1   **Incorporating Linear Programing and Life Cycle Thinking into environmental**

2   **sustainability decision-making: A case study on anchovy canning industry**

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13   **Abstract**

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15   Life Cycle Assessment (LCA) is a powerful tool to support environmental informed decisions among product and

16   process alternatives. LCA results reflect the process stage contributions to several environmental impacts, which

17   should be made mutually comparable to help in the decision-making process. Aggregated environmental indexes

18   enable the translation of this set of metrics into a one final score, by defining the attached weights to impacts. Weighting

19   values reflect the corresponding relevance assigned to each environmental impact. Current weighing schemes are based

20   on pre-articulation of preferences, without considering the specific features of the system under study.

21   This paper presents a methodology that combines LCA methodology and linear programming optimisation to

22   determine the environmental improvement actions that conduct to a more sustainable production. LCA was applied

23   using the environmental sustainability assessment (ESA) methodology to obtain two main indexes: natural resources

24   (NR) and environmental burdens (EB). Normalised indexes were optimised to determine the optimal joint of weighting

25   factors that lead to an optimised global environmental sustainability index.

26   The proposed methodology was applied to a food sector, in particular, to the anchovy canning industry in Cantabria

27   Region (Northern Spain). By maximising the objective function composed of NR and EB variables, it is possible to

28   find the optimal joint of weights that identify the best environmental sustainable options. This study proves that LCA

can be applied in combination to linear programing tools as a part of the decision-making process in the development of more sustainable processes and products.

Keywords: anchovy, canning industry, life cycle assessment, optimisation, linear programing

Introduction

Life Cycle Assessment (LCA) allows to identify methods of sustainable production and consumption and to support environmental decision-making process (Notarnicola et al. 2005). LCA has been widely applied to improve the design or to optimize a wide range of production processes. In many cases, the main goal of LCA studies is to identify, through a set of process alternatives, the one that could lead to environmental improvements (Vázquez-Rowe et al. 2015). Usually, LCA results in an environmental profile that consists of a set of direct or indirect process stages contributions to various environmental impact categories such as climate change, acidification, eutrophication, toxicity, land use or resource depletion. Although decision-making process is in general straightforward when one option under study scores better than the rest in all environmental categories simultaneously, it becomes difficult otherwise (Cortés-Borda et al. 2013).

In this sense, the use of aggregated environmental indexes represents a useful framework to aid in decision-making, since they are one-dimensional representation of all environmental impact categories for a particular system (Islam et al. 2017). However, the use of aggregated indexes in LCA requires quantifying and comparing the value of different environmental impacts, which represents a major challenge. First, normalisation is required to render the variables comparable. Afterwards, weighting and aggregation can be conducted (EC JCR 2010). The weighting step is the most controversial element in LCA, as this stage requires the incorporation of social, political and ethical values (Finnveden et al. 2009). In addition, the LCA international standard, ISO 14044 (2006a) states that: a) "It should be recognized

that there is no scientific basis for reducing LCA results to a single overall score or number"; and b) "Weighting shall not be used in LCA studies intended to be used in comparative assertions intended to be disclosed to the public". Despite the practicality of using aggregated environmental sustainability indexes, they represent an issue highly debated in the LCA community for a long time and is still considered open (Kägi et al. 2015). Weighting methods are frequently used in practice and the need for a science-based methodology is therefore essential.

Weighting schemes can be classified in different ways (Huppes and van Oers, 2011). A first classification consists of 3 categories: (i) panel methods, in which a group of experts are asked to provide their weighting factors; (ii) monetization methods, in which the weighting factors are expressed in monetary costs according to the estimated economic damage, and (iii) distance-to-target methods, in which these factors are calculated as a function of specified target values (Finnveden et al. 2002). A further distinction can be established between midpoints methods (which reflect the potency of a stressor at some point along the stressor-impact chain but are prior to the endpoint) and endpoints methods (which try to approximate the actual impact of emissions or extractions) (Bare, 2010). A representative example is the impact assessment method ReCIPe (Goedkoop et al. 2009), which is a follow up of Eco-indicator 99 (Goedkoop and Spriensma, 2001) and CML2002 methods (Guinée et al. 2002). It harmonises a midpoint and endpoint approach in a consistent framework that considers panel weighting. Both midpoint and endpoint results are also provided by more recent methods such as IMPACT 2002+ (Jolliet et al. 2003) and IMPACT World+ (Bulle et al. 2012). Such approaches ease the interpretation of multi-variable problems. However, they are sensitive to the normalisation and weighting procedures applied (Cortés-Borda et al. 2013).

Once the weighting scheme is selected and applied, impact categories can be aggregated to construct the composite index. Several computation methods have been proposed to derive sustainability aggregated indexes. For example, Sikdar (2009) defined an aggregated index as the geometric mean of the ratios of the individual metrics for pairwise comparison. Energy intensity, material intensity, potential chemical risk, and potential environmental impact were the sustainability metrics aggregated. Later, an Euclidean based distance index was proposed by the same author to incorporate negative indicators that represent benefits to the system (Sikdar et al. 2012). Other aggregation methods include Camberra distance (Brandi et al. 2014), Mahalonbis distance (Dos Santos and Brandi 2015) and vector space theory (Olinto and Islam 2017). A wide range of indicators were considered for these works, including environmental, social and economic metrics. Olinto and Islam (2017) were focused on LCA impact categories, suggesting their direct relationship to the economic and social domains too. These methods provide transparency in evaluating the impacts of

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weighting factors, where equal weights were assumed as a first approach. Other approaches for constructing a composite sustainability index include statistical models such as Principal Component Analysis (Mainali and Silveira, 2015) and multiple criteria decision analysis (MCDA) (De Luca et al. 2017). Regarding the later, data envelopment analysis (DEA) is gaining importance recently (Galán-Martín et al. 2016), since it can derive objective weights using linear programming tools (LP) without making any previous assumption on data, as opposed to Principal Component Analysis (Zhou et al. 2007).

Since the 1990s, LP has essentially been combined to LCA for system design and optimisation under environmental, economic and/or technical criteria. For example, Azapagic et al. (1999a, b) explored the opportunities of improvement of an existing mineral-processing system producing boron products by satisfying both economic and environmental criteria. Forman et al. (2014) used a refinery LP model to simulate operation of large US refineries and estimate the impacts of major petroleum products subjected to quality and demand changes. Steubing et al. (2016) applied a modular LCA approach to reduce the efforts in scenario analyses and optimisation when choices along a product's value chain lead to many alternative life cycles. Tan et al. (2008) presented a different approach that involves fuzzy target levels for environmental impacts. The outcome of the model is the optimal mix of technologies that embodies the best compromise of the targets defined. Furthermore, LP has also been used to address the problem backwards, i.e., given a set of solutions, find the value ranges where the weights attached to a set of environmental impacts must lie (Cortés-Borda et al. 2013).

This paper proposes a methodology that combines LCA and linear programming (LP) to derive an aggregated sustainability index that determines the environmental improvement actions leading to sustainable production. For this, the LCA impact assessment method selected was the Environmental Sustainability Assessment (ESA) approach developed by Irabien et al. (2009). The best joint of weights attached to a set of impact metrics is determined based on energy-intensity criteria, exploring the system from an environmental efficiency perspective. The results obtained will be compared to those delivered by ReCIPe methodology. The novelty of this study lies in obtaining a composite environmental index as decision-support tool using LP to derive objective and system-specific weights without pre-articulation of preferences. It is worth remarking that the proposed LCA-LP method does not intent to distort the results to reduce the environmental impact. The main idea of this weighing procedure according to the purposes defined by Pizzol et al. (2017), is to facilitate decision making in situations where tradeoffs between impact category results do not allow choosing one preferable solution among the alternatives or one improvement among possible ones. This

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work aims to extend beyond the ISO guidelines and propose a methodology based on single-score index together with the midpoint indicators in order to contribute to sound, understandable and effective decisions.

A food industry case study was considered as a good benchmark to illustrate the possibilities of the proposed methodological approach. In particular, a representative anchovy canning plant located in Santoña (Cantabria, North of Spain) was selected as case study.

Methods

The LCA methodology is applied following the recommendations of ISO 14040/14044 (2006a; 2006b) international standards. These guidelines describe LCA as a four-stage process, including: (i) definition of the goal and scope of the analysis, (ii) life cycle inventory analysis (LCI), (iii) life cycle impact assessment (LCIA) and (iv) interpretation. The LCIA stage includes two mandatory (i.e., classification and characterisation) and two optional (i.e., normalisation and weighting) steps.

The four above-mentioned steps are considered by ESA methodology (Margallo et al. 2014). In this sense, the environmental results are summarised into two macro-categories: natural resources consumption (NR) and environmental burdens (EB), which are integrated by several impact categories and provide a broad overview of the environmental performance of the process. In this work, ESA is extended and combined with linear programming (LP) to estimate the weighting factors that lead to an optimised global environmental sustainability index.

ReCIPe methodology also provides a single score by combining 3 endpoint categories: damage to human health, damage to ecosystems and damage to resource availability, which are in turn integrated by 18 midpoint categories. Three different panel perspectives are distinguished: individualist, hierarchist and egalitarian. Owing to its extensive use and similarity with the structure of the methodology proposed, results from our study are compared to the hierarchist (H) single score provided by ReCIPe.

Classification and characterisation

This stage involves the selection of the impact categories that are relevant for this study and the characterisation models. The impact of each emission or resource consumption from the LCI is assigned to the selected impact categories, according to each substance's ability to contribute to different environmental problems (Bare, 2010). The

impact of each emission or resource consumption is modelled quantitatively using a characterisation factor. This factor expresses the potential impact of each substance, allowing the addition of the different contributions from all emissions and resource consumptions within each impact category. This step enables the transformation of the LCI results into environmental impacts.

In this sense, NR includes the consumption of final useful resources such as energy ($X_{1,1}$) [MJ], materials ($X_{1,2}$) [kg] and water ($X_{1,3}$) [kg] for the process or product under study. On the other hand, EB is based on the environmental sustainability metrics developed by the Institution of Chemical Engineering (IChemE, 2002). These indexes can be used to assess the environmental sustainability performance of a system by providing a balanced view of the environmental impact of inputs (i.e., resource usage) and outputs (i.e., emissions, effluents and waste) (García et al. 2013). An advantage of this methodology is that environmental impacts can be aggregated and the environmental performances of different processes can be effectively compared, facilitating benchmarking (Diniz da Costa and Pagan, 2006).

The EB index comprises 10 environmental impact categories that are grouped according each environmental compartment: air ($X_{2,1}$) and water ($X_{2,2}$). The impact categories that constitutes $X_{2,1}$ are: atmospheric acidification (AA), global warming (GW), human health (carcinogenic) effects (HHE), stratospheric ozone depletion (SOD) and photochemical ozone (smog) formation (POF). The impact categories that are included in $X_{2,2}$ are: aquatic acidification (AqA), aquatic oxygen demand (AOD), ecotoxicity to aquatic life (metals to seawater) (MEco), ecotoxicity to aquatic life (other substances) (NMEco) and eutrophication (EU).

Normalisation and weighting

Normalisation relates the magnitude of impacts among different impact categories to reference values (Finnveden et al. 2002). Two different approaches can be distinguished in the normalisation methodology: (i) adjusting the results to have common dimensions as an operational prerequisite to valuation and (ii) placing the LCIA results in context by assessing the relative significance of the results across the different impact categories (Norris, 2001). The former leads to internal or case-specific normalisation based on the division of the scores in each category by some function of the cases values in that category. The second is achieved through dividing the scores in each category by an estimation of the total impacts in that category for a chosen system or region over a chosen time period (i.e. external normalisation). Normalisation facilitates the comparison among impact categories, while weighting procedure ranks the different

1 environmental categories according to their relative importance (EC JCR, 2010). Accordingly, NR and EB metrics

2 were normalised and weighted to assess the contribution of each variable within a single index.

3 In this way, internal normalisation is applied to the NR and EB impact categories according to Eq. 1-2. The average

4 consumption of natural resources ($\overline{X}_{1,i}^{ref}$) and the average environmental impacts ($\overline{X}_{2,j,k}^{ref}$) for the scenarios under study

5 are used as references, respectively.

$$X_{1,i}^* = \frac{X_{1,i}}{\overline{X}_{1,i}^{ref}} \tag{1}$$

$$X_{2,j,k}^* = \frac{X_{2,j,k}}{\overline{X}_{2,j,k}^{ref}} \tag{2}$$

6 where i represents different NR (energy, materials and water); j represents air and water compartments; k represents

7 the environmental impact categories; $X_{1,i}$ is the consumption of each i NR; $X_{1,i}^*$ is the normalised value of $X_{1,i}$; $\overline{X}_{1,i}^{ref}$ is

8 the NR reference value; $X_{2,j,k}$ describes the k environmental impact category for the environmental compartment j;

9 $\overline{X}_{2,j,k}^{ref}$ is the reference value for each impact category and $X_{2,j,k}^*$ is the normalised value of $X_{2,j,k.}$

10 The $X_{2,j}^*$ macro-categories are obtained by subjecting the normalised impact categories to equal weighting (Eq. 3). This

11 approach is based on the assumption that the same relevance is attributed to each impact category and, thus, equal

12 relative weights are given. Since there are 5 EB impact categories to each environmental burden, 1/5 is assumed as

13 weighting factor ($\delta_{j,k}$). Despite being not science-based, this weighting scheme is usually applied in the literature as a

14 first approximation for constructing a composite index (Pizzol et al. 2017).

$$X_{2,j}^* = \sum_{j=1}^{j} \delta_{j,k} \cdot X_{2,j,k}^* \tag{3}$$

15 Where $\delta_{j,k}$ represents the weighting factor for each k impact category in each j environmental compartment. $X_{2,j}^*$

16 represents the macro-categories EB to air and EB to water.

17 Therefore, the normalised NR macro-categories ($X_{1,i}^*$) that represent the consumption of energy ($X_{1,1}^*$), materials ($X_{1,2}^*$)

18 and water ($X_{1,3}^*$) and the normalised EB macro-categories ($X_{2,j,}^*$) that describe the impact to air ($X_{2,1}^*$) and water ($X_{2,2}^*$),

1 can be grouped into a single index so-called Environmental Sustainability Index (ESI). The weighting procedure

2 proposed is shown in Eq. 4.

$$ESI = \gamma_1 \sum_{i=1}^{i} \alpha_i X_{1,i}^* + \gamma_2 \sum_{j=1}^{j} \beta_j X_{2,j}^* \quad i \in [1,3] \text{ and } j \in [1,2] \tag{4}$$

3 Where ESI is the global environmental sustainability index that combines the consumption of NR and the generation

4 of EB. The weighting factor α_i serves at the aggregation of the 3 NR macro-categories ($X_{1,i}^*$) into a single super-

5 category namely NR (X_1). Similarly, β_j enables the weighting and combination of the 2 EB macro-categories ($X_{2,j}^*$)

6 into EB (X_2) index. Finally, γ_i combines X_1 and X_2 super-categories into the composite ESI index.

7 Most of the composite indexes are constructed by equal weighting (Zhou et al. 2012). Hence, for comparison issues,

8 the three NR and the two EB macro-categories are first assumed to be equally relevant. However, using the same

9 weighting value for α_i and β_j may result in an imbalance structure within the composite index, since the dimension

10 grouping the larger number of variables ($X_{1,i}^*$) will have higher weight than the dimension resulting from the

11 aggregation of lesser variables ($X_{2,j}^*$). For this reason, Eq.5-6 must be satisfied and thus $\alpha_i = 1/3$ for each i and $\beta_j = 1/2$

12 for each j.

$$\text{s.t. } \sum_{i=1}^{i=n} \alpha_i = 1 \tag{5}$$

$$\sum_{j=1}^{j=n} \beta_j = 1 \tag{6}$$

13 Nevertheless, this approach can disguise the absence of statistical or empirical bases for determining the weights (Zhou

14 et al. 2012). Therefore, although equal weighting is assumed at the lower level ($X_{2,j,k}^*$), the current study addresses a

15 first LP weighting approach at the macro- ($X_{1,i}^*$ and $X_{2,j}^*$) and super-categories (X_1 and X_2) extent to propose a method

16 simple enough before discussing other more complex and difficult to implement approaches.

17 The optimum values of α_i and β_j for each scenario are determined by means of the methodological procedure outlined

18 in Fig. 1. To keep the linear nature of the approach, γ_1 and γ_2 are assumed to be equal and further studied through a

19 sensitivity analysis at the end of this study. As the rest of weighting schemes proposed, the aggregation of γ_i must add

20 up to the unity and thus γ_i is 0.5. Eq. 4 is thus optimised to estimate weighting factors that lead to the ESI index. The

21 optimisation of Eq. 4 in terms of maximisation will allow to determine the best environmental performing system in

8

1 the worst case scenario. In this sense, LCA results will include specific, local and global features of the processes under

2 study.

3 The limits of the weighting factors are determined by correlating the different metrics. For example, it has been

4 observed that, when the requirements of an energy-intensive process are varied, the consumption of energy presents a

5 linear relationship with the consumption of NRs, as well as the EBs. So that, a linear correlation has been approximated

6 between the consumption of energy ($X_{1,1}^*$) and the rest of metrics ($X_{i,j}^*$) by simulating the process using different

7 scenarios. Hence, this relationship has been taken as a constraint for determining the lower and upper bounds of α_i and

8 β_j.

$$\text{s.t. } 0.01 \leq \alpha_i \leq w_{1,i}\alpha_1 \tag{6}$$

$$0.01 \leq \beta_j \leq w_{2,j}\alpha_1 \tag{7}$$

9 Where $w_{1,i}$ and $w_{2,j}$ are the factors to determine α_i and β_j, respectively. It has also been assumed that every environmental

10 impact has a minimum weight ($\alpha_{i,min} = \beta_{j,min} = 0.01$) so that none of them are neglected.

11

12 **Case study: the anchovy canning industry**

13 The anchovy canning industry in Cantabria produced in 2014 more than 14,000 ton of canned anchovy, generating

14 more than 100 million €. Cantabrian anchovy constitutes a gourmet product, whose diversification strategy and

15 introduction in new markets must be supported by a specific LCA study. Depending on the Cantabrian anchovy stock

16 level, canning plants may import anchovies from other countries such as Argentine and Chile or Peru. Regarding the

17 consumer's preferences, the filling oil can be sunflower oil, olive oil and extra virgin olive oil. Anchovy packaging

18 can be presented in aluminium and tinplate cans or glass jars. These three variables (origin, type of oil and packaging

19 material) comprise a wide range of possible products and scenarios, whose environmental sustainability assessment is

20 addressed in this work using the proposed methodology.

21 The selected canning plant produced in 2014 160 ton of canned anchovies in oil (Laso et al. 2016a). LCA is applied

22 from cradle-to-grave, as shown in Fig. 2. The first stage considers the fishery of the anchovy based on vessel operation

23 and the production stages of the different ingredients (raw materials) that are then transported to the canning factory

in Santoña. There, anchovy is pre-treated and subjected to several transformation steps until the final fillets are introduced into cans and filled with oil. Cans are packed and distributed to the consumer. Fish residues are valorised into fishmeal and anchovy paste, while effluents are sent to a municipal wastewater treatment plant. Finally, the third stage is comprised by the distribution, use and end-of-life of the canned anchovies.

To carry out this analysis under a life cycle approach, 1 kg of raw anchovy entering the factory was selected as functional unit (FU). The anchovy canning process is a multi-output system. Apart from canned anchovy, additional products are obtained from the valorisation of the fish residues. System expansion is applied to consider the extra functions of the system: the production of fishmeal from fresh anchovy, which is considered for the valorisation of the heads and spines, and the production of tuna pâté, which constitutes the replacement process for anchovy meat valorisation (Laso et al. 2016b). Regarding end-of-life, since the current recycling rates in Spain are high (84.8% metals, 82.3% cardboard and paper, 66.5% plastic and 73% glass), 100% recycling rate is assumed for the recycling of the packaging materials (Ecoembes, 2016; Ecovidrio, 2016). An attributional approach is applied assuming that the material recycled would displace an equivalent quantity of the current mix of virgin and recycled material in the market (Bala et al., 2015). Avoided burdens are then applied using the actual mix of virgin and recycled materials in the market (Laso et al. 2017).The transportation of the raw materials and the packaging is also considered using the most similar options from the database (PE International, 2014). The transportation distances were estimated by means of road guides: salt (900 km), brine (80 km), oil (850 km) and packaging (270 km). The Argentine (10,900 km) and Peruvian/Chilean (13,800 km average) anchovies were transported to Cantabria by an ocean container ship. Finally, the canned anchovies were assumed to be transported from the canning plant to a logistic hub (40 km) and from there to a supermarket (10 km).

Scenarios under study

The combination of the three mentioned variables (i.e. anchovy origin, type of oil and packaging material) can lead to multiple scenarios. Ten different scenarios are studied as described below (Table 1).

- *Anchovy origin.* The canning plant processes anchovies from Cantabria (*Engraulis encrasicolus*), but also imported anchovies from Argentina (*Engraulis anchoita*) and from Chile and Peru (*Engraulis ringens*) to satisfy the depleted stock level in the Cantabrian Sea. Cantabrian anchovies are captured from the Bay of

1	Biscay and transported to the factory. Argentine anchovies are fished in the Southwest Atlantic Sea and pre-
2	treated in Argentine factories before being transported to Santoña. Chilean and Peruvian anchovies caught in
3	the Northwest Pacific Sea are pre-treated and transformed in Chilean and Peruvian companies. The already
4	vacuum-packed anchovies are then transported to Santoña. Hence, the study of this variable is conducted by
5	means of the following scenarios: Cantabrian (S_A), Argentine (S_B) and Chilean/Peruvian (S_C) origin.

6   - *Type of oil.* The filling oil used for canned anchovies depends on the anchovy origin and the geographical

7   distribution of the product. The scenarios to study are defined as: refined olive oil (S_D), extra virgin olive oil

8   (S_E) and sunflower oil (S_F).

9   - *Packaging material.* Aluminium and tinplate cans as well as glass jars are already employed in the Cantabrian

10   factory. Furthermore, it is interesting to study other possible materials such as plastic, which is already used

11   by a Danish company as the usual canned appearance (Almeida et al. 2015). Four different scenarios are thus

12   distinguished for the assessment of the packaging material: aluminium (S_G), tinplate (S_H), glass (S_I) and plastic

13   (S_J).

14 Cantabrian origin, extra virgin olive oil and aluminium packaging are the variables used as base case of study.

15

16 *Life cycle inventory and data acquisition*

17 In the life cycle inventory (LCI), all relevant inputs and output are collected. LCA data can be categorised as primary

18 and secondary data (Tolle et al. 1995). Primary data are plant or process-specific data. Secondary or background data

19 are publicly available data which have not been collected specifically for the purpose of the study. Primary data are

20 comprised of the consumption of salt, brine, oil, electricity and fuels, water, packaging materials and the generation of

21 solid residues and effluents. They were sourced from anchovy canning companies in Santoña and from bibliographic

22 sources. Background data were retrieved from the databases PE International (2014) and Ecoinvent 3.1 (2014). The

23 production of oil was obtained from the OiLCA Tool (SUDOE 2011), while salt (Goetfried et al. 2012) and brine

24 production (NYSDEC 2015) and the specifications of the WWTP (Pasqualino et al. 2009) were collected from the

25 literature.

26

27 *Life cycle impact assessment (LCIA)*

1   The LCI was modelled using the LCA software GaBi 6.0 (PE International 2014). The ESA methodology was applied

2   for the LCIA stage using the metrics developed by the Institution of Chemical Engineers (IChemE 2002). Hence,

3   results are divided into two different variables regarding the use of natural resources (i.e. NR) and the environmental

4   burdens generated by the release of contaminants to air and water compartments (i.e. EB). Land use and soil carbon

5   stocks are not included, as the construction and maintenance of the plant is excluded from the system boundaries of

6   the study due to the fact that the anchovy industry is not an intensive-land use process.

7   The LCA results obtained in a previous work (Laso et al. 2017) were normalised as displayed in Table 2. Within each

8   variable, the environmental scores have been ranked using colour coding. Higher impacts are remarked in red, while

9   lower impacts are coloured in green. As observed, the decision-making process in not straightforward, as there is no

10   scenario that presents the best (or worst) scores for every metric. This is the reason why determining a composite index

11   is key for identifying the best performing environmental improvement actions.

12

13   *Weighting and LP of the ESA results*

14   Once the environmental sustainability of the system is assessed, the approach described is applied to simplify the

15   decision-making process. Given a set of environmental impacts for each alternative scenario, the optimisation

16   methodology determines the weighting factors that conduct to a maximum global environmental impact, represented

17   by ESI. It is assumed that the best environmental alternatives that conduct to a more sustainable production are

18   determined by maximising the weighted sum of these impacts. It is also assumed that the scores of the environmental

19   impacts are related among them and so is the weight attached to them. In particular, the consumption of energy ($X_{1,1}^*$)

20   influences the consumption of materials ($X_{1,2}^*$) and water ($X_{1,3}^*$) for energy production and then the environmental

21   burdens generated to air ($X_{2,1}^*$) and water ($X_{2,2}^*$) compartments. As previously shown, the packaging step is the main

22   hotspot of the system due to the energy intensive production of aluminium. Hence, the influence of this process has

23   been studied to determine the relationship between energy consumption and the remaining of metrics. An accurate

24   linear approximation ($R^2 > 0.999$) has been found between $X_{1,1}^*$ and each impact category and this relationship is

25   assumed as a constraint for determining the weighting factors α_i and β_j (Eq. 5). The slopes ($w_{1,i}$ and $w_{2,j}$) of the linear

26   functions are shown in Table 3, while the optimised weighting factors are outlined in Table 4.

1 Therefore, this procedure does not modify the previously calculated individual impacts ($X^*_{i,j}$), but affects the relevance

2 attributed to each of them. Thereafter, the normalised results for NR ($X^*_{1,i}$) and EB ($X^*_{2,j}$) are posed in mathematical

3 terms using the linear model described in Eq. (3-5). The software GAMS (2017) is used to carry out the optimisation

4 procedure. The influence of the three main variables (anchovy origin, type of oil and packaging material) on the

5 environmental impacts of canned anchovy is studied separately, as detailed below.

6

7 *Anchovy origin*

8 Three alternative scenarios are considered to study the anchovy origin variable and determine the most environmental

9 sustainable option. Rows 1-3 in Table 2 represent the normalised NR and EB scores for anchovies captured in

10 Cantabria, Argentine and Chile/Peru, which include vessel operation in terms of diesel consumption. Regarding NR,

11 Argentine anchovies results the best scenario, while anchovies captured in Cantabrian Sea require the highest NR. On

12 one hand, this is due to the fact that a greater quantity of anchovy is processed per kg of Cantabrian fresh anchovy,

13 since *ca.* 68% of fresh anchovy is wasted in S_A vs. 75% in S_B and S_C. On the other hand, this is owed to the larger

14 diesel consumption in the fishery stage, as Cantabrian vessels consumed 20 times more diesel than Argentine and

15 Chilean or Peruvian. This pattern is maintained for EB to water. However, S_C exhibits the largest EB to air as opposed

16 to S_A, essentially due to the air emissions in the production of fuel oil used to transport by ship.

17 Optimisation is conducted according to Fig. 1. Hence, Eq. 4 is maximised so that the scenarios are compared under

18 worst-case conditions. Rows 1-3 in Table 4 show the optimised weighting values for each impact category. As

19 expected, $X^*_{1,1}$ is the metric to which the highest importance is attributed (α_1=0.78-0.98) since its influence is reflected

20 in the rest of categories, avoiding double-counting. S_A receives the highest weighting factor for energy consumption

21 (α_1=0.98) as $X^*_{1,1}$ is the impact category that presents the largest score for this scenario. At the same time, S_A is the

22 most energy intensive alternative, being more penalised than S_B and S_C. Regarding EB, more relevance is attributed to

23 $X^*_{2,1}$ ($\overline{\beta_2}$=0.71) as opposed to $X^*_{2,2}$ ($\overline{\beta_1}$=0.29) when S_A is excluded. This is mainly due to the fact that the valorisation

24 of the anchovy residues involves lower $X^*_{2,2}$ and thus the highest weighting factors are assigned to $X^*_{2,1}$. The exception

25 is S_A, where the production of diesel results in higher EB to water and $\beta_2 = 0.99$.

1   After optimisation, weighting is conducted to obtain X_1 and X_2 indexes as a pre-requisite step for constructing the

2   composite ESI index. These indexes are depicted in Fig. 3a, where x-axis represents the first term of Eq. 4, obtained

3   from the addition of the weighted energy ($X_{1,1}^*$), materials ($X_{1,2}^*$) and water ($X_{1,3}^*$) variables using α_i, while the second

4   term of the objective function is outline in the vertical axis, comprised by the aggregation of the weighted

5   environmental burdens to air ($X_{2,1}^*$) and to water ($X_{2,2}^*$) using β_j. As can be seen, the highest X_1 is depicted by Cantabrian

6   anchovies, while similar scores are obtained for Argentine and Chilean/Peruvian anchovies. Regarding X_2, S_A exhibits

7   and intermediate position between S_B and S_C, remarking the environmental trade-off between fishery in Cantabrian

8   Sea and long-distance transportation from Argentine or Chile/Peru.

9   The global environmental ESI index is obtained from the combination of X_1 and X_2 (Fig. 4). It can be appreciated that

10   despite generating a lower quantity of wastes and involving lower transport distances, Cantabrian anchovies results

11   the worst scenario. Conversely, the most environmentally sustainable option is the importation of Argentine anchovies,

12   followed closely by Chilean/Peruvian anchovies, due to the marine freight. The reason is that marine freight usually

13   has a very low impact as compared to extraction, such as Vázquez-Rowe et al. (2013) and Ziegler et al. (2011)

14   suggested for hake and shrimp capture, respectively.

15   Although this paper explores anchovy origin from an environmental point of view, it must be highlighted that

16   Cantabrian anchovies represent a high-quality product, for which consumers are willing to pay up to twice the value

17   of other anchovies. A further eco-efficiency study in terms of value added is required to properly assess the

18   sustainability of this product.

19

20   *Type of oil*

21   The three type of oil under study were: extra virgin olive oil, refined olive oil and sunflower oil. The normalised scores

22   are displayed in rows 4-6 from Table 2. As can be observed, before optimisation, there is no scenario that scores better

23   (or worst) than the rest for every metric. Regarding NR, S_F presents the highest energy and material consumption, as

24   opposed to the lowest water requirements (Table 2). This is due to the fact that sunflower olive oil presents a lower

25   yield per hectare than both olive oils and consequently requires more fertilizers. Conversely, it consumes less water

1 for cultivation. S_D depicts slightly higher impact than S_E, since an extra refine step is required for the latter. Regarding

2 EB, S_F presents the highest EB to air, but results the less aggressive scenario to water compartment.

3 Once optimisation and weighting are performed, S_F is penalised due to its high energy consumption (α_1=0.98),

4 presenting a 17% larger X_1 impact than S_E (Fig. 4a). Despite its lower EB to water, this difference between S_F and S_E

5 is almost duplicated when X_2 is assessed, since the largest relevance is attributed to EB to air (β_2= 0.77-0.87). From

6 the combination of X_1 and X_2, it can be appreciated that ESI values for S_F are the highest (ESI= 1.12). Therefore,

7 despite requiring less irrigation than olive oil production, sunflower oil displays a *ca.* 20% worse environmental

8 performance than olive oils (ESI_{SD}= 0.91 and ESI_{SE}= 0.88). The differences between S_D and S_E are due to the fact that

9 extra virgin olive oil is extracted directly from olives using pressure and refined olive oil requires a subsequent refining

10 stage. Therefore, the use of extra virgin olive oil for filling anchovy cans results the best environmental option. Since

11 the difference between S_D and S_E is small, the economic value could be considered in a future work. In this sense, it

12 could be anticipated that Spanish consumers are willing to pay around 4% more for extra virgin olive oil than the

13 refined alternative (IOOM, 2016).

14

15 *Packaging material*

16 Four packaging materials were analysed: aluminium, tinplate, glass and plastic. Their normalised results are shown in

17 rows 7 to 10 in Table 2. As previously discussed, changing the packaging material is one of the variables that influences

18 the most the environmental performance of the process (Almeida et al. 2015; Hospido et al. 2006). However, there is

19 no solution that scores better than the rest for every metric. For example, regarding the consumption of NR, the less

20 energy-intensive scenario is represented by the use of tinplate cans, which involves a 13% reduction with regard to the

21 use of aluminium. Conversely, the usage of glass jars requires 26% more energy than aluminium. Nevertheless,

22 aluminium remains as the best option if the materials demand is considered, while glass is maintained as the worst

23 scenario. When the consumption of water is assessed, the use of plastic packaging becomes the best environmental

24 alternative, followed closely (1%) by aluminium and glass. A different pattern is observed for EB metrics. While

25 aluminium results the worst scenario for both $X^*_{2,1}$ and $X^*_{2,2}$, the use of tinplate and plastic cans involve nearly a 50%

26 reduction of those impacts. Consequently, weighting is necessary to identify the best-performing scenario.

1 The optimisation procedure penalises the packaging scenarios for their energy intensive nature, attributing the highest

2 weighting factor to energy consumption (α_1=0.98, Table 4). The exception is S_H, which receives the highest weighting

3 factor (α_3=0.843) for $X^{*}_{1,3}$ owing to the high water intensity of this scenario in comparison to the rest of alternatives.

4 Regarding EB, the largest relevance is again attributed to air compartment, being β_2 in the 0.74-0.95 range.

5 The resulting X_1 and X_2 indexes are outlined in Fig. 5a. S_G (aluminium) and S_J (plastic) display the best X_1 results.

6 Conversely, S_I (glass) involves a 11% higher X_1 impact, resulting the worst scenario. Regarding the X_2 index, the

7 lowest impact is observed for S_H (tinplate) followed by closely S_J, while S_G becomes the largest score. Although the

8 best performing scenario may be inferred from Fig. 5a because of its closeness to the minimum ESI area, the less

9 environmental sustainable packaging is not as easily identifiable. In this sense, the composite ESI indexes are displayed

10 in Fig. 5b. It can be observed that the worst performing scenario is S_G (ESI= 1.09) followed by 6% lower S_I. S_G

11 represents the use of aluminium packaging, which is the current most used material and presents nearly 20% larger

12 environmental global impacts than plastic.

13 Although S_J is estimated as the most sustainable scenario, the difference between S_J and S_H is below 5% and thus the

14 results are not determinant. In this sense, the assessment of other sustainable development issues (social or economic)

15 in combination to environmental perspective may be given more importance. For example, it can be anticipated that

16 tinplate price is around ten times plastic price (LDPE is assumed) and thus plastic alternative could be preferred (LME

17 2016; Plastics Informat 2016).

18

19 *Overall decision*

20 The decision pathway followed for determining the best environmental alternatives in the Cantabrian anchovy canning

21 industry is outlined in Fig. 6. Three different levels of decision are identified, according to the three variables under

22 study. For each variable, the scenarios studied are ranked according to its environmental global impact. Values near

23 the symbol "+" indicate the highest environmental impacts, while "−" represents the lowest scores. In addition, colour

24 coding is used to differentiate each scenario. Therefore, the green path shows the best environmental improvement

25 actions identified for this industry, which are capturing the anchovy in Argentine and importing it, using extra virgin

26 olive oil for filling the cans and replacing the current packaging by plastic material.

It has been observed that the importation of out-of-season products to satisfy the depleted stock is in fact environmentally more sustainable than capturing the Cantabrian anchovies in the Bay of Biscay. In this sense, in order to improve the competitiveness of the Cantabrian anchovy product and preserve the quality label, it is necessary to reduce the environmental impacts of the fishery stage. The improvement measures should be addressed to the revision of the fishing fleet regarding the use of more efficient engines, green fuels and the replacement of the antiquated vessels. On the other hand, it must be highlighted that, from a circular economy point of view, Cantabrian anchovies could not be much more improved, since they generate around 10% less fish residues than anchovies captured in Argentine and Chile/Peru and are in turn valorised.

1   **Sensitivity analysis**

2   Initially, the super-categories X_1 and X_2 were assumed to be equally relevant, and thus their weighting factors γ_i were

3   estimated at 0.5. In this section, the influence of the relative importance assigned to each super-category is studied.

4   Hence, γ_1 is varied within the 1-0 range (Fig. 7). When γ_1 is assumed at 1, the maximum importance is attributed to

5   the consumption of resources, while the environmental burdens are disregarded. Conversely, the influence of the

6   resources consumption on the environmental sustainability of the process is neglected as opposed to the environmental

7   burdens when γ_1 is estimated at 0.

8   Therefore, as Fig. 7 outlines, S_C results in the worst scenario regarding the anchovy origin when the environmental

9   sustainability assessment is only based on the environmental burdens to air and water. However, S_A becomes the less

10   sustainable scenario when $\gamma_1 > 0$ and both NR and EB are studied. Conversely, the ESI index seems to be insensitive

11   to γ_i values when the type of oil is assessed. This is mainly due to the fact that S_E scores better than the rest in the

12   majority of environmental categories simultaneously, while S_F results the worst for most of them.

13   The largest influence of X_1 and X_2 weighting values is observed in the packaging assessment. When X_1 is only

14   considered (γ_1=1), S_I results the worst performing scenario, followed by S_H. As the importance assigned to X_2

15   increases, the pattern is inversed and S_G becomes the worst alternative, followed by S_I. A similar situation is observed

16   for S_H and S_J. Using tinplate material is the best environmental alternative when the consumption of natural resources

17   is assumed to be unimportant (γ_2<0.25). As X_1 weighting factor is increased, plastic utilisation becomes the most

18   sustainable option.

19   On the other hand, it can be observed that the ranking of the scenarios is maintained for all the variables when X_1

20   contributions to ESI is between 25% and 75%. Consequently, the importance of considering both indexes for the

21   construction of the composite index must be highlighted, being 0.5 a conservative selection for simplifying decision-

22   making process.

23   These results have been compared to that from ReCIPe methodology. Table 5 displays the ReCIPe single score for the

24   scenarios under study. The hierarchical perspective is used, which is the consensus model often considered as default

25   approach. Under this perspective, the endpoint categories damage to human health, damage to ecosystems and damage

to resource availability are weighted using the panel factors 400, 300 and 300, respectively. These values are common to every system, regardless of the specific features of the process under study.

A similar pattern to that obtain with our methodology is observed. The exception is found for the packaging materials, since S_H emerges as the most sustainable option according to ReCIPe, while S_I becomes the worst alternative. However, the difference between S_H and S_J is as small as between S_I and S_G (*ca.* 3%) so that other sustainability pillars should be assessed to take a determinant decision.

It must be highlighted the fact that the ReCIPe single score for sunflower oil is twice the value obtained for both olive oils. Such difference is due to the impact category agricultural land occupation, which is 80 times larger for S_H than S_E.

On one hand, the similarity observed with both methods serves as a validation tool for the methodology proposed. On the other hand, the LCA-LP approach proposed is based on technical criteria that enable penalising the critical variables with the aim of comparing the scenarios under worst-case conditions. This is conducted according to the specific, local and global features of the processes under study.

Conclusions

It is of great importance to consider environmental aspects of manufacturing in industry and lead industry professionals towards using more environmentally-friendly and sustainable systems. However, stake-holders are often in the need of choosing among several environmental profiles provided by the computed alternatives. The variety of existing environmental metrics far from providing a more complete framework usually hinders the decision-making process.

Through the proposed methodology, which combines LCA methodology and linear programming (LP), it is possible to find an aggregated environmental sustainability index to determine the environmental improvement actions that lead to sustainable production. The environmental sustainability assessment (ESA) procedure was applied to obtain two main indexes: natural resources (NR) and environmental burdens (EB). An objective function composed of the variables NR and EB was proposed to obtain a composite index (i.e. ESI).

The novelty of this study lies in obtaining a composite environmental index as decision-support tool without pre-

1   articulation of preferences. Weighting factors were estimated using LP tools based on technical criteria that enable

2   penalising the critical variables.

3   This methodology was applied to a case of study based on the Cantabrian anchovy canning process. Three different

4   variables were studied: anchovy origin, type of filling oil and packaging material. In this sense, energy consumption

5   resulted the critical variable, for which the highest weighting factor was mathematically assigned. Consequently, larger

6   weighting factors are attributed to the super-categories EB to air than to EB to water. A sensitivity analysis was

7   performed to provide transparency in evaluating the weighting factors attributed to both super-categories, finding out

8   that results were not disturb for weighting factors fluctuations between 25% and 75%.

9   Results from the study were compared to the panel ReCiPe approach, which serves as a validation tool since similar

10 conclusions were observed. However, it must be highlighted that the ranking of alternatives established by ReCiPe for

11 the anchovy packaging variable was slightly different from our approach. This is due to the fact that the environmental

12 scores of the best performing packaging materials (plastic and tinplate) barely differed between them (3%). In such

13 cases, the need of developing a composite index that encompasses other sustainability criteria such as the economic

14 score is remarked. Same situation was observed for the worst performing packaging scenarios (aluminium and glass).

15 This case study constitutes a first weighting approach at the upper level (macro- and super-categories) to propose a

16 method simple enough before discussing other more complex and difficult to implement approaches. Moreover, using

17 system-specific weighting factors instead of panel values provides a technical discussion by attributing more relevance

18 to the critical system variables.

19 The methodology proposed demonstrates how LCA tool can be applied in combination to linear program optimisation

20 to simplify the decision-making process. The LCA methodology is currently evolving towards the inclusion of more

21 impact categories, taking into account both economic and social dimensions. According to this, forthcoming efforts

22 will be addressed to the development of an aggregation strategy. This work provides an appropriate framework for the

23 future definition of a global composite index that takes into account other sustainability dimensions besides the

24 environmental pillar.

25 The conclusions of this study are in accordance to those LCA practitioners who consider the need to include the

26 grouping, weighting and normalisation steps in ISO 14044, as well as reconsider their prohibited use in comparative

27 assertions intended to be disclosed to the public. Nevertheless, we conclude that even though single-score indicators

can be very helpful for decision-making, midpoint indicators remain helpful in identifying reduction targets and measures for specific environmental concerns, both in terms of consumption of natural resources and environmental burdens.

References

Almeida C, Vaz S, Ziegler F (2015) Environmental life cycle assessment of a canned sardine product from Portugal. J Ind Ecol 19(4):607-617

Azapagic A, Clift R (1999a) The application of life cycle assessment to process optimisation. Comput Chem Eng 23(10):1509-1526

Azapagic A, Clift R (1999b) Life cycle assessment and multiobjective optimisation. J Clean Prod 7(2):135-143

Bala A, Raugei M, Fullana-i-Palmer P (2015) Introducing a new method for calculating the environmental credits of end-of-life material recovery in attributional LCA. Int J Life Cycle Assess 20(5):645-654

Bare JC (2010). Life cycle impact assessment research developments and needs. Clean Technol Environ Policy 12(4):341-351

Brandi HS, Daroda RJ, Olinto AC (2014) The use of the canberra metrics to aggregate metrics to sustainability. Clean Technol Environ Policy 16(5):911-920

Bulle C, Jolliet O, Humbert S, Rosenbaum R, Margni M (2012) IMPACT world: A new global regionalized life cycle impact assessment method. LCA XII, United States, Washington, Tacoma

Cortés-Borda D, Guillén-Gosálbez G, Esteller LJ (2013) On the use of weighting in LCA: Translating decision makers' preferences into weights via linear programming. Int J Life Cycle Assess 18(5):948-957

De Luca AI, Iofrida N, Leskinen P, Stillitano T, Falcone G, Strano A et al. (2017). Life cycle tools combined with multi-criteria and participatory methods for agricultural sustainability: Insights from a systematic and critical review. Sci Total Environ 595:352-370

1   Dos Santos SF, Brandi HS (2015) Model framework to construct a single aggregate sustainability indicator: An

2   application to the biodiesel supply chain. Clean Technol Environ Policy 17(7):1963-1973

3   EC JCR (2010) ILCD handbook: General guide for life cycle assessment - provisions and action steps. Publications

4   office of the European Union, Luxembourg.

5   Ecoembes (2016) Results report 2016 (in Spanish).https://www.ecoembes.com/sites/default/files/reciclaje-en-datos-

6   2016.pdf. Accessed 12 May 2017

7   Ecovidrio (2016) Glass recycling in Spain (in Spanish). El reciclado del vidrio en España 2016.

8   http://www.ecovidrio.es/Accessed 12 May 2017

9   Ecoinvent (2014) Ecoinvent Database Version 3.1. Swiss Federal Institute of Technology Zurich (ETH Zurich)

10   http://www.ecoinvent.org/. Accessed 15 May 2017

11   Finnveden G, Hofstetter P, Bare J, Basson L, Ciroth A, Mettier T et al. (2002) Normalization, grouping and

12   weighting in life cycle impact assessment. Haes et al (ed.), Life Cycle Impact Assessment: Striving Towards Best

13   Practice. SETAC Press, Pensacola FL:177-208

14   Finnveden G, Hauschild MZ, Ekvall T, Guinée J, Heijungs R, Hellweg S et al. (2009) Recent developments in life

15   cycle assessment. J Environ Manage 91(1):1-21

16   Forman GS, Divita VB, Han J, Cai H, Elgowainy A, Wang M (2014) U.S. refinery efficiency: Impacts analysis and

17   implications for fuel carbon policy implementation. Environ Sci Technol 48(13):7625-7633

18   Frischknecht R, Steiner R, Jungbluth N (2009) The ecological scarcity Method–Eco-factors 2006-A method for

19   impact assessment in LCA. http://esu-services.ch/. Accessed 12 May 2017

20   Galán-Martín Á, Guillén-Gosálbez G, Stamford L, Azapagic A (2016) Enhanced data envelopment analysis for

21   sustainability assessment: A novel methodology and application to electricity technologies. Comput Chem Eng

22   90:188-200

GAMS Development Corporation (2017) GAMS: General algebraic modeling system, Washington DC, USA. https://www.gams.com/ Accessed 15 May 2017

García V, Margallo M, Aldaco R, Urtiaga A, Irabien A (2013) Environmental sustainability assessment of an innovative cr (III) passivation process. ACS Sustain Chem Eng 1(5):481-487

Goedkoop M, Spriensma R (2001) The eco-indicator99: A damage oriented method for life cycle impact assessment: Methodology report. Ministry of VROM, The Hague

Goedkoop M, Heijungs R, Huijbregts MAJ, De Schryver A, Struijs J, Van Zelm R (2009) ReCiPe 2008. A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level, 1st ed., Report I: Characterisation, Ministry of VROM, The Hague

Goetfried F, Stratmann B, Quack D (2012) Life cycle assessment of sodium chloride production and transport. International Conference on Biodiversity, Sustainability and Solar Salt Proceedings: 59-63

Guinée JB, Gorrée M, Heijungs R, Huppes G, Kleijn R, Koning A et al (2002) An operational guide to the ISO standards. Centrum Milieukunde Leiden (CML) Kluwer: Dordrecht, NL, Leiden University

Hauschild MZ, Potting J (2004) Background for spatial differentiation in life cycle impact assessment. The EDIP2003 methodology. Institute for Product Development Technical University of Denmark. Environmental Protection Agency, Danish Ministry of the Environment, Denmark

Hospido A, Vazquez ME, Cuevas A, Feijoo G, Moreira MT (2006) Environmental assessment of canned tuna manufacture with a life-cycle perspective. Resour Conserv Recy 47(1):56-72

Huppes G, van Oers L (2011) Background review of existing weighting approaches in life cycle impact assessment (LCIA). Publications Office. http://eplca.jrc.ec.europa.eu/. Accessed 12 May 2017

IChemE (2002) The sustainability metrics: Sustainable development progress metrics recommended for use in the process industry. The Institution of Chemical Engineers. Rugby, UK

1   IOOM (2016) International Olive Oil Market. https://www.oliveoilmarket.eu/. Accessed 12 May 2017

2   Irabien A, Aldaco R, Dominguez-Ramos A (2009) Environmental Sustainability Normalization of Industrial

3   Processes. Comput Aided Chem Eng 26:1105-1109

4   Islam S, Ponnambalam SG, Lam HL (2017) A novel framework for analyzing the green value of food supply chain

5   based on life cycle assessment. Clean Technol Environ Policy 19(1):93-103

6   ISO (2006a) ISO 14044: Environmental Management - Life Cycle Assessment - Requirements and Guidelines. ISO,

7   Geneva

8   ISO (2006b) ISO 14040: Environmental Management - Life Cycle Assessment - Principles and Framework. ISO,

9   Geneva

10   Jolliet O, Margni M, Charles R, Humbert S, Payet J, Rebitzer G, et al. (2003) IMPACT 2002+: A new life cycle

11   impact assessment methodology. Int J Life Cycle Assess 8(6):324-330

12   Kägi T, Dinkel F, Frischknecht R, Humbert S, Lindberg J, De Mester S et al (2016) Session "Midpoint, endpoint or

13   single score for decision-making?"—SETAC europe 25th annual meeting, may 5th, 2015. Int J Life Cycle

14   Assess 21(1):129-132

15   Laso J, Margallo M, Fullana P, Bala A, Gazulla C, Irabien A, et al. (2017) When product diversification influences

16   life cycle impact assessment: A case study of canned anchovy. Sci Total Environ 581-582:629-639

17   Laso J, Margallo M, Fullana P, Bala A, Gazulla C, Irabien A, Aldaco R (2016a) Introducing life cycle thinking to

18   define best available techniques for products: Application to the anchovy canning industry. J Clean Prod. In press.

19   DOI: 10.1016/j.jclepro.2016.08.040

20   Laso J, Margallo M, Celaya J, Fullana P, Bala A, Gazulla C, Irabien A, Aldaco R (2016b) Waste management under

21   a life cycle approach as a tool for a circular economy in the canned anchovy industry. Waste Management and

22   Research 34(8):724-733

1   LME (2016) London Metal Exchange. https://www.lme.com/. Accessed 12 May 2017

2   Mainali B, Silveira S (2015) Using a sustainability index to assess energy technologies for rural electrification.

3   Renewable and Sustainable Energy Reviews, 41, 1351-1365

4   Margallo M, Dominguez-Ramos A, Aldaco R, Bala A, Fullana P, Irabien A (2014) Environmental sustainability

5   assessment in the process industry: A case study of waste-to-energy plants in Spain. Resour Conserv Recy 93:144-

6   155

7   Norris GA (2001) The requirement for congruence in normalization. Int J Life Cycle Assess 6(2):85-88

8   Notarnicola B, Salomone R, Petti L, Renzulli PA, Roma R, Cerutti AK Life cycle assessment in the agri-food sector.

9   Case Studies, Methodological Issues and Best Practices, Springer, Cham, Heidelberg, New York, Dordrecht, London

10   NYSDEC (2015) Solution Salt Mining Summary. New York State Department of environmental Conservation

11   Olinto AC, Islam S (2017) Optimal aggregate sustainability assessment of total and selected factors of industrial

12   processes. Clean Technol Environ Policy. In press. DOI: 10.1007/s10098-017-1350-0

13   Pasqualino JC, Meneses M, Abella M, Castells F (2009) LCA as a decision support tool for the environmental

14   improvement of the operation of a municipal wastewater treatment plant. Environ Sci Technol 43(9):3300-3307

15   PE International (2014) GaBi 6 software and database on life cycle assessment. Leinfelden-Echterdingen, Germany

16   Pizzol M, Laurent A, Sala S, Weidema B, Verones F, Koffler C (2017) Normalisation and weighting in life cycle

17   assessment: Quo vadis? Int J Life Cycle Assess 22 (6):853–866

18   Plastics Informat (2016) LDPE Price Chart. http://www.plasticsinfomart.com/ldpe-price-chart/. Accessed 12 May

19   2017

20   Sikdar SK (2009) On aggregating multiple indicators into a single metric for sustainability. Clean Technol Environ

21   Policy 11(2):157-161

1 Sikdar SK, Sengupta D, Harten P (2012) More on aggregating multiple indicators into a single index for

2 sustainability analyses. Clean Technol Environ Policy 14(5):765-773

3 Sleeswijk AA, Sevenster MM, Croezen HH, Davidson MM (2010) On weighting: A review of LCA weigthing

4 methods. http://www.cedelft.eu/. Accessed 12 May 2017

5 Steen B (1999) A systematic approach to environmental priority strategies in product development (EPS). Version

6 2000-General System Characteristics/ Models and Data of the Default Method. Chalmers University of Technology,

7 Gothenburg, Sweden

8 Steubing B, Mutel C, Suter F, Hellweg S (2016) Streamlining scenario analysis and optimization of key choices in

9 value chains using a modular LCA approach. Int J Life Cycle Assess 21(4):510-522

10 SUDOE (2011) Olive oil life cycle assessment. OILCA tool. http://www.oilca.eu/oilcatool/. Accessed 12 May 2017

11 Tan RR, Culaba AB, Aviso KB (2008) A fuzzy linear programming extension of the general matrix-based life cycle

12 model. J Clean Prod16(13):1358-1367

13 Tolle DA, Lang RT, Becker JR, Pugsley KD, Vigon BW (1995) Life-cycle assessment: Public data sources for the

14 LCA practitioner. United States Environmental Protection Agency (EPA) publication. EP A530-R-95-009

15 Vázquez-Rowe I, Golkowska K, Lebuf V, Vaneeckhaute C, Michels E, Meers E, et al. (2015) Environmental

16 assessment of digestate treatment technologies using LCA methodology. Waste Manage 43:442-459

17 Vázquez-Rowe I, Villanueva-Rey P, Mallo J, De La Cerda JJ, Moreira MT, Feijoo G (2013) Carbon footprint of a

18 multi-ingredient seafood product from a business-to-business perspective. J Clean Prod 44:200-210

19 Zhou L, Tokos H, Krajnc D, Yang Y (2012) Sustainability performance evaluation in industry by composite

20 sustainability index. Clean Technol Environ Policy 14(5):789-803

21 Zhou P, Ang BW, Poh KL (2007) A mathematical programming approach to constructing composite indicators. Ecol

22 Econ 62(2): 291-297

1 Ziegler F, Emanuelsson A, Eichelsheim JL, Flysjö A, Ndiaye V, Thrane M (2011). Extended life cycle assessment of

2 southern pink shrimp products originating in senegalese artisanal and industrial fisheries for export to Europe. J Ind
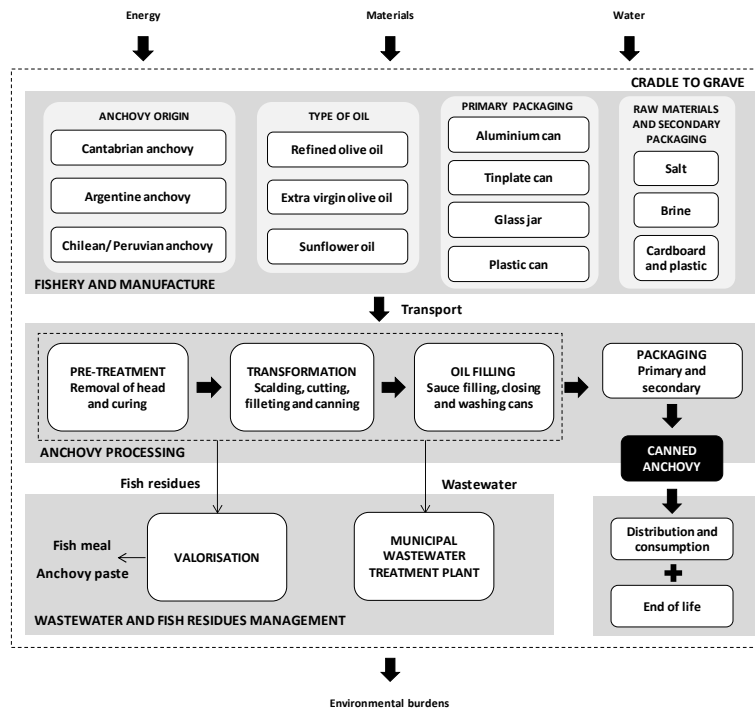
3 Ecol 15(4):527-538

4

1　**FIGURES**
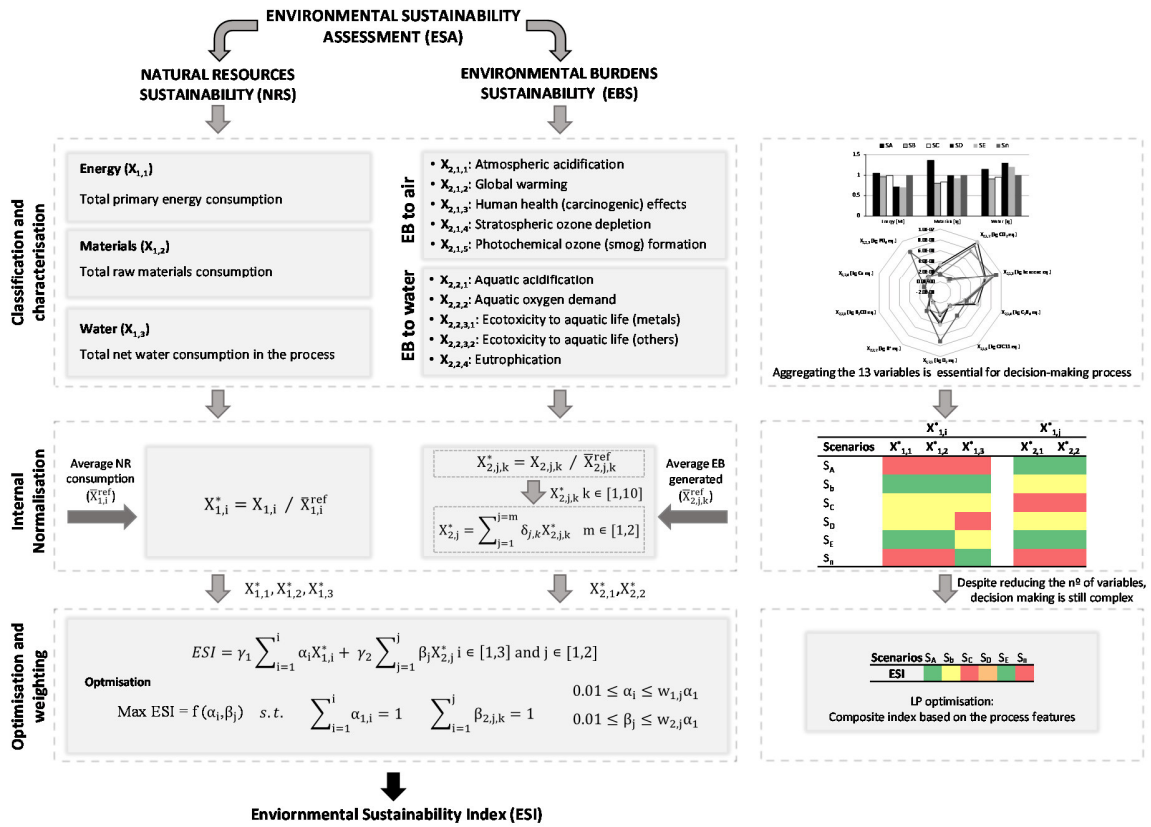
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3　　　**Fig. 1** Life Cycle Impact Assessment steps based on the use of the ESA method coupled to mathematical
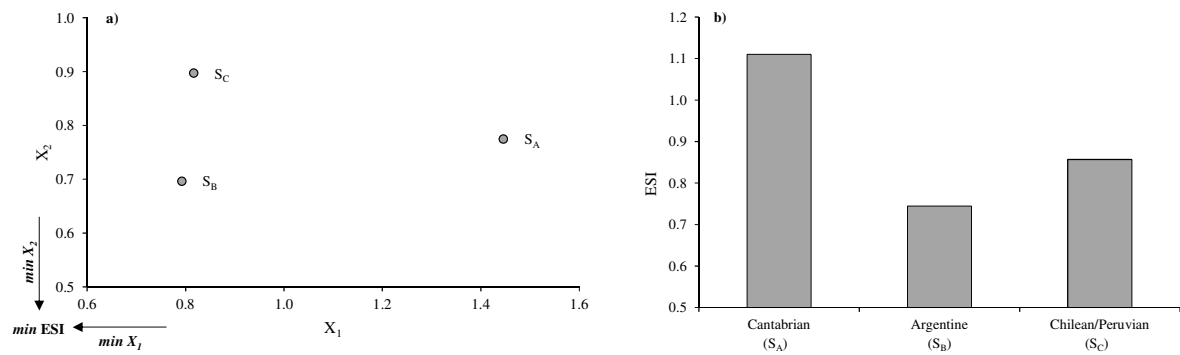
4　　　　　　　　　　　　　　　　　　　　optimisation

5

1

2   **Fig**. **2** Flow diagram of the anchovy canning system showing the system boundaries

3

Fig. 3 a) Bi-criteria scatterplot for natural resources consumption (X_1) vs environmental burdens (X_2) and b) Environmental Sustainability Index (ESI) for canned anchovy according to its origin. S_A: Cantabrian anchovy; S_B: Argentine anchovy; S_C: Chilean/Peruvian anchovy.
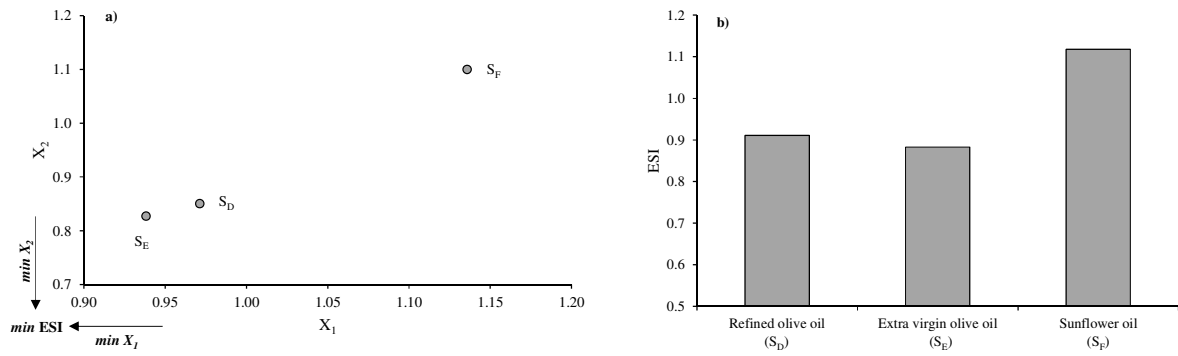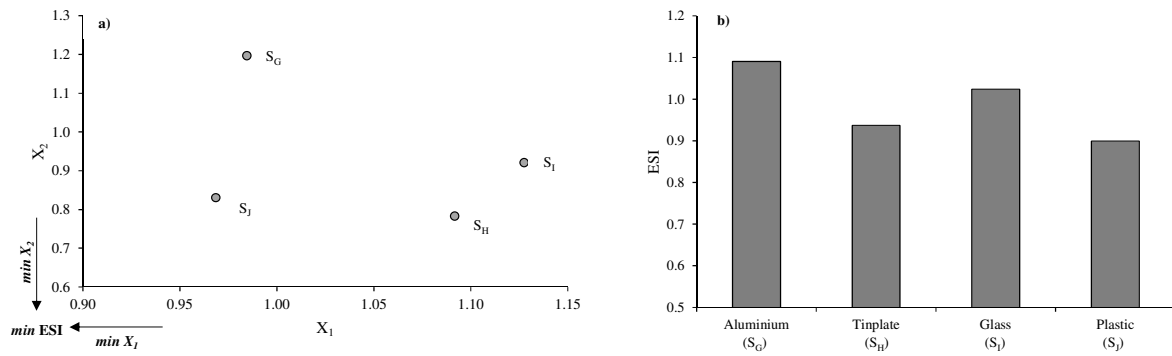
1   **Fig. 4** a) Bi-criteria scatterplot for natural resources consumption (X_1) vs environmental burdens (X_2) and b)
2   Environmental Sustainability Index (ESI) for canned anchovy using different types of oil. S_D: refined olive oil; S_E:
3   extra virgin olive oil; S_F: sunflower oil.

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Fig. 5 a) Bi-criteria scatterplot for natural resources consumption (X_1) vs environmental burdens (X_2) and Environmental Sustainability Index (ESI) for canned anchovy according to the type of oil used. S_D: refined olive oil; S_E: extra virgin olive oil; S_F: sunflower oil. B) ESI index for Cantabrian anchovy using different packaging. S_G: aluminium; S_H: tinplate; S_I: glass; S_J: plastic.
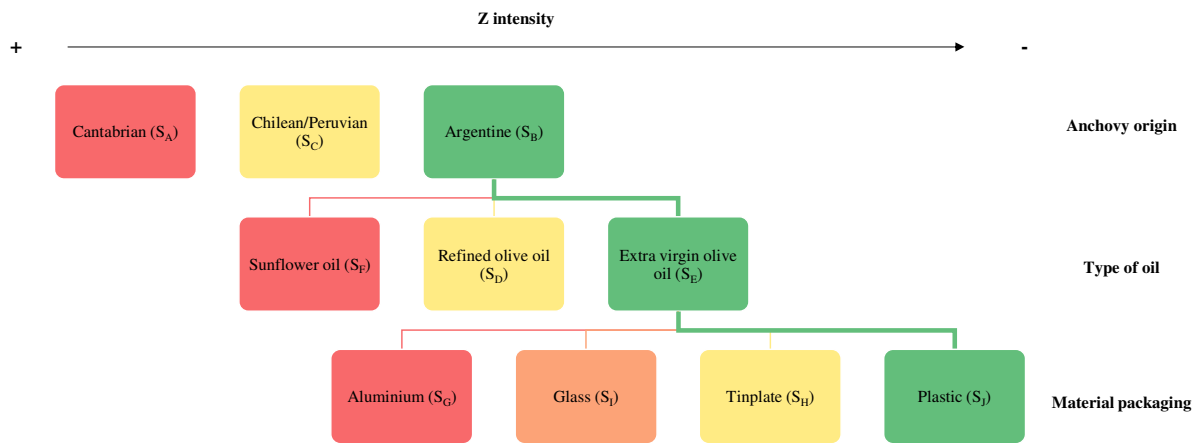
Fig. 6 Analytical decision tree for determining the best environmental alternatives in the anchovy canning industry under study.
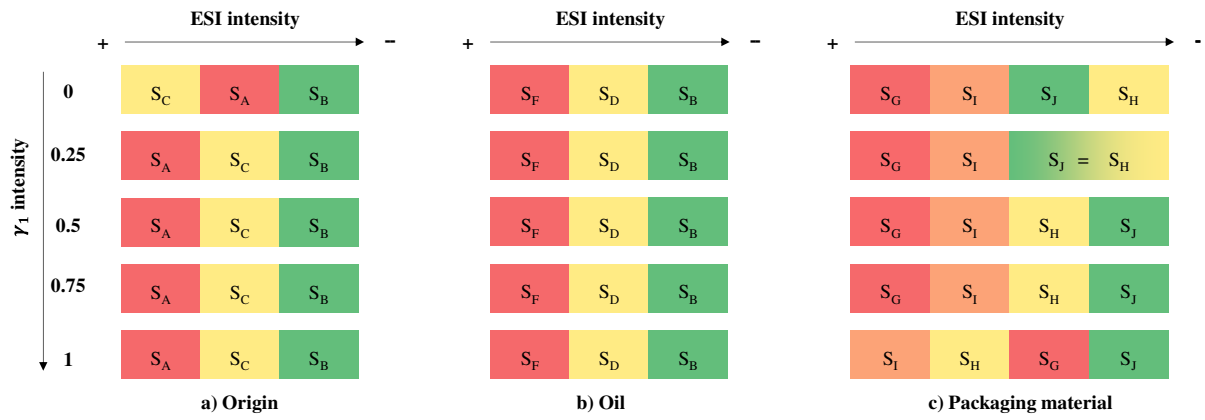
Fig. 7 Influence of the weighting factor γ_1 in the ESI index for the scenarios under study

1 **TABLES**

2 <center>**Table 1** Scenarios description.</center>

Origin	Oil	Packaging
Cantabrian	Refined olive oil (S_D)	Aluminium (S_G)
Argentine	Extra virgin olive oil (S_E)	Tinplate (S_H)
Chilean/Peruvian	Sunflower oil (S_F)	Glass (S_I)
		Plastic (S_J)

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1 **Table 2** Normalised environmental results for the scenarios under study. Colour coding is used to reflect NR and EB
2 intensity. Higher impacts are coloured in red, while lower impacts are coded with green.

Variable	Scenarios	Energy $(X^*_{1,1})$	Materials $(X^*_{1,2})$	Water $(X^*_{1,3})$	EB to air $(X^*_{2,1})$	EB to water $(X^*_{2,2})$
		Natural Resources (NR)			**Environmental Burdens (EB)**	
Origin	Cantabrian (S$_A$)	1.45	1.16	1.22	0.76	0.77
	Argentine (S$_B$)	0.76	0.90	0.87	0.78	0.51
	Chilean/Peruvian (S$_C$)	0.79	0.93	0.91	1.05	0.52
Oil	Refined olive oil (S$_D$)	0.94	1.03	1.10	0.92	0.64
	Extra virgin olive oil (S$_E$)	0.92	0.94	1.03	0.89	0.64
	Sunflower oil (S$_F$)	1.14	1.04	0.87	1.19	0.52
Packaging material	Aluminium (S$_G$)	0.99	0.91	0.96	1.36	0.75
	Tinplate (S$_H$)	0.92	1.03	1.12	0.81	0.54
	Glass (S$_I$)	1.13	1.13	0.96	0.99	0.57
	Plastic (S$_J$)	0.97	0.93	0.95	0.85	0.54

Intensity
+ ——→ -
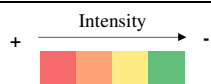
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1 **Table 3** $W_{1,i}$ and $w_{2,j}$ coefficients of the LP modelling (Eq. 6-7) for the scenarios under study

Variable	Scenarios	$w_{1,i}$		$w_{2,j}$	
		$w_{1,2}$	$w_{1,3}$	$w_{2,1}$	$w_{2,2}$
	Cantabrian (S_A)	0.249	0.172	5.12	1.82
Origin	Argentine (S_B)	0.171	0.129	3.24	1.37
	Chilean/Peruvian (S_C)	0.150	0.139	1.87	0.742
	Refined olive oil (S_D)	0.199	0.175	3.15	0.974
Oil	Extra virgin olive oil (S_E)	0.249	0.172	5.12	1.82
	Sunflower oil (S_F)	0.278	0.248	2.88	0.443
	Aluminium (S_G)	0.249	0.172	5.12	1.82
Packaging material	Tinplate (S_H)	5.24	5.74	2.06	0.209
	Glass (S_I)	1.04	0.061	0.892	0.177
	Plastic (S_J)	0.661	0.091	1.16	0.062

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1   **Table 4** Optimised results for the anchovy canning industry in Cantabria after applying the LP model

Variable	Scenarios	Energy α_1	Materials α_2	Water α_3	EB to air β_1	EB to water β_2
Origin	Cantabrian (S$_A$)	0.980	0.010	0.010	0.01	0.99
	Argentine (S$_B$)	0.769	0.132	0.099	0.703	0.297
	Chilean/Peruvian (S$_C$)	0.776	0.116	0.108	0.716	0.284
Oil	Refined olive oil (S$_D$)	0.728	0.145	0.127	0.764	0.236
	Extra virgin olive oil (S$_E$)	0.704	0.175	0.121	0.738	0.262
	Sunflower oil (S$_F$)	0.980	0.010	0.010	0.866	0.134
Packaging material	Aluminium (S$_G$)	0.980	0.010	0.010	0.737	0.263
	Tinplate (S$_H$)	0.147	0.010	0.843	0.908	0.092
	Glass (S$_I$)	0.980	0.010	0.010	0.835	0.165
	Plastic (S$_J$)	0.980	0.010	0.010	0.949	0.051

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Table 5. Single score for the scenarios under study using ReCIPe methodology

Variable	Scenarios	ReCIPe Single score
Origin	Cantabrian (S_A)	0.140
	Argentine (S_B)	0.083
	Chilean/Peruvian (S_C)	0.087
Oil	Refined olive oil (S_D)	0.140
	Extra virgin olive oil (S_E)	0.145
	Sunflower oil (S_F)	0.299
Packaging material	Aluminium (S_G)	0.140
	Tinplate (S_H)	0.127
	Glass (S_I)	0.145
	Plastic (S_J)	0.131

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