

Addressing the circularity and sustainability of different single-use and reusable crates used for fresh fruit and vegetables packaging

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ABSTRACT

This study assesses the sustainability of single-use (cardboard and wooden) crates and reusable plastic crates (RPC) from three perspective life cycle assessment (LCA), life cycle costing (LCC) and circular indicators: Material Circular Indicator (MCI), Material Reutilisation Score (MRS), and Circular Index (CI). A cradle-to-grave life cycle approach was applied to transporting 1 kg of fresh fruit or vegetables to the retailer point. The base case assumed single use for cardboard and wooden crates and 140 rotations for RPC. An optimised scenario was also assessed for RPC, assuming full recycling at its end-of-life. LCA, LCC, and circular indicators (MCI and CI) results indicated that RPC was the most sustainable and circular packaging. Increasing RPC recyclability to 100 % improved circularity scores, however its environmental impact slight increased. These results reflect that the food industry should adopt RPC use because of their higher economic, environmental performance, and circularity.

1. Introduction

Packaging is the act of enclosing or protecting the product using a container to aid its distribution, identification, storage, promotion, and usage. Packaging contributes positively to sustainable development as it improves food preservation and extends shelf life, thus reducing food waste. However, it may also cause significant degradation to the natural environment during its manufacturing and at the end of its useful lifetime (Liu et al., 2023).

Eurostat (2020) identified that 41.1 % (or 32.7 million tonnes) of the packaging waste produced in the European Union is paper and cardboard, 19.4 % is plastic (15.5 million tonnes), and 15.2 % is wood (12.1 million tonnes). Plastic packaging plays a vital role in the food industry as it is a durable material with a range of properties that ensure the safety and quality of food products (Rossi & Bianchini, 2022; Springle et al., 2022). Plastic pollution is a growing problem, with more than eight million tonnes of plastic entering the oceans each year (World Economic Forum, 2016).

Packaging management currently follows an unsustainable linear pattern ("take-make-use-waste") that needs to shift to a circular model (Liu et al., 2023). By 2018, the greenhouse gas (GHG) emissions from the whole packaging sector amounted to 59 million tonnes of CO₂ per year, a figure that is expected to rise due to economic growth and increased consumption of goods (European Commission, 2022). This growing trend can also be seen in the consumption of raw materials. In this context, the European Union (2020) has established the Circular Economy Action Plan and developed measures to increase the reusability or recyclability of packaging materials, with the aim of recovering at least 75 % of waste by 2030.

Kirchherr et al. (2017) defined Circular Economy (CE) as "an economic system that replaces the 'end-of-life' concept with reducing, alternatively reusing, recycling and recovering materials in production/distribution and consumption processes... ..with the aim to accomplish sustainable development, thus simultaneously creating environmental quality, economic prosperity, and social equity, for the benefit of present and future generations". This concept operates at different levels: the micro level relates to

Abbreviations: C2C, Cradle to cradle; CE, Circular Economy; C-indicator, Circular indicator; CI, Circular Index; EoL, End-of-life; GHG, Greenhouse gases; LCA, Life Cycle Assessment; LCC, Life Cycle Costing; LFI, Linear Flow Index; NPV, Net Present Value; MCI, Material Circularity Index; MDF, Medium Density Fibreboard; MRS, Material Reutilisation Score; PET, Polyethylene Terephthalate; PP, Polypropylene; RPC, Reusable Plastic Crate.

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the circularity of products, companies, and consumers; the meso level refers to production systems and industrial activities; whilst the macro level relates to its operation in cities, regions, nations and beyond. In the context of CE, circularity indicators (C-indicators) can be used to compare product and industry performance, to benchmark end-of-life alternatives during product manufacturing, to support changes in product design, and as a tool for policy formulation. Overall, C-indicators are a stepping stone towards more circular practices (Saidani et al., 2019).

Numerous C-indicators have been developed in recent years. For example, Akerman (2016) compiled 180 C-indicators for sustainability, and the Science Advisory Council of the European Academies identified more than 300 CE indicators (EASAC, 2016), intended to operate at the different levels (micro, meso, macro). In addition, specific C-indicators are still being developed, for instance to determine nutrient use efficiency in agricultural systems (van Loon et al., 2023) and to measure urban water circularity (Kakwani & Kalbar, 2022). In this regard, Saidani et al. (2019) developed “The C-indicators Advisor”, a tool that incorporates 55 indicators to help practitioners, decision-makers and policy-makers to identify and select the best C-indicators according to their specific needs. Corona et al. (2019) evaluated the quality of several C-indicators based on eight criteria (resources, emissions, material waste, recycled content, durability and utility, jobs, value added, and social well-being) and concluded that they do not always succeed in identifying the products and scenarios with the best environmental performance. Similarly, de Oliveira & Oliveira (2023) analysed eight C-indicators and concluded that they often fail to align with the three pillars of sustainability (economic, social, and environmental).

While there is no doubt that the use of circularity indicators is becoming more widespread, there is also a need to monitor their usefulness. For example, Niero & Kalbar, (2019) assessed the circularity of different beer packaging options; Roos et al. (2021) conducted a case study of glass and PET bottles; Vadoudi et al. (2022) studied the circularity of three-layer plastic packaging in a recycling loop. The results of these studies often showed that environmental performance and the circularity values do not align in terms of which alternative is better, which may call into question the effectiveness of these circularity indicators.

Regarding packaging used in bulk fruit and vegetable transport, most studies have only determined the environmental impact of single-use (cardboard and wooden crates) or reusable plastic crates (RPC) through LCA (Accorsi et al., 2014; ECOBILAN, 2000; FEFCO, 2022; Krieg et al., 2018; López-Gálvez et al., 2021). Few studies have additionally included the environmental Life Cycle Cost (E-LCC) (Accorsi et al., 2014; Albrecht et al., 2013). To date, none of these studies has assessed the material circularity of these crates. Therefore, the present study aims to contribute to the understanding of the sustainability of alternative bulk packaging materials through the combined use of circularity indicators, environmental life cycle analysis, and life cycle cost assessment.

2. Methodology

2.1. Environmental performance

2.1.1. Goal and scope

Environmental LCA of single-use (cardboard and wooden crates) and reusable plastic crates (RPC) was conducted according to 14040:2006 and 14044:2006 (ISO, 2006a, 2006b) standards and following the life cycle framework of Product Environmental Footprint (PEF) (European Commission, 2021a). The environmental performance of the crates refers to European management (cradle to grave), where the distribution distances from fruit and vegetable processing to retailer were obtained from weighted food production shares considering the top six producer countries (France, Italy, Greece, Netherlands, Poland, and Spain). Those weighted factors were the highest for Spain because it has the greatest fruit and vegetable production and export shares (FEFCO, 2022).

In total, this study evaluates four packaging scenarios: one for the cardboard crates, one for the wooden crates, and two for the RPC (Figure S1). All the crates were assumed to have the same dimensions ($60 \times 40 \times 20$ cm) and to contain the same amount of product (15 kg). Fig. 1 outlines the system boundaries of the conventional packaging scenarios: single-use crates (cardboard and wooden) and RPC, and the optimised scenario for RPC (with a recycling rate $R=100\%$). The value chain of the product systems considered the following three main stages: i) raw material production and crates manufacturing; ii) distribution and washing, which included the transport of these crates from the packaging centre to the retailer and the reverse transport for the RPC and a washing process, to be returned to the packaging centre; and iii) the transport of single-use and reusable containers for waste disposal and their management.

In terms of data collection, the weight and composition of the crates were primary data obtained from a representative fruit and vegetable cooperative in the Region of Murcia. This Spanish region is known for its favourable climate for producing a wide variety of fruit and vegetables and is a major supplier to many European countries. In fact, the value of fruit and vegetables in the EU27 exceeded €82 880 million in 2022, with Spain taking first place with 19 % of the total. This Spanish fresh fruit and vegetable sector destined almost 50 % of its production to exports, with the figure reaching over 70 % for certain greenhouse vegetables such as lettuce or lemons (MAPA, 2022).

However, the information on the transport of these crates was obtained from industry-specific reports on single use and reusable crates for the transport of bulk fruit and vegetables. Details of the transport of raw materials from source to the crate manufacturing plant were as follows: 467 km for cardboard (CCB & FEFCO, 2018), 100 km for wooden crates (ECOBILAN, 2000) and 1000 km for plastic crates (FEFCO, 2022). Once the crates had been manufactured, they were transported to the fresh vegetable packaging centre; the following distances were assumed: 55 km for cardboard crates (FEFCO, 2022), 100 km for wooden crates (ECOBILAN, 2000) and 370 km for plastic crates (FEFCO, 2022). All the crates were filled with 15 kg of fresh fruit or vegetables and distributed through a logistics platform (840 km) and were then sent to retailers (50 km). The distances refer to the European context using the average weighted kilometres of six European countries (FEFCO, 2022). The RPC were returned to the logistics platform and from there sent to the washing centre (380 km) to be reused in the fresh vegetable packaging centre (FEFCO, 2022). The demands of energy, water, disinfectant, and detergent for RPC cleaning were obtained from (Tua et al., 2019). The average transport distances to each of the end-of-life processes were obtained from (FEFCO, 2022; MITECO, 2020).

The conventional scenario considers the end-of-life rates for packaging materials reported by Eurostat (2020); the optimised scenario considers 100 % recycling of the RPC ($R:100\%$) at the end of its useful life, as claimed by the company behind the RPC system ARECO (2023).

The circular footprint formula (CFF) was used to determine the environmental impact of the crate's recycled materials and its end-of-life. This was done by applying the default allocation values which appear in Annex C (European Commission, 2021b). The background data of raw material production, related emissions to transport, and waste management were obtained from the Ecoinvent 3.9 database (Ecoinvent, 2023) and especially the ELCD 3.2 database (European Commission, 2018a) for incineration with energy recovery.

The reusable crates were used for 140 rotations (the average lifetime of a reusable plastic crate) (Albrecht et al., 2022). Each time the RPC was used, a new cardboard or wooden crate had to be used. The functional unit used in this study was the transport of 1 kg of fresh fruit or vegetables to the retailer point in either cardboard, wooden, or reusable plastic crates. The weight of crates needed to satisfy the functional unit was 3.33×10^{-2} kg cardboard crate, 9.94×10^{-2} kg wooden crate and 9.05×10^{-4} kg of RPC.

The EF 3.1 impact assessment method (adapted) v.1.0 (Fazio et al.,

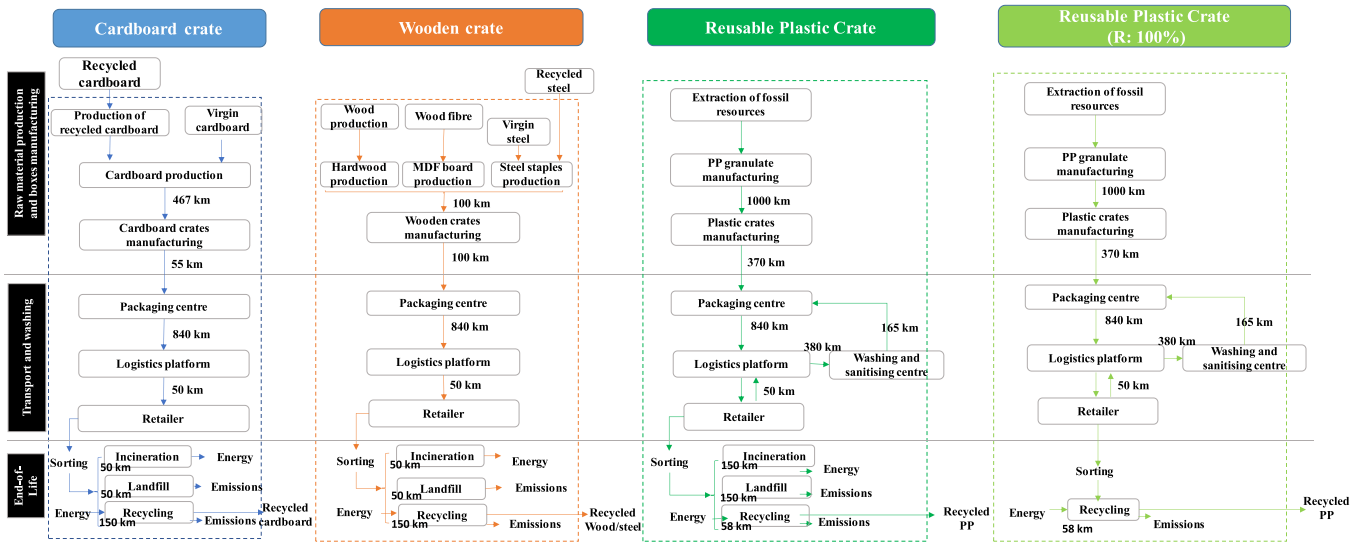


Fig. 1. System boundaries applied to carry out the process-based LCA for conventional packaging scenarios: single-use crates (cardboard and wooden) and RPC (Reusable Plastic Crate), and the optimised scenario for RPC (R:100 %).

2018) and SimaPro 9.5 software (PRé Consultant, 2023) were used to calculate the impacts of the analysed product systems.

2.1.2. Life cycle inventory

The Life Cycle Inventory for the environmental assessment is detailed in the [supplementary information](#). (Table S1). Additionally, Table S2 shows the materials, weights and recycled content, and Table S3 the processed used for crate manufacturing. The cardboard crate manufacturing process was modified to obtain the required recycled content in cardboard (see Table S4). Finally, Table S5 shows the shares of management options and processes selected.

2.2. Economic considerations

The environmental Life Cycle Costing (E-LCC) of single-use crates and RPC was carried out for a time horizon of 11 years (N), with an average lifespan of 140 rotations for RPC (Albrecht et al., 2022), between 2023 and 2034. Future cash flows were discounted to net present value (NPV) using Eq.1, where i represents the real discount rate (nominal minus inflation) (5 %), t is the cash flow period (from 0 to 11 years) and R_t is the annual net cash flow (€). This 5 % rate is considered appropriate for the evaluation of a medium to long-term private project with a strong environmental driver.

$$NPV(i, 11) = \sum_{t=0}^{11} \frac{R_t}{(1+i)^t} \quad (1)$$

The procurement cost of single-use crates and the leasing of RPC were obtained by interviewing a local fruit distributor. Additionally, the costs of other expenses were also quantified for Spain, including road transport (MITMA, 2023), water (INE, 2020), electricity (OCU, 2023), disinfectant and detergent, and waste management (Ecoembes, 2023). A complete life cycle inventory analysis for economic assessment is available in the [supplementary information](#) (Table S6).

2.3. Description of circularity indicators

The following three C-indicators were selected to assess CE at micro level of single-use cardboard and wooden crates and of reusable plastic crates (RPC): i) Material Circularity Indicator developed by (Ellen MacArthur Foundation, Granta Design (2019)), which is the most cited and widely used CE indicator (Garza-Reyes et al., 2019; Linder et al., 2017; Vadoudi et al., 2022); ii) Material Reutilisation Score, which

describes the criteria for the well-known Cradle to Cradle® Certified Product Standard (C2C, 2016) (Ross Lindgreen et al., 2021); and iii) Circular Index described by (Cullen, 2017), which includes both the loss of material quantity as well as quality (in terms of energy).

2.3.1. Material Circular Indicator (MCI)

The Material Circular Indicator (MCI) was developed by (Ellen MacArthur Foundation, Granta Design (2019)). It is fundamentally based on a combination of three product features: the mass of virgin material (V), the mass of non-recoverable waste (W) and the utility factor (F(X)). MCI is a dimensionless indicator that ranges from 0 to 1. A value of 0 describes a linear product, which is produced by virgin material and landfilled at its end-of-life (EoL) whilst MCI = 1 represents a fully circular product that is recycled or reused at 100 %.

As shown below, Material Circular Indicator of a Product (MCI^*_p) is obtained by multiplying the Linear Flow Index (LFI) and the Utility Factor (F(X)), (see Eq.2). The Utility Factor (see Eq.3) depends on the practicality or usefulness of a product (see Eq.4), where L represents its actual average lifetime, L_{av} is the average lifetime of an industry-average product of the same type, U is the actual average number of functional units achieved during the use phase of a product and U_{av} is the number of functional units achieved during the use phase of an industry-average product of the same type. In the present study the parameter L/L_{av} was not considered, it was calculated based on the number of functional units provided by each of the packaging alternatives (U/U_{av}).

Likewise, LFI depends on the following parameters: the mass of virgin material (V), the mass of unrecoverable waste (W), the mass of unrecoverable waste generated when producing recycled feedstock for a product (W_F), the mass of unrecoverable waste generated in the process of recycling parts of a product (W_C), and the mass of the product (M) (see Eq.5). V is obtained by subtracting the fraction of the product obtained from recycled sources (F_R), the fraction of the product obtained from reused sources (F_U), and the fraction of the total that derives from biological feedstock from sustainable production (see Eq.6). In order to calculate the W (see Eq. 7) the W_0 must first be calculated (see Eq.8), where C_R is the fraction of mass of a product being collected to be recycled, C_U is the fraction of a product that is to be reused, C_C is the fraction of a product used for composting, and C_E is the fraction being collected for energy recovery. C_E (see Eq. 9) is obtained by subtracting E_E , the efficiency of the energy recovery process for biological materials, from B_C , the carbon content of a biological material. Eq. 10 reveals how to obtain the E_E parameter, where E_R is the energy recovered; HHV is the

higher heating value, and M_B represents the mass of eligible biological material. For W_F calculation (see Eq. 11), where E_F is the efficiency of the recycling process used to produce recycled feedstock; and for W_C calculation (see Eq. 12), where E_C is the efficiency of the recycling process used for the portion of a product collected for recycling.

$$MCI_p^* = 1 - LFI \times F(X) \quad (2)$$

$$F(X) = \frac{0.9}{X} \quad (3)$$

$$X = \left(\frac{L}{L_{av}} \right) \times \left(\frac{U}{U_{av}} \right) \quad (4)$$

$$LFI = \frac{V + W}{2M + \frac{W_F - W_C}{2}} \quad (5)$$

$$V = M(1 - F_R - F_U - F_S) \quad (6)$$

$$W = W_0 + \frac{W_F + W_C}{2} \quad (7)$$

$$W_0 = M(1 - C_R - C_U - C_C - C_E) \quad (8)$$

$$C_E = (E_E - B_C) \quad (9)$$

$$E_E = \left(\frac{E_R}{HHV \times M_B} \right) \quad (10)$$

$$W_F = M \times \frac{(1 - E_F)F_R}{E_F} \quad (11)$$

$$W_C = M \times (1 - E_C) \times C_R \quad (12)$$

2.3.2. Material Reutilisation Score (MRS)

The Cradle to Cradle certification (C2C, 2016) developed the Material Reutilisation Score (MRS). This indicator assesses the circularity based on material composition, considering the following five material categories: recyclable, biodegradable, compostable, recycled, and rapidly renewable material (i.e., a material grown and harvested less than 10 years ago or Forest Stewardship Council certified wood products even if they were harvested more than 10 years ago) (see Eq. 13). The MRS indicator is a nondimensional parameter that ranges from 0 to 100.

$$MRS = \frac{\left[\frac{\% \text{recycled or rapidly renewable product content}}{3} \right] + 2 \left[\frac{\% \text{of product recyclable or biodegradable/compostable}}{3} \right]}{\times 100} \quad (13)$$

2.3.3. Circular Index (CI)

The Circular Index (CI) was developed by (Cullen, 2017) and consists of two factors, as described in Eq. (14), which represent the quantity of material conservation (α) (see Eq. 15) and the quality of material conservation (β) (see Eq. 16). For both parameters, a value of 1 would describe a totally circular scenario where all the material is recoverable at its EoL ($\alpha=1$) and there is no loss of material quality in each recovery cycle ($\beta=1$). The CI is a dimensionless value that ranges from 0 to 1.

$$CI = \alpha \times \beta \quad (14)$$

$$\alpha = \frac{\text{recovered EOL material}}{\text{total material demand}} \quad (15)$$

$$\beta = 1 - \frac{\text{energy required to recover material}}{\text{energy required for primary production}} \quad (16)$$

2.3.4. Assumptions of set parameters

- It was assumed that the sustainable fraction and rapidly renewable material were equivalents. For wooden and cardboard products it represented 45.0 % of the total (MITECO, 2021).
- The efficiency of energy recovery in Europe was estimated to be 16.5 % (Equanimator Ltd, 2023).
- The waste management options are reported in the supplementary information, and the data is representative for Europe (Eurostat 2020).
- For MCI the efficiency of the recycling process was estimated to be: 88.5 % for cardboard (Merrild et al., 2008; Wang et al., 2012), 65.0 % for wood (European Commission, 2018b), 91.0 % for MDF (Zeng et al., 2018), 85.0 % for steel (Broadbent, 2016), and 71.0 % for PP (Antonopoulos et al., 2021).
- Regarding crates utility (U), this was obtained assuming that the number of functional units was 1 for single-use crates (cardboard and wooden crates), while 140 was estimated for RPC (Albrecht et al., 2022). For all the crates, the number of functional units achieved during the use of an industry-average was 1.
- For CI, the electricity demand for material recycling was as follows: 2.8×10^{-1} kWh/kg cardboard (CCB & FEFCO, 2015); 1.5 kWh/kg wood (Piekariski et al., 2012); 2.08×10^{-2} kWh/kg MDF (Stubdrup et al., 2016); 2.6×10^{-2} GJ/kg steel (Johnson et al., 2008), and 1.80 kWh/kg for PP (Shan et al., 2023).

3. Results and discussion

3.1. Environmental performance

The weighted results known as a single score (European Commission, 2021b), covering 16 midpoint categories was assessed. Based on these results, the interpretation is focused on the most relevant categories, which represented 80 % of the single score, as follows: Climate change, Photochemical ozone formation, Particulate matter, Acidification, Freshwater eutrophication, Land use, Resource use - mineral and metals, and Resource use - fossil. (Table 1). The environmental impact of all impact categories is shown in the supplementary information (Table S8). For all impact categories, the highest environmental impacts were produced by the wooden crate. For example, the results for climate change per FU (1 kg of fresh fruit or vegetables at the retailer point) were 0.11 kg CO₂ for the wooden crate, 0.05 kg CO₂ for the cardboard crate, and 0.04 kg CO₂ for the RPC. For the optimised scenario (RPC 100 % recycled), the environmental impact decreased slightly in almost all the impact categories.

As shown in Fig. 2, the dominant life cycle stage for the wooden and cardboard crates was the production of the raw materials in both cases. The production of the MDF in the case of the wooden crates and the production of the corrugated fluting medium and the linerboard in the case of the cardboard crates are very intensive in terms of their energy consumption and use of potentially toxic chemicals (Albrecht et al., 2013). The second life cycle stage was wood and cardboard recycling, owing to the electricity consumption of the recycling process.

In contrast, transport was by far the largest life cycle stage for RPC in all the impact categories (except freshwater eutrophication) due to the accumulated emissions generated by the transport of plastic crates throughout their life cycle from retail to washing. This concurs with the findings reported by Albrecht et al. (2013). Energy consumption during the washing of RPC was the second largest contributor. Compared to single-use crates, the environmental impact of RPC production was lower (e.g., 2.8×10^{-3} kg CO₂eq of RPC/FU, compared to 3.5×10^{-2} kg CO₂eq of cardboard crate/FU and 9.8×10^{-2} kg CO₂eq of wooden crate/FU). The contribution of the end-of-life stage of the RPC was negligible, as were the emission credits associated with the recycling of plastic materials.

The environmental performance of the optimised RPC scenario

Table 1

Characterised environmental impact and single score of single-use crates (cardboard and wooden) and RPC, per FU (1 kg of vegetables at retailer point) of transported fruits and vegetables.

Impact category	Unit	Cardboard crates ^a	Wooden crates ^a	RPC ^a	RPC (R:100 %) ^b
Climate change	kg CO ₂ eq	4.52×10^{-2}	1.15×10^{-1}	3.52×10^{-2}	3.49×10^{-2}
Photochemical ozone formation	kg NMVOC eq	2.16×10^{-4}	6.03×10^{-4}	1.25×10^{-4}	1.23×10^{-4}
Particulate matter	disease incidence	4.43×10^{-9}	1.05×10^{-8}	2.30×10^{-9}	2.29×10^{-9}
Acidification	mol H ⁺ eq	1.39×10^{-4}	5.78×10^{-4}	1.00×10^{-4}	9.92×10^{-4}
Eutrophication, freshwater	kg P eq	2.40×10^{-5}	4.91×10^{-5}	6.22×10^{-6}	6.34×10^{-6}
Land use	Pt	3.72×10^0	1.68×10^1	2.70×10^{-1}	2.70×10^{-1}
Resource use, fossils	MJ	6.38×10^{-1}	2.00×10^0	5.70×10^{-1}	5.55×10^{-1}
Resources use, minerals and metals	kg Sb eq	1.99×10^{-7}	7.54×10^{-7}	1.50×10^{-7}	1.50×10^{-7}
Single score	mPt	4.91×10^{-3}	1.42×10^{-2}	2.94×10^{-3}	2.91×10^{-3}

^a End-of-life: European waste management shares (Eurostat, 2020a).

^b End-of-life: 100 % recyclable. RPC: Reusable Plastic Crate.

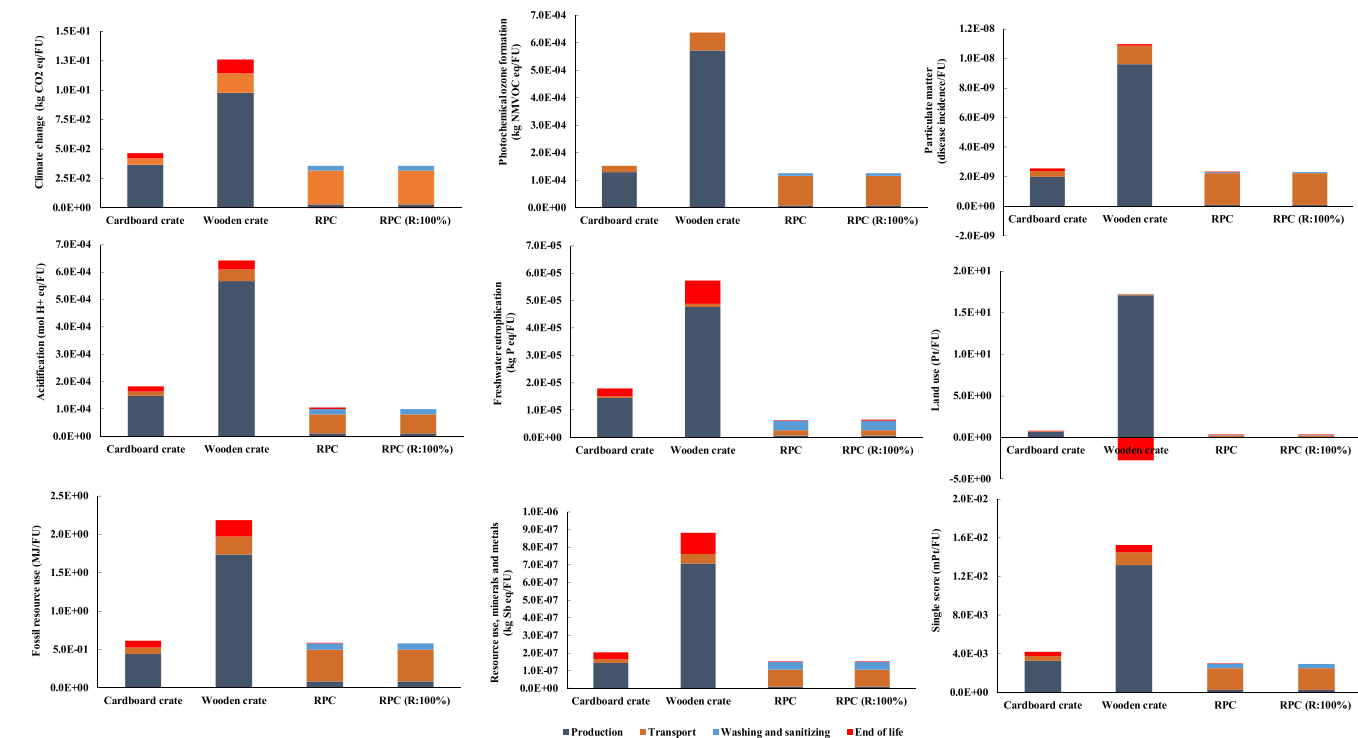


Fig. 2. Environmental impact per FU (1 kg of fresh fruit or vegetables) for single-use crates (cardboard and wooden) and RPC (Reusable Plastic Crate).

(R:100 %) was very similar to that of the conventional RPC scenario, with small decreases in the categories of climate change (0.9 %), photochemical ozone formation (1 %), particulate matter (0.8 %), acidification (1 %), resource fossil use (2.6 %), and the single score (1.1 %). On the other hand, the impacts on land use (0.1 %) freshwater eutrophication (1.9 %) and minerals and metals resource use (0.1 %) increased slightly, due to the energy required to recycle the PP. The reason for the limited benefits of recyclability was the low contribution of the end-of-life stage to the whole impact of the RPC, where the environmental impact of this packaging option was dominated by its transport stage.

There are many comparative LCA studies in the literature between single-use and reusable crates, although the methodological options used in the studies do differ. Most of those studies agree that RPC is the most sustainable packaging (Accorsi et al., 2014; ECOBILAN, 2000; FEFCO, 2022; Krieg et al., 2018; López-Gálvez et al., 2021), with carbon footprints ranging from 1.6×10^{-3} to 8.1×10^{-2} kgCO₂ eq/kg of product. The aforementioned studies found that the carbon footprint of the cardboard crate ranged from 2.79×10^{-2} to 1.47×10^{-1} kgCO₂ eq/kg of

transported product, which is in line with the result obtained in the present study. Regarding the environmental impact of the wooden crate, it was similar to that described by López-Gálvez et al. (2021); overall the environmental impact of this crate was higher due to the consumption of MDF. However, the carbon footprints of wooden crates obtained by ADEME (2000) and Accorsi et al. (2014) do not align with our results (1.16×10^{-3} to 1.29×10^{-2} kgCO₂ eq/kg of product transported), this is likely due to the difference in the crate's material (solid wood). The carbon footprint's variability among crates is due to two main factors: firstly, the number of rotations of the RPC, which ranged from 24 (FEFCO, 2022) to 150 (López-Gálvez et al., 2021); and secondly, the transport distance of single use and RPC from packaging centre to retailer, which ranged from 200 km (Accorsi et al., 2014) to 890 km (FEFCO, 2022).

3.1.1. Sensitivity analysis: number of rotations for RPC

A sensitivity analysis was carried out to evaluate the variability of the number of rotations of the RPC on the climate change. Figure S2 shows that a minimum of five rotations of the RPC or RPC (R:100 %)

would be required in order to obtain a carbon footprint lower than that for the wooden crates. This would increase to 24 rotations for RPC (R:100 %) or 29 rotations for RPC in order to obtain a lower carbon footprint than for the cardboard crate. López-Gálvez et al. (2021) reported that a minimum of 15 rotations for RPC would be required to obtain a lower environmental footprint than for the wooden and cardboard crates. The main difference between that study and the present study lies in the assumed transported network. In the present study, the assumed distance between the packaging centre and the retailer was 890 km, whilst López-Gálvez et al. (2021) considered a lower figure of 500 km. Therefore, the larger the RPC network, the higher the number of rotations required to reduce their impact in comparison with single-use crates.

3.2. Economic performance

Fig. 3 shows that the wooden crate was the costliest ($6.7 \times 10^{-2} \text{ €}_{2023}/\text{FU}$), followed by the cardboard crate ($3.8 \times 10^{-2} \text{ €}_{2023}/\text{FU}$) and the RPC ($2.5 \times 10^{-2} \text{ €}_{2023}/\text{FU}$). The cost profile was similar for the cardboard and wooden crates. Life cycle costs of single-use crates were dominated by the production stage (86 %), followed by transport (7–12 %) and end-of-life (2–7 %). In contrast, for RPC, the costs were dominated by the transportation stage (47 %), followed by end-of-life (25 %), and crate leasing (15 %). It must be noted that the costs presented are specific to Spain. Therefore, generalising these costs to the European level may not be appropriate. However, Spain ranks among the countries with the highest agricultural production and exports in Europe, suggesting that these costs are particularly sensitive to the Spanish situation.

Albrecht et al. (2013) reported that RPC were the least costly packaging ($3 \times 10^{-2} \text{ €}_{2014}/\text{kg}$), and that the most economically intensive were the cardboard crates ($8.0 \times 10^{-2} \text{ €}_{2014}/\text{kg}$). In that study, the production phase was the largest contributor to the total cost of single-use crates. Our results differed slightly from those obtained by Accorsi et al. (2014), who reported that the use of RPC resulted in a cost of 0.058 €/kg for the delivered goods, as their cost was obtained from 50 rotations.

3.3. Scores of material circularity

The parameters obtained for C-indicators calculations are reported in Table 2. The circularity values represented on a scale of 0–100 % are shown in Fig. 4. The results show a remarkable variability in the profile of the packaging scenarios depending on the circularity indicator used. This is mainly due to the fact that MCI and MRS are material-based metrics (Niero & Kalbar, 2019), whilst CI incorporates a material quality conservation factor (β), related to the energy inputs for material upcycling and primary production (Cullen, 2017).

In the conventional scenarios (single-use cardboard and wooden crates, and RPC) that consider the waste management rates described in European statistics for the end-of-life stage (Table S5), the highest circularity scores were obtained using the MCI indicator. That score was particularly high for RPC and cardboard crates. In the case of RPC, the high MCI score was attributed to the fact that this C-indicator considered reusable packaging utility, enhancing the circularity score when it is used more than average functional units (Uav). For the cardboard crate, the high MCI was due to the inclusion of processes such as: recycled content, biological feedstock from sustained production, and incineration with energy recovery, which led to lower waste and therefore higher circularity scores.

In contrast, the CI indicator produced the lowest circularity scores for single-use packaging, and the MRS indicator gave the lowest score for the wooden crate. Fig. 4 also shows the circularity values for the optimised RPC scenario, which considers full recyclability of plastics (R:100 %). All circular indicators produced higher scores for the optimised RPC scenario, particularly in the case of CI.

For the conventional scenarios, a very high variability was observed in the circularity scores generated by the different indicators for the cardboard crate option. The cardboard crate achieved the second highest circularity for both MCI (90.1 %) and MRS (55.3 %). These scores (Table 2) were related to the high recycled content (88 %) (Zampori & Pant, 2019), recyclability at the end of the useful life (81.5 %) (Eurostat, 2020a) and, in the case of MCI, the high efficiency of the cardboard recycling process (88.5 %) (Merrild et al., 2008; Wang et al., 2012). Moreover, the MCI score included incineration with energy recovery, which reduced the proportion of non-recyclable waste (W_o), and the biological origin of the material, thereby reducing the proportion of virgin cardboard and making this product more circular. On the other hand, the CI score was the lowest (26.2 %) for the cardboard crate because the cardboard quality decreased during its recycling ($\beta = 0.32$; see Table S5), making the product less circular. Delgado-Aguilar et al., (2015) reported that paper fibres may be recycled up to three times whilst retaining their physical properties.

The wooden crate was the third in terms of circularity for both MCI (46.6 %) and MRS (21.8 %). The MCI score also included energy recovery from incineration and the proportion of wood produced using sustainable practices, which made that C-indicator the highest. On the other hand, the MRS score was lower due to the low amount of wood recycled in European countries, around 39 %. This low percentage was supported by Faraca et al. (2019), who reported that the recycling of municipal wood waste represented a major challenge, while industrial wood waste was easier to recycle and reuse. This low recovery rate and the low-quality factor of MDF ($\beta = 0.02$) were the main factors for the lowest CI score (10.3 %).

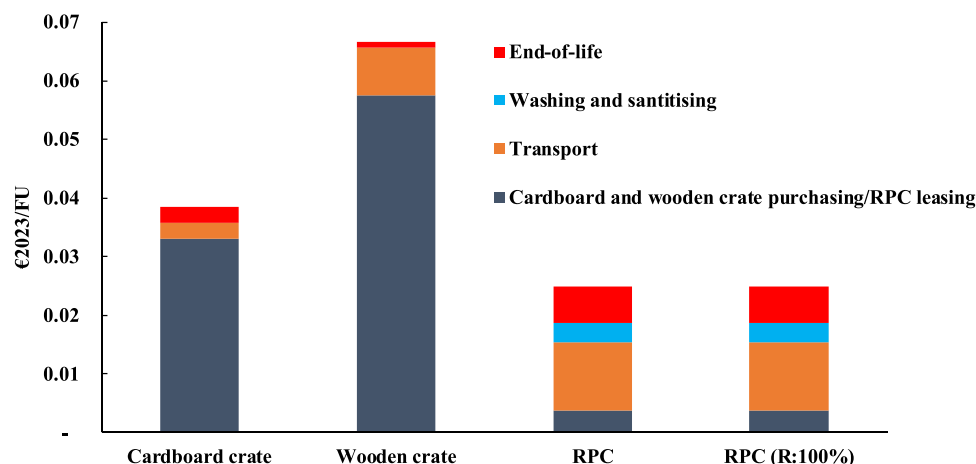
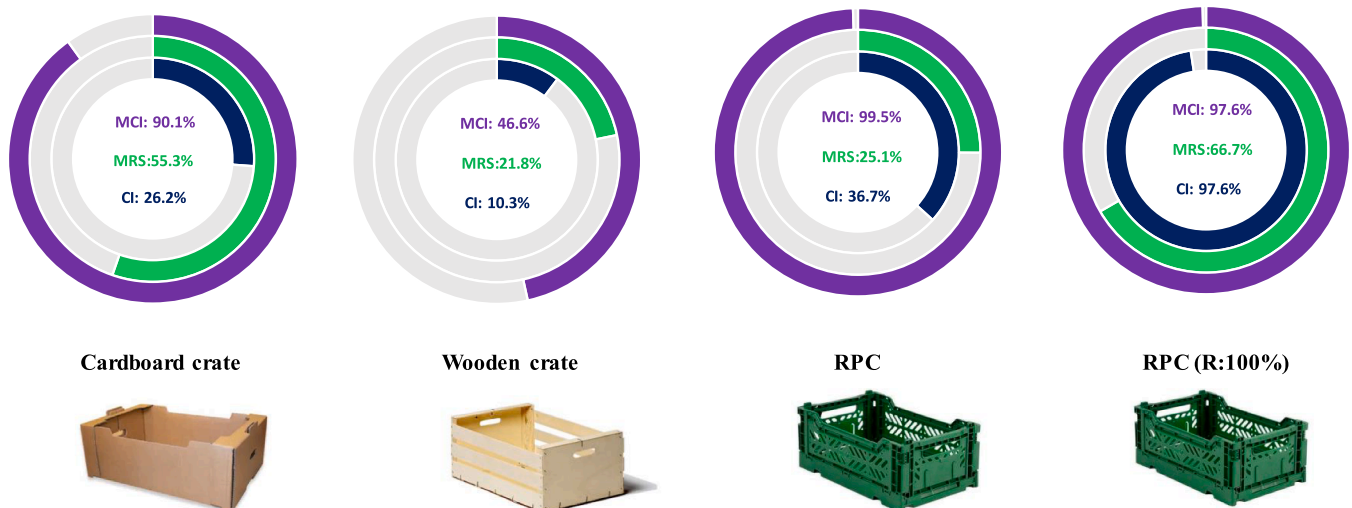


Fig. 3. Life cycle cost per FU (1 kg of fresh fruit or vegetables) for single-use crates (cardboard and wooden) and RPC (Reusable Plastic Crate).

Table 2

Set of parameters used for the calculation of circular indicators and results of MCI, MRS, and CI.

C-indicators	Parameters	Cardboard crate ^a	Wooden crate ^a	Hardwood ^a	MDF ^a	Steel ^a	RPC ^a	RPC (R:100 %) ^b
Material Circular Indicator (MCI)	M	5.00×10^{-1}		4.57×10^{-1}	$1.03 \times 10^{+0}$	2.92×10^{-3}	$1.90 \times 10^{+0}$	$1.90 \times 10^{+0}$
	F _R	8.80×10^{-1} <i>a</i>		-	-	2.00×10^{-1} <i>a</i>	-	-
	F _S	1.20×10^{-1} <i>b</i>		4.52×10^{-1}	4.07×10^{-1}	-	-	-
	V	-		2.50×10^{-1}	6.11×10^{-1}	2.34×10^{-3}	$1.90 \times 10^{+0}$	$1.90 \times 10^{+0}$
	C _R	8.15×10^{-1} <i>c</i>		3.19×10^{-1} <i>c</i>	3.19×10^{-1} <i>c</i>	9.84×10^{-1} <i>c</i>	3.76×10^{-1} <i>c</i>	$1.00 \times 10^{+0}$ <i>m</i>
	C _U	-		-	-	-	-	-
	C _C	-		-	-	-	-	-
	C _E	6.68×10^{-2}		8.41×10^{-2}	1.30×10^{-1}	-	-	-
	E _E	$1.39 \times 10^{+1}$ <i>d</i>		$2.56 \times 10^{+0}$	$1.94 \times 10^{+0}$	-	-	-
	B _C	4.80×10^{-3} <i>e</i>		3.29×10^{-2} <i>f</i>	6.67×10^{-2} <i>f</i>	-	-	-
	W _O	5.91×10^{-2}		2.73×10^{-1}	5.68×10^{-1}	4.67×10^{-5}	$1.19 \times 10^{+0}$	-
	Ec	8.85×10^{-1} <i>g</i>		6.50×10^{-1} <i>h</i>	9.10×10^{-1} <i>i</i>	8.50×10^{-1} <i>j</i>	7.10×10^{-1} <i>k</i>	7.10×10^{-1} <i>k</i>
	W _C	4.69×10^{-2}		5.10×10^{-2}	2.96×10^{-2}	4.31×10^{-4}	1.69×10^{-1}	4.50×10^{-1}
	W _F	5.72×10^{-2}		-	-	1.03×10^{-4}	-	-
	W	1.11×10^{-1}		2.98×10^{-1}	5.83×10^{-1}	3.14×10^{-4}	$1.27 \times 10^{+0}$	2.25×10^{-1}
	LFI	1.11×10^{-1}		6.18×10^{-1}	5.84×10^{-1}	4.67×10^{-1}	8.53×10^{-1}	5.94×10^{-1}
	U	1.00×10^0		$1.00 \times 10^{+0}$	$1.00 \times 10^{+0}$	$1.00 \times 10^{+0}$	$1.40 \times 10^{+2}$ <i>l</i>	$1.40 \times 10^{+2}$ <i>l</i>
	U _{av}	1.00×10^0		$1.00 \times 10^{+0}$	$1.00 \times 10^{+0}$	$1.00 \times 10^{+0}$	$1.00 \times 10^{+0}$	$1.00 \times 10^{+0}$
	X	1.00×10^0		$1.00 \times 10^{+0}$	$1.00 \times 10^{+0}$	$1.00 \times 10^{+0}$	$1.40 \times 10^{+2}$	$1.40 \times 10^{+2}$
	F(X)	9.00×10^{-1}		9.00×10^{-1}	9.00×10^{-1}	9.00×10^{-1}	6.43×10^{-3}	6.43×10^{-3}
	MCI	9.01×10^{-1}	4.66×10^{-1}	4.44×10^{-1}	4.75×10^{-1}	5.80×10^{-1}	9.95×10^{-1}	$9.96 \times 10^{+0}$
Material Reutilisation Score (MRS)	% recycled material	8.80×10^{-1} <i>a</i>		-	-	2.00×10^{-1} <i>a</i>	-	-
	% rapidly renewable material	1.20×10^{-1} <i>b</i>		4.52×10^{-1} <i>b</i>	4.07×10^{-1} <i>b</i>	-	-	-
	% recyclable	8.15×10^{-1} <i>c</i>		3.19×10^{-1} <i>c</i>	3.19×10^{-1} <i>c</i>	9.84×10^{-1} <i>c</i>	3.76×10^{-1} <i>c</i>	1.00×10^0 <i>m</i>
Circular Index (CI)	MRS	5.53×10^{-1}	$2.18 \times 10^{+1}$	$2.17 \times 10^{+1}$	$2.17 \times 10^{+1}$	$6.58 \times 10^{+1}$	$2.51 \times 10^{+1}$	$6.67 \times 10^{+1}$
	α	8.15×10^{-1} <i>c</i>		3.19×10^{-1} <i>c</i>	3.19×10^{-1} <i>c</i>	9.84×10^{-1} <i>c</i>	3.76×10^{-1} <i>c</i>	$1.00 \times 10^{+0}$ <i>m</i>
	Energy required to recover material	2.78×10^{-1}		$1.59 \times 10^{+3}$	$2.08 \times 10^{+2}$	$2.60 \times 10^{+1}$	$1.80 \times 10^{+0}$	$1.80 \times 10^{+0}$
	Energy required for primary production	4.10×10^{-1}		$1.80 \times 10^{+5}$	$2.12 \times 10^{+2}$	$7.90 \times 10^{+1}$	$7.36 \times 10^{+1}$	$7.36 \times 10^{+1}$
	β	3.21×10^{-1} <i>n</i>		9.91×10^{-1}	2.02×10^{-2}	6.71×10^{-1} <i>q</i>	9.76×10^{-1} <i>r</i>	9.76×10^{-1} <i>r</i>
	CI	2.62×10^{-1}	1.03×10^{-1}	3.16×10^{-1}	6.45×10^{-3}	6.600×10^{-1}	3.67×10^{-1}	9.76×10^{-1}

^a End-of-life: European statistics.^b End-of-life: Recyclable at 100 %. *a*: Annex C of PEF method described by [Zampori & Pant, \(2019\)](#). *b*: [MITECO \(2021\)](#). *c*: [Eurostat \(2020b\)](#). *d*: [Ioelovich \(2017\)](#). *e*: [Francou et al. \(2008\)](#). *f*: [Arevalo et al. \(2011\)](#). *g*: [Merrild et al. \(2008\)](#) and [Wang et al. \(2012\)](#). *h*: European Commission (2018). *i*: [Zeng et al. \(2018\)](#). *j*: [Broadbent \(2016\)](#). *k*: [Antonopoulos et al. \(2021\)](#). *l*: [Albrecht et al. \(2022\)](#). *m*: [ARECO \(2023\)](#). *n*: [CCB & FEFCO \(2015\)](#). *o*: [Laurijssen et al. \(2010\)](#) and [Piekarski et al. \(2012\)](#). *p*: [Stubdrup et al. \(2016\)](#). *q*: [Johnson et al. \(2008\)](#). *r*: [Marczak \(2022\)](#) and [Shan et al. \(2023\)](#).**Fig. 4.** Circularity scores for single-use crates (cardboard and wooden) and RPC (Reusable Plastic Crate). Material Circular Indicator (MCI), Material Reutilisation Score (MRS), and Circular Index (CI).

In contrast to single use crates, the RPC obtained the highest circularity for MCI (99.5 %) and CI (36.7 %). The higher value of MCI was due to its reusability, whilst for CI, the better performance was due to the higher quality of the plastic after its upcycling ($\beta = 0.98$). Despite this high value, the recycling process causes a loss of mechanical properties, for example, recycled PET (the most studied plastic in recycling systems) is weaker and cannot be used alone, so additional virgin material is needed in order to achieve the same performance (Mohan Raj, 2019). Similarly, Lonca et al. (2018) show that increasing the recycling content could lead to a decrease in material quality, reducing the material lifetime and therefore its circularity score.

For the optimised RPC scenario, the circularity of fully recyclable RPC (R:100 %) is shown in Fig. 4. Full circularity was almost achieved for the MCI (99.6 %) and CI (97.6 %) scores. In comparison with the conventional scenario, the MCI score increase was negligible (0.1 %). On the other hand, the circularity scores increased more for CI and MRS due to the full recyclability of the RPC, reaching 60.9 % and 41.6 %, respectively. All circularity scores of the RPC (R:100 %) were higher than those of the conventional scenario. Therefore, the RPC should be fully recycled, closing the loop in order to increase its circularity (Colijn et al., 2022).

3.4. Comparing circularity scores with the environmental and economic profiles

The environmental and economic profiles of single-use crates (cardboard and wooden) and RPC were normalised (up to 100 %) to compare those results with the circularity scores (Fig. 5). The figure shows a range from 0 % to 100 %, for economic and environmental performance, 100 % represents the highest environmental impact (worse environmental option), while 0 % represents the lowest (Fig. 5a). For circularity scores, 100 % represents the highest circularity (best option), while 0 % indicates the lowest (Fig. 5b). LCA was established as the reference method for sustainability assessment, its plot shows that the MRS scores were not in line with the environmental and/or economic performance of the packaging products. On the other hand, the material circularity profiles obtained by the MCI and CI indicators were in line with that produced by the LCA, since those circularity scores drawn the same conclusions as the LCA, the preferred crate was RPC followed by, cardboard and wooden crates.

The difference between C-indicators and environmental LCA lies in

the system boundaries of each approach. While MCI and MRS only consider material circularity, the scope of environmental LCA goes further and also includes the background processes and their related environmental impact (Lonca et al., 2018). Among the material-based C-indicators, MCI was the most complete, since it included: recycled content, biological feedstock, incineration with energy recovery, and material utility. By including such parameters, the material circularity improves, and the results could be more aligned with LCA.

Brändström & Saidani (2022) pointed out that MCI could provide a valid comparison with LCA with a higher consistency, as this C-indicator includes parameters such as lifetime, sharing efficiency, and recycling rate, parameters related to climate change and resource use impact categories. Nevertheless, Corona et al. (2019) concluded that the MCI indicator based on material recirculation (circulation of a material in a closed loop) is not suitable for measuring overall savings in the use of resources, since it may hide a burden shift towards higher energy consumption or pollutant emissions. This was observed in our results, which showed higher environmental performance for freshwater eutrophication and minerals and metals resource use of the fully recyclable RPC (R:100 %) compared to the conventional RPC scenario.

The MRS values for the wooden crate (with the highest environmental impact) and the RPC (with the lowest) were similar, 21.8 % and 25.1 %, respectively. These products share a common trait of low material recovery rates. On the other hand, this study is also in line with Niero & Kalbar (2019) who calculated the LCA, MRS, and MCI C-indicators for beer packaging options. When they linked the environmental and circularity indicators, the results differed: that work showed that the two C-indicators gave opposing results, as occurred in the present study for the base case scenario, where RPC was the most circular for MCI and for MRS it was the cardboard crate.

Finally, as mentioned above, the ranking resulting from the CI scores was also in line with the one deduced from the environmental and economic profiles. This was caused by the inclusion of material quality in that circularity score, which was higher for the RPC scenarios. However, for the fully recyclable RPC (R:100 %) the circularity increased very markedly, despite having the same carbon footprint as the conventional scenario and a slightly higher impact on freshwater eutrophication, according to LCA. This fact relied on the nature of the analysed system; in RPC the LCA emissions were completely governed by transport, while for CI only the reuse of PP material affected the value. This discrepancy between LCA and C-indicator has already been

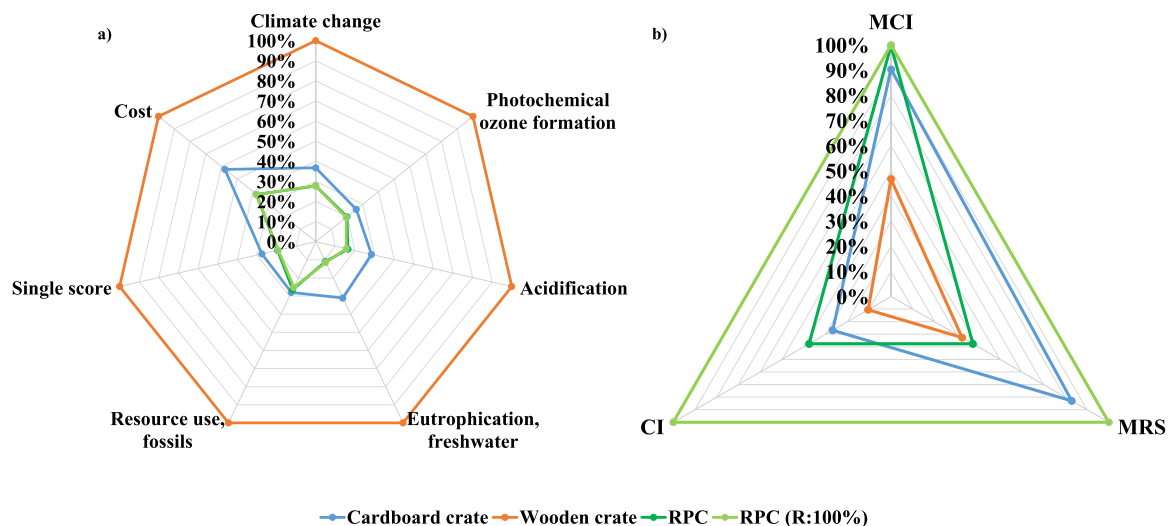


Fig. 5. Economic and environmental performance (a) and circularity scores (b) of packaging options normalised to 100 %. For economic and environmental performance, 100 % represents the highest impact (worse environmental option), while 0 % represents the lowest. Conversely, for circularity scores, 100 % represents the highest circularity (best option), while 0 % indicates the lowest. Material Circularity Indicator (MCI), Material Reutilization Score (MRS), and Circular Index (CI). RPC: Reusable Plastic Crate.

reported by other authors. For example, [Lonca et al. \(2018\)](#) calculated the MCI and LCA of the tyre industry considering recycling, retreading, and regrooving scenarios. They showed that increasing the recycling content of tyres up to 4 % improved the MCI, but also increased the impact on human health and resources impact categories, due to the raw material production. [Vadoudi et al. \(2022\)](#) evaluated the environmental impact and MCI of three-layer plastic film, considering up to three recycling loops. Increasing the recycled content improved the MCI score, whilst the impacts on freshwater eutrophication and land use increased with each recycling cycle, as the recycling process had a higher impact than the linear model in those impact categories.

4. Conclusions

Given the proliferation of C-indicators and the lack of consensus on circularity measurement, the present study helps to identify the most sustainable packaging option for fresh fruits and vegetables, and contributes to the discussion regarding the relationship between circularity scores and overall sustainability performance. To this end, three different circularity metrics (MCI, MRS, and CI) were evaluated for the case study of single-use crates and RPC. The environmental and economic profiles of these packaging options were also compared with these circularity scores. It must be noted that the environmental profile was calculated at European context, using the distribution distances from fruit and vegetable processing to retailer from weighted of to six food producer countries (France, Italy, Greece, Netherlands, Poland and Spain). However, the economic assessment was focused on Spain, which had the highest share among these countries, because it has the highest fruit and vegetable production and the highest export share.

RPC have the lowest environmental impact and the lowest costs. When the environmental and economic profiles were compared with the circularity scores, different conclusions were drawn depending on the C-indicator. Considering the base case (European packaging waste management ratios), the most circular packaging was the RPC for MCI and CI. On the other hand, the cardboard crate achieved the highest score for MRS because this latter indicator puts more weight on recycled content and its recyclability, but without considering material utility or material quality.

The discrepancy between LCA and circularity scores can be attributed to the limited scope of circularity metrics. Both MCI and MRS focus primarily on material recirculation, while the CI score also encompasses material quality in its assessment. Our findings highlight that inclusion of material utility in MCI and the quality of PP in CI has a substantial impact on the circularity scores of RPC, resulting in circularity scores that align closely with those results obtained from LCA.

When the recycling rate of the RPC was increased up to 100 %, the circularity scores of the packaging system increased greatly, while its environmental performance (as determined by LCA) remained essentially unchanged. This is due to the fact that most of the impacts generated by the reused RPC were dominated by the transport stage, with only minimal contributions from the raw materials, fabrication, and end-of-life.

In conclusion, the food industry would benefit from the use of reusable crates for transporting bulk fruit and vegetables, as this option was the most economically and environmentally sustainable. The circularity scores of MCI and CI confirm the findings of LCA, which demonstrate that RPC is the most circular packaging. Moreover, from the alignment between circularity scores and LCA, it can be concluded that C-indicators should not be addressed in isolation from LCA, as it is possible that significant insights might be lost. In this sense, although the environmental impact improved due to full recycling, emissions did not decrease significantly, because the main contributor was the transport stage, which was not covered by the circularity indicators. Therefore, further exploration is needed to mitigate the environmental impact in the transport stage.

CRediT authorship contribution statement

Encarna Aguayo: Writing – review & editing, Validation, Supervision, Project administration, Investigation, Funding acquisition, Data curation, Conceptualization. **Blanca Corona:** Writing – review & editing, Validation, Supervision, Methodology, Investigation. **Guillermo San Miguel:** Writing – review & editing, Validation, Supervision, Methodology, Investigation. **Laura Rasines:** Writing – original draft, Software, Methodology, Investigation, Formal analysis, Data curation.

Declaration of Competing Interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests. As mentioned in the paper: This research was supported by the Autonomous Community of the Region of Murcia through the Fundación Séneca - Agencia de Ciencia y Tecnología de la Región de Murcia (Seneca Foundation - Agency for Science and Technology in the Region of Murcia) and European programme NextGenerationEU. Thanks also go to the Spanish Ministry of Science and Innovation - National Research Agency (MCIN/AEI/10.13039/501100011033) and by “ERDF A way of making Europe”, of the “European Union” [RTI2018-099139-B-C21]. Laura Rasines acknowledges financial support by MCIN/AEI and by “ESF Investing in your future” grant [PRE 2019-090573]. If there are other authors, they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at [doi:10.1016/j.fpsl.2024.101391](https://doi.org/10.1016/j.fpsl.2024.101391).

Data Availability

Data will be made available on request.

References

- Accorsi, R., Cascini, A., Cholette, S., Manzini, R., & Mora, C. (2014). Economic and environmental assessment of reusable plastic containers: A food catering supply chain case study. *International Journal of Production Economics*, 152, 88–101. <https://doi.org/10.1016/j.ijpe.2013.12.014>
- Akerman, E., 2016. Development of Circular Economy Core Indicators for Natural Resources. Royal Institute of Technology.
- Albrecht, S., Bertling, J., Fischer, M., Kabasci, S., Prescher, T., Schulte, A., 2022. Reusable plastic crates vs. single-use cardboard boxes [WWW Document]. <https://doi.org/10.24406/publica-456>.
- Albrecht, S., Brandstetter, P., Beck, T., Fullana-I-Palmer, P., Grönman, K., Baitz, M., Deimling, S., Sandilands, J., & Fischer, M. (2013). An extended life cycle analysis of packaging systems for fruit and vegetable transport in Europe. *The International Journal of Life Cycle Assessment*, 18, 1549–1567. <https://doi.org/10.1007/s11367-013-0590-4>
- Antonopoulos, I., Faraca, G., & Tonini, D. (2021). Recycling of post-consumer plastic packaging waste in EU: Process efficiencies, material flows, and barriers. *Waste Manag.* 126, 694–705. <https://doi.org/10.1016/j.wasman.2021.04.002>

- ARECO, 2023. Sostenibilidad [WWW Document]. URL (<https://areco.org.es/sostenibilidad/>) (accessed 7.5.23).
- Arevalo, C. B. M., Bhatti, J. S., Chang, S. X., & Sidders, D. (2011). Land use change effects on ecosystem carbon balance: From agricultural to hybrid poplar plantation. *Agriculture, Ecosystems & Environment*, 141, 342–349. <https://doi.org/10.1016/j.agee.2011.03.013>
- Brändström, J., & Saidani, M. (2022). Comparison between circularity metrics and LCA: A case study on circular economy strategies. *Journal of Cleaner Production*, 371. <https://doi.org/10.1016/j.jclepro.2022.133537>
- Broadbent, C. (2016). Steel's recyclability: demonstrating the benefits of recycling steel to achieve a circular economy. *The International Journal of Life Cycle Assessment*, 21, 1658–1665. <https://doi.org/10.1007/s11367-016-1081-1>
- C2C, 2016. Cradle to Cradle Certified. Product Standard Version 3.1 [WWW Document]. URL (<http://s3.amazonaws.com/c2c-website/resources/certification/standard/C2CCertifiedProductStandardV3Nov42013.pdf>) (accessed 5.20.23).
- CCB and FEFCO, 2015. European Database for Corrugated Board Life Cycle Studies 1–36.
- CCB, FEFCO, 2018. European Database for Corrugated Board Life Cycle Studies [WWW Document]. URL (<https://www.fefco.org/2018-european-database-corrugated-board-life-cycle-studies-revised/>) (accessed 3.18.23).
- Colijn, L., Fraiture, F., Gommeh, E., Schroën, K., & Metze, T. (2022). Science and media framing of the future of plastics in relation to transitioning to a circular economy. *Journal of Cleaner Production*, 370. <https://doi.org/10.1016/j.jclepro.2022.133472>
- Corona, B., Shen, L., Reike, D., Rosales Carreón, J., & Worrell, E. (2019). Towards sustainable development through the circular economy—A review and critical assessment on current circularity metrics. *Resources, Conservation and Recycling*, 151, Article 104498. <https://doi.org/10.1016/j.resconrec.2019.104498>
- Cullen, J. M. (2017). Circular economy: Theoretical benchmark or perpetual motion machine? *Journal of Industrial Ecology*. <https://doi.org/10.1111/jiec.12599>
- de Oliveira, C. T., & Oliveira, G. G. A. (2023). What Circular economy indicators really measure? An overview of circular economy principles and sustainable development goals. *Resources, Conservation and Recycling*, 190, Article 106850. <https://doi.org/10.1016/j.resconrec.2022.106850>
- Delgado-Aguilar, M., Tarrés, Q., Pélach, M.À., Mutjé, P., & Fullana-I-Palmer, P. (2015). Are cellulose nanofibers a solution for a more circular economy of paper products? *Environmental Science & Technology*, 49, 12206–12213. <https://doi.org/10.1021/acs.est.5b02676>
- EASAC, 2016. Indicators for a circular economy - European Academies Science Advisory Council. German National Academy of Sciences Leopoldina, Germany.
- ECOBILAN, 2000. Analyse du cycle de vie des caisses en bois, carton ondulé et plastique pour pommes 83.
- Ecoembes, 2023. Tarifas de Punto Verde 2023 [WWW Document]. URL (<https://www.fedeto.es/wp-content/uploads/2022/07/fedeto-comunicado-tarifas-punto-verde-2023.pdf>) (accessed 6.13.23).
- Ecoinvent, 2023. The ecoinvent Database 3.9. Zurich [WWW Document]. URL (<https://www.ecoinvent.org/>).
- Ellen MacArthur Foundation, Granta Design, 2019. Circular indicators. An approach to measuring circularity [WWW Document]. URL (<http://www.ellenmacarthurfoundation.org/circularity-indicators/>). (accessed 3.10.23).
- Equanimator Ltd, 2023. Debunking Efficient Recovery/Debunking efficiency recovery. The performance of EU incineration facilities [WWW Document]. URL (<https://zerowasteurope.eu/wp-content/uploads/2023/01/Exec-Summary-Debunking-Efficient-Recovery-EN.docx-1.pdf>) (accessed 7.3.23).
- European Commission, 2022. Commission staff working document. Impact assessment report. Proposal for a regulation of the European Parliament and the Council on packaging and packaging waste, amending Regulation (EU) 2019/1020, and repealing Directive 94/62/EC [WWW Document]. URL (https://eur-lex.europa.eu/resource.html?uri=cellar:0567fd10-7165-11ed-9887-01aa75ed701a1.0001.02/DOC_1&format=PDF) (accessed 9.12.23).
- European Commission, 2021a. Commission Recommendation on the use of the Environmental Footprint methods [WWW Document]. C(2021) 9332 Final. URL (https://environment.ec.europa.eu/publications/recommendation-use-environmental-footprint-methods_en) (accessed 10.25.23).
- European Commission, 2021b. Annexes 1 to 2. Product environmental footprint method [WWW Document]. URL (https://environment.ec.europa.eu/publications/recommendation-use-environmental-footprint-methods_en) (accessed 11.4.23).
- European Commission, 2018a. ELCD [WWW Document]. URL (<https://eplca.jrc.ec.europa.eu/ELCD3/>) (accessed 10.24.22).
- European Commission, 2018b. European Wood Waste Statistics Report for Recipient and Model Regions 1–48.
- European Union, 2020. Circular Economy Action Plan [WWW Document]. URL (https://ec.europa.eu/environment/pdf/circular-economy/new_circular_economy_action_plan.pdf) (accessed 8.5.23).
- Eurostat, 2020a. Statistics Explained. Packaging waste statistics [WWW Document]. URL (https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Packaging_waste_statistics#Waste_generation_by_packaging_material) (accessed 5.25.23).
- Eurostat, 2020b. Statistics Explained. Packaging waste statistics [WWW Document]. URL (https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Packaging_waste_statistics#Waste_generation_by_packaging_material).
- Faraca, G., Boldrin, A., & Astrup, T. (2019). Resource quality of wood waste: The importance of physical and chemical impurities in wood waste for recycling. *Waste Manag*, 87, 135–147. <https://doi.org/10.1016/j.wasman.2019.02.005>
- Fazio, S., Biganzoli, F., De Laurentis, V., Zampori, L., Sala, S., Diaconu, E., 2018. Supporting information to the characterisation factors of recommended EF Life Cycle Impact Assessment methods, version 2, from ILCD to EF 3.0, EUR 29600 EN, European Commission, Ispra, 2018, ISBN 978-92-79-98584-3, doi:10.2760/002447, PUBSY No. JRC114822., New Models and Differences with ILCD, EUR. <https://doi.org/10.2760/002447>.
- FEFCO, 2022. COMPARATIVE LIFE CYCLE ASSESSMENT (LCA) PACKAGING SOLUTIONS FOR THE FOOD SEGMENT [WWW Document]. URL (https://www.fefco.org/sites/default/files/2022/FEFCO_Comparative_LCA_study.pdf) (accessed 6.8.23).
- Francou, C., Linères, M., Derenne, S., Villio-Poitrenaud, M. Le, & Houot, S. (2008). Influence of green waste, biowaste and paper-cardboard initial ratios on organic matter transformations during composting. *Biorescience Technology*, 99, 8926–8934. <https://doi.org/10.1016/j.biortech.2008.04.071>
- Garza-Reyes, J. A., Salomé Valls, A., Peter Nadeem, S., Anosike, A., & Kumar, V. (2019). A circularity measurement toolkit for manufacturing SMEs. *International Journal of Production Research*, 57, 7319–7343. <https://doi.org/10.1080/00207543.2018.1559961>
- INE, 2020. Estadística sobre el suministro y saneamiento del agua [WWW Document]. URL (<https://www.ine.es/prensa/essa2020.pdf>) (accessed 6.13.23).
- Ioelovich, M. Y. (2017). Comparison of methods for the calculation of heat of combustion of bi-polymers. *Khimiya Rastit Syr'ya*, 1, 49–56. <https://doi.org/10.14258/jcpm.2017021625>
- ISO, 2006a. ISO 14040:2006. Environmental management. Life cycle assessment. Principles and framework.
- ISO, 2006b. ISO 14044: Environmental Management e Life Cycle Assessment e Requirements and Guidelines. Bruxelles.
- Johnson, J., Reck, B. K., Wang, T., & Graedel, T. E. (2008). The energy benefit of stainless steel recycling. *Energy Policy*, 36, 181–192. <https://doi.org/10.1016/j.enpol.2007.08.028>
- Kakwani, N. S., & Kalbar, P. P. (2022). Measuring urban water circularity: Development and implementation of a Water Circularity Indicator. *Sustainable Production and Consumption*, 31, 723–735. <https://doi.org/10.1016/j.spc.2022.03.029>
- Kirchherr, J., Reike, D., & Hekkert, M. (2017). Conceptualizing the circular economy: An analysis of 114 definitions. *Resources, Conservation and Recycling*, 127, 221–232. <https://doi.org/10.1016/j.resconrec.2017.09.005>
- Krieg, H., Gehring, F., Fischer, M., Albrecht, S., 2018. Carbon Footprint of Packaging Systems for Fruit and Vegetable Transports in Europe [WWW Document]. URL (https://www.stiftung-mehrweg.de/fileadmin/user_upload/SIM_CF_Final_report_for_publication.pdf) (accessed 7.14.23).
- Laurijssen, J., Marsidi, M., Westenbroek, A., Worrell, E., & Faaij, A. (2010). Paper and biomass for energy? The impact of paper recycling on energy and CO2 emissions. *Resources, Conservation and Recycling*, 54, 1208–1218. <https://doi.org/10.1016/j.resconrec.2010.03.016>
- Linder, M., Sarasini, S., & van Loon, P. (2017). A Metric for Quantifying Product-Level Circularity. *Journal of Industrial Ecology*, 21, 545–558. <https://doi.org/10.1111/jiec.12552>
- Liu, W., Zhu, Z., & Ye, S. (2023). A framework towards design for circular packaging (DFCP): Design knowledge, challenges and opportunities. *Circular Economy and Sustainability*. <https://doi.org/10.1007/s43615-023-00264-3>
- Lonca, G., Muggéo, R., Imbeault-Tétrault, H., Bernard, S., & Marni, M. (2018). Does material circularity rhyme with environmental efficiency? Case studies on used tires. *Journal of Cleaner Production*, 183, 424–435. <https://doi.org/10.1016/j.jclepro.2018.02.108>
- López-Gálvez, F., Rasines, L., Perla, A. G., Artés-Hernández, F., & Aguayo, E. (2021). Reusable plastic crates (RPCs) for fresh produce (Case Study on Cauliflowers): sustainable packaging but potential salmonella survival and risk of cross-contamination. *Foods*, 10, 1254. <https://doi.org/10.3390/foods10061254>
- MAPA, 2022. Frutas y hortalizas [WWW Document]. URL (<https://www.mapa.gob.es/es/agricultura/temas/producciones-agricolas/frutas-y-hortalizas/informacion-general.aspx>) (accessed 10.6.24).
- Marczak, H. (2022). Energy inputs on the production of plastic products. *Journal of Ecological Engineering*, 23, 146–156. <https://doi.org/10.12911/22998993/151815>
- Merrill, H., Damgaard, A., & Christensen, T. H. (2008). Life cycle assessment of waste paper management: The importance of technology data and system boundaries in assessing recycling and incineration. *Resources, Conservation and Recycling*, 52, 1391–1398. <https://doi.org/10.1016/j.resconrec.2008.08.004>
- MITECO, 2021. Estadística forestal [WWW Document]. URL (https://www.miteco.gob.es/es/biodiversidad/estadisticas/forestal/anuarios_todos.aspx) (accessed 7.24.23).
- MITECO, 2020. Memoria anual de generación y gestión de residuos. Residuos de competencia municipal 2020, Instalaciones de tratamiento: entradas y salidas. [WWW Document]. URL (<https://www.miteco.gob.es/es/calidad-y-evaluacion-ambiental/publicaciones/Memoria-anual-generacion-gestion-residuos.aspx>) (accessed 2.10.21).
- MITMA, 2023. Observatorio de costes del transporte de mercancías por carretera [WWW Document]. URL (<https://www.mitma.gob.es/recursos/mfom/comodin/recursos/observatoriocostesenero2023v1.pdf>) (accessed 6.15.23).
- Mohan Raj, J. (2019). Picnic benches and the circular economy. *Reinforced Plastics*, 63, 213–215. <https://doi.org/10.1016/j.jrepl.2019.05.001>
- Niero, M., & Kalbar, P. P. (2019). Coupling material circularity indicators and life cycle based indicators: A proposal to advance the assessment of circular economy strategies at the product level. *Resources, Conservation and Recycling*, 140, 305–312. <https://doi.org/10.1016/j.resconrec.2018.10.002>
- OCU, 2023. Evolución del precio de la electricidad [WWW Document]. URL (<https://www.ocu.org/vivienda-y-energia/gas-luz/informe/precio-luz>) (accessed 6.28.23).
- Piekarski, C. M., Francisco, A. C., De, & Mendes, L. (2012). Energy consumption in MDF production: Overview of use renewable and non-fossil energy sources in a Brazilian mill. *International Conference on Industrial Engineering and Operations Management*, 1–8.
- PRé Consultant, 2023. SimaPro v 9.5.

- Ross Lindgreen, E., Mondello, G., Salomone, R., Lanuzza, F., & Saija, G. (2021). Exploring the effectiveness of grey literature indicators and life cycle assessment in assessing circular economy at the micro level: a comparative analysis. *The International Journal of Life Cycle Assessment*, 2171–2191. <https://doi.org/10.1007/s11367-021-01972-4>
- Rossi, J., & Bianchini, A. (2022). Plastic waste free” – A new circular model for the management of plastic packaging in food value chain. *Transportation Research Procedia*, 67, 153–162. <https://doi.org/10.1016/j.trpro.2022.12.046>
- Saidani, M., Yannou, B., Leroy, Y., Cluzel, F., & Kendall, A. (2019). A taxonomy of circular economy indicators. *Journal of Cleaner Production*, 207, 542–559. <https://doi.org/10.1016/j.jclepro.2018.10.014>
- Shan, C., Pandyaswargo, A. H., & Onoda, H. (2023). Environmental impact of plastic recycling in terms of energy consumption: A comparison of japan's mechanical and chemical recycling technologies. *Energies*, 16. <https://doi.org/10.3390/en16052199>
- Springle, N., Li, B., Soma, T., & Shulman, T. (2022). The complex role of single-use compostable bioplastic food packaging and foodservice ware in a circular economy: Findings from a social innovation lab. *Sustainable Production and Consumption*, 33, 664–673. <https://doi.org/10.1016/j.spc.2022.08.006>
- Stubdrup, K.R., Karlis, P., Roudier, S., 2016. Production of Wood – based Panels Best Available Techniques (BAT) Reference Document for the Production of Wood-based Panels. <https://doi.org/10.2791/21807>.
- Tua, C., Biganzoli, L., Grosso, M., & Rigamonti, L. (2019). Life cycle assessment of reusable plastic crates (RPCs). *Resources*, 8. <https://doi.org/10.3390/resources8020110>
- Vadoudi, K., Deckers, P., Demuytere, C., Askanian, H., & Verney, V. (2022). Comparing a material circularity indicator to life cycle assessment: The case of a three-layer plastic packaging. *Sustainable Production and Consumption*, 33, 820–830. <https://doi.org/10.1016/j.spc.2022.08.004>
- van Loon, M. P., Vonk, W. J., Hijbeek, R., van Ittersum, M. K., & ten Berge, H. F. M. (2023). Circularity indicators and their relation with nutrient use efficiency in agriculture and food systems. *Agricultural Systems*, 207, Article 103610. <https://doi.org/10.1016/j.agsy.2023.103610>
- Wang, L., Templer, R., & Murphy, R. J. (2012). A Life Cycle Assessment (LCA) comparison of three management options for waste papers: Bioethanol production, recycling and incineration with energy recovery. *Bioresource Technology*, 120, 89–98. <https://doi.org/10.1016/j.biortech.2012.05.130>
- World Economic Forum, 2016. The New Plastic Economy: Rethinking the Future of Plastics [WWW Document]. URL www.greenpeace.org/static/planet4-philippines-stateless-release/2019/05/b1e5a437-b1e5a437-wef_the_new_plastics_economy.pdf (accessed 7.18.23).
- Zampori, L., & Pant, R. (2019). Suggestions for updating the Product Environmental Footprint (PEF) method. *JRC Publications Repository*, 248. <https://doi.org/10.2760/424613>
- Zeng, Q., Lu, Q., Zhou, Y., Chen, N., Rao, J., & Fan, M. (2018). Circular development of recycled natural fibers from medium density fiberboard wastes. *Journal of Cleaner Production*, 202, 456–464. <https://doi.org/10.1016/j.jclepro.2018.08.166>