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How to increase sustainability in the Finnish wine supply chain? Insights from a country of origin based greenhouse gas emissions analysis

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ABSTRACT

As wine supply chains become increasingly globalized, sustainability issues take on ever greater importance. This is the first study to analyse the environmental sustainability aspect of greenhouse gas (GHG) emissions from a global wine supply chain perspective, covering just over 90% of Finland's wine imports. Lacking substantial domestic production capacity, virtually all wine consumed in Finland is imported. Finland is comparable to its Nordic neighbours, Sweden and Norway, in this respect.

The Life Cycle Assessment (LCA) methodology was combined with sensitivity and scenario analyses to investigate GHG emissions implications from prospective policy changes. Our results spotlight differences related to wine production in the eight main wine producing countries for the Finnish market (Australia, Chile, France, Germany, Italy, Spain, South Africa, and the United States), related logistics, and all packaging types for wine used in Finland (glass bottle, Bag-in-Box, PET bottle, beverage carton, and pouch). We found an average value of 1.23 kg CO_2e for 0.75 L wine consumed in Finland, ranging from 0.59 kg CO_2e for French wine in a bag-in-box packaging to 1.92 kg CO_2e for Australian wine in a glass bottle. After identifying the main GHG emission hotspots in the wine supply chain, our scenario analyses highlight the effects of reducing glass bottle weight, moving away from glass packaging toward bag-in-box, increasing bulk wine export volumes to Finland, and following the European Commission's Energy 2020 strategy which targets increasing energy efficiency by 20 percent.

1. Introduction

Seeking to improve the sustainability of supply chains, practitioners and scholars of sustainable supply chain management are increasingly using systems approaches, including analyses of greenhouse gas emissions (GHG), environmentally extended input-output analyses, and Life Cycle Assessment (LCA) (Blass and Corbett, 2018; Pfister et al., 2017; Ahi and Searcy, 2013). These approaches, and in particular LCA, have gained in importance due to rising demands for transparency and principle-based sustainability standards (Ayuso et al., 2016), and the need for a harmonization of sustainability claims, as the EU Commission's Environmental Footprint pilot evidences (European Commission, 2018). LCA is widely used to assess the environmental impacts of a product, organization, or service, focusing on the resources used throughout its lifecycle, i.e. from raw material acquisition to waste management (ISO, 2006a and 2006b; Finnveden et al., 2009; Hellweg and i Canals, 2014). These LCA and related systems approaches can help supply chain members to identify cost saving opportunities through energy efficiency initiatives (Matthews et al., 2008; Song et al., 2018) or provide opportunities for restructuring entire supply chains (Steiner and Jäger, 2018; Linton et al., 2007).

The reduction of the overall agri-food sector's carbon footprint (CF) is seen as one important potential contribution to mitigate such anthropogenic GHG emissions (Vermeulen et al., 2012). The

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GHG emissions are indeed the most frequently used environmental indicator to address adverse environmental impacts arising from the agri-food chain (Bosco et al., 2011; Pattara et al., 2012; Steiner et al., 2017; Ferrara and De Feo, 2018). It is thus not surprising that the CF has emerged as a key indicator for sustainability in a number of wine studies (Bonamente et al., 2016; Merli et al., 2018; Flores, 2018; Szolnoki, 2013).

The wine supply chain was found to contribute 0.3% to the total of annual global GHG emissions (Rugani et al., 2013), which is remarkable for a single product category. With the increasingly global sourcing of wines as part of a highly globalized wine supply chain (Anderson et al., 2017), we anticipate that the issue of GHG emissions as part of a global supply chain perspective will only gain in importance in the future global wine supply chain. With around 11 L per year, Finland has a modest per-capita consumption of wine by global standards (Statistics Finland, 2016; Alko, 2017; Wine Institute, 2017). Nevertheless, wine plays an important role in the Finnish alcohol supply chain (Wine Intelligence, 2018). Lacking significant domestic production capacity, virtually all wine consumed in Finland is imported (CBI, 2016), which heightens the likely emissions impact of its underlying wine supply chain relative to less wine import-dependent countries.

The purpose of this study is to identify key sources of GHG emissions in the Finnish wine supply chain and to model GHG reduction potentials that could be used as inputs for policy changes. While our analysis captures around 90% of import volumes of wines entering Finland, we focus on the Country of Origin (COO) of the wine grape production and vinification, packaging types (glass bottle, Bag-in-Box (BiB), beverage cartons, PET bottles, pouch), and transportation (bulk wine and bottled wine) from the COO to the Point of Sale (POS) in the Finnish wine market. Finland is notable, not only for its focus on ecological modernization through innovation (Mickwitz et al., 2011) and on corporate social sustainability (Panapanaan et al., 2003), but also for its institutional parallels with other Nordic countries with regard to a state-controlled alcohol monopoly (Norway: Rossow and Storvoll, 2014; Sweden: Norström et al., 2010). Furthermore, like many other developed economies, the Finnish wine market is influenced by a rising demand from increasingly ecologically conscious consumers (Euromonitor, 2018).

A related study from Weidema et al. (2016) describes environmental impacts from the Nordic alcoholic beverages industry. Contrasting our study, the results represent an aggregate supply chain perspective for all Nordic countries, not specifically focusing on Finland and not displaying differences in GHG emissions per COO or packaging type. On behalf of Alko, the Finnish national alcoholic beverage retailing monopoly, Päällysaho et al. (2018) investigate the environmental impacts from primary packaging materials of wine consumed in Finland using Ecoinvent data.

In contrast to previous studies reviewed by Ferrara and De Feo (2018), the majority of which have omitted to perform a sensitivity analysis, we employ a Monte Carlo-based sensitivity analysis and scenario analyses. The former allows for the determination of the robustness of the analysis, evaluating the extent to which the estimated results are influenced by uncertainties in the data; the latter test the potential effects of variations in the life cycle process. Therefore, our study not only extends previous LCA-based work from the wine sector, it also provides insights into the global wine supply chains, identifying leverage points for improving its sustainability, while enhancing the understanding of GHG emissions from the wine supply chain in Finland.

The remainder of the paper is structured as follows. Section (2) provides a brief literature review, and section (3) introduces methods and data. Section (4) presents the results, including sensitivity and scenario analyses, followed by a discussion (5) and concluding section (6).

2. Literature

The literature on sustainable supply chain management has rapidly been integrating concepts from industrial ecology, related systems perspectives, and a variety of life-cycle approaches (e.g. Seuring, 2004; Seuring and Müller, 2008; Genovese et al., 2017; Blass and Corbett, 2018). Nevertheless, currently there is still no comprehensive, empirically grounded understanding of how companies address sustainability in their supply chains (Thorlakson et al., 2018). On this background, LCAs have been developed and implemented by companies to assess environmental impacts of supply chains connected to products or services from cradle to grave (Guinée and van der Voet, 2017) and applied at various additional levels (e.g. organizations and regions), to at least partly investigate and pinpoint environmental hotspots in complex supply chains (Hellweg and i Canals, 2014). In spite of an ongoing discussion about the relationship between LCA and footprint analyses (e.g. Pfister et al., 2017), a widespread use of LCA has ensued in both industrial ecology and supply chain management (e.g. Guinée and Heijungs, 2017; Notarnicola et al., 2017; Blass and Corbett, 2018; Crawford et al., 2018).

Considering a special case of LCA where the focus is on carbon footprints, a growing number of studies has conducted product-level carbon footprint analyses, where the focus is on one environmental impact category only, namely the global warming potential impact category corresponding to GHG emissions (Finkbeiner, 2009; Kronborg Jensen, 2012; Navarro et al., 2017a; Blass and Corbett, 2018; Scrucca et al., 2018). A large number of such studies on GHG emissions has been conducted outside of the wine sector, including in the wind electricity sector (Padey et al., 2012) and the construction industry (Akan et al., 2017).

In the wine sector, an increasing number of studies on GHG emissions have focused on various stages of the wine supply chain, contributing to a better understanding of emission "hotspots" and resulting emissions reduction options (e.g. Jradi et al., 2018; Ponstein et al., 2019; Navarro et al., 2017b; Marras et al., 2015; Steenwerth et al., 2015; Villanueva-Rey et al., 2014; Vázquez-Rowe et al., 2013; Point et al., 2012). The analysis by Jradi et al. (2018) tracks carbon footprint in French vineyards using a data envelopment analysis as a quantitative approach to measuring efficiency. In contrast, Marras et al. (2015) analyse the CF of a mature vineyard during the grape production process in a typical vineyard in southern Italy, using the Eddy Covariance technique to measure the CO_2 exchange in vineyards. Steenwerth et al. (2015) provide an analysis of GHG emissions, energy use, and freshwater use in wine grape production across common vineyard management scenarios in two representative wine-growing areas of California, highlighting the importance of regional distinctions in wine grape production. Villanueva-Rey et al. (2014) provide a comparative LCA, contrasting biodynamic vs. conventional viticulture activities in Spain, while integrating land use and labour in their LCA.

Those studies that are closely related to our analysis include Amienyo et al. (2014), Point et al. (2012), Vázquez-Rowe et al. (2013), and Navarro et al. (2017b). In particular, Amienyo et al. (2014) assessed wine produced in Australia and consumed in England based on primary data from an Australian wine producer. They model a change in the packaging location from Australia to England, estimating the GHG emission reduction arising from the lower transport weight of wine that was then shipped as bulk. Point et al. (2012) perform a comprehensive analysis of the complete supply chain of wine produced in Canada, including the transportation from the winery to the POS, the shopping trip by the consumer and refrigeration before consumption. Vázquez-Rowe et al. (2013) calculate the CF of nine different types of wine in three different European nations (Italy, Luxembourg, and Spain) to determine the main reasons for varying CF results, highlighting the role of input optimization and the importance of legislative restrictions for achieving good environmental standards. Navarro et al. (2017b) use inventory data on wine production systems from a total of eighteen wineries located in major wine producing regions in Spain and the South of France. In sum, the above-mentioned literature documents the prevalence of case-study types of LCA analyses in the wine sector, concentrating on single countries or production regions, but not providing a comprehensive approach to a global wine supply chain.

Table 1 provides a comparative summary of the above-named LCA studies in the wine sector to better clarify the contributions of our study. For further literature reviews, see Rugani et al. (2013), Navarro et al. (2017a), Ferrara and De Feo (2018) and Scrucca et al. (2018).

3. Methods and data

3.1. Methodological framework

The underlying method for our subsequent analysis is based on the ISO 14040, and 14044 methodology for LCA (ISO, 2006a; 2006b). The strength of LCA lies with the standardization of the assessment process, replicable for different products and each production process or sub-process related to these products (Curran, 2008). The ISO 14040: 2006 standard (reviewed and confirmed in 2016) specifies the definition of the goal and scope of the LCA, the life cycle inventory analysis phase, the life cycle impact assessment phase, the life cycle interpretation phase, reporting and critical review of the LCA, limitations of the LCA, and the relationship between the LCA phases. In the subsequent analysis, we follow the Corporate Value Chain (Scope 3) Accounting and Reporting Standard (WRI, WBCSD, 2013), which opts for the measurement of GHG emissions originating across the full supply chain of a product or service. This standard provides guidance for the assessment of the CF of the wine supply chain from the perspective of the distributor of wine in Finland, which is the alcohol monopoly Alko. We define CF as a measure of '*the exclusive total amount of carbon dioxide emissions that is directly and indirectly caused by an activity or is accumulated over the life stages of a product*' (Wiedmann and Minx, 2008: 4), where the '*total amount of carbon dioxide*' is measured in kilograms or other mass units.

3.2. System description

As illustrated in Fig. 1, the system boundaries were comprised of the four main steps of value creation of the wine supply chain, notably viticulture, vinification, packaging and transport (Ponstein et al., 2019). Looking at a global supply chain, the value creation took place at several different locations. Viticulture and vinification occurred in the respective COO. The bottling process was performed either in the COO or in Finland. When wine was bottled in the COO, we assumed this to take place at the same winery where the wine was produced and filled into glass bottles which were placed into cardboard boxes of six bottles. We did not assume the use of other packaging material than glass bottles in this instance. Subsequently, the bottled wine was trucked to the port, shipped to Finland, and trucked to the warehouse and POS. For wine bottled in Finland, we assumed the packaging into flexitanks at the winery in the respective COO and transport to Finland. Once unloaded in Helsinki, the bulk wine was trucked to a bottling facility. Here, we assumed the bulk wine to be bottled into glass bottles, BiB, beverage cartons, PET bottles, and pouches based on data provided by Alko, and trucked to the POS subsequently.

We excluded GHG emissions from wrapping foil, additives and cleaning agents, since previous studies showed a low impact of these on the final results (Ponstein et al., 2019; Navarro et al., 2017b). Due to lack of data, we ignored GHG emissions related to eventual losses of cooling agents, disposal of cooling agents, and related to the consumption of wine (incl. refrigeration and washing of glasses). Although shopping trips by the final consumer can have an important effect on overall GHG emissions (e.g. Point et al., 2012; Amienyo et al., 2014), they were neglected due to lack of data.

3.3. Modelling and assumptions

3.3.1. Wine imports

UN Comtrade (2017) was used to identify the main COOs of wine consumed in Finland, providing import and export flows of 293 (reporting) countries. From this database, we extracted the 41 countries which exported wine to Finland in 2017, disaggregating the wine trade flows into two main categories: bottled and bulk wine. We followed the methodology proposed by Mariani et al. (2012) to identify such categories, i.e. bottled wine was coded as '220421' with the Harmonised System at a six-digit level of disaggregation, while bulk wine was defined as the difference between the total volume of wine exported to Finland and those volumes of bottled still and sparkling ('220410' code) wine.

To derive the most relevant COOs, those countries individually responsible for at least 5% of the total volume of bulk and bottled wine imports of Finland were included in our analysis. Sparkling wine was excluded as it is a different product, even though Mediterranean countries (France, Italy, and Spain) supply almost 80% of the total volume of sparkling wine consumed in Finland.

Consequently, the COOs chosen for the subsequent analysis were Australia, Chile, France, Germany, Italy, Spain, South Africa, and the U.S. (California). They jointly accounted for 88% of the total bottled wine and 93% of the total bulk wine consumed in Finland in 2017 (54.2 million L), with varying shares of bulk and bottled wine (Table 2).

The functional unit (FU) was 0.75 L of wine and we assumed a wine yield of 0.75 (Ponstein et al., 2019), hence 1 kg of wine grapes equals 0.75 L of wine.

3.3.2. Emission factors

Emission factors were retrieved from the Ecoinvent database V.3.4 (Ecoinvent, 2017) and from Defra (2016) for fossil fuels, which include pre-chain emissions. Emission factors for packaging materials other than glass bottles were retrieved from Päällysaho et al. (2018), who provided data specifically for the Finnish market (cf. Fig. 2). Concerning the direct GHG emissions from the use of organic and synthetic nitrogen (N) fertilizers, we relied on the IPCC (2006) methodology and employed an emission factor of 1% of N_2O–N per kg N-fertilizer applied to vineyard soil. Indirect emissions from N losses were not considered (Ponstein et al., 2019). To include GHG emissions from the production and transport of a range of possible sources of synthetic N fertilizers, we consider the range provided at regional level by Kool et al. (2012). Further, transports based on the inventory 'market for nitrogen fertilizer, as N - GLO' (Ecoinvent, 2017) were included. We did not attribute GHG emission to the production of organic N fertilizers, assuming them to be waste streams from the winery, but accounted for a transport distance of 50 km ['market for transport, freight, lorry, unspecified - GLO'].

3.3.3. Sensitivity analysis

In order to assess the uncertainties underlying LCA which arise from parameters and model uncertainty, spatial and temporal variability, and variability between sources (Bjorklund, 2002), and to derive a robust decision from a supply chain management

Table 1
Comparison of literature concerning system boundaries, scenario analysis, and sensitivity analysis.

Authors	Location(s)	Type of wine	System boundaries							Environmental impact indicator(s)	Scenario analysis	Sensitivity analysis
			Vineyard planting	Viticulture	Wine making	Packaging	Transport distribution	Storage and consumption	End-of life			
Ponstein et al. (2019)	Germany	Red and white	X	X	X	X				GHG	1) reduction in bottle weight; 2) reuse of wine bottles; 3) increasing bottle volume; 4) replacement of grid electricity by renewable energy	Yes
Navarro et al. (2017b)	France and Spain	Red and white	X	X	X	X				GHG	No	No
Meneses et al. (2016)	Spain	Red		X	X	X	X	X	X	ALOP, CC, FE, HT, TA, WDP	No	Yes
Steenwerth et al. (2015)	USA (California)	Red and white	X	X	X	X	X			ED, GWP, WDP	240 production scenarios	Yes (10% reduction in pumping energy)
Marras et al. (2015)	Italy	White	X	X						GHG	No	No
Villanueva-Rey et al. (2014)	Spain	White	X	X	X					ADP, AP, EP, GWP, ODP, POF	No	No
Amienyo et al. (2014)	Australia (production) and England (consumption)	Red		X	X	X	X	X	X	ADP, AP, ED, EP, FAETP, GWP, HT, MAETP, ODP, POF, TETP, WDP	1) bulk shipping of wine; 2) bottling closer to the final market; 3) recycling glass content; 4) reduced bottle weight; 5) exchanging glass bottles by beverage cartons	No
Vázquez-Rowe et al. (2013)	Italy, Luxembourg, and Spain	Red and white	X	X	X	X				GHG	No	No
Neto et al. (2013)	Portugal	White	X	X	X	X	X			ADP, AP, EP, FAETP, GWP, HT, LC, MAETP, MSE, ODP, POF, TETP	No	Yes
Benedetto, 2013	Italy	White	X	X	X	X				ADP, AP, EP, GWP	No	No
Vázquez-Rowe et al. (2012)	Spain	White	X	X	X					AP, EP, GWP, LC, POF, TETP	No	No
Point et al. (2012)	Canada	Red and white		X	X	X	X	X	X	ADP, AP, ED, EP, FAETP, GWP, ODP, POF, TETP	1) organic viticulture; 2) lighter bottle; 3) transport mode	No
Bosco et al. (2011)	Italy	Red and white	X	X	X	X	X	X	X	GHG	No	Yes
Gazulla et al. (2010)	Spain	Red		X	X	X	X	X	X	AP, ED, EP, GWP, ODP, WDP	1) national distribution within Spain; 2) international distribution to the UK	No

Environmental impact indicator(s): Abiotic depletion (ADP); Acidification potential (AP); Agricultural land occupation (ALOP); Climate change (CC); Energy demand (ED); Eutrophication potential (EP); Freshwater ecotoxicity (FAETP); Freshwater eutrophication (FE); Global warming potential (GWP); Greenhouse gas emissions (GHG); Human toxicity (HT); Land competition (LC); Marine ecotoxicity (MAETP); Marine sediment ecotoxicity (MSE); Ozone layer depletion (ODP); Photochemical oxidant formation (POF); Terrestrial acidification (TA); Terrestrial ecotoxicity (TETP); Water depletion (WDP).

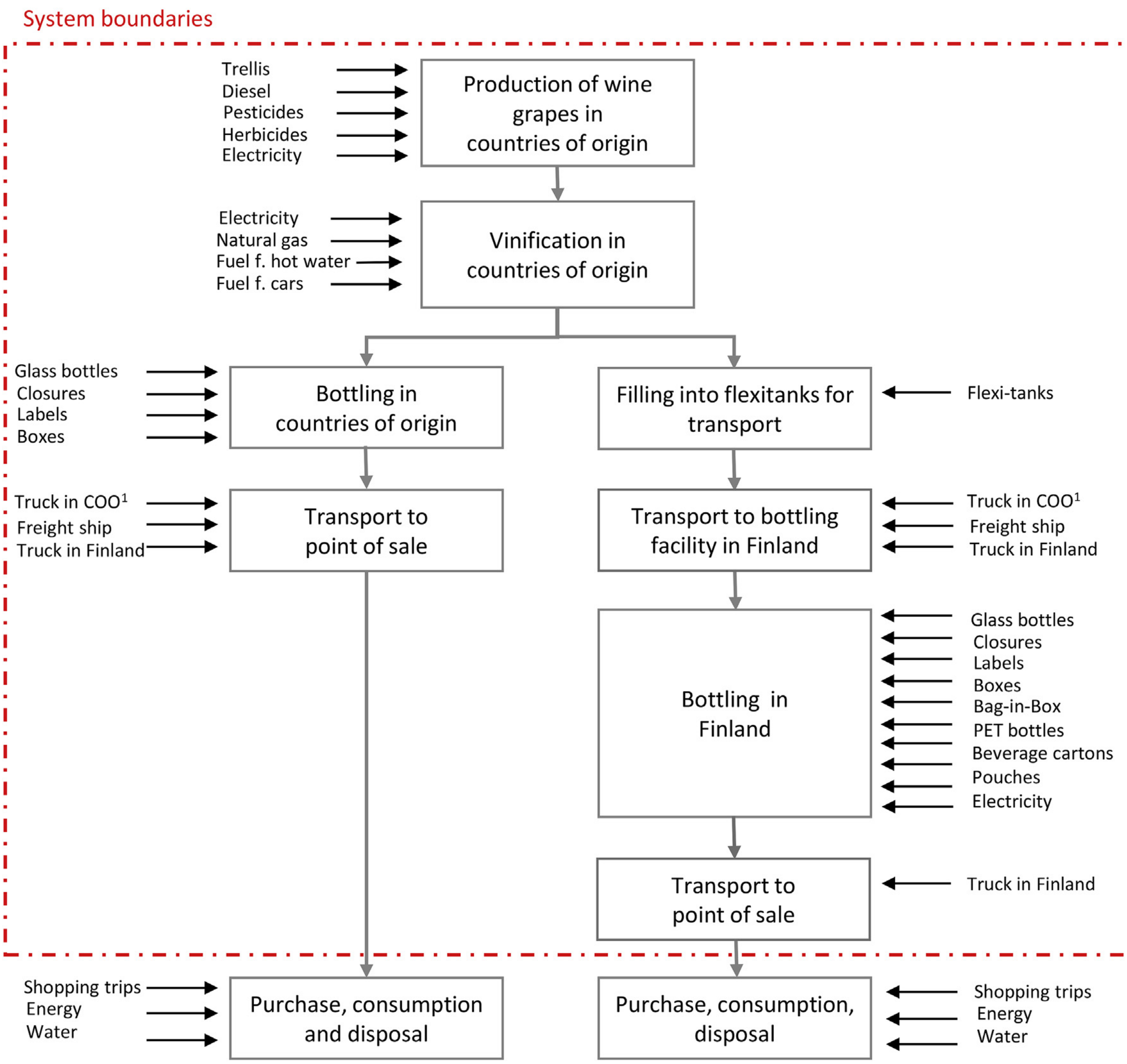

Fig. 1. System boundaries (Source: Own).

Table 2
Imports of bulk and bottled wine from main countries of origin.

Country of origin	Bulk wine (%)	Bottled wine (%)	Bulk & bottled wine (%)
Australia	19.0	9.2	13.4
Chile	23.2	19.1	20.9
France	3.0	10.5	7.2
Germany	6.0	7.7	7.0
Italy	4.0	18.0	11.9
South Africa	12.6	7.7	9.9
Spain	14.6	13.2	13.8
USA (California)	11.0	2.5	6.2
Total	*93.3*	*87.8*	*90.2*

Source: UN Comtrade (2017).

perspective (Blass et al., 2018), we included a sensitivity analysis. Since the high variability of agri-inputs and resulting GHG emissions for wine (Ponstein et al., 2019; Rugani et al., 2013; Vázquez-Rowe et al., 2013) reflect not only differences in agri-inputs, production methods, and yield (Vázquez-Rowe et al., 2013), but also differences in methodology (such as assumptions, emission factors, and system boundaries; Ferrara and De Feo, 2018; Rugani et al., 2013), a sensitivity analysis is important to evaluate the extent to which such input changes affect the output.

First, a particular probability distribution was assigned to each parameter used for estimating the final output (Wei et al., 2015; Meneses et al., 2016). In principle, and depending on the available sample size, the probability distribution of each parameter in the inventory can be derived from statistical methods. However, since we lacked corresponding inventory data, we relied on expert judgment, as previous analyses have done (Lloyd and Ries, 2007; Clavreul et al., 2013). Information concerning emission factors was retrieved from Ecoinvent (Falcone et al., 2016), which led to the modelling of the distributions as lognormal. Also, we followed the IPCC guidelines (2006), as suggested by Neto et al. (2013) and Meneses et al. (2016). For emission factors, the log-normal distribution was used when no information was found, as it is assumed to properly describe the distribution of environmental and ecological parameters (Limpert et al., 2001; Mattila et al., 2012). Regarding the range provided for the production and supply of synthetic N fertilizer (Kool et al., 2012), we assumed a uniform distribution, which implies that all values within the range have the same probability. For the life cycle inventory, we assumed a PERT distribution: this distribution requires minimum, maximum, and the mode, while having some flexibility in the shapes and attributing less weight to extreme values (Muller et al., 2018). All input variables were assumed to be independently distributed of each

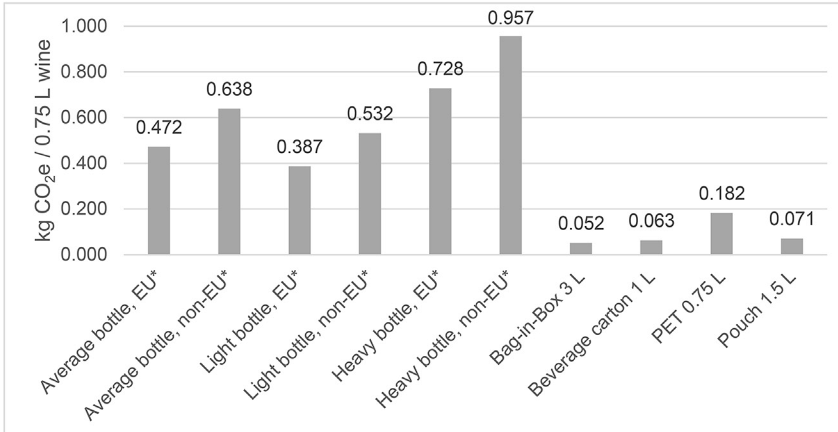

Fig. 2. GHG emissions per packaging type. *Including closure. (Source: Päällysaho (2018) and for glass bottles own calculation based on Päällysaho (2018) and Ecoinvent (2017).

other.

Following similar works on the wine sector (Ponstein et al., 2019; Falcone et al., 2016; Meneses et al., 2016), the second analysis step consisted of employing a Monte Carlo simulation (10,000 iterations), which is the predominant method used in LCA studies (Gregory et al., 2013; Henriksson et al., 2015; Pryshlakivsky and Searcy, 2017; Ross and Cheah, 2017; Song et al., 2018). This method generates random variables selected by the imposed distribution functions for each input parameter. The output is a final distribution for the above-named factors, providing an expected value as well as lower and upper bounds of the 90% confidence interval.

3.3.4. Scenario analysis

A scenario analysis explores the implications of variations in the life cycle process, detecting potential strategies suitable for the enhancement of the sustainability of the Finnish wine supply chain.

Since logistics, packaging, and energy consumption are typically amongst the main emission sources in a supply chain, we developed four scenarios: 1) changes in the bottling location for the whole supply chain (import only bulk wine, bottling in Finland); 2) changes in bottle weight for wine in glass bottles; 3) change in packaging type (increase share in BiB at the cost of glass bottles); 4) increase energy efficiency.

In particular, scenario 1 assumed the establishment of a full bottling process in Finland will impact the production of GHG. Bulk wine has a significantly lower mass per FU compared to bottled wine, thus leading to lower GHG emissions from transport (Amienyo et al., 2014; Harris et al., 2016). For the bottling of bulk wine in Finland we assumed an additional electricity demand of 0.09 kWh per 0.75 L wine (BIER, 2017). Scenario 2 assumed that all bottled wine was filled in 0.380 kg light-weight bottles. Scenario 3 modelled an increase of BiB from 29% to 59% at the cost of glass bottles, aligning Finland with the Swedish market characterized by the highest percentage of wine in BiB in the Nordic countries (Alko, 2017). Scenario 4 proposed an increase of 20% in energy efficiency, following the vision of the EURO 2020 strategy (European Commission, 2010), imposing a reduction of 20% of (i.) the usage of electricity for vineyard irrigation, diesel use (Table 3), (ii.) all energy inputs into vinification (Table 4), and (iii.) the electricity usage of the bottling facility in Finland.

4. Results

4.1. Inventory

4.1.1. Viticulture

A number of studies on GHG emissions from wine production included the production of grapes, and represent this as an inventory (e.g. Amienyo et al., 2014; Benedetto, 2013; Bosco et al., 2011; Gazulla et al., 2010; Neto et al., 2013; Point et al., 2012; Ponstein et al., 2019; Vázquez-Rowe et al., 2012 and 2013). However, the scope of the above-mentioned studies varied due to data availability, regional focus of the analysis, and system boundaries. Due to small sample sizes, none was representative for the specific geographic region (Weidema et al., 2016), except for Navarro et al. (2017b). These authors provided a regional coverage of several wine-growing areas in Spain and the South of France, presenting a detailed inventory based on a sample size of eighteen wineries covering more than 5200 ha of vineyards and a yield of 3.7–11.4 tons of grapes per hectare. Therefore, we based our inventory on their data with regard to organic and synthetic fertilizer, herbicides,

Table 3
Inventory of agri-inputs for wine grape production, per kg grapes.

From the technosphere	Unit	mean	min	max	STD	VAR[a]	Source
Trellis	kg	0.0154	0.0084	0.0283	0.0052	0.3382	[1]
Organic fertilizer	kg N	0.0048	0.0000	0.0102	0.0037	0.7970	[2]
Synthetic N fertilizer	kg N	0.0037	0.0020	0.0060	0.0019	0.5352	[2]
Phosphorous fertilizer	kg P_2O_5	0.0114	0.0036	0.0357	0.0137	1.8832	[2]
Sulphur	kg	0.0070	0.0002	0.0220	0.0080	0.2015	[2]
Unspecified fungicides	kg	0.0020	0.0002	0.0046	0.0010	0.5540	[2]
Herbicides	kg	0.0007	0.0002	0.0017	0.0005	0.8734	[2]
Insecticides	kg	0.0003	0.0001	0.0009	0.0004	2.0253	[2]
Diesel	L	0.0310	0.0120	0.0600	0.0150	0.5263	[2]
Electricity	kWh	0.0450	0.0009	0.0770	0.0280	0.5830	[2]

[a] Own calculation. Source [1] Ponstein et al. (2019); [2] Navarro et al. (2017b).

Table 4
Percentage of vineyards under irrigation.

Country	Irrigation %	Year	Source
Australia	90%	2008	Australian Bureau of Statistics (2009)
Chile	87%	2016	Servicio Agrícola y Ganadero (2017)
France	4%	2010	Eurostat (2016)
Germany	2%	2010	Eurostat (2016)
Italy	27%	2010	Eurostat (2016)
South Africa	85%	2016	Briers-Louw (2016) (using data from the South African Wine Industry Information & Systems)
Spain	23%	2010	Eurostat (2016)
USA (California)	90%	2014	Williams, L. (2014) (newspaper' interview)

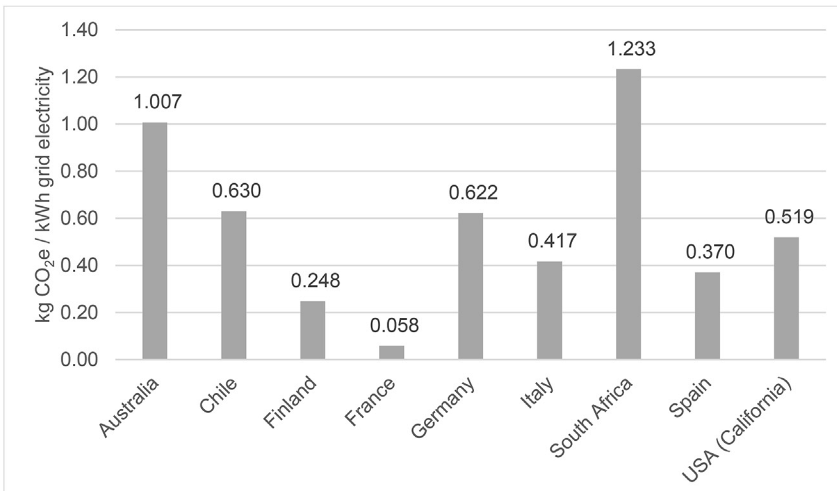

Fig. 3. GHG emissions from grid electricity per country of origin. Own illustration based on Ecoinvent (2017).

pesticides, diesel and electricity for irrigation. Data on trellis was derived from Ponstein et al. (2019), who include wood poles, metal poles, and wires, assuming a life-span of 30 years. Due to a lower deviation of data per kg of grape compared to per hectare of vineyard as unit of analysis (Navarro et al., 2017b), we chose the inventory per kg grape (Table 3). The GHG emissions from the irrigation of vineyards were calculated based on the activity data provided by Navarro et al. (2017b), which was adjusted to the share of irrigated vineyards per COO (Table 4), and the respective emission factor for grid electricity (Ecoinvent, 2017). The GHG emissions from the production of grid electricity vary strongly amongst the COOs (Fig. 3), reflecting the respective shares of fossil fuels, nuclear, and renewable energies, as well as distribution losses (Ecoinvent, 2017). Electricity produced in France caused only 0.058 kg CO_2e per kWh, contrasting South Africa with 1.233 kg CO_2e per kWh.

4.1.2. Vinification

For electricity consumption at the winery level and for the consumption of natural gas to heat water, we relied on Navarro et al. (2017b). The figures for the range of fuel used in winery vehicles exclude transport of wine to the final customers and were adopted from Ponstein et al. (2019) (Table 5).

4.1.3. Bottling/packaging

For wine packaged in the respective COOs, we assumed a range of glass bottle weight retrieved from the literature (minimum weight 0.380 kg: Point et al., 2012; maximum weight of wine bottles in Nordic countries 0.774 kg: Weidema et al., 2016), while the expected value equals the average weight of wine glass bottles used in Finland (0.480 kg, Päällysaho et al., 2018).

For bulk wine bottled in Finland, we assumed the transportation

Table 5
Inventory of inputs for vinification per 0.75 L wine.

Inputs per FU	Unit	Mean	Min	Max	STD	VAR	Source
Electricity	kWh	0.4425	0.1193	1.8075	0.4993	1.1284	[1]
Heat	kWh	0.0458	0.0031	0.0902	0.0040	0.0873	[1]
Fuel to heat water	kWh	0.1478	0.0005	0.3079	0.0107	0.0724	[1]
Fuel for cars	kWh	0.0859	0.0086	0.3758	0.0104	1.5276	[2]

[1] Own calculation. Source [1] Navarro et al. (2017b); [2] Ponstein et al. (2019).

from the respective wine producers to a bottling facility in Finland in flexitanks, which is a bladder that fits 24,000 L of wine made from layers of different plastic types (Weidema et al., 2016). This results in 2.5 g of plastic per FU. For the bottling in Finland, an additional electricity consumption for the bottling process of 0.09 kWh was assumed based on BIER (2017).

GHG emissions from glass bottle production were adopted to the respective regional origin. Hence, GHG emissions from the production of the glass bottle per kg glass were 0.743 kg CO_2e for wine bottled in Germany (BV Glas, 2013; Ponstein et al., 2019), 0.866 kg CO_2e per kg glass for other European countries (RER w/o CH + DE), and 1.079 kg CO_2e per kg glass for non-European countries (Ecoinvent, 2017).

We assumed a weight of 5.5 g per aluminium closure (Päällysaho et al., 2018) and modelled GHG emissions based on the data sets for the production of aluminium ingots and sheet rolling (Ecoinvent, 2017), again taking into account the regional origin (EU or non-EU). For secondary packaging, we assumed 250 g per box that fits 6 bottles of wine, and 0.35 g shrink foil per FU.

Regarding the BiB, we assumed a packaging volume of 3 L with a weight of 0.179 g consisting of cardboard, several extruded types of plastic, and aluminium foil, with an attributed emission factor of 0.052 kg CO_2e based on Päällysaho et al. (2018). Weights and emission factors for beverage cartons, PET bottles, and pouches were equally retrieved from Päällysaho et al. (2018) (Fig. 2).

4.1.4. Transport

The calculation of GHG emissions was based on tonne-kilometres (tkm) and considered the following modes of transport: for wine produced in the EU, the USA, and Australia, we assumed transportation from cellar to port with a truck with EURO 5, while for Chile and South Africa we assumed a truck with EURO 3. For the calculation of the respective transport weights we considered the packaging material for bottled wine and flexitanks for bulk wine. All wines were assumed to be trucked from the cellar to the port in the COO, and per ocean freight to Helsinki. The following ports were selected: Adelaide for Australia; Valparaiso for Chile; Marseille for France; Hamburg for Germany; Genova for Italy; Cape Town for South Africa; Valencia for Spain; San Francisco for the USA. Our selection criteria were based on the relevance for the maritime transport of goods, considering data from Eurostat (2018) and from the American Association of Port Authorities (AAPA) World Port Rankings (American Association of Port Authorities, 2018). For "New World" wines, we assumed that the freight changed ships in Rotterdam before reaching Helsinki. Distances were calculated based on the website www.seadistances.org. Bottled wine was assumed to be trucked to the warehouse and the POS subsequently, while bulk wine was assumed to be trucked to the bottling facility in Finland, and then to the POS. The average distance from cellar to port for each country was calculated using different data sources. For Australia, Chile, South Africa, and USA-California the average distance from cellar to port has been obtained from the reports produced by JF Hillebrand, a beverage logistics company. For France, Germany, Italy, and Spain we made assumptions based on the georeferencing of a sample of wineries within the wine-growing areas. The average transport distance of wine in Finland was provided by Alko (Kokkonen, 2018, personal communication). Total transport distances ranged from 2853 km (Germany) to 22,604 km (Australia) (Table 6).

4.2. GHG emissions

Based on the above analysis, the wine supply chain in Finland caused 88,668 tons of CO_2e in 2017 with a 90% confidence interval of 79,499 to 121,775 tons of CO_2e. The three largest sourcing destinations by volume (Chile, Italy, Spain) contributed 50% of the total import volume and 48% of total GHG emissions (Fig. 4).

Fifty-nine percent of the wine was bottled in glass bottles, causing 72% of GHG emissions owing to the relatively high GHG emissions associated with glass bottles. Meanwhile, 29% of wine was packaged in BiB, which accounted for only 20% of GHG emissions. The remaining 12% were bottled in beverage cartons, PET bottles, and pouches, which contributed almost 9% to the GHG balance (cf. Table 7).

On average, 0.75 L of wine consumed in Finland caused 1.226 kg CO_2e, which is comparable to the range derived by previous authors for other countries (e.g. Ponstein et al., 2019; Scrucca et al., 2018; Rugani et al., 2013). The minimum value of 0.587 kg CO_2e was found for wine produced in France and bottled in BiB in Finland, while the maximum value of 1.932 kg CO_2e was derived for wine produced and filled in glass bottles in Australia. This large range of results can be explained by structural and country-specific differences and packaging material. Concerning the country differences, drivers of GHG emissions are high irrigation intensity, present in Australia, Chile, South Africa, and USA (over 85%, Table 3), high GHG emissions from electricity production, prevalent for Australia, Chile, Germany, and South Africa (Fig. 3), and transport distances, which are particularly large for Australia (22,604 km), Chile (16,459 km), the USA (17,628), and South Africa (13,853 km, Table 5). Consequently, wine from European countries had an advantage with respect to transport distance (over 6500 km), and rather low emission factors for grid electricity (except for Germany).

Despite these important structural differences on country-level, the most influential factor for GHG emissions per FU was the packaging type. As depicted in Fig. 2, an average glass bottle would result in 0.638 kg CO_2e when produced in non-EU countries, and 0.472 kg CO_2e when produced within the EU. Compared to that, BiB would cause only 0.052 kg CO_2e, exceeded by beverage cartons (0.063 kg CO_2e), PET bottles (0.182 kg CO_2e), and pouches (0.071 kg CO_2e).

Consequently, GHG emissions from wine produced in EU countries filled in packaging other than glass bottles are at the lower end of the range of emissions results, while those from wine produced in non-EU-countries, specifically in Australia and South Africa, are at the higher end. Since the type of packaging has such a strong effect on the final result for wine imported from non-EU countries, wine bottled in packaging other than glass can cause less GHG emissions than wine from European countries in glass bottles (Table 8).

With regard to the sensitivity analysis, and as illustrated in Table 8 (row 'confidence interval 90%'), the lower and upper bounds of variability related to the total GHG emissions per FU (for each COO) obtained through the Monte Carlo simulation vary across countries. This suggests that uncertainty associated with a country's GHG emissions is more influential on the final results for some countries compared to others. Australia and South Africa have the wider ranges of variability for each type of packaging. This is caused by the range of electricity usage assumed for the vinification phase in the life cycle inventory, combined with high emission factors for grid electricity in these countries. Furthermore, the range of glass bottle weight in combination with rather high GHG emissions from packaging glass production, which is more pronounced for non-EU countries, are important drivers of variability. Consequently, France and Spain provide the narrower differentials from the upper and bounds for each type of packaging, as electricity-based GHG

Table 6
Transport distances per country of origin.

Country of origin	Truck in country of origin (km)	Ocean freight (km)	Truck in Finland[a] (km)	Total transport distance (km)
Australia	300	22,096	208	22,604
Chile	300	15,951	208	16,459
France	200	5838	208	6246
Germany	500	2145	208	2853
Italy	130	6128	208	6466
South Africa	86	13,558	208	13,853
Spain	250	5276	208	5734
USA (California)	300	17,120	208	17,628

[a] For bulk wine we assume an additional 50 km of transport within Finland (Source: Own).

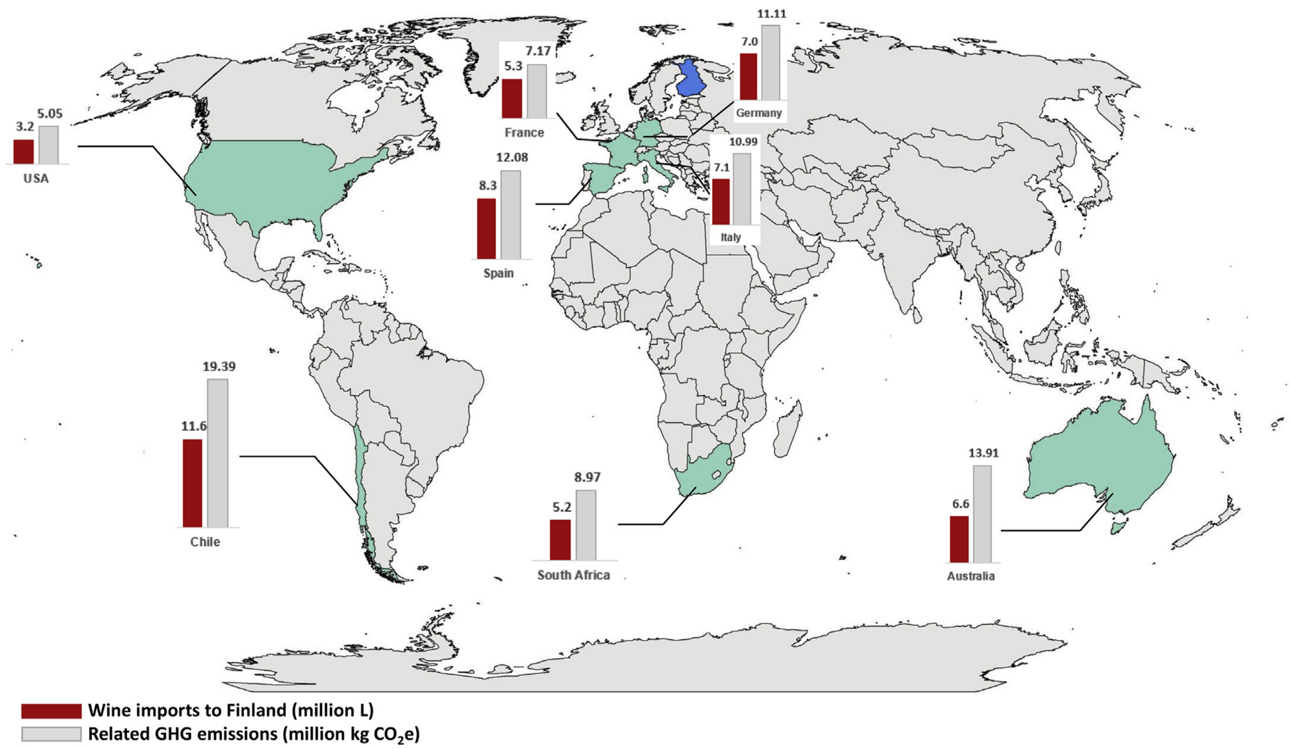

Fig. 4. Finnish wine imports and GHG emissions per country of origin (Source: Own).

Table 7
GHG emissions (in tons of CO_2e) of 90% of the wine supply chain of Finland.

Country of origin (COO)	Bottled at COO	Bottled in Finland					Total	Confidence Interval 90%
	Glass bottle	*Glass bottle*	*BiB*	*Beverage carton*	*PET bottle*	*Pouch*		
Australia	4522	5263	2856	861	399	8	13,909	11,896–20,274
Chile	10,607	–	6260	1530	573	423	19,394	17,101–26,680
France	5742	1095	262	–	43	27	7168	6976–9020
Germany	7824	2022	984	200	79	–	11,110	9704–15,434
Italy	8884	1424	445	63	173	–	10,991	10,151–14,673
South Africa	3122	–	3714	834	440	855	8966	7243–13,840
Spain	6893	3224	1300	345	273	47	12,083	11,237–15,991
USA	932	1853	1592	465	178	28	5048	4592–6875
Total	48,526	14,882	17,415	4299	2158	1389	88,668	79,499–121,775

emissions are rather low and glass bottles are produced within the EU.

4.3. Scenario analysis

The potentials for reducing GHG emission vary on a country-level as a function of the underlying assumptions and circumstances (cf. 4.2, Fig. 5). The increase of BiB at the cost of glass bottles (Scenario 3) offered the greatest potential for reducing GHG emission in the Finnish wine supply chain (−12), followed by the increase in energy efficiency across the full supply chain (Scenario 4, -8%). Here, the impact was greatest for wine from countries with a high emission factor for electricity, particularly for South Africa (−11%) and Australia (−9%). Reducing the bottle weight to 0.380 kg (Scenario 2) throughout the supply chain would avoid 6% of the GHG emissions on a supply-chain level, while the effects are greater for COOs with a high share of bottled wine, such as France, Italy, Germany, and Spain (Table 2). Notably, a change in bottling location (Scenario 1, -2%) would reduce the GHG emissions from wine from overseas destinations, but not from EU countries. Here, the avoided GHGs from transporting the bottle are exceeded by emissions from the additional energy and transport required for bottling in Finland.

5. Discussion

Comparing our results for wine from the various COOs in a glass bottle to the literature (Ponstein et al. (2019); Scrucca et al., 2018; Navarro et al., 2017b; Weidema et al., 2016; Rugani et al., 2013; Point et al., 2012), our findings are within the range identified by earlier studies. Nevertheless, the average value for wine consumed in Finland is at the lower end of the range derived by previous work when considering the same comprehensive system boundaries. This can be attributed to the high share of packaging materials other than glass bottles (41%) typical for the Finnish wine market: We found considerably lower results for wine bottled in alternative packagings, such as beverage cartons, PET bottles, pouches, and BiB. As demonstrated in our scenario analysis, increasing the share of wine bottled in BiB from 29% to 59% at the cost of glass bottles (Scenario 3) has the strongest potential to decrease value-chain based GHGs (−12%). This finding is supported by Amienyo et al. (2013) who analysed the environmental impacts of exchanging 10% of glass bottles for beverage cartons. GHG emissions from wine

Table 8
GHG emissions (kg CO_2e) per 0.75 L wine in various types of packaging.

Country of origin (COO)	Australia	Chile	France	Germany	Italy	South Africa	Spain	USA
Viticulture	0.298	0.282	0.257	0.257	0.261	0.304	0.260	0.278
Vinification	0.526	0.359	0.106	0.356	0.265	0.544	0.245	0.310
Transport from COO to POS in Finland								
Glass, bottled in COO	0.424	0.336	0.170	0.180	0.159	0.257	0.172	0.353
Glass, bottled in Finland	0.309	0.254	0.157	0.161	0.150	0.204	0.157	0.264
BiB[2]	0.294	0.239	0.142	0.146	0.135	0.189	0.141	0.248
Beverage carton	0.293	0.238	0.143	0.145	0.134	0.189	0.141	0.248
PET bottle	0.294	0.239	0.143	0.146	0.135	0.190	0.142	0.249
Pouch	0.293	0.238	0.141	0.147	0.135	0.188	0.140	0.248
Packaging material & bottling								
Glass, bottled in COO	0.684	0.684	0.518	0.459	0.518	0.684	0.518	0.684
Glass, bottled in Finland	0.548	0.548	0.548	0.548	0.548	0.548	0.548	0.548
BiB	0.082	0.082	0.082	0.082	0.082	0.082	0.082	0.082
Beverage carton	0.093	0.093	0.093	0.093	0.093	0.093	0.093	0.093
PET bottle	0.212	0.212	0.212	0.212	0.212	0.212	0.212	0.212
Pouch	0.101	0.101	0.101	0.101	0.101	0.101	0.101	0.101
Total GHG emissions per 0.75L wine								
Glass, bottled in COO	1.932	1.661	1.050	1.252	1.204	1.789	1.195	1.625
90% CI[3]	1.659–2.647	1.473–2.152	1.021–1.326	1.090–1.712	1.113–1.603	1.494–2.501	1.115–1.572	1.461–2.059
Glass, bottled in Finland	1.681	1.443	1.068	1.322	1.224	1.601	1.209	1.400
90% CI	1.469–2.431	1.318–1.970	1.040–1.345	1.187–1.829	1.135–1.624	1.368–2.361	1.130–1.587	1.299–1.869
BiB	1.200	0.962	0.587	0.841	0.743	1.120	0.728	0.919
90% CI	0.971–1.898	0.826–1.430	0.562–0.747	0.698–1.291	0.651–1.075	0.873–1.831	0.646–1.033	0.913–1.326
Beverage carton	1.210	0.972	0.599	0.851	0.754	1.130	0.739	0.930
90% CI	0.981–1.908	0.837–1.440	0.574–0.760	0.709–1.302	0.661–1.086	0.883–1.842	0.657–1.044	0.824–1.338
PET bottle	1.330	1.092	0.717	0.971	0.874	1.250	0.859	1.050
90% CI	1.100–2.029	0.956–1.560	0.692–0.878	0.827–1.421	0.780–1.205	1.003–1.960	0.776–1.164	0.944–1.455
Pouch	1.218	0.980	0.605	0.861	0.763	1.138	0.746	0.937
90% CI	1.006–1.933	0.862–1.466	0.598–0.784	0.733–1.327	0.685–1.112	0.909–1.867	0.682–1.069	0.849–1.362

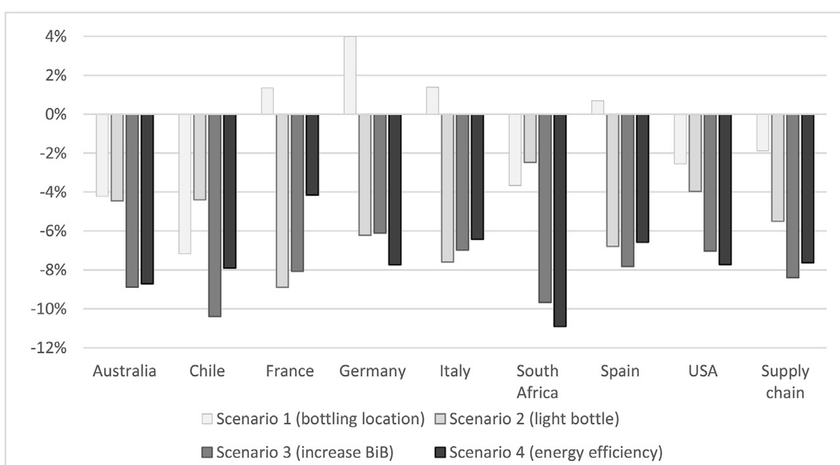

Fig. 5. Scenario analysis – reduction in GHG emissions compared to the baseline.

consumed in Finland can be lower than when consumed in the respective COOs despite long transport distances – given the packaging material is not glass. This is supported by empirical findings of previous work: alternatives to conventional single-use glass bottles are at the core of the decarbonization of the wine supply chain (e.g. Ponstein et al., 2019; Weidema et al., 2016; Amienyo et al., 2013).

Considering that glass bottles have been identified as the single largest source of GHG emissions within the wine supply chain also by previous authors (e.g. Ponstein et al., 2019; Navarro et al., 2017b), the exchange of glass bottles for alternative packaging material with a low emission intensity or the reduction of bottle weight is intuitive. We found that the use of light-weight bottles throughout the supply chain (Scenario 2) would reduce emissions from the supply chain by 6%. With regard to the practical implementation of both scenarios discussed above, there is a need for consumer education, since consumers would otherwise likely be nudged by existing quality proxies like bottle weight, noting previous evidence which suggests that consumers make wine purchase decisions not only based on price and label attributes (Henley et al., 2011; Steiner, 2004), but also on packaging aspects other than labelling (Reynolds et al., 2018; Barber and Almanza, 2007). In particular, studies suggest that quality can be a function of bottle weight and the haptic characteristics of the glass bottle (Szocs et al., 2016; Piqueras-Fiszman and Spence, 2012). Furthermore, price was found to correlate positively with the bottle weight, suggesting that consumers associate higher quality with higher bottle weight (Piqueras-Fiszman and Spence, 2012). This result implies that consumers and industry stakeholders may face a significant trade-off between promoting sustainability gains at the expense of perceived quality. Alternatively, the reuse of wine bottles in Finland should be considered as a low-carbon strategy, following Ponstein et al. (2019).

The scenario analysis shows that possible changes would not impact all countries of origin equally and that it is worthwhile assessing the potential GHG emissions reductions effects individually. For example, while a shift of the bottling location from the COO to Finland (Scenario 1) would imply an overall reduction of merely 2%, the reduction potential is more pronounced for the supply chain of wines from countries with a larger transport distance, such as Australia (−4%), Chile (−7%), and South Africa (−4%), and will be higher with an increased share of bottled wine. For European countries, a shift in the bottling location would actually add GHG emissions to the supply chain, as additional energy usage and transports related to the bottling facility in Finland would not be compensated for by reduced GHG emissions from a lowered transport mass of bulk wine. Our findings are comparable to those by Amienyo et al. (2014), who assessed the environmental impacts associated with Australian wine consumed in the UK. The authors found that 0.33 kg CO_2e occurred from the shipping of bottled wine from Australia to the UK, based on WRAP (2007) who stated a reduction in 0.16 kg CO_2e from shipping bulk as opposed to bottled wine. Amienyo et al. (2014) claimed a reduction of 0.192 kg CO_2e for wine shipped in bulk and bottled in England. This finding compares with our result of 0.424 kg CO_2e from transports of bottled wine from Australia and reduction of 0.123 kg CO_2e when shipped as bulk and bottled in Finland.

Our application of the EU2020 strategy (European Commission, 2010), proposing an energy efficiency gain of 20% (Scenario 4), showed that the reduction in GHG emissions per FU depends on the emission intensity of grid electricity, which varies strongly amongst COOs (Fig. 2). While the mitigation potential would merely be 2% for France, it could reach as much as 11% for wine from South Africa and 9% from Australia. One could conclude that focusing on energy efficiency efforts in those countries with a high emission intensity of grid electricity was preferable. However, limitations of the scope chosen for our LCA should be considered, since we focus on GHG emissions only, not on other environmental indicators, or on economic drivers, such as anticipated cost contributions (Blass and Corbett, 2018).

Our analysis faces a number of limitations that need to be acknowledged. As a function of data availability, we had to assume that the range of inventory data was the same for all factors except for irrigation for all countries of origin. Our main data weakness relates to life cycle inventory data for viticulture, which is not country-specific because such data does not exist. Future research should obtain representative production data for all main wine-growing areas, addressing the high degree of natural variability (Björklund, 2002) in wine production (Rugani et al., 2013; Ponstein et al., 2019). Nevertheless, our results shed light on the structural differences across the eight main wine producing countries under consideration, highlighting major differences in GHG emissions embedded in local input-specifics, e.g. related to electricity production, and irrigation. In light of our focus on one environmental sustainability indicator only, it is clear that future analyses would benefit from a more holistic assessment of the wine supply chain sustainability performance, keeping a triple-bottom-line sustainability assessment in mind, and including other relevant environmental indicators, such as water footprint (Scrucca et al., 2018).

From the perspective of the Finnish Alcohol Monopoly Alko, virtually all GHG emissions arise outside of the organization from upstream activities (production and packaging of wine and transports to Finland) and, to a minor extent, downstream activities (domestic transport and subsequent processes, Fig. 1). Therefore, the decarbonization of the Finnish wine supply chain depends on the joint effort of all members of the supply chain and the wine consumers.

6. Conclusions

This paper provides a detailed analysis of greenhouse gas (GHG) emissions from the global wine supply chain of Finland. As a partial LCA, it conducts not only sensitivity analyses, but also scenario analyses (change in bottling location; reduced bottle weight, increase the share of Bag-in-Box (BiB) at the cost of glass bottles; energy efficiency) for 90% of wines entering Finland, hereby identifying low-carbon strategies potentially of interest to industry and government stakeholders seeking out new strategies for enhancing sustainability in global wine supply chains. Compared to previous work on GHG emissions in the wine supply chain which has not differentiated the analysis by country of origin (COO), our paper provides detailed results by COO. On average, the consumption of 0.75 L wine in Finland caused 1.226 kg CO_2e, which is not only within the range found by comparable studies, but also a plausible value given the high share of Bag-in-Box (BiB) packaging (29%) and packaging types other than glass bottles (12%) currently observed in Finland. Further, we found a large range of results, from 0.587kg CO_2e for French wine in BiB to 1.932 kg CO_2e in the case of Australian wine bottled at source, largely as a function of emission factors for electricity, transport distances, modes of transport, and, most importantly, packaging. Across the supply chain, wine bottled in glass in the respective COOs contributed with the highest amount of GHG emissions. Contrasting, wine imported as bulk wine and bottled in Finland in BiB caused the lowest amount of GHG emissions per Functional Unit.

Our four scenario analyses suggest that increasing the amount of wine in BiB has the greatest environmental sustainability gains with respect to GHG emissions, irrespective of the COOs. Assuming that wine producers would follow the 20% energy efficiency increase recommendation by the European Commission's Energy 2020 strategy, we found the highest GHG reduction potential for wine producers in countries with high electricity-bound GHG emissions (South Africa, Australia, USA). A shift in bottling location from the COOs to Finland, associated with bulk wine exports to Finland, could reduce GHG emissions for wine from Chile, Australia, and South Africa, but not for wine from European countries. Similarly, the reduction in bottle weight would have a higher impact on wine from non-EU countries.

The results from the four scenarios are of particular interest in light of the similarities in industry structure in Finland and the other Nordic countries. Since the Nordic countries are characterized by a retail alcohol monopoly, there may be substantial leverage for making significant sustainability improvements via a single institution that is directly associated with a government.

Future research should obtain representative production data for all main wine-growing areas which consider the high degree of natural variability (Björklund, 2002) in wine production (Rugani et al., 2013; Ponstein et al., 2019).

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References

Ahi, P., Searcy, C., 2013. A comparative literature analysis of definitions for green and sustainable supply chain management. J. Clean. Prod. 52, 329–341.

Akan, M.Ö.A., Dhavale, D.G., Sarkis, J., 2017. Greenhouse gas emissions in the construction industry: an analysis and evaluation of a concrete supply chain. J. Clean. Prod. 167, 1195–1207.

Alko, 2017. Information on the Nordic Alcohol Market. Helsinki.

American Association of Port Authorities, 2018. Port Industry Statistics. Available online at: https://www.aapa-ports.org/unifying/content.aspx?ItemNumber=21048#Statistics. (Accessed 23 June 2018).

Amienyo, D., Camilleri, C., Azapagic, A., 2014. Environmental impacts of consumption of Australian red wine in the UK. J. Clean. Prod. 72, 110–119.

Anderson, K., Nelgen, S., Pinilla, V., 2017. Global Wine Markets, 1860 to 2016: A Statistical Compendium. University of Adelaide Press, Adelaide.

Australian Bureau of Statistics, 2009. Australian Wine and Grape Industry, 2008 (Reissue). Available at: http://www.abs.gov.au/AUSSTATS/abs@.nsf/Previousproducts/1329.0Main%20Features22008%20(Re-Issue)?opendocument&tabname=Summary&prodno=1329.0&issue=2008%20. (Accessed 23 June 2018).

Ayuso, S., Roca, M., Arevalo, J.A., Aravind, D., 2016. What determines principle-based standards implementation? Reporting on Global Compact adoption in Spanish firms. J. Bus. Ethics 133 (3), 553–565.

Barber, N., Almanza, B.A., 2007. Influence of wine packaging on consumers' decision to purchase. J. Foodserv. Bus. Res. 9 (4), 83–98.

Benedetto, G., 2013. The environmental impact of a Sardinian wine by partial life cycle assessment. Wine Econ. Policy 2, 33–41.

BIER, 2017. Beverage Industry Continues to Drive Improvement in Water and Energy Use. Trends and Observations 2016. Beverage Industry Roundtable. Available at: https://www.bieroundtable.com/blank-c1gkm. (Accessed 23 June 2018).

Bjorklund, A.E., 2002. Survey of approaches to improve reliability in lca. Int. J. Life Cycle Assess. 7 (2), 64–72.

Blass, V., Corbett, C.J., 2018. Same supply chain, different models: integrating perspectives from life cycle assessment and supply chain management. J. Ind. Ecol. 22 (1), 18–30.

Bonamente, E., Scrucca, F., Rinaldi, S., Merico, M.C., Asdrubali, F., Lamastra, L., 2016. Environmental impact of an Italian wine bottle: carbon and water footprint assessment. Sci. Total Environ. 560–561, 274–283.

Bosco, S., Di Bene, C., Galli, M., Remorini, D., Massai, R., Bonari, E., 2011. Greenhouse gas emissions in the agricultural phase of wine production in the Maremma rural district in Tuscany, Italy. Ital. J. Agron. 6, 93–100.

Briers-Louw, J., 2016. Dryland Viticulture: an Overview of the South African Situation. Dissertation Submitted to the Cape Wine Academy in Partial Fulfillment of the Requirements for the Diploma of Cape Wine Master. Available at: https://www.icwm.co.za/dissertations/downloadable-dissertations/60-2016-briers-louw-janno-dryland/file. (Accessed 23 June 2018).

BV Glas, 2013. BV Glas Nachhaltigkeitsstudie. Zusammenfassung. January 2013. Düsseldorf, Germany.

CBI, 2016. CBI Product Factsheet: Wine in Nordic Countries. CBI Market Intelligence, The Netherlands.

Clavreul, J., Guyonnet, D., Tonini, D., Christensen, T.H., 2013. Stochastic and epistemic uncertainty propagation in LCA. Int. J. Life Cycle Assess. 18 (7), 1393–1403.

Comtrade, U.N., 2017. UN comtrade database. Available at: https://comtrade.un.org/data/. (Accessed 23 June 2018).

Crawford, R.H., Bontinck, P.A., Stephan, A., Wiedmann, T., Yu, M., 2018. Hybrid life cycle inventory methods - a review. J. Clean. Prod. 172, 1273–1288.

Curran, M.A., 2008. Life-Cycle Assessment. Encyclopedia of Ecology, pp. 2168–2174.

Defra, 2016. Conversion Factors 2016 - Full Set (For Advanced Users). Available at: https://www.gov.uk/government/publications/greenhouse-gas-reporting-conversion-factors-2016. (Accessed 28 May 2018).

Ecoinvent, 2017. Ecoinvent Database V.3.4. Swiss Centre for Life Cycle Inventory. CH. Available at: http://www.ecoinvent.org/database/. (Accessed 1 December 2017).

Euromonitor, 2018. Wine in Finland, p. 32. June 2018. http://www.euromonitor.com/wine-in-finland/report. (Accessed 15 July 2018).

European Commission, 2010. Energy 2020 A Strategy for Competitive, Sustainable and Secure Energy. Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions, Brussels.

European Commission, 2018. Results and Deliverables of the Environmental Footprint Pilot Phase. http://ec.europa.eu/environment/eussd/smgp/PEFCR_OEFSR_en.htm. (Accessed 23 October 2018).

Eurostat, 2016. Agri-environmental Indicator - Irrigation. Available at: http://ec.europa.eu/eurostat/statistics-explained/index.php?title=Agri-environmental_indicator_-_irrigation. (Accessed 23 October 2018).

Eurostat, 2018. Maritime Ports Freight and Passenger Statistics. Luxembourg.

Falcone, G., De Luca, A.I., Stillitano, T., Strano, A., Romeo, G., Gulisano, G., 2016. Assessment of environmental and economic impacts of vine-growing combining life cycle assessment, life cycle costing and multicriterial analysis. Sustainability 8 (8), 793.

Ferrara, C., De Feo, G., 2018. Life cycle assessment application to the wine sector: a critical review. Sustainability 10 (2), 395.

Finkbeiner, M., 2009. Carbon footprinting—opportunities and threats. Int. J. Life Cycle Assess. 14 (2), 91–94.

Finnveden, G., Hauschild, M.Z., Ekvall, T., Guinée, J., Heijungs, R., Hellweg, S., Koehler, A., Pennington, D., Suh, S., 2009. Recent developments in life cycle assessment. J. Environ. Manag. 91 (1), 1–21.

Flores, S.S., 2018. What is sustainability in the wine world? A cross-country analysis of wine sustainability frameworks. J. Clean. Prod. 172, 2301–2312.

Gazulla, C., Raugei, M., Fullana-i-Palmer, P., 2010. Taking a life cycle look at crianza wine production in Spain: where are the bottlenecks? Int. J. Life Cycle Assess. 15 (4), 330–337.

Genovese, A., Acquaye, A.A., Figueroa, A., Koh, S.L., 2017. Sustainable supply chain management and the transition towards a circular economy: evidence and some applications. Omega 66, 344–357.

Gregory, J.R., Montalbo, T.M., Kirchain, R.E., 2013. Analyzing uncertainty in a comparative life cycle assessment of hand drying systems. Int. J. Life Cycle Assess. 18, 1605–1617.

Guinée, J.B., Heijungs, R., 2017. Introduction to life cycle assessment. In: Bouchery, Y., Corbett, C.J., Fransoo, J.C., Tan, T. (Eds.), Sustainable Supply Chains – A Research-Based Textbook on Operations and Strategy. Springer Series in Supply Chain Management No. 4. Springer.

Guinée, J.B., van der Voet, E., 2017. Life Cycle Assessment for resource nexus analysis. In: Bleischwitz, R., Hoff, H., Spataru, C., van der Voet, E., VanDeveer, S.D. (Eds.), Routledge Handbook of the Resource Nexus. Routledge, pp. 67–78.

Harris, I., Rodrigues, V.S., Pettit, S., Beresford, A., Liashko, R., 2016. The impact of alternative routeing and packaging scenarios on carbon and sulphate emissions in international wine distribution. Transport. Res. Transport Environ. 58, 261–279.

Hellweg, S., i Canals, L.M., 2014. Emerging approaches, challenges and opportunities in life cycle assessment. Science 344 (6188), 1109–1113.

Henley, C.D., Fowler, D.C., Yuan, J., Stout, B.L., Goh, B.K., 2011. Label design: impact on millennials' perceptions of wine. Int. J. Wine Bus. Res. 23 (1), 7–20.

Henriksson, P.J., Heijungs, R., Dao, H.M., Phan, L.T., de Snoo, G.R., Guinée, J.B., 2015. Product carbon footprints and their uncertainties in comparative decision contexts. PLoS One 10 (3), e0121221.

IPCC, 2006. Guidelines for national greenhouse gas inventories. Agriculture, forestry and other land use, Intergovernmental Panel of Climate Change 4.

ISO 14040, 2006a. Greenhouse Gases – Part 1: Specification with Guidance at the Organization Level for Quantification and Reporting of Greenhouse Gas Emissions and Removals.

ISO 14040, 2006b. Environmental Management-Life Cycle Assessment-Principles and Framework. International Organization for Standardization, Geneva, Switzerland.

Jradi, S., Chameeva, T.B., Delhomme, B., Jaegler, A., 2018. Tracking carbon footprint in French vineyards: a DEA performance assessment. J. Clean. Prod. 192, 43–54.

Kokkonen, T., 2018. Personal Communication, Telephone Call with Alko's Logistics Development Manager, 12 June 2018.

Kool, A., Marinussen, M., Blonk, H., 2012. LCI Data for the Calculation Tool Feedprint for Greenhouse Gas Emissions of Feed Production and Utilization. GHG Emissions of N, P, and K Fertilizer Production. Blonk Consultants, Gouda, Netherlands.

Kronborg Jensen, J., 2012. Product carbon footprint developments and gaps. Int. J. Phys. Distrib. Logist. Manag. 42 (4), 338–354.

Limpert, E., Stahel, W.A., Abbt, M., 2001. Log-normal distributions across the sciences: keys and clues. Bioscience 51, 341–352.

Linton, J.D., Klassen, R., Jayaraman, V., 2007. Sustainable supply chains: an introduction. J. Oper. Manag. 25 (6), 1075–1082.

Lloyd, S.M., Ries, R., 2007. Characterizing, propagating, and analyzing uncertainty in life-cycle assessment: a survey of quantitative approaches. J. Ind. Ecol. 11 (1), 161–179.

Mariani, A., Pomarici, E., Boatto, V., 2012. The international wine trade: recent trends and critical issues. Wine Economics and Policy 1 (1), 24–40.

Marras, S., Masia, S., Duce, P., Spano, D., Sirca, C., 2015. Carbon footprint assessment on a mature vineyard. Agric. For. Meteorol. 214–215, 350–356.

Matthews, H.S., Hendrickson, C.T., Weber, C.L., 2008. The importance of carbon footprint estimation boundaries. Environ. Sci. Technol. 42 (16), 5839–5842.

Mattila, T., Leskinen, P., Soimakallio, S., Sironen, S., 2012. Uncertainty in environmentally conscious decision making: beer or wine? Int. J. Life Cycle Assess. 17 (6), 696–705.

Meneses, M., Torres, C.M., Castells, F., 2016. Sensitivity analysis in a life cycle assessment of an aged red wine production from Catalonia, Spain. Sci. Total Environ. 562, 571–579.

Merli, R., Preziosi, M., Acampora, A., 2018. Sustainability experiences in the wine sector: towards the development of an international indicators system. J. Clean. Prod. 172, 3791–3805.

Mickwitz, P., Hildén, M., Seppälä, J., Melanen, M., 2011. Sustainability through system transformation: lessons from Finnish efforts. J. Clean. Prod. 19 (16), 1779–1787.

Muller, S., Mutel, C., Lesage, P., Samson, R., 2018. Effects of distribution choice on the modeling of life cycle inventory uncertainty: an assessment on the ecoinvent v2.2 database. J. Ind. Ecol. 22 (2), 300–313.

Navarro, A., Puig, R., Fullana-i-Palmer, P., 2017a. Product vs corporate carbon footprint: some methodological issues. A case study and review on the wine sector. Sci. Total Environ. 581–582, 722–733.

Navarro, A., Puig, R., Kiliç, E., Penavayre, S., Fullana-i-Palmer, P., 2017b. Eco-innovation and benchmarking of carbon footprint data for vineyards and wineries in Spain and France. J. Clean. Prod. 142, 1661–1671.

Neto, B., Dias, A.C., Machado, M., 2013. Life cycle assessment of the supply chain of a Portuguese wine: from viticulture to distribution. Int. J. Life Cycle Assess. 18,

590–602.
Norström, T., Miller, T., Holder, H., Österberg, E., Ramstedt, M., Rossow, I., Stockwell, T., 2010. Potential consequences of replacing a retail alcohol monopoly with a private licence system: results from Sweden. Addiction 105 (12), 2113–2119.
Notarnicola, B., Sala, S., Anton, A., McLaren, S.J., Saouter, E., Sonesson, U., 2017. The role of life cycle assessment in supporting sustainable agri-food systems: a review of the challenges. J. Clean. Prod. 140, 399–409.
Päällysaho, M., Leoino, K., Saario, M., 2018. Update of Wine Packaging LCA - Final Report Alko Oy. Gaia consulting Oy, Helsinki, Finland, p. 46.
Padey, P., Blanc, I., Le Boulch, D., Xiusheng, Z., 2012. A simplified life cycle approach for assessing greenhouse gas emissions of wind electricity. J. Ind. Ecol. 16 (s1).
Panapanaan, V.M., Linnanen, L., Karvonen, M.M., Phan, V.T., 2003. Roadmapping corporate social responsibility in Finnish companies. J. Bus. Ethics 44 (2–3), 133–148.
Pattara, C., Raggi, A., Cichelli, A., 2012. Life cycle assessment and carbon footprint in the wine supply chain. Environ. Manag. 49, 1247–1258.
Pfister, S., Boulay, A.M., Berger, M., Hadjikakou, M., Motoshita, M., Hess, T., Ridoutt, B., Weinzettel, J., Scherer, L., Döll, P., Manzardo, A., Núñez, M., Verones, F., Humbert, S., Buxmann, K., Harding, K., Benini, L., Oki, T., Finkbeiner, M., Henderson, A., 2017. Understanding the LCA and ISO water footprint: a response to Hoekstra (2016) "A critique on the water-scarcity weighted water footprint in LCA". Ecol. Indicat. 72, 352–359.
Piqueras-Fiszman, B., Spence, C., 2012. The weight of the bottle as a possible extrinsic cue with which to estimate the price (and quality) of the wine? Observed correlations. Food Qual. Prefer. 25 (1), 41–45.
Point, E., Tyedmers, P., Naugler, C., 2012. Life cycle environmental impacts of wine production and consumption in Nova Scotia, Canada. J. Clean. Prod. 27, 11–20.
Ponstein, H.J., Meyer-Aurich, A., Prochnow, A., 2018. Greenhouse gas emissions and mitigation options for German wine production. J. Clean. Prod. 212, 800–809. https://doi.org/10.1016/j.jclepro.2018.11.206.
Pryshlakivsky, J., Searcy, C., 2017. Uncertainty analysis focusing on the variance of energy intensity of vehicle materials. J. Clean. Prod. 143, 1165–1182.
Reynolds, D., Rahman, I., Bernard, S., Holbrook, A., 2018. What effect does wine bottle closure type have on perceptions of wine attributes? Int. J. Hosp. Manag. (in press).
Ross, S.A., Cheah, L., 2017. Uncertainty quantification in life cycle assessments: interindividual variability and sensitivity analysis in LCA of air-conditioning systems. J. Ind. Ecol. 21 (5), 1103–1114.
Rossow, I., Storvoll, E.E., 2014. Long-term trends in alcohol policy attitudes in Norway. Drug Alcohol Rev. 33 (3), 220–226.
Rugani, B., Vázquez-Rowe, I., Benedetto, G., Benetto, E., 2013. A comprehensive review of carbon footprint analysis as an extended environmental indicator in the wine sector. J. Clean. Prod. 54, 61–77.
Scrucca, F., Bonamente, E., Rinaldi, S., 2018. Carbon footprint in the wine industry. In: Muthu, S.S. (Ed.), Environmental Carbon Footprints. Industrial Case Studies. Butterworth-Heinemann, pp. 161–196.
Servicio Agrícola y Ganadero, 2017. Catastro Vitícola Nacional. Available at: https://www.odepa.gob.cl/wp-content/uploads/2018/03/catastro-vides-2017.pdf. (Accessed 23 June 2018).
Seuring, S., 2004. Industrial ecology, life cycles, supply chains: differences and interrelations. Bus. Strateg. Environ. 13 (5), 306–319.
Seuring, S., Müller, M., 2008. From a literature review to a conceptual framework for sustainable supply chain management. J. Clean. Prod. 16 (15), 1699–1710.
Song, R., Clemon, L., Telenko, C., 2018. Uncertainty and variability of energy and material use by fused deposition modeling printers in makerspaces. J. Ind. Ecol. (in press).
Statistics Finland, 2016. Income and consumption. Available at: http://www.stat.fi/tup/suoluk/suoluk_tulot_en.html. (Accessed 23 June 2018).
Steenwerth, K.L., Strong, E.B., Greenhut, R.F., Williams, L., Kendall, A., 2015. Life cycle greenhouse gas, energy, and water assessment of wine grape production in California. Int. J. Life Cycle Assess. 20, 1243–1253.
Steiner, B.E., 2004. Australian wines in the British wine market: a hedonic price analysis. Agribusiness: Int. J. 20 (3), 287–307.
Steiner, B.E., Jäger, V., 2018. Anwendung der Gemeinwohl-Bilanz auf Wertschöpfungsketten der Agrar-und Ernährungswirtschaft: einzelbetriebliche Erfahrungen von „Taifun". In: Baumast, A., Pape, J., Weihofen, S., Wellge, S. (Eds.), Betriebliche Nachhaltigkeitsleistung messen und steuern: Grundlagen und Praxisbeispiele. UTB Verlag, Ulmer, Stuttgart.
Steiner, B.E., Peschel, A.O., Grebitus, C., 2017. Multi-product category choices labeled for ecological footprints: exploring psychographics and evolved psychological biases for characterizing latent consumer classes. Ecol. Econ. 140, 251–264.
Szocs, C., Biswas, D., Borges, A., 2016. Cheers to haptic sensations and alcohol consumption: how glassware weight impacts perceived intoxication and positive emotions. Journal of the Association for Consumer Research 1 (4), 569–578.
Szolnoki, G., 2013. A cross-national comparison of sustainability in the wine industry. J. Clean. Prod. 53, 243–251.
Thorlakson, T., de Zegher, J.F., Lambin, E.F., 2018. Companies' contribution to sustainability through global supply chains. In: Proceedings of the National Academy of Sciences, 201716695.
Vázquez-Rowe, I., Villanueva-Rey, P., Iribarren, D., Teresa Moreira, M., Feijoo, G., 2012. Joint life cycle assessment and data envelopment analysis of grape production for vinification in the Rías Baixas appellation (NW Spain). J. Clean. Prod. 27, 92–102.
Vázquez-Rowe, I., Rugani, B., Benetto, E., 2013. Tapping carbon footprint variations in the European wine sector. J. Clean. Prod. 43, 146–155.
Vermeulen, S., Campbell, B., Ingram, J., 2012. Climate change and food systems. Annu. Rev. Environ. Resour. 37, 195–222.
Villanueva-Rey, P., Vázquez-Rowe, I., Moreira, M.T., Feijoo, G., 2014. Comparative life cycle assessment in the wine sector: biodynamic vs. conventional viticulture activities in NW Spain. J Journal of Cleaner Production 65, 330–341.
Wei, W., Larrey-Lassalle, P., Faure, T., Dumoulin, N., Roux, P., Mathias, J.-D., 2015. How to conduct a proper sensitivity analysis in life cycle assessment: taking into account correlations within LCI data and interactions within the LCA calculation model. Environ. Sci. Technol. 49 (1), 377–385.
Weidema, B.P., de Saxcé, M., Muñoz, I., 2016. Environmental Impacts of Alcoholic Beverages as Distributed by the Nordic Alcohol Monopolies 2014.
Wiedmann, T., Minx, J., 2008. A definition of 'carbon footprint'. In: Pertsova, C.C. (Ed.), Ecological Economics Research Trends. Nova Science Publishers, Hauppauge NY, USA, pp. 1–11.
Williams, L., 2014. Dunne on Wine: Water Used to Make Wine Becomes Issue during Drought. Newspaper article available at: https://www.sacbee.com/food-drink/wine/dunne-on-wine/article2622749.html. (Accessed 23 June 2018).
Wine Institute, 2017. World Wine Consumption by Country. Available at: https://www.wineinstitute.org/files/World_Wine_Consumption_by_Country_2015.pdf. (Accessed 23 June 2018).
Wine Intelligence, 2018. Finland Landscapes 2018. Wine Intelligence Report, UK.
WRI and WBCSD, 2013. Corporate Value Chain (scope 3) Accounting and Reporting Standard. Supplement to the GHG Protocol Corporate. Accounting and Reporting Standard. Available at: https://ghgprotocol.org/sites/default/files/standards/Corporate-Value-Chain-Accounting-Reporing-Standard_041613_2.pdf. (Accessed 23 June 2018).