Implementation of resource supply risk characterisation factors in the life cycle assessment of food products: Application to contrasting bread supply chains

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Abstract

Purpose

In addition to generate environmental impacts, food systems are vulnerable to shortages of the resources on which they rely (e.g. critical minerals, water, land). To eco-design these systems, their environmental footprint and vulnerability to resources must be assessed simultaneously. Resource supply risk methods have been applied to Life Cycle Assessments (LCA) of high-tech products to provide information on resource accessibility. The aim of this paper is to discuss the applicability of these methods to the LCA of food products.

Methods

Supply risk characterisation factors (CF) are derived from the two acknowledged resource criticality methods of i) the Joint Research Centre and ii) the Yale University. These methods characterise mineral, land and water resources, which are essential for agricultural products. CFs are matched with Life Cycle Inventory (LCI) data, and a comparative LCA on both environmental impacts and resource supply risk is performed on three contrasting bread supply chains. These supply chains differ by agricultural practices, flour milling and bread baking processes, as well as transport distances between intermediaries.

Results and discussion

The results of the case study show that trade-offs can occur between environmental impact and resource supply risk. Indeed, the supply chain with the greatest environmental impacts is also the one with the lowest mineral and water supply risk potential. Analysis of the results indicates that fertilisers contribute the most to the mineral supply risk of agricultural products, that land supply risk potential is due to agricultural production and forestry for energy and packaging, and that the water supply risk potential is mainly related to electricity production for baking. The comparison between the Yale and JRC methods highlights differences in modelling choices, mainly due to their different coverage and scope.

Conclusion

This case study highlights the value of considering resource supply risks as a complement to conventional LCA of food products, as it makes it possible to identify potential trade-offs between environmental impacts and vulnerability to resource supply shortage. The development of additional case studies, using other supply risk methods and including processed resources such as agricultural products would enable further research on food systems criticality.

Keywords

resource criticality, environmental performance, trade-offs, impact assessment, vulnerability, agri-food systems

1) Introduction

The increasing world population and rising standards of living are leading to a growing demand for food worldwide. Projections estimate that, compared with 2010, food demand will have increased by between 35% and 56% (in kcal consumed) by 2050. (van Dijk et al. 2021). These socio-economic trends place increasing pressure on natural resources on which food systems depend to feed humanity (Westhoek et al. 2016), such as minerals, energy, land or water (Chowdhury et al. 2017; Ringler, Bhaduri, and Lawford 2013).

Certain mineral resources, such as phosphate, which forms the basis of mineral fertilisation for agriculture, are set to become increasingly scarce (Cordell and Neset 2014). Estimates of the lifespan of phosphate rock reserves range from 60 to 400 years (Cordell and White 2011). In the same time, agriculture accounts for 38% of available land use (FAOSTAT 2020) and more than 70% of water withdrawals (FAO 2020). Land and water resources are subject to competition and tension (Nonhebel 2005; Petersen-Perlman et al. 2017). Consequently, the dependence of food systems on these key resources make them vulnerable to resource supply disruptions.

Food systems also generate multiple environmental impacts. They are responsible for 26% of global greenhouse gas emissions (Poore and Nemecek 2018), 64 % of intentional nitrogen fixation and 80 % of phosphorus leakage flows (Springmann et al. 2018; Steffen et al. 2015) and they are the primary cause for loss in biodiversity (Benton et al. 2021).

In response to these challenges, emerging strategies aim to design food supply chains that generate less environmental impact and are less dependent on resources (Petit-boix et al. 2022). These include the adoption of new farming practices (e.g. organic farming, no-tillage,…) (Gomiero, Pimentel, and Paoletti 2011), relocation of food industries (Watts, Lbery, and Maye 2005), and the development of short food supply chains (Renting, Marsden, and Banks 2003). In order to assess the environmental sustainability of these new strategies, quantitative tools are required to identify potential trade-offs between the environmental impacts and the resource vulnerability of food supply chains.

Life cycle assessment (LCA) is a reference method for quantifying the environmental impacts of products or services, and has been successfully applied to agri-food supply chains (Notarnicola et al., 2017; Roy et al., 2009). However, to date, Life Cycle Impact Assessment (LCIA) methods lack CFs assessing the accessibility, dependence or vulnerability of resources (Van Der Werf et al., 2014).

LCA describes the impacts of human activities on three Areas of Protection (AoP), i.e. Ecosystem Quality, Human Health and Resources. The most widely used models for the resources AoP quantify the impact of resource consumption on the availability of resources for future generations (Dewulf et al. 2015; Huijbregts et al. 2016). However, other perspectives are now being proposed that assess new resource issues (e.g. resource functionality or resource access), such as resource criticality (Dewulf et al. 2015; Sonderegger et al. 2017, 2020).

Criticality methods evaluate the economic and technical dependency on a certain materials, as well as the probability of supply disruptions, for a defined stakeholder group within a certain time frame (Schrijvers et al. 2020). Criticality assessment methods first propose to quantify a supply risk index, which represents the possibility of supply disruption for a given material, according to its geological, technological, economic, social or geopolitical availability. The methods also entail a second dimension related to the impact of supply restriction, by analysing the substitution and importance in use of the studied material (importance can be economic, strategic, etc.) (Schrijvers et al. 2020). Criticality was initially intended for non-energetic mineral resources to help secure the supply of key resources for specific industries (Graedel et al. 2015), for instance low carbon energy technologies that depend on rare-earth elements.

Implementing the concept of resource criticality in LCA enables the potential risk of resource supply disruption to be assessed (Sonnemann et al. 2015), and therefore take into account the short-term accessibility of resources. This approach is different from the evaluation of resources currently carried out in LCA, i.e. the evaluation of the dissipation or depletion of resources for future generations (Sonnemann et al. 2015). It provides LCA with a criticality midpoint indicator, which quantifies a different category of impact than traditional resource midpoint indicators (Sonderegger et al. 2020). The criticality assessment of a product or service within the LCA framework thus complements conventional environmental impact assessments (André and Ljunggren 2021; Berger et al. 2020; Cimprich et al. 2019; Mancini et al. 2015; Sonderegger et al. 2020). From an operational point of view, criticality is implemented in LCA by deriving characterisation factors from resource supply risk indexes (Bach et al. 2016; Santillán-Saldivar et al. 2022).

Proposals to implement the concept of criticality within LCA have so far focused on high-tech products such as batteries (Pelzeter et al. 2022; Terlouw et al. 2019), medical equipment (Cimprich, Young, et al. 2017) or electronic devices (Mancini, Benini, and Sala 2018). Yet, to date, there are no criticality assessments for food products. In addition, LCA criticality studies have focused on the criticality of mineral (i.e. aluminium, cobalt, etc.) and fossil (i.e. natural gas, coal etc.) resources only. However, food products depend on other abiotic resources, such as water and land, whose criticality has also been characterised (Deteix et al. 2023; Sonderegger, Pfister, and Hellweg 2015) but not tested in case studies. Furthermore, of the studies that incorporate criticality into LCA, only the studies of Cimprich et al., (2017a), Henßler et al., (2016) and Penaherrera et al. (2022)

present both environmental impacts and criticality results. These works highlight the complementarity of the two approaches, and the possible trade-offs between environmental impacts and supply risks.

The aim of this work is to apply criticality methods to the food sector, by integrating supply risk metrics into the LCA of food products.

To this end, two criticality methods are used to derive supply risk CFs. These CFs are then applied to an illustrative LCA case study, the LCA of three contrasting bread supply chains. These supply chains were specifically modelled for this work to be used as a proof of concept, and are a combination of existing processes related to bread supply chains found in the Agribalyse database (Koch and Salou 2020a), and the work of Kulak et al. (2015), who performed the LCA of different food networks for bread.

Section 2 first describes the choice for the two criticality methods among existing methods, and presents the methodology for deriving CFs from the supply risk indexes. These indexes were provided by methods applied to a wide range of resources and countries. Secondly, it describes the relationship between CFs and Life Cycle Inventory (LCI) flows. Thirdly, LCA is carried out on three contrasting bread supply chains according to the four standardised LCA phases (ISO 2006b, 2006a). Section 3 presents the results from this LCA study including the indicator of supply risks for three main categories of resources (i.e. mineral, water and land).The fourth and final section assesses the applicability and current limitations in using supply risk methods to complement food product LCA.

2) Materials and Methods

2.1 Choice of the criticality methods

Although many criticality methods exist, there is no prevailing scientific consensus concerning the method to be used (Schrijvers et al. 2020). For the characterisation of Supply Risk (SR) in LCA, Sonderegger et al., (2020) and Berger et al., (2020) recommended the GeoPolRisk (Cimprich, Young, et al. 2017; Gemechu et al. 2015; Santillán-Saldivar et al. 2022) and ESSENZ (Bach et al. 2016; Pelzeter et al. 2022) methods. In addition to GeoPolRisk, Hackenhaar et al., (2022) also found the European Union (EU) Joint Research Centre (JRC) method (Blengini et al. 2017) to be compatible with LCA, owing to its scientific robustness (data quality, uncertainty, peer-reviewing), transparency (traceability of modelling and documentation, reproducibility), applicability (technical feasibility, data availability,…) and high level of acceptance (by policy-makers, industry, academia).

To date, the GeoPolRisk method has been applied to 8 LCA case studies, the ESSENZ method to 6 and the JRC method to 5. All these case studies involved high-tech products such as battery (Pelzeter et al. 2022; Santillán-Saldivar et al. 2021; Terlouw et al. 2019) vehicle (Berger et al. 2020; Cimprich et al. 2019; Gemechu et al. 2015; Henßler et al. 2016; Lütkehaus et al. 2022) or electronic devices (Koch et al., 2019; Mancini et al., 2018; Yavor et al., 2021) except for the study of Bach et al., (2018) that was based on shelves. The 3 papers that compared criticality methods in LCA case studies highlighted the variability in results due to the methodology of supply risk modelling (Berger et al. 2020; Cimprich et al. 2019; Terlouw et al. 2019). The studies of Henßler et al., (2016), Mancini et al., (2018) and Cimprich, Karim et al., (2017) also highlighted the complementarity between criticality assessment and LCA. In particular, Henßler et al., (2016) highlighted the occurrence of potential trade-offs between environmental performance and resource criticality, as well as the importance of the life cycle perspective for assessing product criticality. However, Cimprich et al., (2017) and Helbig et al., (2016) also pointed out the challenges in matching Supply Risk CF with LCI environmental flows due to the differences in goal and scope of criticality assessments and LCA.

Other criticality methods have also been implemented in LCA case studies (Hackenhaar et al. 2022; Schrijvers et al. 2020), in particular the Yale University method (Graedel et al. 2012), which was used by Terlouw et al. (2019) on the LCA of two batteries.

In the four criticality methods mentioned above, SR indexes are quantified on the basis of a set of parameters, which can vary between methods. Some parameters are common across all four methods (i.e., concentration of production, or political stability of supplier countries), but each method includes its own distinct parameters (e.g. trade restrictions for the JRC method, depletion time for Yale, price elasticity for GeoPolRisk, demand growth for ESSENZ). ESSENZ is the only method not to provide a single aggregate SR index, but a set of parameters influencing the SR of resources. Finally, GeoPolRisk is spatialised at country level, the JRC at European level, and Yale and ESSENZ are not spatialised.

In addition to providing supply risk indexes for mineral resources, the Yale methodological framework has been adapted to take the criticality of water resources into account (Sonderegger et al. 2015). Similarly, Deteix et al., (2023) proposed to adapt both the Yale and the JRC methods to derive a SR index for land resources.

Water and land being key resources for food systems, the JRC and Yale methods have been selected for this case study to derive SR CFs. They also provide SR indexes for a wide range of mineral resources (i.e. 82 for the JRC and 86 for the Yale method, compared to 31 for GeoPolRisk or 60 for ESSENZ). To ensure an in-depth comparison between the two criticality methods for the three types of resources, water SR indexes have been computed for this study following the JRC method.

The detailed methodology of the water SR developments and the data sources are available in the Supplementary Material 1 (Appendix A) of this article.

Table B.2 (from Supplementary Material 1 Appendix B) summarises the main parameters included in the three resource SR indexes within the Yale and JRC criticality frameworks.

2.2 Deriving Supply Risk Characterisation Factors (CFs)

For the LCIA, the aim is to compute CFs based on the resource SR indexes from the criticality methods. CFs were obtained for mineral, land and water resources, both for the Yale and JRC criticality methods. However, the two methods can differ in terms of material or spatial coverage (see

Table 1).

***Table 1** : Summary of the 6 Characterisation Factor sets; SI : Supplementary Information; NC : Not Characterised*

	Mineral		**Land**		**Water**	
	Yale	**JRC**	**Yale**	**JRC**	**Yale**	**JRC**
Method publication	Graedel et al., (2012)	Blengini et al., (2017)	Deteix et al., (2023)	Deteix et al., (2023)	Sonderegger et al., (2015)	This publication (see SI)
Spatial coverage	Global	European Union	Global	European Union	Global	Global
Spatial resolution (number of units covered)	World as a whole	European Union as a whole	Countries (76)	Countries (24)	Countries (159)	Countries (90)
Number of resources covered	86	82	1	1	1	1

Strategy to fill data gaps	NC	NC	World average	World average	World average	World average
SR index availability	https://doi.org/10.5281/zenodo.2561882	European Commission, (2020b)	https://doi.org/10.57745/RALP5G		SI of Sonderegger et al., (2015)	https://doi.org/10.57745/U8TLHN (method in SI)

For mineral resources, the available Yale SR indexes are non-spatialized, meaning the criticality of a mineral is only assessed from the worldwide perspective, unlike those from the JRC, that assess SR from the European Union perspective. The SR indexes were used directly as CFs for both methods. In cases where a mineral was not included in a method, it was not characterised in the impact assessment.

Concerning water resources, in the Yale framework, the SR indexes from Sonderegger et al., (2015) are spatialized at country level and were also used directly as CFs. However, for large-scale countries such as the United States, China or Brazil, the SR indexes are available at county levels. For these large countries, the national water SR CFs were obtained by computing the average of their county SR indexes. Within the JRC framework, water SR indexes were computed at the country level and used directly as CFs.

For land resources in both frameworks, land SR indexes were computed at the country level (Deteix et al. 2023) and used directly as CFs.

For land and water resources, when a country was not characterised by a CF, the average of the SRs weighted by the surface areas of the countries was used, equivalent to a global average CF.

2.3 Matching SR CFs with LCI

The LCIA resource supply risk characterisation model was developed according to the approaches proposed by Santillán-Saldivar et al., (2021, 2022) and Terlouw et al., (2019). The importance of resource use was quantified by the physical amount of resource recorded in LCIs of products or services (i.e., mass, area*time, volume, see Table **2**).

As described in Eq. (1), the SR CF was multiplied by the resource flows (obtained from the LCI). The resulting Supply Risk Potential (SRP) indicators (Santillán-Saldivar et al. 2022) are equivalent to classical LCIA midpoint indicators, but providing information on resource accessibility.

$$resource\ SRP = \sum_i F_i * SR_i$$

(1)

With:

- F_i: flow of resource i (dimension: see
- Table **2**)
- SR_i: Supply Risk CF of resource i (dimensionless)

***Table 2** : Natural resource flow (F_i) definitions and units according to the type of resource considered. FU: Functional Unit.*

	Mineral	**Land**	**Water**
F_i definition	Mass flow of mineral i	Land occupied in country i	Water withdrawn in country i
F_i unit	$kg.FU^{-1}$	$m^2.year.FU^{-1}$	$m^3.FU^{-1}$

Resource flows include the resource that is occupied (land) or extracted from the environment (mineral, water) and correspond to the inputs from the ecosphere into the technosphere. Resource outputs to the ecosphere are not taken into account in the calculation of SRs, unlike impact assessment procedures for certain resources such as water. In conventional LCIA, the water flow accounted for is the amount of water consumed in a given location, i.e. water withdrawn minus water released (Bayart et al. 2010; Pfister, Koehler, and Hellweg 2009). Therefore, the impact of certain processes such as electricity production (electricity dam, nuclear power plant) is low due to low water consumption. Supply Risk assessment focuses on access to a resource, and not on the impact induced by its consumption. If water is not accessible, for instance due to a drought, dams and nuclear power plants would be affected, no matter how little water they consume. This explains why the water flow accounted for in this model only concerns withdrawn water (see

Table **2**). In this case study, all withdrawn water flows were considered, without applying cut-off.

With this model, the potential supply risk increases if a system consumes more of the same resources, but also if it consumes additional resources. Conversely, the first way to reduce vulnerability to resources is to reduce the use of resources, by reducing quantities consumed, as highlighted by Cimprich et al., (2019).

2.3 Application to three contrasting bread supply chains

2.3.1 Goal and scope

The goal of the study is to compare the environmental impact and resource supply risk potential of three contrasting bread supply chains (BSC). Among food products, bread represents a staple food that is widely consumed worldwide and provides a number of essential nutrients (Dewettinck et al. 2008; Weegels 2019). Its supply should therefore be guaranteed while its environmental impact is reduced.

In addition, many different BSCs can coexist within a country. They differ according to agricultural production techniques (crop rotation, fertilisation, crop protection), milling and baking technologies or distribution channels (Kulak et al. 2015).

In this case study, the BSCs contrast according to four different aspects, i.e. i) agricultural production modes, ii) baking technologies, iii) total transport distances from field to point of sale and iv) number of intermediaries (see **Fig. 1**).

Bread from BSC A is produced with organic wheat grown in France without any irrigation. Wheat milling is performed on the farm, using an electric mill. The bread is baked in a wood furnace on the same site, and is transported to the point of sale by a small truck (30 km).

Bread from BSC B is made with protein-improved wheat from Spain grown with fertilisers, pesticides and irrigation. The wheat is transported to France to the milling site by lorry. The wheat flour is transported in paper bags (made from Portuguese eucalyptus wood plantations) to the industrial baking plant. The bread is then further transported by a small truck to the point of sale (total distance 750 km).

BSC C bread is a combination of the two previous BSCs. The wheat is identical to that of BSC A, and the processing and transport processes are those of BSC B.

The functional unit chosen is 1 kg of bread ready for consumption, delivered to the point of sale in a French city. **Fig. 1** describes the three BSCs modelled in this study. Production tools (farm machinery, mill, furnaces) as well as infrastructure (farm buildings, factory buildings, storage sheds, etc.) are included in the inventory.

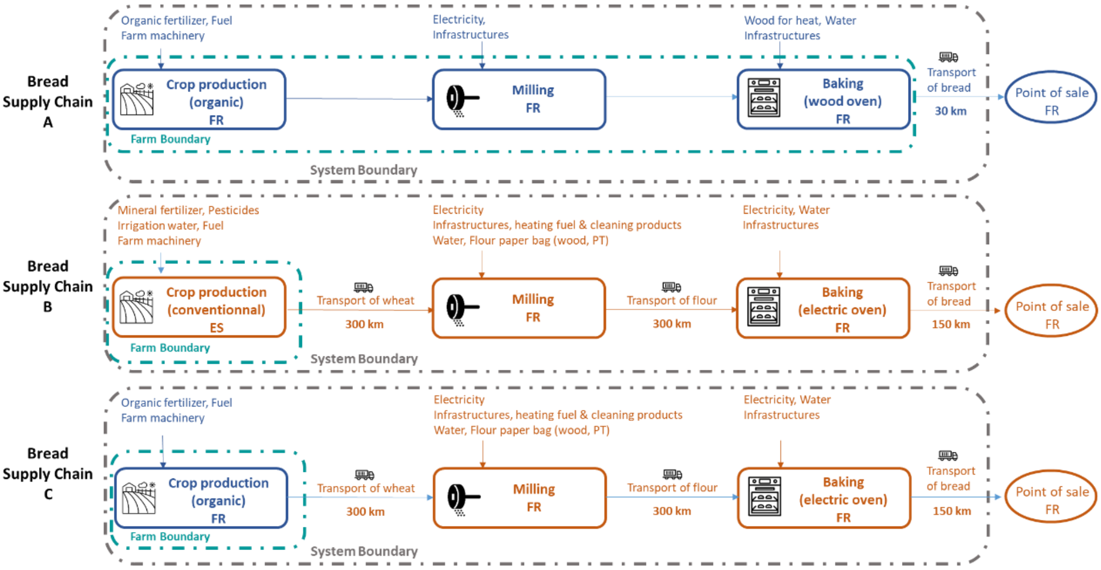

Fig. 1: System boundaries of the three bread supply chains. FR : France, ES : Spain, PT : Portugal

2.3.2 Life Cycle Inventory (LCI)

The three BSCs involve four stages, i.e. crop production, milling, baking, and final distribution. Data for the main parameters of the three scenarios are listed in Table 3 (e.g., transport distances, amounts of water and electricity consumed, etc.). The process inventory data (e.g. wheat growing stages, oven baking process, milling process etc.) and (field emissions) come from the Agribalyse (v 3.0) database (Koch and Salou 2020b). The background data (electricity mix, trucks, etc.) come from the Ecoinvent (v 3.9) databases. For BSC A, the quantities of energy and wheat for the milling process, as well as the bread recipe and baking energy, are taken from the work of Kulak et al. (2015).

Table D.1 D.2 and D.3 (see Supplementary Material 2) provide the detailed LCIs for the three BSCs.

Table 3 : Main parameters of the three bread supply chains. FR: France

	Bread supply chain A	**Bread supply chain B**	**Bread supply chain C**
Wheat country production	France	Spain	France
Irrigation volume	0 m^3/ha	1200 m^3/ha	0 m3/ha
Fertilizers	Organic	Minerals	Organic
Pesticides	No	Yes	No
Yield	4,00 tonnes/ha	5,96 tonnes/ha	4,00 tonnes/ha

Milling energy source	Electricity (FR)	Electricity (FR)	Electricity (FR)
Wheat conditioning	None	Kraft paper	Kraft paper
Baking energy source	Wood	Electricity (FR)	Electricity (FR)
Total transport distance (field to point of sale)	30 km	750 km	750 km
Transport type	Small truck	Lorry	Lorry
Point of sale	Small grocery	Supermarket	Supermarket

2.3.3 Life Cycle Impact Assessment (LCIA)

The environmental impacts of the three BSCs are compared at the endpoint level, as the synthesis of information at this level is straightforward and relevant for decision making (Van Hoof et al. 2013). The environmental impact assessment was carried out with the ReCiPe method, which provides endpoint impact result levels (Huijbregts et al. 2016). In addition, the Recipe method also provides midpoints results that allow for more in-depth analysis.

In addition to the environmental impact assessment, the potential supply risks of mineral, land and water resources are assessed with the Supply Risk characterisation model defined in section 2.1, providing three resource SRP indicators. The SRPs are compared to the ReCiPe midpoint impact results that characterise the same resource. In this case, the ReCiPe midpoint impacts are used, because the land resource is not considered for damage on the Resource AoP.

The LCA calculations are performed with the LCA software: Simapro. Supplementary Material 3 provides the elementary flows from Simapro that are included in the assessment as well as their corresponding Supply Risk CFs.

3) Results

This section first presents the results from the conventional LCA for environmental damage and secondly, the resource supply risk assessment, with a focus on each type of resource, i.e., mineral, land and water resources.

3.1 Environmental Impact

As shown in **Fig. 2**, BSC A presents a higher impact on both Ecosystem quality and Human health AoPs than BSC B and BSC C. Regarding Ecosystem Quality, the impact of BSC B is slightly less than that of BSC C, but for Human Health, the impact of BSC B is twice that of BSC C.

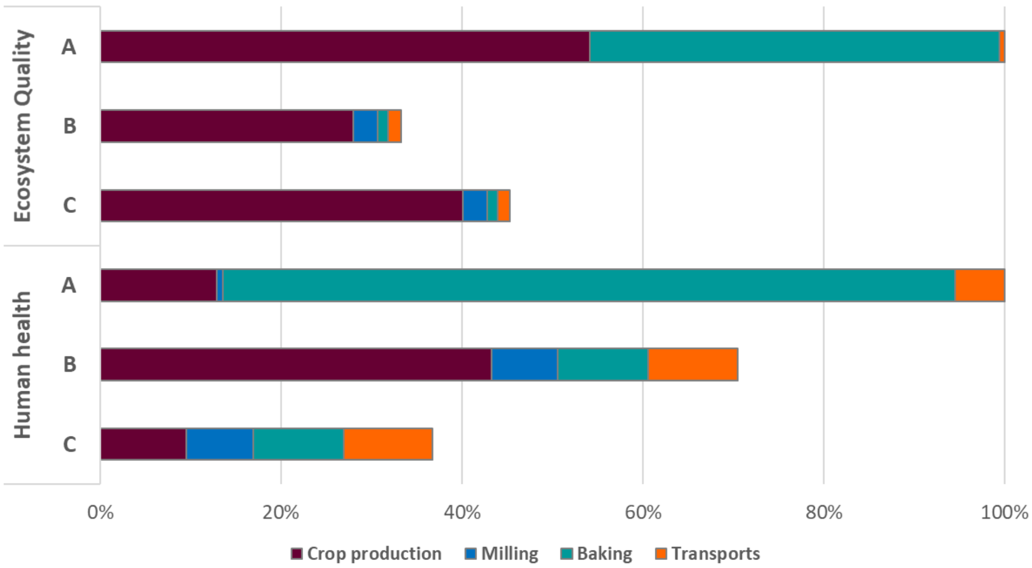

Fig. 2*: Environmental Impact at the endpoint level of the production and delivery to point of sale of 1kg of bread in three contrasted supply chains*

For Ecosystem Quality, the result for BSC A is explained primarily by the impact of land use on biodiversity, due to wheat production and the consumption of wood for baking. For the crop production phase, BSC A and C have a lower wheat yield than BSC B (Table 3). Thus, per kg of bread, BSC A&C require more land than BSC B, resulting in a greater impact on biodiversity due to land use (see Appendix C, Fig C6).

For BSC B, the impacts are mainly due to land use during the crop production phase, but also, to a lesser extent, to water consumption for irrigation, as well as field emissions due to the application of nitrogen fertilizer, which result in fine particulate matter (NH_3) and global warming (N_2O) (see Appendix C, Fig. C6). The milling, baking and transport phases are identical between BSC B and BSC C and contribute little to the impacts (see **Fig. 2**). However, the impact of BSC C is mainly due to land use, as for BSC A, because the agricultural processes in BSCs A and C are identical.

For Human Health, the major contributor for BSC A is the wood burning process associated with the baking phase, which is responsible for fine particulate matter and human toxicity. In addition, the agricultural production phase contributes to global warming (see Appendix C, Fig. C6) through N_2O emissions and tractor fuel consumption. For BSC B, the impacts occur mainly during the crop production phase and are due to NH_3 and NO_X emissions and their effect on fine particulate matter, N_2O and CO_2 emissions and their effect on global warming, and finally water consumption for irrigation in Spain (see Appendix C , Fig. C6). Damage on human health due to the milling, baking and transport phases are mainly incurred by global warming impact related to energy consumption. The impact of BSC C on Human Health is equally spread across the 4 phases of the supply chain. For the agricultural phase, the same impact pathways apply as for BSC A, while for the milling, baking and transport phases, the same impact pathways apply as for BSC B.

3.2 Resource supply risk and comparison with selected ReCiPe midpoint impacts

3.2.1 Mineral Supply Risk Potential (SRP)

Fig. 3 shows the results of the mineral SRP using the Yale and JRC criticality methods, in comparison with the ReCiPe midpoint impact category “mineral resource scarcity”.

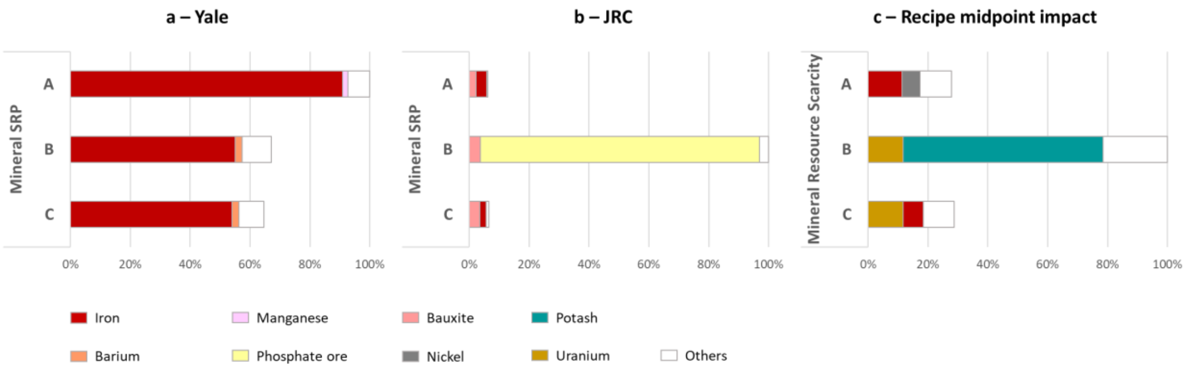

__Fig. 3__: Mineral Supply Risk Potential (SRP) (a & b) and mineral resource midpoints impacts (c) of the production and delivery to point of sale of 1kg of bread in three contrasting supply chains.

The Yale and JRC methods provide contrasted results. Whereas with the Yale method, the mineral SRP of BSC A is 30% higher than BSC B and BSC C (see **Fig. 3**-a), using the JRC method, the mineral SRP of BSC B is 90% higher than BSC A and BSC C (see **Fig. 3**-b). These differences can be explained by the fact that the two methods do not cover the same range of minerals. For example, phosphate is not included in the Yale method.

With the Yale method, iron, which is needed for infrastructure, is the main contributor to the impact for all types of supply chain (see **Fig. 3**-a). For BSC A, the infrastructures that contribute most to the impact through their iron consumption are the bread furnace (baking phase), agricultural machinery and transport for final delivery (see Appendix C, Fig. C2). For BSC B and BSC C, the infrastructures that contribute most are transport during the milling and baking phases and for final delivery, as well as industrial production tools (mill and furnace) (see Appendix C, Fig. C2).

With the JRC method, the main contributors to mineral SRP for BSC A and BSC C are iron and bauxite, which are used for furnaces, agricultural machinery and transport in both supply chains (see Appendix C, Fig. C3). For BSC B, phosphate ore shows, by far, the highest contribution to the mineral SRP (see **Fig. 3**-b), as phosphate is used as fertilizer for wheat production. BSC A and BSC C have no phosphate contribution to mineral SRP as phosphate used for crop production is not of mineral origin. Instead, it is found in organic manure, which is considered as supply risk free in this model.

The ReCiPe method also provides different results (see **Fig. 3**-c). For BSC A, the biggest contributors to the mineral resource scarcity impact were iron and nickel, both for furnace and agricultural machinery. For BSC B, the greatest contributors are potash for wheat fertilizing, and uranium, for nuclear electricity during the baking phase. For BSC C, the main contributor is uranium, for the production of electricity during the baking phase, like BSC B. The second largest contributor is iron for agricultural machinery, like BSC A. While uranium is not included in the Yale and JRC methods, the SR of potash is characterised by the JRC method. The difference between both results lies in the fact that they do not focus on the same mechanisms. The midpoint impact "Mineral Resource Scarcity" from ReCiPe is based on the Surplus Ore Potential method (Huijbregts et al. 2016), a future effort method, which characterises the consequences of mineral resource use for future generations (Berger et al. 2020; Sonderegger et al. 2020), while criticality methods characterise vulnerability to potential resource access restrictions.

3.2.2 Land Supply Risk Potential

Regarding land SRP, both methods converge and show that BSC A has the highest land SRP, BSC B has the lowest impact, and the impact of BSC C lies between the two (**Fig. 4**-a and **Fig. 4**-b). These results are similar to the biodiversity impact due to land use with the ReCiPe method (**Fig. 4**-c) and can be explained by two factors. Firstly, by the yield effect during the crop production phase and, secondly, by the fact that BSC A uses forest land to produce wood for baking, whereas BSCs B and C only use wood to make paper bags for transporting wheat flour.

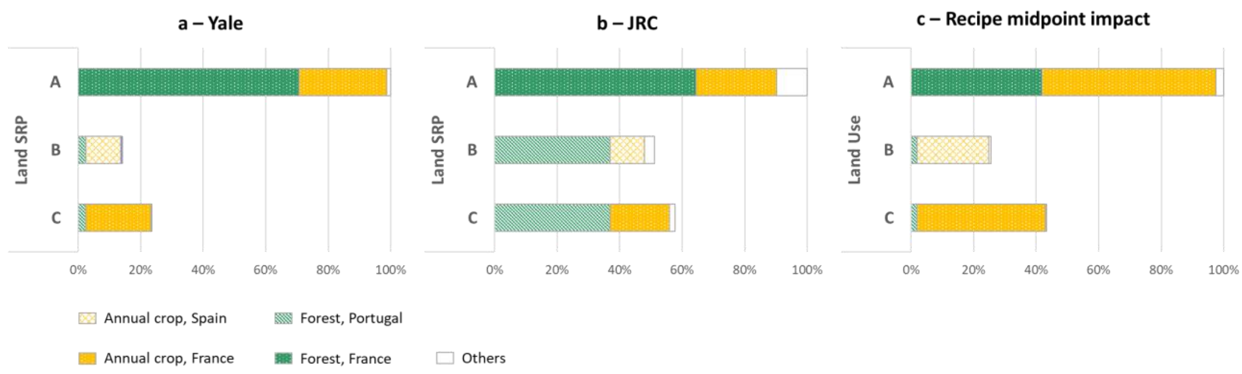

Fig. 4: Land Supply Risk Potential (SRP) (a & b) and land use midpoints impacts (c) of the production and delivery to point of sale of 1kg of bread in three contrasting supply chains.

With the JRC method, BSC B&C present a much larger forest contribution to land SRP than with the Yale method. Indeed, the JRC method considers the land SR in Portugal (producer of wood for paper) to be high, whereas the land SRP for Portugal in the Yale method is significantly lower.

For both methods, BSC A land SRP is mainly due to forest occupation (see **Fig. 4**-a and **Fig. 4**-b), whereas in ReCiPe, the land use impact on biodiversity is equally attributed between agricultural land and forest occupation (**Fig. 4**-c), because these two impact categories do not focus on the same mechanisms. Indeed, land use impact on biodiversity involves changes in species composition (Huijbregts et al. 2016), whereas the land SRP focusses on the vulnerability to potential land access restriction. In addition, land SRP is unchanged, regardless of the type of land use (only surface areas matter) (Deteix et al. 2023), while the impacts on biodiversity differ according to the type of land use (Koellner and Scholz 2007).

3.2.3 Water Supply Risk Potential

In terms of water SRP, the two methods converge on the same conclusions, both BSC B and BSC C have a similar water SRP, which is higher than that of BSC A (see **Fig. 5**-a and **Fig. 5**-b), mainly due to higher electricity consumption in France. This results from the high volumes of water withdrawn for hydroelectricity production in the French electrical mix, and not for nuclear electricity production.

In this section, unlike the analyses of the contribution of land and mineral resources, the contribution analysis is carried out by considering the contributions of technosphere processes (e.g. electricity, France) and not elementary flows (e.g. water, France). This analysis provides information on the processes vulnerable to water supply restrictions.

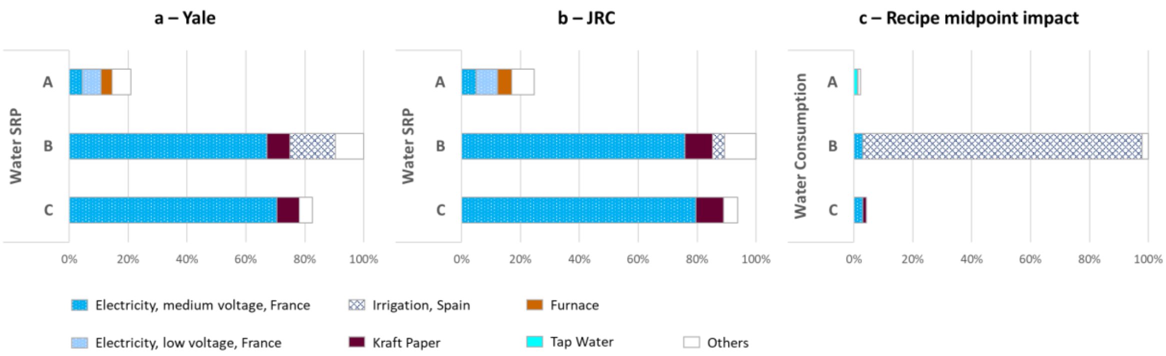

***Fig. 5**: Water Supply Risk Potential (SRP) (a, b) and water scarcity midpoints impacts (c) of the production and delivery to point of sale of 1kg of bread in three contrasting supply chains.*

For both methods, BSC A presents a water SRP that is essentially due to electricity consumption and furnace production (see **Fig. 5**-a and **Fig. 5**-b). The phase in which the most electricity is consumed is the baking phase (see Appendix C, Fig. C4 and Fig. C5). For BSC B and C the water SRP is primarily due to electricity consumption (see **Fig. 5**-a and **Fig. 5**-b) for baking (see Appendix C, Fig. C4 and Fig. C5), and water withdrawals for the wheat flour paper bags. For BSC B there is also the contribution of water withdrawals for irrigation, which is more marked with the Yale method (see **Fig. 5**-a).

These results are quite different to the water consumption impact from ReCiPe (see **Fig. 5**-c), which is mainly related to irrigation for BSC B, tap water for BSC A and electricity production for BSC C. Indeed, water SRP accounts for water withdrawal, while the ReCiPe method only considers consumed water (see Table **2**).

4) Discussion

4.1 Key findings from the case study

Resource SRPs were quantified along with the environmental impacts of three contrasting BSCs, using supply risk indexes from two criticality methods.

The environmental impact assessment of the BSCs reveals that crop production is the most impactful phase. These results concur with previous works where the agricultural production stage is the stage that contributes most to the environmental impacts of agri-food products (Castellani, Beylot, and Sala 2019; Poore and Nemecek 2018). The yield effect explains why the impact of organic wheat in BSCs A and C is higher than that of conventional wheat in BSC B. This result is in line with the literature, which shows that yield is an important parameter in explaining differences in results between LCAs of organic and conventional products (van der Werf, Knudsen, and Cederberg 2020). However, these differences vary according to the type of crop and agricultural practices (Boschiero et al. 2023).

For the Human Health AoP, it is the crop production and food processing stages (milling and baking phases) that contribute most to the damage in the three supply chains. In addition, for BSC B and C, which have much longer transport distances than BSC A, transport makes a substantial contribution to damage. These results are in line with the literature, which shows that food processing and logistics stages contribute significantly to the environmental impact of food products, with the exception of meat-based food products (Notarnicola et al. 2017).

Regarding resources SRPs, with the JRC method, BSC A and C have a lower SRP for mineral and water resources than BSC B, but higher for land. With the Yale method, BSC A has a lower SRP for water resources than BSC B and C, while mineral and land resource SRPs are higher. Results for agricultural products indicate that, similarly to technological products (Bach et al. 2018), trade-offs can occur between environmental impacts and vulnerability to resource supply shortages. Moreover, the application of criticality metrics to agricultural products provides complementary information to environmental impact assessment. Indeed, environmental and criticality assessments are found to characterise different impact pathways (André and Ljunggren 2021; Berger et al. 2020; Cimprich et al. 2019; Sonnemann et al. 2015).

The contribution analysis reveals that for the BSC A and C, the mineral SRP hotspots are the machinery. For the BSC B, the mineral SRP hotspots are agricultural machinery according to the Yale method, and phosphorous fertilizer according to the JRC method. With both methods and for all three supply chains, land SRP results from agricultural and forestry production, whether for food (wheat production), fibre (wood for paper) or energy production (firewood). Finally, water SRP is primarily related to electricity production.

These results highlight the importance of considering a life cycle perspective for food product criticality assessments. Firstly, taking into account all life cycle phases allows for vulnerability hot spots to be identified along the whole supply chain. These hotspots remain generally unnoticed in studies that focus on territory food self-sufficiency and are limited to the agricultural production phase (Clapp 2017; Fader et al. 2013). Secondly, the multi-criteria approach (i.e. three resources SRP indexes are taken into account) also allows for vulnerabilities to be identified due to the different resources required for agricultural products.

These results therefore reveal significant application perspectives, even though certain improvements could still be made from methodological and operational points of view. These issues are further discussed below.

4.2 Importance of the criticality method

The contribution analysis of resources and processes highlights the differences between the two studied criticality methods due to divergent methodological aspects.

Firstly, both methods are not based on the same resource or spatial coverage. For example, for BSC B, phosphate is not included in the Yale method, whereas it is the largest contributor to the BSC B mineral SRP using the JRC method. Similarly, bauxite, the second largest contributor to BSC B mineral SRP using the Yale method, is not covered by the JRC method.

Secondly, both methods do not take into account the same parameters that affect the SR of a resource. For example for mineral resources, the Yale method considers geological availability as a supply risk parameter, whereas the JRC only focuses on economic and geopolitical accessibility. This explains the contrasting contributions of iron to mineral SRP between the two methods for both supply chains. For land and water, political stability is a parameter considered by the Yale method and not by the JRC. Conversely, the JRC method comprises a land recycling parameter for land supply risk as well as a non-conventional water resource rate for water supply risk, while these are not part of the Yale method.

Finally, the two methods do not apply the same equations when aggregating the different parameters into a final SR index. The parameters affecting the SR in both methods do not share the same weight. This discrepancy leads, for example, to the higher contribution of the Portuguese forest to the BSC B and land SRP with the JRC method than with the Yale method (**Fig. 3**-a and **Fig. 3**-b). According to the JRC method, the land SR of Portugal is driven by a relatively high concentration in land ownership and low quality of land administration. While the effect of these two parameters is amplified with the JRC aggregation method, it is attenuated with the Yale method (Deteix et al. 2023).

These results therefore highlight the strong influence that the choice of a method may have on the outputs of a criticality assessment, in line with conclusions of previous studies that have compared several criticality methods applied to the same products (Cimprich et al. 2019; Terlouw et al. 2019).

Consequently, the choice of a supply risk method should first be based on the purpose of the study and the scope of the method, particularly in terms of resource coverage (i.e. the types of resource that are included, depending on the application considered, the spatial distribution available, etc.).

For example, for the food sector, it is crucial that phosphate is included, due to its importance for food systems (Cordell and Neset 2014). For criticality assessment in this sector, the Yale method should therefore be extended to phosphate. However, the main constraint to accomplishing this extension lies in calculating the depletion time parameter. This parameter requires knowledge of the lifetime of the phosphate when it is used, which is not as easily quantifiable as for metals.

More generally, to use a criticality method in a case study, it is necessary to ensure that the critical minerals for the sector under study are well covered (European Commission 2020a). If several methods can be applied, a sensitivity analysis must be carried out on the results calculated with the different methods.

In the longer term, it would be preferable to have a harmonised criticality assessment method applicable to all types of products and sectors, like the impact methods used in LCA. To this end, the method should exhaustively cover the natural resources used for human activities. This method could also cover all the parameters taken into account in each of the four criticality methods potentially adapted to LCA. However, care must be taken to avoid double counting by discarding certain redundant parameters, such as the trade barrier index from ESSENZ and that from the JRC method.

In this case study, the Yale and JRC criticality methods were chosen as they are renowned, operational and characterise land and water, two key resources for agricultural products. Nevertheless, other methods are presently recommended for the characterisation of supply risk in LCA such as the GeoPolRisk and ESSENZ methods (Hackenhaar et al. 2022; Sonderegger et al. 2020). These recent methods are still under development, and extensions could be made to add new resources such as phosphate, water and land to provide a criticality assessment tailored to agricultural products. Building on the work of Sonderegger et al., (2015) and Deteix et al., (2023), it would be possible to take up the conceptual framework of each method. Therefore, with additional data collection (for phosphate notably), and a few adaptations (e.g. adapting the Supply Risk parameters to the land and water resources), it would be possible to extend the range of resources considered.

4.3 Spatial representativeness

The land and water SR indexes are computed at the country level for both the Yale and JRC criticality methods. However, the Yale water SR from Sonderegger et al., (2015) are also available at finer resolution. For instance, water SRs for Brazil or Australia are available for different administrative counties. In the present study, the water SRs for these large countries were obtained by averaging the county SR. This simple approach does not allow for in depth spatial representativeness, so future versions could apply methods where counties are weighted by share of country space or population, or other more sophisticated methods such as those described by Mutel et al., (2012).

Concerning the issue of missing values, for both methods, the global average value is calculated by weighting the SRs by the surface area of each country. In this way, more weight is given to the largest countries. While this approach is relevant for land, other approaches such as the one mentioned above could be used for water. Furthermore for both methods, the calculation of the world average water SR CF, used for characterising water flows in countries where the specific CF is not available, does not take into account the same countries (159 for the Yale method, 90 for the JRC method). Nevertheless, the two SR indexes present similar distributions, implying that the mean values have the same representativeness.

The mineral SR indexes from Yale are not regionalised, while those from the JRC are regionalised at the European level. For finer analysis, the criticality methods would need to be spatially differentiated, as suggested by Ioannidou et al., (2019). In line with land and water resources, they could be first computed at the country level, as for the SCARCE method for Germany (Arendt et al. 2020), and with GeoPolRisk for several countries and groups of regions (Koyamparambath et al. 2022). However, this means LCI would need to be spatialized, and LCI and CF would have to be matched, which is one of the challenges in LCA spatialization (Patouillard et al. 2018).

4.4 Aggregating resources

The LCA characterisation model developed in this study is based on the important assumption that each resource has the same importance for the system, whatever its substitutability, i.e. its ability to be replaced by another resource fulfilling the same function. Within a single type of resource, i.e. mineral, land or water, the aggregation between resources is done by summing up the individual SRPs. The impact of supply disruption is therefore assessed by taking into account only the physical relevance of the resource, and not the resource substitutability. With this approach, the impact on the system of a resource restriction remains the same whatever the resource. Further developments on the model could integrate substitutability metrics, as has begun to be done in the GeoPolRisk method (Santillán-Saldivar et al. 2021). Concerning mineral resources, this latter method proposes to use the price elasticity of the different minerals as an aggregation factor to reflect the economic impact of a supply disruption. As the data is currently only available for a small number of minerals, this method could not be used in the present case study. Concerning land and water, price elasticity data have been studied (Garrone, Grilli, and Marzano 2019; Tabeau, Helming, and Philippidis 2017), but are restricted to a certain type of resource use (agricultural land, residential water) and/or to a particular location (a city, a state,…). However, as water and land do not share the same functions and governance (Ostrom 1990) as mineral resources, the use of price elasticity as a proxy for substitutability might not be relevant.

Finally, the present case study focuses on three types of resources, although food systems also require other inputs such as energy, transport or agricultural commodities. These inputs belong to the technosphere in LCA and can also undergo supply risk restrictions. To provide a better integrated assessment, future LCA criticality assessments should strive to characterise these input supply risks by exploring other approaches that focus on technosphere products, as suggested by Helbig et al., (2016) and Berr et al., (2022).

5) Conclusion

The integration of criticality methods into LCA allows for the assessment of both the vulnerability of a product to resource supply risks as well as its environmental impacts. This joint assessment has so far only been applied to high-tech products such as batteries. In spite of the vulnerability of agricultural products to supply risks, no study has yet investigated their criticality.

The present work therefore proposed to integrate criticality into LCA for agricultural and food products, by taking into account three key resources, i.e. minerals, water and land.

The integration of criticality into LCA was carried out by deriving characterisation factors from the SR indexes of the Yale and JRC methods, which were in turn applied to a case study: the LCA of three contrasting bread supply chains.

The LCA results showed that BSC B has fewer environmental impacts per kilogram of bread produced than BSC A, but a higher mineral and water Supply Risk Potential and a lower land Supply Risk Potential than BSC A. As BSC C is a combination of BSC A and BSC B, its SRP are close to those of A or B, depending on the contribution of the supply chain stages to the three resources SRP. The analysis of the results highlights the relevance in characterising Supply Risk profiles for agri-food products, since potential trade-offs between environmental performances and vulnerability to resource shortages can arise. Furthermore, the contribution analysis stresses the importance of applying a life cycle perspective in order to perform a food product supply risk assessment, because vulnerability trade-offs are known to occur between life cycle phases or resource types.

Nevertheless, the comparison between the two criticality methods also reveals the variability in terms of methodological choices, and calls for the development of further food product case studies implementing different criticality methods. In order to provide more meaningful insights, future investigations should also integrate other types of key resources for an agricultural product, such as transport, energy or agricultural commodities.

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CRediT authorship contribution statement

Lazare Deteix: Conceptualization; Methodology; Investigation; Software; Data Curation; Writing-Original Draft. **Thibault Salou:** Conceptualization; Methodology; Investigation; Validation; Supervision. **Eléonore Loiseau:** Conceptualization; Methodology; Investigation; Validation; Supervision; Writing-Review & Editing; Project administration.

Declaration of competing interest

The authors declare no competing interests.

Data Availability

All data generated or analysed during this study are included in this published article and its supplementary information files, and the water supply risk indexes are available in the DATA INRAE repository : https://doi.org/10.57745/U8TLHN

Supplementary information associated with this article include 3 Supplementary Material files. Supplementary Material 1 contains Appendix A that describes the water supply risk methodology, Appendix B that presents the harmonization across resources within the Yale and JRC criticality methods, Appendix C that provides additional results of the case study, and a description of the files from Supplementary Material 2 and 3.

Supplementary Material 2 is an excel file providing the Life Cycle Inventory of the three BSCs.

Supplementary Material 3 is a zip file containing 3 files that provide the elementary flows from the Simapro software that are included in the assessment as well as their corresponding Supply Risk CFs.

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