purchased in store - 20% virgin HDPE, 80% recycled PCW[1]
- Paper bags - 100% virgin material, flat handle
- “Bag for life” shopping bag - 100% virgin nylon
- 100% recycled PET (Polyethylene terephthalate)
- Textile shopping bag- 100% virgin organic cotton
- Cardboard box - 46% recycled material
- Rucksack - Polyester, nylon, foam, plastic buckles and steel zips

A base case of 50% volume utilisation was defined for the carrying solutions (see Table 2), and lower and upper ranges defined for “small shopping” (1.1 litres) and large shopping (80% of carrier volume) for sensitivity analysis.

Sensitivity analyses were also performed on:

- the proportion of recycled materials used to make the carrier;
- the number of uses of the carrier;
- carrier volume utilisation;
- transport distances;
- the cotton bag volume;

[1] Post consumer waste

- changing electricity mixes from specific mixes for given locations to the average European electricity production mix.

3.3 Number of carriers per functional unit

In order to calculate the reference material flows, the number of carriers per functional unit needs to be calculated (to then be multiplied by the mass of material per carrier). This chapter presents the work performed in order to calculate the number of carriers per functional unit.

The capacity of shopping carriers and how they are utilised is a complicated issue, and most related LCA studies to date appear fairly sketchy in their treatment of this. The functional unit and number of carriers influencing material reference flow considerations as described in this section, are more extensive and detailed than for most other studies in the literature (for example, Bisinella et al., 2018).

Capacity utilisation depends on carrier size and strength, characteristics of the goods themselves (such as bulk density) and elements of consumer behaviour including carrier packing practice and shopping patterns. Maximum capacities, either by mass or by volume, may not be very meaningful in practice. For example, a study for the Danish Environmental Protection Agency (Bisinella et al., 2018) collated volume and maximum weight holding capacity for carrier bags of various sorts, with the latter described as the capacity at which the material began to break. However, particularly for larger bags this gave rise to maximum mass capacities in excess of 30 kg and in some cases up to 50 kg. This is clearly unrealistic in practice, as few consumers would even be capable of lifting those carriers and would very seldom pack them in that fashion even if it were possible.

The ability of consumers to lift carriers is clearly a limiting capacity factor in practice. The bulk density of goods as packed will be significant. This was estimated in a recent study (Civancik-Uslu et al., 2019) as 450 kg/m^3 or 0.45 kg per litre. This seems reasonable for groceries as a whole – some items such as liquids or pastes in glass jars will clearly have a higher density, but many other goods are much less dense, and obviously there are large empty spaces in any packing structure within a shopping carrier.

A simple, yet seemingly reasonable, assumption for the present study is to take this estimate of packed bulk density and propose that carrier packing is entirely volume limited. This means that every 10 litres of volume provide 4.5 kg of carrying capacity, and that volume is the limiting characteristic of the carrier (i.e., the only capacity factor that matters). It means that typical carriers in the range of 15-25 litres provide around 6-10 kg of mass capacity when full, which seems reasonable in practice. An experimental study of grocery packing behaviour in the US by Kimmel et al. (2014) suggested that bags of various types with around 20 litres nominal capacity were typically packed to hold 4-5 kg of groceries. This clearly established the idea that packing is volume limited in practice (the weight-bearing capacity of the bags was much higher).

For volume-limited packing, the maximum volumetric capacity of carriers is clearly not always used. The practice of packing goods and the size of individual shopping events will be important. In some cases, consumers will use a carrier for small shopping events where only a few items are purchased. In contrast, for large shopping events where the goods fill multiple carriers, a much higher fraction of the available volume per carrier will be utilised. A Canadian study by CIRAIG (2017) attempted to account for different shopping event sizes using two functional units (1 and 100 litres of shopping), the approach of defining a ”small shopping” and “big shopping” event for sensitivity analysis was also deemed relevant for this study.

The limiting values for volumetric capacity used are clearly not 0% and 100% respectively. Small shopping events consist of a consumer purchasing 2-3 items and using a bag, irrespective of size. For this study, observational data was collected on actual shopping habits in Norway during the summer of 2021, see Appendix 1. 1536 shopping events were observed, and the capacity utilisation of carriers is summed up in Table 1.

Table 1: Capacity utilisation of different carriers

Carrier Type	Number of carriers	Average items per carrier	Volume of items (litres)	Nominal volume of carrier (litres)	Overall capacity utilisation
Plastic bags (in-store purchase)	923	8.02	9.17	12	76%
No carrier used	625	3.51	4.01	-	-
"Bag for life"	111	10.82	12.37	20	62%
Woven plastic bag	17	17.12	19.56	20	98%
Paper bag	13	7.92	9.05	24	38%
Textile bag	50	9.14	10.45	25	42%
Rucksack or hand bag	78	7.82	8.94	30	30%
Own single-use plastic bag	63	7.56	8.63	12	72%

This observational study found that at 625 of these events, consumers did not use a carrier. It should be noted that this finding has not been used to affect the number of carriers needed, as it was not possible to survey the proportion of a consumer's monthly shopping that is carried out without a carrier. A broader consumer survey would be needed for that purpose. However, this may weigh up somewhat for the underestimate of grocery shopping volume noted above, as the monthly amount of groceries is based on food consumption (excluding packaging and non-food items).

In the shopping events where a carrier was used, 30-98% of the carrier volume was utilised. Examples of all types of carriers analysed in this study (including bags for life and rucksacks) were observed, except for the cardboard box. This is unsurprising as the cardboard box is currently a home delivery carrying solution and not used for grocery shopping in stores. The average shopping event item was found to occupy 1.1 litres of volume. Thus a "small shopping" event that involves a carrier is assumed to use the lower volume limit of 1.1 litre. Examining the raw data, 203 shopping events involved 1 item (see Figure 1 in Appendix 1), and 194 of these single item events used no carrier at all. If 2 item shopping events are considered (e.g., approximately 2.2 litre of shopping), then there are 242 of these events, of which 223 were carried out without using a carrier. Assuming that the carriers assessed in this study (Table 2) will be used in small shopping events is thus unlikely, but is a good basis for an example (outlier) that can be considered for a sensitivity analysis.

For the larger shopping events, complete capacity utilisation depends on the size of the shopping event relative to the carrier(s) and the way in which they are packed. It is proposed that a maximum of 80% of total carrier capacity is possible; Table 1 shows that such capacity utilisations are possible. Figure 4 in Appendix 1 shows that 75% of items were purchased in shopping events where 20 or more items were purchased at a time, where an average of 10.4 items were packed per bag (approx. 11.4 litres of shopping, or a capacity utilisation of 95% for the in-store purchase plastic bags).

Based on the observational data documented in Appendix 1 and the description above, the volume utilisation assumed for the carriers in this study is shown in Table 2.

Table 2: Carrier materials and volume utilisation per functional unit
LDPE = Low density polyethylene, PCW = post consumer waste (recycled plastic), HDPE = high density polyethylene, PET = Polyethylene terephthalate.

Carrying solution	Composition	Material weight (g)	Maximum volume (litres)	Capacity utilisation (litre)		
				Baseline case (50%)	Small shopping event	Large shopping event (80%)
Single use plastic bags purchased in store	20% virgin LDPE 80% recycled LDPE	19	12	6	1.1	9.6
	20% virgin HDPE 80% recycled PCW	19	12	6	1.1	9.6
Paper bags	100% virgin material, flat handle	63	24	12	1.1	19.2
"Bag for life" shopping bag	100% virgin nylon	45	23	11.5	1.1	18.4
	100% recycled PET	56	15	7.5	1.1	12
Textile shopping bag	100% virgin cotton	125	25	12.5	1.1	20
Cardboard box	46% recycled material	227	40	20	1.1	32
Rucksack	Polyester, nylon, foam, plastic buckles and steel zips	846	30	15	1.1	24

The material weight of the carriers in Table 2 was obtained from measurements, or carrier supplier data (see Appendix 4).

Having established a volumetric basis for the analysis, it is first necessary to convert the functional unit into volume. Taking the packed bulk density as 0.45 kg per litre, the functional unit is 51.4 / 0.45 = 114.2 litres. The number of carriers (N_i) is defined as the number of carrying solutions needed in order to fulfil the functional unit, i.e., to provide that available carrying volume. The number will depend on the effective volumetric capacity of carriers, see Table 1. Note that differences in function between the carrying solutions, other than their weight and volume carrier capacity, such as consumer perception on the ease of use or aesthetics, will not be considered in this study. The number of carriers *i* (N_i) is calculated according to:

$$N_i = \frac{G}{V_i \cdot U_i} \qquad \text{Eq. 1}$$

where one month's amounts of groceries in litre (G) are divided by its effective volumetric capacity in litre (V_i) times the number of uses (U_i) during the lifetime of the carrier. Thus, the reference flow is readily calculated as the total volume required to carry the groceries divided by its lifetime volumetric capacity. For example, for 12 litre single-use plastic bags where 50% capacity is utilised on average, then:

$$\frac{114.2}{12 \cdot 0.5} = 19.03$$ bags are required.

For multiuse carriers, the number of uses must be considered since only a fraction of the carrier is effectively used for fulfilment of the functional unit. Hence, for example, for 35 litre rucksacks where 50% capacity is utilised on average, then:

$$\frac{114.2}{35 \cdot 0.5} = 6.53$$ rucksack uses are required.

However, as outlined in Appendix 1, if the rucksacks are used 2600 times in their lifetime, then 6.53 uses are equivalent to:

$$\frac{6.53}{2600} = 0.0025$$ rucksacks.

Technically this "fraction of carrier" consideration is also valid for single-use carriers, but it results in division by 1 which is not normally expressed explicitly in the calculation. A similar approach is adopted for multiuse textile bags with 1000 lifetime uses. The number of carriers applied for the functional unit is shown in Table 3.

Table 3: Number of carriers per functional unit

			Number of bags per functional unit			Number of bin liners needed
Carrying solution	**Composition**	Number of uses	**Baseline case (50%)**	**Small shopping event**	**Large shopping event**	
Single use plastic bags purchased in store	20% virgin LDPE 80% recycled LDPE	1	19	103.8	11.9	0
	20% virgin HDPE 80% recycled PCW	1	19	103.8	11.9	0
Paper bags	100% virgin material, flat handle	1	9.5	103.8	5.9	15.2
"Bag for life" shopping bag	100% virgin nylon	1000	0.0099	0.1	0.006	15.2
	100% recycled PET	1000	0.015	0.1	0.010	15.2
Textile shopping bag	100% virgin cotton	1000	0.0091	0.1	0.006	15.2
Cardboard box	46% recycled material	1	5.7	103.8	7.1	15.2
Rucksack	Polyester, nylon, foam, plastic buckles and steel zips	2600	0.0029	0.040	0.0018	15.2

It is noteworthy that multiuse carriers may impose some constraints on shopping patterns and behaviour. For single-use carriers, the requisite number of carriers to hold the goods are assumed to be taken at the point of purchase; this means the solution can accommodate shopping events of any size. This may also effectively be the case for multiuse carriers such as textile bags if the user has multiple textile bags available at any given time. In contrast, for rucksacks, the user might only be expected to carry one at a time and this therefore places an implicit constraint (a full rucksack) on the size of single shopping events. We note such considerations but consider them too complicated for inclusion in the present study. Moreover, the study focuses on carriers of a single type. In practice, shoppers may well combine single-use and multiuse carriers of different types in individual shopping events. The respondents to the survey presented in Appendix 2 also mentioned combining rucksacks with textile bags. This kind of combination of different carrying solutions is beyond the scope of the present study.

Moreover, the "number of lifetime uses" for multiuse carriers is not straightforward, especially for carriers such as rucksacks that could be used for purposes other than shopping. The basis for the number of uses and lifetime values shown in Table 3 are provided in Appendices 2 and 3 for the multiuse carriers and

rucksack. For the multiuse bags the base case scenario assumptions of 1000 uses have been found to be reasonable, both though the survey conducted in this study (see Appendix 2 and 3) and other literature sources (e.g., Briedis et al. (2019)'s number of uses was 1414 for a multiuse plastic bag with a lifetime of 10 years). It should also be noted that there are multiuse bags that are given to consumers and are not used. None of the respondents to the survey about rucksacks had purchased the rucksack in order to use it for shopping, shopping was an additional use function for their rucksack, not the main or intended use. Some respondents had been given the rucksack as a gift, or as a prize. The definition of the number of lifetime uses is a sensitive parameter and is therefore subject to sensitivity analysis in Chapter 6.5.

When considering allocation of the multiuse carriers, it is also relevant to consider the scenario where the multiuse carrier has been purchased for other purposes, not for the purpose of shopping. This is particularly relevant for the rucksack. It could be relevant to allocate the production of the rucksack to shopping in different ways: (a) the rucksack's shopping use is assumed to be incidental next to other uses; (b) it is assumed to be used solely for shopping; (c) an intermediate scenario. In the first case, none of the environmental burdens of manufacture and disposal are allocated to shopping events. In the second case, all of the burdens are connected with shopping. This study uses approach (c) having surveyed respondents 30% of the rucksack use was allocated to shopping (70% to other functions). If approach (a) had been chosen the rucksack could be considered as having zero impacts for the shopping function, leaving only the bin-liner impacts. Conversely if the rucksack was purchased for shopping and considered as used only for shopping (approach b), then the rucksack burdens allocated to the shopping function would be higher than shown in this study.

3.4 System boundaries

The carrying solutions listed in Table 1 will be assessed applying a cradle-to-grave perspective, implying that the solutions will be assessed over their entire life cycles. A simplified flowchart illustrating the system boundary applied in this study is provided in Figure 1. All life cycle processes from raw material extraction and production to the use and waste management of the assessed carrying solutions will be included with two exceptions. The store, including the storage of carrying solutions, and the transport of the carrying solution, including groceries, to the consumer's home will be excluded from this study as these processes can be assumed to have a smaller contribution to the total environmental impact and/or be similar for the different carrying solutions. Transport packaging for the different carrier solutions from their production site to the store is also not included in this study. This seemed reasonable as Edwards et al. (2011) concluded that secondary packaging had a minimal influence on carrier environmental performance. Figure 1 illustrates a generic, simplified flowchart for the single use plastic bags purchased in store.

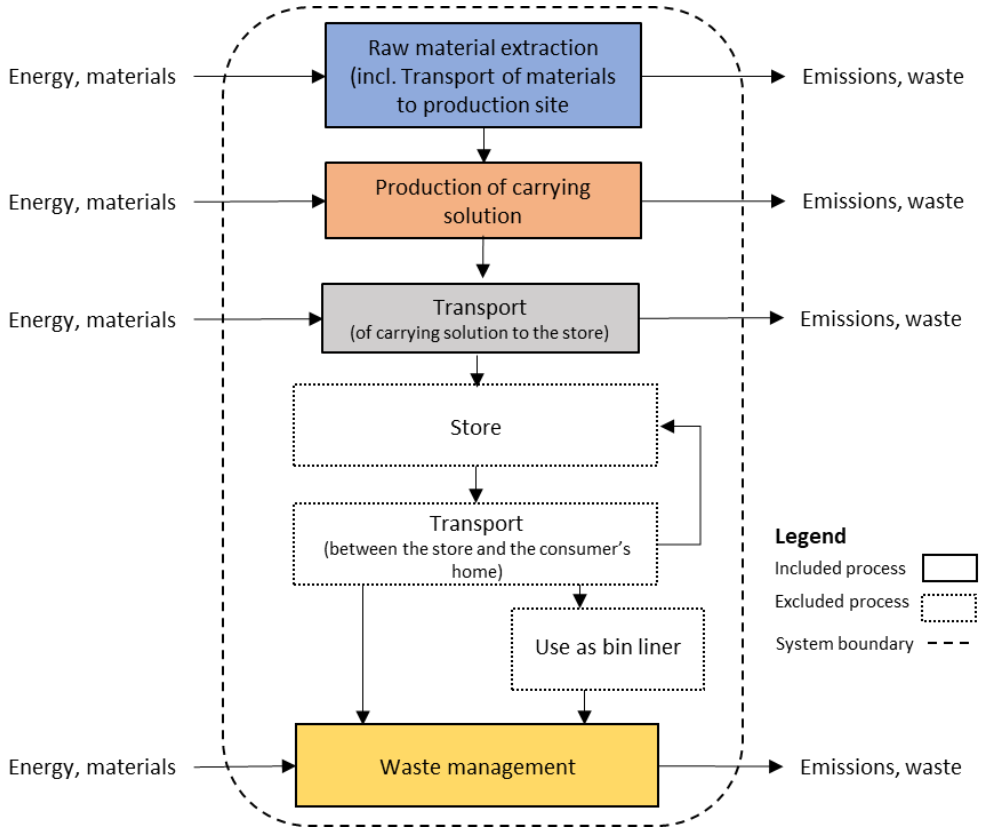


Figure 1. Simplified flowchart illustrating the system boundary applied in this study for LDPE and HDPE light weight plastic bags. The colours used in the boxes correspond to the colours used in the results figures (Chapter 6).

The lightweight plastic carrier bags made from LDPE and HDPE can also be used as bin liners. For all other carrier systems Figure 2 illustrates their simplified flow chart. These other systems need to have a bin liner produced and transported to the home in order to fulfil the same functions (i.e. carrying groceries and bin liner).

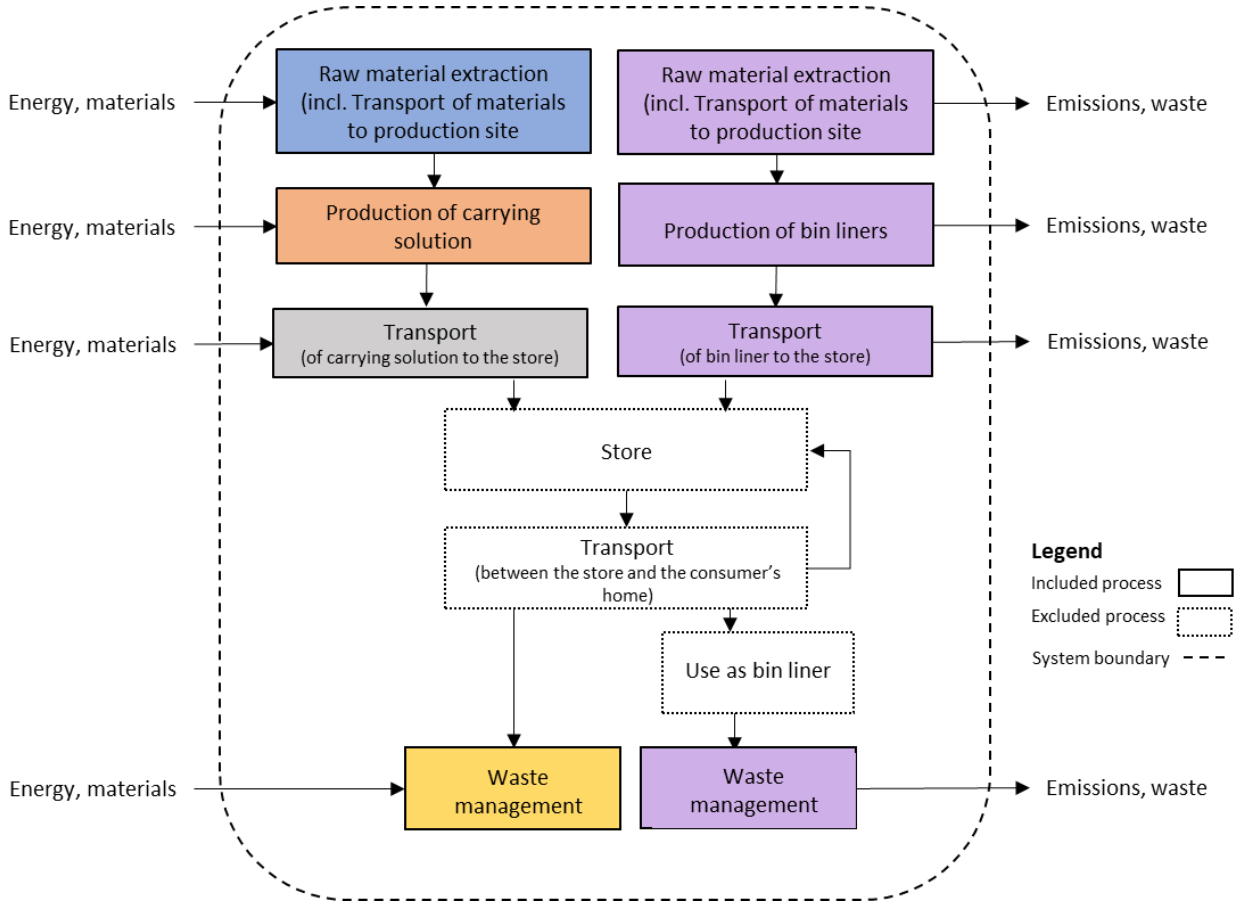


Figure 2: Simplified flowchart illustrating the system boundary applied in this study for all the carriers studied except for the lightweight plastic carriers. The colours used in the boxes correspond to the colours used in the results figures (Chapter 6).

Allocation of impacts related to multi-input and multi-output processes can be avoided by increasing the level of detail of the system or by system expansion (ISO 2006b), i.e., a stepwise procedure. Firstly, allocation should be avoided by increasing the level of detail of the system or by system expansion. If this is not possible, then impacts should be allocated based on physical entities, such as mass. The third option is allocation based on economic value. Two common methods to allocate impacts between life cycles at recycling of bags after use; the simple cut off and the end-of-life approach (Ekvall et al., 2020), will be applied and compared in this study. The simple cut of approach implies that the burdens directly caused by the product is assigned to the product. The end-of-life approach implies that the burden of virgin production and disposal are allocated to the life cycle where the material is lost from the Technosphere, while the burden of recycling is allocated to the product that generates recyclable material. Importantly, the simple cut off and the end-of-life approaches give quite different incentives. Namely, the simple cut off approach give incentives to use recycled materials while the end-of-life approach give incentives to recycle the product after use but gives no incentive to use recycled material. Therefore, both approaches will be applied in this study and the results from applying these will be compared. These approaches are described in more detail in Chapter 3.5.

3.4.1 Cut-off criteria

Cut-off criteria are described as the specification of the amount of material or energy flow, or the level of environmental significance associated with unit processes or product system to be excluded from a study.

The energy and materials used in the foreground systems are entered as shown in the tables in Chapter 5. There has been no conscious cut-off in foreground data. It is possible that data providers have used some form of cut-off, but to the authors' knowledge this is not the case. The database datasets used for energy and materials data described in the tables for each system (see Chapter 5) have their approaches to cut-off, which are documented in the references associated with these databases. The Ecoinvent database has for example a documented approach to cut-off described as consistently seeking to report all datasets as completely as possible, including all by-products and potentially recyclable materials, and consistently include these in allocation and/or substitution calculations (Weidema et al., 2013).

3.5 Modelling recycling in LCA

There are two main modelling approaches used for waste management in this study. For **the base case scenarios** for each carrying solution the **cut-off approach** has been used (so no credits for energy and materials recovered at end of life). Then for **scenario modelling**, system expansion using the **"net scrap approach"** (also known as End of Life substitution, EoL) as described in Bergsma and Sevenster (2013) has been used. These two modelling approaches for material recycling and re-use are described in Lyng and De Sadeleer (2021), from which the text in this chapter has been largely reproduced.

3.5.1 The allocation problem

When a material is recycled, the materials leave one product system to become part of another, as shown in Figure 3.

Figure 3: Recycling is a common process for both the first and the second product system

The material recycling system is a common process for both the first and second product systems. The discussion about how these kinds of common processes should be modelled in LCA has been ongoing since the 90s: there are several different methods for this, but a common consensus about which is the correct method has not yet been reached (Bergsma and Sevenster, 2013), (Allacker et al., 2017), (Ekvall et al., 2020).

The challenge with this type of common process is whether it is to be modelled on an individual product level, or on a system level, with cascades of products. On a product level all processes are assessed in relation to the specific product, while a system level assessment considers all of the products that are connected through the relevant recycling process. This means that one needs to prioritise between these levels / perspectives. Correct modelling of physical flows on a product level leads to double-counting on a system level (Allacker et al., 2014). This can be illustrated by considering a product made from 100% recycled material (recycled content = 100) that is fully recycled (100% recycled) after use. In order to calculate the material flows physically correctly, a recycling process should be included at the start of the product's life cycle and a second recycling process should be included at the end of the product life cycle. This results in two recycling processes in total on the product level. However, on the system level, this leads to double-counting as the recycling process at the start of the product life cycle, which is included as a production process, is also a recycling process for the previous product in the cascade, represented as a

waste management process. In the same way, the recycling process occurring at the end of the product life cycle will also be the start of the next product life cycle in the cascade.

According to PAS 2050 (BSI, 2011), both the recycling and use of recycled material potentially lead to reduced production of virgin material and the associated environmental impacts. The reduced environmental impacts must be allocated either to the use of recycled material, or to recycling of the material at the end of the products life cycle – but not 100% to both. The ISO standards for LCA ISO 14040/44 (ISO, 2006, ISO, 2020) describe a general conceptual framework for allocation in recycling systems, but described in general form and with quite a lot of room for interpretation. This has led to the development of different ways to analyse recycling systems in LCA. Despite the lack of consensus about the scientifically «correct» modelling method, there is agreement that the choice of method can affect the environmental profile results for products (Bergsma and Sevenster, 2013), (Allacker et al., 2017), (Ekvall et al., 2020).

When product systems with different waste management solutions (energy recovery, recycling and re-use) are compared, it is important to ensure that the system boundaries for each product system are defined such that the comparison is fair. As described above, different modelling approaches have been developed for recycling and re-use. It is because of this that we propose to use two different sets of system boundaries in the analyses for this project: 1. Cut-off (recycled content) and 2. System expansion (End-of-Life, EoL, substitution using the net scrap approach). This is proposed in order to assess how robust the conclusions are.

3.5.2 Cut-off/Recycled content

The *cut-off method (or recycled content)* approach is most often used in Environmental Product Declarations (EPD). EPDs for the construction sector include three main modules: A which is raw material extraction / use of recycled material and production, B which is the use phase and C which is waste management. In the cut-off method the system boundaries are set between the first and second product system at the point where the life cycle for the first product ends and the life cycle of the second product begins. Thus, the system boundary is set between transport to recycling and the recycling process, as shown in Figure 4. This means that *Product 1* will only have the impacts associated with the transport to the recycling facility assigned to it, while the recycling process is defined as a production process that is included in the life cycle of *Product 2*. For energy recovery the system boundaries are set after incineration, such that the incineration process is part of the waste management process for *Product 1*, while *Product 2* (the energy generated) is assigned no impacts from burning the waste.

Figure 4: Cut-off method for modelling recycling

3.5.3 Net scrap approach

Also called system expansion or end of life (EoL) substitution. The “net scrap approach” is described (Lyng and de Sadeleer 2021) as in line with the principle «100:100: crediting for avoided virgin production as the difference between R2 and recycled content» (Allacker et al., 2017) and EPD methodology (CEN, 2019)

when taking Module D into account. This modelling approach avoids double-counting of the material recycling benefits on a system level.

System expansion is where no system boundary is set between the first and second product when the environmental impact from *Product 1* is assessed. Instead, the system boundaries are expanded such that the beneficial effect of the waste management process for *Product 1,* which generates one or more new products (in this case *product 2*) which replace other products (substitution), are included. An example for plastic is that recycled plastic can replace virgin plastic. This replacement means that some negative emissions will be included (emission reduction) as it is assumed that the system contributes to reducing the need for extraction of and production of virgin raw materials.

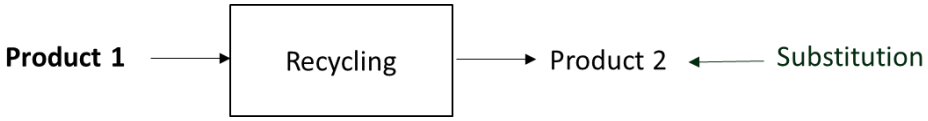


Figure 5: System expansion method for modelling recycling

As described above, the cut-off method is used in EPDs, and the life cycle of a product split up into three modules: A, B and C. A module D is also included in EPDs for the building industry; module D includes *burdens and benefits beyond system boundaries (ref EN 15804)*. The sum of A, B, C and D is equivalent to modelling with system expansion (note EN 15804 does not allow for making the sum. The information in Module D should be kept separate).

Modelling with system expansion means that a series of assumptions need to be made about what is replaced and how much after the product is used. Thus, the following need to be defined:

- The rate of material recycling: the rate at which used carrying solutions are recycled and replace new products.
- Replacement rate: The amount of recycled material that replaces new, virgin material.
- The amount of recycled material the carrying solution is made from (recycled content) when it is put on the market.

3.6 Data Requirements and Data Quality

This study uses specific data representative for the Norwegian context to the extent that this was possible. Specific data for the types and amounts of materials used in various carrying solutions, amounts of groceries to be transported in one shopping trip and the carrying capacity of the different carrying solutions have been collected to represent the current situation in Norway. See Chapter 5 for more details of the specific Norwegian data used. Data was also obtained for Norwegian waste management systems relevant for the carrying solutions assessed. The specific data sources include (but are not limited to) Handelens Miljøfond and their members, Norwegian statistics (SSB), Norwegian Environmental Agency, Green Dot Norway, the consumer research institute in Norway (SIFO) and various industry partners. This Norwegian-specific data was applied as the foreground system data in this study.

Where retailers are listed as the source in Appendix 4, the data was provided by contacting a Norwegian retailer that has supplied the carrier to its customers. Retailers were helpful in providing specific data for their carriers (via e-mail). The data gathering stage and consumer study were performed during restrictions due to the corona pandemic, which imposed limitations on the approaches that could be used.

NORSUS performed a limited observational study to include Norwegian consumer behaviour (see Appendix 1 for details). The data set was gathered by observing individual shopping events in grocery stores. A completely representative sample of Norwegian shopping as a whole would capture events from stores of different supermarket brand, store size, location, and times of the day / week / season. This limited study made observations at two different stores in Oslo and Halden. The observations were conducted in two summer weeks by two master's degree students on secondment to NORSUS. The observations needed to be as unobtrusive as possible, to preserve customers' privacy and also to ensure compliance with infection control requirements. Thus, the size of shopping events was gauged by the number of items purchased (via signals from the cash register or by observing the cashier making a manual entry).

Generic database data has mostly been applied to model the production of the carrying solutions (which typically occurs outside of Norway), with some exceptions. One example is the rucksack for which the required electricity in production was obtained from the factory. Generic database data was used for various raw materials, commonly traded on the global market, and for the transport of these materials. This generic data has primarily been obtained from the Ecoinvent database version 3.7.1, "allocation, cut-off by classification" (Ecoinvent 2020). If data were unavailable in Ecoinvent other sources of information were sought (e.g., scientific literature and reports). Details on the data used for each carrier system can be found in Chapter 5 and Appendix 4. The data quality guideline for the Ecoinvent database can be accessed online, see Weidema et al. (2013).

Recent specific data (2019-2020) was used as far as possible for the current system for carrying solutions and their relevant waste management technologies in Norway. The time-related and technology coverage of the generic data is dependent on the Ecoinvent database and other sources if used.

A data quality assessment of the data used in this study has been performed based on the Annex E (Table E.1 Data quality level and criteria of the UN Environment Global Guidance on LCA database development) of ISO15804 (CEN 2019). The quality level assessment is included in the tables in Chapter 5, where the foreground system data is provided.

The background system data sets (see Appendix 4) were mainly obtained from the Ecoinvent database version 3.7.1 (Ecoinvent 2020) and is presented in more detail in Appendix 4. A data quality assessment column is also included in the tables in Appendix 4. The Ecoinvent database system model "Allocation, cut-off by classification", also called cut-off system model, have been applied; this is based on the Recycled Content, or Cut-off, approach. The underlying philosophy of this approach is that primary (first) production of materials is allocated to the primary user of a material. If a material is recycled, the primary producer does not receive any credit for the provision of recyclable materials. Consequently, recyclable materials are available burden-free to recycling processes, and secondary (recycled) materials bear only the impacts of the recycling processes. Furthermore, producers of wastes do not receive any credit for recycling or re-use of products resulting out of any waste treatment (Ecoinvent 2021).

Where data gaps existed, these were filled using assumptions and estimates. The details of which data this applies to, and the basis / source of the assumptions and estimates are included in Chapter 5 and Appendix 4.

3.7 Life Cycle Impact Assessment

In LCA the LCIA step is where the life cycle inventory (LCI) results are used to calculate potential environmental impacts (also called midpoint categories) and potential damage that can be caused as a result of the life cycle system under study. The figure below describes this process (Jolliet et al. 2016).

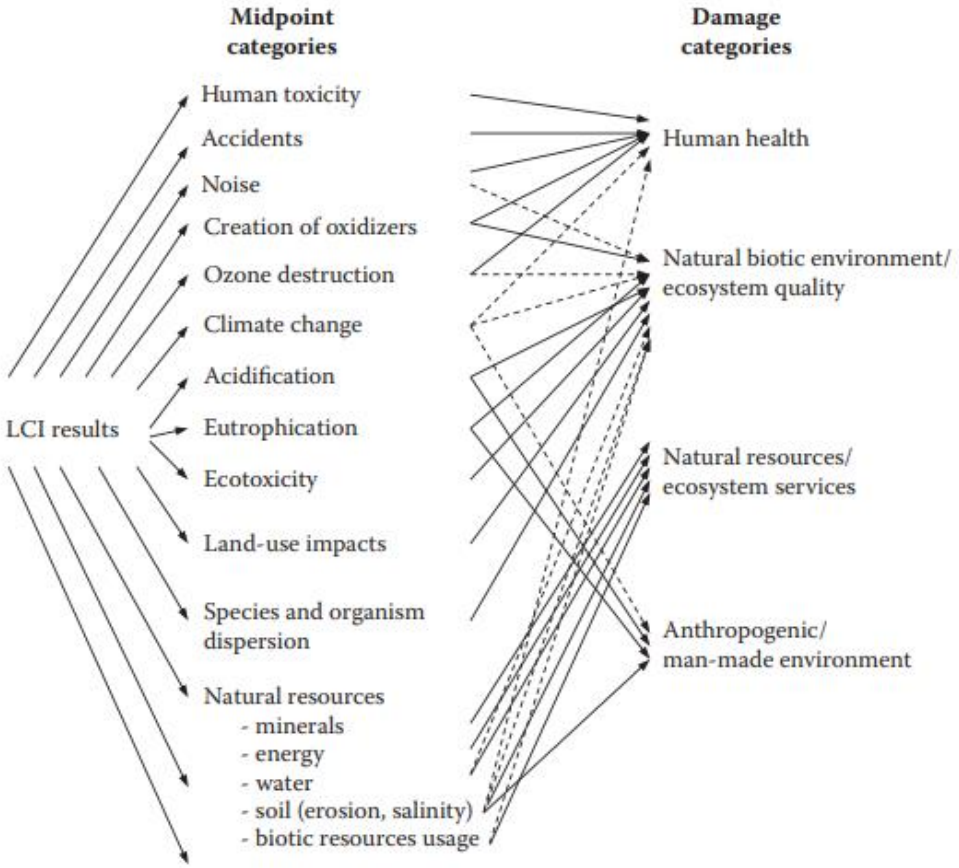


Figure 6: The UNEP-SETAC impact assessment framework (Jolliet et al. 2016).

Appendix 6 provides general equations and descriptions of the calculation procedures involved. As shown, each inventory result links to one or more damage categories through midpoint categories (e.g. ozone damaging emissions can also have climate change effects). Using this analytical framework means that results can be calculated on a midpoint level (see Table 4 for the choices made for midpoint in this study) or at the damage level (i.e. ReCiPe Endpoint in this study).

It was important for this study to cover many different impact categories, as we are comparing product systems with fossil-based and bio-based materials, as well as systems where material is recycled, or incinerated. It was also possible that different transport and energy systems could lead to different types of impacts dominating the results for different systems. Thus, several environmental impact categories were included in this study to be able to identify potential risks for trade-offs (Table 4).

The life cycle impact assessment (LCIA) method of ReCiPe version 2016 (Huijbregts et al., 2017) was applied for the LCIA calculations using the LCA software tool SimaPro, version 9.2 (Pré, 2021a). ReCiPe 2016 was developed in collaboration between the Dutch National Institute for Public Health and the Environment (RIVM), Radboud University Nijmegen, Norwegian University of Science and Technology, and PRé (Pré, 2021c https://pre-sustainability.com/articles/recipe/). ReCiPe was chosen, as it has a broad set of midpoint

impact categories. The method factors are updated regularly to incorporate new data and new research (the latest update was led by Radboud University, The Netherlands). ReCiPe was also one of the recommended methods for several midpoint impact categories by ILCD and UNEP's Life Cycle Initiative (ILCD 2011). The midpoint and endpoint modelling based on ReCiPe 2016 is used, with the exception of two of the midpoint impact categories, water consumption and littering.

For water consumption at midpoint, the AWARE method was applied instead (AWARE 1.04, 2021). This method is the result of a consensus building international process initiated by the UNEP SETAC Life Cycle Initiative to achieve a harmonized method for assessing water use in LCA, involving key method developers and stakeholders through an international collaborative effort.

Another impact that has received a lot of attention recently is plastic littering and its negative effects on the marine environment. Methods that would enable the inclusion of impacts related to plastic littering in LCA are under development, but they are not yet finalized and implemented (UNEP, 2020, de Sadeleer et al., 2021). However, there is sufficient background information to estimate the littering potential associated with different carrying solutions and materials therein (see de Sadeleer et al., 2021). Chapter 3.7.1 details how this was done for this study.

Table 4: Impact categories applied in this study (Huijbregts et al., 2017, Huijbregts et al., 2016, AWARE 1.04, 2021).

Midpoint Impact category and method	**Unit**	**Environmental relevance**
Climate change, ReCiPe 2016 v1.1 midpoint method, Hierarchist[2] version.	kg CO_2-eqv.	Emissions of greenhouse gases cause an increased atmospheric concentration of these gases. This leads to increased radiative forcing and mean global temperature, which ultimately results in damage to human health and ecosystems.
Stratospheric ozone depletion, ReCiPe 2016 v1.1 midpoint method, Hierarchist version.	kg CFC-11-eqv.	Emissions of ozone depleting substances cause an increase of these substances in the stratosphere. This leads to a decrease in the atmospheric ozone concentration, which enables a larger portion of UVB radiation, which is harmful for e.g., human health, to hit the earth.
Photochemical oxidant formation: ecosystem quality, ReCiPe 2016 v1.1 midpoint method, Hierarchist version.	kg NOx	The creation of ozone in the atmosphere, due to photochemical reactions between emitted NOx and non-methane volatile organic compounds, can lead to for example respiratory diseases and negative impacts on vegetation such as growth reductions.
Photochemical oxidant formation: human health, ReCiPe 2016 v1.1 midpoint method, Hierarchist version.		
Terrestrial acidification, ReCiPe 2016 v1.1 midpoint method, Hierarchist version.	kg SO_2-eqv.	Atmospheric deposition of acidic substances that have become emitted, such as sulphates, causes the acidity level in soil and water bodies to deviate from its optimum which affects plants and animals negatively.
Freshwater eutrophication, ReCiPe 2016 v1.1 midpoint method, Hierarchist version.	kg P-eqv.	Emissions of nutrients into the environment leads to that some species, such as algae, grow fast and further causes a depletion of oxygen in, for example, lakes and rivers in turn causing loss of biodiversity.

[2] The ReCiPe method has three possible value perspectives that can be chosen. These are: individualist, hierachist and egalitarian, see Table 5: The ReCiPe method cultural perspectives.

Midpoint Impact category and method	Unit	Environmental relevance
Marine eutrophication, ReCiPe 2016 v1.1 midpoint method, Hierarchist version.	kg N-eqv.	Similarly to freshwater eutrophication, Marine eutrophication is a process driven by the enrichment of marine water by nutrients, especially compounds of nitrogen and/or phosphorus, leading to: increased growth, primary production and biomass of algae; changes in the balance of organisms; and water quality degradation. The consequences of eutrophication are undesirable if they appreciably degrade ecosystem health and biodiversity and/or the sustainable provision of goods and services.
Mineral resource scarcity, ReCiPe 2016 v1.1 midpoint method, Hierarchist version.	kg Cu-eqv.	The extraction of mineral resources leads to an overall decrease in ore grade, which in turn will cause an increase in the amounts of ore mined per kg of extracted mineral resource. The extraction of minerals and metals leads to resource depletion.
Fossil resource scarcity, ReCiPe 2016 v1.1 midpoint method, Hierarchist version.	kg oil-eqv.	Increased extraction of fossil resources causes increased costs to extract such resources since fossil fuels with the lowest extraction costs becomes extracted first. The extraction of fossil fuels leads to resource depletion.
Fine particulate matter formation, ReCiPe 2016 v1.1 midpoint method, Hierarchist version.	kg PM2.5-eqv.	Exposure to emissions of particulate matter can cause substantial health effects in humans and other organisms.
Human toxicity: cancer, ReCiPe 2016 v1.1 midpoint method, Hierarchist version.	kg 1,4-DCB-eqv.	Exposure to emitted substances can lead to cancer in humans.
Human toxicity: non-cancer, ReCiPe 2016 v1.1 midpoint method, Hierarchist version.	kg 1,4-DCB-eqv.	Exposure to emitted substances can lead to negative (non-cancerogenic) effects in humans.
Freshwater ecotoxicity, ReCiPe 2016 v1.1 midpoint method, Hierarchist version.	kg 1,4-DCB-eqv.	Exposure to emitted substances can lead to disease in organisms and further losses of biodiversity.
Terrestrial ecotoxicity, ReCiPe 2016 v1.1 midpoint method, Hierarchist version.	kg 1,4-DCB-eqv.	
Marine ecotoxicity, ReCiPe 2016 v1.1 midpoint method, Hierarchist version.	kg 1,4-DCB-eqv.	
Ionizing radiation, ReCiPe 2016 v1.1 midpoint method, Hierarchist version.	kBq Co-60 eq	Exposure to ionizing radiation can cause chemical changes in cells and damage DNA. This may increase the risk of developing certain health conditions, such as cancer.
Land use, ReCiPe 2016 v1.1 midpoint method, Hierarchist version.	m^2a crop eq	Land use includes the effect on biodiversity of anthropogenic land transformation or land occupation. Anthropogenic land use influences the species richness and species loss in terrestrial ecosystems. Certain land use types are related to species loss resulting from annual crop production
Water deprivation, AWARE	m^3 world-eq.	The AWARE method will be used to calculate the relative Available WAter REmaining per area in a watershed after the demand of humans and aquatic ecosystems has been met. It assesses the potential of water deprivation, to either humans or ecosystems, building on the assumption that the less water remaining available per area, the more likely another user will be deprived

The ReCiPe method provides the option for the LCA practitioner to choose from three different perspectives: individualist, hierachist and egalitarian. The table below describes these in brief (Pré 2021b).

In the absence of strong positive reasons to choose either the individualist or egalitarian perspective, the hierarchist perspective has been selected. There appear to be no particular reasons for strong technological-optimism (or pessimism) in the context of the study and the value chains under consideration.

Table 5: The ReCiPe method cultural perspectives

Perspective	Description
Individualist	Short term, optimism that technology can avoid many problems in future
Hierarchist	Consensus model, as often encountered in scientific models, this is often considered to be the default model.
Egalitarian	Long term based on precautionary principle thinking

3.7.1 Littering

The amount of plastic litter emitted to the environment has been calculated using littering rates taken from Briedis et al. (2019), the relevant extract of Table 14 *ibid* is included below. Note that the focus for this calculation is on the carrier product itself and not on the whole product life cycle.

Table 14 - Summary of Litter Rates Modelled for Items/Alternatives (estimated rates)

Item	SUP	SUNP	MU
Lightweight plastic carrier bags	5%	5%	0.0117%

Factors for lightweight plastic carrier bags (both single use, SUP, and multiuse, MU) are combined with the weight of plastic material per functional unit (FU) to provide mass littered per FU. Briedis et al. (2019) also states that *It is assumed that a large proportion of the littered items are subsequently intercepted (i.e. collected and disposed of either through waterway cleaning, street cleaning, etc.) in Norway (~90% litter interception), through street cleaning/ sweeping efforts. Of the remaining 10% of litter in the wider environment, it is assumed for the majority of items that 50% of this stream final enters the marine environment (through drains, canals, rivers, direct disposal in marine environment).* This would mean that only 5% of the littered plastic would become marine litter.

If the value chosen for this study (5% of the littered amount, which is 2.5% of the bag mass per functional unit) is compared to the values used for Norway by the plastic leakage project (Peano et al., 2020, referred to as PLP in the following) the values are similar. Plastic bags littered during use would be products of a large size (>25cm), on the go littering of this type of product is expected to be 1%. Mismanaged waste in Norway has a factor of 2.4% loss. Plastic bags are a product with “low residual value” (i.e. the likelihood of collection through the informal waste collection system, due to the value of the material or product) and thus the factors from the PLP study that are relevant are: initial release rate to ocean and freshwater is 5%, the share of mismanaged waste not released to the environment 0% and redistribution rate (% of macroplastics released into ocean and freshwater are assumed to be ultimately redistributed into the ocean) is 100%. Thus, if one combined mismanaged waste and littering factors in the PLP reference (and supplementary material), the littered amount (1%) would end up in the ocean and of the mismanaged waste 5% would end in the ocean, i.e. 5% of 2.4%, which is 1.2%. The total factor from the PLP study would be 2.4%.

Plastic carriers have the potential to entangle sea creatures when they are littered, and to break down into micro- and nano-plastic (MNP) particles in the marine environment. There is a lot of ongoing work on effects

of plastic in marine environments. The research front for this subject includes Woods et al. (2021) and Lavoie et al. (2021). In Woods et al. (2019) an effect factor approach for entanglement was developed, providing effect factors that are spatially differentiated, and specific to species group. A more generalised macro effect factor has been submitted for scientific peer review, but is not yet published (Verones 2021). Lavoie et al. (2021) who have developed an effect factor for the quantification of the physical impact of MNPs on biota. This factor for physical effects is:

- The recommended EF value of 72.9 PAFm3kg^{-1} (Lavoie et al., 2021)

There are several other potential effects MNPs can have on the environment, also outside of the marine compartment (e.g., soil and freshwater, see Woods et al. 2021), but work on these effects is ongoing and have not been quantified in this study.

The impact assessment method chosen for marine ecotoxicity (ReCiPe) provides midpoint impact results in units of "1,4-DCB" in the results tables in Chapter 6; this unit is 1,4- dichlorobenzene equivalents. The units for this effect factor are PAFm3kg−1, which is the Potentially Affected Fraction of species that experience an increase in stress for a change in contaminant exposure above a predefined effect level. In life cycle impact assessment, it is normal to combine an effect factor with a fate factor to provide the characterisation factor needed to calculate an impact (this is also described in Woods et al. 2019 and Lavoie et al. 2021). The approach above amounts to the fate factor being considered as 1. The assumptions above mean that all the bags that enter the ocean remain in the ocean (disregarding degradation, fragmentation, and settling/sedimentation for example). This makes the assessment more conservative (overestimating the macroplastic impacts) for the use phase.

3.7.1 Endpoint damage categories

As described in 3.7 (see also Figure 6), the life cycle impact assessment (LCIA) method of ReCiPe version 2016 (Huijbregts et al., 2017) was applied for the LCIA calculations using the LCA software tool SimaPro, version 9.2 (Pré, 2021a). The damage level, or endpoint part of this ReCiPe model uses the endpoint damage categories shown in Table 6.

Table 6: End-point damage categories applied in this study (Huijbregts et al., 2017, Huijbregts et al., 2016).

Endpoint damage category and method	Unit	Midpoint categories
Human health, ReCiPe 2016 v1.1 Endpoint, Hierachist perspective.	DALY (disability adjusted life years)	This endpoint combines human health damage from the following mid-point impact categories: Climate change, human health (HH); Stratospheric ozone depletion; Ionizing radiation; Photochemical oxidant formation: human health; Fine particulate matter formation; Human toxicity: cancer; Human toxicity: non-cancer; Water consumption HH.
Ecosystems, ReCiPe 2016 v1.1 Endpoint, Hierachist perspective.	Species.year	This endpoint combines Climate change, terrestrial ecosystems (TE); Climate change, freshwater ecosystems; Photochemical oxidant formation TE, Terrestrial acidification; Freshwater eutrophication; Marine eutrophication; Terrestrial ecotoxicity; Freshwater ecotoxicity; Marine ecotoxicity; Land use; Water consumption TE and Water consumption aquatic ecosystems.
Resources, ReCiPe 2016 v1.1 Endpoint, Hierachist perspective.	USD 2013	This endpoint combines Mineral resource scarcity and Fossil resource scarcity.

See Table 4 for the environmental relevance of the impact categories included in each endpoint shown in Table 6.

3.8 Sensitivity Analysis

A sensitivity analysis will be conducted to assess parameter uncertainties. Foreground data parameter values, e.g., the number of uses of carriers, will be altered one at a time and changes in results will be investigated relative to the defined base case scenario. The scenarios are detailed in tables in Chapter 6.5, where the sensitivity analysis results are also presented.

3.9 Reporting and Critical Review

The study is documented in this scientific report. Since the results from the study will include comparative assertions that are intended to be disclosed to the public, a critical review has been carried out in accordance with ISO (2006a, b). The purpose of the critical review was to ensure a robust and transparent study. The review was be conducted by a committee consisting of three LCA experts, two industry experts and an observer:

- Tomas Ekvall (chairman), Chalmers University of Technology
- Anne-Marie Boulay, Polytechnique Montreal and CIRAIG
- Ian Vázquez-Rowe, PUCP
- Rolf Erling Eriksen, Leader environment, Vinmonopolet (industry expert)
- Halvard Hauger, Fagsjef miljø, Norgesgruppen (industry expert)
- Sjur Kvifte Nesheim, Analyst environment & society, Norwegian Retailers' Environment Fund (observer)

The committee will be involved in the project from the start and three meetings of the panel are proposed with the following meeting themes:

1. Goal and scope proposal
2. Dataset and calculation methods for environmental impacts to be included in the analysis
3. Presentation and discussion of results, input to the final report

In addition to the input on the study provided from the critical review, the Norwegian Retailer's Environment Fund's observer was invited to comment and provide information to be used in the study. The panel provided detailed comments to a report draft sent to them in October 2021. NORSUS received their comments and provided a second, revised draft, as the basis for their final comments, a summary of these can be found in Chapter 8.

4 Comparison of goal and scope for other carrier studies

The goal and scope for this study has been developed based on experience with LCA studies, dialogue with the critical review panel and literature. This study did not aim to be a literature study, but as the results are somewhat different to some newer studies, a brief overview of the differences between the goal and scope noted by the authors of this study are included here. This is in order to assist in interpretation of the results presented in Chapter 6 in relation to other studies. The studies below are those considered when defining the goal and scope for this study. This does not represent an extensive, exhaustive list of literature, please refer to other literature studies for that (e.g. Ekvall et al. 2020 or Gaudreault 2020).

Table 7: A summary of some relevant LCA studies on, or including, carrying solutions for shopping.

Author	Year	Country	Functional unit	Functional unit (operationalization)	Alternatives assessed	Comments on data requirements and data quality	LCIA indicators included
Civancik-Uslu et al.	2019	Spain	The bags required to bring supermarket goods home to an average Spanish household for a year: 408 single-use bags.	The number of bags required is assumed not to vary between the bags depending on the size of the bag but only on the number of times the bag is used.	(5) HDPE (10% recycled content), LDPE (100% virgin), paper (100% recycled), biodegradable (50% starch and 50% polycaprolactone), PP woven (100% virgin PP and 100% recycled paper for carton layer)	Data on HDPE, LDPE and PP production from Ecoinvent. Data on biodegradable and paper bag production from producers.	Climate change, acidification, eutrophication, photochemical ozone creation, abiotic resource depletion, water use, littering potential.
BIOSPRI	2018	Europe	1 single-use all-purpose lightweight plastic carrier bag with the volume of 20 litres and 10 kg weight holding capacity	The bags compared were functionally approximately equivalent (compared on a 1-1 basis)	(3) LDPE (100% virgin), biodegradable (polyester-complexed starch biopolymer bags produced from maize and potato), bio-based LDPE (sugarcane ethanol)	Aggregated data on LDPE production from PlasticsEurope. Data on biodegradable alternative production from large industrial producer. Data on bio-based LDPE from the relevant region	Climate change, particulate matter, photochemical ozone formation, acidification, terrestrial eutrophication, marine eutrophication, abiotic depletion, non-renewable energy use. (n.b.,

						or adjusted from PlasticsEurope.	some more impacts were analysed for some carriers, but the ones listed were available for all 3)
Kimmel	2014	USA	Bags used by U.S. consumers to transport the shopping items for one trip (52 items) from the grocery store to the consumer's home in 2012	Four different scenarios of numbers of trips were analysed: 1, 3.1, 14.6 and 44 trips to the grocery store. For all these functional units, a single set of reusable bags was assumed to be sufficient, while a new set of single-use bags where used for each trip.	(6) HDPE (100% virgin), HDPE (30% recycled content), paper (40% recycled content), paper (100% recycled content), LDPE, non-woven polypropylene (with LDPE stability insert)	Data from US-EI2.2 database in turn derived from Ecoinvent.	Climate change, water depletion, cumulative energy demand, terrestrial acidification, freshwater eutrophication, marine eutrophication, human toxicity, terrestrial ecotoxicity, freshwater and marine ecotoxicity, fossil fuel depletion, photochemical oxidant formation
Edwards et al.	2011	UK	Carrying one month's shopping (483 items) from the supermarket to home in the UK in 2006/07	The volume of the bags is used to calculate the number of bags needed to meet the functional unit. Calculated how many times a paper bag/reusable bag must be used before it affects climate less than conventional HDPE bag.	(7) HDPE, oxo-degradable HDPE, biodegradable starch-polymer/bipolymer blend ("starch polyester"), paper, LDPE bag for life, non-woven PP durable bag (multiuse), cotton (multiuse).	Most data used in this study are from Ecoinvent v2, specific foreground data, e.g. on UK shopping and size and weight of carriers, from retailers.	Global warming potential, depletion of abiotic resources, photo-oxidant formation, eutrophication, acidification, human toxicity, aquatic and terrestrial toxicity.
Edwards and	2012	UK	"The production, use and disposal	The study builds on / is a response to Edwards	(3) HDPE, oxo-biodegradable HDPE and	HDPE: Plastics Europe data, energy	Global warming potential, litter

Parker 2012			of a single conventional light weight bag and alternatives of the same capacity"	et al 2011, focussing on comparison of conventional, oxo-biodegradable and bio-based bags	bio-based (starch-polymer/bipolymer blend ("starch polyester") Recycled content in HDPE (conventional and oxo) is also analysed.	required for recycling is based on internal confidential data and is assumed to take place in China. Recyclable material assumed collected in the UK and transported recycling in China. Ecoinvent data on the production of modified starch is used for the starch polyester bag The quantity of carbon dioxide absorbed from the atmosphere was adapted based on a 2010 reference. Additional data from data collected by Edwards et al. 2011.	effects, abiotic resource depletion, acidification, eutrophication, ozone layer depletion, photochemical oxidation, human toxicity, freshwater aquatic ecotoxicity, marine aquatic ecotoxicity, terrestrial ecotoxicity.
Muthu et al.	2011, 2012	China, Hong Kong and India	The annual number of bags used for grocery shopping by an average person	In China and Hong Kong, they estimate the number of single-use bags to be 1095 bags per person and year. For India the estimate is 150 bags per person and year. Assume that more durable bags (PP	(2, 2011 study) HDPE (100% virgin), kraft paper (100% virgin) (4. 2012 study) HDPE (100% virgin), kraft paper (100% virgin), non-woven PP, woven cotton	Cradle to gate data from literature (i.e., studies for Environment Australia in 2002 and 2004) for material consumption and energy needed for	Eco-indicator 99 single score results, calculated using Simapro 7.1. in the 2011 reference. In the 2012 reference climate change results were

				or cotton) replaces a certain amount of single-use bags (paper and plastic)		manufacturing process. Data for disposal scenarios was specific data collected through surveys of users of plastic and paper bags in China, Hong Kong and India.	calculated using SIMAPRO 7.2.
Mattila et al.	2011	Finland	The bags required to carry groceries for a year to an average Finnish household	This is translated into 100 single-use bags, based on how many bags were actually bought in Finland in the year 2007. All single-use bags considered to be functional equivalent (same capacity). Cotton bag assumed to replace about 50 single-use bags.	(5) Biodegradable (thermoplastic starch with additives), cotton, bleached paper, LDPE (100% virgin), LDPE (60% recycled content)	Primary process data collected from the manufacturers of Finnish plastic, paper, and biodegradable bags and for paper recycling. Finnish averages were used for energy production and waste treatment. The Ecoinvent 2.0 database was used for filling data gaps and for primary production of polyethylene, cotton, and modified starch.	Carbon footprint (CO_2-equ.)
Khoo et al	2010, 2010	Singapore	The carrying capacity of the standard bag: 20 kg	Reuse of bags not considered. One of the compared bags has lower carrying capacity	(2) polyethylene PP (fossil-based), polyhydroxyalkanoate (PHA, bio-based)	Literature and databases.	Global warming, acidification and photochemical ozone formation.

				(thus need >1 bag of this one to be in line with the functional unit).			
Ahamed et al	2021	Singapore	820 million bag equivalents	The annual uptake of plastic bags from supermarkets in Singapore amounted to 820 million bags, which was roughly 141 bags per person in that year. For reusable bags it was assumed that the bags were reused 50 times amounting to 16.4 million bags per FU.	(5) single-use (HDPE, biodegradable plastic, kraft paper) bags and reusable (cotton, polypropylene non-woven) bags	LCI data was developed from various sources including laboratory analysis, Ecoinvent database, compu-tation and estimation from published literature and reports, and calculations from industrial and commercial sources.	Global warming potential, freshwater- and marine-aquatic ecotoxicity potentials, terrestrial ecotoxicity potential, human toxicity potential, acidification potential, eutrophication potential, and abiotic depletion potential (fossil)
Bisinella et al	2018	Denmark	Carrying one time grocery shopping with an average volume of 22 litres and with an average weight of 12 kilograms from Danish supermarkets to homes in 2017 with a (newly purchased) carrier bag	Number of bags required calculated based on volume and weight capacities.	(14) Low-density polyethylene (LDPE), 4 types: an LDPE carrier bag with average characteris-tics, an LDPE carrier bag with soft handle, an LDPE carrier bag with rigid handle and a recycled LDPE carrier bag; Polypropylene (PP), 2 types: non-woven and woven; Recycled polyethylene terephthalate (PET); Polyester (of virgin PET polymers); Starch-complexed biopolymer;	Material production data was mostly based on the consequential version of the Ecoinvent database, version 3.4. Data for production of the carrier bags was obtained from a literature review of existing LCA studies on supermarket carrier bags. Additional data on	Climate change, Ozone depletion, Human toxicity, cancer effects, Human toxicity, non-cancer effects, Photochemical ozone formation, Ionizing radiation, Particulate matter, Terrestrial acidification, Terrestrial eutrophication, Freshwater eutrophication,

			Note: the functional unit basis of a newly purchased one time grocery shopping bag, means that material quantity used for multiuse bags for this one time shopping event (e.g. cotton) is modelled as if the multiuse bag was a single use bag in practice.			the material composition and on the waste management technologies were obtained from the library of the LCA model EASETECH. EASETECH data and process models were used in order to model waste incineration in Denmark, as well as recycling in Europe. Management of rejects from recycling outside Denmark was modelled using generic waste management processes for Europe. Note new plastic production data used as a proxy for recycled material.	Marine eutrophication, Ecosystem toxicity, Resource depletion, fossil, Resource depletion, abiotic, Water resource depletion.
Ciraig	2017	Canada	Two functional units were explored: The bags needed to transport 1	Number of bags required calculated based on volume.	(8) Single use: -HDPE - Oxodegradable HDPE	primary data are collected through a electronic questionnaire sent to stakeholders and	Endpoint (damage category) LCIA results were calculated for: - Human health

			litre of products purchased by a private individual when shopping in Quebec in 2016 And The bags needed to transport 100 litres of products purchased by the private individual when shopping at Quebec in 2016		- Compostable bioplastic bag with straps (starch-polyester mixture - Thick plastic bag (LDPE, with cut handles) - Paper bag (unbleached kraft paper) Re-usable bags: - Woven PP bag - Non-woven PP bag - Cotton bag	through telephone interviews. Secodary data from Ecoinvent, an internal CIRAIG database, public databases, literature and expert judgements.	- Ecosystem quality - Use of fossil resources Using IMPACT World + ReCiPe (perspective "Hierarchist") was also used for a sensitivity analysis A littering indicator "Abandonment in the environment" was also calculated.

All of the studies in the table above have a different geographical focus than this study. Many have different functional units, although the functional units used by Civancik-Uslu et al. (2019), Kimmel 2014, Edwards et al. (2011 and 2012), Muthu et al. (2011 and 2012) and Mattila et al. (2011) are similar, in that they are related to an amount of shopping over a given period of time. However, none of these studies has the bin liner function included in their functional unit. In Chapter 7 there is some discussion about similarities and differences in conclusions from some of these studies.

5 Systems studied

In this chapter, the systems for the different carrying solutions and the bin liner are described. The foreground system data applied in this study are summarised in Table 8 - Table 16. The background system data sets were mainly obtained from the Ecoinvent database version 3.7.1 (Ecoinvent 2020) and is presented in more detail in Appendix 4.

Specific data, including locations for production facilities were provided by specific Norwegian retailers. Five different retailers and carrier suppliers provided such information for this study. In order to preserve confidentiality these companies are labelled “Norwegian retailer”.

The tables in this chapter have a data quality assessment column. The data quality assessment approach is described in Chapter 3.6. Assumptions made are clearly marked, but no data quality assessment is made for these.

5.1 Low-density polyethylene (LDPE) plastic bag

Light weight carrier bags made of LDPE are a common solution for carrying groceries home from stores in Norway. LDPE granulate bought on the global plastic market is transported to Sweden. A proportion of the granulate (80%) is from recycled plastic, while the remainder is from virgin plastic. The LDPE bags included in this study are produced in a factory in Sweden and then become transported by lorry to stores in Norway. The bags are bought in-store and used for carrying groceries. After use, 80% of the plastic bags are used as bin liners (see SIFO (2019)). In the waste treatment phase, 89% of discarded plastic bags (including the plastic bags reused as bin liners and then discarded) were assumed to become incinerated with energy recovery in Norway. The remainder (11%) was assumed to become sent to material recycling in Europe. Data on the recycled content and weight of LDPE bags were supplied by a Norwegian home delivery shopping service while the production site was set based on information from a Norwegian grocery store company. The foreground system data applied for the LDPE bag are summarised in Table 8 and the type of background system data applied is presented in Appendix 4.

Table 8. Foreground system data applied for the low-density polyethylene (LDPE) plastic bag

Parameter [unit]	Value	Reference	Data quality assessment[3]
Parameters included in base case scenario (and in the end-of-life scenario)			
Recycled content in LDPE bag [%]	80	Norwegian retailer	Very good[4]
Weight of LDPE bag [g]	19	Norwegian retailer	Very good
Distance from production site (Arvika) to store (Oslo) [km]	160	Norwegian retailer; Google maps	Good[5] (as for Oslo location and not the whole of Norway)
Share of discarded LDPE bags that becomes incinerated (rest is recycled) [-]	0.89	Handelens Miløfond (2021a)	Very good
Distance from consumer to incineration [km]	85	Raadal et al. (2009)	Fair[6] (geography and technical aspects are very good, but the time scale is "poor", as the source data is 12 years old).
Distance from consumer to recycling [km]	905	Raadal et al. (2009)	Fair (see previous row).
Parameters included in net scrap approach scenario only			
Lower heating value (LHV) for LDPE [MJ/kg]	38.9	Areepraserta et al. (2016)	LHV data is physical property data, so although the timescale is more than 3 years ago, this data is deemed very good.
Efficiency of incineration process in Norway	0.69	Calculated from 81% energy efficiency (energiutnyttelsesgrad) for Norwegian incineration plants in 2019 Avfall Norge (2020) and 85% efficiency for waste incineration, Raadal et al. (2009).	Good (geography and technical aspects are very good, the time scale of the energy efficiency is very good. The efficiency of waste incineration is by expert assessment good, but classified as "poor"[7], as the source for the detailed data behind this is 12 years old).
Substitution rate for material in LDPE bag[8]	0.75	Zampori and Pant. (2019)	Very good

[3]See Chapter 3.6, synopses of the definitions of the data quality assessment terms are given in foot notes.
[4] Geographically representative for this study, data for processes and products under study, less than 3 years between the reference year and year of study.
[5] Average data from a larger area of study, data for processes and products with similar technology, less than 6 years between the reference year and year of study.
[6] Data from an area with similar production conditions, data for processes or products under study but from different technology, less than 10 years of difference between the reference year and year of study.
[7] Data from an area with slightly similar production conditions, data for related processes or products, less than 15 years of difference between the reference year and year of study.
[8] This is the fraction of material that is assumed to replace virgin material when recycled.

Share of plastic bags that replaces bin liners after use (the rest becomes incinerated)	0.8	Assumed based on SIFO (2019) and Handelens Miløfond (2020)	Very good

5.2 High-density polyethylene (HDPE) plastic bag

The HDPE bags, with a recycled content at 80%, are produced in Southeast Asia and then transported by boat to Europe (Oslo). They are bought in-store and used for groceries. Similar to the LDPE bags, the HDPE bags also become reused as bin liners to a certain extent and the modelling of the waste management of the HDPE bags is similar to that for the LDPE bag (Chapter 5.1). The foreground system data applied for the HDPE bag was partly provided by a Norwegian grocery store company. The data are summarised in Table 9 and the type of background system data applied is presented in Appendix 4.

Table 9. Foreground system data applied for the high-density polyethylene (HDPE) plastic bag

Parameter [unit]	Value	Reference	Data quality assessment[9]
Parameters included in base case scenario (and in the net scrap approach scenario)			
Recycled content in HDPE bag [%]	80	Norwegian retailer	Very good
Weight of HDPE bag [g]	19	Norwegian retailer	Very good
Distance from production site (Southeast Asia) to store (Oslo) [km]	15 900	Norwegian retailer, Sea-distancesorg. (2021)	Good (as for Oslo location and not the whole of Norway)
Share of discarded HDPE bags that becomes incinerated (rest is recycled) [-]	0.89	Handelens Miløfond (2021a)	Very good
Distance from consumer to incineration [km]	85	Raadal et al. (2009)	Fair (geography and technical aspects are very good, but the time scale is "poor", as the source data is 12 years old).
Distance from consumer to recycling [km]	905	Raadal et al. (2009)	Fair (see previous row).
Parameters included in net scrap approach scenario only			
Lower heating value (LHV) for HDPE [MJ/kg]	34.1	Areepraserta et al. (2016)	LHV data is physical property data, so although the timescale is more than 3 years ago, this data is deemed very good.
Efficiency of incineration process in Norway	0.69	Calculated from 81% energy efficiency (energiutnyttelsesgrad) for Norwegian incineration plants in 2019 Avfall Norge	Good (geography and technical aspects are very good, the time scale of the energy efficiency is very good. The efficiency of waste incineration is

[9] See Table 8 for footnotes about the data quality assessment and definitions of the terms "very good", "good" etc.

		(2020) and 85% efficiency for waste incineration, Raadal et al. (2009).	by expert assessment good, but classified as "poor", as the source for the detailed data behind this is 12 years old).
Substitution rate for material in HDPE bag	0.9	Zampori and Pant. (2019)	Very good
Share of plastic bags that replaces bin liners after use (the rest becomes incinerated)	0.8	SIFO (2019) and Handelens Miløfond (2020)	Very good

5.3 Bin liner

In Norway, it is quite common that single-use plastic bags are reused as bin-liners (SIFO, 2019). The other carriers assessed in this study, i.e. paper bag, nylon bags PET bag, cotton bag, cardboard box and rucksack, will not be reused as bin liners. Therefore, additional bin liners need to be considered for these carriers to fulfil the functional unit of this study and thus be comparable to the plastic bags. The data for the bin liner were provided by a Norwegian company that sells packs of bin liners to consumers and by a Norwegian grocery store company. Based on the data for LDPE bags (Chapter 5.1) it was assumed that the bin liners were produced in Sweden. After use, bin liners are assumed to become incinerated with the residual waste, which is typical for Norwegian conditions. The foreground system data applied for the bin liner is summarised in Table 10 and the type of background system data applied is presented in Appendix 4.

Table 10. Foreground system data applied for the low-density polyethylene (LDPE) bin liner

Parameter [unit]	Value	Reference	Data quality assessment[10]
Parameters included in the base case scenario (and in the net scrap approach scenario)			
Recycled content in LDPE bin liner [%]	100	Norwegian retailer	Very good
Weight of LDPE bin liner [g]	9	Norwegian retailer	Very good
Distance from production site (Sweden) to store (Oslo) [km]	160	Assumed based on data on production site for LDPE bags from Norwegian retailer	Good (as for Oslo location and not the whole of Norway)
Share of discarded bin liners that becomes incinerated [-]	1	Assumption by the authors	
Distance from consumer to incineration [km]	85	Raadal et al. (2009)	Fair (geography and technical aspects are very good, but the time scale is "poor", as the source data is 12 years old).

[10] See Table 8 for footnotes about the data quality assessment and definitions of the terms "very good", "good" etc.

Parameters included in the net scrap approach scenario only			
Lower heating value (LHV) for LDPE [MJ/kg]	38.9	Areepraserta et al. (2016)	LHV data is physical property data, so although the timescale is more than 3 years ago, this data is deemed very good.
Efficiency of incineration process in Norway	0.69	Calculated from 81% energy efficiency (energiutnyttelsesgrad) for Norwegian incineration plants in 2019 Avfall Norge (2020) and 85% efficiency for waste incineration, Raadal et al. (2009).	Good (geography and technical aspects are very good, the time scale of the energy efficiency is very good. The efficiency of waste incineration is by expert assessment good, but classified as "poor", as the source for the detailed data behind this is 12 years old).

5.4 Paper bag

Retailers selling light weight plastic bags in Norway, commonly have a paper bag (0% recycled content) that the consumer can choose to purchase as an alternative to plastic bags. These bags are heavier than the plastic bags and usually made of brown kraft paper with handles. The paper bags included in this study are produced in Sweden and then transported to Norway by lorry. The raw materials for the paper bag were modelled to represent average European production. In the waste management of the paper bag, 29% was considered to be incinerated with energy recovery in Norway and the remainder sent to material recycling in Norway or abroad. This was based on data from Envir (2015) who conducted a pick analysis of residual waste in Tromsø and Karlsøy municipalities in Norway and stated that about 72% of generated paper waste is collected for material recycling while the remainder ends up being incinerated together with the residual waste. Furthermore, SSB (2021), stated that in 2019 about 98% of the collected paper, cardboard and carton waste in Norway was sent to material recycling while about 2% was sent to incineration (about 0.3% sent to other treatment was excluded). About 50% of the collected paper was assumed to be recycled in Norway while the other 50% was assumed to be recycled abroad based on Raadal et al. (2009). The foreground system data applied for the paper bag are summarised in

Table 11 and the type of background system data applied is presented in Appendix 4.

Table 11. Foreground system data applied for the paper bag

Parameter [unit]	**Value**	**Reference**	**Data quality assessment[11]**
Parameters included in base case scenario (and in the net scrap approach scenario)			
Recycled content in paper bag [%]	0	Norwegian retailer	Very good
Weight of paper bag [g]	63	Norwegian retailer	Very good
Distance from production site (Sweden) to store (Oslo) [km]	440	Norwegian retailer; Google maps	Good (Oslo location, not the whole of Norway)

[11] See Table 8 for footnotes about the data quality assessment and definitions of the terms "very good", "good" etc.

Share of discarded paper bags that becomes incinerated (rest is recycled) [-]	0.29	Envir (2015) and SSB (2021)	Good (some of the data is very good, but the time scale aspect of Envir 2015 is poor).
Distance from consumer to incineration [km]	85	Raadal et al. (2009)	Fair (geography and technical aspects are very good, but the time scale is "poor", as the source data is 12 years old).
Share of paper bags that are recycled in Norway (remainder are recycled abroad) [-]	0.5	Raadal et al. (2009)	Fair (see previous row).
Distance from consumer to recycling in Norway [km]	806	Raadal et al. (2009)	Fair (see above).
Distance from consumer to recycling abroad [km]	806[12]	Raadal et al. (2009)	Fair (see above).
Parameters included in net scrap approach scenario only			
Lower heating value (LHV) for paper [MJ/kg]	15	Raadal et al. (2009)	LHV data is physical property data, so although the timescale is more than 3 years ago, this data is deemed very good.
Efficiency of incineration process in Norway	0.69	Calculated from 81% energy efficiency (energiutnyttelsesgrad) for Norwegian incineration plants in 2019 Avfall Norge (2020) and 85% efficiency for waste incineration, Raadal et al. (2009).	Good (geography and technical aspects are very good, the time scale of the energy efficiency is very good. The efficiency of waste incineration is by expert assessment good, but classified as "poor", as the source for the detailed data behind this is 12 years old).
Substitution rate for material in paper bag	1	Zampori and Pant. (2019)	Very good

5.5 Nylon bag

The nylon bag, with a recycled content of 0%, is produced in East Asia and then transported to shops in Norway by ship. The waste management of nylon bags was mainly based on data from Watson et al. (2020) and Schmidt et al. (2016). In line with these studies, about 83.5% of discarded nylon bags are incinerated and the remainder recycled. The nylon bags collected for recycling are sorted and then sent to washing,

[12] Note: in Raadal et al. the distances are split into 3 transport stages: household to waste reception centre, waste reception centre to central sorting, sentral sorting to final treatment. The transport distance for recycling within Norway is assumed the same as for abroad. The relevant recycling facility in Norway was in Skogn (Mid-Norway) and 50% was transporten abroad. Towns like Tromsø in Northern Norway can be more than 1000 km away from Skogn and recycling plants in southern Sweden, e.g. Norrkjöping, can be about 400km or less from Oslo. Thus the same transport distance assumption has been deemed reasonable.

drying and further recycling abroad. Textiles lost in the recycling process become incinerated with energy recovery. Data on recycled content, weight and production site of nylon bags were provided by a Norwegian grocery store company. All foreground system data applied for the nylon bag is summarised in Table 12 and the type of background system data applied is presented in Appendix 4.

Table 12. Foreground system data applied for the nylon bag

Parameter [unit]	Value	Reference	Data quality assessment[13]
Parameters included in base case scenario (and in the net scrap approach scenario)			
Recycled content in nylon bag [%]	0	Assumption by Norwegian retailer	
Weight of nylon bag [g]	45	Norwegian retailer	Very good
Distance from production site (East Asia) to store (Oslo) [km]	20 400	Norwegian retailer; Sea-distancesorg. (2021)	Good (Oslo location, not the whole of Norway)
Share of discarded nylon bags that becomes incinerated (rest is recycled) [-]	0.835	Watson et al. (2020)	Very good
Distance from consumer to incineration [km]	85	Assumed by the authors based on Raadal et al. (2009)	Fair (geography and technical aspects are very good, but the time scale is "poor", as the source data is 12 years old).
Distance from consumer to sorting [km]	150	Schmidt et al. (2016)	Good (geography and technical aspects are very good, but the source data is 5-6 years old).
Electricity consumption in sorting process [MJ/kg]	0.25	Schmidt et al. (2016)	Good (see previous row)
Distance from sorting to recycling abroad (Europe) [km]	1 600	Schmidt et al. (2016)	Good (see above)
Parameters included in net scrap approach scenario only			
Electricity use in recycling of nylon (washing, drying, recycling) [kWh/kg textile output]	1.05	Schmidt et al. (2016)	Fair (laundry process data from another reference from 2012)
Thermal energy use in recycling of nylon [MJ/kg textile output]	6.84	Schmidt et al. (2016)	Good
Detergent consumption in recycling of nylon [kg/kg textile output]	0.009	Schmidt et al. (2016)	Fair (laundry process data from another reference from 2012)
Tap water consumption in	12	Schmidt et al. (2016)	

[13] See Table 8 for footnotes about the data quality assessment and definitions of the terms "very good", "good" etc.

recycling of nylon [kg/kg textile output]			
Loss of textiles in recycling of nylon [weight-%]	10	Schmidt et al. (2016)	Good
Share of lost textiles from recycling that become incinerated for electricity generation (remainder for heat generation) [-]	0.52	Eurostat (2021a, b)	Good
Lower heating value (LHV) for nylon (PA-6) [MJ/kg]	30.2	Ioelovich (2018)	LHV data is physical property data, so although the timescale is more than 3 years ago, this data is deemed very good.
Efficiency of incineration process in Norway	0.69	Calculated from 81% energy efficiency (energiutnyttelsesgrad) for Norwegian incineration plants in 2019 Avfall Norge (2020) and 85% efficiency for waste incineration, Raadal et al. (2009).	Good (geography and technical aspects are very good, the time scale of the energy efficiency is very good. The efficiency of waste incineration is by expert assessment good, but classified as "poor", as the source for the detailed data behind this is 12 years old).
Efficiency of incineration process in Europe	0.3	Swedenergy (2021)	Very good (the source is knowledgeable about the industry and European market and it is a new source. Although the source of the efficiency data is not referenced).
Substitution rate for material in nylon bag	0.9	Schmidt et al. (2016)	Good

5.6 Polyethylene terephthalate (PET) bag

The PET bag is made from 100% recycled materials and produced in East Asia. After production, the bag is transported to Norway by ship. The waste management of PET bags was modelled in a similar fashion as for the nylon bags, see Chapter 5.5. Data on the recycled content, weight and production site of nylon bags were provided by a Norwegian grocery store company. The foreground system data applied for the PET bag is summarised in Table 13 and the type of background system data applied is presented in Appendix 4.

Table 13. Foreground system data applied for the polyethylene terephthalate (PET) bag

Parameter [unit]	Value	Reference	Data quality assessment[14]
Parameters included in base case scenario (and in the net scrap approach scenario)			
Recycled content in PET bag [%]	100	Norwegian retailer	Very good
Weight of PET bag [g]	56	Norwegian retailer	Very good
Distance from production site (East Asia) to store (Oslo) [km]	20 400	Norwegian retailer; Sea-distancesorg. (2021)	Good (Oslo location, not the whole of Norway)

[14] See Table 8 for footnotes about the data quality assessment and definitions of the terms "very good", "good" etc.

Share of discarded PET bags that becomes incinerated (rest is recycled) [-]	0.835	Watson et al. (2020)	Very good
Distance from consumer to incineration [km]	85	Assumed by the authors based on Raadal et al. (2009)	Fair (geography and technical aspects are very good, but the time scale is "poor", as the source data is 12 years old).
Distance from consumer to sorting [km]	150	Schmidt et al. (2016)	Good (geography and technical aspects are very good, but the source data is 5-6 years old).
Electricity consumption in sorting process [MJ/kg]	0.25	Schmidt et al. (2016)	
Distance from sorting to recycling abroad (Polen) [km]	1 600	Schmidt et al. (2016)	
Parameters included in net scrap approach scenario only			
Electricity use in recycling of PET (washing, drying, recycling) [kWh/kg textile output]	1.05	Schmidt et al. (2016)	Fair (laundry process data from another reference from 2012)
Thermal energy use in recycling of PET [MJ/kg textile output]	6.84	Schmidt et al. (2016)	Good
Detergent consumption in recycling of PET [kg/kg textile output]	0.009	Schmidt et al. (2016)	Fair (laundry process data from another reference from 2012)
Tap water consumption in recycling of PET [kg/kg textile output]	12	Schmidt et al. (2016)	Fair (laundry process data from another reference from 2012)
Loss of textiles in recycling of PET [weight-%]	10	Schmidt et al. (2016)	Good
Share of lost textiles from recycling that become incinerated for electricity generation (remainder for heat generation) [-]	0.52	Eurostat (2021a, b)	Good
Lower heating value (LHV) for PET [MJ/kg]	21.3	Areepraserta et al. (2016)	LHV data is physical property data, so although the timescale is more than 3 years ago, this data is deemed very good.
Efficiency of incineration process in Norway	0.69	Calculated from 81% energy efficiency (energiutnyttelsesgrad) for Norwegian incineration plants in 2019 Avfall Norge (2020) and 85% efficiency for waste incineration, Raadal et al. (2009).	Good (geography and technical aspects are very good, the time scale of the energy efficiency is very good. The efficiency of waste incineration is by expert assessment good, but classified as "poor", as the source for the detailed data behind this is 12 years old).
Efficiency of incineration process in Europe	0.3	Swedenergy (2021)	Very good (the source is knowledgeable about the industry and European market and it is a new source. Although the source of the efficiency data is not referenced).

Substitution rate for material in PET bag	0.9	Schmidt et al. (2016)	Good

5.7 Cotton bag

The example bag chosen was one purchased in a Norwegian interior textiles shop chain some years ago by one of the authors. The pragmatic choice was made, as it was an organic cotton bag, purchased for shopping in a Norwegian store and available to weigh and measure during the corona restrictions. This bag is no longer sold by that retail chain. The system described here is based on specific information about that type of bag, provided by the specific shop chain (from earlier records). The example bag is still of good quality and being used for shopping. The cotton bag is made from organic cotton and was produced in South Asia. After production, the bag was transported to Norway by ship. The waste management of the cotton bag was modelled in a similar manner to the nylon bags, see Chapter 5.5. Data on the location for cotton bag production as well as the type of cotton used were provided by the Norwegian shop chain that used to sell these bags. The foreground system data applied for the cotton bag is summarised in Table 14 and the type of background system data applied is presented in Appendix 4.

Table 14. Foreground system data applied for the cotton bag

Parameter [unit]	Value	Reference	Data quality assessment[15]
Parameters included in base case scenario (and in the net scrap approach scenario)			
Recycled content in organic cotton bag [%]	0	Norwegian retailer	Fair (due to the age of the data, geographical and technical aspects are very good).
Weight of organic cotton bag [g]	125	Measurements by the authors	Fair – the bag was weighed, but the weight was for a specific, single bag.
Distance from production site (South Asia) to (Oslo) [km]	12 600	Norwegian retailer; Sea-distancesorg. (2021)	Fair (due to the age of the data, geographical and technical aspects are very good).
Share of discarded cotton bags that becomes incinerated (rest is recycled) [-]	0.835	Watson et al. (2020)	Very good
Distance from consumer to incineration [km]	85	Assumed by the authors based on Raadal et al. (2009)	Fair (geography and technical aspects are very good, but the time scale is "poor", as the source data is 12 years old).
Distance from consumer to sorting [km]	150	Schmidt et al. (2016)	Good (geography and technical aspects are very good, but the source data is 5-6 years old).
Electricity consumption in sorting process [MJ/kg]	0.25	Schmidt et al. (2016)	Good (see previous row)
Distance from sorting to recycling abroad (Polen) [km]	1 600	Schmidt et al. (2016)	Good (see previous row)
Parameters included in net scrap approach scenario only			

[15] See Table 8 for footnotes about the data quality assessment and definitions of the terms "very good", "good" etc.

Electricity use in recycling of cotton (washing, drying, shredding and carding) [kWh/kg textile output]	0.89	Schmidt et al. (2016)	Fair (laundry process data from another reference from 2012)
Thermal energy use in recycling of cotton [MJ/kg textile output]	6.84	Schmidt et al. (2016)	Good
Detergent consumption in recycling of cotton [kg/kg textile output]	0.009	Schmidt et al. (2016)	Fair (laundry process data from another reference from 2012)
Tap water consumption in recycling of cotton [kg/kg textile output]	12	Schmidt et al. (2016)	Fair (laundry process data from another reference from 2012)
Loss of textiles in shredding and carding of cotton [weight-%]	20	Schmidt et al. (2016)	Good
Share of lost textiles in recycling that become incinerated for electricity generation (remainder for heat generation) [-]	0.52	Eurostat (2021a, b)	Good
Lower heating value (LHV) for cotton [MJ/kg]	20.2	Schmidt et al. (2016)	LHV data is physical property data, so although the timescale is more than 3 years ago, this data is deemed very good.
Efficiency of incineration process in Norway	0.69	Calculated from 81% energy efficiency (energiutnyttelsesgrad) for Norwegian incineration plants in 2019 Avfall Norge (2020) and 85% efficiency for waste incineration, Raadal et al. (2009).	Good (geography and technical aspects are very good, the time scale of the energy efficiency is very good. The efficiency of waste incineration is by expert assessment good, but classified as “poor”, as the source for the detailed data behind this is 12 years old).
Efficiency of incineration process in Europe	0.3	Swedenergy (2021)	Very good (the source is knowledgeable about the industry and European market and it is a new source. Although the source of the efficiency data is not referenced).
Substitution rate for material in cotton bag	0.8	Schmidt et al. (2016)	Good

5.8 Cardboard box

A corrugated cardboard box currently used for home delivery shopping is included as a possible carrying solution in this study. Thus, the cardboard box is not currently used for shopping in grocery stores but was included as a potential future alternative. The box is a single use box, as this is the current design of the system. If these boxes were to be provided as an alternative to light weight plastic bags in shops, then more storage space would most probably be required in the shops and should then be considered in the analysis.

These logistics aspects are, however, not included within the scope of this study. The cardboard box is produced in Southeast Norway with a recycled content at 46%. The produced box is assumed transported from the production site to stores by truck. For waste management of the cardboard box, 64% was assumed to be sent to material recycling in Norway, 30% to material recycling abroad and 6% was assumed to be incinerated with energy recovery in Norway. This was based on data from Norsk Resy (2021 a, b), who stated a collection rate at 96% in 2020 for cardboard waste. Norsk Resy has the responsibility to ensure that cardboard waste in Norway is collected and recycled and they primarily gather solid and corrugated cardboard transport packaging. Uncollected cardboard waste was assumed incinerated together with residual waste. Note that the data from Norsk Resy is a combination of data for cardboard collection from industry, retail and households and are likely to represent a higher average collection rate than for households alone. Furthermore, SSB (2021), stated that in 2019 about 98% of the collected paper, cardboard and carton waste in Norway was sent to material recycling while about 2% was sent to incineration (about 0.3% was sent to other treatment but this was not considered). About 68% of the collected cardboard for recycling was assumed to be recycled in Norway while the remainder (32%) was assumed to be recycled abroad based on Raadal et al. (2009). A Norwegian home delivery shopping service provided data on recycled content, weight and production site for the cardboard box studied. The foreground system data used for the cardboard box is summarised in Table 15 and the type of background system data applied is presented in Appendix 4.

Table 15. Foreground system data applied for the cardboard box

Parameter [unit]	Value	Reference	Data quality assessment[16]
Parameters included in base case scenario (and in the net scrap approach scenario)			
Recycled content in cardboard box [%]	46	Norwegian retailer	Very good
Weight of cardboard box [g]	227	Norwegian retailer	Very good
Distance from production site (Southeast Norway) to store (Oslo) [km]	50	Norwegian retailer; Google maps	Good (Oslo location, not the whole of Norway)
Share of discarded cardboard boxes that becomes incinerated (rest is recycled) [-]	0.06	Norsk Resy (2021 a, b) and SSB (2021)	Very good
Distance from consumer to incineration [km]	85	Raadal et al. (2009)	Fair (geography and technical aspects are very good, but the time scale is "poor", as the source data is 12 years old).
Share of cardboard boxes that become recycled in Norway (remainder become recycled abroad) [-]	0.68	Raadal et al. (2009)	Fair (see previous row)
Distance from consumer to	498	Raadal et al. (2009)	Fair (see previous row)

[16] See Table 8 for footnotes about the data quality assessment and definitions of the terms "very good", "good" etc.

recycling in Norway [km]			
Distance from consumer to recycling abroad [km]	498	Raadal et al. (2009)	Fair (see previous row)
Parameters included in net scrap approach scenario only			
Lower heating value (LHV) for cardboard [MJ/kg]	15	Raadal et al. (2009)	LHV data is physical property data, so although the timescale is more than 3 years ago, this data is deemed very good.
Efficiency of incineration process in Norway	0.69	Calculated from 81% energy efficiency (energiutnyttelsesgrad) for Norwegian incineration plants in 2019 Avfall Norge (2020) and 85% efficiency for waste incineration, Raadal et al. (2009).	Good (geography and technical aspects are very good, the time scale of the energy efficiency is very good. The efficiency of waste incineration is by expert assessment good, but classified as "poor", as the source for the detailed data behind this is 12 years old).
Substitution rate for material in cardboard box	1	Zampori and Pant. (2019)	Very good

5.9 Rucksack

Some Norwegian consumers use a rucksack to carry their groceries home from the store. The rucksack consists of different materials, e.g., textiles and a steel frame, in turn produced in various countries. The rucksack itself is produced in Southeast Asia and then transported to Norway by ship. The waste management of the rucksack textiles were modelled in a similar manner to the nylon bags, see Chapter 5.5. The rucksack steel frame was assumed to become recycled in Europe. A rucksack supplier provided data for materials use, energy use for production of the rucksack and the location of the factory. Appendix 2 summarises data from a set of questions asked to a sample of environmentally aware consumers about the use of rucksacks for shopping. Their responses informed the choice of size of rucksack (30 litres) that the supplier was asked to provide data for. Appendix 2 also provides data used in the sensitivity analysis concerning rucksack allocation to shopping, lifetime and number of uses in that lifetime. The foreground system data used for the rucksack is summarised in Table 16 and the type of background system data applied is presented in Appendix 4.

Table 16. Foreground system data applied for the rucksack

Parameter [unit]	**Value**	**Reference**	**Data quality assessment[17]**
Parameters included in base case scenario (and in the net scrap approach scenario)			
Recycled content (all materials) in rucksack [%]	0	Norwegian retailer	Very good

[17] See Table 8 for footnotes about the data quality assessment and definitions of the terms "very good", "good" etc.

Weight of foam in rucksack [g]	88	Norwegian retailer	Very good
Weight of nylon in rucksack [g]	193	Norwegian retailer	Very good
Weight of POM in rucksack [g]	95	Norwegian retailer	Very good
Weight of steel parts and frame in rucksack [g]	115	Norwegian retailer	Very good
Weight of polyester in rucksack (produced in East Asia) [g]	208	Norwegian retailer	Very good
Weight of polyester in rucksack (produced in East Asia) [g]	54	Norwegian retailer	Very good
Weight of polyester in rucksack (produced in Southeast Asia) [g]	93	Norwegian retailer	Very good
Electricity use in weaving of polyester [kWh/kg textile]	0.775	Roos et al. (2015)	Fair (technology and processes very good, but the data is just over 6 years old, so falls into the fair category).
Use of heat in weaving of polyester [kWh/kg textile]	4.15	Roos et al. (2015)	Fair (see previous row)
Starch input in weaving of polyester [kg/kg textile]	0.05	Roos et al. (2015)	Fair (see above)
Electricity use in production of a rucksack [kWh/rucksack]	1.65	Norwegian retailer	Good. The data is from the specific factory producing the rucksack, but differentiation between products produced was not possible, so allocation of electricity production to all products was performed.
Distance from production site (Southeast Asia) to store (Oslo) [km]	17 480	Norwegian retailer; Sea-distancesorg. (2021)	Good (Oslo location, not the whole of Norway)
Share of discarded rucksack textiles that becomes incinerated (rest is recycled) [-]	0.835	Assumption by the authors based on Watson et al. (2020)	
Share of discarded steel parts and frame in rucksack that become incinerated (rest recycled) [-]	0	Assumption by the authors	
Distance from consumer to incineration, textiles [km]	85	Assumed by the authors based on Raadal et al. (2009)	Fair (geography and technical aspects are very good, but the time scale is “poor”, as the source data is 12 years old).
Distance from consumer to sorting, textiles [km]	150	Schmidt et al. (2016)	Good (geography and technical aspects are very good, but the source data is 5-6 years old).

Electricity consumption in sorting, textiles, process [MJ/kg]	0.25	Schmidt et al. (2016)	Good (see previous row)
Distance from sorting to recycling (textiles) abroad (Europe) [km]	1 600	Schmidt et al. (2016)	Good (see previous row)
Distance from consumer to recycling (steel) [km]	1 050	Assumption by the authors; Google maps	
Parameters included in net scrap approach scenario only			
Electricity use in recycling of rucksack textiles (washing, drying, recycling) [kWh/kg textile output]	1.05	Schmidt et al. (2016)	Fair (laundry process data from another reference from 2012)
Thermal energy use in recycling of rucksack textiles [MJ/kg textile output]	6.84	Schmidt et al. (2016)	Good
Detergent consumption in recycling of rucksack textiles [kg/kg textile output]	0.009	Schmidt et al. (2016)	Fair (laundry process data from another reference from 2012)
Tap water consumption in recycling of rucksack textiles [kg/kg textile output]	12	Schmidt et al. (2016)	Fair (laundry process data from another reference from 2012)
Loss of textiles in recycling of rucksack textiles [weight-%]	10	Schmidt et al. (2016)	Good
Share of lost textiles in recycling that become incinerated for electricity generation (remainder for heat generation) [-]	0.52	Eurostat (2021a, b)	Good
Lower heating value (LHV) for PET (used as proxy) [MJ/kg]	21.3	Areepraserta et al. (2016)	LHV data is physical property data, so although the timescale is more than 3 years ago, this data is deemed very good.
Efficiency of incineration process in Norway	0.69	Calculated from 81% energy efficiency (energiutnyttelsesgrad) for Norwegian incineration plants in 2019 Avfall Norge (2020) and 85% efficiency for waste incineration, Raadal et al. (2009).	Good (geography and technical aspects are very good, the time scale of the energy efficiency is very good. The efficiency of waste incineration is by expert assessment good, but classified as “poor”, as the source for the detailed data behind this is 12 years old).
Efficiency of incineration process in Norway	0.64	Raadal et al. (2009)	Good (geography and technical aspects are very good, the time scale of the energy efficiency is very good. The efficiency of waste incineration is by expert assessment good, but classified as “poor”, as the source for

			the detailed data behind this is 12 years old).
Efficiency of incineration process in Europe	0.3	Swedenergy (2021)	Very good (the source is knowledgeable about the industry and European market and it is a new source. Although the source of the efficiency data is not referenced).
Substitution rate for textile material in rucksack bag	0.9	Assumed by the authors based on values for PET and nylon from Schmidt et al. (2016)	Good
Substitution rate for steel material in rucksack bag	1	Zampori and Pant. (2019)	Very good

6 Results

The results for the different analyses are shown in tables and figures in this chapter. Endpoint results are shown first, then midpoint. This is done in order to give an overview of the endpoint results, with three damage assessment indicators, before the midpoint results (covering 19 midpoint impact categories). Endpoint and midpoint results are shown for the two modelling approaches (cut-off and net scrap approach) and sensitivity analyses.

The midpoint results would be a very extensive if figures were shown for every impact category. In order to choose which figures are presented, the authors examined potential correlations between the midpoint environmental impacts analysed. Where there was a correlation (linear regression analysis gives R^2 results[18] close to 1), the figure for only one of the correlated impact categories is shown, this approach is similar to that used in Askham et al. 2012). The results for all impact categories are included in tables.

It is important for the reader to note that in accordance with ISO 14044 (2006) the LCIA results are relative expressions and do not predict actual impacts on category endpoints, the exceeding of thresholds, safety margins or risks.

6.1 Endpoint results - Cut-off

6.1.1 Human health

The figure below shows the human health damage assessment results for the different carrying solutions.

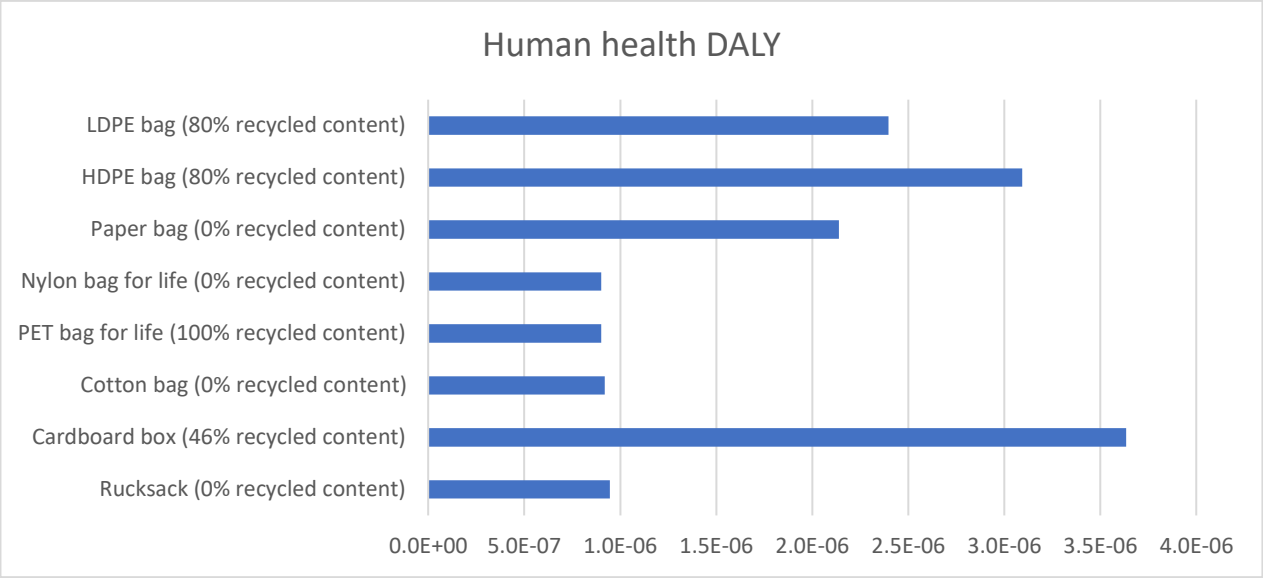


Figure 7: Potential human health damage (DALY) results per functional unit for the different carrying solutions analysed

Global warming impacts on human health dominate the results for this damage assessment category, fine particulate matter formation is also important.

[18] In this approach, R^2 is a statistical measure of how well two variables correlate, if R^2 is 1 the data for the two variables correlate exactly.

The single use carriers (LDPE, HDPE, paper and cardboard) show contributions to potential human health damage that are greater than the multiuse carriers. A large contributing factor to the difference between the paper and cardboard results is that the paper bag has a lower mass than the cardboard box. Of the single use carriers assessed the paper bag shows the lowest human health damage potential. Although the results for the paper bag are unlikely to be significantly different to those of the LDPE bag.

6.1.2 Ecosystems

The figure below shows the ecosystems damage assessment results for the different carrying solutions.

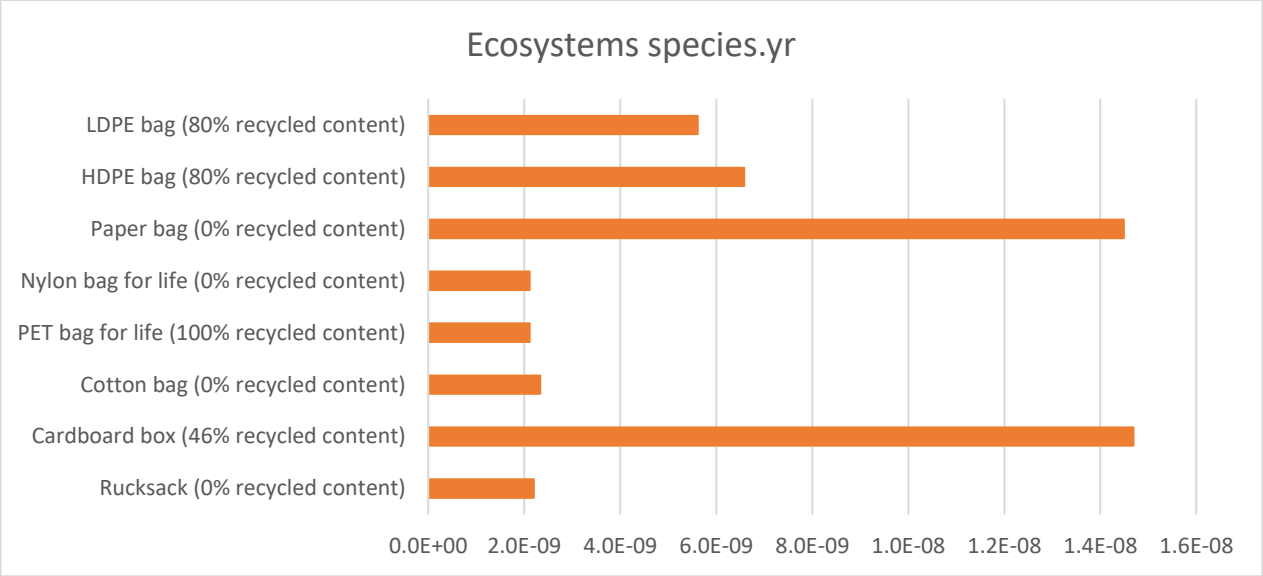


Figure 8: Potential ecosystems damage (species.yr) results per functional unit for the different carrying solutions analysed

Global warming impacts on ecosystems (particularly terrestrial) dominate the results for this damage assessment category. Land use is important for carrier systems using bio-based materials (i.e, paper, cotton and cardboard).

The single use carriers (LDPE, HDPE, paper and cardboard) show contributions to potential ecosystems damage that are greater than the multiuse carriers. Of the single use carriers assessed the HDPE and LDPE bags show the lowest ecosystems damage potential.

6.1.3 Resources

The figure below shows the resources damage assessment results for the different carrying solutions.

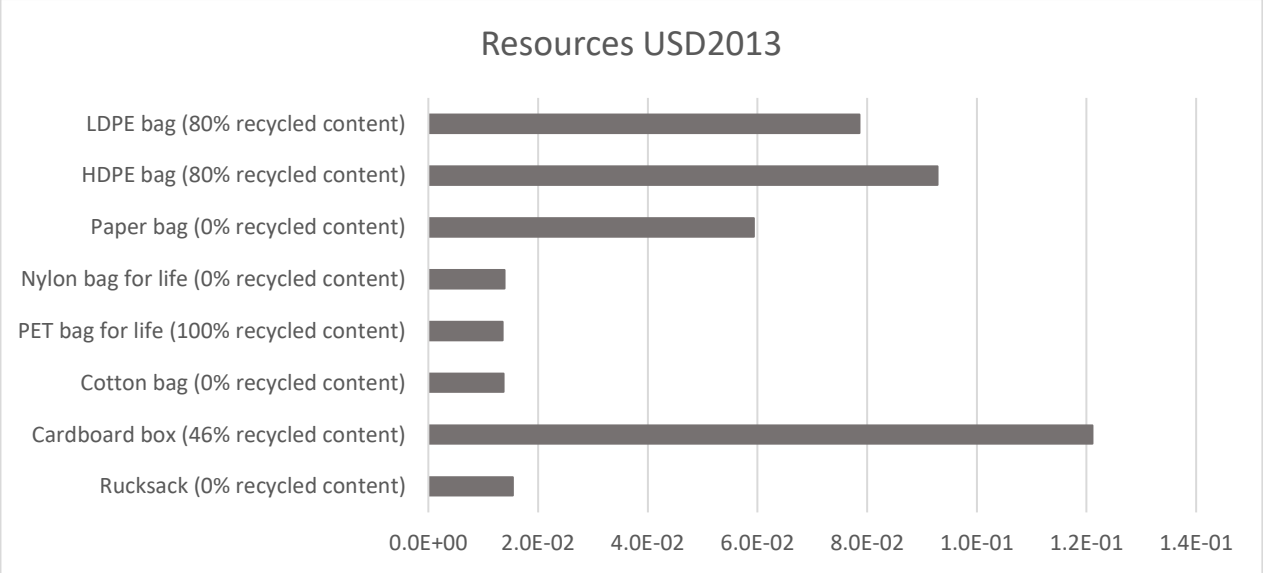


Figure 9: Potential resources damage (USD 2013) results per functional unit for the different carrying solutions analysed

Fossil resource scarcity is the most important contributor to the results for this damage assessment category.

The resources analysis results show that multiuse carriers are less resource intensive than single use carriers. The use of non-renewable fossil resources dominates these results, for both single use and multiuse carriers. The paper bag has a lower resource damage cost than the single use cardboard carrier. This is largely related to the different masses of these two carriers. For the paper bag system, the bin liner accounts for just over 20% of the contributions to resources damage. The other contributions are linked to the energy carriers used in the paper bag life cycle (i.e., materials production and transport). The picture is similar for the cardboard box, but the bin liner contribution is the same in number (thus amounting to approximately 10% for the cardboard box, as the total USD 2013 result is higher) and the main contributions are from fossil fuel energy carriers used during raw material and box production and transport.

6.1.4 Number of times a given carrier must be used – Endpoint cut-off

If the LDPE single use carrier is used as the reference system, the results for the analysis of the cut-off modelling approach for all of the systems can be used to calculate the number of times a given carrier must be used to be equivalent to the LDPE reference carrier. The impact of a carrier, i.e. an alternative carrier to the LDPE bag, (with the exception of the HDPE bag) consists of two parts, since the functional unit covers two functions. One part is the impact of the number of alternatives needed to fulfil the functional unit of a certain amount of groceries to be carried, in turn dependent on the number of times that the alternative is used during its lifetime. The other part is the impact of the bin liners needed to provide as many bin liners as the LDPE bag does per functional unit. Importantly, the number of times that the carrier is used will not affect the impact associated with the bin liners. This was considered in the calculation of the break-even point, i.e. the number of times a given carrier must be used to be equivalent to the LDPE reference carrier, and the calculation was conducted applying the following equation:

$$\frac{\frac{\text{impact carrier (used once) per f.u}}{x} + \text{impact bin liners per f.u.}}{\text{impact LDPE bag (used once) per f.u.}} = 1 \qquad \text{Eq. 2}$$

Where f.u = functional unit and *x* is the number of uses required for a carrier to be equivalent to the LDPE bag in terms of environmental impact. Note that the "impact bin liners per f.u" equals zero for the HDPE bag.

Table 17. The number of times a given carrier must be used to be equivalent to the reference carrier (cut-off) Endpoint damage assessment

Environmental impact	HDPE	Paper Bag (virgin)	Bag for life Nylon	Bag for life recycled PET	Bag for life Cotton	Cardboard box	Rucksack
Potential human health damage	1,3	0,8	7,3	7,5	19,9	1,8	95,7
Potential ecosystems damage	1,17	3,52	7,18	6,10	68,01	3,58	81,17
Potential resources damage	1,2	0,7	8	4	6	1,7	84

If the value in Table 17 is greater than 1, then the carrier needs to be used more than the reference carrier in order to have equivalent impacts. In order to be better than the LDPE single use carrier for the three damage assessment endpoints, the nylon and PET multiuse bags for life will need to be used 8 times. Similarly, the organic cotton bag 68 times and the rucksack 96 times.

The number of uses shown in the table above are much lower than the assumed base case scenario of 1000 uses for the multiuse bags (2600 for the rucksack, see Table 3). This assumption has been found to be reasonable, both though the survey conducted in this study (see Appendix 2 and 3) and other literature sources (e.g., Briedis et al. 2019's number of uses was 1414 for a multiuse plastic bag with a lifetime of 10 years).

6.2 Midpoint results - Cut-off

The following correlations were found for the cut-off modelling approach:

Table 18. Correlation between impacts for the cut-off modelling approach

Impact category correlation tested for	Impacts that correlate (R^2 value >0.8)
Global warming potential	Freshwater ecotoxicity (R^2 = 0.97) Marine ecotoxicity (R^2 = 0.97) Human carcinogenic toxicity (R^2 = 0.97) Fossil resource scarcity (R^2 = 0.88)
Marine Eutrophication	Stratospheric ozone depletion (R^2 = 0.91) Land use (R^2 = 0.92)
Water deprivation	Ozone formation, Human health (R^2=0.9) Ozone formation, Terrestrial ecosystems (R^2=0.88) Fine particulate matter formation (R^2=0.81) Terrestrial ecotoxicity (R^2=0.81) Human non-carcinogenic toxicity (R^2=0.86)

Fine particulate matter formation	Ozone formation, Human health (R^2=0.84) Terrestrial acidification (R^2=0.99) Terrestrial ecotoxicity (R^2=0.87) Human non-carcinogenic toxicity (R^2=0.95) Mineral resource scarcity (R^2=0.85) Fossil resource scarcity (R^2=0.88) Water deprivation (R^2=0.81)
Ionizing radiation	No correlation found
Plastic littering	No correlation found

In order to facilitate interpretation the critical review panel suggested colour coding the results table below. Table 19 provides an overview of the results based on the impact assessment methods described in Chapter 3.7. The colour coding in the table represents the smallest to the largest results (lightest shading is smallest and darkest is largest) for each impact category.

As stated above, LCIA results are relative expressions and do not predict actual impacts on category endpoints, the exceeding of thresholds, safety margins or risks. This table does not show relative importance of categories, or whether the size of an impact is large, or small for society as a whole. The difference from light to dark only colour codes the smallest to largest in the range of results, so there could be only small (and insignificant) differences from light to dark shading. Colour coding can subconsciously impart a lot of these aspects and the reader is advised that the categories shown in Table 19 should be interpreted independently from each other and with care. Endpoint results shown above (Chapter 6.1), are a way of combining the midpoint effects into a damage assessment that facilitates a more aggregated interpretation of these midpoint results.

6.2.1 Results for all impact categories

Table 19. Environmental impacts calculated per functional unit for the cut-off modelling approach (darkest shading is the largest result and lightest smallest for each impact category)[19]

Impact category	Units	LDPE bag	HDPE bag	Paper bag	Nylon bag	PET bag	Cotton bag	Cardboard box	Rucksack
Global warming	kg CO_2 eq	1.56	1.74	1.08	0.60	0.60	0.61	1.85	0.62
Stratospheric ozone depletion	kg CFC-11 eq	2.5E-07	2.7E-07	6.6E-07	1.0E-07	9.7E-08	1.4E-07	1.2E-06	1.7E-07
Ionizing radiation	kBq Co-60 eq	0.16	1.2E-02	0.14	6.5E-02	6.6E-02	6.6E-02	0.17	6.6E-02
Ozone formation, Human health	kg NOx eq	1.5E-03	2.7E-03	2.8E-03	4.7E-04	4.7E-04	4.9E-04	4.3E-03	5.1E-04
Fine particulate matter formation	kg PM2.5 eq	7.1E-04	1.6E-03	1.0E-03	2.2E-04	2.2E-04	2.4E-04	1.9E-03	2.5E-04
Ozone formation, Terrestrial ecosystems	kg NOx eq	1.5E-03	2.7E-03	2.9E-03	4.8E-04	4.8E-04	5.0E-04	4.4E-03	5.3E-04
Terrestrial acidification	kg SO2 eq	1.6E-03	3.2E-03	2.8E-03	5.2E-04	5.2E-04	5.7E-04	4.5E-03	5.9E-04
Freshwater eutrophication	kg P eq	2.2E-04	2.0E-04	8.5E-04	8.5E-05	8.6E-05	1.1E-04	6.7E-04	9.0E-05
Marine eutrophication	kg N eq	7.4E-05	6.3E-05	1.7E-04	3.4E-05	3.3E-05	6.6E-05	5.7E-04	3.3E-05
Terrestrial ecotoxicity	kg 1,4-DCB	2.16	1.95	8.57	0.87	0.88	0.88	5.47	0.89
Freshwater ecotoxicity	kg 1,4-DCB	6.9E-02	5.9E-02	4.8E-02	2.9E-02	2.9E-02	2.9E-02	7.5E-02	2.9E-02

[19] Please note littering data included here is for the consumer use phase only. No other life cycle stages are included in this calculation

Marine ecotoxicity	kg 1,4-DCB	9,3E-02	8,1E-02	6,8E-02	3,9E-02	3,9E-02	3,9E-02	1,0E-01	4,0E-02
Human carcinogenic toxicity	kg 1,4-DCB	7,6E-02	7,4E-02	7,9E-02	3,1E-02	3,2E-02	3,2E-02	1,0E-01	3,3E-02
Human non-carcinogenic toxicity	kg 1,4-DCB	0,96	0,87	0,91	0,39	0,40	0,40	1,46	0,40
Land use	m2a crop eq	3,5E-02	3,7E-02	1,09	1,4E-02	1,4E-02	3,3E-02	0,82	1,4E-02
Mineral resource scarcity	kg Cu eq	1,7E-03	1,4E-03	1,8E-03	6,1E-04	6,2E-04	6,1E-04	3,2E-03	6,7E-04
Fossil resource scarcity	kg oil eq	0,23	0,28	0,19	4,6E-02	4,6E-02	4,7E-02	0,38	5,2E-02
Water Use	m3	0,46	0,42	0,41	0,14	0,14	0,14	0,57	0,14
Marine Littering[19]	g	0,9	0,9	0	2,6E-06	4,9E-06	0	0	0
Marine litter impact	PAF-m^3-day	6,6E-02	6,6E-02	0	1,9E-07	3,6E-07	0	0	0

It should be noted that the bin liner contribution for the relevant carriers (paper bag, nylon bag, PET bag, organic cotton bag, cardboard box and rucksack) contributes to the results equally for these systems. In order to aid transparency, the impacts for the bin liners for the cut-off model are given in the table below.

Table 20. Environmental impacts calculated per functional unit for the bin liner contribution to the results presented in Table 19 for the cut-off modelling approach (relevant for the paper bag, nylon bag, PET bag, organic cotton bag, cardboard box and rucksack carrier systems)

Impact category	Unit	Total
Global warming	kg CO2 eq	0.59
Stratospheric ozone depletion	kg CFC11 eq	9.5E-08
Ionizing radiation	kBq Co-60 eq	6.5E-02
Ozone formation, Human health	kg NOx eq	4.5E-04
Fine particulate matter formation	kg PM2.5 eq	2.2E-04
Ozone formation, Terrestrial ecosystems	kg NOx eq	4.6E-04
Terrestrial acidification	kg SO2 eq	5.1E-04
Freshwater eutrophication	kg P eq	8.4E-05
Marine eutrophication	kg N eq	3.2E-05
Terrestrial ecotoxicity	kg 1,4-DCB	8.6E-01
Freshwater ecotoxicity	kg 1,4-DCB	2.9E-02
Marine ecotoxicity	kg 1,4-DCB	3.9E-02
Human carcinogenic toxicity	kg 1,4-DCB	3.1E-02
Human non-carcinogenic toxicity	kg 1,4-DCB	3.9E-01
Land use	m2a crop eq	1.4E-02
Mineral resource scarcity	kg Cu eq	6.0E-04
Fossil resource scarcity	kg oil eq	4.5E-02
AWARE Water deprivation	m3 world eq	0.14

6.2.2 Global warming potential

The figure below shows the global warming potential (GWP) results for the different carrying solutions

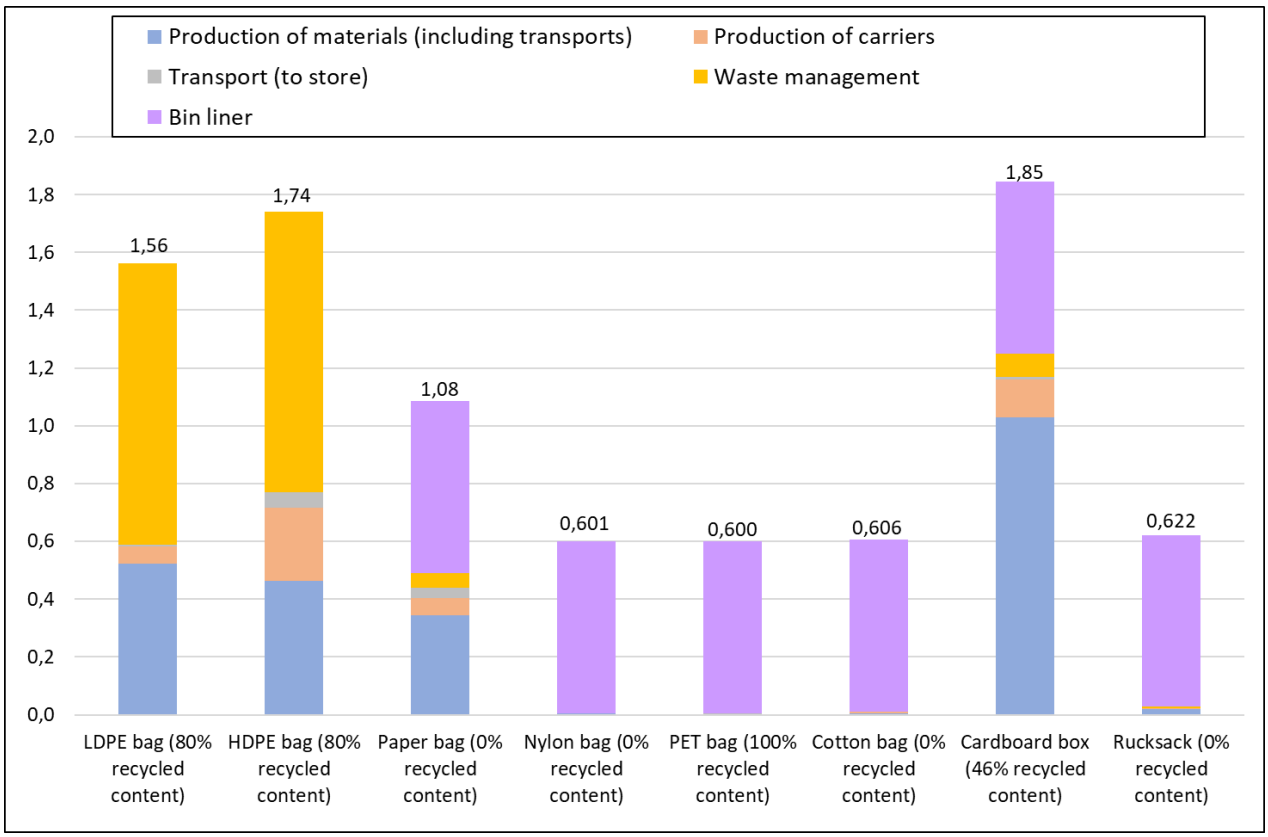


Figure 10: Global warming potential (kg CO_2-equivalents) results per functional unit for the different carrying solutions analysed

The single use carriers are LDPE, HDPE, Paper bag and cardboard box. These have a greater GWP impact than the multiuse carriers. The impacts of the materials used to make these carriers and their production processes are shared for the number of uses of the given carrier. This is why the impacts shown for production of materials, production of carriers, transport to store and waste management are small for multiuse carriers. The dominating GWP result for the multiuse carriers is the bin liner that is required to fulfil the same function as single use carriers. The bin liner is also significant for the paper bag and cardboard carriers, as they are assumed to be unsuitable for use as bin liners.

It is notable that for carriers made of plastic, waste incineration of the plastic is the most important contributing process. This is also the greatest contribution for the bin liners (shown as purple in Figure 10). The difference between the two single use plastic bags (LDPE and HDPE) in production of the carrier is because of the different electricity mixes. The HDPE bag analysed in this study is produced in Southeast Asia, while the LDPE bag is produced in Sweden.

The most important emissions contributing to the results shown for the LDPE bag system are carbon dioxide and methane from fossil sources (99% of the total GWP). This is also the case for the analysed systems for HDPE, nylon, PET and cotton bags, and the rucksack system. 97% of the GWP for the paper bag system arises from carbon dioxide and methane from fossil sources. The main process contribution here is also incineration of plastic (waste management of the bin liner), but the energy mix used for kraft paper production is important for production of materials (shown in blue). 93% of the GWP for the cardboard box

system arises from carbon dioxide and methane from fossil sources. The main process contribution here is also incineration of plastic (waste management of the bin liner), but the energy mix used for production of materials (e.g., linerboard, fluting and corrugated board) is a larger contributor for the cardboard box system than the paper bag system.

6.2.3 Marine eutrophication potential

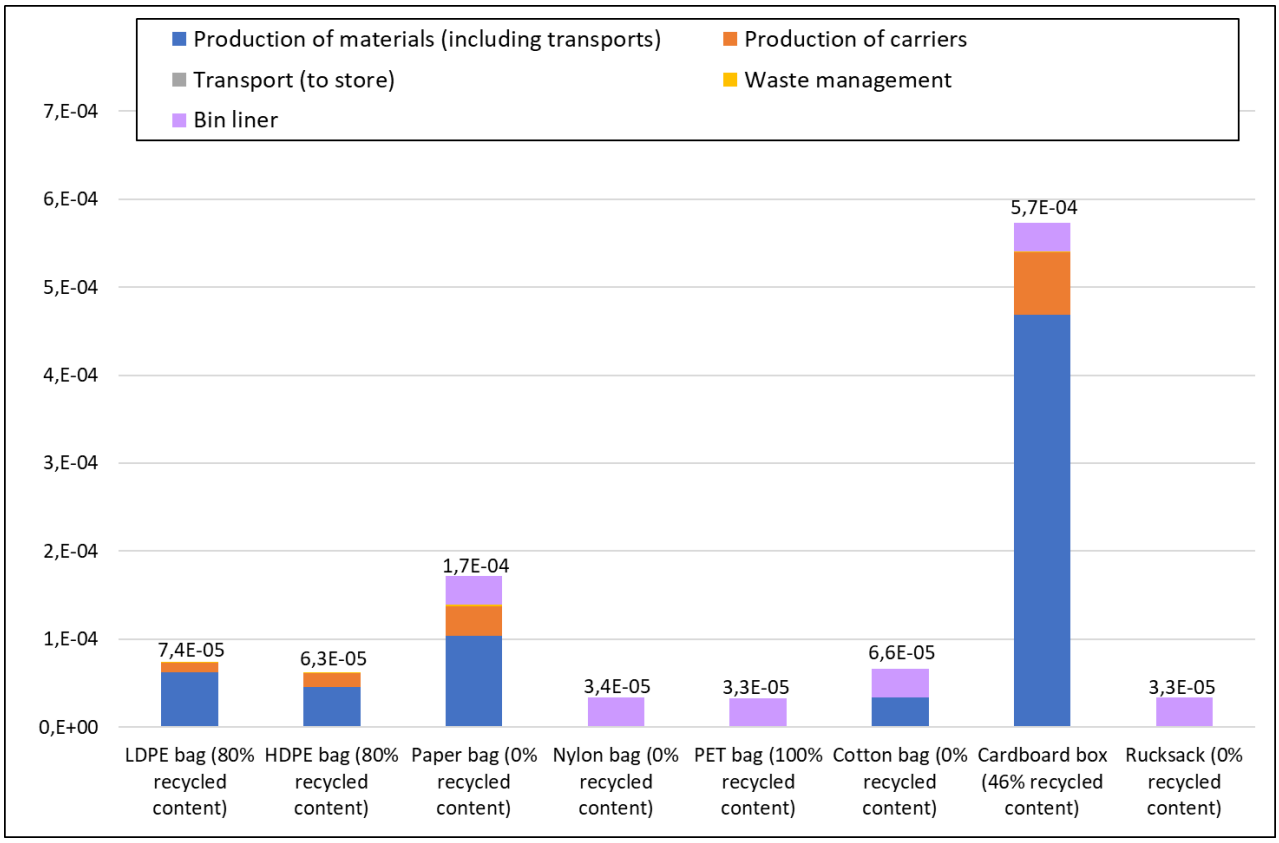


Figure 11: Marine eutrophication potential (kg N-equivalents) results per functional unit for the different carrying solutions analysed

The most important emissions contributing to marine eutrophication contain nitrogen. Nitrate, ammonium (ion), organically bound nitrogen, nitrite and nitrogen dioxide to water are important emissions for this impact indicator. The top three emissions for the LDPE, HDPE, nylon, PET and rucksack systems are nitrogen organically bound, nitrate and ammonium, accounting for approximately 99% of these systems' marine eutrophication potential. The processes that contribute the most for these five carrier systems are associated with granulate production. A certain amount of waste from granulate production internationally ends its life on landfill, which accounts for much (60%) of the marine eutrophication potential associated with these plastic-based systems. For the paper and cardboard system sludge from pulp and paper production accounts for approximately 40 and 60% of the marine eutrophication potential for these systems respectively.

50% of the marine eutrophication potential impact for the cotton bag systems arises from organic seed cotton production, the remaining 50% is as for the plastic-based systems, due to the bin-liner.

It should be noted that emissions contributing to marine eutrophication potential (i.e. nitrogen organically bound, nitrate and ammonium) need to reach the sea in order to contribute to marine eutrophication. Thus, the locations of the relevant facilities are important for whether the potential marine eutrophication

impacts are realised. Due to confidentiality aspects, the locations of suppliers of carriers are not provided in this report (country is provided, see Appendix 4).

6.2.4 Water deprivation potential

For the carrier solutions that are plastic, one of the main contributors to water use is cooling of equipment, such as extruders, needed to process the materials from granulate into the finished carrier.

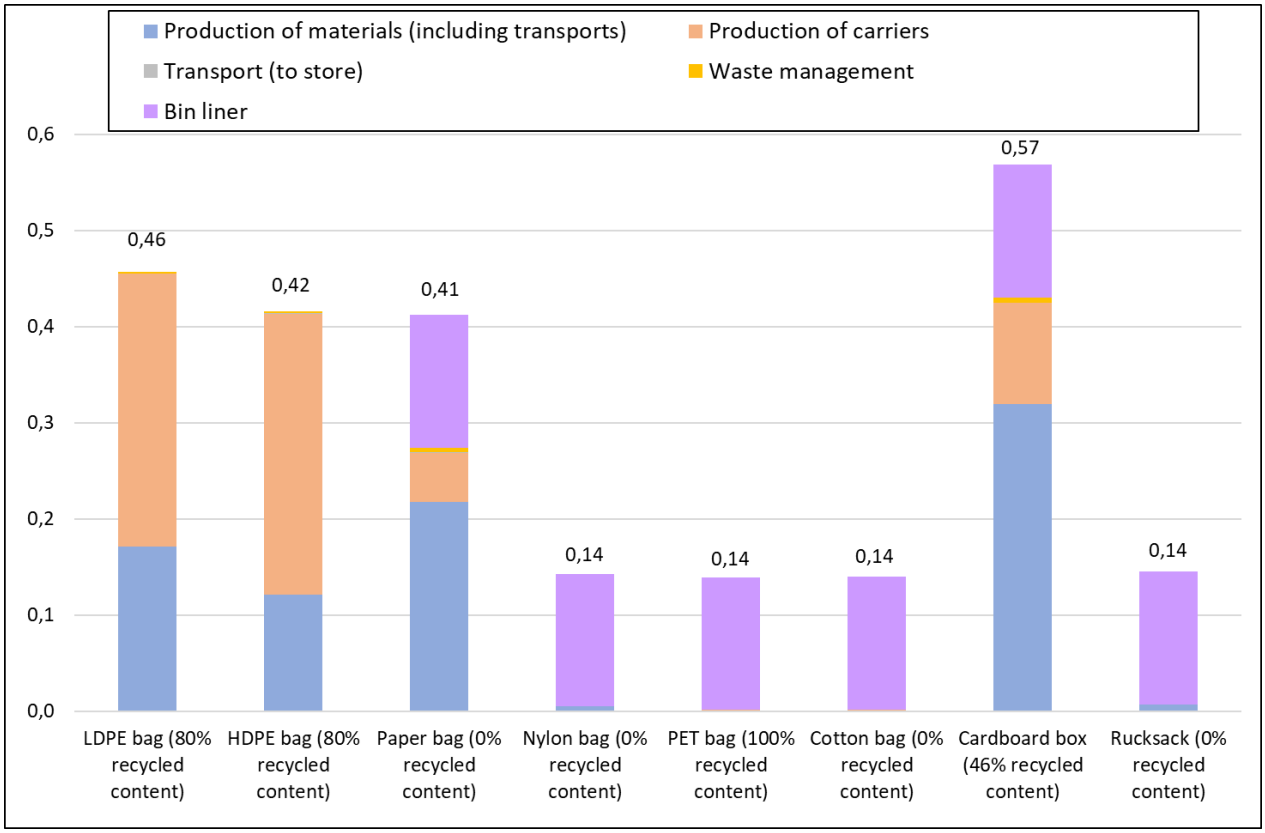


Figure 12: Water deprivation potential (m^3 world-eq.) results per functional unit for the different carrying solutions analysed

The single use carrier systems are LDPE, HDPE, Paper bag and cardboard box. These have a greater water deprivation impact than the multiuse carriers. The impacts of the materials used to make carriers and their production processes are shared for the number of uses of the given carrier. This is why the impacts shown for production of materials, production of carriers, transport to store and waste management are small for multiuse carriers. The dominating contributor for the water deprivation result for the multiuse carriers is the bin liner, that is required for multiuse carriers. The bin liner is also significant for the paper bag and cardboard carriers, as they are assumed unsuitable for use as bin liners.

The plastic-based systems and bin liner use water for cooling in extrusion and thus, the production of carrier (and production of the bin liner) is important for this impact. Pulp and paper production is also water intensive, explaining the large water use impact associated with material production for the paper and cardboard systems.

The critical review panel has commented that water use for different types of polymers can vary quite widely and this can lead to a high variability and uncertainty in water use results for plastic carriers. In order to check how this is reflected in the Ecoinvent database a comparison calculation was run for water

deprivation potential for several different plastics and processing techniques. The results for plastic granulate of different types could vary as much as up to nine times the lowest value. This was due to a combination of amounts of water for production and where the water is assumed to be drawn from. Processing of plastic varies also. The thermoforming of plastic sheets has much lower water deprivation potential than stretch blow moulding for example (a factor of 26). Neither of these processes are chosen for plastic bag production in this study; extrusion of plastic film is chosen, which has a water deprivation potential that is in the mid-range for the plastic processing technologies provided in the Ecoinvent database. This check was performed in the Ecoinvent database (version 3.7.1), as implemented in SimaPro (version 9.2) and not across any other sources of data.

Water use for the organic cotton data set chosen is low. If a global cotton data set was used for cotton, the water consumption would increase considerable per kilogram cotton used. If the cotton material in "production of materials" and "production of carriers" were to be replaced with the global cotton data set, then the result bar for the cotton bag system in Figure 8 would increase to approximately 0.31 m^3 world-eq. This would bring it closer to the water use associated with the single use carrier alternatives, and higher than the other multiuse systems. This shows that the choice of «organic cotton bag» as an option has important effect in the results and if a non-organic cotton bag had been chosen it would affect the results. Other impacts besides water deprivation would also be affected if the bag is not made from organic cotton.

6.2.5 Fine particulate matter formation potential

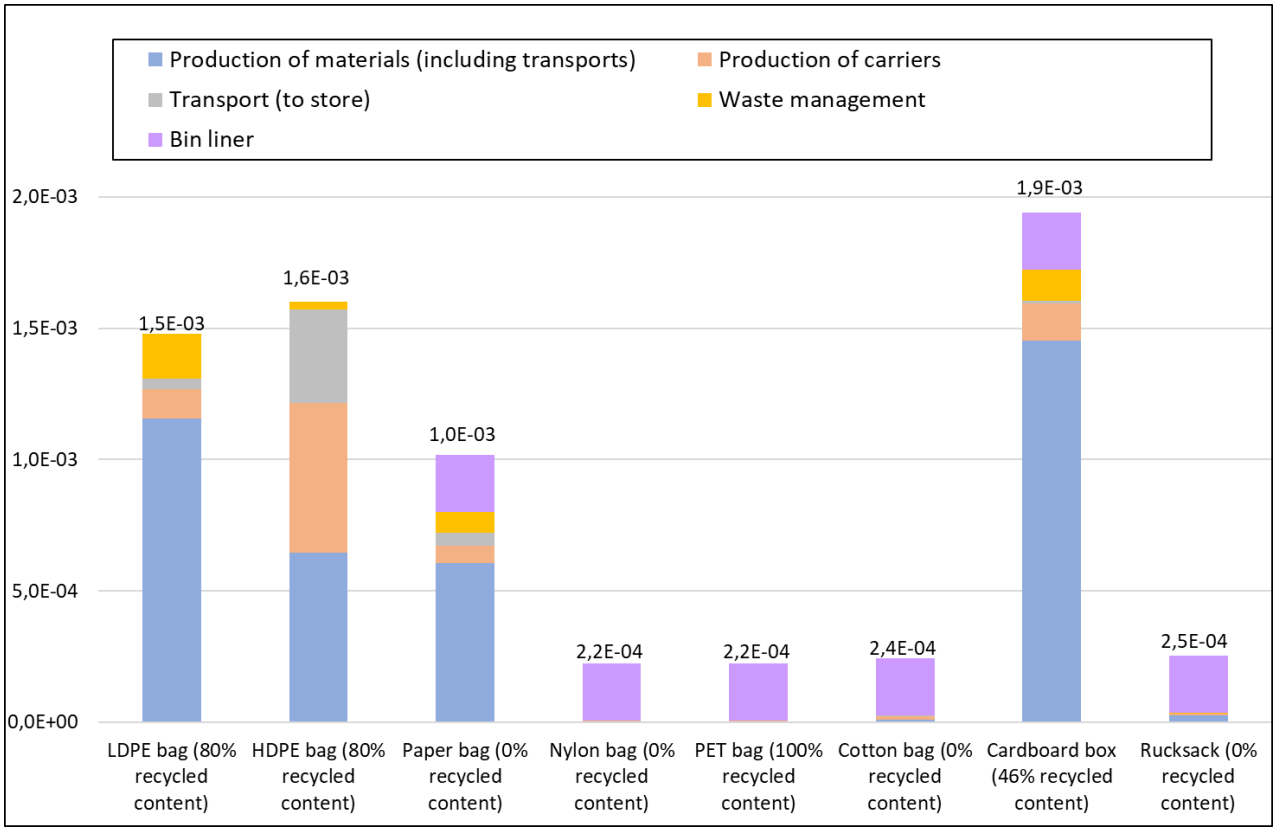


Figure 13: Fine particulate matter formation potential (kg PM2.5 equivalents) results per functional unit for the different carrying solutions analysed

Fine particulate matter can be generated when burning fuel (both bio-based and fossil-based). The difference between the two single use plastic bags (LDPE and HDPE) in production of the carrier is because

of the different electricity mixes. Transport over long distances by ship (using heavy fuel oil for fuel) is the reason for the relatively large transport impact for the HDPE bag. The HDPE bag analysed in this study is produced in Southeast Asia, while the LDPE bag is produced in Sweden.

The most important contributing emissions to this potential impact are sulphur oxides, direct emission of fine particulates (<2.5 mm) and nitrogen oxides. These are closely linked to combustion of fossil fuels, even for the cardboard box system.

6.2.6 Ionizing radiation

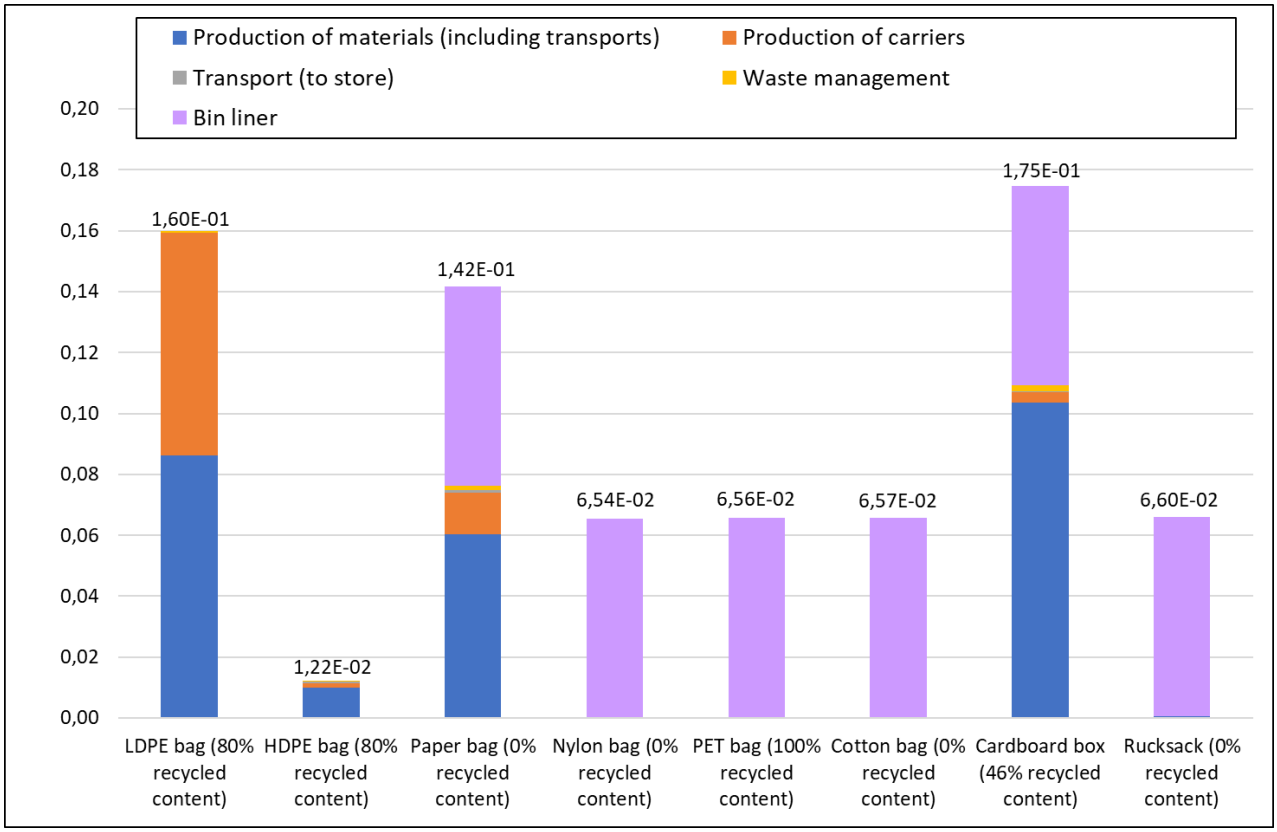


Figure 14: Ionizing radiation potential (kBq Co-60 equivalents) results per functional unit for the different carrying solutions analysed

As previously, the single use carriers (with the exception of HDPE) have greater ionizing radiation impacts than the multiuse carriers, due to the multiuse carrier impacts for production processes being shared across the number of uses for the given carrier. The bin liner impacts are significant for the systems requiring a bin liner.

Examining the details of the results, and the processes and life cycle stages that contribute to this impact, shows that the majority of the contributions to the ionizing radiation impact potential are from nuclear power processes within the electricity mixes used. More specifically Radon-222 emissions to air from treatment of tailings from uranium milling. The reason why the HDPE bag has results much lower than the LDPE bag is because the electricity mix included in the model for the production locations is different; LDPE bags are assumed produced in Sweden and HDPE in Southeast Asia.

6.2.7 Littering

The calculation of littering data has been performed, as described in Chapter 3.7.1 . This calculation has been performed for littering from the consumer use phase only. The stages in the calculation and calculation results are presented in the table below. It is assumed that a ruck sack will not be littered at the end of its life.

Table 21. Littering per functional unit

	LDPE SU[20] bag	HDPE SU bag	Paper Bag	Nylon MU[21] bag	PET MU bag	Cotton MU bag	Cardboard box SU	Rucksack MU
Reference flow [# carriers per functional unit]	19	19	9.5	0.01	0.015	0.009	5.7	0.003
Mass pr carrier (g)	19	19	63	45	56	125	227	846
Total mass of material	361	361	599	0.45	0.84	1.1	1294	2.5
Plastic discarded [g/FU]	18	18	0	5.2E-05	9.8E-05	0	0	0
Plastic litter not picked up [g/FU]	1.8	1.8	0	5.2E-06	9.8E-06	0	0	0
Marine litter [g/FU]	0.9	0.9	0	2.6E-06	4.9E-06	0	0	0
Marine litter effect [PAF-m^3-day]	0.07	0.07	0	1.9E-07	3.6E-07	0	0	0

LCIA impact assessment results for other impact categories are calculated using characterisation factors, and can be described using the following equation:

$$LCIA\ result = emission \times characterisation\ factor\ (CF)$$

Characterisation factors (CF) can be described by (Rosenbaum et al. 2008):

$$CF = effect\ factor\ (EF) \times exposure\ factor\ (XF) \times fate\ factor\ (FF)$$

Please see Chapter 3.7.1 for more information about how these calculations are performed, and references for the factors used. The emission is the mass entering the marine environment, 100% of the bag is assumed to be fragmented in micro and nano particles (MNPs). This implies no biodegradation, and no macro effect (e.g., entanglement). It also means that 100% of the MNPs are assumed to have an effect (i.e. no settling into the sediment, no degradation). This is «conservative», as marine litter impacts resulting from MNPs are overestimated, while entanglement effects are not included (which is an underestimation of this effect). The effect factor used does not discriminate between different plastic types, as there was not adequate data to enable this (see Lavoie et al. 2021).

[20] Single use
[21] Multiuse

6.2.8 Number of times a given carrier must be used - Midpoint cut-off

If the LDPE single use carrier is used as the reference system, the results for the analysis of the cut-off modelling approach for all of the systems can be used to calculate the number of times a given carrier must be used to be equivalent to the LDPE reference carrier. The calculation procedure for these results is given in Chapter 6.1.4 . The number of times a given carrier must be used to be equivalent to the LDPE reference carrier is given in the table below.

Table 22. The number of times a given carrier must be used to be equivalent to the reference carrier (cut-off)

Environmental impact	HDPE	Paper Bag (virgin)	Bag for life Nylon	Bag for life recycled PET	Bag for life Cotton	Cardboard box	Rucksack
Global warming	1.1	0.5	6	5	12	1.3	73
Stratospheric ozone depletion	1.1	4	29	14	301	7	1376
Ionizing radiation	0.1	0.8	0.7	3	4	1.2	20
Ozone formation, Human health	1.8	2.3	12	13	31	3.8	148
Fine particulate matter formation	2.3	1.6	12	12	52	3.5	183
Ozone formation, Terrestrial ecosystems	1.8	2.3	12	13	30	3.7	145
Terrestrial acidification	1.9	2	16	13	55	3.5	183
Freshwater eutrophication	0.9	6	4	8	155	4	97
Marine eutrophication	0.8	3	30	10	799	13	71
Terrestrial ecotoxicity	0.9	5.9	2	10	8	3.5	53
Freshwater ecotoxicity	0.9	0.5	2	9	10	1.2	38
Marine ecotoxicity	0.9	0.5	2	9	10	1.1	38
Human carcinogenic toxicity	1.0	1.1	5	10	11	1.5	85
Human non-carcinogenic toxicity	0.9	0.9	3	10	13	1.9	60
Land use	1.1	50	4	8	922	38	36
Mineral resource scarcity	0.8	1.1	6	13	6	2.4	170
Fossil resource scarcity	1.2	0.8	9	5	14	1.9	96
Water deprivation	0.9	0.9	15	3	5	1.3	55
Marine littering	1.0	N/A[22]	2.9E-03	5.4E-03	N/A	N/A	N/A

[22] Not applicable for this carrier (see Chapter 6.2.7

Marine litter impact	1.0	N/A	2.9E-06	5.4E-06	N/A	N/A	N/A

If the value in Table 22 is greater than 1, then the carrier needs to be used more than the reference carrier in order to have equivalent impacts. In order to be better than LDPE for all of the impacts considered, with the exception of land use, the paper bag will need to be used 6 times. If the land use category is included, it would need to be used more than 50 times to be better than the LDPE reference bag. Similarly, the nylon bag would need to be used 30 times, the PET bag 14, the cotton bag 922 times, the cardboard box 38 and the rucksack 1376 times.

For the multiuse bags the base case scenario assumptions of 1000 uses have been found to be reasonable, both though the survey conducted in this study (see Appendix 2 and 3) and other literature sources (e.g., Briedis et al. (2019)'s number of uses was 1414 for a multiuse plastic bag with a lifetime of 10 years). For the rucksack the number of uses shown in Table 23 is just over half the assumed number of uses for a rucksack in this study (see Table 3).

6.3 Endpoint results – Net scrap approach

The results in this chapter are from the system expansion modelling using the net scrap approach described in Chapter 3.5.3 .

6.3.1 Human health

The figure below shows the human health damage assessment results for the different carrying solutions.

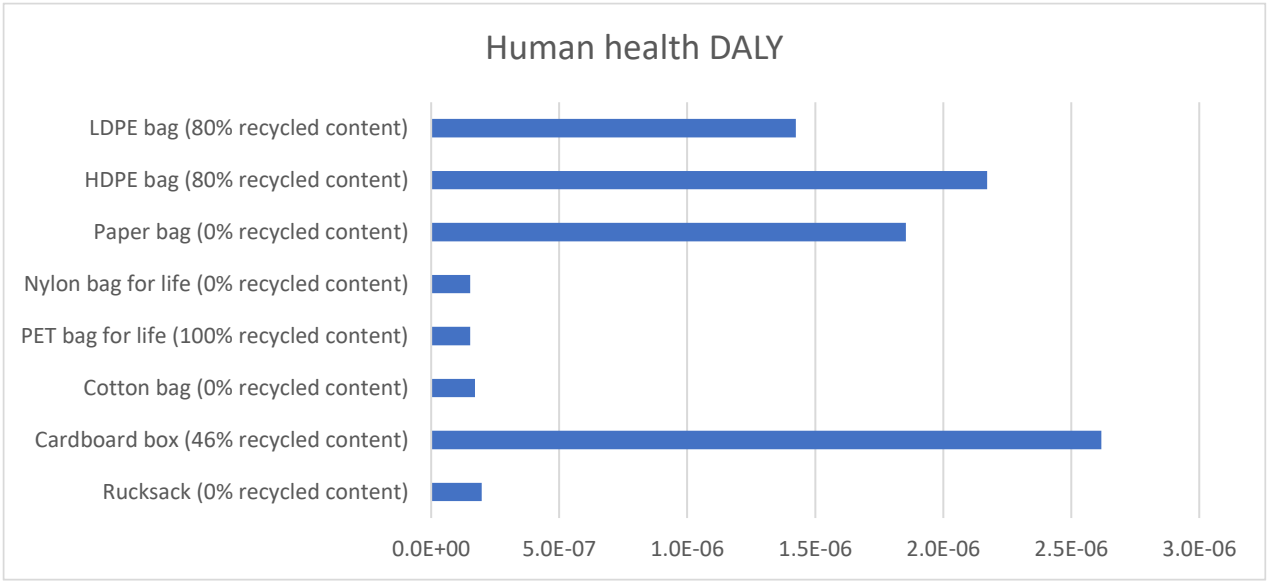


Figure 15: Potential human health damage (DALY) results per functional unit for the different carrying solutions analysed

The net scrap approach is described in Chapter 3.5.3 it includes credits for the energy recovered from waste incinerated in the (avoided burdens from energy produced by other means), as well as credits for material that is recycled (avoided production of primary material). For details of which energy and materials are assumed replaced, see Appendix 4.

Global warming impacts on human health dominate the results for this damage assessment category, fine particulate matter formation is also important.

The multiuse carriers have lower potential human health damage than the single use carriers. Of the single use carrier alternatives, the LDPE bag system has the lowest potential human health damage.

6.3.2 Ecosystems

The figure below shows the ecosystems damage assessment results for the different carrying solutions

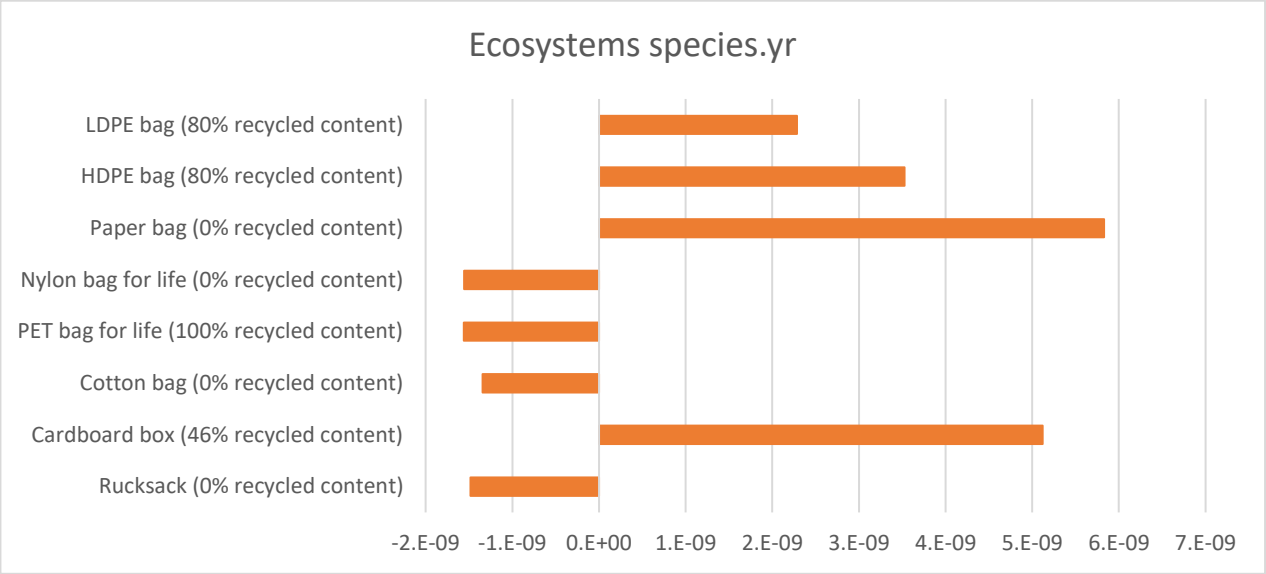


Figure 16: Potential ecosystems damage (species.yr) results per functional unit for the different carrying solutions analysed

Global warming impacts on ecosystems (particularly terrestrial) and land use dominate the results for this damage assessment category. For the paper bag ozone formation (effects of terrestrial ecosystems) more significant (about 20% of the result in Figure 16) than for other systems.

The multiuse carriers have lower potential ecosystems damage than the single use carriers. Of the single use carrier alternatives, the LDPE bag system has the lowest potential human health damage.

6.3.3 Resources

The figure below shows the resources damage assessment results for the different carrying solutions

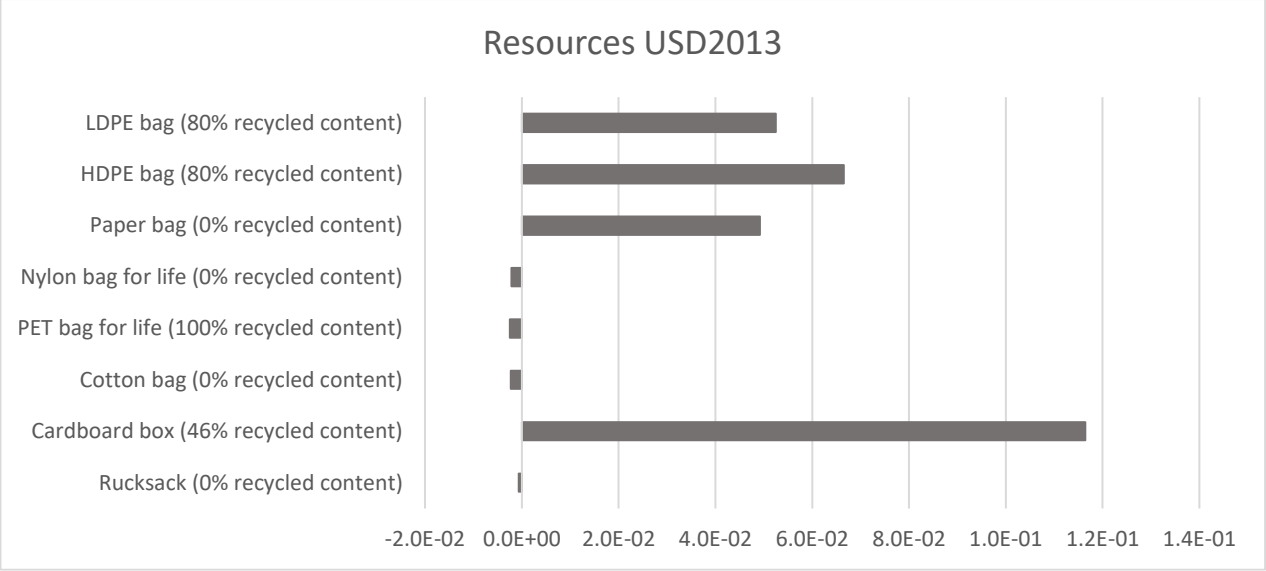


Figure 17: Potential resources damage (USD 2013) results per functional unit for the different carrying solutions analysed

Fossil resource scarcity is the most important contributor to the results for this damage assessment category.

The resources analysis results show that multiuse carriers are less resource intensive than single use carriers. The negative results for these are a result of the waste management stage of the carrier life cycle. This net scrap approach results in negative values due to avoided production of materials and energy. The paper bag has a lower resource damage cost than the single use cardboard carrier. This is largely related to the different masses of these two carriers.

6.3.4 Number of times a given carrier must be used – Endpoint net scrap

The approach used to calculate the number us times a given carrier must be used in order to be equivalent to the LDPE single use carrier is described in Chapter 6.1.4 The results of the calculation (see Equation 2) are given in the table below.

Table 23. The number of times a given carrier must be used to be equivalent to the reference carrier (cut-off) Endpoint damage assessment

Environmental impact	HDPE	Paper Bag (virgin)	Bag for life Nylon	Bag for life recycled PET	Bag for life Cotton	Card-board box	Rucksack
Potential human health damage	1.5	1.3	8	8	23	1.9	113
Potential ecosystems damage	1.5	1.9	6	5	61	1.7	65
Potential resources damage	1.3	0.9	10	4	7	2.2	96

If the value in Table 23 is greater than 1, then the carrier needs to be used more than the reference carrier in order to have equivalent impacts. In order to be better than the LDPE single use carrier for the three

damage assessment endpoints, the nylon multiuse bag for life will need to be used 10 times. Similarly, the PET multiuse bag would need to be used 8 times, the organic cotton bag 61 times and the rucksack 113 times.

The single use cardboard box would need to be used 2-3 times to compete with the LDPE single use alternative when the net scrap modelling approach is applied.

The number of uses shown in the table above are lower than the assumed baseline scenario of 1000 uses for the multiuse bags. This assumption has been found to be reasonable, both though the survey conducted in this study (see Appendix 2 and 3) and other literature sources (e.g., Briedis et al. (2019)'s number of uses was 1414 for a multiuse plastic bag with a lifetime of 10 years).

6.4 Midpoint results - Net scrap approach (EoL substitution)

The results in this chapter are from the system expansion modelling using the net scrap approach described in Chapter 3.5.3 .

The following correlations were found for this modelling approach:

Table 24. Correlation between impacts for the net scrap modelling approach

Impact category correlation tested for	**Impacts that correlate (R^2 value >0.8)**
Global warming potential	Water deprivation (R^2 = 0.8) Fine particulate matter formation (R^2 = 0.87) Terrestrial acidification (R^2 = 0.86) Freshwater ecotoxicity (R^2 = 0.96) Marine ecotoxicity (R^2 = 0.96) Human carcinogenic toxicity (R^2= 0.83) Human non-carcinogenic toxicity (R^2 = 0.81)
Marine Eutrophication	Ozone formation, Human health (R^2=0.82) Freshwater eutrophication (R^2=0.97) Terrestrial ecotoxicity (R^2= 0.94) Mineral resource scarcity (R^2= 0.83)
Water deprivation	Global warming (R^2= 0.8) Stratospheric ozone depletion (R^2= 0.93) Terrestrial acidification (R^2= 0.87) Freshwater eutrophication (R^2= 0.88) Terrestrial ecotoxicity (R^2 = 0.93) Freshwater ecotoxicity (R^2 = 0.83) Marine ecotoxicity (R^2 = 0.85) Human carcinogenic toxicity (R^2=0.98) Human non-carcinogenic toxicity (R^2 = 0.97) Mineral resource scarcity (R^2 = 0.96) Fossil resource scarcity (r^2 = 0.85)
Fine particulate matter formation	Global warming (R^2 = 0.87) Terrestrial acidification (R^2=0.88)
Ionizing radiation	No correlation found
Land use	Ozone formation, human health (R^2 = 0.94) Ozone formation, terrestrial ecosystems (R^2 = 0.89)
Plastic littering	No correlation found

Please note that the plastic littering results are related to the use phase for the carriers only, and can be found in Chapter 6.2.7 , so are not repeated in this chapter.

Table 25. Environmental impacts calculated per functional unit for the net scrap approach modelling approach (darkest shading is the largest result and lightest smallest for each impact category)[23]

Impact category	Units	LDPE bag	HDPE bag	Paper bag	Nylon bag	PET bag	Cotton bag	Cardboard box	Rucksack
Global warming	kg CO_2 eq	1.27	1.46	1.25	0.45	0.45	0.45	1.78	0.47
Stratospheric ozone depletion	kg CFC-11 eq	-1.4E-07	-8.3E-08	4.0E-07	-3.7E-07	-3.7E-07	-3.3E-07	7.8E-07	-3.0E-07
Ionizing radiation	kBq Co-60 eq	0.08	-6.3E-02	0.14	4.8E-02	4.8E-02	4.8E-02	0.10	4.9E-02
Ozone formation, Human health	kg NOx eq	3.3E-04	1.6E-03	6.2E-03	-6.8E-04	-6.8E-04	-6.6E-04	2.5E-03	-6.4E-04
Fine particulate matter formation	kg PM2.5 eq	7.6E-05	1.0E-03	5.6E-04	-4.3E-04	-4.3E-04	-4.1E-04	7.7E-04	-4.0E-04
Ozone formation, Terrestrial ecosystems	kg NOx eq	3.4E-04	1.6E-03	9.2E-03	-7.3E-04	-7.3E-04	-7.1E-04	2.5E-03	-6.9E-04
Terrestrial acidification	kg SO2 eq	7.9E-04	2.3E-03	2.7E-03	-3.7E-05	-3.9E-05	6.0E-06	3.4E-03	1.9E-05
Freshwater eutrophication	kg P eq	1.2E-04	1.0E-04	5.0E-04	5.2E-05	5.3E-05	7.3E-05	4.7E-04	5.6E-05
Marine eutrophication	kg N eq	4.1E-05	2.9E-05	1.0E-03	3.0E-05	3.0E-05	6.2E-05	7.8E-04	2.9E-05
Terrestrial ecotoxicity	kg 1,4-DCB	0.65	0.51	3.82	-0.19	-0.18	-0.18	3.82	-0.17
Freshwater ecotoxicity	kg 1,4-DCB	2.6E-02	1.9E-02	1.8E-02	-9.6E-03	-9.3E-03	-9.3E-03	4.2E-02	-9.1E-03

[23] Please note littering data included here is for the consumer use phase only. No other life cycle stages are included in this calculation

Marine ecotoxicity	kg 1,4-DCB	3,8E-02	2,9E-02	3,0E-02	-8,5E-03	-8,2E-03	-8,1E-03	5,9E-02	-7,9E-03
Human carcinogenic toxicity	kg 1,4-DCB	3,6E-02	3,5E-02	5,6E-02	1,3E-02	1,3E-02	1,3E-02	7,6E-02	1,5E-02
Human non-carcinogenic toxicity	kg 1,4-DCB	0,37	0,32	0,71	-0,02	-0,02	-0,02	1,09	-0,01
Land use	m2a crop eq	-1,9E-01	-1,6E-01	0,01	-3,1E-01	-3,1E-01	-2,9E-01	-0,16	-3,1E-01
Mineral resource scarcity	kg Cu eq	5,9E-04	3,5E-04	1,8E-03	-1,1E-04	-1,0E-04	-1,1E-04	2,4E-03	-9,8E-05
Fossil resource scarcity	kg oil eq	0,15	0,20	0,17	2,3E-03	1,6E-03	3,2E-03	0,36	7,3E-03
Water Use	m3	0,28	0,25	0,46	0,10	0,10	0,10	0,51	0,10
Marine Littering[23]	g	0,9	0,9	0	2,6E-06	4,9E-06	0	0	0
Marine litter impact	PAF-m^3-day	6,6E-02	6,6E-02	0	1,9E-07	3,6E-07	0	0	0

It should be noted that the bin liner contribution for the relevant carriers (paper bag, nylon bag, PET bag, organic cotton bag, cardboard box and rucksack) contributes to the results equally for these systems. In order to aid transparency, the impacts for the bin liners for the net scrap approach model are given in the table below.

Note that negative values are a result of the net scrap approach modelling including credits for the energy recovered from waste incinerated in the (avoided burdens from energy produced by other means), as well as credits for material that is recycled (avoided production of primary material). For details of which energy and materials are assumed replaced, see Appendix 4.

Table 26. Environmental impacts calculated per functional unit for the bin liner contribution to the results presented in Table 19 for the net scrap modelling approach (relevant for the paper bag, nylon bag, PET bag, organic cotton bag, cardboard box and rucksack carrier systems)

Impact category	Unit	Total
Global warming	kg CO2 eq	0.44
Stratospheric ozone depletion	kg CFC11 eq	-3.7E-07
Ionizing radiation	kBq Co-60 eq	4.8E-02
Ozone formation, Human health	kg NOx eq	-6.9E-04
Fine particulate matter formation	kg PM2.5 eq	-4.3E-04
Ozone formation, Terrestrial ecosystems	kg NOx eq	-7.4E-04
Terrestrial acidification	kg SO2 eq	-5.3E-05
Freshwater eutrophication	kg P eq	5.2E-05
Marine eutrophication	kg N eq	2.9E-05
Terrestrial ecotoxicity	kg 1,4-DCB	-0.19
Freshwater ecotoxicity	kg 1,4-DCB	-9.6E-03
Marine ecotoxicity	kg 1,4-DCB	-8.6E-03
Human carcinogenic toxicity	kg 1,4-DCB	1.3E-02
Human non-carcinogenic toxicity	kg 1,4-DCB	-2.5E-02
Land use	m2a crop eq	-0.31
Mineral resource scarcity	kg Cu eq	-1.2E-04
Fossil resource scarcity	kg oil eq	7.9E-04
Water deprivation	m3 world eq	9.9E-02

6.4.1 Global warming potential

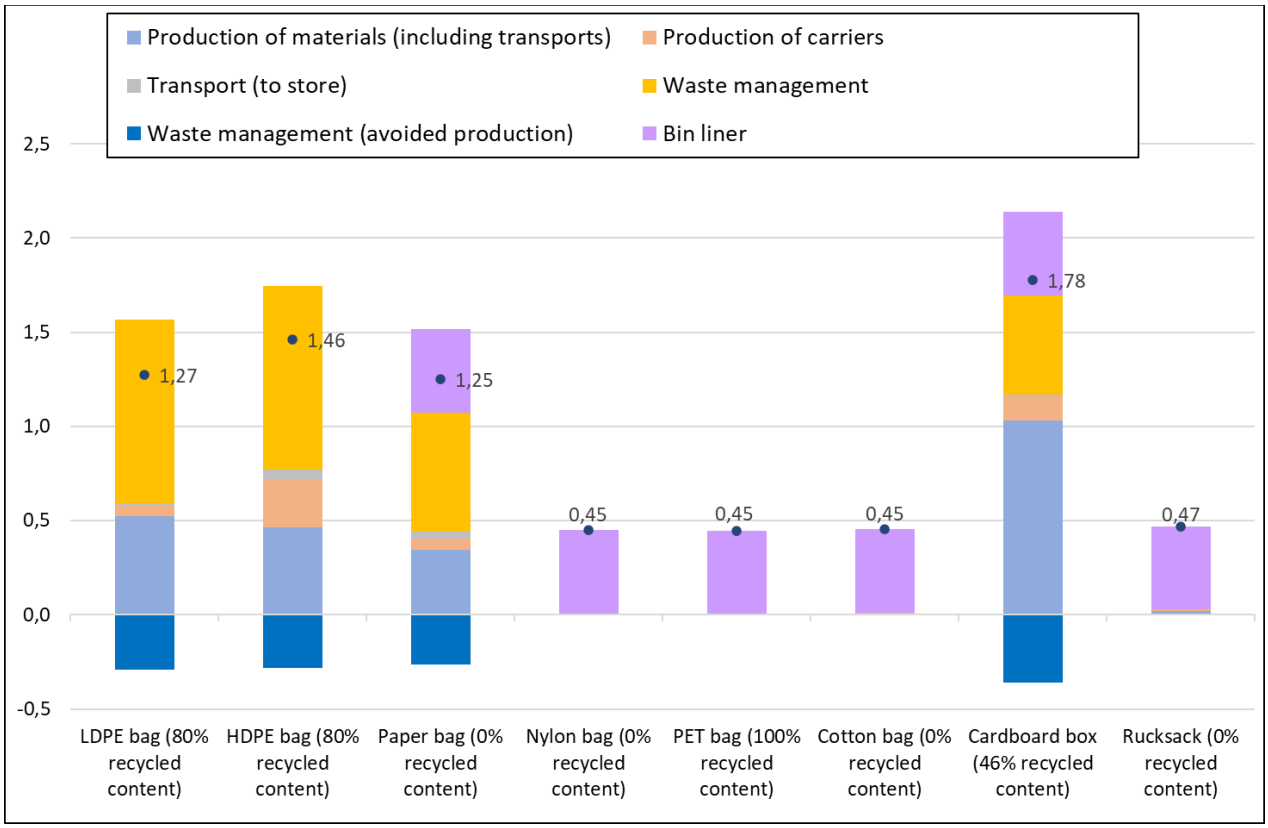


Figure 18: Global warming potential (kg CO_2-equivalents) results per functional unit for the different carrying solutions analysed

The results for GWP shown in Figure 18 now include credits for the energy recovered from the plastic incinerated in the plastic-based carrier systems, avoiding burdens from energy produced by other means. The "waste management (avoided production)" part of the bars in Figure 18 include credits for plastic material that is recycled also. For the paper and cardboard systems there are also credits for avoided production of primary material. As for the cut-off system model, the main impacts (above the x-axis) are from energy use for materials production and the emissions of fossil-based CO_2 from plastic incineration. Please note that in order to make the bin liner burdens and benefits clear and separate from the impacts of the main carrying solution value chain, the bin liner results are presented in sum for the bin-liner value chain. This means that the credits from replaced energy are not obvious on the graph above, but if this graph is compared to Figure 10, it becomes clear that they are included.

As for the cut-off modelling approach, the single use carriers have in general greater GWP impacts than the multiuse carriers, due to the multiuse carrier impacts for production processes being shared across the number of uses for the given carrier. The bin liner impacts are significant for the systems requiring a bin liner.

6.4.2 Marine eutrophication potential

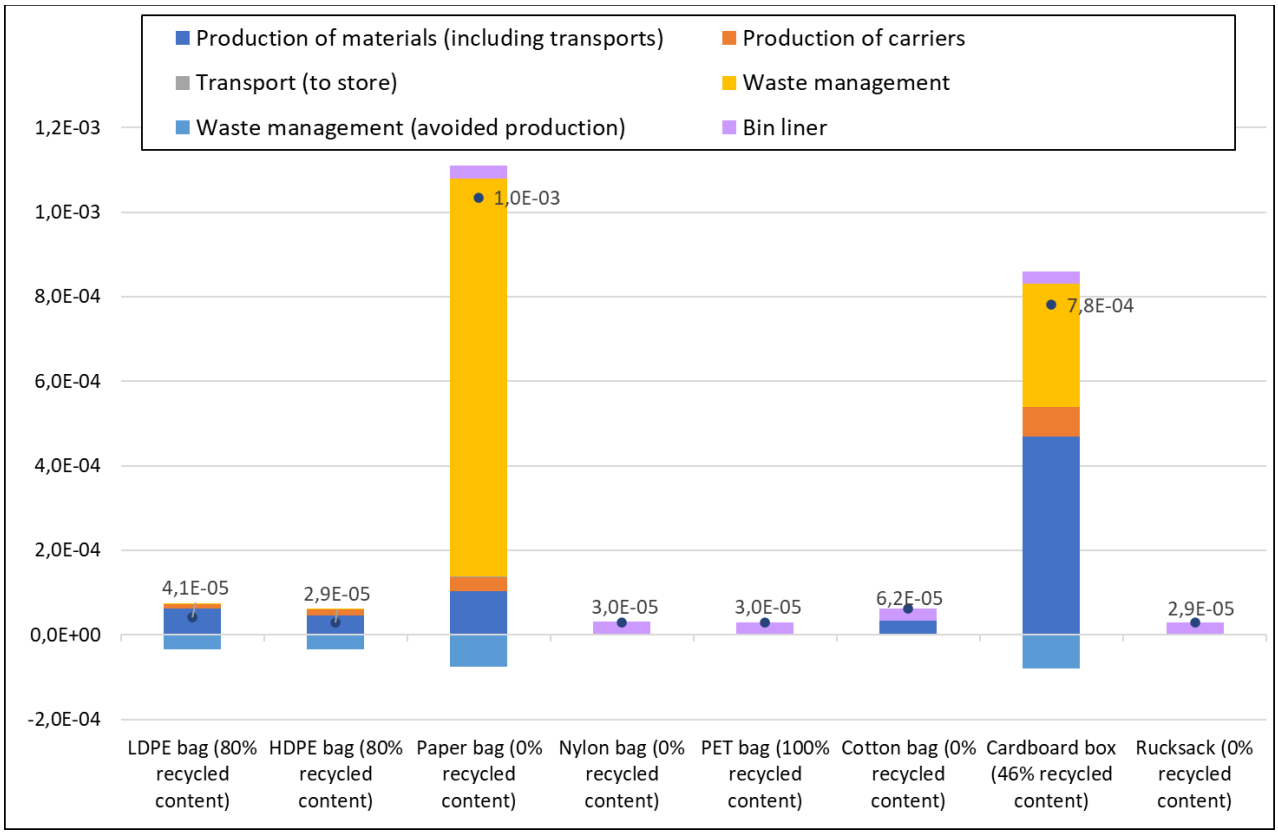


Figure 19: Marine eutrophication potential (kg N-equivalents) results per functional unit for the different carrying solutions analysed

As for the cut-off modelling approach, the top three emissions for the LDPE, HDPE, nylon, PET and rucksack systems are nitrogen organically bound, nitrate and ammonium, accounting for approximately 99% of these systems' marine eutrophication potential. These numbers are relatively low, compared to the paper and cardboard carriers. Treatment of plastic waste in landfill accounts for about 60% of the marine eutrophication potential associated with plastic-based systems. Further investigation of this result was performed, as deposition of plastic waste on landfill would not logically seem to contribute to the emissions listed previously. Infrastructure for sanitary landfill is about as important as the process specific burdens. The process specific burdens come from machine and electricity use (fossil fuels), incineration of sludge from wastewater treatment of leachate as well as from incineration of landfill gas at landfill sites. The leachate data are relevant for landfill in general, but their relevance for landfill of plastic is uncertain, biological activity on the surface of the plastic particles can lead so some small relevance, but the decomposition of the plastic itself is not likely to lead to these emissions. For the paper and cardboard system sludge from pulp and paper production is significant for the marine eutrophication potential for these systems. The rates of recycling for the paper bag and the cardboard box are different (71% and 94% respectively) influence the differences in waste management for these two carriers. The amount of material that is not recycled is assumed incinerated, contributing to this impact category in addition to process and sludge emissions from the recycling processes. The paper bag is assumed recycled in a paper recycling process, while the cardboard box is assumed recycled in a carboard process (see Appendix 4).

Approximately 50% of the marine eutrophication potential impact for the cotton bag systems arises from organic seed cotton production, the bin liner dominates the other half of the impacts.

It should be noted that emissions contributing to marine eutrophication potential (i.e. nitrogen organically bound, nitrate and ammonium) need to reach the sea in order to contribute to marine eutrophication. Thus, the locations of the relevant facilities are important for whether the potential marine eutrophication impacts are realised. Due to confidentiality aspects, the locations of suppliers of carriers are not provided in this report (country is provided, see Appendix 4).

6.4.3 Water deprivation potential

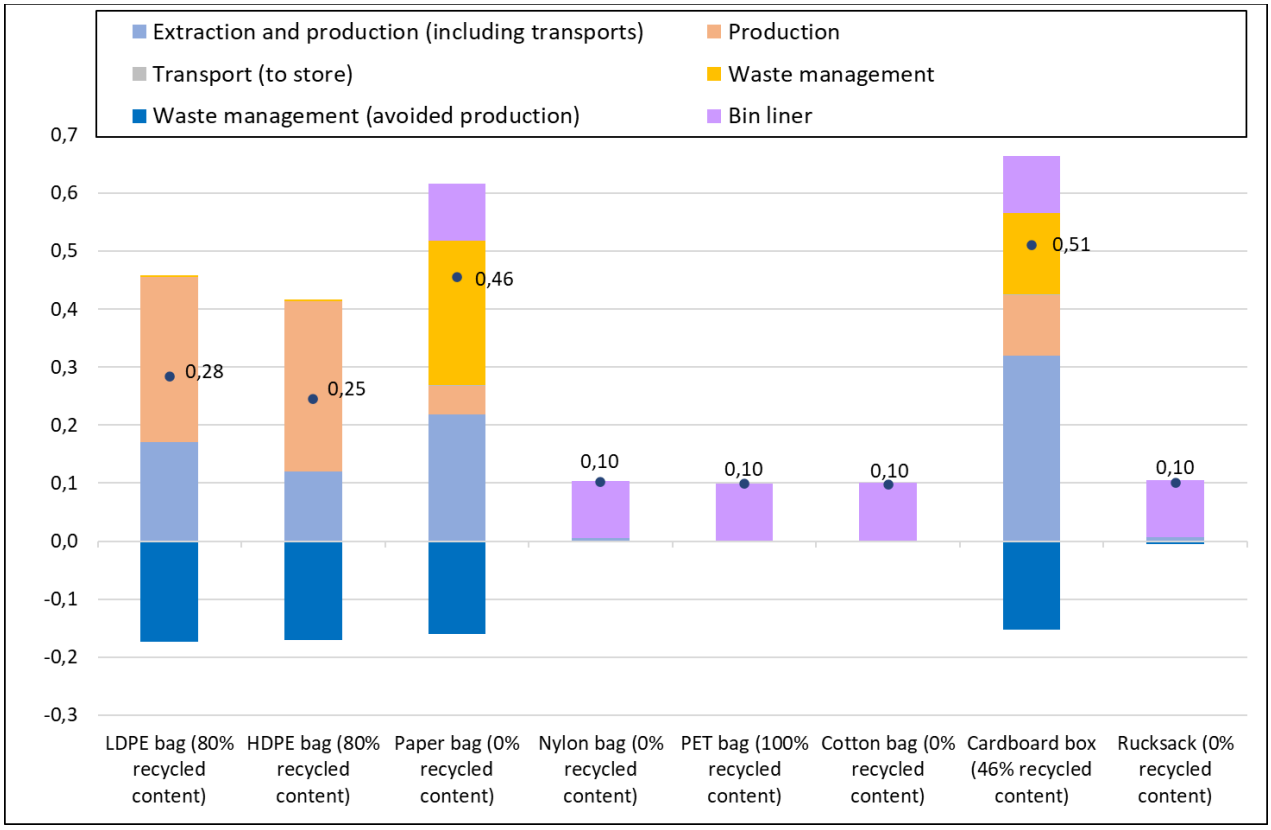


Figure 20: Water deprivation potential (m^3 world eq.) results per functional unit for the different carrying solutions analysed.

As for the cut-off modelling approach, the single use carriers have in general greater water use impacts than the multiuse carriers, due to the multiuse carrier impacts for production processes being shared across the number of uses for the given carrier. The bin liner impacts are significant for the systems requiring a bin liner.

As described for the cut-off modelling approach, water deprivation potential for the organic cotton data set chosen is low. However, when modelling with the net scrap approach the final results are dominated by the bin-liner. The choice of «organic cotton bag» is not so significant in this modelling approach, as for the cut-off modelling. The reason for this is the replaced material when cotton is recycled, which is conventionally farmed flax (see Chapter 5.7, Appendix 4 and Appendix 5 for the source of this assumption).

6.4.4 Fine particulate matter formation potential

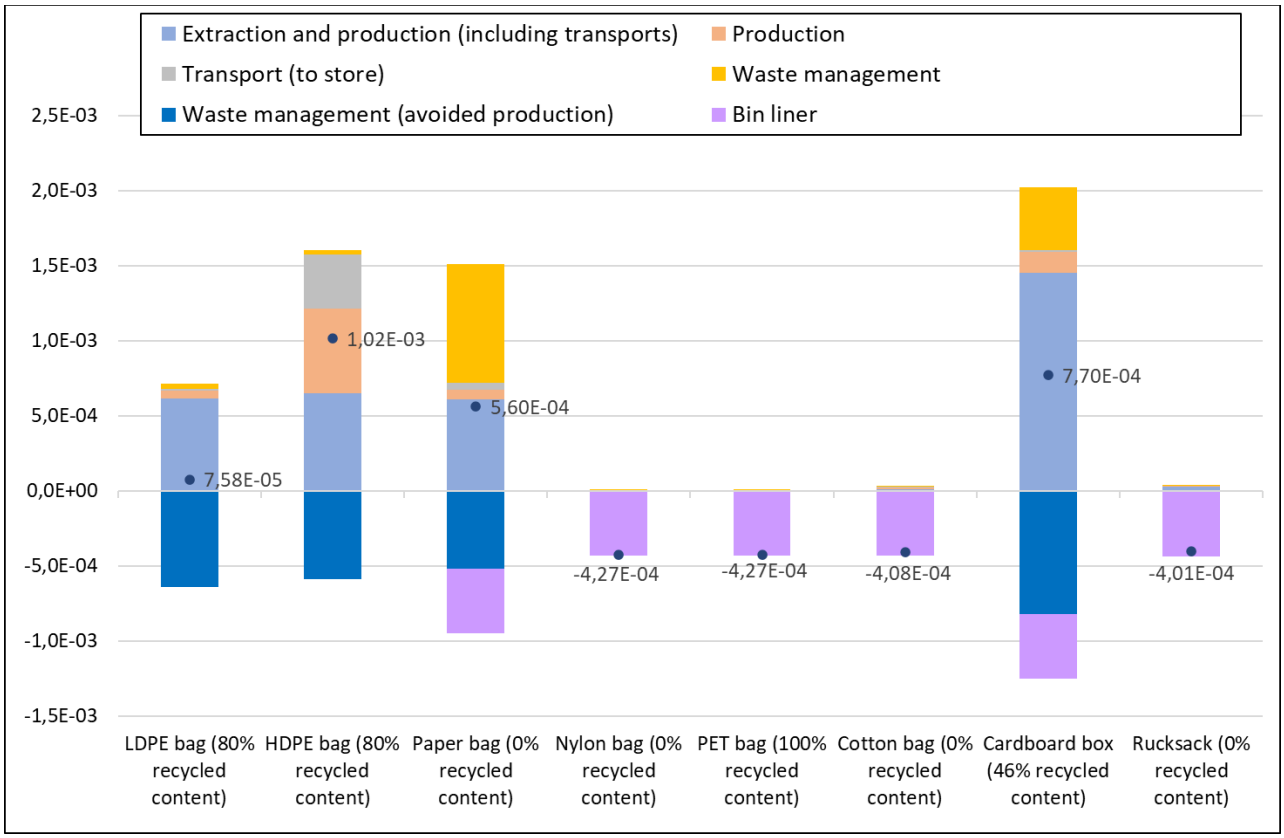


Figure 21: Fine particulate matter formation potential (kg PM2.5 equivalents) results per functional unit for the different carrying solutions analysed

The dominating emission, with the greatest contribution to the fine particulate matter potential for these carrier systems is sulphur dioxide. Nitrogen oxides are also important. Plastic production (raw materials) and extrusion and their energy sources are the most important contributors for the plastic-based systems. Transport by ship, over long distances is also important for this impact category (often heavy fuels are used for long distance transport by ship, thus sulphur dioxide, nitrogen oxide and direct fine particulate emissions arise). This can be clearly seen in the “transport to store” (grey part of the bar) result for the HDPE carrier system (transport from Southeast Asia). Fossil-fuel heavy electricity mixes also have more sulphur dioxide, nitrogen oxide and direct fine particulate emissions. The replacement of energy and materials from the waste management of the carriers and bin liners has a significant impact on the results for all systems.

For the paper and cardboard systems waste management shows up as more significant than for the other systems. Sulphur dioxide accounts for the majority of the impacts, followed by nitrogen oxides. There are direct nitrogen oxide and sulphur dioxide emissions reported for the processes involved in paper and board manufacture and the energy sources used. The recycling process also involves similar emissions and energy sources.

As for the cut-off modelling approach, the single use carriers have in general greater fine particulate matter impacts than the multiuse carriers, due to the multiuse carrier impacts for production processes being shared across the number of uses for the given carrier. The bin liner impacts are significant for the systems requiring a bin liner.

6.4.5 Ionizing radiation potential

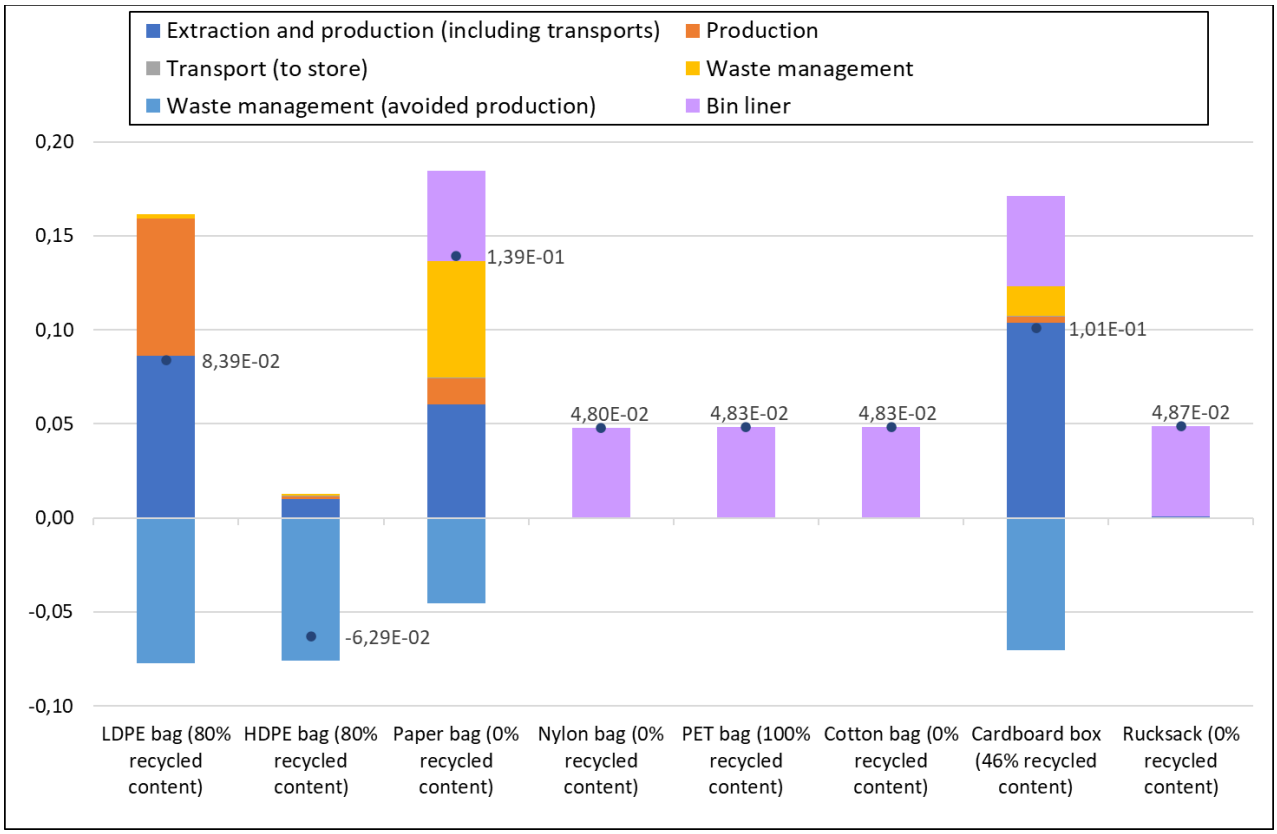


Figure 22: Ionizing radiation potential (kBq Co-60 equivalents) results per functional unit for the different carrying solutions analysed

As for the results for ionizing radiation for the cut-off modelling approach, the majority of the contributions to the ionizing radiation impact potential are from nuclear power processes within the electricity mixes used. Swedish electricity has a larger proportion of nuclear power in it than Norwegian, or Southeast Asian electricity. It is also the case here that the single use carriers have in general greater impacts than the multiuse carriers, due to the multiuse carrier impacts for production processes being shared across the number of uses for the given carrier. The bin liner impacts are significant for the systems requiring a bin liner.

6.4.6 Land use potential

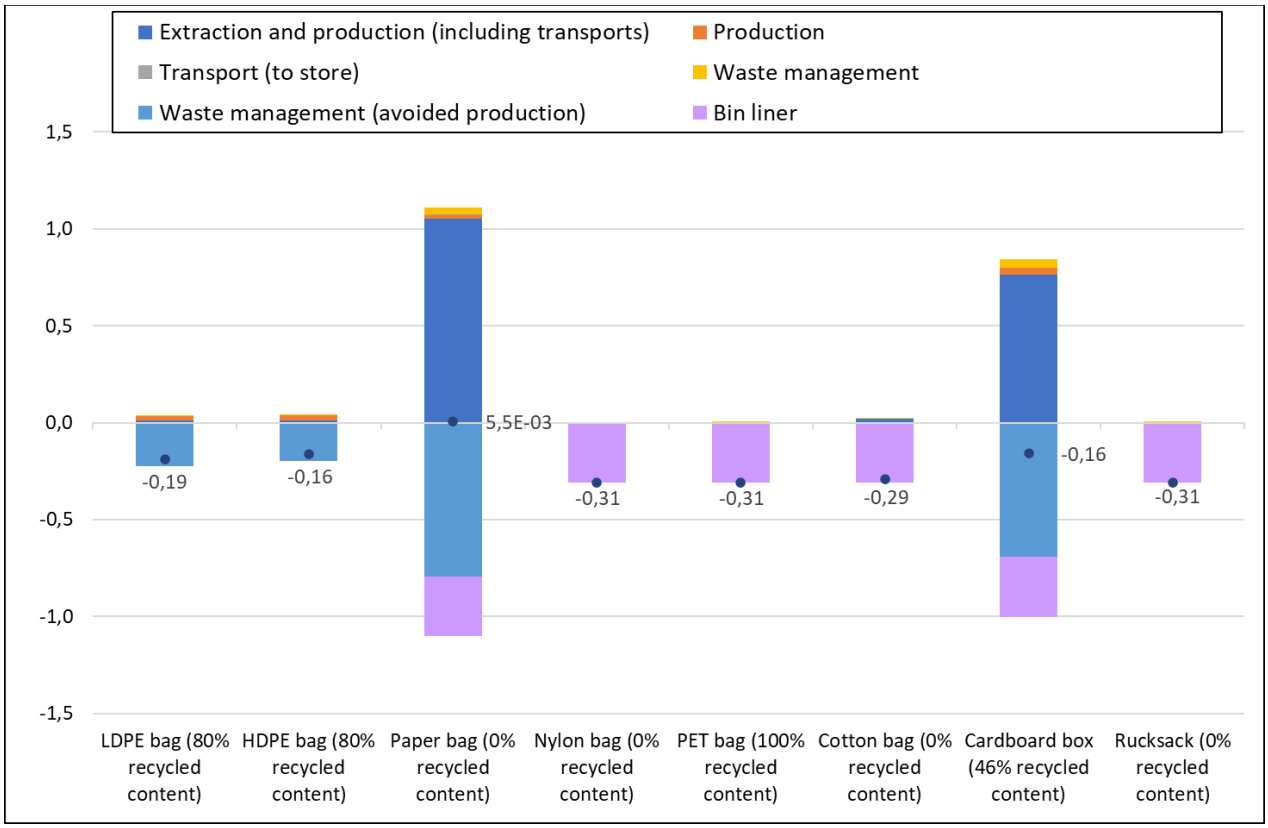


Figure 23: Land use potential (m^2a crop equivalents) results per functional unit for the different carrying solutions analysed

The results for land use are dominated by the replacement of energy that is modelled to replace Norwegian district heat. This district heat has a large proportion of wood-based fuel, thus forestry land area is not required. This gives a land use benefit for the EoL modelling approach for the carriers where energy is generated from incinerating the carrier materials. The forestry component of the avoided materials from recycling is the reason that the area use benefits are larger for the paper bag and cardboard box systems.

As for the cut-off modelling approach, the single use carriers have in general greater land use impacts than the multiuse carriers, due to the multiuse carrier impacts for production processes being shared across the number of uses for the given carrier.

6.4.7 Number of times a given carrier must be used - Midpoint net scrap

If the LDPE single use carrier is used as the reference system, the results for the analysis of the net scrap modelling approach for all of the systems can be used to calculate the number of times a given carrier must be used to be equivalent to the LDPE reference carrier. For further details on how these calculations were performed, see Chapter 6.2.8 .

Table 27. The number of times a given carrier must be used to be equivalent to the reference carrier (net scrap).

Environmental impact	HDPE	Paper Bag (virgin)	Bag for life Nylon	Bag for life recycled PET	Bag for life Cotton	Cardboard box	Rucksack
Global warming	1.1	1.0	7	6	13	1.6	82
Stratospheric ozone depletion	0.6	3.4	14	7	192	5.0	855
Ionizing radiation	-0.7	2.5	2	9	11	1.5	55
Ozone formation, Human health	4.9	6.7	11	12	30	3.1	136
Fine particulate matter formation	13.4	2.0	11	11	49	2.4	161
Ozone formation, Terrestrial ecosystems	4.7	9.2	11	12	28	3.0	132
Terrestrial acidification	3	3.3	20	17	71	4.1	224
Freshwater eutrophication	0.9	7	8	17	320	6	191
Marine eutrophication	0.7	85	84	34	2796	63.4	-57
Terrestrial ecotoxicity	0.8	4.8	3	15	11	4.8	67
Freshwater ecotoxicity	0.7	0.8	1	9	10	1.4	34
Marine ecotoxicity	0.8	0.8	1	9	10	1.4	35
Human carcinogenic toxicity	1.0	1.9	10	19	20	2.7	280
Human non-carcinogenic toxicity	0.9	1.9	4	14	19	2.8	84
Land use	0.9	2.6	-1	-1	161	1.3	-10
Mineral resource scarcity	0.6	2.8	8	19	7	3.6	70
Fossil resource scarcity	1.3	1.2	10	6	17	2.4	114
Water deprivation	0.9	1.9	21	5	-2	2.2	30
Marine littering	1.0	N/A[24]	2.9E-03	5.4E-03	N/A	N/A	N/A
Marine litter impact	1.0	N/A	2.9E-06	5.4E-06	N/A	N/A	N/A

If the value in Table 27 is greater than 1, then the carrier needs to be used more than the reference carrier in order to have equivalent impacts. There are some negative values in the table above. This due to the subtraction of avoided burdens. The negative values result from the cases where the impact of the LDPE bag is negative, or the impact of the bin liners are negative and sometimes the impact of the compared

[24] Not applicable for this carrier (see Chapter 6.2.7

carrier is negative. In order to be better than the LDPE bag system for all of the impacts considered the paper bag will need to be used 85 times. Similarly, the nylon bag would need to be used 84 times, the PET bag 34, the cotton bag 2796 times, the cardboard box 63 and the rucksack 855 times.

Briedis et al. (2019) suggests a number of uses value of 1414 for a multiuse plastic bag (equivalent to this study's nylon and PET bags) with a lifetime of 10 years. For the multiuse bags the maximum number of uses (from the surveys reported in appendices 2 and 3) are tested in the sensitivity analysis. For multiuse bags made from nylon and PET, these upper values are 3120. For cotton the upper value is 4680, while for the rucksack it is 7800. Thus, the number of uses required in Table 27 are higher than the base case scenarios (1000 times for multiuse bags and 2600 for the rucksack, see Table 3, Chapter 3.3), but not as high as the upper limits chosen for the sensitivity analysis.

6.5 Sensitivity analysis

The Baseline value shown in Table 28 is based on the cut-off modelling approach, which is also known as the "base case" in previous chapters. "-" in the table means that an analysis was not performed (e.g., there is no lower value tested than the baseline value for number of uses for the LDPE bag).

Table 28. Parameters tested in the sensitivity analysis and the values applied

Parameter [unit]	Baseline value	Low value	High value
LDPE bag			
Recycled content in LDPE bag [%]	80	0	100
Number of uses of LDPE bag [-]	1	-	5
Volume utilisation of LDPE bag [%]	50	9.17	80
Distance from production site to store [km]	160	-	500
Distance from consumer to incineration [km]	85	-	500
Distance from consumer to recycling [km]	905	-	1 200
HDPE bag			
Recycled content in HDPE bag [%]	80	0	100
Number of uses of HDPE bag [-]	1	-	5
Volume utilisation of HDPE bag [%]	50	9.17	80
Distance from production site to store [km]	15 900	160	-
Distance from consumer to incineration [km]	85	-	500
Distance from consumer to recycling [km]	905	-	1 200
Paper bag			
Recycled content in paper bag [%]	0	-	100
Number of uses of paper bag [-]	1	-	5
Volume utilisation of paper bag [%]	50	4.58	80
Nylon bag			
Number of uses of nylon bag [-]	1 000	2	3120
Volume utilisation of nylon bag [%]	50	4.78	80
PET bag			
Number of uses of PET bag [-]	1 000	2	3120
Volume utilisation of PET bag [%]	50	7.33	80
Cotton bag			
Number of uses of cotton bag [-]	1 000	2	4 680
Maximum volume cotton bag [L]	25	12	30
Volume utilisation of cotton bag [%]	50	4.4	80
Cardboard box			

Recycled content in cardboard box [%]	46	0	100
Number of uses of cardboard box [-]	1	-	50
Volume utilisation of cardboard box [%]	50	2.75	80
Rucksack			
Number of uses of rucksack [-]	2 600	39	7 800
Volume utilisation of rucksack [%]	50	3.67	80

The sensitivity analysis for carrier volume includes a cardboard box for small shopping events. It is very unlikely that a customer would use a cardboard box with a volume of 40 litres for a small shopping purchase of 1.1 litres. It has been included in the small shopping sensitivity for consistency.

The ranges shown for number of uses were established through the surveys presented in Appendices 2 and 3, as well as dialogue with the critical panel members.

Electricity models can affect the outcome of the analyses. Thus, a sensitivity analysis associated with changing the electricity models used for carrier production from country specific to European has been performed.

The results for sensitivity analyses are shown using the endpoint damage assessment indicators.

6.5.1 Recycled content

The table below shows the results for changes in recycled content shown in Table 28 for the endpoint damage assessment indicators.

Table 29. Recycled content sensitivity analysis results

		LDPE			HDPE			Paper		Cardboard		
Damage category	**Unit**	**Baseline scenario**	**Recycled content 0%**	**Recycled content 100%**	**Baseline scenario**	**Recycled content 0%**	**Recycled content 100%**	**Baseline scenario**	**Recycled content 100%**	**Baseline scenario**	**Recycled content 0%**	**Recycled content 100%**
Human health	DALY	2.40E-06	3.03E-06	2.24E-06	3.09E-06	3.64E-06	2.96E-06	2.14E-06	3.18E-06	3.63E-06	3.83E-06	3.41E-06
Ecosystems	species.yr	5.61E-09	7.28E-09	5.19E-09	6.58E-09	8.23E-09	6.16E-09	1.45E-08	8.68E-09	1.47E-08	1.98E-08	8.71E-09
Resources	USD2013	7.86E-02	2.50E-01	3.57E-02	9.29E-02	2.68E-01	4.92E-02	5.93E-02	7.60E-02	1.21E-01	1.02E-01	1.44E-01

The baseline scenario is the cut-off modelling scenario, as shown in Chapter 6.2. The changes in results are shown for the potential human health damage in Figure 24.

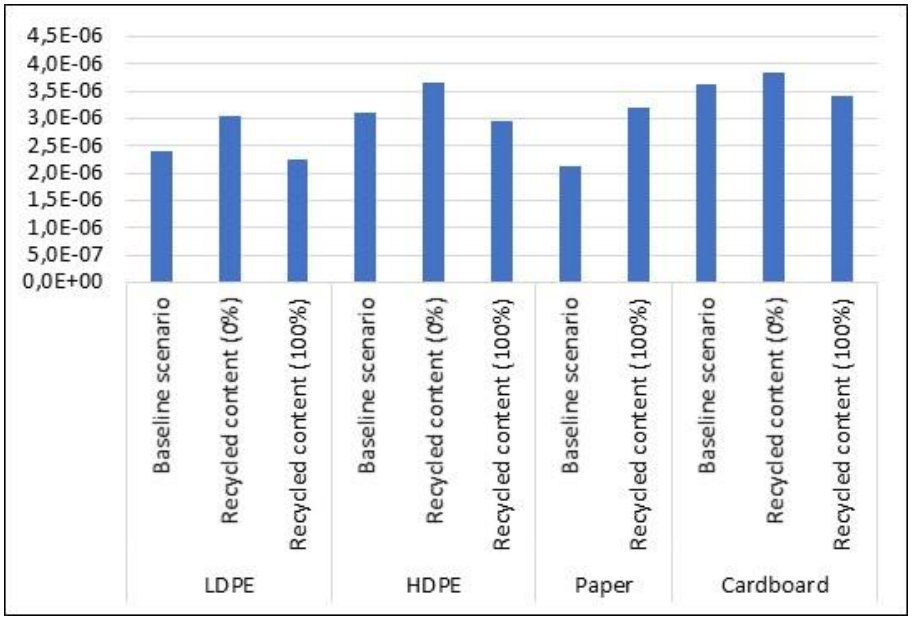


Figure 24: Potential human health damage (DALY) results per functional unit for the different recycled content scenarios analysed

For the plastic bags (LDPE and HDPE) and cardboard systems, an increase or reduction in recycled content will similarly increase, or reduce the potential human health damage. The increase in potential human health damage associated with increased paper bag recycling seems counterintuitive. Further analysis of the results show that, as previously for potential human health damage, potential effects of global warming and fine particulate matter formation on human health are most significant. The contributors increase with increased use of recycled material. This is due to the use of fossil-based energy carriers. The recycling process energy mix is described in the database as using natural gas, heavy fuel oil and biogas. Electricity use in the factory is with European electricity (see Appendix 4). This energy mix results in higher global warming and particulate formation potential than for the virgin kraft paper, where the baseline scenario for European kraft paper. Assessing these processes in detail shows that there is a large discrepancy in energy carrier use per kg in the factory data for recycled paper (about 14 MJ/kg) when compared to the equivalent data for virgin material (about 2.5 MJ/kg). Thus, the discrepancy in results for recycled paper vs virgin paper should not be given weight by the reader. The data used for recycled paper should be re-evaluated in further work.

The figure below shows the ecosystems damage assessment results for the different recycled content analyses.

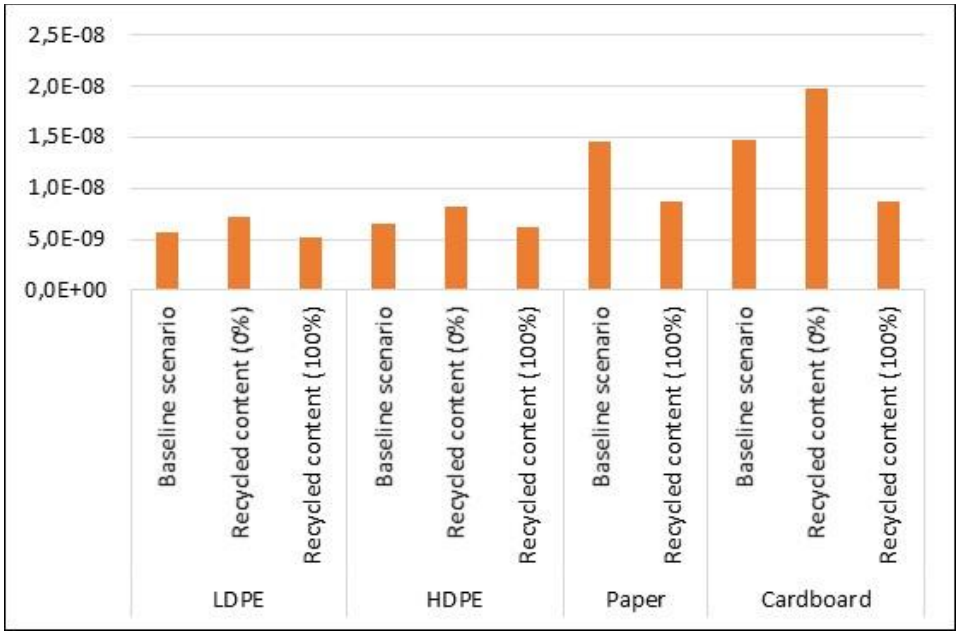


Figure 25: Potential ecosystems damage (species.yr) results per functional unit for the different recycled content scenarios analysed.

For the single use carrier systems analysed, an increase or reduction in recycled content will similarly increase, or reduce the potential for ecosystems damage.

The figure below shows the resources damage assessment results for the different carrying solutions.

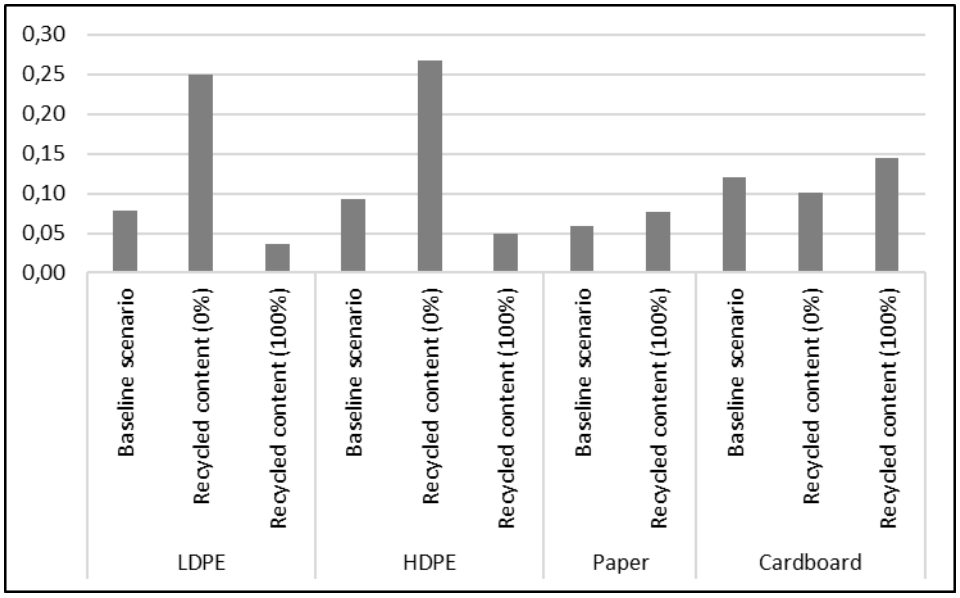


Figure 26: Potential resources damage (USD 2013) results per functional unit for the different recycled content scenarios analysed.

For the plastic bags (LDPE and HDPE) an increase or reduction in recycled content will similarly increase, or reduce the potential resources damage. The increase in potential resources damage associated with increased paper bag and cardboard recycling seems counterintuitive. Fossil energy carriers dominate the potential resources damage category, therefore the aspects highlighted under Figure 24, where the quality of the recycled paper data is questioned are also important here.

6.5.2 Number of uses

The table below shows the results for changes in number of uses shown in Table 28 for the potential endpoint damage categories human health, ecosystems and resources.

Table 30. Number of uses sensitivity analysis results

		LDPE		HDPE		Paper		Nylon			PET		
Damage category	**Unit**	**Baseline scenario (single use)**	**Number of uses = 5**	**Baseline scenario (single use)**	**Number of uses = 5**	**Baseline scenario (single use)**	**Number of uses = 5**	**Baseline scenario (1 000 uses)**	**Number of uses = 2**	**Number of uses = 3120**	**Baseline scenario (1 000 uses)**	**Number of uses = 2**	**Number of uses = 3120**
Human health	DALY	2.40E-06	1.15E-06	3.09E-06	1.29E-06	2.14E-06	1.14E-06	9.01E-07	6.36E-06	8.93E-07	9.01E-07	6.51E-06	8.93E-07
Ecosystems	species.yr	5.61E-09	2.69E-09	6.58E-09	2.88E-09	1.45E-08	4.57E-09	2.11E-09	1.47E-08	2.10E-09	2.11E-09	1.28E-08	2.10E-09
Resources	USD2013	7.86E-02	2.57E-02	9.29E-02	2.86E-02	5.93E-02	2.25E-02	1.39E-02	2.84E-01	1.35E-02	1.36E-02	1.35E-01	1.34E-02

		Cotton			Cardboard		Rucksack		
Damage category	**Unit**	**Baseline scenario (1 000 uses)**	**Number of uses = 2**	**Number of uses = 4680**	**Baseline scenario (single use)**	**Number of uses = 50**	**Baseline scenario (1 000 uses)**	**Number of uses = 39**	**Number of uses = 7800**
Human health	DALY	9.20E-07	1.59E-05	8.96E-07	3.63E-06	9.45E-07	9.45E-07	4.59E-06	9.08E-07
Ecosystems	species.yr	2.33E-09	1.22E-07	2.14E-09	1.47E-08	2.34E-09	2.20E-09	9.42E-09	2.13E-09
Resources	USD2013	1.37E-02	2.09E-01	1.34E-02	1.21E-01	1.55E-02	1.54E-02	1.53E-01	1.40E-02

The baseline scenario is the cut-off modelling scenario, as shown in Chapter 6.2. The changes in results are shown using the potential human health damage in Figure 27.

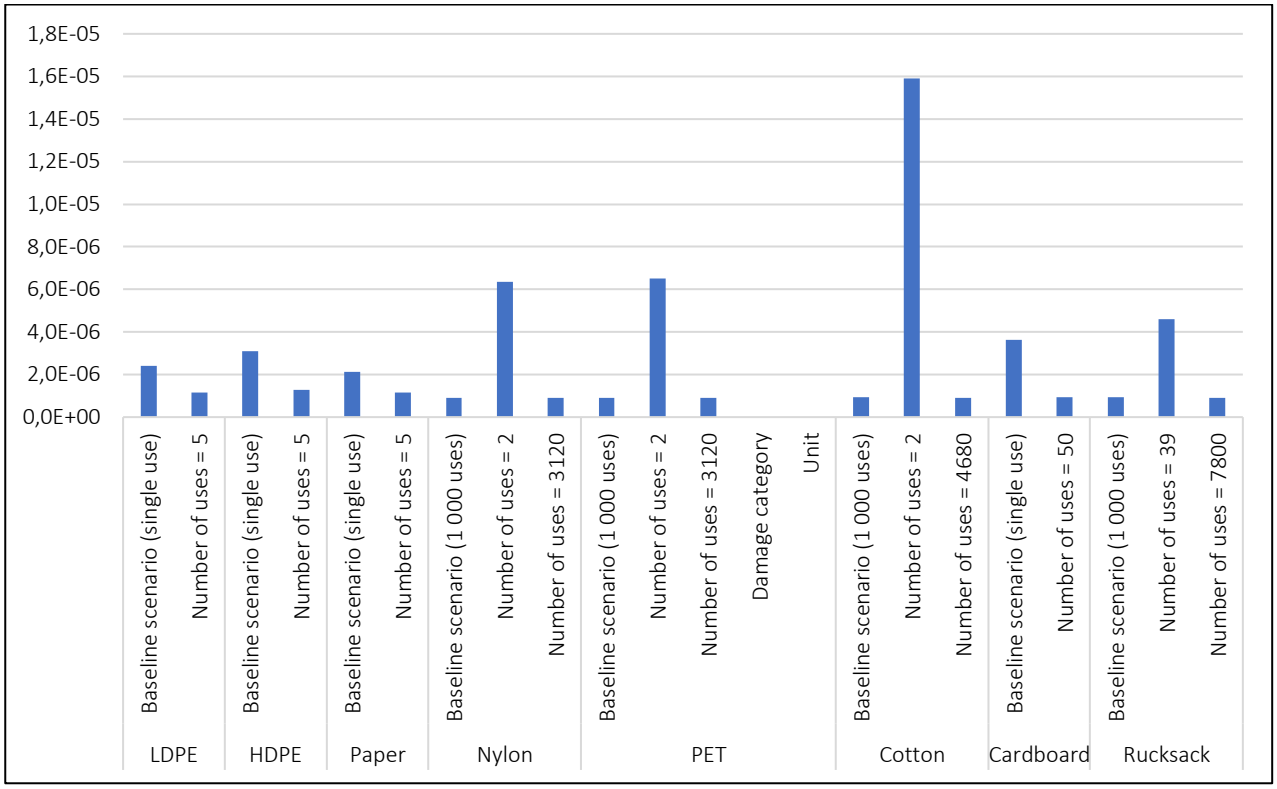


Figure 27: Potential human health damage (DALY) results per functional unit for the different scenarios analysed for number of uses.

It is logical to see that the number of uses affects the results. If the number of uses increases the potential damage reduces.

One notable result for the single use plastic bags is that if they are used 5 times, they then become competitive with the multiuse system baseline scenarios. The need for bin liners as the single use bags become multi use has been included in the calculation as follows:

> If the single use bags become multiuse (5 times) a consumer would then purchase not 19 single use carriers, but one fifth of this number, or 3.8 carrier bags. As the functional unit requires 15.2 bin liner bags, the single use carriers would only fulfil part of this bin liner function and 11.4 bin liners would need to be purchased.

For multiuse carriers where the number of uses is very large, the results are dominated by the bin liners and thus the high value does not differ significantly from the baseline.

The results for potential damage to ecosystems and resources show very similar trends to Figure 27 and are included below.

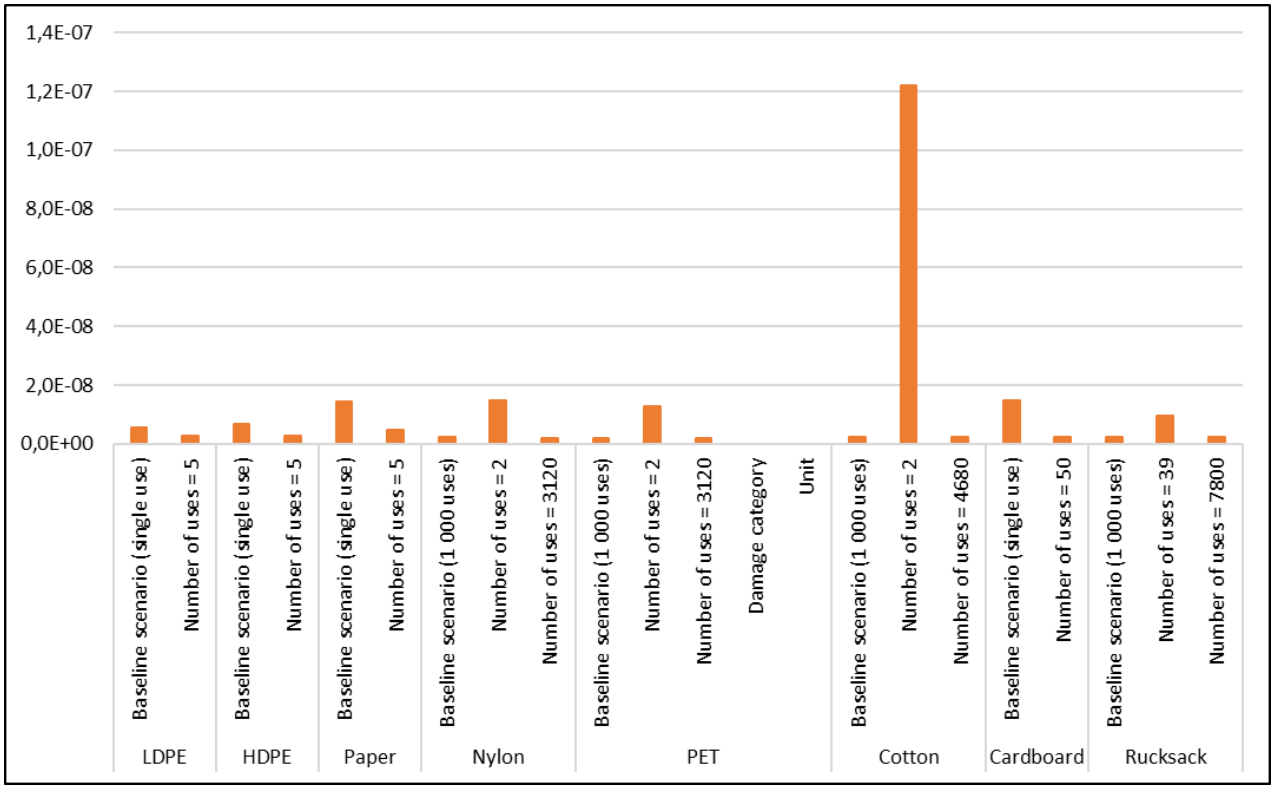


Figure 28: Potential ecosystems damage (species.yr) results per functional unit for the different scenarios analysed for number of uses.

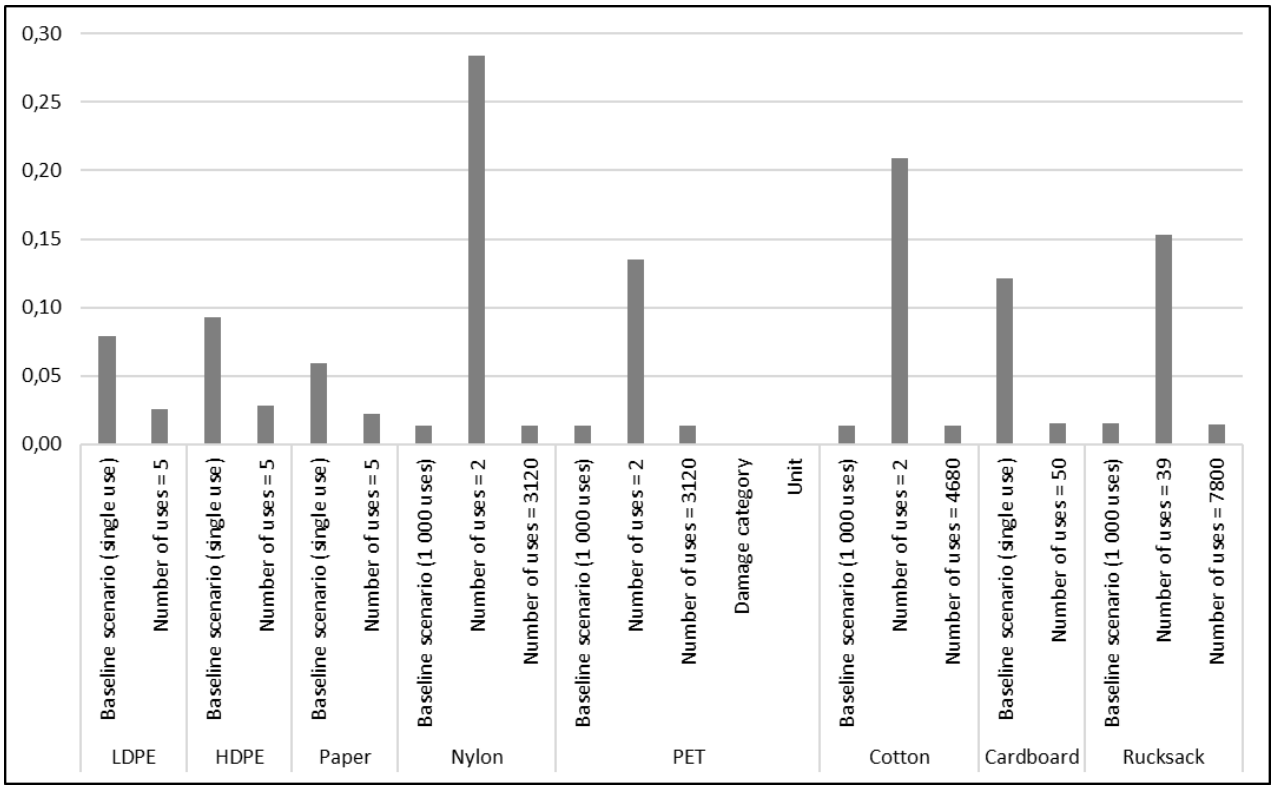


Figure 29: Potential resources damage (USD 2013) results per functional unit for the different scenarios analysed for number of uses.

6.5.3 Volume utilisation

The table below shows the results for changes in volume utilisation shown in Table 28 for the three endpoint potential damage categories.

Note that the number of bin liners changes for a small shopping, or a large shopping event. The LDPE baseline case includes the replacement of 15.2 bin liners (80% of the 19 LDPE bags are assumed used as bin liners). The small shopping case (with a volume utilisation of 1.1 litre, or 9.17%) leads to 104 bags used, replacing 83 bin liners. The large shopping event uses 12 bags, replacing 10 bin liners. The results shown in the figures below include the changes in the number of bin liners needed.

Table 31. Volume utilisation sensitivity analysis results

		LDPE			HDPE			Paper			Nylon		
Damage category	**Unit**	**Baseline scenario (50%)**	**Capacity utilisa-tion (9.17%)**	**Capacity utilisa-tion (80%)**	**Baseline scenario (50%)**	**Capacity utilisa-tion (9.17%)**	**Capacity utilisa-tion (80%)**	**Baseline scenario (50%)**	**Capacity utilisa-tion (4.58%)**	**Capacity utilisa-tion (80%)**	**Baseline scenario (50%)**	**Capacity utilisa-tion (4.78%)**	**Capacity utilisa-tion (80%)**
Human health	DALY	2.40E-06	1.31E-05	1.50E-06	3.09E-06	1.69E-05	1.93E-06	2.14E-06	1.45E-05	1.67E-06	9.01E-07	1.00E-06	8.97E-07
Ecosystems	species.yr	5.61E-09	3.06E-08	3.51E-09	6.58E-09	3.59E-08	4.11E-09	1.45E-08	1.37E-07	9.84E-09	2.11E-09	2.35E-09	2.10E-09
Resources	USD2013	7.86E-02	4.29E-01	4.91E-02	9.29E-02	5.06E-01	5.80E-02	5.93E-02	5.16E-01	4.21E-02	1.39E-02	1.90E-02	1.37E-02

		PET			Cotton			Cardboard			Rucksack		
Damage category	**Unit**	**Baseline scenario (50%)**	**Capacity utilisa-tion (7.33%)**	**Capacity utilisa-tion (80%)**	**Baseline scenario (50%)**	**Capacity utilisa-tion (4.4%)**	**Capacity utilisa-tion (80%)**	**Baseline scenario (50%)**	**Capacity utilisa-tion (2.75%)**	**Capacity utilisa-tion (80%)**	**Baseline scenario (50%)**	**Capacity utilisa-tion (3.67%)**	**Capacity utilisa-tion (80%)**
Human health	DALY	9.01E-07	9.67E-07	8.97E-07	9.20E-07	1.23E-06	9.09E-07	3.63E-06	5.08E-05	2.61E-06	9.45E-07	1.65E-06	9.25E-07
Ecosystems	species.yr	2.11E-09	2.24E-09	2.10E-09	2.33E-09	4.81E-09	2.24E-09	1.47E-08	2.31E-07	9.96E-09	2.20E-09	3.59E-09	2.16E-09
Resources	USD2013	1.36E-02	1.50E-02	1.35E-02	1.37E-02	1.78E-02	1.36E-02	1.21E-01	1.97E+00	8.07E-02	1.54E-02	4.19E-02	1.46E-02

The baseline scenario is the cut-off modelling scenario, as shown in Chapter 6.2. The changes in results are shown using the three end point potential damage categories.

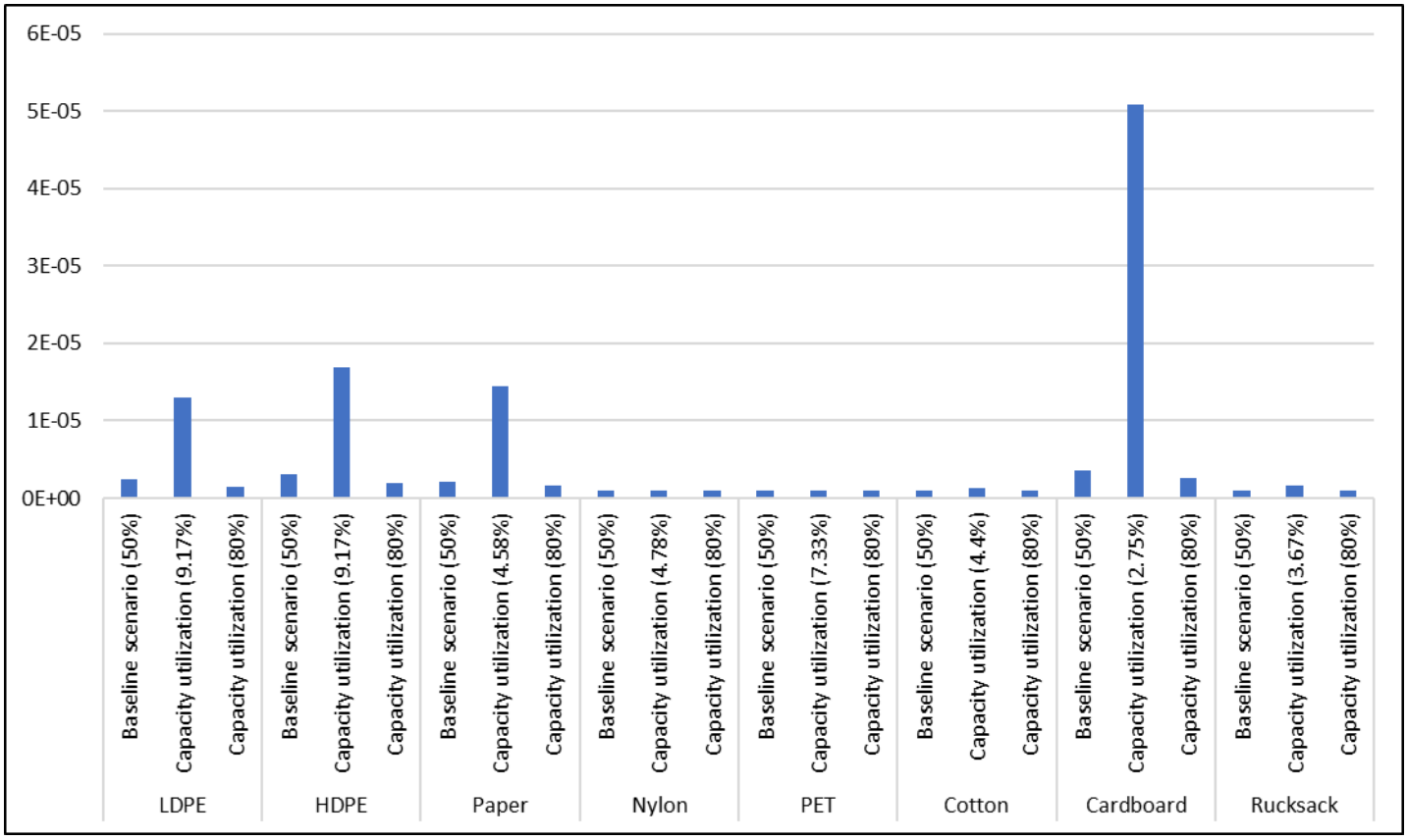


Figure 30: Potential human health damage (DALY) results per functional unit for the different scenarios analysed for volume utilisation.

The volume utilisation assumption affects the results, particularly for the single use carriers. If the volume utilised increases, the number of bags required reduces, and the converse is also true. Thus, for a small shopping event where only 1.1 litres of the carrier volume (the lowest capacity scenarios in Figure 30) is used, then the number of carriers to fulfil the functional unit increases. The same main conclusions hold for the other damage categories shown below.

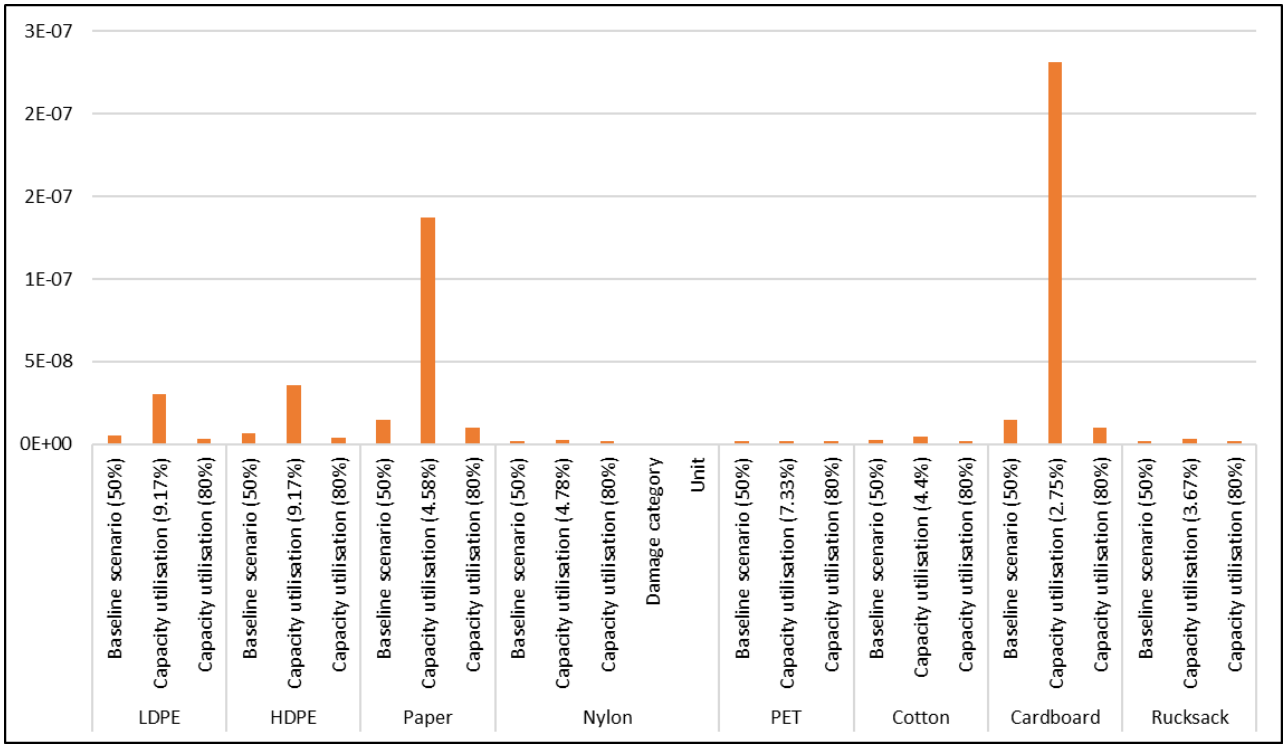


Figure 31: Potential ecosystems damage (species.yr) results per functional unit for the different scenarios analysed for volume utilisation.

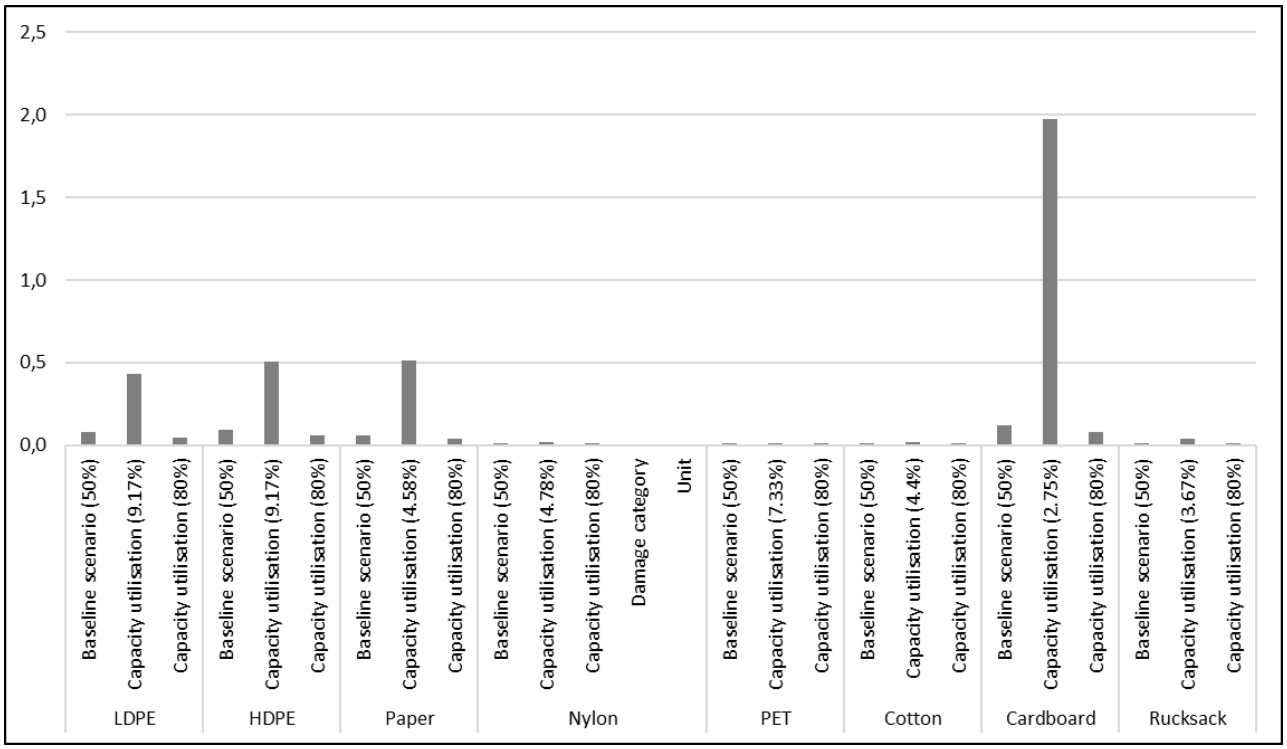


Figure 32: Potential resources damage (USD 2013) results per functional unit for the different scenarios analysed for volume utilisation.

6.5.4 Transport distances

The table below shows the results for changes in transport distances shown in Table 28 for the three endpoint potential damage categories.

Table 32. Transport distances sensitivity analysis results

		LDPE				HDPE			
Damage category	**Unit**	**Baseline scenario**	**Distance from carrier production to use (500 km)**	**Distance consumer to incineration (500 km)**	**Distance consumer to recycling (1 200 km)**	**Baseline scenario**	**Distance carrier production to use (160 km)**	**Distance consumer to incineration (500 km)**	**Distance consumer to recycling (1 200 km)**
Human health	DALY	2.40E-06	2.43E-06	2.43E-06	2.40E-06	3.09E-06	2.81E-06	3.13E-06	3.10E-06
Ecosystems	species.yr	5.61E-09	5.69E-09	5.70E-09	5.62E-09	6.58E-09	6.05E-09	6.66E-09	6.59E-09
Resources	USD2013	7.86E-02	8.11E-02	8.13E-02	7.89E-02	9.29E-02	8.59E-02	9.55E-02	9.31E-02

The baseline scenario is the cut-off modelling scenario, as shown in Chapter 6.2. The changes in results for these damage categories are shown below.

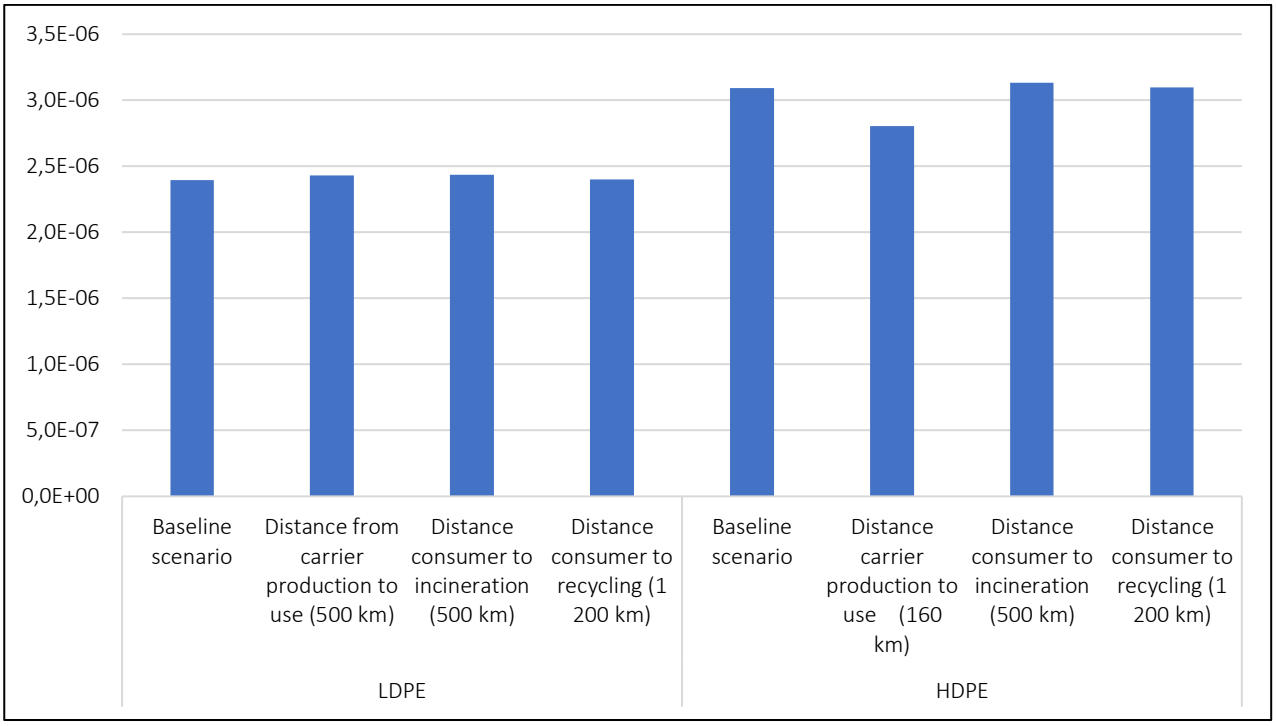


Figure 33: Potential human health damage (DALY) results per functional unit for the different transport distance scenarios analysed.

Figure 33 shows how sensitive the results are to different transport distances. For the LDPE carrier, if all of the distances were set to their longest extreme, the results would be the baseline result plus the sum of the differences between each of the scenarios and the baseline, shown in the table above. For the HDPE case long distance transport by ship from Southeast Asia is replaced by 160 km by truck which gives a reduction in emissions. If the HDPE bag was produced closer to Norway (e.g., in Sweden) and the other distances were the longer distances, the result for the HDPE carrier system is shown in Table 33 below. The increases in truck transport for HDPE do not outweigh the reduction transport emissions from the long distance freight ship. Thus, production of carriers closer to Norway would be beneficial for the environmental impacts of carriers.

Table 33. Potential human health damage (DALY) results if all changes in transport distances are implemented in the model

		LDPE		HDPE	
Damage category	**Unit**	**Baseline**	**With all long transport distances** (all transport sensitivity scenarios at once)	**Baseline HDPE**	**HDPE with 160 km from production to use and long transport distances from consumer to incineration & recycling** (all transport sensitivity scenarios at once).
Human health	DALY	2.40E-06	2.47E-06	3.09E-06	2.85E-06
Ecosystems	species.yr	5.61E-09	5.79E-09	6.58E-09	6.14E-09
Resources	USD2013	7.86E-02	8.39E-02	9.29E-02	8.88E-02

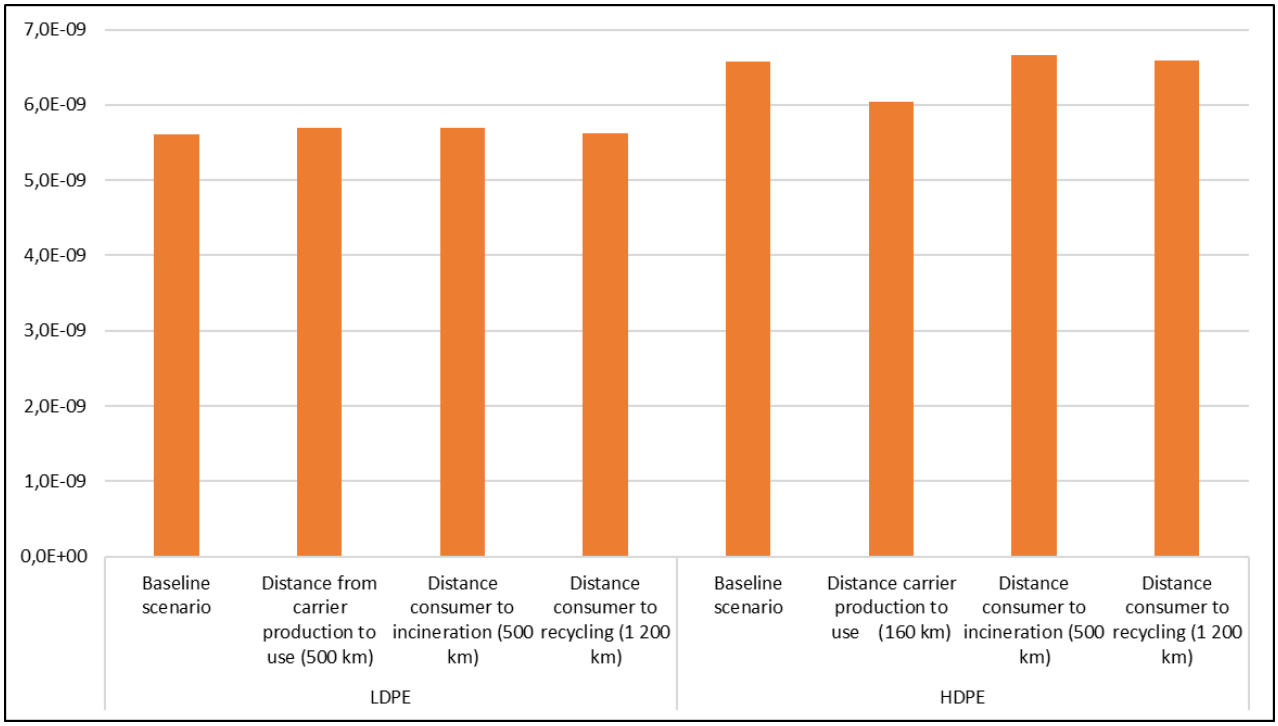


Figure 34: Potential ecosystems damage (species.yr) results per functional unit for the different transport distance scenarios analysed.

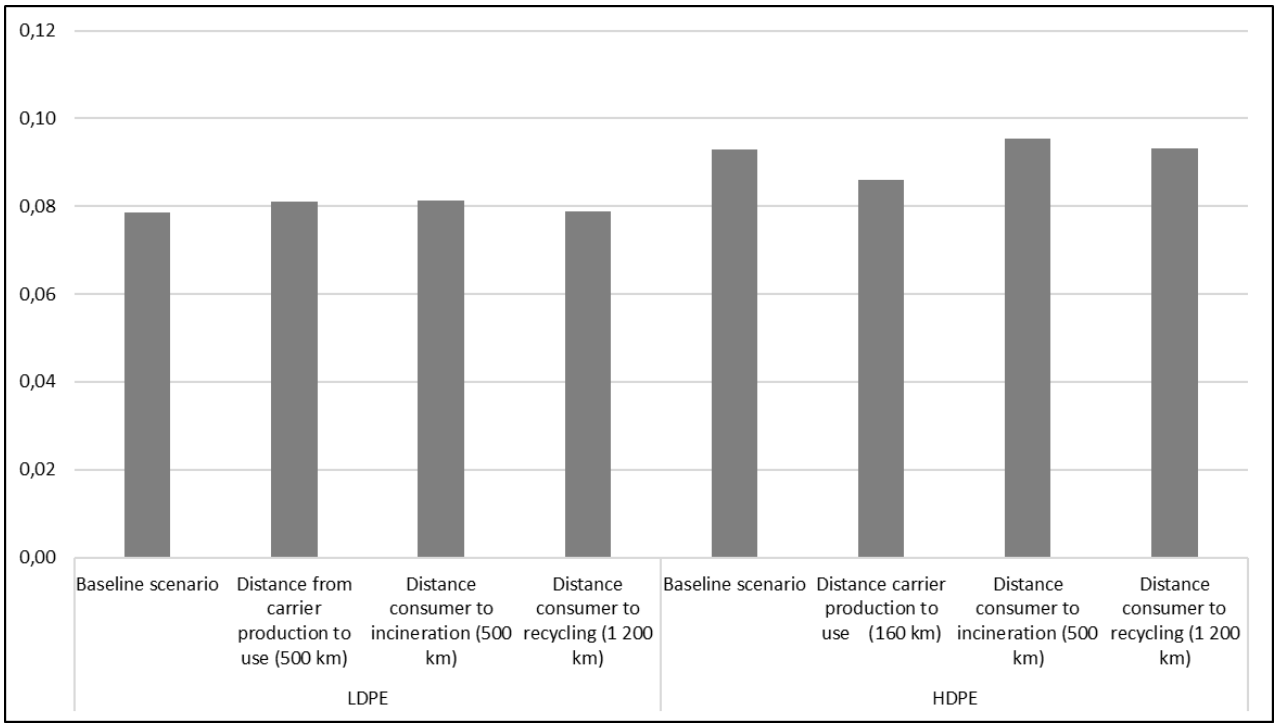


Figure 35: Potential resources damage (USD 2013) results per functional unit for the different transport distance scenarios analysed.

6.5.5 Cotton bag volume

As cotton bags can vary greatly in size and volume, the table below shows the results for the changes in volume for the cotton bag shown in Table 28.

Table 34. Cotton bag volume sensitivity analysis results

		Cotton		
Damage category	**Unit**	**Baseline scenario (25 litres)**	**Bag volume 12 litres**	**Bag volume 30 litres**
Human health	DALY	9.20E-07	9.52E-07	9.15E-07
Ecosystems	species.yr	2.33E-09	2.59E-09	2.29E-09
Resources	USD2013	1.37E-02	1.41E-02	1.36E-02

The baseline scenario is the cut-off modelling scenario, as shown in Chapter 6.2. The changes in results for the damage categories are shown in the figures below.

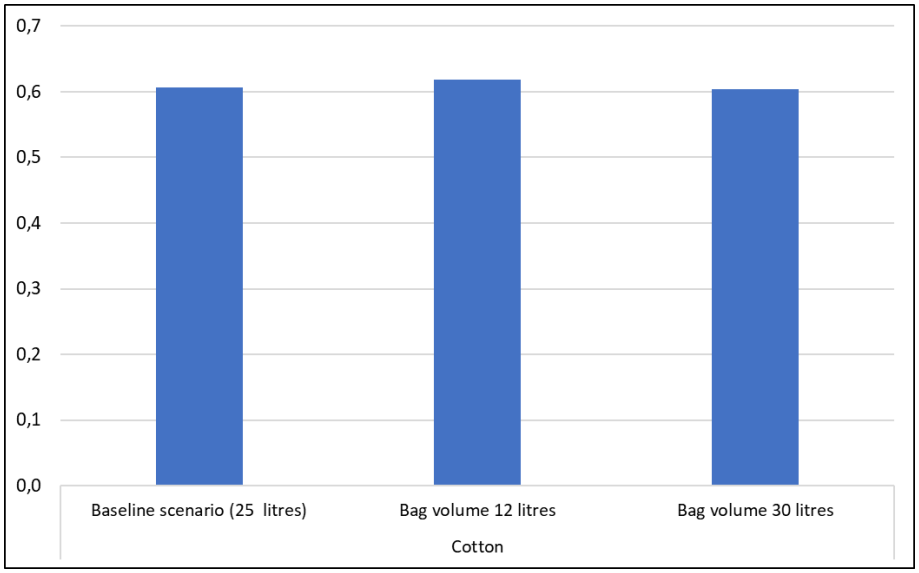


Figure 36: Potential human health damage (DALY) results per functional unit for the different cotton bag volumes analysed.

Figure 36 shows how the results are not sensitive to different bag volumes. The cotton bag system is dominated by the bin liners needed. Thus, changes in amounts of material due to changes in number of bags needed per functional unit do not have a significant impact on the overall results for the cotton bag system. The same main conclusions hold for the other damage categories shown below.

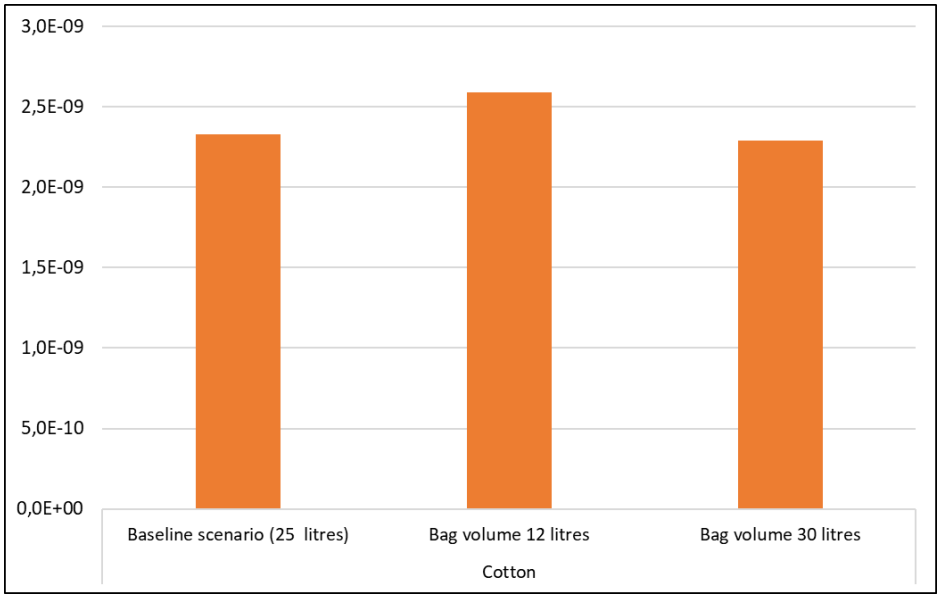


Figure 37: Potential ecosystems damage (species.yr) results per functional unit for the different cotton bag volumes analysed.

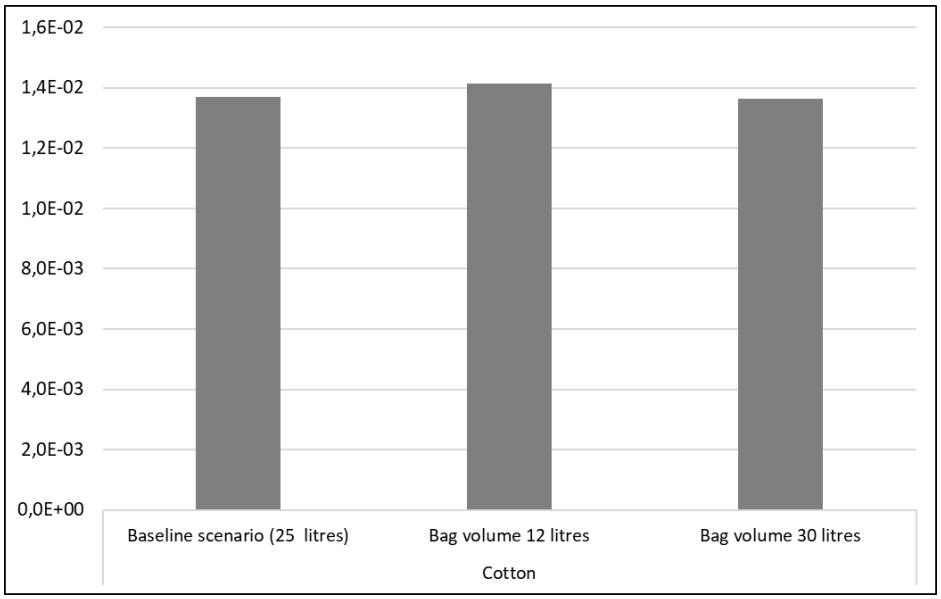


Figure 38: Potential resources damage (USD 2013) results per functional unit for the different cotton bag volumes analysed.

6.5.6 Electricity mixes

The baseline cases were performed with bag production process electricity data chosen for the specific country where the activity takes place (based on data from Norwegian retailers and their suppliers). The country names have been replaced by regional names, where confidentiality is risked if the specific country is named. The critical review panel has received the information about the specific production sites.

Some countries, such as Norway, have an electricity production mix from largely renewable sources. The choice of electricity mix will affect the life cycle impact assessment results for life cycles where electricity use is significant. Plastic bag extrusion uses electricity, as does textile production and many other processes in the carrier systems life cycles.

Table 35 shows the results where the specific production mix for the country where the bag production takes place is replaced with a European average electricity mix (labelled “European electricity” below).

Table 35. Electricity mix sensitivity analysis results

		LDPE		HDPE		Paper		Nylon	
Damage category	**Unit**	**Baseline scenario (Swedish electricity)**	**bag prod. using European electricity**	**Baseline scenario (Southeast Asian electricity)**	**bag prod. using European electricity**	**Baseline scenario (Swedish electricity)**	**bag prod. using European electricity**	**Baseline scenario (East Asian electricity)**	**bag prod. using European electricity**
Human health	DALY	2.40E-06	2.61E-06	3.09E-06	2.61E-06	2.14E-06	2.18E-06	9.01E-07	9.01E-07
Ecosystems	species.yr	5.61E-09	6.02E-09	6.58E-09	6.02E-09	1.45E-08	1.46E-08	2.11E-09	2.11E-09
Resources	USD2013	7.86E-02	8.32E-02	9.29E-02	8.32E-02	5.93E-02	6.01E-02	1.39E-02	1.39E-02

		PET		Cotton		Cardboard		Rucksack	
Damage category	**Unit**	**Baseline scenario (East Asian electricity)**	**bag prod. using European electricity**	**Baseline scenario (South Asian electricity)**	**bag prod. using European electricity**	**Baseline scenario (Norwegian electricity)**	**bag prod. using European electricity**	**Baseline scenario (Southeast Asian electricity)**	**bag prod. using European electricity**
Human health	DALY	9.01E-07	9.01E-07	9.20E-07	9.08E-07	3.63E-06	3.72E-06	9.45E-07	9.44E-07
Ecosystems	species.yr	2.11E-09	2.11E-09	2.33E-09	2.31E-09	1.47E-08	1.49E-08	2.20E-09	2.20E-09
Resources	USD2013	1.36E-02	1.36E-02	1.37E-02	1.36E-02	1.21E-01	1.23E-01	1.54E-02	1.54E-02

The changes in results for the damage categories are shown in the figures below.

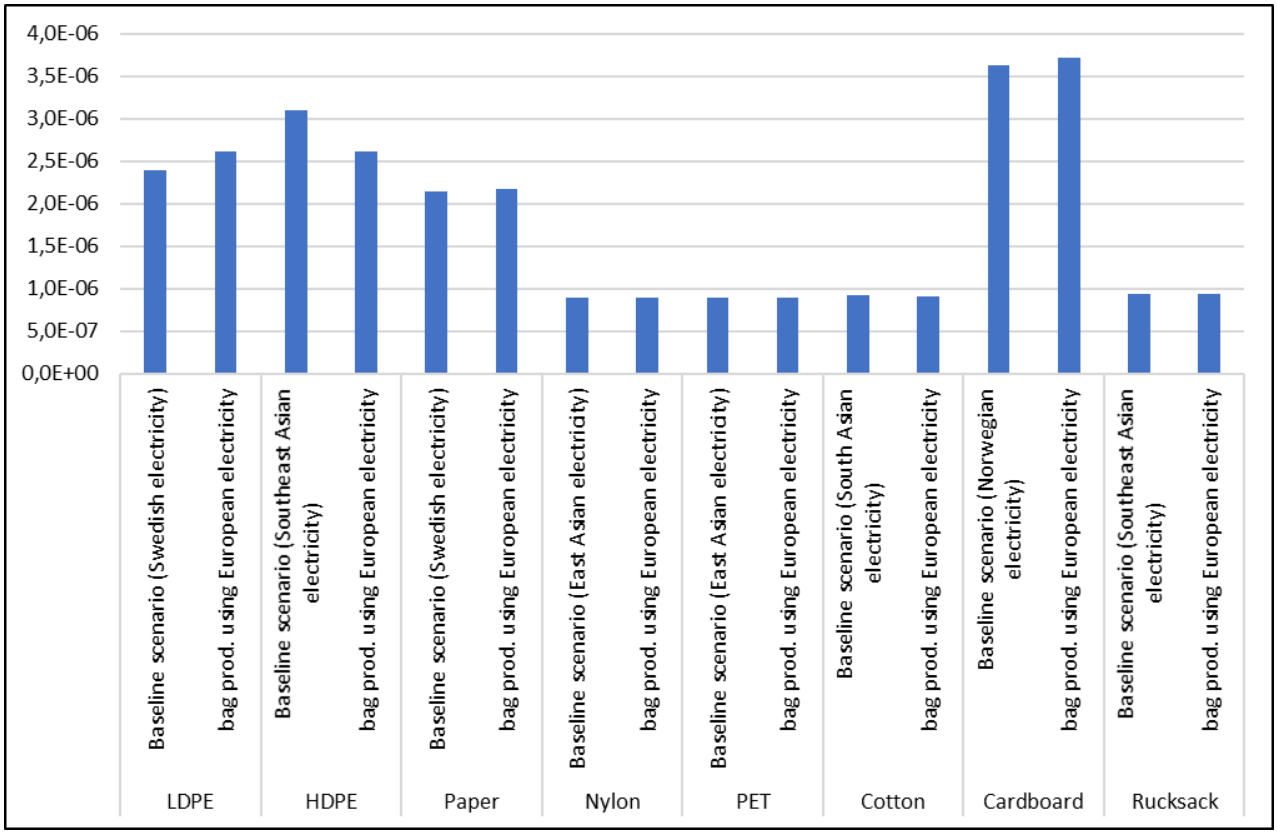


Figure 39: Potential human health damage (DALY) results per functional unit for the different electricity mix scenarios analysed.

Figure 39 shows that the two single use plastic bags (LDPE and HDPE) are comparable when produced with the same electricity mix. The multiuse carriers are still dominated by the bin liner, so the differences are not visible. There is a slight increase in potential damage for the cases where the "cleaner" (Swedish, or Norwegian) electricity mix is changed to the European average. The bin liner is produced with Swedish electricity in all cases. It could be argued that the bin liner production data should be modelled with the same change (Swedish electricity substituted for European electricity). However, the results for the LDPE bag can be used to infer the size of the change in results that this could make. The electricity use data for bin liner and bag production are provided per kg material and the bin liner weighs less than the LDPE bag. As the bin liner mass is smaller than the LDPE carrier bag, the bin liner result will change no more than the LDPE result changes in Figure 39 if bin liner production were to use European electricity, rather than Swedish.

The same main conclusions hold for the other damage categories shown below.

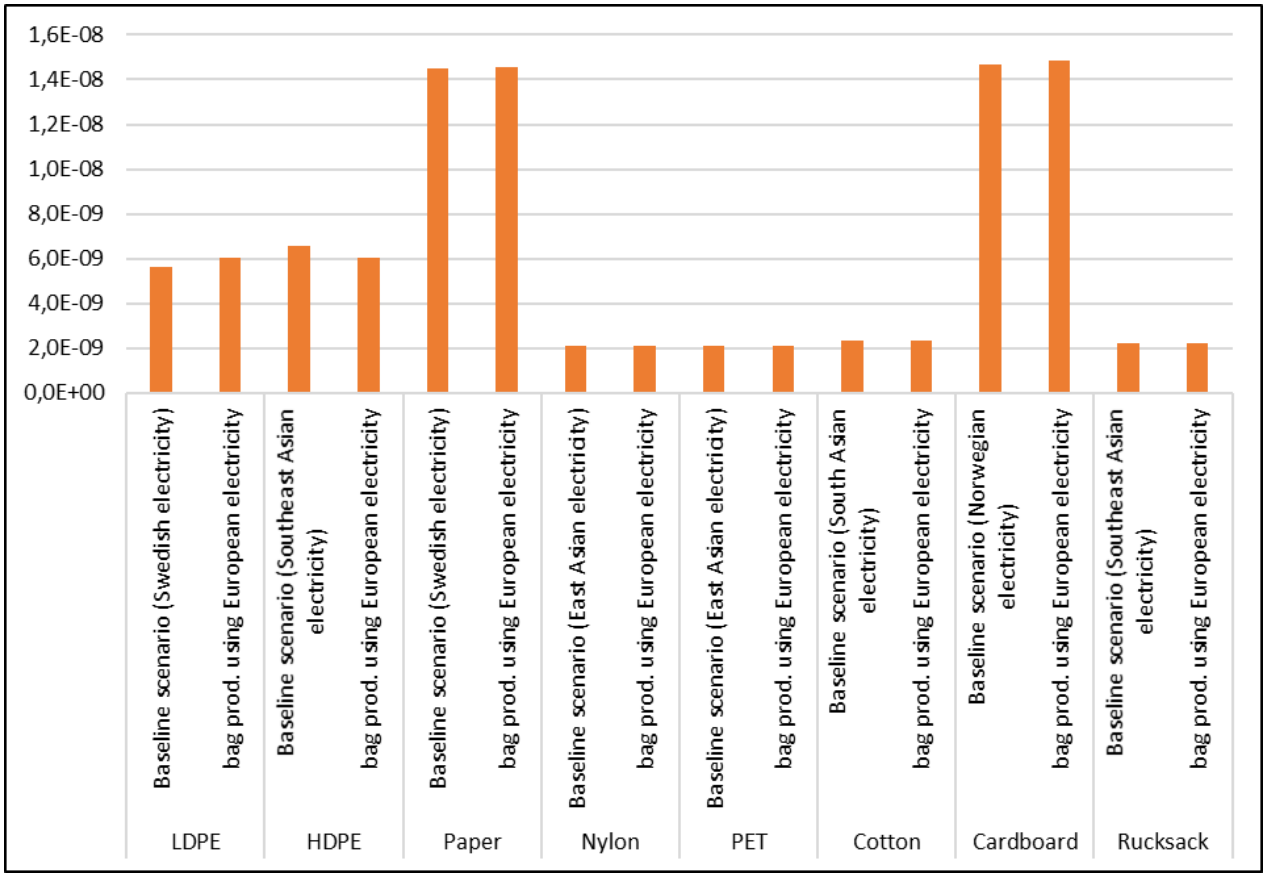


Figure 40: Potential ecosystems damage (species.yr) results per functional unit for the different electricity mix scenarios analysed.

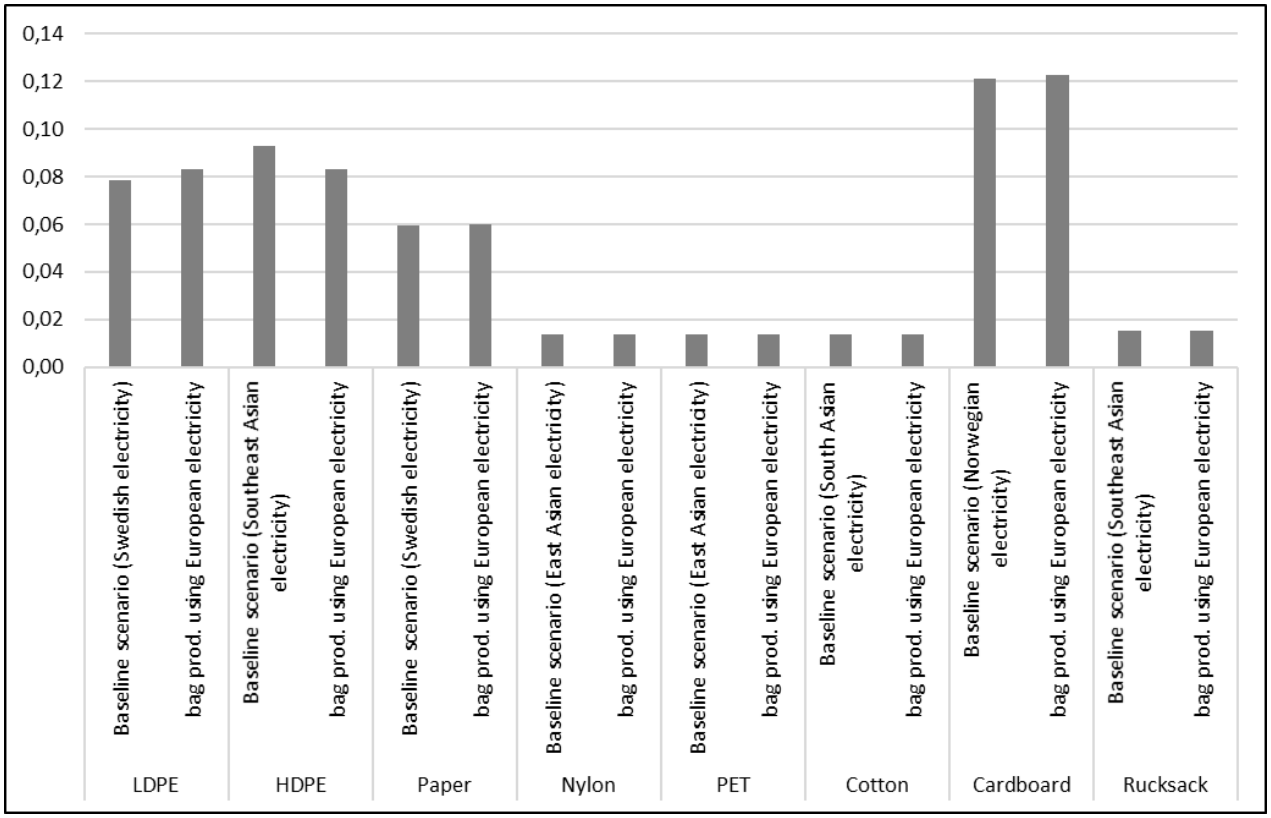


Figure 41: Potential resources damage (USD 2013) results per functional unit for the different electricity mix scenarios analysed.

7 Discussion

Goal and scope

This study compares carrier solutions available and suitable for the Norwegian market. The goal, scope and data used are relevant for Norway, Norwegian retailers and consumers. This means that the results and conclusions are relevant for this and not necessarily applicable to other regions.

The functional unit reflects an amount of shopping for a Norwegian consumer for one month. The basis for this is discussed in detail in Chapter 3.1. The inclusion of the bin liner function is something that makes this study different from many other studies. Norwegian consumers use their single use plastic shopping bags as bin liners. This is an important part of their function for Norwegian consumers and was therefore included in this study. Single use plastic carrier bags are made of thicker material than bin liners sold by retailers for the same purpose.

Transport packaging for the different carrier solutions from their production site to the store was not included in this study, as Edwards et al. (2011) concluded that secondary packaging had a minimal influence on carrier environmental performance. The importance of this exclusion has not been analysed.

Allocation of the rucksack's burdens to shopping is described in Chapter 3.3. If the rucksack has been purchased for other purposes, not for the purpose of shopping, it could be relevant to allocate the production of the rucksack to shopping in different ways: (a) the rucksack's shopping use is assumed to be incidental next to other uses; (b) it is assumed to be used solely for shopping; (c) an intermediate scenario. In the first case, none of the environmental burdens of manufacture and disposal are allocated to shopping events. In the second case, all of the burdens are connected with shopping. This study uses approach (c) having surveyed respondents 30% of the rucksack use was allocated to shopping (70% to other functions). If approach (a) had been chosen the rucksack could be considered as having zero impacts for the shopping function, leaving only the bin-liner impacts. Conversely if the rucksack was purchased for shopping and considered as used only for shopping (approach b), then the rucksack burdens allocated to the shopping function would be higher than shown in this study. This can be quantified for midpoint based on the values shown in Table 19 (midpoint results) and Table 20 (bin liner midpoint results) for the cut-off modelling approach and Table 25 and Table 26 for the net scrap approach. If the bin liner results are subtracted from the midpoint results and a scale up factor of 1/0.3 is used. This will allocate the whole of the rucksack to the shopping function, rather than the 30% chosen. Similarly, if the rucksack is assumed to have its main function elsewhere and not be allocated to shopping at all, then the bin liner results are those that are relevant for the rucksack system (no additional carrier impacts).

The consumer survey was an important part of the goal and scope definition and testing of assumptions (see Appendix 1), as well as the survey of environmentally conscious consumers (see appendices 2 and 3) for multiuse bags and rucksacks. It would be preferable for such surveys were performed in a more robust manner, with larger sampling events (observations of shopping behaviour) in a broader set of locations over time. A form of survey for multiuse carriers and rucksacks expanded to a much larger number of respondents would be more robust and shed more light on how consumers use carriers in practice. Involving consumer survey expertise would also be beneficial. NORSUS asked about recent shopping events and what was done, rather than intended behaviour, during the survey of environmentally conscious consumers and the shopping behaviour survey in shops was based on real observations. This was done to avoid problems with the intention-action gap (Carrington et al. 2010).

The data quality assessments shown in the tables in Chapter 5 and Appendix 4 show data of a reasonable quality, given the resources available and the complexity of the study. It is comparable to other studies and better than some. Data quality across the different carrier systems is similar. The data set that stands out as not adequate is for paper recycling. This is used in the sensitivity analysis about the recycled content of paper bags. The investigation of these results and the reasons for them, alongside the data quality assessment, shows that the results from the comparison for increased recycling rate of paper cannot be relied upon. Data quality and specificity could be improved in general by gathering more specific data for the processes involved in the life cycles of the carriers studied.

The Study provides results for two modelling approaches: cut-off and system expansion (called net scrap in this study). These approaches for modelling recycling and energy recovery are both commonly used approaches for modelling recycling and energy recovery in LCA. The net scrap approach (see Chapter 3.5.3) means in practice that the recycled content of the carrier has little or no impact on the results; it is only the recycling rate at end of life that matters. The rates of material recycling and replacement rates for new materials are given in the tables in Chapter 5. The higher the recycling rate and replacement rate of new materials, the larger the credits assigned for this will be. Thus, the environmental profile for the carrier system will be positively impacts by high recycling and replacement rates at end of life. The same holds true for incineration efficiency and energy replacement where avoided energy is credited to the carrier system. It is therefore important that these factors are chosen with care.

Uncertainty

Zampori et al. (2016) describe uncertainty in LCA. Uncertainty can be introduced in the results of a life cycle inventory analysis due to the cumulative effects of model imprecision, input uncertainty and data variability. The PEF Guide (EC 2012) identifies two key sources of uncertainty in PEF studies: 1. Stochastic uncertainties for the inventory data (i.e., statistical descriptions of variance around a mean/average. For normally distributed data, this variance is typically described in terms of an average and standard deviation). 2. Choice-related uncertainties (i.e. from methodological choices including modelling principles, system boundaries, allocation choices, choice of impact assessment methods and other assumptions related to time, technology, geographical aspects, etc.). The latter can be explored using scenario model assessments. The type of modelling has been explored in this report, showing two possible modelling choices (cut-off and net scrap). Sensitivity analyses (showing how sensitive the results are to a change in an input parameter) are also included in this study. However, the first, stochastic assessment of uncertainties in inventory data has not been performed. In order for this to be done, uncertainty should be assessed at inventory level (e.g. standard deviation associated to a data point) and at characterization level (e.g. linked to characterization factors for both the midpoint and endpoint modelling). If stochastic uncertainty had been included, the results could be interpreted with a higher level of certainty about the differences between the systems and whether these are likely to be within the uncertainty ranges. This could be the subject of further work.

LCIA results

This report presents life cycle impact assessment results per functional unit. The impact assessment methods used calculate midpoint impacts (see Chapter 16) and endpoints, reflecting areas of protection, i.e. human health (disability adjusted life years, DALY), species loss (species.yr) and resources (USD2013). The choice of impact categories shown in figures in the results chapter is the result of the regression analysis described, not any form of significant importance of these impact categories in relation to the others.

The contribution analysis in the results chapter is only valid for the given impact category, and the regression analysis for correlation between results is based on the total impacts and not the contributions. Thus it can be argued that the results chapter can miss some contributions that would be different, or of interest to the reader.

The midpoint impact categories that contribute the most to potential endpoint damages are global warming (human health and terrestrial ecosystems), fine particulate matter formation (human health) and fossil resource scarcity. For bio-based materials (e.g. paper, cotton and cardboard) land use is important for potential damage to ecosystems. This holds for both the cut-off and net scrap approaches to the system model. Using the net scrap approach means that potential ozone formation and land use are more significant contributors to ecosystem damage for some systems.

For water emissions to cause marine eutrophication, they must reach the sea and for freshwater eutrophication and freshwater ecotoxicity, water emissions are not likely to have an impact on freshwater systems if the factories are located by the sea. The impacts shown in the results are potential impacts and thus the emissions from the systems are compared on the same basis. If the locations of the production sites were disclosed the emissions could be ascribed specific freshwater and seawater compartments to make them more site specific.

For both the cut-off and net scrap modelling approaches, the ***multiuse carriers are preferable*** to the single use carriers, due to the multiuse carrier impacts for production processes being shared across the number of uses for the given carrier. A high number of uses, means that for many impacts the multiuse carrier itself is almost not visible in the results figure. For the multiuse carriers it is the bin liner impacts that dominate the overall results. The grocery retailers do not sell a significant number of multiuse carrier bags or rucksacks, and given that these should definitely not be single use products, this is a good thing. Consumers should be encouraged to use a carrier that they already own instead of buying a new one. However, in situations where a multiuse carrier (already owned) is not available the retailer wishes to know which single use carrier is preferable. For cut-off modelling the endpoint results show the following:

- For potential human health damage: the paper bag, but the plastic single use bag made in Scandanavia is not likely to be significantly worse.
- For potential ecosystem damage (excl. litter impacts): the single use plastic carriers.
- For resources: the paper bag.

For net scrap modelling the endpoint results show the following:

- For potential human health damage: the plastic single use bag made in Scandinavia, but if made far away, then the paper bag is better.
- For potential ecosystem damage (excl. litter impacts): the single use plastic carrier made in Scandinavia.
- For resources: the paper bag, but the single use plastic carrier made in Scandinavia is not likely to be significantly different.

Where ***littering*** is an aspect of concern, the bio-based carrier alternatives (paper, cotton and cardboard) are preferable, as well as the rucksack, which is extremely unlikely to become litter. If the littering problem is weighted strongly for the decision-maker, the single use option that is environmentally preferable is the paper bag.

The cardboard box is the single use carrier with the worst environmental performance for the three potential endpoint damage results and most midpoint results. The net scrap approach modelling is a little more favourable for the cardboard box carrier, where potential ecosystems damage is better than, but quite close to the result for the paper bag. It is also not the worst performer for a few more midpoint categories (see Table 25).

The cardboard box case included in this study is for a box produced for the purpose of transporting goods from the store to the home. However, an alternative scenario that has not been analysed in this study would be if cardboard boxes used for transporting groceries to the store (before they are put on the shelves) were used by consumers to transport their shopping to their home. It is very likely that this would be environmentally beneficial to purchasing a carrier. Recycling rates for cardboard can be higher for businesses than residential homes, but the recycling rates for paper and cardboard from Norwegian homes are high, so the difference in recycling rates would not affect this conclusion.

The results shows that the industry perception that the use of single use bags as bin liners is environmentally beneficial is only partially correct. If the alternative is to dispose of the bags in the waste management system without achieving this second function, bin liners would have to be purchased by the consumer. The inclusion of bin liners in the functional unit has a significant impact on the results for all of the multiuse carriers, independent of the modelling approach. The bin liner is light weight, lighter than the single use plastic carriers (LDPE and HDPE). This means that a plastic carrier bag is over-designed (has too much material, or is of too good quality) to be used as a bin liner. If a multiuse carrier is successfully used by the consumer (remembered when they go to the shop), it is environmentally preferable that they purchase a lightweight, adequately sized bin liners for their household waste bin, rather than purchase carrier bags because they have run out of bin liners.

The energy carriers are important for all of the product systems. Using energy sources with lower environmental impacts, e.g. global warming potential and fine particulate matter formation for both materials production and transport would be beneficial for the environmental impacts of the carriers, whichever are chosen. This is something the purchasers of the carriers may be able to influence.

The results were used to calculate the number of times multiuse carriers would need to be used in order to be competitive with the LDPE bag system for all endpoint and midpoint results (see chapters 6.1.4 , 6.2.8 6.3.4 6.4.7). For the cut-off endpoint results the multiuse carriers made from nylon and PET needed to be used eight times to be better than the single use LDPE reference bag. For the cotton bag and rucksack this rose to sixty-eight and ninety-six times respectively. The endpoint results represent an aggregation of the midpoint results, which should reflect their environmental importance for the damage endpoints. In order to focus the discussion, the midpoint results are not repeated here, but can be found in Chapter 6.2.4 for the cut-off modelling approach. For the net scrap endpoint results the multiuse carriers made from nylon, PET and cotton needed to be used ten, eight and sixty-one times respectively to be better than the single use LDPE reference bag. The single use cardboard box would need to be used two-three times to be equivalent to the single use LDPE bag. For the rucksack this number of uses was one hundred and thirteen. As for the cut-off modelling approach, the midpoint results are not repeated here, but can be found in Chapter 6.4.7 .

Whichever modelling approach is used, the number of uses calculated for multiuse carriers was below the number defined for the multiuse carrier systems in the reference case. The basis for the number of uses and lifetime values shown in Table 3 are provided in Appendices 2 and 3 for the multiuse carriers and rucksack. When considering the number of uses that are realistic, there are also outliers, such as multiuse bags where consumers are given these and do not use them. There is also a case for considering that a rucksack produced for use by the consumer should not be allocated to shopping activities at all (as previously discussed). None of the respondents to the survey about rucksacks had purchased the rucksack in order to use it for shopping, shopping was an additional use function for their rucksack, not the main or intended use. Some respondents had been given the rucksack as a gift, or as a prize. The definition of the number of lifetime uses is a sensitive parameter, as shown in the sensitivity analysis Chapter 6.5.2

Sensitivity analysis

The sensitivity analysis tested parameters perceived to be of importance to the results and relative rankings of the carrying solutions. The proportion of recycled materials used to make the carrier, number of uses, carrier volume utilisation, transport distances, cotton bag volume, and changing electricity mixes for carrier production electricity from location specific mixes to the average European electricity production mix were all tested.

If one discounts the paper bag results from this discussion (for reasons of data quality previously explained), the general conclusion taken from the results of the sensitivity analysis about the proportion of recycled materials used to make the carrier is that an increased proportion of recycled material is environmentally preferable.

The sensitivity analysis about the number of uses shows the result that an increase in the number of uses reduces the environmental impacts of the given carrier, even where an increased need for bin liners is included. A small number of uses (i.e. two) were analysed for a multiuse carrier. The environmental performance becomes worse than that of a single use plastic carrier with so few uses. If carriers that are intended to be multiuse carriers are used only a few times, then this is extremely inefficient use of resources. Giving multiuse carriers away or selling them at a low price could cause an excess production of multiuse carriers and increase the environmental impacts of the system. A multiuse carrier used many times is the environmentally preferable option. It is important to identify and assess measures that stimulate a transition to multiuse carriers, but also avoid excess production of these.

It is perhaps of use to also reflect on the observation that several shopping events carried out by Norwegian consumer are without bags (Appendix 1). These are small purchases, with only a few items at a time. For this type of small purchasing event the retailer could encourage the consumer not to use a carrier as the environmentally preferable option. However, please note that this recommendation is valid when there is no significant risk of dropping and breaking the items bought. The environmental impacts of producing a bag are typically small compared to the impacts of producing the groceries.

The volume utilisation of the carrier can be expressed as the difference between a consumer doing their shopping as a series of very many small shopping events, rather than larger shopping events, where they buy groceries for several days at a time. As previously described, the survey (Appendix 1) showed that there are many small shopping events, often without a carrier employed. The sensitivity analysis shows that the single use carriers have the worst environmental performance when purchased for small shopping events. Unsurprisingly increased volume utilisation led to better environmental performance.

Changes in transport distances within Norway did not have much impact on the results for the LDPE carrier. Reducing long distance ship transport for the HDPE carrier and replacing this with a more local lorry transport, reduced the potential damage arising from the HDPE single use carrier system. This reduces fossil energy carrier use and transfers the energy used to a slightly cleaner form of fossil energy than shipping fuel.

The sensitivity analysis for cotton bag volume does not show significant changes, as the bin liner results dominate. The only damage category that one can discern a small increase is for potential ecosystem damage, when the volume of the bag is approximately halved (to 12 litres from 25).

Changing electricity mixes for carrier production electricity from location specific mixes to the average European electricity production mix was tested for all carriers. The two single use plastic bags (LDPE and HDPE) were comparable when produced with the same electricity mix. The multiuse carriers are still dominated by the bin liner, so the differences are not visible.

Comparison with conclusions from other studies

This part of the discussion takes findings from some of the studies described in Chapter 4 and describes similarities and differences between the relevant conclusions in these studies and the study presented in this report.

Edwards et al. (2011) concluded that the environmental impact of all types of carrier bag is dominated by resource use and production stages. Our results do not concur with this across all types of carriers and modelling approaches. However, we have included more and different carriers in our study, as well as two different modelling approaches. They also conclude that whatever type of bag is used, the key to reducing the impacts is to reuse it as many times as possible and where reuse for shopping is not practicable, other reuse, e.g. to replace bin liners, is beneficial. This is in agreement with our findings. Edwards et al. (2011) found that the reuse of conventional HDPE and other lightweight carrier bags for shopping and/or as bin-liners is pivotal to their environmental performance and reuse as bin liners produces greater benefits than recycling bags. This has informed this study's approach to the importance of including bin liners. However, the best disposal option for bags has not been directly analysed in the current study.

Mattila et al. (2011) analysed climate change only. The results were found to be highly sensitive to the weight and number of bin liners replaced by the secondary use of plastic bags. In scenarios where waste was incinerated, assumptions about replaced energy played a key role in the ranking. They conclude that the shopping bag comparisons should be considered as being conditional on the waste treatment infrastructure analysed. Since the study presented in this report focusses strongly on conditions in Norway and applies data for waste management relevant for Norway, the carrier systems are assessed with similar assumptions for these aspects. Mattila et al.'s findings highlight the importance of obtaining good data for these aspects in the model. Mattila et al. describe the Scandanavian paper industry, with use of a high proportion of wood energy and integrated paper mills as giving better results for paper. The study presented in this report uses European average paper. It is possible that if data for Norwegian, or Scandinavian paper mills were used, that the paper bags could produce more favourable results.

Ahmed et al (2021) focusses on different carriers used in Singapore (given the waste management system that is present). Usage characteristics (reusability, dimensions, carrying capacity) of bags, the production process (raw materials extraction, production processes), and emissions were significant factors contributing to the environmental impacts. The reusable polypropylene non-woven bag (PNB) was found to cause the lowest environmental impacts (50 instances of reuse), followed by single use HDPE plastic bag. The global warming potential was 14, 81, 17 and 16 times higher for HDPE plastic, kraft paper, cotton woven and biodegradable polymer bags, respectively, when compared to PNB. The sensitivity analysis performed in Ahmed et al. (2021) indicated that a minimum of 4 reuses of the PNB was needed to avoid emissions equivalent to the HPB.

Binsella et al. (2018) uses an approach to functional unit that focusses on a single shopping event with a newly purchased carrier. They conclude that after reusing the carrier bag as many times as possible, reusing the carrier bag as a bin liner is better than simply throwing away the bag in the residual waste and it is better than recycling. The finding that re-use as many times as possible before using as a bin liner is in line with circular economy principles and is also true for the findings presented in this study. Whether recycling or incineration is a better waste management approach for the carriers has (as described above) not a focus for this present study. It is sensible to re-use any carrier as many times as possible before it reaches waste treatment. This maximises the function of the product for the same resource input.

Binsella et al. (2018) conclude that LDPE carrier bags provide the over-all lowest environmental impacts for most environmental indicators. There are notable differences in goal and scope of the study Binsella et al.

(2018) study compared to the study presented in this report, including functional unit, number of uses per bag, modelling approach and databases used. The ranking of the other bags studied *ibid.* is (lowest to highest impacts): woven PP carrier bags, non-woven PP bags, unbleached paper, bleached paper, conventional cotton and then organic cotton.

Quantification of the benefits from the reduction in use of SUP bags

The Norwegian Retailer's Environment Fund (Handelens Miljøfond) reports that Norwegian consumers each buy about 12 new plastic bags per month (HMF 2021). This amounts to 782 million plastic bags per year. If the consumers switched from using a single use plastic (SUP) bag like the reference bag to a multiuse carrier for every shopping trip, this would mean savings in environmental impacts that can be calculated using the tables in Chapter 6. Considering global warming potential, a complete change for all consumers would mean savings in global warming emissions from 19 thousand tonnes CO_2 equivalents – 40 thousand tonnes CO_2 equivalents, depending on which multiuse carrier was chosen and which modelling approach. It is unlikely that one could change the shopping habits of every consumer in Norway, but if Handelens Miljøfond manages a 20% reduction in the purchase of SUPs through its work, this is still a large overall saving. If levels like those seen in other countries (e.g. 60% in the England, Guardian 2020), then the environmental benefits would increase proportionately. Ambitious reduction targets would be environmentally beneficial. A similar calculation (if all consumers changed to multiuse carriers) gives a littering reduction of plastic emitted to the marine environment of 37 tonnes per annum. The reduced climate impacts calculated here are from activities across the globe, whereas the littering is a use-phase impact taking place in Norway.

8 Conclusion

This study compares carrier solutions available and suitable for the Norwegian market. The goal, scope and data used are relevant for Norway, Norwegian retailers and consumers. This means that the results and conclusions are relevant for this and not necessarily applicable to other regions.

For both the cut-off and net scrap modelling approaches, the multiuse carriers are preferable to the single use carriers, if they are used many times. This is due to the impacts for production processes for multiuse carriers being shared across the number of uses for the given carrier. Consumers should be encouraged to use a carrier that they already own instead of buying a new one. However, in situations where a multiuse carrier (already owned) is not available the retailer wishes to know which single use carrier is preferable. For cut-off modelling the endpoint results show the following:

- For potential human health damage: the paper bag, but the plastic single use bag made in Scandanavia is not likely to be significantly worse.
- For potential ecosystem damage (excl. litter impacts): the single use plastic carriers.
- For resources: the paper bag.

For net scrap modelling the endpoint results show the following:

- For potential human health damage: the plastic single use bag made in Scandinavia, but if made far away, then the paper bag is better.
- For potential ecosystem damage (excl. litter impacts): the single use plastic carrier made in Scandinavia.
- For resources: the paper bag, but the single use plastic carrier made in Scandinavia is not likely to be significantly different.

Where ***littering*** is an aspect of concern, the bio-based carrier alternatives (paper, cotton and cardboard) are preferable, as well as the rucksack, which is extremely unlikely to become litter. If the littering problem is weighted strongly for the decision-maker, the single use option that is environmentally preferable is the paper bag.

It would be environmentally beneficial for purchasers of carriers to influence their suppliers to use cleaner energy for production processes and transport, as well as use recycled raw materials.

Norwegian consumers use their single use plastic shopping bags as bin liners. This is an important part of their function for Norwegian consumers included in this study. Whichever carrier is chosen, re-use as many times as possible before it reaches waste treatment is important. If it can be used as a bin liner for its last use, this maximises the function of the product for the same resource input. The perception that the use of single use bags as bin liners is environmentally beneficial is however only partially correct. If the alternative is to dispose of the bags in the waste management system without achieving this second function, bin liners would have to be purchased by the consumer. The inclusion of bin liners in the functional unit has a significant impact on the results for all of the multiuse carriers, independent of the modelling approach. The bin liner is light weight, lighter than the single use plastic carriers (LDPE and HDPE). This means that a plastic carrier bag is over-designed (has too much material, or is of too good quality) to be used as a bin liner. If a multiuse carrier is successfully used by the consumer (remembered when they go to the shop), it is environmentally preferable that they purchase a lightweight, adequately sized bin liners for their household waste bin, rather than purchase carrier bags because they have run out of bin liners.

9 Critical Review Summary

This study is a comprehensive environmental assessment of grocery carriers in Norway. A panel of international LCA experts and representatives from the Norwegian retail sector reviewed the study based on the international standards for LCA. Overall, the review panel finds that the methodological choices in the study are scientifically valid. The LCA report complies with most of the many reporting requirements posed by ISO 14044 on a comparative assertion disclosed to the public. It includes an adequate assessment of key uncertainties and a full-bodied discussion of the results.

However, the review panel found several problems in the application of the methods and in the presentation of the study. While the report is transparent in theory, the effective transparency is low, making it difficult to understand how the results are produced. The numerical results should be used with care, because other results would have been obtained with a different set of input data or calculation methods. The study also appears to include a few significant calculation errors.[25]

Most of the conclusions are still sound. The environmentally best option is to use a carrier from home. The ranking order between plastic and paper single-use bags depends on the environmental impacts in focus, on how the life cycle is modelled, and on where the bags are produced. However, the conclusion that multiuse carriers are environmentally preferable to single-use bags is valid only if each multiuse carrier is used many times.

9.1 Rebuttal

The report authors address some of the points made in the critical review in the table below. The points addressed are taken from the critical review in Appendix 7. The reader should note that the text sections included in the table below are only excerpts from the appendix and that the appendix should be read in detail to get the full picture.

Review comment	Author response & action taken
Summary "…The study also appears to include a few significant calculation errors."	Any calculation queries raised during the work have been addressed by the authors, providing the panel with detailed data and calculations to show what lies behind the results. The endpoint results were a new addition in the final version of the report that has been reviewed. There was a calculation error, that the authors are grateful that the panel spotted. The calculation error has been corrected (see the detailed comments below about Table 17, A7.3.10) in this version of the report.
A7.3.3 LCI modelling approaches • Calculations with the cut-off approach seem to go under different names in different parts of the report: for example, main scenarios or reference scenarios at the beginning of Section 3.5, baseline scenarios in	This is confusing for the reader. The authors have tried to find these texts and make them consistently refer to "base case scenarios". However, the term "baseline" is still to be found in the sensitivity analysis chapter (6.5). Text at

[25] The report authors have added this footnote: the calculation error in Table 17 has been found and corrected in this version of the report. Please see the detailed comments in Chapter 9.1 (Rebuttal).

Section 3.7.2 and Chapter 5, and Cut-off in Sections 6.1-6.2.	the start of Chapter 6.5 informs the reader that the baseline values shown are based on the cut-off modelling approach and the "base case".
A7.3.3 LCI modelling approaches • Calculations with the net-scrap approach also seem to go under different names in the report: for example, system expansion in the Summary, scenario modelling, End of Life substitution and EoL at the beginning of Section 3.5, end-of-life scenario in Chapter 5, and Net scrap approach in Section 6.1.	This is confusing for the reader. The authors have revised this version of the report to ensure that the net scrap approach is always present, even if listing, or describing other terminology that can be used. In Chapter 5 "end-of-life" is consistently replaced with "net scrap approach" in all tables.
A7.3.4 Transparency The report includes sources with references to the input data, the characterization factors, and the conversion factors used in the calculation of end-point impacts. It also presents the methods used in the calculations and outlines the calculation procedure. This makes the study highly transparent in theory. It also meets most of the many requirements posed by ISO 14044 on a comparative assertion disclosed to the public. An important exception is that input data on important processes, such as material production, are not included in the report. These data are from the Ecoinvent database, which prohibits publication of the data. The absence of important input data makes it difficult for the reviewers to assess them. However, the report includes an assessment of the data quality made by the LCA team, as required by ISO 14044. The report includes separate presentation of the mid-point LCIA results for bin liners, which dominates the total results for the multi-use carriers. This contributes to explaining the results. However, other important partial results, such as the LCI results, are missing in the report. Characterization factors and end-pointy conversion factors are also absent. This makes it difficult to assess the importance of input data with low quality, to check the calculations, and to assess whether the results are reasonable. The effective transparency of the study is low.	The authors have striven to meet the requirements in ISO 14044 for a comparative assertion disclosed to the public. The Ecoinvent license agreement prevents publication of inventory data from their database. According to ISO 14044 critical review requirements, the review panel is not expected to perform detailed checks of calculations, but ensure that methods used are consistent with ISO 14044 and are scientifically and technically valid. The data should be appropriate and reasonable; the interpretations reflect limitations and the report be transparent and consistent. The authors have supported all queries from the panel about specific calculations and responded with detailed data and calculation steps during the review period. All relevant foreground system data, gathered and derived for the purpose of this specific study, are presented in Chapter 5 in the report. The different carrier systems are described and the quality of the data is assessed. All relevant background system data applied in this study are presented in Appendix 4 in the report. The LCIA methods applied are specified with references in Chapter 3.7 in the report. Since detailed presentations of foreground as well as background system data are provided in the report and characterisation factors as well as end-point conversion factors are available from the references cited in the report, the calculations can be checked. The authors have striven to provide transparency related to the type of data that have been used and, when not possible to present these values directly in the report e.g. due to intellectual property rights, made sure to cite references to enable readers as well as the reviewers to be able to check the calculations.
A7.3.5 Utilization of carrier capacity	

The baseline calculations assume that all carriers are filled to 50%. This contrasts with the results from the observational investigation of consumer behavior ... It is unclear how the results from the observational study affected the assumptions on the utilization of the carrier capacity. The investigation included a small number of observations only for paper bags, but if it is correct that the average utilization of a 24 liter paper bag is 38% and the utilization of a 12 liter plastic bag is 76%, the two carriers on average hold the same amount of groceries. This indicates that they should be compared based on the results for small shopping events, where the size of the bag does not matter. The conclusions from such a comparison would be more positive for the plastic bag and less for the paper bag.	The base case scenario had to be established before the observational data was available for the study. This base case was agreed in the goal and scope phase of the study, informed by a literature search and the first critical panel meeting. The filling rate is an important assumption and if different filling rates were to be used for different carrying solutions then this would affect the results (see sensitivity analysis Chapter 6.5.1). As noted in the review report, there were only a few observations for some types of carriers. A more comprehensive and robust observational study would be needed to be certain of using observed filling rates as the basis for comparison.
A7.3.6 The number of uses "… The conclusions largely build on calculations that assume single-use bags to be used 1 time only, and durable carriers to be used 1000 times or more. The latter is a much higher number than in previous studies. Contrary to what is common in other studies, the high number is not a pure assumption but an estimate based on a survey (Appendix 3)…"	In Chapter 3.3 many of these issues are discussed. The range for multiuse carriers in the survey was from 39 – 7800 for rucksacks and 156 – 4680 for bags for life (cotton or plastic). The baseline scenario assumptions of 1000 uses was found to be reasonable, both though the survey conducted in this study (see Appendix 2 and 3) and other literature sources (e.g., in Briedis et al. 2019 the number of uses was 1414 for a multiuse plastic bag with a lifetime of 10 years).
A7.3.7 Cotton production "… In conclusion, the results on cotton bags appears to be valid for relatively light, rain-fed organic cotton but not for cotton bags in general or for irrigated organic cotton."	During the projects' data gathering phase, there was a lockdown in Norway, making visits to shops to survey where cotton bags were sold not possible. E-mails were sent to several retailers (covering a large proportion of the Norwegian grocery shopping market), asking for information on cotton bags, but none of the retailers approached sold cotton bags in 2021. The authors used a bag purchased in Norway at an interior furnishings retailer. The bag is robust and is used for grocery shopping on a regular basis. The data for this bag was provided by the retailer that originally sold the bag. We agree there are several limitations to this approach. It is also noted that many that sell multiuse bags choose organic cotton, as the consumers interested in multiuse bags are perceived as also interested in organic cotton. Thus, we find that the choice of organic cotton is not unreasonable for this product. The original source for the cotton in the study is relevant also for the rain-fed cotton production region covered by the data set.

	We agree that not all cotton production is covered here, and the global water equivalent results would be different for irrigated cotton production. In Chapter 6.2.4 we write: Water use for the organic cotton data set chosen is low. If a global cotton data set was used for cotton, the water consumption would increase considerable per kilogram cotton used. If …replaced with the global cotton data set, then the result bar for the cotton bag system in Figure 8 would increase to approximately 0.31 m3 world-eq. This would bring it closer to the water use associated with the single use carrier alternatives, and higher than the other multiuse systems. This shows that the choice of «organic cotton bag» as an option has important effect in the results and if a non-organic cotton bag had been chosen it would affect the results." We also note that "Other impacts besides water deprivation would also be affected if the bag is not made from organic cotton."
A7.3.8 Life cycle impact assessment "…a table similar to Table 4 with the endpoint aggregated categories would be good to include in the report." "…Endpoint results are derived from the midpoint results, which constitutes a translation into damage. This information, including the utility of presenting one or the other (or both) is not specified in the report, an issue that could hinder the understanding of the results by unexperienced stakeholders." "An additional point of confusion is the order in which the results are presented: a lot of focus is given to midpoints in Section 3.7, but Chapter 6 presents the results in terms of endpoint first and then midpoints."	This table has now been included and is in Chapter 3.7, Table 6. This also means that any further tables referred to in the review report (Appendix 7 and this chapter) will have a number one lower than in this version of the report (if they are numbered 6 or higher in the review report). Table 6 now makes this clearer to the reader. The authors chose to present endpoint first, as many of those interested in the report are interested in overall results and not detail, thus the style choice was made to present the endpoint (with 3 sets of results figures per scenario) first. Then the midpoint details after. It is purely a presentation style choice.
A7.3.9 Littering "Marine littering is presented as part of the LCIA Section 3.7. However, much of Subsection 3.7.1 is about estimating the quantity of plastic lost (released) to the ocean per FU. This is a physical flow rather than an impact and, hence, fits better in the description of the LCI. Section 3.7.1 should be revised to avoid mixing	There is no chapter in the report presenting the inventory in detail, thus the results for the calculation steps involved remain in the results chapter. The quantity of plastic littered is now clearly labelled as that. The description of the methods and references has been improved in Chapter 3.7.1.

LCI and LCIA concepts, with those linked to LCI moved to another section." The equations below Table 20 (in the results section on littering with the cut-off approach) are, on the other hand, general for LCIA in general. They fit better in Section 3.7. The equation for the characterization factor is over-simplified, since it does not include an exposure factor. References are also missing for these equations. …	The omission of the exposure factor has been corrected and a reference has been added.
The unit for units of marine litter impacts (rather than effects) should be PAF-m3-day as the units of the fate factor (even though value is one) should be in time units, or more correctly in (kg ending up in the compartment)/(kg emitted per day). Litter impacts does not seem to be included in the end-point calculations. A method for end-point assessment of littering has been published. This involves converting PAF to PDF using existing conversion factors.	The units have been corrected to PAF-m3-day and description in the results tables corrected to "marine litter impact". It is true that they are not included in the endpoint calculations. The ReCiPe endpoint method implemented in SimaPro does not include these yet and as the panel advised against trying to include AWARE water consumption into the ReCiPe endpoint model (advising that the two approaches modelled different impact pathways and such connection would not make sense nor improve the Recipe endpoint models), the authors have assumed that the panel wished that the ReCiPe endpoint model should be used as is, without any adaptions to include specific impacts calculated at midpoint for this study. From this panel comment the authors are now aware that this was a misunderstanding on our side.
A7.3.10 Calculations and results "…significant errors have been made in the calculation of the number of times a carrier must be used to be environmentally equivalent to the conventional LDPE bag. For example, Figures 7-9 indicate a higher impact for 1 HDPE bag compared to 1 LDPE bag, but Table 16 states that the HDPE bag must be used only 0.6 times to be equivalent or better in all environmental aspects. All numbers on human-health impacts in Table 16 seem to be approximately a factor 106 too low. The results on ecosystem damage impacts are also too low – most evident for the HDPE bag, the paper bag and the cardboard box." "When the net-scrap approach is applied on multiuse carriers, the results are negative for many impacts (see Figures 16-17 and Table 24):	The reviewers are correct. There was a link to the mid-point results for the LDPE bag, rather than the endpoint results in the excel file. This led to significant errors, which have now been corrected. This version of this report contains the correct values in the table that was Table 16, but is now Table 17 in this version of the report. The paragraph discussing these results in the Discussion chapter has also been amended. The results for the bin liner applying the net-scrap approach are net positive for seven of the

• overall ecosystem damage, • overall resource depletion, • stratospheric ozone depletion, • tropospheric ozone formation, • fine particulate formation, • terrestrial acidification, • terrestrial, marine and freshwater ecotoxicity, • human non-carcinogenic toxicity, • land use, and • mineral resource depletion. The negative results are generated because the net-scrap approach indicates less than zero environmental impacts for the bin liners (Table 25). This, in turn, is probably because when bin liners are incinerated together with the household waste, the net-scrap approach gives a credit for the impacts avoided when heat from the incineration displaces Norwegian district heating mix. The negative results indicate that the impacts of the Norwegian district heating mix are worse than the combined impacts of production, transport and combustion of the bin liner. This is reasonable for some impacts: for example, the bin liners might be associated with less land use than the biofuel in the district heating systems. Most of the negative results might instead have other explanations, however: data gaps, inconsistent measurement methods, variability between processes, calculation errors, etc. A deeper analysis is required to find the explanations. Meanwhile, the negative results should not be interpreted as proof that the use of bin liners are good for ecosystems, resource depletion, or the environment in general." … "The units and indicators of water use are inconsistent between Tables 18, 19, 24 and 25. We suggest the indicator be called "water deprivation". The unit should be m^3 world eq." "To be consistent with other figures and avoid misleading the reader to think the transport distance is very important for the ecosystem damage, the y axis in Figure 34 should start at zero and not at 5.0E-09."	in total 18 impact categories assessed. These seven impact categories include, for example, global warming, ionizing radiation, freshwater eutrophication, and fossil resource scarcity. The impact categories with a net negative result stem from the fact that the net-scrap approach gives a credit for the impacts avoided when heat from the incineration displaces Norwegian district heating mix. Note that all negative results range between -0.31 (land use) and -3.7E-07 (stratospheric ozone depletion) and that most negative results are only slightly negative. The data applied for the bin liner and a quality assessment of this data are provided in Chapter 5 and Appendix 4 in the report. The authors have now changed this. This has now been changed.
A7.3.11 Conclusions in the report "…A more accurate conclusion is to say that the results indicate that multiuse carriers are environmentally preferable, if they are used many times (cf. Tables 16, 21-22, 26, and 29)…" "…Clarity is improved if "ecosystem damage" is replaced by "ecosystem damage (excl. litter impacts)"	This modification has been made. This suggestion has been implemented both in the Summary and Conclusions.

in the Summary and possible also in Chapter 8 Conclusions."	
A7.3.12 Recommendations in the report "… • The Discussion includes a suggestion to encourage consumers not to use a carrier when buying a few items only. This recommendation is valid only when there is no significant risk of dropping and breaking the items. The environmental impacts of producing a bag are typically small compared to the impacts of producing the groceries. • The Discussion also indicates that a transition from single-use to multiuse carriers brings large environmental benefits. This is valid only if the transition does not lead to an excess production of multiuse carriers. Giving multiuse carriers away or selling them at a low price might cause such an excess production and increase the environmental impacts of the system. It is important to identify and assess measures that stimulate a transition to multiuse carriers while avoiding excess production of these.	This additional text was added to the recommendation: "However, please note that this recommendation is valid when there is no significant risk of dropping and breaking the items bought. The environmental impacts of producing a bag are typically small compared to the impacts of producing the groceries." The discussion section discussing the number of uses and sensitivity analysis for this states: "If carriers that are intended to be multiuse carriers are used only a few times, then this is extremely inefficient use of resources." We have now added two more sentences: "Giving multiuse carriers away or selling them at a low price could cause an excess production of multiuse carriers and increase the environmental impacts of the system... It is important to identify and assess measures that stimulate a transition to multiuse carriers, but also avoid excess production of these."

10 References

Allacker, K., Mathieux, F., Manfredi, S., Pelletier, N., De Camillis, C., Ardente, F. & Pant, R. 2014. Allocation solutions for secondary material production and end of life recovery: Proposals for product policy initiatives. Resources, Conservation and Recycling, 88, 1-12.

Allacker, K., Mathieux, F., Pennington, D. & Pant, R. 2017. The search for an appropriate end-of-life formula for the purpose of the European Commission Environmental Footprint initiative. The International Journal of Life Cycle Assessment, 22, 1441-1458.

Areepraserta, C., Asingsamanuntb, J., Srisawatb, S., Kaharna, J., Inseemeesaka, B., Phaseea, P., Khaobanga, C., Siwakosita, W., Chiemchaisri, C. (2016) Municipal Plastic Waste Composition Study at Transfer Station of Bangkok and Possibility of its Energy Recovery by Pyrolysis, 3rd International Conference on Energy and Environment Research, ICEER 2016, 7-11 September, Barcelona, Spain, Energy Procedia (107) 222-226, https://www.researchgate.net/publication/314490983_Municipal_Plastic_Waste_Composition_Study_at_Transfer_Station_of_Bangkok_and_Possibility_of_its_Energy_Recovery_by_Pyrolysis/figures?lo=1

Askham C, Hanssen OJ, Gade AL, Christensen P (2012) Strategy Tool Trial for Office Furniture, Int. J. of Life Cycle Assessment 17 (6) 666-677.

Avfall Norge (2020) Høyeste energiutnyttelsesgrad fra avfallsforbrenning i 2019, https://avfallnorge.no/bransjen/nyheter/h%C3%B8yeste-energiutnyttelsesgrad-fra-avfallsforbrenning-i-2019. Accessed 10.11.2021.

AWARE 1.04 (2021) Available WAter Remaining method 1.2c from https://wulca-waterlca.org/, implemented in SimaPro 9.2, 2021.

Briedis, R., Kirkevaag, K., Elliott, T., Darrah, C., Bapasola, A., Sherrington, C. (2019) Reduced Littering of Single-Use Plastics. Report number: M-1360|2019. The Norwegian Environment Agency (https://www.miljodirektoratet.no/globalassets/publikasjoner/m1360/m1360.pdf).

Bergsma, G. & Sevenster, M. 2013. End-of-life best approach for allocating recycling benefits in LCAs of metal packaging. Publication no.: 13.8231.02, CE Delft, Delft, The Netherlands.

BSI 2011. PAS 2050:2011. Specification for the assessment of the life cycle greenhouse gas emissions of goods and services. British Standards Institution.

Carrington, M. J., Neville, B. A., & Whitwell, G. J. (2010). Why ethical consumers don't walk their talk: Towards a framework for understanding the gap between the ethical purchase intentions and actual buying behaviour of ethically minded consumers. *Journal of Business Ethics*, 97(1), 139-158.

CEN 2019. EN 15804:2012+A2:2019 Sustainability of construction works - Environmental product declarations - Core rules for the product category of construction products. Brussels, Belgium: European Committee for Standardization.

CIRAIG (2017) Analyse Du Cycle De Vie Des Sacs D'emplettes Au Québec https://www.recyc-quebec.gouv.qc.ca/sites/default/files/documents/acv-sacs-emplettes-rapport-complet.pdf

COWI (2019) Study about Plastic Sorting and Recycling, COWI A/S, Lyngby, Denmark. https://plastikviden.dk/media/212448/study-about-plastic-sorting-and-recycling.pdf

de Sadeleer, I., Askham, C., Baxter, J., and Stensgård, A (2021) Integration of plastic littering in LCA methodology and eco-design tips for the avoidance of littering. NORSUS. OR.08.21.

EC (2012) Product Environmental Footprint (PEF) Guide. European Commission, Joint Research Centre, Institute for Environment and Sustainability, https://ec.europa.eu/environment/eussd/pdf/footprint/PEF%20methodology%20final%20draft.pdf

Ecoinvent (2021) https://www.ecoinvent.org/database/system-models-in-ecoinvent-3/cut-off-system-model/allocation-cut-off-by-classification.html Accessed 12.05.2021.

Ecoinvent (2020) Ecoinvent database version 3.7.1 https://ecoinvent.org/the-ecoinvent-database/data-releases/ecoinvent-3-7-1/, implemented in SimaPro 9.2.0.2.

Ekvall, T., Björklund, A. Sandin, G., Jelse, K., Lagergren, J., and Rydberg, M. (2020) Modeling recycling in life cycle assessment. Report number: 2020:05. Swedish Life Cycle Center, Gothenburg, Sweden.

Envir (2015) Plukkanalyse av avfall fra Karlsøy og Tromsø. https://avfallnorge.no/fagomraader-og-faggrupper/plukkanalyser/plukkanalyse-av-avfall-fra-karlsøy-og-tromsø Accessed on 2021-10-14

Eurostat (2021a) Gross electricity production by fuel, https://ec.europa.eu/eurostat/statistics-explained/index.php?title=File:Gross_electricity_production_by_fuel,_EU-27,_2000-2018_(GWh).png#file Accessed 27.05.2021

Eurostat (2021b) https://ec.europa.eu/eurostat/statistics-explained/index.php?title=File:Gross_derived_heat_generation_by_fuel,_EU-27,_2000-2018_(GWh)_T4.png Accessed 27.05.2021

Gaudrealt, C. (2020) White Paper. Review Of Life Cycle Assessments Comparing Paper and Plastic Products, NCASI, Montreal, QC, Canada.

GPN (2021) Grønt Punkt Norge Plastemballasje fra husholdning. https://www.grontpunkt.no/gjenvinning/plastemballasje-fra-husholdninger/ Accessed on 12.05.2021.

Handelens Miløfond (2021a) Personal communication with Sjur Kvifte Nesheim with data from an as yet unpublished study “Plastposerapporten 2020”. This report is a study about plastic bags used on the Norwegian market in 2020.

Handelens Miløfond (2021b) Ta med plastposen på flere handleturer! https://handelensmiljofond.no/ta-med-plastposen-pa-flere-handleturer. Acessed 30.12.2021.

Huijbregts MAJ, Steinmann ZJN, Elshout PMF, et al. (2016) ReCiPe2016: a harmonised life cycle impact assessment method at midpoint and endpoint level. Report I: Characterization. RIVM Report 2016-0104.

Huijbregts MAJ, Steinmann ZJN, Elshout PMF, et al. (2017) ReCiPe2016: a harmonised life cycle impact assessment method at midpoint and endpoint level. International Journal of Life Cycle Assessment 22(2): 138-147.

ILCD (2011) European Commission-Joint Research Centre - Institute for Environment and Sustainability: International. Reference Life Cycle Data System (ILCD) Handbook- Recommendations for Life Cycle Impact Assessment in the European context. First edition November 2011. EUR 24571 EN. Luxemburg. Publications Office of the European Union. https://eplca.jrc.ec.europa.eu/uploads/ILCD-Recommendation-of-methods-for-LCIA-def.pdf

Ioelovich, M. (2018) Energy Potential of Natural, Synthetic Polymers and Waste Materials - A Review. Academ J Polym Sci. 2018; 1(1): 555553. DOI: 10.19080/AJOP.2018.01.555553.

ISO (2006a) ISO 14040:2006 Environmental Management -- Life cycle assessment -- Principles and framework. International Organization for Standardization, Geneva, Switzerland.

ISO (2006b) Environmental Management—Life Cycle Assessment—Requirements and Guidelines (ISO 14044). International Organization for Standardization, Geneva, Switzerland.

ISO/TR 14049:2012 Environmental management — Life cycle assessment — Illustrative examples on how to apply ISO 14044 to goal and scope definition and inventory analysis. International Standardisation Organisation.

ISO 2020. ISO 14044:2006/A2:2020 Environmental Management - Life Cycle Assessment - Requirements and guidelines. International Standardisation Organisation.

Jolliet, J., Saadé-Sbeih, M., Shaked, S., Jolliet, A., Crettaz, P. (2016) Environmental Life Cycle Assessment, CRC Press, Taylor & Francis Group, 6000 Broken Sound Parkway NW, Suite 300, Boca Raton, FL 33487-2742, ISBN 978-1-4398-8770-7.

Lavoie, J., Boulay, A.-M., & Bulle, C. 2021. Micro- and Nanoplastics in LCA: Development of an effect factor for the quantification of their physical impact on aquatic biota. J. Ind. Ecol., 2015, 1–13. https://doi.org/10.1111/jiec.13140.

Lopez, J.P., Girones, J., Mendez, J.A. et al. Recycling Ability of Biodegradable Matrices and Their Cellulose-Reinforced Composites in a Plastic Recycling Stream. J Polym Environ 20, 96–103 (2012). https://doi.org/10.1007/s10924-011-0333-1.

Lyng, K.A. and de Sadeleer, I. (2021) Miljøvurdering av ølservering på festivaler (Environmental Assessment of Beer Serving at Festivals), NORSUS report OR 18.21.

Mattila, T, Kujanpää, M., Dahlbo, H., Soukka, R., Myllymaa, T. (2011) Uncertainty and Sensitivity in the Carbon Footprint of Shopping Bags. J. Ind. Ecol. 15 (2) 217-227. DOI: 10.1111/j.1530-9290.2010.00326.x

“Ny Teknik” (2018) Billerud Korsnäs in Norrköping was Europe’s largest paper plant for kraft paper production. https://www.nyteknik.se/premium/har-har-europas-storsta-pappersmaskin-flyttat-in-6935173. Accessed 25.05.2021.

Norsk Fjernvarme. (2021). Fjernvarme - Enegikilder 2020 (Eng. Norwegian district heating. District heating - Energy sources 2020). Retrieved from https://www.fjernkontrollen.no. Accessed 14.06.2021

Norsk Resy. (2021a). Bølgepappskolen. Retrieved from https://www.resy.no/nb/bolgepapp/bolgepappskolen. Accessed 15.06.2021.

Norsk Resy. (2021b). Om Norsk Resy - Det Naturlige Kretsløpet. Retrieved from https://www.resy.no/nb/resy/om-norsk-resy. Accessed 15.06.2021.

Norwegian Directorate of Health (2021). Utviklingen i norsk kosthold 2020: Matforsyningsstatistikk, https://www.helsedirektoratet.no/rapporter/utviklingen-i-norsk-kosthold/Utviklingen%20i%20norsk%20kosthold%202020%20%E2%80%93%20Fullversjon.pdf

Peano, L., Kounina, A., Magaud, V., Chalumeau, S., Zgola, M. and Boucher, J. (2020). Plastic Leak Project, Methodological Guidelines, v.1.0. Quantis + ea.

Pré (2021a). SimaPro. 9.2.0.2 ed. Amersfoort, Netherlands: PRé Sustainability B.V.

Pré (2021b) https://pre-sustainability.com/articles/recipe/. Accessed 16.06.2021.

Pré, (2021c) https://pre-sustainability.com/articles/recipe/. Accessed 16.12.21.

Ports Malaysia (rvo.nl): https://www.rvo.nl/sites/default/files/2020/10/Port%20Development%20in%20Malaysia%20An%20introduction%20to%20the%20countrys%20evolving%20port%20landscape.pdf

Raadal, H. L., Modahl, I. S., & Lyng, K.-A. (2009). Klimaregnskap for avfallshåndtering, Fae I og II. Fase I: Glassemballasje, metallemballasje, papir, papp, plastemballasje, våtorganiskt avfall. Fase II: Treavfall og restavfall fra husholdninger. Østfoldforskning AS. OR.18.09.

Roos, S. Sandin, G. Zamani, B. and Peters, G. (2015) Environmental assessment of Swedish fashion consumption. Five garments – sustainable futures. http://mistrafuturefashion.com/wp-content/uploads/2015/06/Environmental-assessment-of-Swedish-fashion-consumption-LCA.pdf Accessed 14.10.2021.

Rosenbaum R.K., Bachmann, T.M., Gold, L.S., Huijbregts, M.A.J., Jolliet, O., Juraske, R., Koehler, A., Larsen, H.F., MacLeod, M., Margni, M., McKone, T.E., Payet, J., Schuhmacher, M., van de Meent, D., Hauschild, M.Z. (2008) USEtox—the UNEP-SETAC toxicity model: recommended characterisation factors for human toxicity and freshwater ecotoxicity in life cycle impact assessment. Int J Life Cycle Assess (2008) 13:532–546. DOI:10.1007/s11367-008-0038-4.

Sea-distancesorg. (2021). Retrieved from https://sea-distances.org. Accessed 20 September 2021.

SIFO (2019) Forbruksreduksjon av plastposer - en litteraturstudie. FORBRUKSFORSKNINGSINSTITUTTET SIFO. SIFO NOTAT NR 4-19.

Smithers R. (2020) Use of plastic bags in England drops by 59% in a year. The Guardian, UK. https://www.theguardian.com/environment/2020/jul/30/use-of-plastic-bags-in-england-drops-by-59-in-a-year.

SSB 2021 Waste accounts. https://www.ssb.no/en/natur-og-miljo/avfall/statistikk/avfallsregnskapet Accessed 14.10.2021.

SSB 2016 *https://www.ssb.no/en/transport-og-reiseliv/statistikker/havn/kvartal/2016-06-29*

Schmidt, A., Watson, D., Roos, S., Askham, C. & Brunn Poulsen, P. (2016) Gaining Benefits from Discarded Textiles: Life Cycle Assessment (LCA) of different treatment pathways. TemaNord 2016:537, Nordic Council of Ministers.

Swedenergy (2021) Energiföretagen i Sverige. https://www.energiforetagen.se/energifakta/kraftvarme/ Accessed 13.10.2021.

Syversen and Bjørnerud 2020 https://dl8y9d78cbd9m.cloudfront.net/reports/Materialstr%C3%B8manalyse-for-b%C3%A6reposer-2019.pdf?mtime=20210505212456&focal=none

Verones F. (2021) Communication with Askham C. and Pauna V. in ATLANTIS / Dsolve meeting 05.12.2021.

VPK group (2021a) Paper Recycling https://www.vpkgroup.com/en/sustainability/paper-recycling

VPK group (2021b) VPK Halden https://www.vpkgroup.com/plant-overview/vpk-halden

Watson et al. 2020. Kartlegging av brukte tekstiler og tekstilavfall i Norge. https://norsus.no/publikasjon/kartlegging-av-brukte-tekstiler-og-tekstilavfall-i-norge/ Accessed 14.10.2021.

Weidema B P, Bauer C, Hischier R, Mutel C, Nemecek T, Reinhard J, Vadenbo C O, Wernet G. (2013). Overview and methodology. Data quality guideline for the ecoinvent database version 3. Ecoinvent Report 1(v3). St. Gallen: The ecoinvent Centre https://ecoinvent.org/wp-content/uploads/2021/09/dataqualityguideline_ecoinvent_3_20130506.pdf

Wood Z. (2021) Co-op to ditch plastic 'bags for life' over pollution concerns. The Guardian, UK. https://www.theguardian.com/business/2021/apr/30/co-op-to-ditch-plastic-bags-for-life-over-pollution-concerns.

Woods, J. S., Verones, F., Jolliet, O., Vázquez-Rowe, I., Boulay, A. M. (2021) A framework for the assessment of marine litter impacts in life cycle impact assessment. Ecol. Indic., 129 (107918). https://doi.org/10.1016/j.ecolind.2021.107918.

UNEP (2020). (United Nations Environment Programme) Single-use plastic bags and their alternatives - Recommendations from Life Cycle Assessments.

Woods, J. S., Rødder, G., Verones, F. (2019) An effect factor approach for quantifying the entanglement impact onmarine species of macroplastic debris within life cycle impact assessment. Ecol. Indic., 99 (61-66).

WULCA (2021) http://www.wulca-waterlca.org. Accessed 11.05.2021.

Zampori L., Saouter E., Castellani V., Schau E., Cristobal J., Sala S. (2016) Guide for interpreting life cycle assessment result; EUR 28266 EN; doi:10.2788/171315.

Zampori, L., & Pant, R. (2019). Suggestions for updating the Product Environmental Footprint (PEF) method. EUR 29682 EN.

Appendix 1: Shopping habits investigation and reference flows

As mentioned in the main report (section 2.2.2), the reference flows for the LCA depend on the capacity utilisation of different carrying solutions, and in turn on the size distribution of individual shopping events and the carriers used. An observational investigation was carried out to gauge both the number and capacity of utilised carriers.

Methods

The data set was gathered by observing individual shopping events in grocery stores. A completely representative sample of Norwegian shopping as a whole would capture events from stores of different supermarket brand, store size, location, and times of the day / week / season. These factors were taken into account as far as was practicable. Permission was gained to observe at the following locations:

- Oslo: Kiwi Konowsgate, Joker Jens Bjelkes gate
- Halden: Kiwi Brødløs, MENY Halden

The observations were conducted by Astrid Lye Moum (Master student at NMBU) and Ina Charlotte Berntsen (NTNU), on secondment to NORSUS. They were conducted across Weeks 25 and 26 in 2021. Around 40 hours of observation, divided approximately equally between Oslo and Halden, yielded data on over 1500 individual shopping events.

Observations needed to be as unobtrusive as possible, to preserve customers' privacy and also to ensure compliance with infection control requirements. As such, the size of shopping events was gauged simply by the number of items purchased, specifically the number of individual items registered via signals from the cash register or by observing the cashier making a manual entry. All of the cash registers monitored were cashier-operated; there was no self-scanning. It was not always possible to see the groceries on the belt before scanning and/or whilst being scanned, and some mistakes in observations seem inevitable. Also, the number of items is a metric giving some obvious anomalies and inconsistencies for individual shopping events. For example, a multi-pack purchase of 6 items might count as one item, whereas four separate identical items would be registered as four. Also, individual items clearly vary greatly in size and volume. However, it was judged that, given a sufficiently large data set, such anomalies should be averaged-out and the number of items should give a statistically meaningful representation of shopping event size.

More extensive, representative, and accurate data of this sort might be available in principle from sales records / databases from particular retailers. However, it is acknowledged that any such databases, if they even exist, could be both extremely large and difficult to interrogate in practice, and also contain extremely commercially sensitive information. Direct access to such data seems unlikely for our research purposes.

The following specific data were recorded for each shopping event observed:

- Location, date and time
- Number of items purchased
- Number and type of carriers used
- Estimated volumetric filling of carriers

The latter was a completely subjective estimate and only possible in certain cases where observations were sufficiently clear to make an estimate. It was most useful in establishing a link between an average number of items purchased and the volumetric filling of carriers.

Observations

A total of 1535 shopping events, encompassing 11076 items purchased, were observed across the five stores. The overall distribution of shopping event sizes is shown in Figure 1, with most events being relatively small. The modal shopping event size was two items, namely 242 events or just over 15% of the total. The median size was three (nearly four) items; 51% of all events were of four items or less. Because of relatively small numbers of much larger shopping events, the arithmetic mean event size was much higher at 7.22 items.

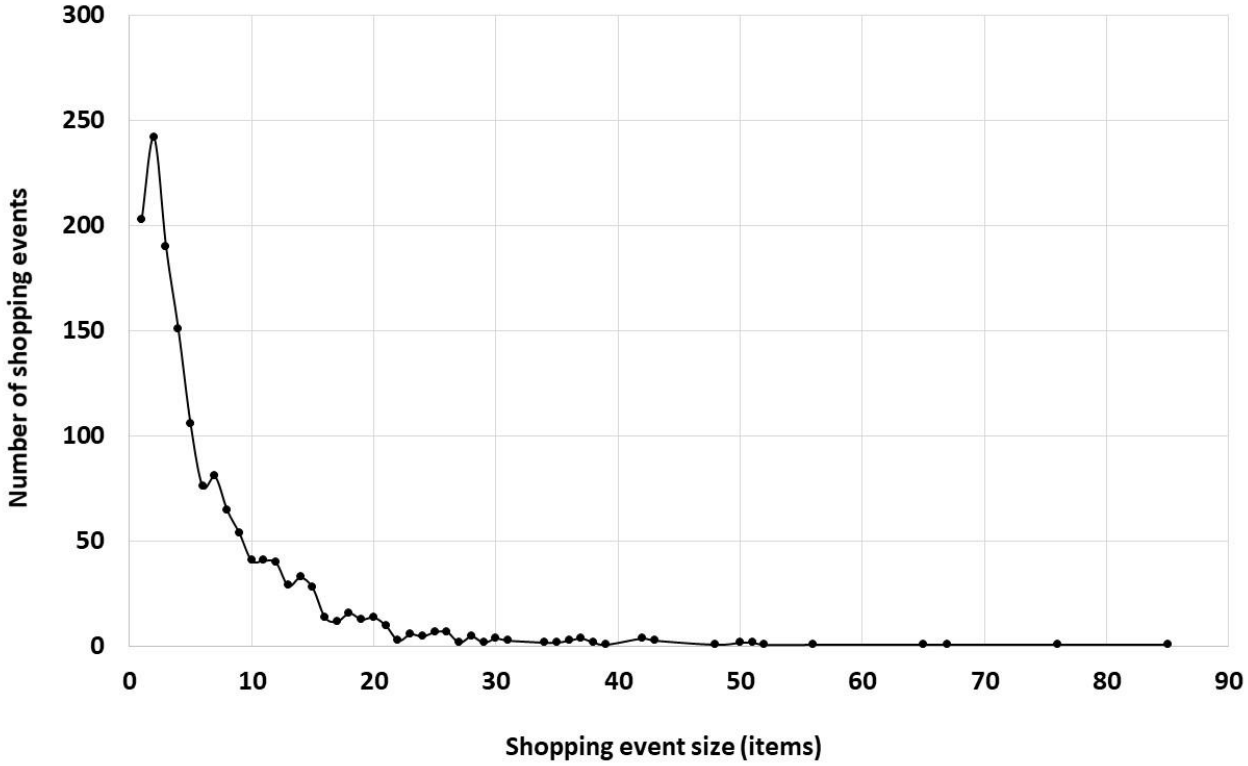


Figure 1: Distribution of observed shopping event sizes

Table 1 shows the data from different stores, showing that the Joker Jens Bjelkes Gate store in particular was quite different from the others (a small store attracting customers buying small numbers of items at a time). Some general observations served to explain these differences – for example the Joker store had a number of other, larger stores in the close vicinity which would naturally attract shoppers undertaking larger shopping events.

	Number of shopping events observed	Arithmetic mean event size (items)	Largest observed event (items)
Kiwi Brødløs	459	7.57	76
Kiwi Konowsgate	567	7.25	85
Joker Jens Bjelkes Gate	49	2.00	7
MENY Halden	297	7.84	67

Table 1: Summary of observations at different stores

The distribution of carrying solutions used is shown in Figure 2. The total number of shopping events slightly exceeds the number of distinct observations (1535) since in a few cases more than one type of

carrying solution was observed. Around 43% of shopping events involved the purchase of a plastic bag and 38% of events involved no carrier at all. The reusable carrying solutions were observed far less frequently.

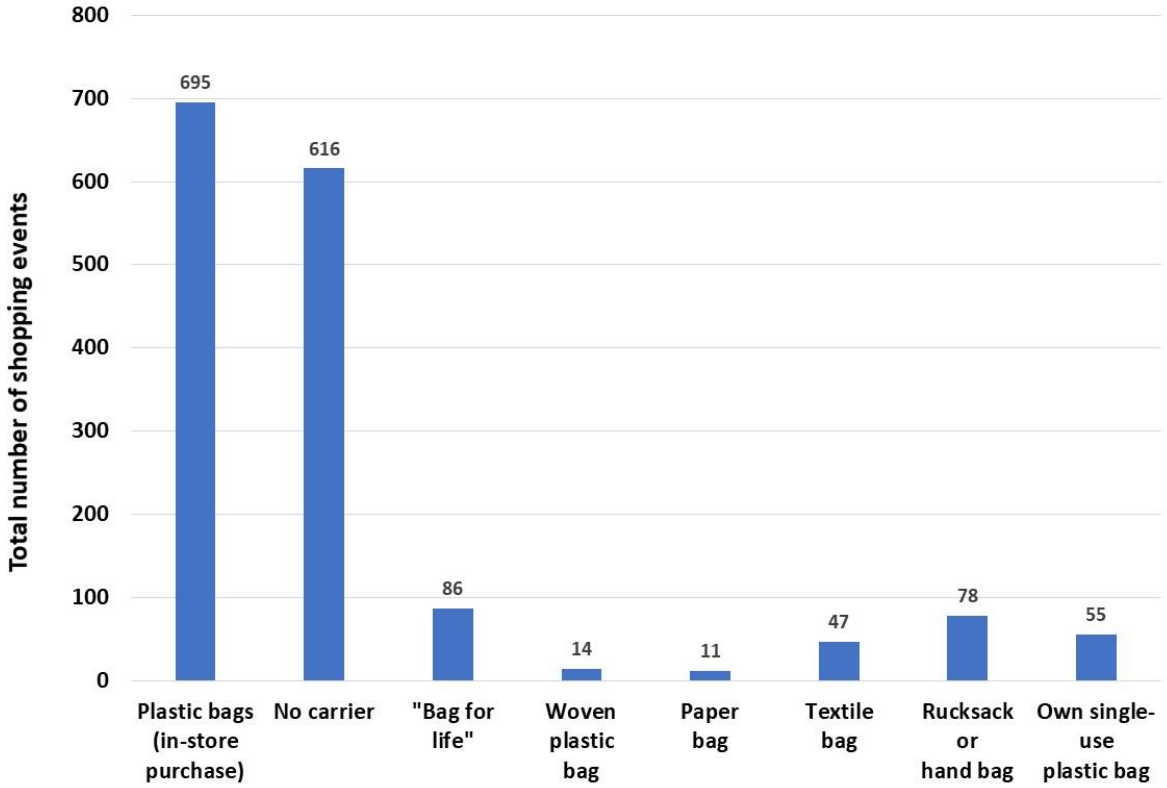


Figure 2: Carriers used in observed shopping events (number of shopping events)

Whilst observations on a per-event basis are interesting and intuitive, for the functional unit the calculations must be placed on a per-item basis, meaning that the fraction of items in shopping events of different sizes, and in events using different carrying solutions, is of most interest. The distinction is important because the utilisation of carriers must relate to the number of items rather than the number of events. Otherwise, a single small item being placed in a single-use plastic bag would be regarded identically as a case where such a bag were filled to maximum capacity.

Figure 3 shows a direct analogue of Figure 1 on a per-item basis, with the total number of items purchased in events of different sizes. However, a cumulative form as shown in Figure 4 is probably more useful. The cumulative distribution illustrates the fraction of all items purchased in events of the indicated size, or less. For example, one data point highlighted in the Figure show that a little over 20% of all items, across all events, are purchased in shopping events of size 5 items or fewer. The actual figures are 2391 items of a total of 11076 or 21.8%. Also, just over half of all items are purchased in shopping events of 12 items or fewer, and only 25% of all items are purchased in events exceeding 20 items (put another way, 75% of items are purchased in events of 20 items or fewer)

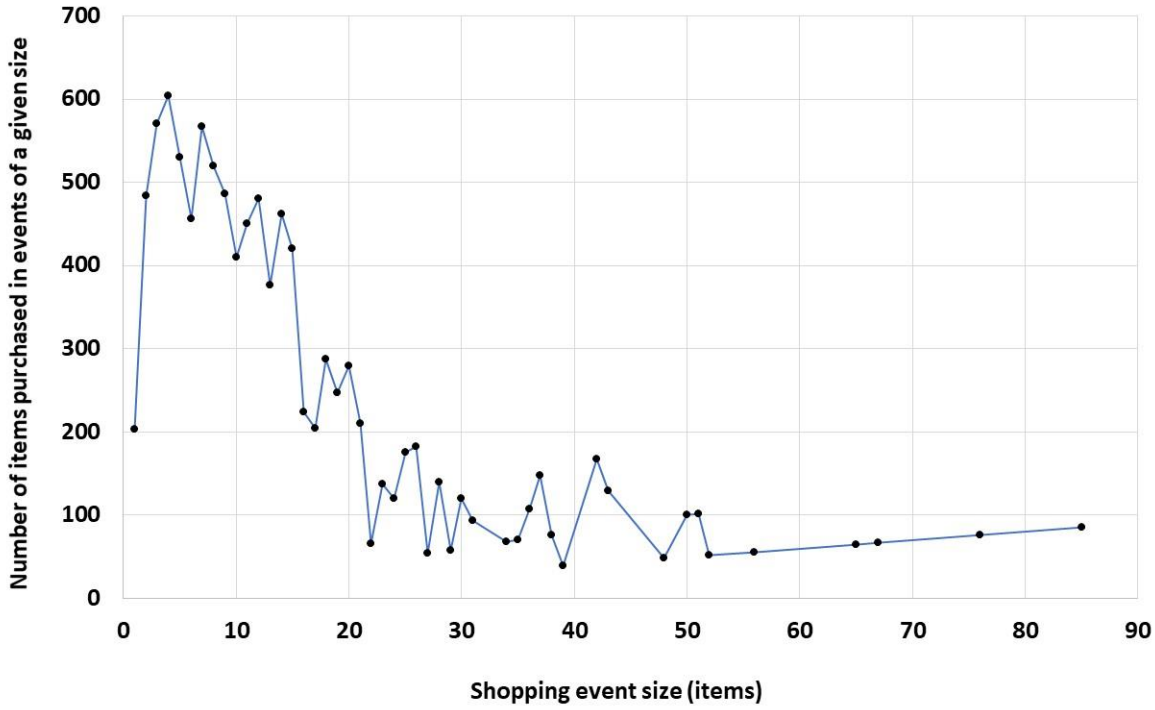


Figure 3: Per-item analogue of Figure 1 (number of items purchased in different-sized events)

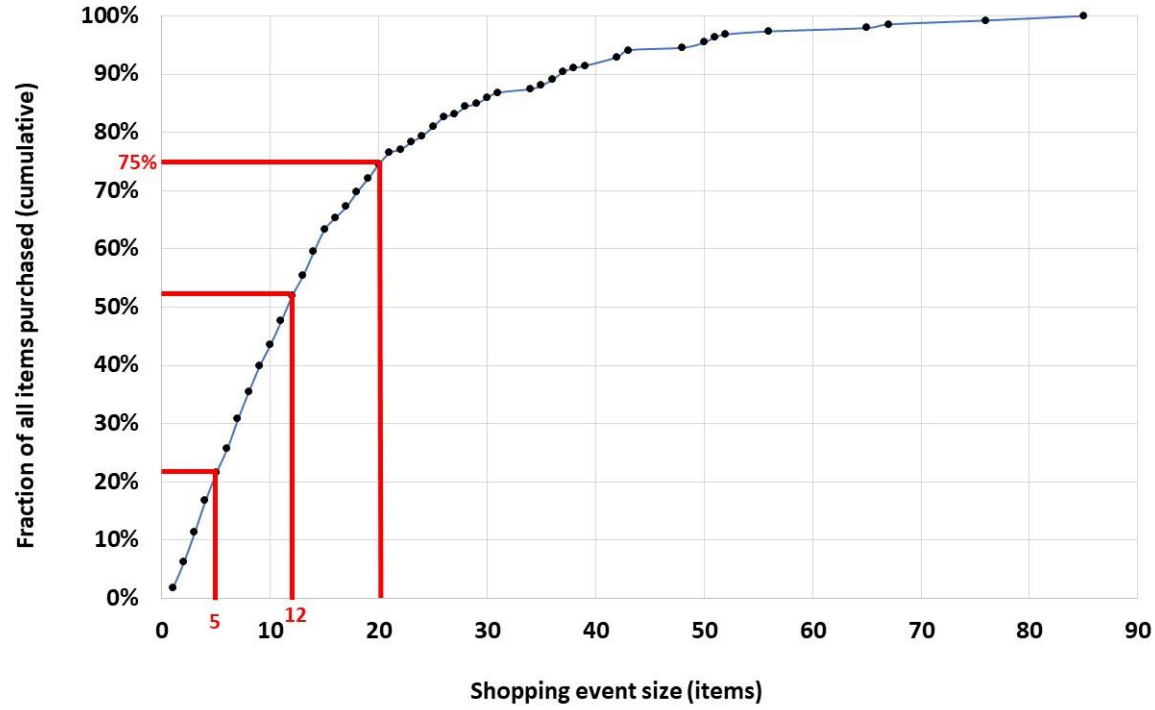


Figure 4: Cumulative distribution of items purchased within shopping events of given size

The data show that whilst small shopping events are the most frequent sort, they only give rise to a minority of all items purchased.

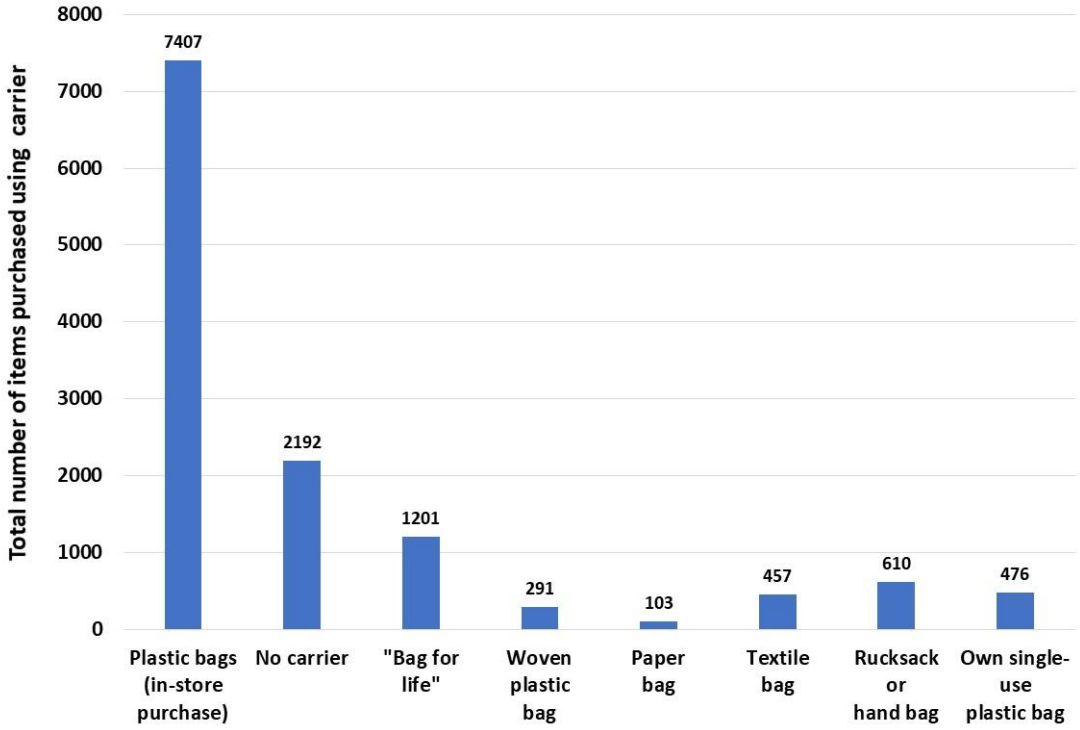


Figure 5: Total number of items purchased in shopping events using different carriers.

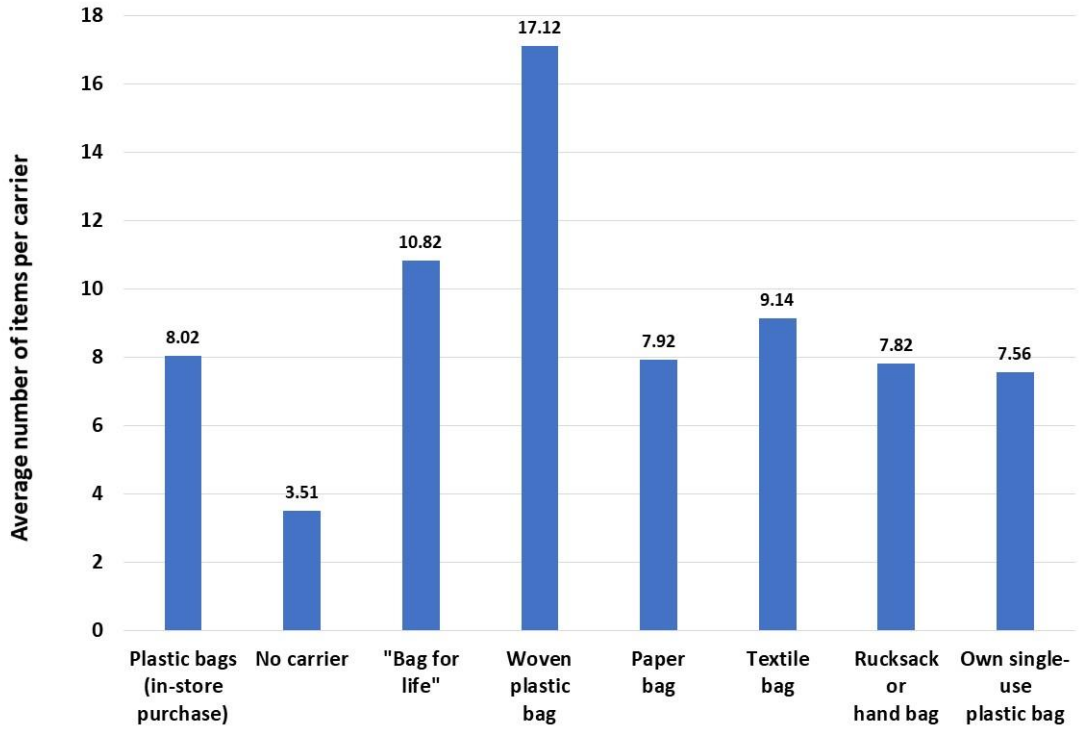


Figure 6: Average number of items per carrier used in observed shopping events

Figure 5 provides the per-item analogue of Figure 2. It presents the number of items purchased in shopping events using different carriers. For example, from Figure 2 we see there were 695 shopping events involving in-store purchased plastic bags. Figure 5 shows that a total of 7407 items were purchased in these 695 events, or 10.7 items per shopping event.

The data in Figure 5 show that around 60% of all items purchased are placed in a plastic bag purchased in-store. Reusable carriers (everything other than in-store plastic bags or "no carrier") account for only around one-quarter of all items purchased.

Reference flows

The key figure for the reference flow is the effective volumetric capacity of the carrier (Vi in Equation 1 in section 2.2.3), specifically the fraction of the true volume that is occupied by goods – the capacity utilisation. This is of interest for all carriers but particularly (single use) plastic bags since it will be important for the allocation of environmental burdens.

Estimates of capacity utilisation were only available for a fraction of all shopping events, moreover the individual observations were subjective. Better estimates are possible if a link between the number of items and the effective volume of groceries can be established, since data on number of items is available for every purchase.

Focusing solely on in-store plastic bags, we isolated the data for events where the shopping bags were judged to be 100% full; there were 50 such events. The observations showed that on average there were 10.2 items per full bag. A further 54 events where the bags were judged to be 90% full suggest that 10.9 items can be expected to be found in a full bag. Based on these data we take a working assumption of 10.5 items per full carrier bag on average. Clearly individual items vary greatly in size, but this average should be reached over a large number of observations. With a measured full volume of 12 litres, this means that the average item is taken to occupy 12/10.5 = 1.1 litres. The overall average capacity utilisation of plastic bags is then found from the average number of items per plastic bag, averaged over all shopping events involving plastic bags.

Figure 6 shows that 8.02 items were placed in each plastic bag on average (7407 items in 923 plastic bags in total), giving an overall capacity utilisation of plastic bags of around 8.02 / 10.5 = 76%.

Equivalent data for all types of carriers are presented in Table 2. The average items per carrier are taken from the data in Figure 6. The volume of items assumes 1.1 litres per item on average, the nominal volume of the carrier is taken from Table 1 in section 2.2.3 and hence the overall capacity utilisation of different carriers is calculated. Note that some of the data sets, notably for woven plastic bags and paper bags, are rather small and hence the calculations may be unreliable.

Carrier Type	Number of carriers	Average items per carrier	Volume of items (litres)	Nominal volume of carrier (litres)	Overall capacity utilisation
Plastic bags (in-store purchase)	923	8.02	9.17	12	76%
No carrier	625	3.51	4.01		
"Bag for life"	111	10.82	12.37	20	62%
Woven plastic bag	17	17.12	19.56	20	98%
Paper bag	13	7.92	9.05	24	38%
Textile bag	50	9.14	10.45	25	42%
Rucksack or hand bag	78	7.82	8.94	30	30%
Own single-use plastic bag	63	7.56	8.63	12	72%

Table 2: Capacity utilisation of different carriers

Appendix 2: Use of rucksacks for grocery transport

In an attempt to gauge important factors regarding the use of rucksacks in grocery shopping, an informal survey was conducted. 28 individuals were polled, and 15 answers received. Subjects were asked if they had ever used rucksacks in grocery shopping, and if so for the last time they did so to specify:

- The size and brand of rucksack used.
- The fraction of the bag's volume used for food.
- The mode of transport to and from the store.

Summary of results

- Rucksacks used varied between 15 and 55 litres capacity. The arithmetic mean and median rucksack sizes were each in the range 30-35 litres.
- In most cases (9 of 15) the entire rucksack volume was used for groceries (100% of volume) and in almost all cases (13 of 15) at least half of the rucksack's total volume was used for groceries. The arithmetic mean of the responses was around 80% and the median 100%.
- None of the respondents drove to the grocery store: 10 of 15 respondents were on foot, 4 cycled (manual or electric cycles) and one used a kick scooter.

Analysis / Comments

The respondents were sustainability professionals and hence presumably more engaged in sustainability matters than the population at large. The mode of transport mix is also probably unrepresentative of the general population – reflecting a relatively young, urban, and environmentally aware group. It also seems likely that such a group are more likely to use rucksacks at all than the population at large. It is perhaps reasonable to propose that the data on average rucksack size (30-35 litres) and utilisation (100%) are reasonable for the subset of the population that use rucksacks. However, this gives little insight into how prevalent rucksack use is in the general population.

The capacity utilisation figures should be subject to interpretation. As established in the main text above, "completely filling" a carrier does not mean that 100% of its volumetric capacity is actually used; 80% is established as an upper limit. The survey was not explicit on this point but it reasonable to conclude when respondents said "100%" they meant the carrier was "filled with groceries" and thus to the upper limit.

Additional questions were asked referring to:

- The reason for purchasing the rucksack
- The fraction of overall rucksack use for shopping, both now and before the pandemic.
- The number of uses per week of the rucksack for shopping, both now and before pandemic
- The age and expected lifetime (future estimated life) of the rucksack

It should be noted that the perceived lifetime might not be correct, respondents may both under- and overestimate this.

Summary of results

- None of the respondents had purchased the rucksack for purpose of using it for shopping. 4 respondents had been given the rucksack as a gift, or as a prize.
- Rucksack use for shopping (rather than an alternative use) varied widely (from 0-100%) with an average value of around 30%, with little difference pre- and during pandemic.

- Uses per week for shopping with rucksacks varied widely between 0 and 5, with an average of about 1 currently. Some difference pre- and during pandemic was seen. The average pre-pandemic was 1.7.
- Rucksack age varied widely up to about 15 years with an average of 6 years. Expected future lifetime of rucksacks again varied widely (from 2 to 15 years) but with an average of 10 years. Thus, the average total lifetime expected for the respondents' rucksacks was 16 years.

For sensitivity analysis, it should be noted that the lowest total lifetime expected by the respondents was 6 years and the highest 30. The lowest number of uses (fewest times per week * lowest age in years (yrs) * 52 weeks/yr) is 39, the highest (most times per week * longest age yrs *52 weeks/yr) is 7800.

Conclusions

We can conclude that the total lifetime of rucksacks was of the order of 15 years. Taking 1 use per week as average gives 15 x 52 = 780 uses. Taking 30% of the overall use being shopping means the effective "number of uses" for all purposes is 780 / 0.3 = 2600 and hence each shopping visit should be attributed 1/2600 of the rucksack's burdens.

Appendix 3: Use of textile bags ("bags-for-life")

The same respondent group (see Appendix 2) was questioned about the use of textile / reusable bags. Estimates of numbers of uses for textile bags vary enormously. Evidence from the UK (Wood 2021) suggests that significant numbers of consumers use "bags-for-life" on effectively a single-use basis, and their sale is being discouraged or abandoned. Previous literature studies have suggested up to 50 uses (e.g. Ekvall 2020) as typical for a lifetime.

However, our respondents suggested that textile bags could have much longer lives with many more potential uses than these studies indicate. Among other questions regarding material composition and such, respondents were asked to estimate:

- The frequency of use of textile bags in the previous 4 weeks and whether that was different to pre-pandemic use.
- The current age and likely lifetime of their textile bags
- The material the textile bag was made from

Across 44 responses, the 50 uses of the previous literature appears consistent with one year of use (the arithmetic mean response was 57 uses per annum and the median 52 uses). However, the lifetimes of such bags were estimated to be much longer than just one year. The arithmetic mean response was just under 22 years and the median response 20 years.

Assuming 50 uses per annum and a lifetime of 20 years, a typical textile bag is taken to have 1000 lifetime uses. For simplicity, the textile bag is assumed to never be used for purposes other than grocery shopping.

For sensitivity analysis the shortest expected bag life was 12 years in total, the longest was 45. The lowest number of uses for the respondents' textile bags in the last 4 weeks was 1, the highest 8. This provides a range of 156 – 4680 uses.

The respondents replied that the materials their bags were made of were cotton (22) and polyester (22), although there was some uncertainty expressed ("?") about which type of plastic the synthetic textile bags were made from. The average number of uses did not change between material types, but the average lifetime expected was 18 years for polyester bags (range: 6-30) and 25 years for cotton bags (range: 8-45).

Another piece of information asked of the respondents was whether they had purchased their textile bag, or been given it. A little under half of the textile bags were purchased.

Appendix 4: Background system data for the studied systems

The data quality assessment approach is described in Chapter 3.6, but the categories for assessment are repeated here for the reader's convenience:

- Very good - Geographically representative for this study, data for processes and products under study, less than 3 years between the reference year and year of study.
- Good - Average data from a larger area of study, data for processes and products with similar technology, less than 6 years between the reference year and year of study.
- Fair - Data from an area with similar production conditions, data for processes or products under study but from different technology, less than 10 years of difference between the reference year and year of study.
- Poor - Data from an area with slightly similar production conditions, data for related processes or products, less than 15 years of difference between the reference year and year of study

Please note that the Ecoinvent data base often operates with extrapolated data sets, rather than gathering a lot of data for each revision. The age of the data considered (see categories above) refers to the time period for which the Ecoinvent database states that the data is valid, rather than the original year of study (unless otherwise commented upon). See Chapter 5 for details of the foreground data used and their data quality assessment.

Table A1. Background system data applied for the low-density polyethylene (LDPE) plastic bag

Process	Location or transport type	Ecoinvent dataset	Comments	Data quality Assessment
Extraction and production (including transports)				
Virgin LDPE production	Global	Polyethylene, low density, granulate {GLO}| market for | Cut-off, U		Good
Recycled LDPE production	Europe	Polyethylene, high density, granulate, recycled {Europe without Switzerland}| market for polyethylene, high density, granulate, recycled | Cut-off, U	A dataset for recycled LDPE was unavailable in Ecoinvent. The dataset for HDPE recycling was applied but corrected using data from COWI (2019) to represent average efficiency and energy use in LDPE recycling, specifically	Good
Production of plastic bag				
Extrusion LDPE film (virgin and/or recycled material)	Sweden	Extrusion, plastic film {RER}| extrusion, plastic film | Cut-off, U Electricity, medium voltage {SE}| market for | Cut-off, U		Fair (older data for the process technology itself, but updated electricity data and database links to other processes and datasets used.
Transport (of plastic bag to the store)				

Transport (Sweden to Oslo)	Truck	Transport, freight, lorry, unspecified {RER}| market for transport, freight, lorry, unspecified | Cut-off, U	European data. Transport from Sweden to Oslo in Norway	Good
Waste management (incineration and plastic recycling)				
Transport to incineration	Truck	Transport, freight, lorry, unspecified {RER}| market for transport, freight, lorry, unspecified | Cut-off, U	European data	Fair. Data for a refuse vehicle may be more appropriate.
Incineration	Norway	Waste polyethylene {CH}| treatment of, municipal incineration | Cut-off, U	Data for Switzerland used as proxy	Fair (older data for the process technology itself, but updated electricity data and database links to other processes and datasets used.
Incineration, avoided heat production (end-of-life scenario)	Norway	Norwegian district heating mix, 2020	Note that this dataset was based on data from Fjernkontrollen (2021) and only applied in the end-of-life scenario	Very good
Transport to recycling	Truck	Transport, freight, lorry, unspecified {RER}| market for transport, freight, lorry, unspecified | Cut-off, U	European data	Fair, a combination of refuse vehicle and lorry is likely to be better.
LDPE recycling (end-of-life scenario)	Europe	Polyethylene, high density, granulate, recycled {Europe without Switzerland}| market for polyethylene, high density, granulate, recycled | Cut-off, U	A dataset for recycled LDPE was unavailable in Ecoinvent. The dataset for HDPE recycling was applied but corrected using data from COWI (2019) to represent average efficiency and energy use in LDPE recycling, specifically. Note that this dataset was only applied in the end-of-life scenario	Good
LDPE recycling, avoided production (end-of-life scenario)	Europe	Polyethylene, low density, granulate {RER}| production | Cut-off, U	Note that this dataset was only applied in the end-of-life scenario	Good

Table A2. Background system data applied for the high-density polyethylene (HDPE) plastic bag

Process	Location or transport type	Ecoinvent dataset	Comments	Data quality Assessment
Extraction and production (including transports)				
Virgin HDPE production	Global	Polyethylene, high density, granulate {GLO}| market for | Cut-off, U		Good
Recycled HDPE production	Europe	Polyethylene, high density, granulate, recycled {Europe without Switzerland}| market for polyethylene, high density, granulate, recycled | Cut-off, U Electricity, medium voltage {MY}| market for | Cut-off, U	European data corrected for Malaysian electricity mix (various electricity mixes were tested in the scenario analysis)	Good
Production of plastic bag				
Extrusion HDPE film (virgin and/or recycled material)	Southeast Asia	Extrusion, plastic film {RER}| extrusion, plastic film | Cut-off, U Electricity, medium voltage {MY}| market for | Cut-off, U		Fair (older data for the process technology itself, but updated electricity data and database links to other processes and datasets used.
Transport (of plastic bag to the store)				
Transport (Southeast Asia to Oslo)	Ship	Transport, freight, sea, container ship {GLO}| market for transport, freight, sea, container ship | Cut-off, U	Transport from Port in Southeast Asia to Port Oslo in Norway	Good
Waste management (incineration and plastic recycling)				
Transport to incineration	Truck	Transport, freight, lorry, unspecified {RER}| market for transport, freight, lorry, unspecified | Cut-off, U	European data	Fair. Data for a refuse vehicle may be more appropriate.
Incineration	Norway	Waste polyethylene {CH}| treatment of, municipal incineration | Cut-off, U	Data for Switzerland used as proxy	Fair (older data for the process technology itself, but updated electricity data and database links to other processes and datasets used.
Incineration, avoided heat production (end-of-life scenario)	Norway	Norwegian district heating mix, 2020	Note that this dataset was based on data from Fjernkontrollen (2021) and only applied in the end-of-life scenario	Very good
Transport to recycling	Truck	Transport, freight, lorry, unspecified {RER}| market for transport, freight,	European data	Fair, a combination of refuse vehicle and lorry is likely to be better.

		lorry, unspecified | Cut-off, U		
HDPE recycling (end-of-life scenario)	Europe	Polyethylene, high density, granulate, recycled {Europe without Switzerland}| market for polyethylene, high density, granulate, recycled | Cut-off, U	Note that this dataset was only applied in the end-of-life scenario	Good
HDPE recycling, avoided production (end-of-life scenario)	Europe	Polyethylene, high density, granulate {RER}| production | Cut-off, U	Note that this dataset was only applied in the end-of-life scenario	Good

Table A3. Background system data applied for the bin liner

Process	**Location or transport type**	**Ecoinvent dataset**	**Comments**	**Data quality Assessment**
Extraction and production (including transports)				
Recycled LDPE production	Europe	Polyethylene, high density, granulate, recycled {Europe without Switzerland}| market for polyethylene, high density, granulate, recycled | Cut-off, U	A dataset for recycled LDPE was unavailable in Ecoinvent. The dataset for HDPE recycling was used but corrected using data from COWI (2019) to represent average efficiency and energy use in LDPE recycling, specifically	Good
Production of plastic bag				
Extrusion LDPE film	Sweden	Extrusion, plastic film {RER}| extrusion, plastic film | Cut-off, U Electricity, medium voltage {SE}| market for | Cut-off, U		Good
Transport (of plastic bag to the store)				
Transport (Arvika to Oslo)	Truck	Transport, freight, lorry, unspecified {RER}| market for transport, freight, lorry, unspecified | Cut-off, U	Transport from Sweden to Oslo in Norway	Good
Waste management (incineration)				
Transport to incineration	Truck	Transport, freight, lorry, unspecified {RER}| market for transport, freight, lorry, unspecified | Cut-off, U	European data	Fair. Data for a refuse vehicle may be more appropriate.
Incineration	Norway	Waste polyethylene {CH}| treatment of, municipal incineration | Cut-off, U	Data for Switzerland used as proxy	Fair (older data for the process technology itself, but updated electricity data and database

				links to other processes and datasets used.
Incineration, avoided heat production (end-of-life scenario)	Norway	Norwegian district heating mix, 2020	Note that this dataset was based on data from Fjernkontrollen (2021) and only applied in the end-of-life scenario	Very good

Table A4. Background system data applied for the paper bag

Process	Location or transport type	Ecoinvent dataset	Comments	Data quality Assessment
Extraction and production (including transports)				
Virgin paper production	European	Kraft paper {RER}| market for kraft paper | Cut-off, U		Good
Recycled paper production	Global/Canada, Québec	Paper, woodfree, uncoated {CA-QC}| paper production, woodfree, uncoated, 100% recycled content, at non-integrated mill | Cut-off, U Electricity, medium voltage {RER}| market group for | Cut-off, U	Data for Canada, Québec, were corrected for European electricity mix	Poor. See Chapter 6.5.1 . Closer examination shows that there is a large discrepancy in energy carrier use per kg in the factory data for recycled paper (about 14 MJ/kg) when compared to the equivalent data for virgin material (about 2.5 MJ/kg). Thus the data used for recycled paper should be re-evaluated in further work.
Production of paper bag				
Paper bag production	Sweden	Corrugated board box {RER}| production | Cut-off, U Electricity, medium voltage {SE}| market for | Cut-off, U	Production processes for paper bags were unavailable in Ecoinvent, therefore, the production process for corrugated board box was used instead as a proxy.	Poor
Transport (of paper bag to the store)				
Transport (Sweden to Oslo)	Truck	Transport, freight, lorry, unspecified {RER}| market for transport, freight, lorry, unspecified | Cut-off, U	Transport from Sweden to Oslo, Norway	Good
Waste management (incineration and paper recycling)				
Transport to incineration	Truck	Transport, freight, lorry, unspecified {RER}| market for transport, freight,	European data	Fair. Data for a refuse vehicle may be more appropriate.

		lorry, unspecified | Cut-off, U		
Incineration	Norway	Waste paperboard {CH}| treatment of, municipal incineration | Cut-off, U	Data for Switzerland used as proxy	Fair (older data for the process technology itself, but updated electricity data and database links to other processes and datasets used.
Incineration, avoided heat production (end-of-life scenario)	Norway	Norwegian district heating mix, 2020	Note that this dataset was based on data from Fjernkontrollen (2021) and only applied in the end-of-life scenario	Very good
Transport to recycling	Truck	Transport, freight, lorry, unspecified {RER}| market for transport, freight, lorry, unspecified | Cut-off, U	European data	Fair, a combination of refuse vehicle and lorry is likely to be better.
Paper recycling (end-of-life scenario)	Norway or Europe	Paper, woodfree, uncoated {CA-QC}| paper production, woodfree, uncoated, 100% recycled content, at non-integrated mill | Cut-off, U Electricity, medium voltage {NO}| market for | Cut-off, U Electricity, medium voltage {Europe without Switzerland}| market group for | Cut-off, U	Note that this dataset was only applied in the end-of-life scenario. The dataset was corrected for Norwegian electricity mix for the share that is recycled in Norway and corrected with European, without Switzerland, electricity mix for the share that is recycled abroad	Poor. See Chapter 6.5.1 . Closer examination shows that there is a large discrepancy in energy carrier use per kg in the factory data for recycled paper (about 14 MJ/kg) when compared to the equivalent data for virgin material (about 2.5 MJ/kg). Thus the data used for recycled paper should be re-evaluated in further work.
Paper recycling, avoided production (end-of-life scenario)	Europe	Kraft paper {RER}| market for kraft paper | Cut-off, U	Note that this dataset was only applied in the end-of-life scenario	Good

Table A5. Background system data applied for the nylon bag

Process	**Location or transport type**	**Ecoinvent dataset**	**Comments**	**Data quality Assessment**
Extraction and production (including transports)				

Virgin nylon fiber production	Global	Fibre, polyester {GLO}| market for fibre, polyester | Cut-off, U - HMF Nylon 6-6 {RER}| market for nylon 6-6 | Cut-off, U	Dataset for polyester fiber production was corrected for virgin nylon input	Good
Production of nylon bag for life				
Nylon bag production	East Asia	Textile, non-woven polyester {GLO}| market for textile, non woven polyester | Cut-off, U Electricity, medium voltage {CN}| market group for | Cut-off, U	The production process for polyester textile was used as a proxy for the nylon bag production. Dataset corrected for East Asian electricity mix	Fair
Transport (of nylon bag for life to the store)				
Transport (East Asia to Oslo)	Ship	Transport, freight, sea, container ship {GLO}| market for transport, freight, sea, container ship | Cut-off, U	Transport from East Asian port to Port Oslo in Norway	Good
Waste management (incineration and nylon recycling)				
Transport to incineration	Truck	Transport, freight, lorry, unspecified {RER}| market for transport, freight, lorry, unspecified | Cut-off, U	European data	Fair. Data for a refuse vehicle may be more appropriate.
Incineration	Norway	Waste plastic, mixture {CH}| treatment of, municipal incineration | Cut-off, U	Data for Switzerland used as proxy	Fair (older data for the process technology itself, but updated electricity data and database links to other processes and datasets used.
Incineration, avoided heat production (end-of-life scenario)	Norway	Norwegian district heating mix, 2020	Note that this dataset was based on data from Fjernkontrollen (2021) and only applied in the end-of-life scenario	Very good
Transport to sorting	Truck	Transport, freight, light commercial vehicle {RER}| market group for transport, freight, light commercial vehicle | Cut-off, U	Transport from collection point to sorting	Fair, a combination of refuse vehicle and lorry is likely to be better.
Sorting	Europe	Electricity, medium voltage {RER}| market group for | Cut-off, U	Process for electricity used in sorting	Good
Transport to recycling	Truck	Transport, freight, lorry, unspecified {RER}| market for transport, freight, lorry, unspecified | Cut-off, U		Good

Nylon recycling (end-of-life scenario)	Europe	*Electricity:* Electricity, medium voltage {Europe without Switzerland}| market group for | Cut-off, U *Thermal energy:* Heat, district or industrial, other than natural gas {Europe without Switzerland}| market for heat, district or industrial, other than natural gas | Cut-off, U *Detergents:* Non-ionic surfactant {GLO}| market for non-ionic surfactant | Cut-off, U *Tap water:* Tap water {Europe without Switzerland}| market for | Cut-off, U	Processes for various inputs to recycling	Good
Incineration	Europe	Waste plastic, mixture {CH}| treatment of, municipal incineration | Cut-off, U	Incineration of losses at recycling facility. Data for Switzerland used as proxy	Fair (older data for the process technology itself, but updated electricity data and database links to other processes and datasets used.
Incineration, avoided electricity production (end-of-life scenario)	Europe	Electricity, medium voltage {RER}| market group for | Cut-off, U		Good
Incineration, avoided heat production (end-of-life scenario)	Europe	Heat, district or industrial, other than natural gas {Europe without Switzerland}| market for heat, district or industrial, other than natural gas | Cut-off, U		Good
Nylon recycling, avoided production (end-of-life scenario)	Global	Fibre, flax {GLO}| market for fibre, flax | Cut-off, U		Good

Table A6. Background system data applied for the polyethylene terephthalate (PET) bag

Process	Location or transport type	Ecoinvent dataset	Comments	Data quality Assessment
Extraction and production (including transports)				
Recycled PET fiber production	Global	Fibre, polyester {GLO}| market for fibre, polyester | Cut-off, U - HMF Polyethylene terephthalate, granulate, amorphous, recycled {Europe without Switzerland}| market for polyethylene terephthalate, granulate, amorphous, recycled | Cut-off, U	Dataset for polyester fiber production was corrected for recycled PET granulate input	Good
Production of PET bag for life				
PET bag production	East Asia	Textile, non-woven polyester {GLO}| market for textile, non woven polyester | Cut-off, U Electricity, medium voltage {CN}| market group for | Cut-off, U	The production process for polyester textile was used as a proxy for the PET bag production. Dataset corrected for East Asian electricity mix	Fair
Transport (of PET bag to the store)				
Transport (Shanghai to Oslo)	Ship	Transport, freight, sea, container ship {GLO}| market for transport, freight, sea, container ship | Cut-off, U	Transport from port in East Asia to Port Oslo in Norway	Good
Waste management (incineration and PET recycling)				
Transport to incineration	Truck	Transport, freight, lorry, unspecified {RER}| market for transport, freight, lorry, unspecified | Cut-off, U	European data	Fair. Data for a refuse vehicle may be more appropriate
Incineration	Norway	waste polyethylene terephthalate {RoW}| treatment of waste polyethylene terephthalate, municipal incineration | Cut-off, U		Fair
Incineration, avoided heat production (end-of-life scenario)	Norway	Norwegian district heating mix, 2020	Note that this dataset was based on data from Fjernkontrollen (2021) and only applied in the end-of-life scenario	Very good
Transport to sorting	Truck	Transport, freight, light commercial vehicle {RER}| market group for transport, freight, light commercial vehicle | Cut-off, U	Transport from collection point to sorting	Fair, a combination of refuse vehicle and lorry is likely to be better.
Sorting	Europe	Electricity, medium voltage {RER}| market group for | Cut-off, U	Process for electricity used in sorting	Good

Transport to recycling	Truck	Transport, freight, lorry, unspecified {RER}| market for transport, freight, lorry, unspecified | Cut-off, U		Good
PET recycling (end-of-life scenario)	Europe	*Electricity:* Electricity, medium voltage {Europe without Switzerland}| market group for | Cut-off, U *Thermal energy:* Heat, district or industrial, other than natural gas {Europe without Switzerland}| market for heat, district or industrial, other than natural gas | Cut-off, U *Detergents:* Non-ionic surfactant {GLO}| market for non-ionic surfactant | Cut-off, U *Tap water:* Tap water {Europe without Switzerland}| market for | Cut-off, U	Processes for various inputs to recycling	Good
Incineration	Europe	waste polyethylene terephthalate {RoW}| treatment of waste polyethylene terephthalate, municipal incineration | Cut-off, U	Incineration of losses at recycling facility.	Fair
Incineration, avoided electricity production (end-of-life scenario)	Europe	Electricity, medium voltage {RER}| market group for | Cut-off, U		Good
Incineration, avoided heat production (end-of-life scenario)	Europe	Heat, district or industrial, other than natural gas {Europe without Switzerland}| market for heat, district or industrial, other than natural gas | Cut-off, U		Good
PET recycling, avoided production (end-of-life scenario)	Global	Fibre, flax {GLO}| market for fibre, flax | Cut-off, U		Good

Table A7. Background system data applied for the cotton bag

Process	Location or transport type	Ecoinvent dataset	Comments	Data quality Assessment
Extraction and production (including transports)				
Virgin cotton yarn production	Global	Yarn, cotton {GLO}| yarn production, cotton, ring spinning, for weaving | Cut-off, U Fibre, cotton, organic {GLO}| market for fibre, cotton, organic | Cut-off, U	Dataset for yarn production corrected for organic cotton fiber input	Good
Production of cotton bag				
Cotton bag production	South Asia	Textile, woven cotton {GLO}| market for | Cut-off, U (for transports) Textile, woven cotton {RoW}| textile production, cotton, weaving | Cut-off, U (for weaving) Finishing, textile, woven cotton {GLO}| market for finishing, textile, woven cotton | Cut-off, U Electricity, medium voltage {IN}| market group for electricity, medium voltage | Cut-off, U	Datasets corrected for South Asian electricity mix	Good
Transport (of cotton bag to the store)				
Transport (Mumbai to Oslo)	Ship	Transport, freight, sea, container ship {GLO}| market for transport, freight, sea, container ship | Cut-off, U	Transport from Mumbai in India, to Port Oslo in Norway	Good
Waste management (incineration and cotton recycling)				
Transport to incineration	Truck	Transport, freight, lorry, unspecified {RER}| market for transport, freight, lorry, unspecified | Cut-off, U	European data	Fair. Data for a refuse vehicle may be more appropriate
Incineration	Norway	Waste textile, soiled {RoW}| treatment of, municipal incineration | Cut-off, U		Fair
Incineration, avoided heat production (end-of-life scenario)	Norway	Norwegian district heating mix, 2020	Note that this dataset was based on data from Fjernkontrollen (2021) and only applied in the end-of-life scenario	Very good
Transport to sorting	Truck	Transport, freight, light commercial vehicle {RER}| market group for transport, freight, light commercial vehicle | Cut-off, U	Transport from collection point to sorting	Fair, a combination of refuse vehicle and lorry is likely to be better.

Sorting	Europe	Electricity, medium voltage {RER}| market group for | Cut-off, U	Process for electricity used in sorting	Good
Transport to recycling	Truck	Transport, freight, lorry, unspecified {RER}| market for transport, freight, lorry, unspecified | Cut-off, U		Good
Cotton recycling (end-of-life scenario)	Europe	*Electricity:* Electricity, medium voltage {Europe without Switzerland}| market group for | Cut-off, U *Thermal energy:* Heat, district or industrial, other than natural gas {Europe without Switzerland}| market for heat, district or industrial, other than natural gas | Cut-off, U *Detergents:* Non-ionic surfactant {GLO}| market for non-ionic surfactant | Cut-off, U *Tap water:* Tap water {Europe without Switzerland}| market for | Cut-off, U	Processes for various inputs to recycling	Good
Incineration	Europe	Waste textile, soiled {RoW}| treatment of, municipal incineration | Cut-off, U	Incineration of losses at recycling facility.	Fair
Incineration, avoided electricity production (end-of-life scenario)	Europe	Electricity, medium voltage {RER}| market group for | Cut-off, U		Good
Incineration, avoided heat production (end-of-life scenario)	Europe	Heat, district or industrial, other than natural gas {Europe without Switzerland}| market for heat, district or industrial, other than natural gas | Cut-off, U		Good
Cotton recycling, avoided production (end-of-life scenario)	Global	Fibre, flax {GLO}| market for fibre, flax | Cut-off, U		Good

Table A8. Background system data applied for the cardboard box

Process	Location or transport type	Ecoinvent dataset	Comments	Data quality Assessment
Extraction and production (including transports)				
Virgin containerboard, linerboard production	Europe	Containerboard, linerboard {RER}| market for containerboard, linerboard | Cut-off, U	The dataset was corrected so that only kraftliner (virgin) linerboard and not testliner (recycled) linerboard is used	Good
Recycled containerboard, linerboard production	Europe	Containerboard, linerboard {RER}| market for containerboard, linerboard | Cut-off, U	The dataset was corrected so that only testliner (recycled) linerboard and not kraftliner (virgin) linerboard is used	Good
Virgin containerboard, fluting medium production	Europe	Containerboard, fluting medium {RER}| market for containerboard, fluting medium | Cut-off, U	The dataset was corrected so that only semichemical (virgin) and not recycled fluting medium is used	Good
Recycled containerboard, fluting medium production	Europe	Containerboard, fluting medium {RER}| market for containerboard, fluting medium | Cut-off, U	The dataset was corrected so that only recycled and not semichemical (virgin) fluting medium is used d	Good
Production of cardboard box				
Cardboard box production	Norway	Corrugated board box {RER}| production | Cut-off, U Electricity, medium voltage {NO}| market for | Cut-off, U	European data corrected for Swedish electricity mix	Good
Transport (of paper bag to the store)				
Transport (Southeast Norway to Oslo)	Truck	Transport, freight, lorry, unspecified {RER}| market for transport, freight, lorry, unspecified | Cut-off, U	Transport from a specific location in Southeast Norway to Oslo, Norway	Good
Waste management (incineration and cardboard box recycling)				
Transport to incineration	Truck	Transport, freight, lorry, unspecified {RER}| market for transport, freight, lorry, unspecified | Cut-off, U	European data	Fair. Data for a refuse vehicle may be more appropriate
Incineration	Norway	Waste paperboard {CH}| treatment of, municipal incineration | Cut-off, U	Data for Switzerland used as proxy	Fair (older data for the process technology itself, but updated electricity

				data and database links to other processes and datasets used.
Incineration, avoided heat production (end-of-life scenario)	Norway	Norwegian district heating mix, 2020	Note that this dataset was based on data from Fjernkontrollen (2021) and only applied in the end-of-life scenario	Very good
Transport to recycling	Truck	Transport, freight, lorry, unspecified {RER}| market for transport, freight, lorry, unspecified | Cut-off, U	European data	Fair, a combination of refuse vehicle and lorry is likely to be better.
Cardboard box recycling (end-of-life scenario)	Norway or Europe	Containerboard, linerboard {RER}| containerboard production, linerboard, testliner | Cut-off, U Electricity, medium voltage {NO}| market for | Cut-off, U Electricity, medium voltage {Europe without Switzerland}| market group for | Cut-off, U	Note that this dataset was only applied in the end-of-life scenario. The dataset was corrected for Norwegian electricity mix for the share that is recycled in Norway and corrected with European, without Switzerland, electricity mix for the share that is recycled abroad	Good
Cardboard box recycling, avoided production (end-of-life scenario)	Europe	Containerboard, linerboard {RER}| containerboard production, linerboard, kraftliner | Cut-off, U	Note that this dataset was only applied in the end-of-life scenario	Good

Table A9. Background system data applied for the rucksack

Process	Location or transport type	Ecoinvent dataset (unless otherwise noted)	Comments	Data quality Assessment
Extraction and production (including transports)				
Virgin polyester textile production	East Asia / Southeast Asia	Fibre, polyester {GLO}| market for fibre, polyester | Cut-off, U For weaving: *Electricity* Electricity, medium voltage {TW}| market for | Cut-off, Electricity, medium voltage {CN}| market group for | Cut-off, U Electricity, medium voltage {VN}| market for electricity, medium voltage | Cut-off, U *Heat* Heat, central or small-scale, other than natural gas {RoW}| heat production, light fuel oil, at boiler 10kW condensing, non-modulating | Cut-off, U *Starch* Maize starch {GLO}| market for | Cut-off, U		Good
Virgin Nylon 6 production	Southeast Asia	nylon 6 {RoW}| market for nylon 6 | Cut-off, U Polar fleece production, energy use only {RoW}| production | Cut-off, U		Fair
Virgin Foam EVA/PU production	Southeast Asia	Polyurethane, flexible foam {RoW}| market for polyurethane, flexible foam | Cut-off, U		Good
Virgin Polyoxymethylene (POM) buckles production	Southeast Asia	Polyoxymethylene (POM)/EU-27 Injection moulding {GLO}| market for | Cut-off, U	Note that the dataset "Polyoxymethylene (POM)/EU-27" is from the Industry 2.0 database in SimaPro	Fair
Virgin steel production	Global / Europe	Steel, low-alloyed {GLO}| market for | Cut-off, U *combined with*		Good

		Steel, low-alloyed (RER)| steel production, converter, low-alloyed | Cut-off, U Deep drawing, steel, 3500 kN press, single stroke (GLO)| market for | Cut-off, U		
Production of rucksack				
Rucksack production	Southeast Asia	Electricity, medium voltage {VN}| market for electricity, medium voltage | Cut-off, U		Good
Transport (of rucksack to the store)				
Transport (Vietnam to Oslo)	Ship	Transport, freight, sea, container ship {GLO}| market for transport, freight, sea, container ship | Cut-off, U	Transport from Ho Shi Minh city in Vietnam, to Port Oslo in Norway	Good
Waste management (incineration and rucksack recycling)				
Transport to incineration	Truck	Transport, freight, lorry, unspecified {RER}| market for transport, freight, lorry, unspecified | Cut-off, U	European data	Fair. Data for a refuse vehicle may be more appropriate
Incineration	Norway	Waste plastic, mixture {CH}| treatment of, municipal incineration | Cut-off, U	Data for Switzerland used as proxy	Fair (older data for the process technology itself, but updated electricity data and database links to other processes and datasets used.
Incineration, avoided heat production (end-of-life scenario)	Norway	Norwegian district heating mix, 2020	Note that this dataset was based on data from Fjernkontrollen (2021) and only applied in the end-of-life scenario	Very good
Transport to sorting	Truck	Transport, freight, light commercial vehicle {RER}| market group for transport, freight, light commercial vehicle | Cut-off, U	Transport from collection point to sorting	Fair, a combination of refuse vehicle and lorry is likely to be better.
Sorting	Europe	Electricity, medium voltage {RER}| market group for | Cut-off, U	Process for electricity used in sorting	Good
Transport to recycling	Truck	Transport, freight, lorry, unspecified {RER}| market for transport, freight, lorry, unspecified | Cut-off, U		Good

Rucksack, textiles, recycling (end-of-life scenario)	Europe	*Electricity:* Electricity, medium voltage {Europe without Switzerland}| market group for | Cut-off, U *Thermal energy:* Heat, district or industrial, other than natural gas {Europe without Switzerland}| market for heat, district or industrial, other than natural gas | Cut-off, U *Detergents:* Non-ionic surfactant {GLO}| market for non-ionic surfactant | Cut-off, U *Tap water:* Tap water {Europe without Switzerland}| market for | Cut-off, U	Processes for various inputs to recycling	Good
Rucksack, steel, recycling (end-of-life scenario)	Europe	Steel, low-alloyed (RER)| steel production, electric, low-alloyed | Cut-off, U		Good
Incineration	Europe	Waste plastic, mixture {CH}| treatment of, municipal incineration | Cut-off, U	Incineration of losses at recycling facility. Data for Switzerland used as proxy	Fair (older data for the process technology itself, but updated electricity data and database links to other processes and datasets used.
Incineration, avoided electricity production (end-of-life scenario)	Europe	Electricity, medium voltage {RER}| market group for | Cut-off, U		Good
Incineration, avoided heat production (end-of-life scenario)	Europe	Heat, district or industrial, other than natural gas {Europe without Switzerland}| market for heat, district or industrial, other than natural gas | Cut-off, U		Good
Rucksack, textiles, recycling, avoided production (end-of-life scenario)	Global	Fibre, flax {GLO}| market for fibre, flax | Cut-off, U		Good
Rucksack, steel, recycling, avoided production (end-of-life scenario)	Global	Steel, low-alloyed (RER)| steel production, converter, low-alloyed | Cut-off, U		Good

Appendix 5: Textile waste Management in Norway

Amounts of Norwegian textile waste incinerated and recycled

The data and text presented in this part of the appendix is from the reference Watson et al. (2020): https://norsus.no/wp-content/uploads/or1120-kartlegging-av-brukte-tekstiler-og-tekstilavfall-i-norge_Versjon-2.pdf (eller https://norsus.no/publikasjon/kartlegging-av-brukte-tekstiler-og-tekstilavfall-i-norge/).

PlanMiljø (DK) and Ostfold Research (NO) were commissioned by the Norwegian Environment Agency to carry out a study called "Mapping of used textiles and textile waste in Norway" (published in 2020).

The study made use of available data on imports, exports and production of new and used textiles and data on shares of textiles in mixed household waste, supplemented by primary data and information gathering. Primary data was gathered through surveys of used textile collectors, municipal waste companies, clothing retailers and international wholesalers and recyclers of textiles.

This study found that 45% of the volume of new textiles purchased by private households was collected in textile collection schemes. In Norway separate textile collection is carried out by charitable organisations (79%), municipal waste companies (13%) and private collectors (8%). 93% of the collection is achieved via bring-banks placed on public or private ground. Kerbside (door-to-door) collection represents just 4% of the total. Almost no used textiles are collected from government or business.

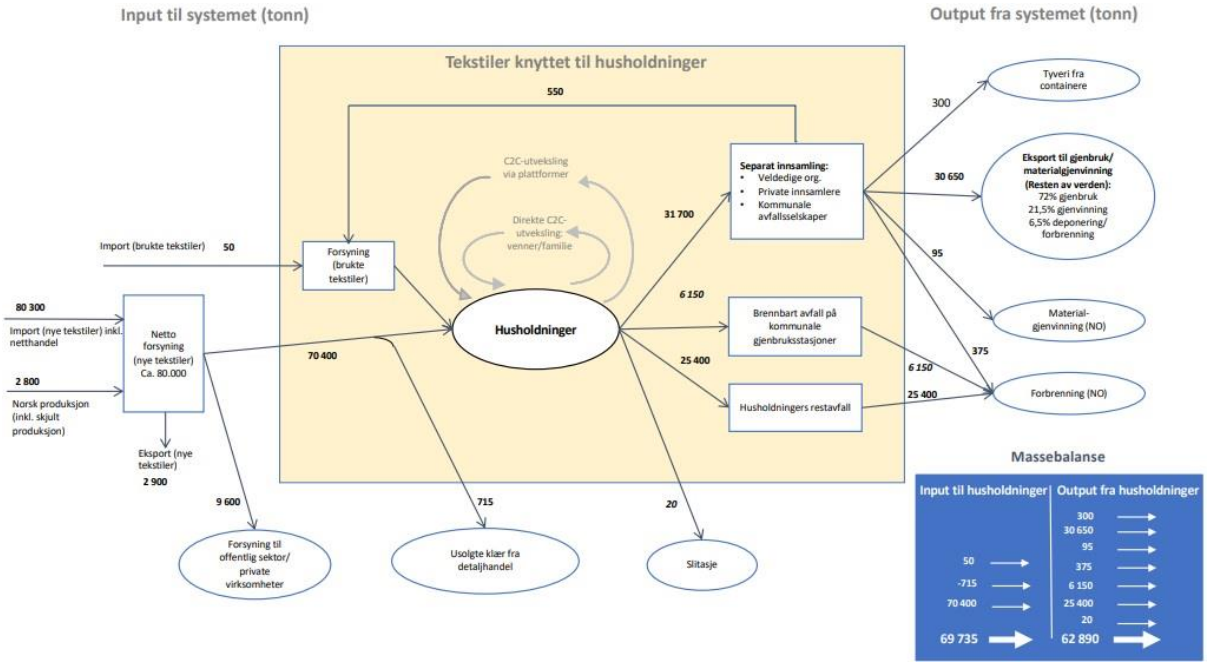
97% of used textiles collected in Norway were subsequently exported for detailed sorting, reuse and recycling elsewhere. Only 3% stayed in Norway for reuse (550 tonnes), recycling (90 tonnes) or incineration (375 tonnes). Sorting takes place in EU countries with responsible waste management systems. The report states that 6.5 % of non-reusable or recyclable waste that arises from the sorting processes can, therefore, assumed to be disposed of safely. 72% of the exported textiles are reused on global reuse markets, while 21,5% are recycled as industry wipes, upholstery, insulation, non-woven rugs etc. Reuse gives a much higher price than recycling.

Norwegian retailers are left with at least 700 tonnes annually, and potentially much more, of unsold garments (dead-stock) which they must find ways to dispose of. 45% of retailers claim to donate at least some of these garments to charitable organisations for reuse. Collectors reported receiving 600 tonnes of unsold textiles from retailers in 2018, usually under the agreement that these may only be sold or donated in countries where the retailing brand isn't active. Retailers reported sending 95 tonnes of unsold textiles to incineration but the figure could be significantly higher since the larger retailers were not willing or able to report on quantities. Recirculation of deadstock is hindered by a number of barriers including VAT rules that penalise recirculation and a lack of viable material recycling options.

At least half the textiles consumed by households are thought to end in mixed waste for incineration. Data from picking analyses of mixed waste streams from households gave an estimated 31 550 tonnes of textiles disposed of in mixed waste. However, the method is relatively inaccurate and it is thought that a good share of the unaccounted 6 745 tonnes of textiles emerging from mass balance calculations, may also be disposed of in mixed waste.

Markets for non-reusable and lower quality reusable textiles are showing signs of saturation driven by an increasing supply but a stable demand.

Many see fibre-to-fibre recycling and other forms of textile-to-textile recycling as a solution to the problems of lacking recycling markets and the worsening economics of used textile collection. However, fibre-to-fibre recycling faces a range of technical, economic, regulatory and market barriers. Mechanical and chemical recycling with high quality outputs requires inputs of used textiles with single fibre types and which are free of persistent chemicals. Chemical recycling technologies that can extract quality fibres from fibre mixes, are still at the development stage. Automated sorting of textile waste into fibre types is on the verge of being launched at industrial scale. However, demand for recycled content yarns and fabrics is low in the textile industry, and textile products are not designed with ease of recycling in mind.



The data shown in the figure above has been used to derive the proportions of textile waste that are incinerated and recycled for the bag for life scenarios in this report.

Recycling of textiles

The data and information in this appendix is from Schmidt, A., Watson, D., Roos, S., Askham, C. & Brunn Poulsen, P. (2016) Gaining Benefits from Discarded Textiles: Life Cycle Assessment (LCA) of different treatment pathways. TemaNord 2016:537, Nordic Council of Ministers

In Schmidt et al. (2016) discarded textiles were assumed to undergo a series of processes before they were in a form fit for use in production where they can substitute a virgin material that would otherwise have been used. Avoided impacts from production of the substituted material were subtracted from the impacts of the recycling processes to give an estimation of the overall impacts or benefits of the recycling scenario. It was assumed that textiles were collected, sorted, transported and shredded to replace virgin textile fibres, insulation fibres, or cellulose (tissue).

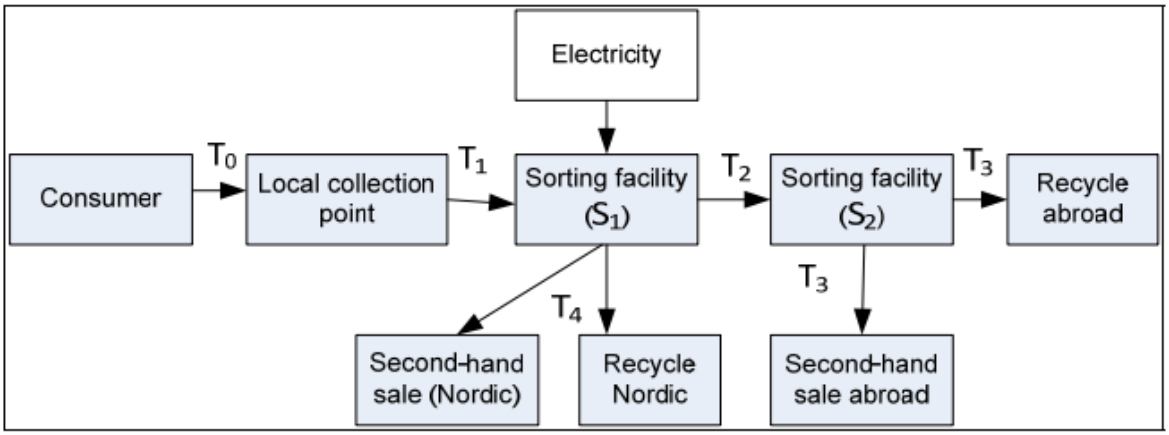
Substitution factors in recycling scenarios

In the recycling scenarios, a main challenge is to identify the point of substitution, i.e. the point where the output from a recycling process can substitute an alternative product produced in a different way. If technical properties are important there is often a need for more recycled material (measured by weight) to achieve the same functionality as if virgin materials are used. An example of this is that secondary (re-spun) fibres do not have the same strength as virgin fibres. Nevertheless, secondary fibres can (and do) substitute virgin fibres in a 1:1 ratio in some products. They cannot substitute the full amount of virgin fibres without compromising the quality of the product, but as much as 20% recycled fibres are being used in products that are considered to have the same functionality as products made entirely from virgin fibres.

Even though the substitution ratio is assumed to be 1:1, losses in the recycling process must also be taken into account. When shredding fibres, some will inevitably lose their functionality and have to be discarded as waste. Typically, one kg of fibres for recycling will only yield 0.8 kg for further processing and can, accordingly, only substitute 0.8 kg of virgin fibres. The remaining 0.2 kg is assumed to be incinerated with energy recovery in Europe.

Data for collection and sorting of textile waste

Figure 5: Basic activities in collection and sorting of used textiles



- T_0: The consumer brings the textile to the collection point in conjunction with another errand and thus 0 km is allocated to textile collection.
- T_1: Transport to a Nordic sorting facility. Data obtained: Vehicle size used for collection: 15 tonne or 7.5 tonne gross weight. Amount of textiles collected: 2 tonnes on average. Distance driven: 20 km (variation from 10 to 150 km is possible). Modelled using data for a 12–14 tonne truck with 10-ton capacity and a utilisation rate of 0.2. The distance was set to 150 km.
- S1 and S2: Sorting facilities. A typical sorting process involves conveyor transport, with largely manual sorting, followed by bailing. S1 energy consumption is approximately 70 kWh electricity per tonne of clothes sorted, equal to 0.25 MJ/kg. Electricity consumption in a second sorting facility is not included in the calculations but will in relevant cases be of the same magnitude.
- T2: Transport to second sorting facility in Europe. Data obtained: Vehicle size used for collection: 15–19 tonne (load weight of clothes. Distance driven: 1,600 km (Drammen-Krakow, Poland assumed). Modelled using data for a 20–26 tonne lorry with 17.9-ton capacity and a utilisation rate of 0.4. Distance driven is 1 600 km.
- T3: Transport to reuse/recycling in ROW. Same truck and distance as for transport to Poland in T2. Additionally, a transport with a container ship is assumed (distance = 12,000 km, using Pakistan as an example country). This is the scenario used when addressing ROW scenarios, giving the most conservative estimate for transport related impacts.

- T4: Transport to Nordic second-hand shop or recycling. Vehicle size used for distribution: 15 tonne or 7.5 tonne gross weight. Amount of textiles collected: 2 tonnes on average. Distance driven: 150 km (variation from 10 to 150 km is possible). Modelled using data for a 12–14 tonne truck with 10-ton capacity and a utilisation rate of 0.2.

Washing and drying of textiles for reuse and recycling

Washing and drying of separately collected textiles has not been included as an induced impact. It may be relevant for some garments but it is judged from the descriptions of well-established sorting facilities that it is not a general activity. It is estimated the washing and drying in relevant cases will cause an energy consumption of about 0.65 kWh, increasing the consumption of primary energy with about 5 MJ/kg.

Energy recovery from incineration of fibres

Four fibre types are included in the calculations of impacts and benefits associated with incineration of textiles. The lower heat value of the fibre types is as follows:

- Cotton: 20.2 MJ/kg.
- Polyester: 21.2 MJ/kg.
- Wool: 23.2 MJ/kg.
- Flax: 20.2 MJ/kg.

Please note that the tables in Appendix 4 of this report detail which electricity and heat mixes are replaced for the relevant carrier.

The choice of recycling process and replaced material

Cotton recycling scenarios

The main challenge in defining the recycling scenarios is to find a “point of substitution” where the recycled material has the same technical properties as the (virgin) material it is substituting. The scenarios described …are very basic in the sense that the discarded textiles are subjected to one or a few relatively simple processes where after it can be further processed into a new fibre-based product using the same processes as for the substituted virgin material.

Options for recycled cotton substitution

Recycled cotton fibres substituting virgin cotton fibres for production of yarn

This specific option is not considered for bags for life in this “I pose og sekk” study. However the information about how it could be modelled is included here for reference and comparison.

There is fairly limited experience with respect to recycling of used cotton textiles into fibres that subsequently can be used in the production of new textiles after spinning, weaving, dyeing, etc. However, it is known that some companies such as H&M and G-Star Raw are making use of recycled cotton fibres in new products along with virgin fibres.

This need to mix recycled fibres with new fibres is a result of a shortening of fibres during wear and laundering of the original textile product and further during the recycling process. It is, therefore, currently not possible to achieve a high quality textile product using only recycled fibres. In practice, the recycled fibres are blended with virgin fibres at some point in the production process. This can happen during spinning, but it can also take place in weaving with up to 20% of the finished product being produced from recycled yarn as is the

case for H&M and G-Star Raw producing denim products like jeans with 20% recycled cotton content. There is also a third part assured global standard for inclusion of recycled cotton in new garments (Textile exchange, 2014).32 In the mechanical recycling process, the cotton waste is sorted by type and colour, cut into small pieces, passed through a rotating drum and finally turned into fibres. The physical quality of the fibres produced using this method is low due to the shortened fibre lengths. One way to improve the quality of this product is to mix these fibres with virgin fibres and blend them into yarns (Zamani, 2014).

In the calculations it has been assumed that the substitution factor is 1, in recognition of the fact that an economically satisfactory market already exists for secondary cotton fibres and that the resulting products with up to 20% recycled cotton are of as good quality as products entirely from new cotton fibres.

The elements in this scenario are shown in Figure 25. It is noted that it is assumed that the collected cotton is transported to Asia for processing. If this type of recycling takes place in Europe, the environmental benefits will be marginally higher.

It is also noted that only the basic processes in (avoided) production of virgin cotton fibres and (induced) recycling into secondary fibres are included. In practice, it is assumed that cotton collected for recycling after shredding into loose fibres can substitute virgin cotton fibres in bales 1:1 (w/w).

If more processes than shredding are needed to give the secondary fibres qualities ready for spinning, this will decrease the benefits accordingly. Mechanical processes, however, are not very demanding in the overall picture. The basic shredding process used in the calculations thus only requires 0.18 MJ of electricity per kg, based on information from a producer of textile shredders. As this process initially produces down-sized textiles, it has been chosen to double the energy consumption to allow for a subsequent pulling process, yielding loose fibres that are suitable for spinning. A 20% loss of material has been assumed for the process. This loss is assumed to be incinerated without energy recovery. The material balance thus shows that for one tonne of cotton being collected, 800 kg of recycled fibres are produced and 200 kg incinerated. The 800 kg of recycled fibres can substitute 800 kg of virgin cotton in bales.

Mechanically recycled cotton fibres substituting flax insulation in Europe

This is the scenario chosen for cotton bags for life in this "I pose og sekk" study

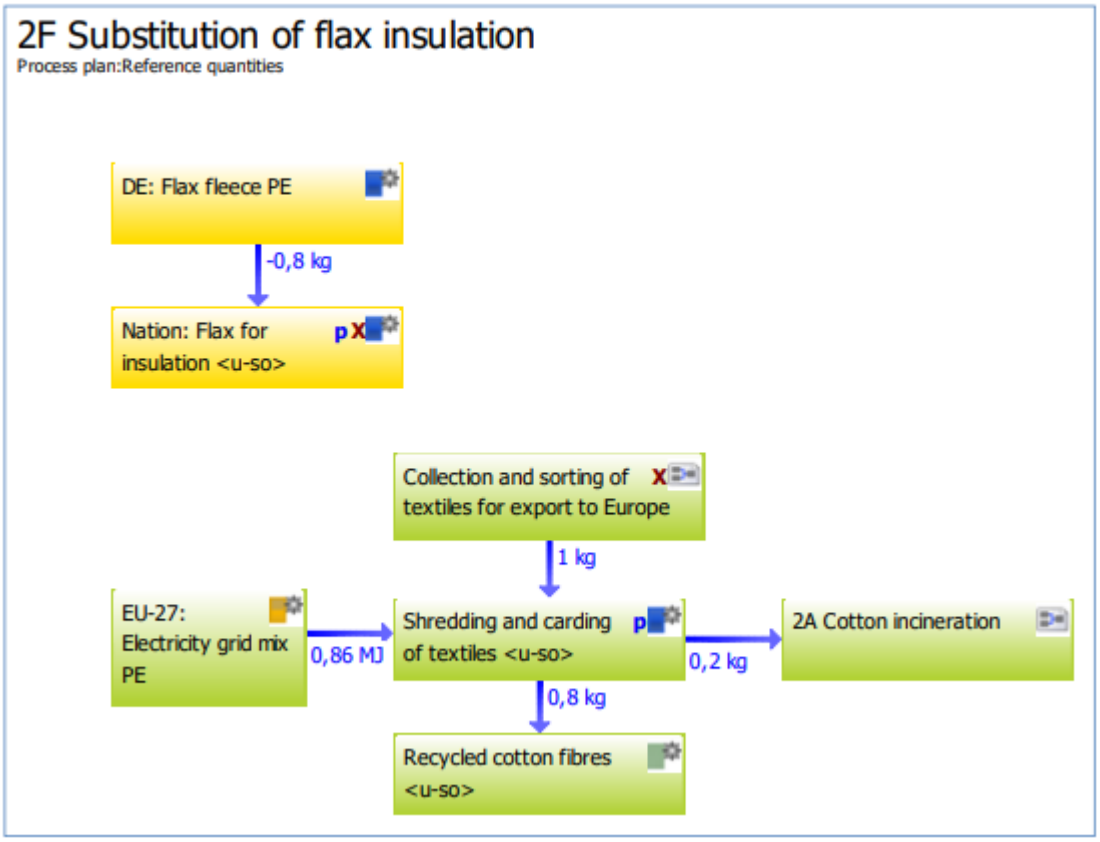
Many different types of fabrics are used in cars for many purposes. Insulation – both acoustic and thermal – is an application where fabrics are commonly used, but seldom visible or identifiable as fabrics. If visible, the appearance is determined by colouring of the plastic matrix of the composite material. According to O'Dell (2011), a wide range of materials are suitable for the purpose: "Acoustic and thermal insulation uses needlepunched nonwoven composites made with natural fibres (kenaf, jute, waste cotton, flax) in blends, often with PP and PET, for floor covering, door panels, headliners, trunk liners and parcel shelves". Pure fractions of recycled cotton and polyester can be used in the same type of applications, and also blends of wool and polyester are reported to be used in the production of fleece and fabric with noise-reducing properties, e.g., by Recytex (Recytex.com) in Germany.

It is noted that the use of recycled textile materials in cars is not limited to insulation products. A company like Rando (Rando.com) designs random air laid-down manufacturing systems for production of headliners, sun shades, acoustical pads and door panels, using cotton, cotton shoddy, synthetic shoddy, recycled fibre, reclaimed material, carbon, and fiberglass as fibres in a composite.

Of the many possibilities for recycling of fabrics in cars it is chosen to investigate the consequences of substituting virgin flax with mechanically recycled cotton waste in car insulation. It has not been possible to

map the actual use of different fibres in this type of applications. It is therefore assumed that recycled cotton fibres after shredding and carding can substitute flax fibres 1:1 (w/w). It is noted that a similar substitution also is possible with regard to upholstery of chairs in cars and home furniture The elements in this scenario are shown in Figure 26.

Figure 26: Elements included in the assessment of impacts and benefits from substituting flax insulation with secondary cotton in Europe



The cotton recycling processes include shredding and carding, two processes that combined requires 0.238 kWh/kg, based on information in Östlund et al. (2015). The loss in shredding and carding is estimated to be 20%, the full amount of this is assumed to be incinerated with energy recovery in an average municipal incineration plant in the EU.

The EOL treatment of virgin flax and recycled cotton fibres is the same, irrespective of their application. If the EOL treatment involves incineration, this may cause very small differences with respect to impacts and benefits, related to the chemical composition of the cellulosic fibres. There is, however, not specific processes available … for incineration of flax and cotton, and the potential differences in impacts from this stage of the life cycle are therefore not considered in the calculations.

Recycling of mixed fibres as a substitution for cellulose-base industrial wipes and low-quality flax-based filling material

The scenario chosen for nylon and polyester bags for life in this "I pose og sekk" study is this recycling of mixed fibres as a substitution for low-quality flax-based filling material.

For the wipes scenario, the avoided and induced processes are shown in Figure 30. The used mixed textiles are washed and dried and cut into wipe size. This has been modelled using data from a Swedish hospital laundry and data from Thompson et al. (2012), assuming a per kg electricity consumption of 0.4 kWh, thermal energy consumption of 6.84MJ, detergent consumption of 0.009 kg and 12 kg of tap water. A loss of 10% is assumed to be incinerated with energy recovered, using national efficiencies and substitution scenarios.

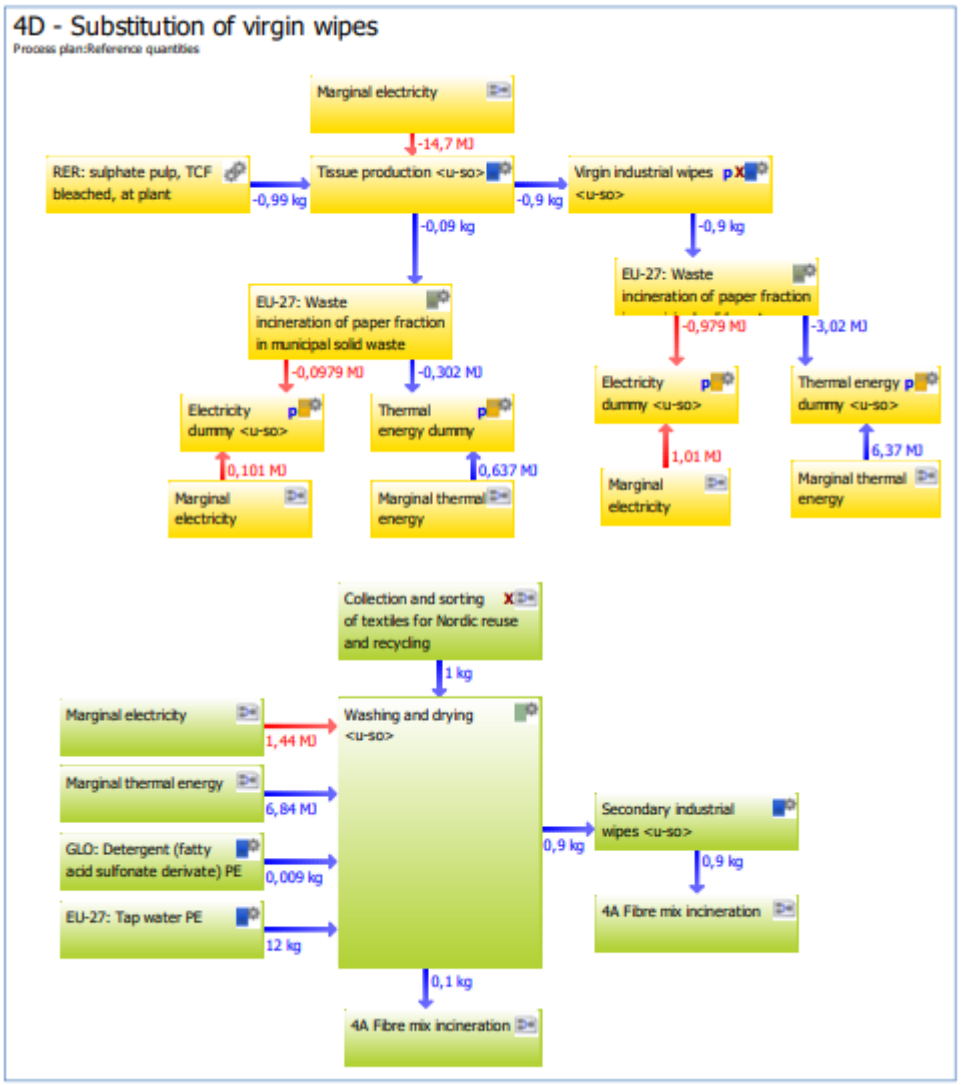
The substituted cellulose based industrial wipes are modelled as produced from sulphate pulp using ecoinvent data, and production data from Pullman et al. (1997) and Ekstrom (2012).

The scenario investigated is that one tonne of recycled mixed fibres results in 900 kg secondary wipes for industrial use, replacing 900 kg of virgin cellulose-based industrial wipes. The loss of 100 kg fibres in the process is assumed to be incinerated in the Nordic countries with energy recovery.

Recycling of mixed fibre as non-woven filling material substituting flax-based filling material

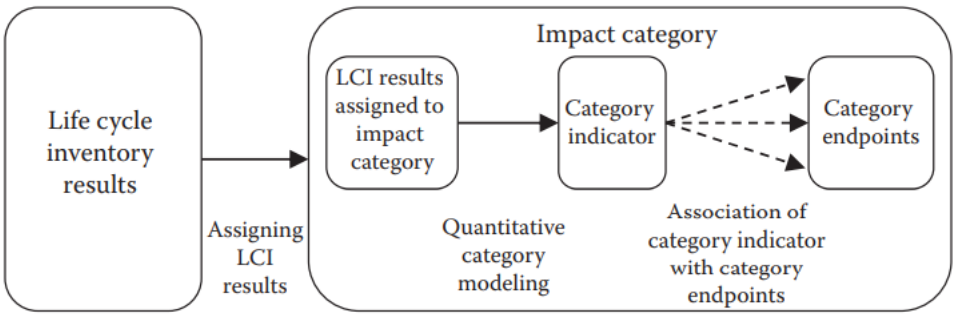
The elements in this scenario are similar to the previous scenario, except for incineration of the waste from shredding and carding of recycled fibres. This difference is judged to be without importance in the overall picture, and the results are accordingly also representative for mixed fibres.

Figure 30: Induced and avoided processes in mixed fibre substitution of cellulose-based wipes



Appendix 6: LCIA calculation procedures

As shown in Figure 6, the LCI data calculated from the reference flow (see Chapter 3.3) is used to calculate impact category results and then category endpoints (see figure below).



Impact assessment scheme to link inventory results with category end point or damage to areas of protection (from Jolliet et al. 2016, ISO 14040, 2006).

Impact categories are calculated using inventory data and characterisation factors, shown in the equation below (Jolliet et al. 2016):

$$S_m^{\text{midpoint}} = \sum_i \left(CF_{m,i}^{\text{midpoint}} u_i \right)$$

where:

$CF_{m,i}^{\text{midpoint}}$ (in, e.g., $kg_{CO2\text{-}eq}/kg_i$) is the midpoint characterization factor of substance i in the midpoint category m

u_i is the emitted or extracted mass of substance i per functional unit as given in the inventory

The relevant characterisation factors used in this study are referred to in Chapter 3.7. Readers can find the details of the characterisation factors used in the references for each method.

The calculation procedure for endpoint (potential damage) can be performed from midpoint to endpoint, multiplying the midpoint impact score of a category by the midpoint-to-damage characterization factor ($MDF_{d,m}$, e.g., in DALY/kgPM2.5-eq) (where DALY refers to disability-adjusted life years). Alternatively damage scores can be directly calculated from the inventory results u_i by combining the midpoint and damage characterization steps for each midpoint impact category m to yield the damage characterization factor for substance i in damage category d ($CF_{d,i}^{damage}$ in, e.g., DALY/kg_i) using the following equation:

$$S_d^{\text{damage}} = \sum_i \left(CF_{d,i}^{\text{damage}} u_i \right)$$

where

$$CF_{d,i}^{\text{damage}} = \sum_m MDF_{d,m} \times CF_{m,i}^{\text{midpoint}}$$

To calculate endpoints with the ReCiPe method, the SimaPro software multiplies the LCI results with a damage characterisation factor to calculate the potential damage to three damage endpoints (human health, ecosystems and resources).

Appendix 7: Critical review report

A7.1 The study

This study compares grocery carriers that can be used by consumers in Norway. It was conducted by three LCA researchers at the Norwegian Institute for Sustainability Research (NORSUS). The first author, Cecilia Askham, has more than 20 years of experience from LCA.

The study (including this review) is funded by the Norwegian Retailers' Environment Fund (Handelens Miljøfond), established to resolve environmental problems associated with plastics and funded by a fee of 1 NOK per plastic carrier sold at the retailers.

The report states in the Introduction that the study is not consistent with the recommendations on allocation in the international standard as interpreted in the descriptive Annex D of ISO 14044 and in the Technical Report ISO/TR 14049. However, the authors aim for the study to be scientifically valid and to adhere to the requirements of ISO 14044.

A7.2 The review

Since the study is a comparative assertion disclosed to the public, the review was conducted by an external panel:

- Tomas Ekvall, adjunct professor in Environmental Systems Analysis at Chalmers University of Technology, and consultant in Tomas Ekvall Research Review & Assessment (chairman),
- Anne-Marie Boulay, assistant professor in Chemical Engineering at Polytechnique Montréal and member of The International Reference Center for Life Cycle of Products, Processes and Services (CIRAIG),
- Ian Vázquez-Rowe, professor in Peruvian LCA & Industrial Ecology Network at Pontificia Universidad Católica del Perú,
- Rolf Erling Eriksen, environmental manager at the Norwegian Wine Monopoly (Vinmonopolet),
- Halvard Hauger, environmental manager at the grocery retail company Norgesgruppen, and
- Sjur Kvifte Nesheim, analyst on environment and society at Norwegian Retailers' Environment Fund (observer).

The panel members were selected by NORSUS and not by the panel chairman, as recommended by ISO 14044. However, Tomas Ekvall, Anne-Marie Boulay, and Ian Vázquez-Rowe are all established LCA researchers with experience from a broad range of case studies, including LCAs of carrier bags. Tomas Ekvall is an expert on modelling in life cycle inventory analysis (LCI), while Anne-Marie Boulay is an expert on life cycle impact assessment (LCIA).

Rolf Erling Eriksen, Halvard Hauger, and Sjur Kvifte Nesheim represent important actors, because Vinmonopolet, Norgesgruppen and Norwegian retailers in general make the decisions on what carrier solutions the shops will make available to consumers in Norway. The Norwegian Retailers' Environment Fund has a particular interest in the study as funding body. As indicated in the report, the participation of industrial representatives in the panel was valuable to alleviate the risk of a biased report, given the preconceptions of the authors.

The scope of the review is consistent with ISO 14044. The main aim of the review is to ensure that the methods used in the study are scientifically and technically valid, that the input data are appropriate, that the interpretations reflect the goal and limitations of the study, and that the report is transparent and consistent. In addition, the panel investigated to what extent the methods are report are consistent with the international LCA standard ISO 14044. The review does not cover the Norwegian summary, however.

The review panel was involved all through the project and gratefully acknowledges a good and constructive collaboration with the authors of the study. Three meetings were held with NORSUS to give feedback and discuss 1) the goal and scope definition, 2) the choice of input data and LCIA methods, and 3) the results of the study. The panel also gave advice on allocation, LCIA approaches, and reporting through email between and after the meetings. This meant most of the perspectives and insights of the panel could be integrated in the study.

In addition, the review panel commented on a draft version of the full LCA report. The report was then revised and extended before this final review report was written. The revision meant many of the original comments were dealt with before the report was submitted to our formal review.

A7.3 Comments

The comments of the review panel are listed below. Note that these comments are valid for the manuscript version on which the review report is based. Some or all of them might be amended before the LCA report is published.

A7.3.1 General

The study is comprehensive: it compares 18 environmental impacts of eight grocery carriers – each in a life cycle perspective. The calculations are done with two ways of modelling waste management and includes sensitivity analyses for the recycled content, the location where the carrier is produced, the carrier volume needed when shopping, and the number of times each carrier bag is used. This makes the calculations a huge task and reporting a challenge.

The authors discuss and justify the methodological choices made in the study. References are plentiful in the report. However, only a small share of the many numbers involved in the calculations are presented: a selection of the input data and a selection of LCIA results. This serves to make the most important results accessible rather than presenting an overwhelming volume of text and numbers. On the other hand, the comprehensiveness of the study and its condensed presentation make it a challenge to understand and assess how the results are produced (see also Transparency below).

A7.3.2 Background of study

The Summary and Introduction of the report states that the industry perceives plastic grocery bags in Norway to have a good quality and multifunctionality, meaning that it can be used as bin liner by Norwegian consumers. During the study, the review panel also argued that the “single-use” bags are sturdy enough to be used for carrying groceries multiple times.

A7.3.3 LCI modelling approaches

Results in the study are presented for the cut-off and net-scrap approaches to modelling recycling. This is reasonable, since they are both well established and common in the LCA community. We have not found any error in how the approaches are applied in the study. However, the presentation of them is confusing:

- The report discusses multiple types of cut-off with no clear distinction between. As a result:
  - Section 3.4.1 states “There has been no conscious cut-off in foreground data”; however, the choice of a cut-off approach is a conscious a cut-off in foreground data, if foreground data are defined as data in Chapter 5.
  - Section 3.6 indicates that Ecoinvent data based on a cut-off model was used not only in the cut-off calculations but also in the net-scrap calculations.
- Calculations with the cut-off approach seem to go under different names in different parts of the report: for example, main scenarios or s at the beginning of Section 3.5, baseline scenarios in Section 3.7.2 and Chapter 5, and Cut-off in Sections 6.1-6.2.
- Calculations with the net-scrap approach also seem to go under different names in the report: for example, system expansion in the Summary, scenario modelling, End of Life substitution and EoL at the beginning of Section 3.5, end-of-life scenario in Chapter 5, and Net scrap approach in Section 6.1.

A consistent terminology that also distinguishes between the types of cut-offs would make the report more accessible to readers.

A7.3.4 Transparency

The report includes sources with references to the input data, the characterization factors, and the conversion factors used in the calculation of end-point impacts. It also presents the methods used in the calculations and outlines the calculation procedure. This makes the study highly transparent in theory. It also meets most of the many requirements posed by ISO 14044 on a comparative assertion disclosed to the public. An important exception is that input data on important processes, such as material production, are not included in the report. These data are from the Ecoinvent database, which prohibits publication of the data.

The absence of important input data makes it difficult for the reviewers to assess them. However, the report includes an assessment of the data quality made by the LCA team, as required by ISO 14044.

The report includes separate presentation of the mid-point LCIA results for bin liners, which dominates the total results for the multi-use carriers. This contributes to explaining the results. However, other important partial results, such as the LCI results, are missing in the report. Characterization factors and end-pointy conversion factors are also absent. This makes it difficult to assess the importance of input data with low quality, to check the calculations, and to assess whether the results are reasonable. The effective transparency of the study is low.

A7.3.5 Utilization of carrier capacity

The baseline calculations assume that all carriers are filled to 50%. This contrasts with the results from the observational investigation of consumer behavior (Table 1 and Appendix 1), where the average capacity utilization varied from 30% for rucksacks and 38% for papers bags to 76% for conventional plastic bags and 98% for woven plastic bags. It is unclear how the results from the observational study affected the assumptions on the utilization of the carrier capacity.

The investigation included a small number of observations only for paper bags, but if it is correct that the average utilization of a 24 liter paper bag is 38% and the utilization of a 12 liter plastic bag is 76%, the two carriers on average hold the same amount of groceries. This indicates that they should be compared based on the results for small shopping events, where the size of the bag does not matter. The conclusions from such a comparison would be more positive for the plastic bag and less for the paper bag.

A7.3.6 The number of uses

Even more important for the results and conclusions is the number of times each carrier is used. The environmental impact of a bag is inversely proportional to the number of times it is used. This number can vary greatly between consumers and between individual bags. The reviewers appreciate the fact that the study includes a sensitivity analysis with a broad range of assumptions on the number of uses (Table 29) and also calculate how many times a carrier has to be used to be environmentally equivalent or superior to the conventional plastic bag (Tables 16, 21-22, and 26). This is important information from the study.

The conclusions largely build on calculations that assume single-use bags to be used 1 time only, and durable carriers to be used 1000 times or more. The latter is a much higher number than in previous studies. Contrary to what is common in other studies, the high number is not a pure assumption but an estimate based on a survey (Appendix 3). However, there are several reasons why the 1000 uses might still be significantly higher than the actual value for an average durable bag in Norway:

- the respondents are sustainability professionals, which means they might live more sustainably than the average Norwegian consumer;
- they responded to colleagues, which might entice them to describe their actions in a positive manner; and
- the respondents were asked about the likely lifetime of their durable bags, which means the estimate can be considered a value for the potential number of uses of the bag, rather than the actual number of uses. The latter might be significantly lower, for example because the bag is lost, or because the respondent gets more bags than necessary.

The assumption that each single-use carrier is used only once is common in LCAs. However, the review panel observes that the “single-use” bags in Norway can be used to carry groceries several times.

This makes the results from the sensitivity analysis particularly relevant. Accounting for multiple uses of “single-use” bags can reduce the LCA results by a factor two or more (see Table 29). When used five times, though, the LCA results with the cut-off approach still indicate a greater environmental impact of “single-use” bags, compared to durable carriers that are used 1000 times. It is only when the durable carriers are used few times that the ranking order changes.

A7.3.7 Cotton production

This study indicates much less environmental impacts for a cotton bag, compared to, e.g., an earlier Danish study (Bisinella et al. 2018). This study indicates that a bag produced from organic cotton needs to be used 12-13 times to have the same climate impact as a conventional plastic bag (Tables 21 and 26). From the Danish study it can be concluded that an organic cotton bag must be used at least 75 times to compete in terms of climate. This is partly because the bag in this study is lighter (125 g instead of 252 g). However, the climate impact per kg organic cotton is also four times less in this study compared to Bisinella et al. 2018. In terms of water use, the difference per kg cotton is even greater: more than a factor 15.

The difference in results is likely due to different sets of input data on the production of organic cotton textile. Both studies apply data from the Ecoinvent database in this part. The Danish study modelled organic cotton production by subtracting fertilizers production data from a consequential data set on conventional cotton production and by lowering the yield by 30 % (Bisinella et al. 2018, Section 4.3.7). This study, in contrast, used an attributional data set specifically on rain-fed organic cotton production and with low land use change-related emissions. This indicates that the data in the present study are more accurate; however, the climate impact, associated land use changes and water use probably depend on the irrigation system, which varies greatly between producers.

In conclusion, the results on cotton bags appears to be valid for relatively light, rain-fed organic cotton but not for cotton bags in general or for irrigated organic cotton.

A7.3.8 Life cycle impact assessment

The selection of assessment methods in LCIA is sound and the report includes a clear description of ReCiPe, the main assessment framework used. The justification for the use of this framework is valid. Midpoint impact categories are listed and presented with some detail in Section 3.7. The inclusion of a hierarchist perspective for endpoint modelling is also justified. However, a table similar to Table 4 with the endpoint aggregated categories would be good to include in the report.

The reasons for using AWARE instead of ReCiPe for water consumption are reasonable and valid. Similar arguments could be made for using IPCC for global warming and Usetox for toxicity, instead of the ReCiPe approaches. However, an important difference is that ReCiPe includes toxicity and GHG emissions but not water use in endpoint calculations. Hence, despite available alternatives, using ReCiPe for global warming and toxicity makes sense, as it makes the mid-point and end-point results consistent.

Although midpoint and endpoint results are complementary, they should not be seen as independent because they come from the same life cycle model. Endpoint results are derived from the midpoint results, which constitutes a translation into damage. This information, including the utility of presenting one or the other (or both) is not specified in the report, an issue that could hinder the understanding of the results by unexperienced stakeholders. An additional point of confusion is the order in which the results are presented: a lot of focus is given to midpoints in Section 3.7, but Chapter 6 presents the results in terms of endpoint first and then midpoints.

A7.3.9 Littering

Marine littering is presented as part of the LCIA Section 3.7. However, much of Subsection 3.7.1 is about estimating the quantity of plastic lost (released) to the ocean per FU. This is a physical flow rather than an impact and, hence, fits better in the description of the LCI. Section 3.7.1 should be revised to avoid mixing LCI and LCIA concepts, with those linked to LCI moved to another section.

The equations below Table 20 (in the results section on littering with the cut-off approach) are, on the other hand, general for LCIA in general. They fit better in Section 3.7. The equation for the characterization factor is over-simplified, since it does not include an exposure factor. References are also missing for these equations.

The report in some places states that plastic littering is only considered for the consumption phase. This is not quite accurate. The littering is only considered for the carrier bags themselves; this excludes other things that can occur in the use phase, for example that the carrier can break causing the groceries to contribute to littering.

The unit for units of marine litter impacts (rather than effects) should be PAF-m3-day as the units of the fate factor (even though value is one) should be in time units, or more correctly in (kg ending up in the compartment)/(kg emitted per day).

Litter impacts does not seem to be included in the end-point calculations. A method for end-point assessment of littering has been published. This involves converting PAF to PDF using existing conversion factors.

A7.3.10 Calculations and results

Most of the calculations could not be checked, since important input data, LCI results, characterization factors, end-point conversion factors, etc. are not included in the report.

However, significant errors have been made in the calculation of the number of times a carrier must be used to be environmentally equivalent to the conventional LDPE bag. For example, Figures 7-9 indicate a higher impact for 1 HDPE bag compared to 1 LDPE bag, but Table 16 states that the HDPE bag must be used only 0.6 times to be equivalent or better in all environmental aspects. All numbers on human-health impacts in Table 16 seem to be approximately a factor 10^6 too low. The results on ecosystem damage impacts are also too low – most evident for the HDPE bag, the paper bag and the cardboard box.

When the net-scrap approach is applied on multiuse carriers, the results are negative for many impacts (see Figures 16-17 and Table 24):

- overall ecosystem damage,
- overall resource depletion,
- stratospheric ozone depletion,
- tropospheric ozone formation,
- fine particulate formation,
- terrestrial acidification,
- terrestrial, marine and freshwater ecotoxicity,
- human non-carcinogenic toxicity,
- land use, and
- mineral resource depletion.

The negative results are generated because the net-scrap approach indicates less than zero environmental impacts for the bin liners (Table 25). This, in turn, is probably because when bin liners are incinerated together with the household waste, the net-scrap approach gives a credit for the impacts avoided when heat from the incineration displaces Norwegian district heating mix. The negative results indicate that the impacts of the Norwegian district heating mix are worse than the combined impacts of production, transport and combustion of the bin liner. This is reasonable for some impacts: for example, the bin liners might be associated with less land use than the biofuel in the district heating systems. Most of the negative results might instead have other explanations, however: data gaps, inconsistent measurement methods, variability between processes, calculation errors, etc. A deeper analysis is required to find the explanations. Meanwhile, the negative results should not be interpreted as proof that the use of bin liners are good for ecosystems, resource depletion, or the environment in general.

The sensitivity analysis on recycled content indicates that an increased use of recycled fibers in paper bags and cardboard boxes increases their impact on climate, particulate formation and overall human health (Figure 24). The report states that further analysis is required on the validity of the input data used in this part of the analysis.

The above are all examples of the fact that numerical LCA results should not be accepted at face value.

The units and indicators of water use are inconsistent between Tables 18, 19, 24 and 25. We suggest the indicator be called “water deprivation”. The unit should be m^3 world eq.

To be consistent with other figures and avoid misleading the reader to think the transport distance is very important for the ecosystem damage, the y axis in Figure 34 should start at zero and not at 5.0E-09.

A7.3.11 Conclusions in the report

The report concludes that multiuse carriers are environmentally preferable to single-use carriers. With the input data selected this conclusion appears to be robust if each multiuse carrier is used 1000 times, as assumed in this study, but does not hold if a multiuse carrier is used a few times only. A more accurate conclusion is to say that the results indicate that multiuse carriers are environmentally preferable, if they are used many times (cf. Tables 16, 21-22, 26, and 29). This is a significant modification, since behavior varies between consumers, as stated in Appendix 3. Multiuse carriers that are mainly hoarded in cupboards etc. are bad for the environment, as indicated in the discussion chapter.

Litter impacts can be an important part of the ecosystem damage. Readers can be confused when conclusions on littering are presented separately from the conclusions on ecosystem damage. Clarity is improved if “ecosystem damage” is replaced by “ecosystem damage (excl. litter impacts)” in the Summary and possible also in Chapter 8 Conclusions.

A7.3.12 Recommendations in the report

The report includes several recommendations relevant for the retail sector. The panel wants to add to or modify some of these:

- The Conclusions and Summary includes the recommendation to stimulate the use of bags already owned. This recommendation is valid.
- The Discussion includes a suggestion to encourage consumers not to use a carrier when buying a few items only. This recommendation is valid only when there is no significant risk of dropping and breaking the items. The environmental impacts of producing a bag are typically small compared to the impacts of producing the groceries.
- The Discussion also indicates that a transition from single-use to multiuse carriers brings large environmental benefits. This is valid only if the transition does not lead to an excess production of multiuse carriers. Giving multiuse carriers away or selling them at a low price might cause such an excess production and increase the environmental impacts of the system. It is important to identify and assess measures that stimulate a transition to multiuse carriers while avoiding excess production of these.

NORSUS
Norwegian Institute for Sustainability Research

The vision of NORSUS Norwegian Institute for Sustainability Research (formerly Ostfold Research) is to provide knowledge for sustainable societal development. We apply and develop knowledge and methods to understand and implement sustainable solutions in society. Together with a wide range of public and private clients, we undertake projects locally, nationally and internationally to enhance environmental performance, often also generating economic benefits.