

An economic-environmental analysis of selected barrier-coating materials used in packaging food products: a Swedish case study

G. Venkatesh¹ · Åsa Nyflött^{1,2} · Chris Bonnerup² · Magnus Lestelius¹

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Abstract The purpose of a barrier coating in food packaging is primarily to increase the shelf life of the foodstuff contained within the packaging, preserve its colour, odour, taste and quality, and thereby reduce food wastage (both at retail outlets and households). While most publications hitherto have compared packaging and barrier-coating materials on the basis of their environmental impacts alone, this paper adopts a more holistic approach by factoring in the economic aspect as well. Four barrier material alternatives—starch, polyethylene, EVOH + kaolin and latex + kaolin are analysed. Two well-defined end-of-life handling scenarios, relevant to Sweden, are: one in which everything except starch is recycled, with starch being composted, and the other in which everything is incinerated. Among the several environmental impact categories which can be analysed, this paper considers only global warming. Two approaches are tested to combine the economic and environmental aspects—normalisation, weighting and aggregating on the one hand, and using the carbon tax to internalise the externality caused by GHG emissions on the other. For the set of weighting factors obtained thanks to a survey conducted by the authors (40.6% for environmental and 59.4% for economic), starch emerges as the most sustainable alternative, followed by polyethylene for both the end-of-life handling scenarios. This tallies with the result obtained by using the carbon tax for internalisation of the externality. The case study, methodology and results presented in this paper, will hopefully be a springboard for more detailed studies of this nature, under the umbrella of sustainability.

✉ G. Venkatesh
venkatesh.govindarajan@kau.se
Åsa Nyflött
asa.nyflott@kau.se
Chris Bonnerup
chris.bonnerup@storaenso.com
Magnus Lestelius
magnus.lestelius@kau.se

¹ Department of Chemical and Engineering Sciences, Karlstad University, Karlstad, Sweden

² Stora Enso Karlstad Research Centre, Karlstad, Sweden

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1 Introduction

The food and beverages industry cannot do without the products and services supplied by the packaging sector (Venkatesh 2016). The packaging sector has its own supplier base, which includes, *inter alia*, firms manufacturing barrier coatings. The purpose of a barrier coating is primarily to increase the shelf life of the foodstuff contained within the packaging, preserve its colour, odour, taste and quality, and thereby reduce food wastage (both at retail outlets and households). A barrier coating accomplishes this by shielding the foodstuff from oxygen (which is detrimental especially if the foodstuff contains lipids) and water vapour. Barriers also successfully inhibit carbon dioxide loss from beer and carbonated soft drinks—a requirement for these beverages. It must also be mentioned at this juncture that barrier packaging reduces the need for preservatives. Barrier coatings enter into the life cycle of the ‘packaging + foodstuff’, having caused environmental impacts during the upstream processes, and are functional all through the use phase. When the packaging has to be handled at the end of its useful lifetime, the barrier coating in it could end up with different destinies such as composting, incineration, landfilling, recycling (downcycling usually) or anaerobic co-digestion to generate biogas.

Among the most desirable properties of food packaging materials in general, are ease of handling (for producer, transporter, retailer and consumer) and recyclability. Packaging materials and barrier coatings are often made of polymeric materials that have these properties, and they are often filled with impermeable inorganic fillers to improve their barrier performance. Polymer-based composite materials can be produced on an industrial scale, by dispersion, inter-lamellar polymerisation or melting, conferring enhanced barrier properties, better mechanical properties and thermal stability. Currently, research is still underway to further improve the performance of the composites and their stability. In addition to the performance, ease of handling and recyclability, renewability and biodegradability are increasingly being considered to be extremely desirable characteristics of packaging materials. The driving force behind the last two sought-after characteristics is the indisputable requirement of this century—to minimise the adverse environmental impacts; be that climate change or other impact categories.

Barrier packaging with added convenience and smart/intelligent features not only will remain relevant in the economically developed markets but will also become more widespread in the developing markets. Also, with consumers increasingly interested in environmental concerns, the need to make packaging more sustainable continues to be a key driver of innovation; and bioplastics will continue to grow in market acceptance though petrochemical-based polymers may be the dominant materials used in packaging for many years to come (Savinov 2014). Of course, one may often wonder how not having any packaging at all figures in the scheme of things. In Pagani et al. (2015), the authors have concluded that packed salads require an elevated energy input ranging from 16 to 37 MJ per kilogram packed. By comparison, unpacked salads require just one-tenth of this, while locally sold produce (at farmers’ markets for instance) demands even less, as they do not need processing, refrigeration and disposable packages. However, one would need to

think in terms of the entire product system (food plus packaging) in such cases and factor in the role played by packaging in reducing food waste.

In this paper, a case study with Karlstad in south-western Sweden was carried out to determine the economic and environmental impacts of three selected barrier coatings that are used in food packaging materials. The analysis was on the 'end-use' and 'end-of-life' handling of these options/combinations of barrier coatings in the food packaging: (1) starch; (2) latex + kaolin (50% by mass for each); (3) ethyl vinyl alcohol copolymer + kaolin (50% by mass for each) and (4) fossil-sourced polyethylene as a reference. Ideally, one would analyse a host of environmental impact categories, but in this paper, the authors have restricted themselves to global warming caused by emissions of greenhouse gases (which leads to climate change and concomitant challenges). There is no reason for that matter, to exclude the other environmental impact categories, other than the fact that global warming due to the emission of greenhouse gases is the one which has been meriting the greatest attention these days. Further, if the other environmental impact categories are considered, one is confronted by the challenge of deciding the relative importance of each of those in environmental decision-making, which the authors wished to consider as being beyond the scope of this paper. The authors are of the view that environmental life-cycle analyses alone cannot and should not support decision-making in the real world. Economics, functionality and social acceptance/approval are also important. However, the relative degrees of importance of all these aspects are not the same—temporally and geographically, which makes decision-making all the more complex, non-standard and thereby interesting. For instance, as gathered from Innventia AB (2016), in an International Innventia Consumer Survey, bio-based materials were seen as far less 'reliable' than many other materials, reliability being regarded as the most important property. Thus, if 'social acceptance' is factored in as a criterion, it may even end up trumping environment friendliness.

2 Literature review

There are many studies that have analysed the role of barrier-coating materials on food packaging materials in the last decade. Chan et al. (2011), in a study on aseptic liquid packaging, have compared polyamide with aluminium as barrier material and concluded that the former is superior to the latter, when it comes to environmental performance. This, the authors have pointed out is because of the difficulty in separating and recycling the aluminium barrier coating from the primary packaging material. However, the authors have not included the use phase of the packaging in their analysis and consider it necessary to also determine whether polyamide is as effective as aluminium in extending the shelf life of the foodstuff it is expected to preserve. While separation of aluminium from polyethylene is difficult, the composite can still be recycled in the manufacture of roof shingles, granules and other products as observed by Varžinskas et al. (2012). To date, about 34% of aluminium-PE composite waste is recycled in Europe, with 30% being incinerated and the remaining landfilled. In another study on aseptic packaging, Xie et al. (2013) have stated that in 2010, in China, 2500 tons of composite packaging waste were generated daily and consigned to sanitary landfills, owing to the absence of efficient separation and recovery systems in that country. In the life-cycle analysis (LCA) study carried out in the said paper, the authors have concluded that separating aluminium from polyethylene (in the composite packaging) will reduce environmental impacts by almost

13% with respect to the incineration scenario and, at the same time, enable the recovery of resources for their material values, which does not occur in landfilling.

Aluminium was regarded as a perfect oxygen and moisture barrier in coffee capsules (Krones et al. 2011). Thus, it becomes clear that any LCA of packaging materials (including barrier coatings) is incomplete and misleading if the effect on shelf life and therefore food spoilage/wastage is not factored in (Castelanelli et al. 2011; Wikström et al. 2016). It is precisely the difficulty in separating materials from composites that prompts Toniolo et al. (2013) to recommend the use of mono-material packaging wherever possible, to tide over the impasse. Silvenius et al. (2014) have shown in a comparative LCA for ham, dark bread and a fermented soy-based drink in Finland, that while the contribution of the life cycle of the packaging (production and waste handling) per se to the environmental footprint of the entire product system (food + packaging) would not be more than 15%, that of household food waste could be high as 26%. The role of packaging materials in preventing/reducing food waste is highlighted in Manfredi et al. (2015). In this paper, the authors have contended that the environmental impacts reduced by avoiding wastage of milk are far greater than those caused by introducing an anti-microbial coating (a synthetic derivative of lauric acid) to milk packaging.

According to Kliugaite and Staniskis (2013), a paper in which three barrier alternatives have been compared, with the system boundary being set around raw material extraction, adhesives, printing ink and plastic polymers production and multilayer high-barrier plastic package manufacturing, using a multi-layered film (including a barrier layer) is not advisable owing to poor or no recyclability—in agreement with Toniolo et al. (2013). However, as mentioned earlier, if this disadvantage more than offsets the life-cycle environmental impacts associated with the food waste that would have happened otherwise, a more holistic outlook (a proper systems approach) would perhaps not prioritise the environmental performance of the packaging per se. When comparisons are made among packaging options, this point has to be borne in mind. Grönman et al. (2013) has advocated a ‘consideration of the properties of both the package and the product itself when designing the final package, in order to reduce both product losses and environmental impacts’. Azzi et al. (2012) have written that the impact of packaging design on supply chain costs and performance can be devastating. Whitworth (2013) writes that barrier-coating technologies have helped to reduce packaging waste and allow recycling efforts by replacing multilayer film structures.

While a replacement of conventional, petroleum-based (or non-renewable abiotic material-based) packaging with renewable biopolymers (or bioplastics) is touted as an environment-friendly strategy to adopt, Garrain et al. (2007) would like to advise some circumspection, by stating that ‘extracting a definite conclusion about the environmental efficiency of biodegradable polymers compared to conventional products is no easy task’—all that glitters, is not gold. Vidal et al. (2007), in a comparison of biodegradable modified starch-poly(lactic acid) barrier film and petroleum-derived polypropylene-polyamide 6 film, have shown that the latter causes less acidification and eutrophication vis-à-vis the former, while having a global warming potential over 100 times that of the biodegradable combination. In a comparative LCA of polystyrene and recycled paper as egg-packaging materials, Zabaniotou and Kassidi (2003) found out that the latter was responsible for higher environmental impacts when it came to emissions of heavy metals and carcinogenic substances, even though, on the whole, it emerged as an environmentally superior alternative than polystyrene. Bayus et al. (2016) have shown in a comparative LCA of barrier film material combinations—aluminium foil, metallised oriented polypropylene and metallised oriented polyethylene terephthalate—that the global warming potential of the

metal-polymer composites is half that of the aluminium foil laminate, while non-renewable embodied energy in the films is reduced by 25% when the metal-polymer composites are chosen over the aluminium foil. In another comparative LCA study conducted for three types of cheese packaging—100% polypropylene, tin and polyethylene (PE), paper carton and PE—Banar and Cokaygil (2009) concluded that the paper carton/PE alternative is environmentally most preferable, while the tin/PE option has the largest environmental footprint among the three. However, the source of the polymers considered in both Bayus et al. (2016) and Banar and Cokaygil (2009) is still fossil oil.

Unless one sets priorities regarding the environmental impact categories, an overall comparison becomes difficult. Further, if landfilled, the biodegradable film, owing to the emission of methane gas, courtesy anaerobic microbial activity, causes 180 times greater global warming than the non-biodegradable petroleum-based plastics. Razza et al. (2015) have compared expanded polystyrene (EPS) packaging systems to a bio-based, biodegradable starch-based prototype (both to be used for protecting washing machines during transport and storage) and concluded that despite its higher density, the bio-based alternative has a carbon footprint 40% smaller than EPS. However, even as one digests the findings and conclusions made by the authors of all these publications, it must always be remembered that even though consumers are concerned about the (environmental) sustainability of products they buy, they are not always willing to pay more for them (Savinov 2014).

Whitworth (2013) had written that nanocomposite barrier coatings were in their infancy and much costlier than the conventional alternatives. At the time of writing (in 2016) though, nanocomposites can be said to hold great promise in flexible packaging, in the years to come, with optimisation likely to bring down the cost and thereby make the nanocomposites competitive with the alternatives. Lorite et al. (2017) have shown that polylactic-acid-based nanocomposite active packaging, vis-à-vis polyethylene terephthalate (PET), can extend the shelf life of the food product, thus reducing food wastage, while being environmentally as good a performer as PET.

3 Methodology

3.1 Processes included and system boundaries

Figure 1 charts out the processes in the life cycle, which have been considered in this LCA. It at once becomes evident that what has been carried out in this paper is a 'partial LCA'. The journey from filling (the package with the food product) to the final disposal for end-of-life handling has not been studied. It must also be pointed out at this juncture that barrier film production and incorporation into primary packaging may be integrated processes not involving inter-stage transportation as shown in the diagram.

3.2 Data sources

The data sources for the different materials which have been studied in this paper and have been listed as follows:

- (a) Starch—Processing data obtained from Avebe, Veendam (the Netherlands). The source of the starch could vary, and hence, the upstream impacts of cultivating the

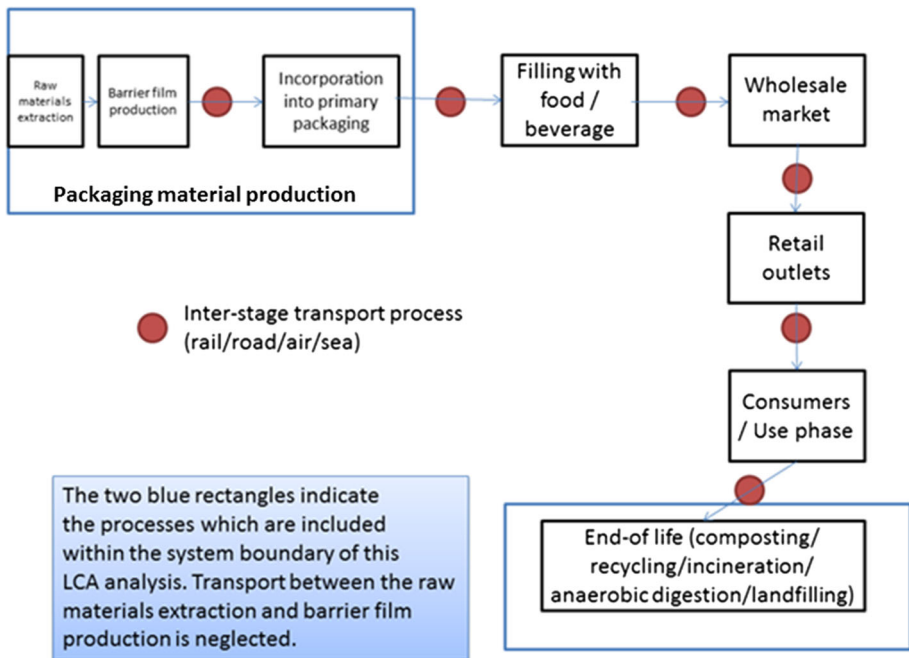


Fig. 1 The processes in the life cycle which have been considered in this partial LCA

source crops have not been considered. One may assume that starch is sourced from plant materials which would otherwise be considered wastes.

- (b) EVOH (ethyl vinyl alcohol copolymer) from Germany—*ecoinvent* database (Swiss Centre for Life-cycle Inventories, 2016).
- (c) Kaolin—*ecoinvent* database (Swiss Centre for Life-cycle Inventories 2016), sourced from either the USA or UK.
- (d) Latex—Production data from CH Polymers, Finland.
- (e) Polyethylene (PE)—Data obtained from Stora Enso of barrier film production; production in Sweden.

While PE is deposited on the substrate packaging material by the extrusion coating method, the others are deposited using the dispersion coating method. The barrier performance of the materials differs slightly, though they are fairly similar to each other, with respect to dry food or frozen food. Fillers are commonly used in order to make the pathways for the penetrants more tortuous. In this study, kaolin was chosen as the filler. PE is used as a reference case in this analysis, owing to the fact that it is very commonly used as a barrier in dry food packaging. EVOH and latex are also in vogue currently, with paperboard packaging. Starch holds great potential to be used widely as a barrier coating, but it is relatively more sensitive to moisture and more research is needed to study its functionality in greater detail.

As global warming is the only environmental impact category considered, it is only the energy use (heating, electricity and transportation) during the stages considered in the analysis, which is of concern. The databases from *ecoinvent* that have been availed of and are listed in “Appendix 1”. The authors were asked by the providers not to publish the received primary data. The economic aspects that are studied in this paper are restricted to

the production costs. “Appendix 2” represents the unit production costs of the different barrier materials that were obtained from their respective suppliers.

4 Assumptions

Environmental life-cycle analyses—or E-LCAs—cannot be carried out without making suitable assumptions along the way. These assumptions may perhaps detract a little from the accuracy of the final results, while enabling comprehensiveness. While awareness of this fact is very much necessary, analysts can, of course, carry out ‘*What-If*’ (sensitivity) analyses, to tide over this unavoidable impasse.

In this paper, a very specific case has been considered. Karlstad is the end-use and end-of-life handling location, for the barrier-coating materials imported from Germany, Finland and the Netherlands. Starch and EVOH are transported by truck from the Netherlands (1157 km) and Germany (1569 km), respectively. Latex is transported from Finland, first over land within Finland (300 km), then across the Gulf of Bothnia (317 km) into Sweden and thereafter by truck within Sweden (311 km). Within Sweden, polyethylene is transported by truck over a relatively shorter distance of 18 km. Kaolin comes either from the UK (in which case, the transport entails 1134 km by sea, and 249 km within Sweden by truck), or from the USA (in which case, transport entails 7221 km across the Atlantic and through the Baltic Sea, and a total of 494 km by truck in the USA and Sweden). Figure 2 depicts the source–destination details (though not the transport distances) for the different materials.

The emissions related to the upstream transportation of raw materials to the manufacturers producing the barrier-coating materials have not been considered in this analysis, as accurate data were not available to the authors at the time of this analysis. The drying process of the barrier coating consumes energy. The specific energy demand in this process is assumed to be 4 GJ/ton of the barrier-coating substance, considering the following dry content percentages—30% for starch, 15% for EVOH + kaolin and 50% for latex + kaolin (this is based on experimental data obtained from Bonnerup, one of the co-authors, from trials conducted in Finland, for the latex + kaolin combination. In this analysis, this is assumed to hold for the Swedish context, and also for the other alternatives). The German, Dutch, Finnish and Swedish electricity mixes are used for the respective cases. The Nordic electricity mix has not been considered for Sweden and Finland, though it must be mentioned that this could be done as part of a sensitivity analysis. Incineration for heat recovery, composting and recycling is considered as the three end-of-life handling options for the barrier-coating materials. In all these cases, it is assumed that the primary packaging to which the barrier coating is applied is the same, and the combination is used for the same application—in other words, for the same foodstuff. Further, we also make an assumption (which of course needs further studies to vet, accept or oppose) that the functionality per unit mass of the barrier coating applied remains the same in all cases. However, the authors agree that this may not really be the case, if a closer examination is carried out, as regards the shelf life of the foodstuff and any deterioration in its quality, over time.

Two different scenarios have been tested—all the wastes being recycled (latex and kaolin recycled along with the plastics as additives in the resulting recycled material), with starch being composted, and all the wastes including the starch being incinerated. Starch—a polymeric carbohydrate—is purely of biological origin and incinerating it releases

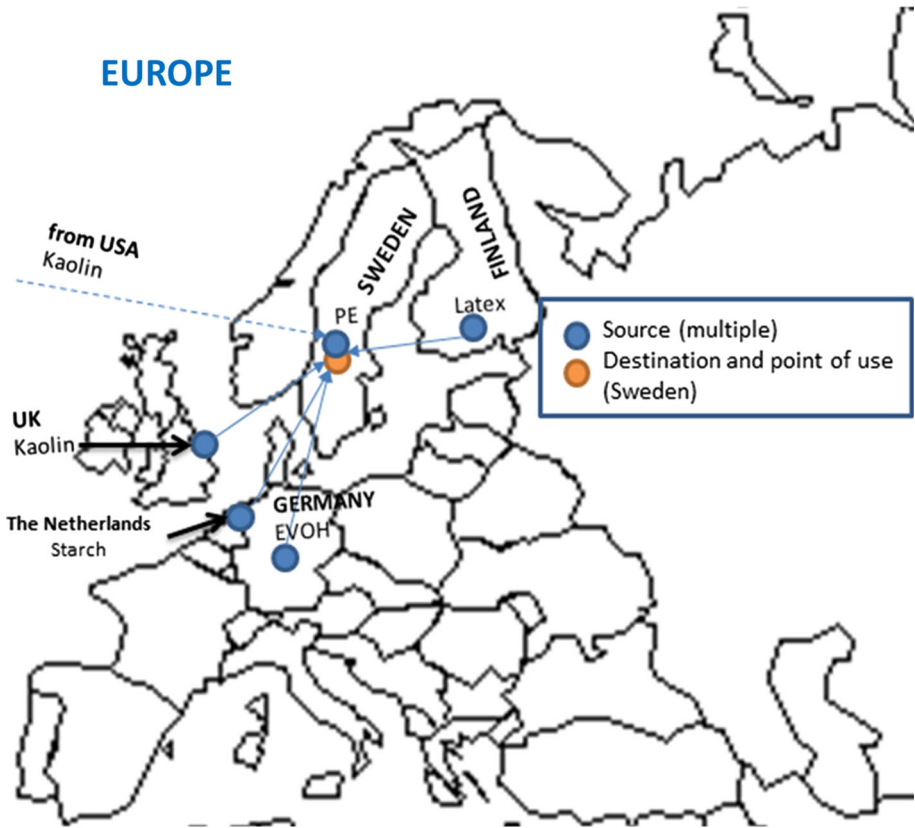


Fig. 2 Source destination for the different barrier film materials

biogenic carbon dioxide (if complete combustion in the incinerator is assumed). Likewise, aerobic composting of starch may release some of the carbon in the starch as biogenic carbon dioxide, while providing most of it as a carbon source to the soil. It thereby does not play any role in the avoidance of production of chemical fertilisers needed to supply nitrogen, phosphorus and potassium. Thereby, the end-of-life handling of starch can be assumed to be perfectly carbon neutral. Anaerobic digestion and landfilling are indicated in Fig. 1 as possible end-of-life handling options; but in this paper, as the geographical scope is limited to Sweden where recycling and incineration are the prevalent end-of-life handling options, those two have not been considered. As far as environmental impact categories are concerned, the analysis in this paper is restricted to global warming. This restriction is, admittedly, a limitation of this paper.

4.1 Tools and techniques

The authors identified a list of individuals in the academia, research institutes and industry, experts in the fields of sustainability/packaging/forestry sector, and contacted them via email with this simple question: *If you had to make a decision on selecting a barrier coating material for food packaging, how would you weight the economic and environmental aspects of the same? (The sum of the two weighting factors must be equal to 100%.*

We assume that there are no other aspects/criteria to be considered in this case). Seventeen experts were identified within Sweden and Norway and contacted over the email. Some declined to assign weighting factors citing the complexity of the issue of weighting as the reason. Nine sets of weighting factors were obtained for the purpose of this analysis—five from the industry, two from academic research (Swedish universities) and two from business-oriented research (from Norway and Sweden). Six of the nine respondents are women. The results were collated, and the weighting factors assigned by the respondents were averaged for use in the last step of this analysis (refer “Appendix 3” for a list of the weighting factors; the respondents have not been directly identified in the table). The average for the economic aspect is 59.4% and that for the environmental 40.6%.

The values (both for GHG emissions and the production cost) were normalised with respect to polyethylene. The normalised values were then multiplied by the averaged weighting factors, and the products were summed up to obtain a single score (Refer Eq. 1). For polyethylene, the reference material, this score is 1.

For GHG emissions, normalised value for alternative ‘i’ would be

$$N_{\text{GW},i} = \text{GHG}_i / \text{GHG}_{\text{PE}} \quad (1)$$

For production cost, normalised value for alternative ‘i’ would be

$$N_{\text{COST},i} = \text{COST}_i / \text{COST}_{\text{PE}} \quad (2)$$

If weighting factors are denoted in % by W_{GW} and W_{COST} ,

Final score is:

$$(N_{\text{GW},i} * W_{\text{GW}} + N_{\text{COST},i} * W_{\text{COST}}) * 0.01 \quad (3)$$

The scores obtained for the other alternatives can thus be compared directly with the reference and also with each other. The lower the score, the more sustainable the alternative, as one would like to lower both the GHG emissions and the production cost, in order to impart a higher degree of sustainability to the process/product. There are variants to this method which was reported by Venkatesh et al. (2015) that compared alternative water treatment approaches from economic and environmental perspectives. Similarly, Venkatesh (2015) calculated sustainability indices by starting with different criteria and indicators that will better help decision-makers to measure sustainable development. The weighting factor-sets for {Economic, Environmental} range from {0,100} to {100,0}, the extremities representing the ‘radical-green environmentalists’ and the ‘climate-change-denying-capitalists’. Plotting the scores for this entire range of weighting factors as a line graph for all the barrier-coating alternatives passes for a sensitivity analysis (one can determine the rankings directly from such a plot for any set of weighting factors one may arrive at).

If one seeks to avoid the weighting factor approach to aggregation, the life-cycle (or partial life cycle in this case) GHG emissions can be converted into monetary units by simple multiplication with the carbon price in vogue in the region/country, and added on to the production cost, thereby internalizing the GHG-emission externality. Then, one could simply compare the ‘total cost’ (or the so-called social cost) of the different alternatives. In this paper, this approach has also been tested, by using a carbon dioxide tax of 0.3 Euro (3 SEK approximately) per kg CO₂-eq emitted (Skatteverket 2016). Intuitively, it becomes clear that as long as the contribution of the carbon tax to the ‘total cost’ is less than 50%, the economic aspect (the production cost in other words) is indirectly assigned a greater

weighting factor than the environmental (here, just the GHG emissions). However, if this intuitive reasoning is adopted, it also follows that the weighting factors are usually not the same for all the alternatives.

5 Results and Discussion

In Fig. 3a, b, polyethylene is considered to be the reference case. Figure 3a is related to recycling and composting of starch as the end-of-life approaches adopted, while Fig. 3b is related to incineration. The aggregated value (normalised cost*weighting factor for the economic aspect + normalised GHG * weighting factor for global warming), which has been plotted along the Y-axis of both the graphs, is equal to one for polyethylene for the entire range of weighting factors, as it is considered to be the reference case. These graphs can be looked upon as sensitivity analysis plots—considering that the ranking is very sensitive to the weighting factors one would choose for the analysis. As one moves from left to right along the X-axis—in both Fig. 3a, b—the weighting factor for the environmental aspect (global warming to be more specific) increases from 0 to 100%, while that for the economic aspect decreases from 100 to 0%. The vertical dotted line in both the graphs indicates the average of the weighting factors obtained from the experts contacted (“Appendix 3”) The set of weighting factors the authors of this papers arrived at, after averaging the responses from nine experts from Sweden and Norway (“Appendix 3”), was {40.6, 59.4%} for the environmental and economic aspects, respectively. The ranking for this particular set can be obtained from Fig. 3a, b for the two end-of-life handling scenarios.

Figure 4 bypasses the need for weighting factors—which makes the final ranking quite subjective—and uses the prevalent carbon tax value to convert the GHG emissions into money units and thereby obtaining a value for the ‘total cost’ which includes the externality caused by the GHG emissions. The ranking in this case, for both recycling/composting and incineration scenarios, would be thus:

- Best option: Starch
- Second best: PE
- Third best: Latex + kaolin from the UK
- Fourth best: Latex + kaolin from the USA
- Fifth best : EVOH + kaolin from the UK
- Worst alternative: EVOH + kaolin from the USA

The benefits of recycling over incineration are also evident from Fig. 4. This ranking matches the one obtained using the average weighting factors. As mentioned in Methodology section, one effectively (indirectly) assigns different weighting factors to the different alternatives to obtain the final score. A costly product with a smaller carbon footprint would have a higher weighting factor for the economic aspect, compared to a cheaper product with a larger carbon footprint. This would simply mean that producers with higher production costs are motivated to reduce the same, while producers whose products have higher carbon footprints are driven towards truncating them. The driver or the motivating factor here is simply the need to compete sustainably. However, the rules of competition ought to change in order to promote sustainability thinking.

6 Conclusions and recommendations

While most publications hitherto have compared packaging and barrier-coating materials on the basis of their environmental impacts alone, this paper has adopted a more holistic approach by factoring in the economic aspect as well. However, the assumption that the functionality of all the alternatives is more or less the same, need to be investigated more closely, as there may be some observable differences in the quality of the foodstuffs

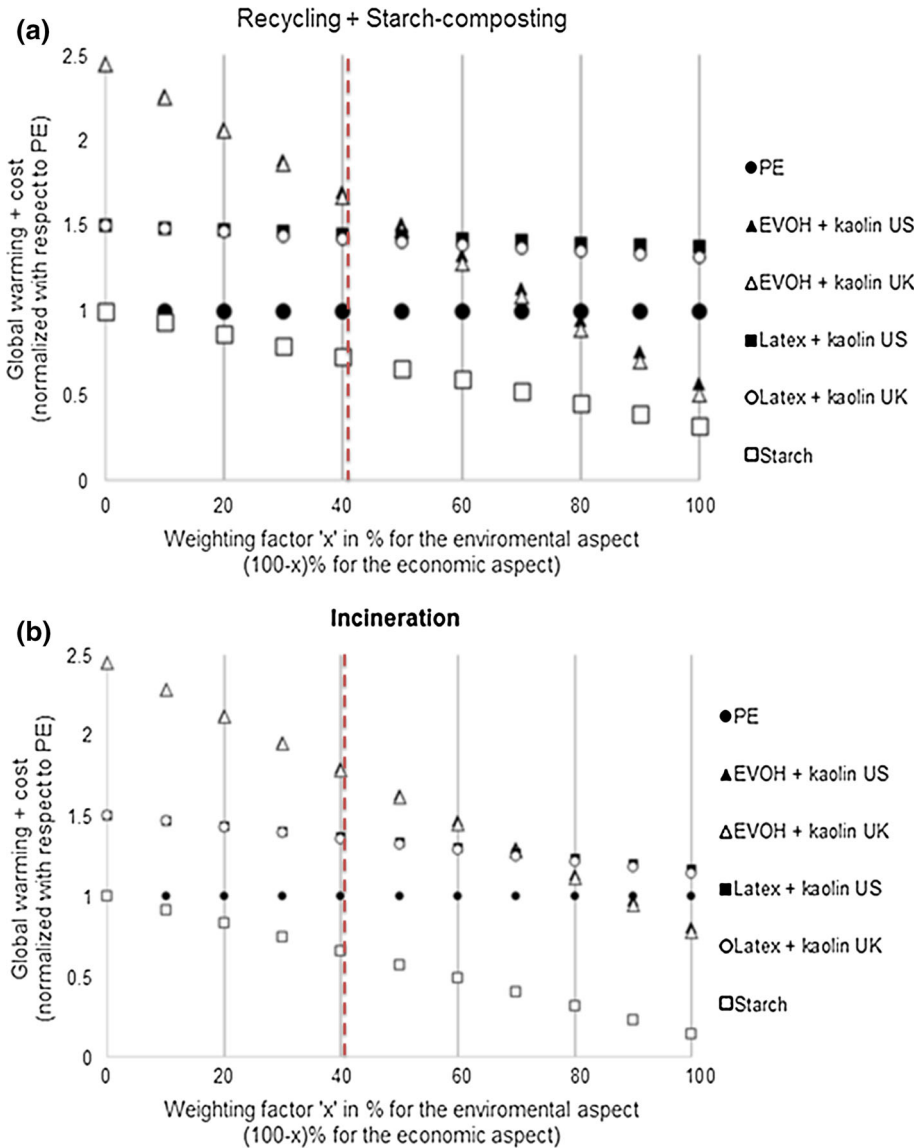


Fig. 3 **a** Sensitivity of ranking of the alternatives to weighting factors chosen in the scenario in which starch is composted and the other materials are recycled, **b** Sensitivity of ranking of the alternatives to weighting factors chosen in the scenario in which all the materials are incinerated

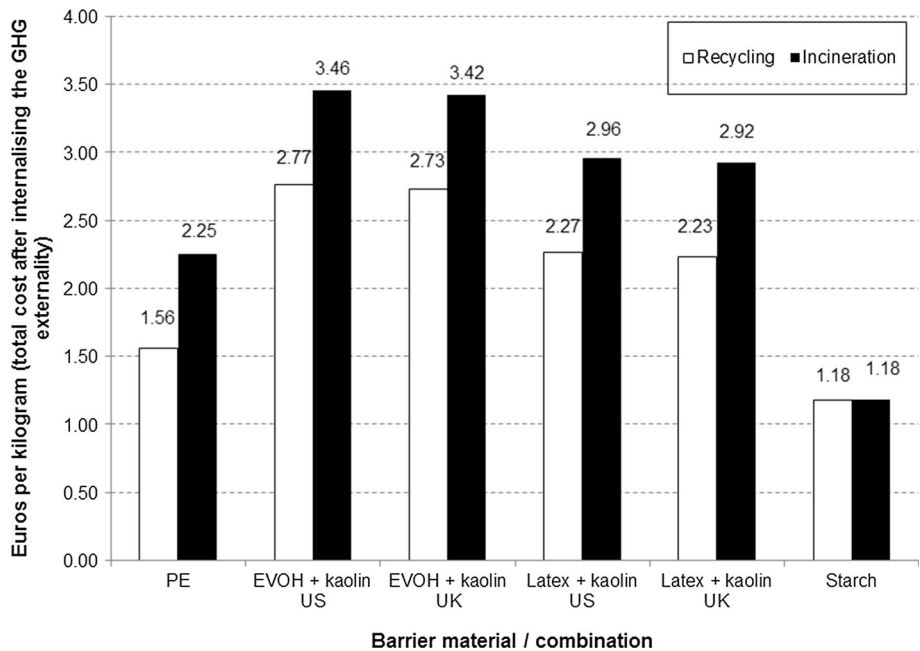


Fig. 4 Aggregated score after considering the carbon tax and expressing GHG emissions in monetary units—internalizing the externality

packaged over a given shelf life. The fact that there are differences in the physicochemical properties of the different materials cannot be overlooked. The number of criteria (and indicators under each of them) considered for the comparison can be increased if analysts have recourse to reliable and accurate data and information.

In this paper, we have considered two well-defined scenarios—one in which everything except starch is recycled and the other in which everything is incinerated. It is quite possible in a real case, where all these barrier materials are in vogue, that some may be recycled while some others may be incinerated, necessitating adjustments in the analysis. The specific energy consumption for the drying process—4 GJ/ton of the barrier-coating substance—has been assumed to be hold for all the alternatives considered in this analysis, though this was the value obtained empirically only for the kaolin + latex combination. Experimental trials may need to carry out separately for the different alternatives in order to determine the divergence of the specific value from the one which has been used in the calculations.

Among the several environmental impact categories that can be analysed, this paper considers only global warming. Most publications in this field—some of which have been cited in Literature Review section—have considered a wider gamut of categories, which makes the analysis more interesting and comprehensive, at the expense of some accuracy. In this paper, as a proxy for the economic aspect, only the production cost has been taken into consideration. For a more detailed analysis, costs incurred throughout the life cycle (including transport and end-of-life handling) need to be considered. While internalizing the GHG externality, a carbon tax of 0.3 Euro per kg-CO₂-eq was considered in this paper. The rankings obtained from Fig. 4 are sensitive to the carbon tax.

Also to be emphasised is the fact that total recycling of all the materials in barrier coatings may not happen in reality owing to institutional and technological hurdles and

lock-ins. But this just means that there are challenges to be overcome to make it possible. A circular material flow, according to Innventia AB (2016), is ‘*indeed possible to achieve*’. However, if incineration as an end-of-life handling approach has entrenched itself to contribute to district heating, the consequences of diverting materials which are being incinerated to recycling facilities need to be studied.

As this analysis is merely a case study, based in the south-western region of Sweden, the results cannot be generalised for other parts of the world. The datasets from the *ecoinvent* database (Swiss Centre for Life-Cycle Inventories) which have been used are global averages and may still apply, but transport distances, barrier-coating costs and also the weighting factors assigned to economic and environmental aspects are likely to vary. The methodology adopted in this paper can still be applied to comparative analyses of this nature for other cases.

To quote from Innventia AB (2016), ‘*...forming a thesis like this one and identifying its limitations, is one thing, but to subsequently expose it to stakeholders in industry and let them challenge facts, quality and credibility is another story entirely*’. It is hoped that this case study, methodology and results presented will be a springboard for more detailed studies of this nature, under the umbrella of sustainability thinking.

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Appendix 1: Datasets used (EcoInvent database, Version 3.0.2.1)

1. Ethylene vinyl acetate copolymer {GLO}|market for|alloc def s. Germany’s electricity mix
2. Kaolin {GLO}|market for|alloc def s.
3. For latex, Finland’s electricity mix + natural gas for heating (data for energy requirements were provided by the supplier)
4. For starch, Dutch electricity mix and Steam heat {GLO} (data for energy requirements were provided by the supplier)
5. For PE, Swedish electricity mix + natural gas for heating (data for energy requirements were provided by the supplier)
6. Transport by road: Transport freight, lorry 16–32 metric ton, Euro3 {GLO}|market for|alloc def, s.
7. Transport by sea: Transport, freight, sea, transoceanic ship {GLO}|market for|alloc def s
8. Waste treatment:
 - i. Mixed plastic (waste treatment){GLO}|recycling of mixed plastics|alloc def, s
 - ii. Average incineration residue (waste treatment) {GLO}|market for|alloc def, s

Appendix 2

See Table 1.

Table 1 Unit production costs in Euro per kilogram, of the different barrier material/combination alternatives (obtained from the suppliers)

Material	Euro per kg, unit production cost, year-2016
PE	1
EVOH + kaolin from the USA (50% each)	2.45
EVOH + kaolin from the UK (50% each)	2.45
Latex + kaolin from the USA (50% each)	1.5
Latex + kaolin from the UK (50% each)	1.5
Starch	1

Appendix 3

See Table 2.

Table 2 List of weighting factors obtained

Sector	Person	Weighting factors (%)	
		Economic	Environmental
Industry (Sweden)	A	90	10
Academics/research (Sweden)	B	0	100
Academics/research (Sweden)	C	90	10
Industry (Sweden)	D	80	20
Research/business-oriented (Norway)	E	75	25
Industry (Sweden)	F	40	60
Industry (Sweden)	G	60	40
Industry (Sweden)	H	70	30
Research/business-oriented (Sweden)	I	30	70
Average (%)		59.4	40.6

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