UPDATING THE CARBON FOOTPRINTS FOR SELECTED NEW ZEALAND AGRICULTURAL PRODUCTS

an update for apples, kiwifruit and wine

Sarah McLaren[1], Brent Clothier[2], Andrew Barber[3], Sam McNally[2], Louise Bullen[1], Andre Mazzetto[4], Stewart Ledgard[4]

1 – Massey University

2 – Plant and Food Research

3 – Agrilink

4 - Agresearch

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Executive Summary

Goal, review of emission factors and methodology

This study updates the earlier carbon footprint studies undertaken for selected New Zealand horticultural products (apples, kiwifruit and grapes/wine). A life cycle assessment (LCA) methodology was applied using the most recent international guidelines and standards, and updated emission factors were used where they were available. This document also summarises the changes in the methodology compared with the previous studies (c2009) and provides guidance for future studies.

As part of this update, we reviewed the emission factors applied in the previous study (Section 2). Based on current practices, we provided recommendations on the use of emission factors for the main sources of greenhouse gas (GHG) emissions when performing an LCA. Considering the hotspots for the horticultural products, a broader review was done for the emission factors for shipping the products to overseas markets, generating emission factors for different ship types and sizes.

The nature of perennial crops gives rise to several methodological issues which need to be considered when calculating the GHG emissions of these products. In particular, the different stages over the lifetime of perennial crops and the annual variability of data need to be taken into consideration. In addition, a number of activities are the subject of an ongoing debate regarding the most appropriate modelling approach. This report also provides advice on which methodology should be used regarding GHG metrics, the allocation between co-products, emissions rising from land-use change, the inclusion of infrastructure and the general cut-off rules to be applied in future LCA studies.

Apples

A re-analysis of the carbon footprint of New Zealand's export apples from 2009 has been completed. A consultation was carried out with three large apple companies who, in sum, export about 60% of New Zealand's fresh apples. Apple productivity in 2019, in terms of t/ha, is found to be 1.25 times that in 2008, and the apple pack-out from the packhouse-coolstore is 1.2 times that of 2008. On the debit side, it is considered that on-orchard trafficking would have increased by 25% due to higher density plantings. Through quantitative and qualitative assessments of changed orchard practices and protocols in the packhouse-coolstore, we consider there to have been a reduction in the footprint of these two stages of 0.03 $kgCO_{2eq}$/kg apples.

A quantitative assessment of the relative global weightings of the changes in the destinations of New Zealand's export apples between 2008 and 2019 was carried out. In 2008, some 55% of New Zealand's apples were exported to the United Kingdom and Europe. This has now dropped to 29%. Apple export to East and Southeast Asia has risen from 21% to 56%. So, despite the increase in apple exports, they are travelling shorter distances to their destination. This leads us to conclude that the emissions from the shipping stage of the supply chain have dropped from 0.38 to 0.30 $kgCO_{2eq}$/kg apples, a reduction of 0.08 kCO_{2eq}/kg apples, or a drop of 21%. The total reduction between 2008 and 2019 in the supply-chain stages of the orchard, packhouse, coolstore and shipping amounts to a saving of 0.11 $kgCO_{2eq}$/kg apples. As well, apples are now being exported on larger ships carrying more refrigerated containers. The emission factor for reefer shipping is now likely to be less than the 0.016 $kgCO_{2eq}$/kg apples used by Hume et al. (2009), which is the same value we used here. We did not consider any changes beyond the destination port. With the changed destinations, the post-port emissions are now likely to be quite different from the assessments carried out by Hume et al. (2009) for just the UK.

Thus between 2008 and 2019, we consider that the carbon footprint of New Zealand's export apples has dropped from 0.83 to 0.72 $kgCO_{2eq}$/kg apples, a reduction of 13%.

A range of studies calculating the total GHG emissions using LCA have been carried out internationally. The cradle –to-gate footprint was largely driven by fertiliser, fuel emissions, and pest and disease management. Beyond the orchard gate, electricity for storage and transportation to market were the key contributors to the GHG emissions. Most international studies on apple production reported smaller carbon footprints than New Zealand apples, particularly if using a cradle to consumer system boundary. However, the cradle to farm-gate footprint would be similar to international studies, particularly when considering the shipping to market aspect of New Zealand grown apples accounted for up to 57% of the total GHG emission in 2008, but now much less. A detailed quantitative survey, akin to that of Hume et al. (2009), would seem to be merited to confirm these numbers.

Kiwifruit

Zespri commissioned a carbon footprint assessment in 2019, updating an assessment conducted in 2010. The 2019 study included emissions for fruit produced in 2017 delivered to a German consumer, as well as a revision of the original 2010 emissions to enable a system changes-based-comparisons, i.e., applying the latest methodology to both data sets. The assessment was for Green ('Hayward' variety) kiwifruit produced in NZ (specifically the Bay of Plenty) then distributed to and consumed in Germany. The 2019 values have now been updated in this study based on emission factors, methodology, and system boundary changes, but using the same resource inputs and production values. Based on the system boundary being to the German retailer, excluding the consumer and their associated waste, 2021 orchard emissions (including capital) were found to account for 7.6% of total emissions, 49% lower per kilogram of kiwifruit harvested compared to the revised 2010 study. This change has been primarily driven by a significant 69% increase in production, from 7,400 to 12,500 tray equivalents per hectare. These higher yields were achieved with lower resource use inputs, including lower fuel (-38%), electricity (-61%), nitrogen (-21%), and lime (-14%) inputs.

The original 2010 study, with a functional unit adjustment, found the carbon footprint of kiwifruit was 1.12 $kgCO_{2eq}/kg$ at the retailer. Based on the updated 2021 study, emissions have increased by 11% to 1.24 $kgCO_{2eq}/kg$ at the retailer. However, when compared to the revised 2010 values (recalculated using the same methodologies and emission factors used in 2019), there has been a 24% decrease in the carbon footprint of kiwifruit delivered to a retailer in Germany from 1.64 to 1.24 $kgCO_{2eq}/kg$ at the retailer.

The shipping stage is the most significant life cycle stage, accounting for over two-thirds of all emissions. Outside of shipping, the next most significant stage is at the packhouse, followed by the orchard. If consumer emissions and their associate wastage were included, then this would be the second-largest emission source.

No international studies of kiwifruit which consider the full life cycle, including the orchard stage, packaging and transport, consumption and end of life, have been identified. A number of studies have focused on the orchard stage (cradle to farm gate). Those studies calculated lower orchard GHG emissions than the New Zealand study (although one of the Iranian studies was within the NZ range of results). The hotspot for orchard operations identified in most studies was nitrogen fertiliser with diesel/fuels and electricity also identified as a significant contributor in several studies.

Wine

The original ‘cradle to retailer’ carbon footprint for a bottle of sauvignon blanc from the case study winery was updated based on more recent emission factors and estimated improvements in productivity, vineyard agrichemical inputs and winery electricity intensity.

There is considerable variability in the productivity of New Zealand vineyards from year to year, as is typical for perennial crops. During the 10-year period from 2010 to 2019, the average productivity of New Zealand sauvignon blanc per hectare varied from a low of 8.0 t / ha in 2012 to a high of 15.4 t / ha in 2014.

The updated ‘cradle to retailer’ carbon footprint for a bottle of wine shipped to the USA is 1.090 $kgCO_{2eq}$/bottle and for a bottle shipped to the UK is 1.263 $kgCO_{2eq}$/bottle. This compares to 1.216 $kgCO_{2eq}$/bottle in the original study. The relative proportion of the carbon footprint due to distribution for the bottle shipped to the UK is 28.8% compared to 17.6% for a bottle shipped to the USA. The packaging stage is the most significant life cycle stage regardless of the destination. Overall, the revised ‘cradle to retailer’ carbon footprint for a bottle of sauvignon blanc is comparable to the carbon footprint calculated in the original case study. This is because the more efficient vineyard and winery operations are offset by the higher packaging-related GHG emissions. However, it is important to note that this comparison is based on just one winery, over a single season, and for one variety of wine, assessed over ten years ago.

The carbon footprint associated with the vineyard stage decreased by 28% compared to the original study due to an estimated improvement in productivity. The winery stage carbon footprint decreased by 39% due to a combination of a lower electricity emission factor and estimated improvements in electricity intensity. However, these improvements in the vineyard and winery stages are considered indicative only as these values were estimated from industry average data, which may not be representative of the original case study.

In contrast to apples and kiwifruit, the international literature on GHG emissions of wine is extensive. A wide range of studies have considered the carbon footprint and other life cycle based environmental impacts of wine and winemaking activities. However, comparisons between studies are not always straightforward due to differences in system boundaries, functional units and different methodological approaches between studies. Those studies conclude that the main environmental hotspots are the viticulture stage (due to fuel, fertiliser and pesticides consumption) and packaging production (due to glass bottles). The impact on the results of the choice of system boundary was noted as well as the shortage of original and site-specific inventory data.

The collection of up-to-date data from a representative range of New Zealand wineries over a number of seasons would enable a more robust and representative assessment of the vineyard and winery stages of the life cycle. Similarly, improved data on wine bottles and other packaging systems used in New Zealand and their production would add robustness to the assessment of the packaging and distribution stages.

Recommendations

The updated carbon footprint results for the apple sector is based on limited data compared with the earlier study. The earlier kiwifruit and apple studies included primary data collection from more than 50 orchards and additional primary data collection for packhouses and coolstores. For this study, no such apple data was available, and so estimates of changes in production and other operational practices were provided by industry bodies and matched with updated data on yields. To a lesser extent, kiwifruit orchard resource use inventories were based on a smaller sample size, and not necessarily the same orchards as were used in the earlier study. The exception being the packhouse stage, which drew on more packhouses and is considered more robust than the original study. Nevertheless, the issue remains that the sample size needs to be at the very least reviewed and most likely increased in order to provide more confidence in the result. Likewise with a larger more regularly collected dataset analysis could be done on a same site/orchard basis to detect change over time.

For the wine sector, the earlier study was of a single winery, and the original report does not specify if this represented average practice at the time. For the updated study, the industry averaged data were used to estimate likely changes in GHG emissions since the original study. Based on this very limited dataset, it is not possible to extrapolate to the wider New Zealand wine sector. Nevertheless, the relatively high contribution of the wine bottle to the wine carbon footprint has also been observed in other wine studies, and the New Zealand wine sector's initiatives to identify alternative packaging types with lower carbon footprints are aligned with this result.

For the horticultural sector, it is therefore recommended that primary data are collected in order to have a robust basis upon which to demonstrate the environmental credentials of horticultural products. An online data collection and environmental assessment system would enable growers to benchmark their practices against industry averaged data and prioritise activities to improve their environmental profiles. This approach, termed the Organisational Plus approach, is described in McLaren et al. (2008, Section 5). It can be developed to complement other data collection efforts for SWNZ Scorecards, GlobalGAP, Farm Environment Plans, and so on. Furthermore, over time, it could be extended to additionally quantify different life cycle-based environmental indicators associated with horticultural systems such as water consumption, water quality, and ecotoxicity. This proposed work programme would benefit from a coordinated approach across the primary sector to develop consensus-based methods, share common datasets (energy, fertilisers, plant protection, etc.), and build online data collection systems. This work should build upon the NZ Sustainability Dashboard Project (https://www.agribusinessgroup.com/new-zealand-sustainability-dashboard).

1.Literature Review of Existing Studies (NZ and International)

1.1 Perennial horticultural crops

There is a large number of international studies on the greenhouse gas (GHG) emissions of a range of perennial horticultural crops (Bessou et al., 2013; Cerutti et al., 2014; Cerutti et al., 2011; Parajuli et al., 2019). This section discusses common methodological issues identified in LCA or carbon footprinting studies of perennial crops, focusing on review articles and those addressing specific methodological issues.

The following methodological issues relevant to perennial crops have been identified and are discussed in the following sections:

- Identifying appropriate functional unit(s) and system boundaries.
- Accounting for the different life-cycle stages of perennial crops.
- Accounting for variability between years, orchards, and orchard management systems.
- Field emissions of fertilisers and pesticides.
- Treatment of soil organic carbon and biogenic carbon sequestration.

1.1.2 Choice of Functional Unit and System Boundary

Cerutti et al. (2014) noted that functional units adopted in LCA studies of perennial crops are generally either mass-based (e.g., per kg of fruit), area-based (e.g., per ha of crop) or economic-based (per unit of farmer income). Mass-based approaches are most commonly adopted. Area-based functional units are generally limited to cradle to orchard gate assessments (Ingrao et al., 2015; Page, 2011; Yan et al., 2016). Recent reports are also linking functional units to nutritional content (Parajuli et al., 2019).

A number of researchers have noted that different functional units can lead to very different conclusions (Cerutti et al., 2011, 2013, 2014; Parajuli et al., 2019). Cerutti et al. (2011) noted that mass-based functional units can lead to a preference for a high input/output system and are suitable for evaluating eco-efficiency. In contrast, area-based functional units can lead to a preference for low input/output systems. They advocated for the use of more than one functional unit to provide complementary assessments.

Many LCA studies of perennial crops are undertaken as cradle-to-gate studies (Cerutti et al., 2011; Parajuli et al., 2019). However, the processing, distribution and consumer stages can be a significant proportion of life cycle impacts for perennial crops due to the emissions associated with transportation, food wastage (Bessou et al., 2013; Cerutti et al., 2014), and storage emissions (Parajuli et al., 2019).

1.1.3 Modelling the Full Lifetime of Perennial Crops

The importance of accounting for the full lifetime of a perennial crop, including low or non-productive stages, has been noted by many researchers (Alaphilippe et al., 2016; Bessou et al., 2016; Bessou et al., 2013; Cerutti et al., 2011, 2014). However, this is not reflected in most studies. In a review of over 100 papers on LCA studies of perennial crops, Bessou et al. (2013) noted that perennial systems are usually oversimplified, with 88% of the studies reviewed assessing the impacts of perennial crops in the same way as annual crops (i.e., based on one production year).

The following six different stages over the lifetime of perennial crops have been identified (Cerutti et al., 2010, 2014; Milà i Canals et al., 2006; Parajuli et al., 2019):

1. Nursery stage
2. Planting and field preparation stage
3. Ealy low production stage due to immature plants
4. Full production stage
5. Low production stage due to plant senescence
6. Removal and disposal of plants at the end of life.

The different stages in the lifetime of a perennial crop give rise to several issues when assessing GHG emissions and environmental impacts more broadly.

A number of researchers have noted that the nursery stage of an orchard should be included in an LCA of perennial crops (Bessou et al., 2013; Cerutti et al., 2011, 2014) although this stage is often ignored. In a review of 26 studies assessing the environmental impacts of fruit production systems, Cerutti et al. (2011) noted that only two studies included the nursery stage. However, those studies used general data rather than case-specific data. It has also been observed that the impacts of the nursery may be negligible (Alaphilippe et al., 2016; Bessou et al., 2013; Cerutti et al., 2014), and the lack of available data for the nursery stage is often a limiting factor (Cerutti et al., 2011, 2014). Cerutti et al. (2014) noted that the importance of the nursery stage would depend on the nature of the crop, with a more significant contribution for crops that need special protection during early growth stages (e.g., special growth substrates, greenhouses). Alaphilippe et al. (2016) recommended the inclusion of the nursery stage but noted that, due to the generally low contribution, it is appropriate to use existing LCI databases for this stage.

Several researchers have demonstrated the important role of non-productive or low-production stages of perennial crops. Bessou et al. (2016) investigated how partial modelling of the perennial cycle affects LCA results using two case studies (oil palm fruits and small citrus) and found that the immature plant stages accounted for 7-40% of impacts for oil palm fruits and 7-29% for small citrus depending on the impact category. Similarly, in an LCA study of apple production in France, Alaphilippe et al. (2016) found that unproductive stages of the life cycle (nursey, orchard creation, establishment and destruction) accounted for 9-28% of impacts.

Data availability is identified as a limiting factor for all stages of perennial crops (Bessou et al., 2013, 2016; Cerutti et al., 2011). Some have noted that the low production stage due to plant senescence and end of life stage are not relevant in many cases as crops are replanted at the end of the full production stage (Cerutti et al., 2011, 2014).

Bessou et al. (2013) defined three potential approaches to modelling all six stages of the perennial crop cycle:

1. Spatial: the perennial system is captured from an area representative of the whole cycle
2. Modular: each stage of the perennial cycle is modelled separately
3. Chronological: describes the whole cycle following the historical course of crop development.

The objectives of the study and data availability are identified as key determinants of which modelling approach should be used (Bessou et al., 2013, 2016). Bessou et al. (2016) noted the importance of being transparent with the selected approach, the representativeness of data used and discussing the implications of modelling choices.

1.1.4 Variability of Perennial Crops

Several researchers have noted the natural variability of perennial crops due to climate, extreme weather events, diseases, pests and biennial bearing (Alaphilippe et al., 2016; Cerutti et al., 2014; McLaren et al., 2010; Parajuli et al., 2019). For example, McLaren et al. (2010) observed a 31% difference between the lowest and highest yields for New Zealand green kiwifruit recorded over six years.

Variability also occurs due to differences in management practices between different management systems (organic, integrated, conventional) and between individual orchards, as well as differences in data collection and quality (Bessou et al., 2013; Parajuli et al., 2019).

The collection of data over at least four seasons has been proposed as a possible solution to account for inter-annual variability with a preference for an even number of years to account for biennial bearing (Alaphilippe et al., 2016; Cerutti et al., 2014). Similarly, averaged data from a large number of orchards or industry-wide data can be used to account for variability between orchards (Cerutti et al., 2014; McLaren et al., 2010). However, Bessou et al. (2013) noted that there could be a wide range of agricultural practices. They recommended that minimum and maximum impact values should be reported as the use of averaged results from aggregated data can potentially obscure the specific impacts of diverse operations.

1.1.5 Field Emissions

Several researchers have found that field emissions associated with fertiliser and pesticide use are an important contributor, which should be included when calculating the GHG emissions from perennial systems. They have also noted the need for improved methods for estimating these emissions (Alaphilippe et al., 2016; Bessou et al., 2016; Cerutti et al., 2011).

Peter et al. (2016) used carbon footprint case studies, including two Italian peach orchards, to compare the impact of calculating field emissions using Tier 1 (based on IPPC default emission factors), Tier 2 (based on national emission factors) and Tier 3 (based on model simulation or in-situ measurement) methods. They found that using a Tier 2 approach reduced the carbon footprint in the peach orchards by 7% compared to a Tier 1 approach and recommended using higher tier methods to improve the accuracy of carbon footprint results.

Alaphilippe et al. (2016) found that fertilisers (production, application and field emissions) contributed 30-83% to the climate change impacts in two cradle-to-gate apple orchard case studies. The contribution varied depending on the intensity of the production system and the stage of the orchard (orchard establishment or productive stage).

Bessou et al. (2013) noted that operational approaches to field emissions, such as the IPPC Tier 1 method, only provide a rough estimation of field emissions due to the high level of variability in space and time of these emissions. They recommended further developing process-based models for perennial systems and collecting field data to calibrate such models to specific crops and local conditions.

1.1.6 Soil Organic Carbon and Potential Biogenic Carbon Sequestration

The role and inclusion within carbon footprinting studies of soil organic carbon and biogenic carbon emissions or sequestration associated with land-use change, crop management, and above ground storage within perennial systems is a subject of ongoing debate in the literature (Bessou et al., 2013; Bessou et al., 2020; McLaren et al., 2010; Peter et al., 2016; Plassmann et al., 2017).

Bessou et al. (2013) found that the vast majority (73%) of perennial LCA studies reviewed did not account for land-use changes. Bessou et al. (2020) observed that there is no scientific consensus on the best methods to assess the impact of land use and land-use change within LCA studies. They used case studies, including some perennial crops, to investigate the influence of different methods for accounting for land-use change and land management change on climate change impacts and found that accounting for land-use change and land management change greatly influenced the results. They recommended using the IPPC Tier 1 approach using default values for soil organic carbon as a minimum and using case-specific data where available.

Peter et al. (2016) found that using a Tier 1 approach to account for soil organic carbon in two Italian peach orchards underestimated emissions and resulted in a decrease of 67% in the overall carbon footprint compared to Tier 3 methods. They concluded that a Tier 1 approach could not adequately represent changes in soil organic carbon due to crop management changes.

Less attention has been given to the potential carbon sequestration role of perennial trees and shrubs. Still, Plassmann et al. (2017) explored some of the issues involved, including the difference between stocks and sequestration, the non-permanence of the removals, the tendency to focus only on the product being analysed, allocating removals between co-products from the same system, and considering the entire system over its full lifetime. They concluded there were three circumstances where it may be meaningful to give credit for biogenic carbon sequestration of woody perennials:

1. If management or land-use changes increase total carbon stocks
2. If biogenic carbon replaces fossil fuels
3. If biogenic carbon is stored for more than 100 years.

1.2 Kiwifruit

1.2.1 NZ studies

A previous study on the full life cycle GHG emission of a tray of New Zealand kiwifruit consumed in Europe determined a value of 5.32 $kgCO_{2eq}$/tray of green kiwifruit or 1.6 $kgCO_{2eq}$/kg of fruit (Mithraratne et al., 2010). The shipping stage was the most significant life cycle stage with the distribution of GHG emissions across the different life cycle stages as follows:

- Orchard operation – 13%
- Packhouse – 10%
- New Zealand port operation – 1%
- Shipping – 44%
- Packaging at Zeebrugge – 3%
- Retail operations – 6%
- Consumer and end of life disposal – 23%

Mithraratne et al. (2010) also concluded that the gold kiwifruit cultivar 'Hort16A' had slightly lower GHG emissions than green and green organic kiwifruit due to higher yields per hectare and shorter storage time at the coolstore. It was also identified that further research was required in the areas of nitrogen fertiliser and compost production and use, coolstore energy use and refrigerant leakage, refrigerated shipping, and changes in soil carbon on orchards. A related report (McLaren et al., 2008, p.18) noted that the modelled refrigerant loss at the coolstore was probably a "worst-case" scenario and that the average was more likely to be about one-third of the value used in the scoping study (equivalent to a reduction of about 1.3% in the total kiwifruit CF). However, it also noted that refrigerant losses can be highly variable between different coolstores and through time at any one coolstore. This variability is due to infrequent catastrophic events such as forklift damage. The same report noted the potential variability in the results due to the modelling assumptions for composting.

In a report undertaken in conjunction with the above study, Deurer et al. (2008) identified nine potential short-term opportunities to reduce kiwifruit's life cycle GHG emissions. The highest short-term reduction potential existed in the areas of shipping and establishing a new orchard. It was estimated that the combined effect of these nine opportunities could reduce life cycle GHG emissions by 20%. It was also estimated that medium-term reduction opportunities could reduce life cycle GHG emissions by a further 15%, with the most significant reduction potential associated with shipping.

In conjunction with the above two reports, an 'organisational plus' GHG accounting system for New Zealand kiwifruit was proposed based on existing organisational and product-oriented GHG accounting systems and standards. This combined approach was proposed to provide a mechanism to measure the GHG emissions for New Zealand kiwifruit and drive continuous improvement in the kiwifruit supply chain (McLaren et al., 2008).

Müller et al. (2015) investigated the carbon footprint of the orchard stage of New Zealand kiwifruit production based on a study of 40 orchards covering both green and gold kiwifruit cultivars and both integrated and organic production systems. The carbon footprint varied between 0.13-0.22 $kgCO_{2eq}$/kg of kiwifruit or 3,948-5,379 $kgCO_{2eq}$/ha and was considered comparable for different cultivars and production methods. This is similar to the earlier Mithraratne et al. (2010) study, which calculated 0.17 $kgCO_{2eq}$/kg green kiwifruit at the orchard gate. The hotspots in the orchard stage were emissions associated with the production, packaging, storage and transportation of fertilisers, emissions associated with the on-farm nitrogen cycle (i.e., field emissions from organic and inorganic fertilisers and crop residues), and fuel use. Potential reduction opportunities were identified as changes in the background system, such as different transport routes and product choices, and improving the nitrogen-use efficiency of the fertilisers.

Robertson et al. (2014) investigated the carbon footprint associated with different packaging and transport options for kiwifruit exported to Europe and Japan. They found that the carbon footprint of packaging and transport varied between 0.33 and 0.67 $kgCO_{2eq}$/kg of fruit delivered to a store depending upon the packaging type and destination. Potential options for reducing the carbon footprint associated with packaging and transport were identified as: increased efficiency in shipping, use of trains for land transport, reductions in the addition of structural packaging, minimising supply chains with a higher carbon footprint, identifying alternative materials, replacing the use of polystyrene clamshells with alternative materials or plastic bags, and maximising recycling rates along all stages of the supply chain.

Robertson et al. (2014) noted that a comparison with the results of their study with Mithraratne et al. (2010) is problematic due to differences in system boundaries and functional units. However, once adjusted to the same functional unit, the results from Mithraratne et al. (2010) showed a carbon footprint for packaging (international tray) of 0.012 $kgCO_{2eq}$/kg of fruit compared to 0.093 $kgCO_{2eq}$/kg of fruit in Robertson et al. (2014). The difference between these two studies was attributed to differences in raw data regarding how much each packaging type was used and differences in cardboard emission factors.

A number of studies have investigated the role of carbon sequestration in determining the carbon footprint of kiwifruit. Page (2011) used a net balance approach that considered sources and sinks of carbon to calculate the carbon sequestration potential of a modelled organic kiwifruit production system and estimated net carbon sequestration of 2.4 tCO_{2eq}/ha/year. However, it was noted that further research was required to improve and validate the approach used.

Holmes et al. (2014) investigated carbon sequestration in kiwifruit soils by measuring soil carbon stocks up to 1m deep at more than 60 kiwifruit orchards throughout New Zealand. They found that the soil of kiwifruit orchards stored an average of 174.9±3 t C/ha to a depth of 1m. They also noted that 51% of the soil carbon stored down to 1m depth was stored in the top 0.3m, but about 72% was captured by increasing the sampling depth to 0.5m. Gentile et al. (2016) investigated soil carbon stocks to 9m depth in a kiwifruit orchard and nearby pasture soil. They calculated an increase of 190 t C/ha in the kiwifruit orchard after 30 years compared to the pasture. This work was developed further by Gentile et al. (2021) by sampling at 19 paired kiwifruit and pasture sites in the Waikato and western Bay of Plenty regions using a different measurement method. They found that cumulative soil organic carbon and nitrogen stocks to 2m depth were not different between the two land uses and that labile water-extractable soil organic carbon pools were lower under kiwifruit in the topsoil (0-0.1m). They concluded that further work was required to understand the contribution of perennial horticulture crops to subsoil organic carbon.

More recently, Zespri (2019) have reported a carbon footprint for kiwifruit produced in 2017 and consumed globally of 2.0 kgCO_{2eq}/kg of fruit compared to 2.6 kgCO_{2eq}/kg in the 2009 season (global weighted average). The biggest contributors were the shipping stage (43%) and the consumer stage (31%).

1.2.2 International studies

International studies on the GHG emissions of kiwifruit include a study by Mistura et al. (2009), which estimated the transport GHG emissions of kiwifruit produced in New Zealand, Australia, Chile or Argentina and consumed in Italy based on an Italian national food survey. They concluded that significant reductions in GHG emissions could be obtained if consumers limited their kiwifruit consumption to locally grown kiwifruit. However, this study was not based on a life cycle approach and only considered transport emissions in isolation.

No international studies of kiwifruit which consider the full life cycle, including the orchard stage, packaging and transport, consumption and end of life, have been identified. A number of studies have focused on the orchard stage (cradle to farm gate) of kiwifruit produced in Iran (Mostashari-Rad et al., 2019; Nabavi-Pelesaraei et al., 2016; Nikkhah et al., 2015; Nikkhah et al., 2016), Italy (Baudino et al., 2017; Lardo et al., 2018) and Greece (Mazis et al., 2021). Baudino et al. (2017) also considered the warehouse stage of fruit processing and packaging. A summary of the carbon footprint calculated by each of these studies is provided in Table 1.

The functional unit varied between the different studies, with some utilising an area-based approach (per ha) and others a production-based functional unit (per kg or ton of kiwifruit produced). Nikkhah et al. (2015) also calculated the GHG emissions of kiwifruit per farmer (3,163 kgCO_{2eq}/farmer/year).

All the studies using an area-based functional unit were based in Northern Iran with a range of 1,310-4,519 kgCO_{2eq}/ha. Electricity use accounted for 47% of the GHG emissions calculated by Nikkhah et al. (2015). Nabavi-Pelesaraei et al. (2016) noted that the significantly lower level of GHG emissions obtained in their study was primarily due to a much lower use of electricity.

Studies using a production-based functional unit calculated GHG emissions between -0.04 and 0.25 kgCO_{2eq}/kg of kiwifruit for cradle to farm gate studies. Baudino et al. (2017) was the only study that included warehouse operations resulting in a higher level of GHG emissions (0.33-0.48 kgCO_{2eq}/kg of kiwifruit).

The role of soil organic carbon sequestration was only considered by Lardo et al. (2018), who compared the carbon footprint of ‘conventional’ and ‘sustainable’ orchard management practices on a number of different fruit orchards. They concluded that orchards could act as either a carbon ‘sink’ or ‘source’ depending on the orchard management practices adopted.

In summary, the limited international studies (for Iran and Greece) calculated lower orchard GHG emissions than the New Zealand study (although one of the Iranian studies was within the NZ range of results). The hotspot for orchard operations identified in most studies was nitrogen fertiliser with diesel/fuels and electricity also identified as a significant contributor in several studies. No other studies were found that calculated the GHG emissions along the whole supply chain.

Table 1: Summary of Recent International Carbon Footprint Studies of Kiwifruit

Author	**Geographical Location**	**System Boundary**	**Carbon Footprint per ha** ($kgCO_{2eq}$/ha)	**Carbon Footprint per kg** ($kgCO_{2eq}$/kg of kiwifruit)	**Identified Hotspots**
Nikkhah et al. (2015)	Northern Iran	Cradle to farm gate	4,519	-	Electricity Farmyard manure Diesel fuel
Nikkhah et al. (2016)	Northern Iran	Cradle to farm gate	-	0.152	Nitrogen fertiliser
Nabavi-Pelesaraei et al. (2016)	Northern Iran	Cradle to farm gate	1,310	-	Nitrogen fertiliser Diesel fuel Machinery Electricity
Mostashari-Rad et al. (2019)	Northern Iran	Cradle to farm gate	2,373	-	Nitrogen fertiliser
Baudino et al. (2017)	Italy	Cradle to farm gate	-	0.14 (*A.arguta*–baby kiwi) 0.11 (*A.deliciosa*)	Nitrogen fertiliser
		Cradle to warehouse gate	-	0.33 (*A.arguta*–baby kiwi) 0.48 (*A.deliciosa*)	Storage electricity
Lardo et al. (2018)	Southern Italy	Cradle to farm gate (incl soil C stock & CO_2 sequestration)	-	-0.04 (sustainable) 0.161 (conventional)	
Mazis et al. (2021)	Western Greece	Cradle to farm gate	-	0.25 (range: 0.16-0.37)	Nitrogen fertiliser Fuels

1.3 Apples

1.3.1 NZ studies

A study on the cradle to orchard-gate emissions using an LCA approach was performed for New Zealand Braeburn apple systems based on data from the 2000-2001 season (Milà i Canals et al., 2006). The carbon footprint ranged between 40-100 $kgCO_{2eq}/t$ (0.41.0 $kgCO_{2eq}/kg$) for grade 1 and 2 Braeburn apples. Apples grown in Central Otago had larger footprints than the apples grown in Hawkes Bay due to the increased need for practices such as frost protection. The largest on-orchard drivers of the GHG emission were associated with energy (i.e., fuel) for machinery and fertiliser use.

A later study using a full LCA approach for Braeburn and royal gala apples consumed in the U.K. (cradle to consumer) had total GHG emissions of 1.2 $kgCO_{2eq}/kg$ (Hume et al., 2009). This study used the total yield of apples. A comparison between integrated and organic apple production systems displayed no difference in carbon footprint between these two systems. The shipping stage of the life cycle was the largest contributor to the total GHG emissions, and the various life cycle stages for both Braeburn and Royal Gala varieties accounted for the following:

- Orchard operations - 4-7%
- Packhouse operations - 5-6%
- Port - 1-2%
- Shipping - 45-57%
- Repackaging in the UK - 7-9%
- Retailer - 9-12%
- Consumer - 8-28%

Perie (2015) explored the methodological requirements to account for any change in carbon dynamics in apple orchards and whether these could be integrated into LCA. The results of this study demonstrated that, through appropriate sampling, the woody biomass carbon could provide a reduction in the cradle to New Zealand port carbon footprint of up to 4.6%. Furthermore, there was no observed change in soil carbon stocks across a 12-year chronosequence of orchards. Methodological considerations raised were that soil carbon stock estimates should be site-specific and communicate the uncertainty in the measurements based on the inherent variability observed within soils.

1.3.2 International studies

A range of studies calculating the total GHG emissions using LCA have been carried out internationally. Within these studies, many have compared different production systems (i.e., organic vs conventional; semi-extensive vs intensive) using a range of system boundaries (e.g., cradle to gate through to cradle to grave). The comparisons made between management systems generally demonstrate small differences in the GHG emissions, driven mainly by differences in fertiliser use and pest management. For example, intensive apple production had a larger cradle to gate carbon footprint than a semi-extensive system, largely driven by greater fertiliser and pesticide use associated with the intensive management (Alaphilippe et al., 2016). However, when the system boundary is extended, organic and conventionally grown apples had similar footprints (Longo et al., 2017) as the post-gate contributions to the GHG emission were greater than any differences associated with on-farm management.

A negative correlation between carbon footprint and yield demonstrated an increase in efficiency within a production system (Yan et al., 2016). This effect was presumably driven by increased efficiency of nutrient use, with the largest contribution to the GHG emissions of this study associated with fertiliser use and the energy requirement for pumping irrigation.

A multi-year approach to LCA was conducted for apple and peach production in Spain (Vinyes et al., 2017). This study used production data over a ten-year period to account for variability within each crop cycle. The total GHG emissions for apples in this study was 0.3 $kgCO_{2eq}/kg$. The agricultural and retail stages were responsible for 37% and 39% of the total emissions, while the consumption and disposal stages contributed to 23% and 1% of the total emissions, respectively.

Keyes et al. (2015) assessed the LCA for apples produced in Nova Scotia, Canada, under conventional and organic apple production. The difference between the cradle to gate footprint of the two production systems was small, with the organic production (0.073 $kgCO_{2eq}/kg$) having a greater footprint than the conventional system (0.064 $kgCO_{2eq}/kg$). This study defined the LCA for different aspects of the supply chain (e.g., cradle to gate and cradle to market) to highlight the opportunities to improve within each key supply chain inputs. Results demonstrated that, up to the farm gate, the major contributors were fuel use for machinery, fertiliser emissions, and pest and disease management. When the system boundary was extended to market locations, the dominant contributors were electricity for storage associated with the coal-based power generation in Nova Scotia. The use of shipping is more favourable for long-distance transportation compared to truck delivery.

Frankowska et al. (2019) summarised the LCA of different fruits consumed in the UK. The study accounted for the fact that only 7% of fruit was produced locally, and most fruit was imported. For this study, apples had a total GHG emission of 1.4 $kgCO_{2eq}/kg$ and the largest contributors to this footprint related to the transport to market and the production within the orchard gate. It should be noted that New Zealand apples were included in this study which may have influenced the footprint reported.

In summary, the cradle to gate footprint was largely driven by fertiliser, fuel emissions, and pest and disease management. Beyond the orchard gate, electricity for storage and transportation to market were the key contributors to the GHG emissions. Most international studies on apple production reported smaller carbon footprints than New Zealand apples, particularly if using a cradle to consumer system boundary. However, the cradle to farm-gate footprint would be similar to international studies, particularly when considering the shipping to market aspect of New Zealand grown apples accounted for up to 57% of the total GHG emission. The most similar comparison to the New Zealand cradle to consumer study by Hume et al. (2009) compares fruit consumed in the UK (Frankowska et al., 2019). The study of Frankowska et al. (2019) was for all 17 types of fruit, including apples, but they did not present the break-down of their assessments by country of origin, only by fruit, In their study, the LCA of apples was found to 1.4 $kgCO_{2eq}/kg$, which is very similar to the 1.2 $kgCO_{2eq}/kg$ calculated for New Zealand apples. This agreement is probably not surprising as 40% of the UK's fresh apples come from South Africa, New Zealand, Chile and Argentina, and so have a footprint burden due to refrigerated sea-transport. Another reason for the similarity would be that another

40% of the apples consumed in the UK come from France where, despite being close-by, orchard productivity is just 60% that of New Zealand's (FAOSTAT). Nonetheless, of all the 17 fruit under consideration by Frankowska et al. (2019), apples had the fifth-equal lowest greenhouse gas LCA footprint. Furthermore, when summed across all 19 LCA environmental-impact categories, apples ranked fifth best of the 17th fruit.

Table 2: Summary of Recent International Carbon Footprint Studies for Apple Production

Author	**Geographical Location**	**System Boundary**	**Carbon Footprint per ha** ($kgCO_{2eq}$/ha)	**Carbon Footprint per kg** ($kgCO_{2eq}$/kg of apples)	**Yield (t/ha)**	**Identified Hotspots**
Milà i Canals et al. (2006)	New Zealand	Cradle to orchard gate	-	0.4-1.0	79-121 (including Grade 1-3)	Fertiliser Energy Fuel
Hume et al. (2009)	New Zealand	Cradle to UK consumer	-	1.2	44-58 36-43 (exported)	Shipping
Alaphilippe et al. (2016)	France	Cradle to gate	2.7-4.3	0.075-0.09	37.8-55.4	Fertiliser Agrichemicals
Yan et al. (2016)	China	Cradle to the point of sale	8.2	0.24	37	Fertiliser
Vinyes et al. (2017)	Spain	Cradle to grave	-	0.3	48.81	Agricultural stage (fertiliser and agrichemical) Retail stage (diesel and refrigeration)
Longo et al. (2017)	Italy	Cradle to Consumer	-	0.59-0.61	50-70	Post-harvest and transportation
Keyes et al. (2015)	Canada	Cradle to gate	-	0.064-0.073	12-24	Fertiliser, fuel use and pest/disease management
		Cradle to consumer	-	0.28-0.85		Electricity for storage Transportation
Bartzas et al. (2017)	Greece	Cradle to gate	-	0.09	32	Fertiliser Field management
Frankowska et al. (2019)	Consumed in the UK	Cradle to UK consumer	-	1.4	-	Farm management Transport

1.4 Grapes/wine

1.4.1 NZ studies

There are a limited number of studies quantifying the life cycle GHG emissions of New Zealand wines. Greenhalgh et al. (2008) published guidelines for GHG products accounting for the New Zealand wine industry, including a case study of a 750ml bottle of Sauvignon Blanc based on data provided by Wairau River Wines. The total GHG emissions were 1.24 $kgCO_{2eq}$ for a 750ml bottle of wine spread over vineyard (20.4%), winery (18.9%), bottling (0.1%), packaging (36.7%), distribution (21.5%), retail (0.2%), and consumer and end of life (2.2%) stages. The glass bottle accounted for 77% of packaging emissions and was also the single largest contributor across the life cycle, accounting for 28% of total GHG emissions. The next largest contributors were international shipping (22%) and electricity use in the winery (18%).

A report by Deurer et al. (2009) used the results of Greenhalgh et al (2008) and other case studies to investigate potential improvement opportunities to reduce the carbon footprint of New Zealand wine. Deurer et al. (2009) identified the following top 10 potential short to medium terms improvement options (percentage values are estimated reduction in total GHG emissions over the life cycle):

- Change to a Tetra-pack packaging system (42%)
- Replace the standard wine bottle with a PET bottle (27%)
- Replace the standard wine bottle with a lighter weight bottle (26%)
- Replace the standard wine bottle with a recycled glass bottle (19%)
- Use grass or crops to cover the vineyard floor (16-40%)
- Ship wine in bulk and package closer to market (9%)
- Reduce fuel use in vineyard operations (4%)
- Increase fuel use efficiency during shipping operations (2-4%)
- Adopt energy-efficient practices in the winery to reduce electricity usage (2-3%)
- Substitute diesel with an 80/20 mixture of diesel/biodiesel (1%).

Consumer acceptance of alternative packaging options was identified as a potential barrier to implementation for improvement options involving plastic bottles or Tetra-packs.

Deurer et al. (2009) identified a longer-term improvement option to reorientate export markets from Europe to closer destinations to reduce shipping distances.

Barry (2011) undertook a full LCA study of a 750ml bottle of sauvignon blanc produced from two case study wineries located in Hawkes Bay and Blenheim. The study considered the life cycle stages of grape growing, wine making, packaging and distribution to a consumer in the UK. A total life cycle GWP of 1.4 $kgCO_{2eq}$/750ml bottle of wine was calculated with the biggest contributors to GWP from glass bottle production (41-42%) and shipping to the UK (26-27%). A sensitivity assessment, which included a number of potential improvement options, indicated that significant reductions in the life cycle GWP could be achieved by the use of a PET bottle (40% reduction), location of the consumer in China rather than the UK (13% reduction), and bulk shipping of wine (12% reduction).

Clothier et al. (2017) used the carbon footprint results per bottle of wine from Greenhalgh et al. (2008) to calculate a carbon footprint per ha of vineyard area. This study was only concerned with the vineyard stage of the life cycle and calculated total vineyard GHG emissions of 3,000 $kgCO_{2eq}$/ha, of which 6% or 170 $kgCO_{2eq}$/ha were estimated to be due to biological GHG emissions (i.e., emissions of N_2O and CO_2 from nitrogen fertiliser, leaf-fall and prunings).

Yealands Wine Group Limited produce the only New Zealand wines that are currently certified as carbonZero products. The carbon footprint of certified Yealands wines on a cradle-to-grave basis is reported as 1.11 $kgCO_{2eq}$/0.75L bottle when bottled in New Zealand using a glass bottle. This compares to 0.62 $kgCO_{2eq}$/0.75L bottle when bottled in New Zealand using a PET bottle and 0.97 $kgCO_{2eq}$/0.75L bottle when the wine is shipped in bulk and bottled in the UK using a glass bottle. Crossings bag in box and PET bottle products have a carbon footprint of 0.85 and 1.31 $kgCO_{2eq}$/0.75L, respectively.

1.4.2 International studies

In contrast to New Zealand specific studies, the international literature on GHG emissions of wine is extensive. A wide range of studies have considered the carbon footprint and other life cycle based environmental impacts of wine and wine making activities. However, comparisons between studies are not always straightforward due to differences in system boundaries, functional units and different methodological approaches between studies (Ferrara et al., 2018; Jourdaine et al., 2020; Rugani et al., 2013).

There have been three main reviews of LCA and carbon footprint studies of wine in recent years. Rugani et al. (2013) reviewed 35 studies to explore the use of carbon footprint as an environmental indicator in the wine sector, including an examination of methodological and conceptual issues. Ferrara et al. (2018) reviewed 34 LCA studies of the wine sector. They concluded that the main environmental hotspots are the viticulture stage (due to fuel, fertiliser and pesticides consumption) and packaging production (due to glass bottles). The impact on the results of the choice of system boundary was noted as well as the shortage of original and site-specific inventory data.

More recently, Jourdaine et al. (2020) reviewed ten LCA papers representing 17 different wines using a harmonization process to reduce methodological discrepancies. In all but one study, the main contributors to climate change were grape production and the packaging stages. The manufacture of synthetic fertilisers, pesticides and the emission of diesel used in agricultural machineries were the main contributors to the grape production stage. Bottle production and disposal were the highest contributors to the packaging stage, with the glass bottle accounting for 24-70% of the total climate change impact.

A recent study by D'Ammaro et al. (2021) was focused on Italian wines but also included a summary of carbon footprint values reported in the literature between 2009 and 2019 from a range of wine-producing countries.

A summary of the carbon footprint values found by the above studies is presented in Table 3. Table 4 shows the carbon footprint results obtained from LCA studies of wine conducted in recent years which were not considered in the review articles listed in Table . This summary of recent studies is limited to studies utilising a functional unit of a 0.75L bottle of wine. The results from different studies may not be directly comparable as the system boundaries are not consistent with most studies covering ‘cradle to winery gate’ or ‘cradle to retailer’ life cycle stages.

Table 3: Summary of Carbon Footprint Values Reported in Review Studies of the Wine Sector (System Boundaries Vary between Studies)

Review Study	**Carbon Footprint ($kgCO_{2eq}$ / 750ml bottle)**	**Number of Studies Estimate Based on**	**Location of Studies**
Rugani et al. (2013)	2.17 ±1.34	29	Europe, North America, Chile, Australia, NZ, Scandinavia, South Africa,
Ferrara et al. (2018)	1.2 (average value)	28	Australia, Italy, Canada, Spain, NZ, Cyprus, Portugal, USA, Luxembourg
Jourdaine et al. (2020)	0.88 - 6.20	10	Australia, Italy, Spain, Portugal, Canada
D'Ammaro et al. (2021)	0.90 - 1.88	14	Italy, France, Spain, Germany, Australia, Portugal, Canada

Table 4: Summary of Recent International Carbon Footprint Studies of Wine and Grapes

Author	Geographical Location	System Boundary	Carbon Footprint ($kgCO_{2eq}$ / 0.75L bottle)	Identified Hotspots
D'Ammaro et al. (2021)	Italy (16 wineries)	Cradle to Grave	1.39 ±0.25	Glass bottle (29%) Electricity used in winery (14%), Distribution of product (13%) Heat used in winery (9%) Fossil fuel used in the vineyard (8%)
Harb et al. (2021)	Lebanon (consumed in UK)	Cradle to Grave	0.98[1]	Grape production (45%) Packaging (33%)
Chiriaco et al. (2019)	Central Italy	Cradle to Winery Gate	0.79 ± 0.14	Vinification and bottling (39%) Wine packaging (46%)
Laca et al. (2021)	Northern Spain Northern Spain (mountain winery)	Cradle to Winery Gate	2.35 (single winery)	Vineyard (incineration) Production of glass bottles
Martins et al. (2018)	Portugal	Gate to Gate (winemaking & bottling)	1.10 (branded wine) 1.23 (Terroir wine)	Packaging materials (58% branded wine; 71% Terroir wine)
Martins et al. (2019)	Portuguese (terroir wine)	Cradle to Retailer	3.51 (3-year average)	Transportation to retailer
Ponstein et al. (2019a)	Wine consumed in Finland from different countries of origin (COO)	Cradle to Retailer [2]	1.659 - 2.647 (COO: Australia) 1.473 - 2.152 (COO: Chile) 1.021 - 1.326 (COO: France) 1.090 - 1.712 (COO: Germany) 1.113 - 1.603 (COO: Italy) 1.494 - 2.501 (COO: South Africa) 1.115 - 1.572 (COO: Spain) 1.461 - 2.059 (COO: USA)	Packaging
Ponstein et al. (2019b)	Germany	Cradle to Winery Gate	0.829 (90% CI: 0.753-1.069)	Packaging (57%) (glass bottle=47%) Viticulture (19%) Electricity use in winery (10%)
Trombly et al. (2019)	New York State, USA	Cradle to Winery Gate	0.617 - 1.03 (range of 3 wineries)	Bottle production Viticulture
Litskas et al. (2020b)	Cyprus	Cradle to Retailer	1.31 (single winery)	Electricity use in winery (46%) Packaging material (18%) Vineyard (16%)

[1] A mass based allocation for the winemaking process allocated only 56% of the impacts to wine with the remainder allocated to the waste products of grape pomace, marc and lees. This approach will have resulted in an underestimate of the winemaking process as well as the total impacts.
[2] Results provided for wine in glass bottle, bottled in country of origin

Overall, a wide range of carbon footprint results from international wine studies can be seen in Tables 3 and 4 (0.62 to 6.2 $kgCO_{2eq}$ per 0.75L bottle of wine). This range reflects both differences in methodological choices between the studies (e.g., system boundaries, excluded processes) and a range of factors that can affect the GHG emissions throughout the life cycle of a bottle of wine.

The lowest carbon footprint reported of 0.62 $kgCO_{2eq}$/0.75L bottle was obtained from a large winery that produces more than 500,000 bottles of wine per year in the Finger Lakes Region in New York State, USA. Several features of the area, and this particular winery, were noted by the authors as contributing to a low carbon footprint, including economies of scale due to the size of the winery, the use of solar panels for electricity production, low winter temperatures reducing energy requirements for cooling during winemaking, and high rainfall reducing the need for energy for irrigation. In addition, this study excluded the distribution and consumption life cycle stages (Trombly et al., 2019).

In contrast, the highest reported carbon footprint of 6.2 $kgCO_{2eq}$ per 0.75L bottle (including the distribution life cycle stage) was for an Italian wine (Bosco et al., 2011) and was considered to be due to several factors. Among these are the diesel used for the transportation of packaging materials, the production and emission of fertilisers, and the production of glass bottles (Jourdaine et al., 2020). This wine also appears to be an outlier compared to other studies listed in Tables 3 and 4. In the review by Jourdaine et al. (2020), the other wines had a carbon footprint of <3 $kgCO_{2eq}$/0.75L bottle.

A number of factors have been noted by researchers as being important contributors to the variation in carbon footprint between different wines. These include interannual variability (Beauchet et al., 2019; Ferrara et al., 2018; Renaud-Gentié et al., 2020; Vendrame et al., 2019), productivity differences due to climatic factors (Laca et al., 2021) and grape variety (Ferrara et al., 2018; Rugani et al., 2013), differences in production methods such as conventional versus organic wine growing (Ferrara et al., 2018; Litskas et al., 2020a; Rugani et al., 2013), and the use of traditional grape growing or wine making methods (Laca et al., 2021). Where the distribution life cycle stage is included, packaging weight, bottling location, and transport distances have also been noted as important factors (Litskas et al., 2020b; Martins et al., 2019; Ponstein et al., 2019a).

Recently, Sun et al. (2020) also considered the impact of wine tourism and cellar door sales based on an Australian case study. They estimated that, for domestic sales, the carbon emissions per bottle of wine purchased from a visit to a winery are about 17-23 times larger than the emissions associated with purchasing from a traditional retailer. For a UK visitor to Australia for wine tourism, the carbon emissions of a bottle of wine are estimated to increase 54 to 88-fold.

The glass bottle, or the packaging stage, is commonly identified as a hotspot in many studies (D'Ammaro et al., 2021; Ferrara et al., 2018; Jourdaine et al., 2020; Laca et al., 2021; Navarro et al., 2017). Studies specifically considering wine packaging are discussed in Section 1.4.3 (below). Other hotspots identified include electricity use in the winery (D'Ammaro et al., 2021; Litskas et al., 2020b), the distribution stage (when included) (D'Ammaro et al., 2021; Martins et al., 2019) and the viticulture or vineyard stage (Ferrara et al., 2018; Jourdaine et al., 2020; Laca et al., 2021; Litskas et al., 2020a; Navarro et al., 2017).

1.4.3 Studies on Wine Packaging

Due to the predominance of the packaging stage in many LCA studies of wine, alternative packaging options such as reduced bottle weight (Litskas et al., 2020b; Martins et al., 2019; Navarro et al., 2017; Ponstein et al., 2019b), bottle recycling (Ferrara et al., 2020a; Landi et al., 2019; Ponstein et al., 2019b), and use of alternative packaging materials (Ferrara et al., 2018, 2020a; Ponstein et al., 2019a) have been identified as potential mitigation options to reduce GHG emissions associated with wine production.

Ferrara et al. (2020a) undertook a comparative LCA of alternative wine packaging systems based on an Italian case study and found that, for most indicators, including GWP, the best packaging alternative was a bag-in-box container followed closely by an aseptic container. Other packaging systems considered were a traditional glass bottle, a refillable glass bottle and a multi-layer PET bottle. Other researchers have also noted the potential for significant reductions in the life cycle GHG emissions of wine by using alternative packaging systems such as bag in a box (Ponstein et al., 2019a), aseptic containers (Ferrara et al., 2018) and PET bottles (Martins et al., 2018).

Ferrara et al. (2020a) found that the performance of a refillable glass bottle became comparable to that of aseptic cartons and bag-in-box system when transport distances of less than 100km were required and therefore were only a convenient alternative when considering a local market. Similarly, Landi et al. (2019) compared end-of-life alternatives of recycling and reusing glass bottles based on an Italian wine consortium and found that reuse of glass bottles performed better than recycled bottles. However, the authors noted that similar results would only be expected in consortiums with similar attributes (i.e., producers located within 50-100km, few wine types, few bottle types, local distribution). In assessing GHG emission mitigation options for wine production in Germany, Ponstein et al. (2019b) found that greater reductions in GHG emissions would be achieved by reuse of the glass bottle compared to reduced bottle weight, increase in packaging volume and increased use of renewable energies.

Opposition to alternative wine packaging systems by consumers and other stakeholders is identified as a barrier to adopting alternative packaging. Ferrara et al. (2020b) undertook a survey of Italian consumers attitudes to alternative wind packaging and found that 91% of respondents were not willing to try alternative packaging options. However, approximately 62% were willing to re-evaluate the purchase of wine in alternative packaging when provided with further information regarding the quality of wine and sustainability of alternative packaging.

2. Review of Common Emission Factors

The Environmental Product Declaration (EPD) Product Category Rules (PCR) for Fruit & Nuts (EPD, 2019) provides advice on system boundaries, cut-off rules, allocation rules and data quality requirements. It provides guidance on recommended databases for generic data, and these are noted below:

PROCESS	GEOGRAPHICAL SCOPE	DATABASE
Agricultural product	Worldwide	Agrifootprint, Ecoinvent
Plastic	Europe	Industry data 2.0, Plastics Europe
Aluminium	Europe	EAA (European Aluminium Association)
Steel	Worldwide	Worldsteel
National residual electricity mix	Europe	Association of Issuing Bodies
Transport	Worldwide	Ecoinvent

It also provides guidance on nutrient emissions to air and water and these are:

	Emission	Paragraph	Source
Emission in air	Ammonia	4.10.2.1	EMEP/CORINAIR, 2013[5], IPCC, 2006[6]
	N_2O, NO – direct emission	4.10.2.2	Bouwman et al., 2002[7]
	N_2O – indirect emission	4.10.2.3	IPCC, 2006
Emission in water	Nitrates	4.10.2.4	IPCC, 2006
	Phosphorus	4.10.2.5	Prahsun, 2006[8]

Given these references are mostly those that were used in the 2009 study on New Zealand apples (Hume et al., 2009) and revised 2010 study on kiwifruit (Mithraratne et al., 2010), there is likely no change in emission factors unless these have changed in databases such as ecoinvent. The following tables are populated with the emission factors from Hume et al. (2009). Note that all ecoinvent datasets are "market for" datasets and global (GLO) unless specified otherwise. Due to copyrights agreement, the ecoinvent values can't be disclosed. Therefore, we marked on the Tables below when the ecoinvent values are used, and practitioners with access to the database will be able to access it in the database.

2.1 Fertiliser, lime, compost

2.1.1 Production

Activity or product	Emission factors in earlier studies				Recent recommendations				
	Dairy	Kiwifruit	Apples	Grapes/wine	Dairy	EPD Int PCR fruit and nuts	ecoinvent/World Food LCA Database	MfE GHG Guidance	Other
Synthetic Nitrogen $kgCO_{2eq}/kgN$		3.38	3.38	3			4.627 for NZ or GLO value used inorganic nitrogen fertiliser $kgCO_{2eq}/kgN$		
Urea, as N, at regional store $kgCO_{2eq}/kg$	3.1				2.12 (or 3.71 including CO_2 from soil after application)				
Soil amendments & compost $kgCO_{2eq}/t$		20 (2010) 50 (2019)	50				GLO value used		-525 to 1,140. 375 default[1]
Greenwaste $kgCO_{2eq}/t$		-	690						
Straw $kgCO_{2eq}/t$		-	181						
Phosphorus $kgCO_{2eq}/kgP$		0.96	0.96	0.9			2.45 for NZ or GLO value used for inorganic phosphorus fertiliser $kgCO_{2eq}/kg\ P_20_5$		
Potassium $kgCO_{2eq}/kgK$		0.64	0.64				1.042 for NZ or GLO value used for inorganic potassium fertiliser $kgCO_2eq/kg\ K_2O$		

Activity or product	Emission factors in earlier studies				Recent recommendations				
	Dairy	**Kiwifruit**	**Apples**	**Grapes/wine**	**Dairy**	**EPD Int PCR fruit and nuts**	**ecoinvent/World Food LCA Database**	**MfE GHG Guidance**	**Other**
Sulphur $kgCO_{2eq}/kgS$		0.32	0.32						
Magnesium $kgCO_{2eq}/kgMg$		0.32	0.32						
Lime $kgCO_{2eq}/kg$	0.412 (includes field reaction emissions)	0.43	0.43 (includes field reaction emissions)		0.004 (excludes field reaction emissions)		GLO value used for $CaCO_3$ as lime		
Limestone $kgCO_{2eq}/kg$		0.036							
Potassium chloride $kgCO_{2eq}/kg$				0.39 (0.74 $kgCO_{2eq}/kg$ ai)	0.430				
Single Superphosphate $kgCO_{2eq}/kg$					0.148				
Dolomite $kgCO_{2eq}/kg$		0.036		0.0116					

[1] The compost EF was updated to 50 in 2018 accounting for processing and transport to orchard only, which has a default figure of 34 and ranges between 29 and 95. For the 2018 kiwifruit study 50 was used with the recommendation to conduct a sensitivity analysis (Barber and Rothmann, 2008). This also aligned with the apple study.

Nitrogen

The original horticultural studies all used a nitrogen emission factor of 3.38 kgCO_{2eq}/kgN. This was based on Wells (2001) who determined an average figure of 3.0 kgCO_2/kgN. Added to this was an allowance for methane and nitrous oxide emissions. This does not include field emissions, which are captured separately. The use of a generic nitrogen figure reflected the limited information on fertiliser emissions, the level of uncertainty, and that the data collected from the growers was often only in units of nitrogen applied without a description of the fertiliser. This was done to overcome the issue of unrecognised fertiliser names or special mixes that could not be related back to the quantity of nutrients.

Fertiliser emission factors were identified as an area for further work. Subsequently, AgResearch prepared a report on the carbon footprint of NZ fertilisers in 2010. This was updated in 2019 using specific data on fertiliser sources and production from the two main fertiliser companies in NZ (Ledgard and Falconer 2019). On a per kilogram of nitrogen basis, urea has an emission factor of 2.12 kgCO_{2eq}/kgN, capturing production, local transport, and shipping. If emissions of CO_2 from soil after application are included, then this becomes 3.71 kgCO_{2eq}/kgN. CAN emissions to an NZ port are 4.2 kgCO_{2eq}/kgN. The study also included the complex fertilisers ammonium sulphate (AS) and diammonium phosphate (DAP). The EF for AS is 0.77 kgCO_{2eq}/kg which equates to 3.75 kgCO_{2eq}/kgN. DAP is 1.28 kgCO_{2eq}/kg which includes an important P component to it. Where the type of N fertiliser is not known, it is recommended that the factor of 3.75 kgCO_{2eq}/kgN (based on AS) is used.

Recommendations

Nitrogen can be a significant component of emissions to the farm gate. Based on the original 2008 studies, nitrogen represented almost 50% of fertiliser emissions for kiwifruit and 42% for pipfruit. The form of nitrogen has a significant impact on its production and transport emissions. Where possible, the emission factors should be based on the specific fertiliser using the known emission factors from Ledgard and Falconer (2019). Alternatively, the fertilisers should be split into urea and non-urea forms, which would also align with the methodology used to calculate field emissions (see Field Emissions Section). Therefore, the best compromise where the type of fertiliser is unknown or where the emission factor has not been determined is to use the emission factor of 3.75 kgCO_{2eq}/kgN.

Phosphorus

The original studies used the Wells (2001) emission factor of 0.9 kgCO_{2eq}/kgP, and then adjusted it to 0.96 kgCO_{2eq}/kgP to account for methane and nitrous oxide in the fuels that are used in the production and transport of phosphorus. It was found to vary between 0.6 to 1.2 kgCO_{2eq}/kgP. Ledgard and Falconer (2019) found that single superphosphate had emissions of 0.148 kgCO_{2eq}/kg, equating to 1.64 kgCO_{2eq}/kgP. The corresponding factor for triple superphosphate is 0.38 kgCO_{2eq}/kg fertiliser, which equates to 1.85 kgCO_{2eq}/kgP.

Recommendations

Phosphorus is a small component of emissions to the farm gate. Based on the original 2008 studies, phosphorus represented 3.1% of fertiliser emissions for kiwifruit. Where single superphosphate is used, it is recommended to use the NZ-specific factor of 1.64 kgCO_{2eq}/kgP, while for other P fertilisers 1.85 kgCO_{2eq}/kgP should be used.

Potassium

Potassium was the highest input fertiliser in the kiwifruit study at 215 kgK/ha and represented 15% of fertiliser emissions, although considerably less in pipfruit at 19 kgK/ha. The original studies used the Wells (2001) emission factor of 0.6 kgCO_{2eq}/kgK, and then adjusted to 0.64 kgCO_{2eq}/kgK to account for methane and nitrous oxide in the fuels that are used in the production and transport of potassium. Ledgard and Falconer (2019) found potassium chloride (KCl) has an emission factor of 0.45 kgCO_{2eq}/kg fertiliser, which (adjusted using latest characterisation factors) equates to 0.86 kgCO_{2eq}/kgK.

Recommendations

An emission factor of 0.86 kgCO_{2eq}/kgK should be used reflecting the latest figure found by Ledgard and Falconer (2019).

Sulphur

Sulphur is a reasonably high input fertiliser in kiwifruit at 90 kgS/ha, although down to around 26 kgS/ha in pipfruit, but due to its low emission factor is a small component of fertiliser emissions at less than 5% in both kiwifruit and pipfruit. All studies used the Wells (2001) emission factor of 0.3 kgCO_{2eq}/kgS, and then adjusted to 0.32 kgCO_{2eq}/kgS. While sulphur is a component of ammonium sulphate in the Ledgard and Falconer (2019) study it isn’t separated out into the component parts.

Recommendations

Where S is being applied and is not accounted for within other applied fertilisers (e.g., single superphosphate or Ammonium Sulphate), an emission factor of 0.32 kgCO_{2eq}/kgS should be used, reflecting the figure used in the previous studies and in the absence of a more recent value.

Lime and Dolomite

The previous studies used the emission factors determined by Wells (2001). This was an aggregated figure for the carbon dioxide emissions associated with mining and crushing at 0.036 kgCO_2/kg limestone or dolomite, plus the carbon dioxide that is evolved on reaction with the soil. These figures were previously reported together, although more accurately, they should be separated into processing and then the main release of carbon dioxide reported in field emissions. Dolomite also includes 11.5% magnesium, which needs to be accounted for. Wells (2001) reported magnesium to have the same primary energy content as sulphur. Therefore, it should have an emission factor of 0.32 kgCO_{2eq}/kg Mg applied. Other values reported are 0.004 kgCO_2/kg limestone in the earlier Ledgard and Falconer report and 0.07 kgCO_2/kg limestone as a recommended default for Oceania by Brentrup et al. (2018).

Recommendations

Lime fertiliser emissions to be reported at 0.038 kgCO_{2eq}/kg lime (0.036 kgCO_2/kg lime adjusted for methane and nitrous oxide). Dolomite to have a fertiliser emission factor of 0.038 kgCO_{2eq}/kg dolomite for crushing and processing plus 0.037 kgCO_{2eq}/kg dolomite for its *magnesium component (0.32 kgCO_{2eq}/kg Mg at 11.5%). This is total fertiliser emissions of 0.075 kgCO_{2eq}/kg dolomite. The carbon dioxide emissions upon reaction with the soil are to be reported in field emissions.*

2.1.2 Field Emissions

Activity or product	Emission factors in earlier studies				Recent recommendations				
	Dairy	**Kiwifruit**	**Apples**	**Grapes/wine**	**Dairy**	**EPD Int PCR fruit and nuts**	**ecoinvent/World Food LCA Database**	**MfE GHG Guidance (NZ GHG Inventory)**	**Other**
NH_3 from N fertiliser application							GLO value used per kgN applied (depending upon type of fertiliser and pH of soil) (WFLCDB, Table 8)	0.055 if urea is coated, 0.1 if urea is uncoated	
N_2O from N fertiliser application		4.904 $kgCO_{2eq}/kgN$ (2010) 4.458 $kgCO_{2eq}/kgN$ (2019)		3.1 $kgCO_{2eq}/kgN$			1% of all N applied in fertiliser plus in crop residues plus mineralized from soil organic matter plus lost as ammonia, plus lost as NO_x plus lost as nitrate	5.40 (Non-urea N), 3.48 (Urea N w/o UI), 3.27 (Urea N w/UI)	
Limestone $kgCO_{2eq}/kg$		0.433						0.440	
Dolomite $kgCO_{2eq}/kg$				0.477				0.477	
Methane from crop residues							Assumed 0 if left on field and incorporated/mulched		
Methane from composting							25-90% of carbon content assumed degraded, <0.1 methane conversion factor (but <0.001 in forced-aeration compost)		
N_2O from composting		4.683 $kgCO_{2eq}/kgN$ (2010)					0.05 kgN_2O-N per kg degraded total ammoniacal N		
N_2O from crop residues		4.683 $kgCO_{2eq}/kgN$ (2019)							

Activity or product	Emission factors in earlier studies				Recent recommendations				
	Dairy	**Kiwifruit**	**Apples**	**Grapes/wine**	**Dairy**	**EPD Int PCR fruit and nuts**	**ecoinvent/World Food LCA Database**	**MfE GHG Guidance (NZ GHG Inventory)**	**Other**
CO_2 from urea/lime application							GLO value for $kgCO_{2eq}$/kg urea N GLO value for $kgCO_{2eq}$/kg limestone GLO value for $kgCO_{2eq}$/kg dolomite	1.59 $kgCO_2$/kg urea-N; 0.44 $kgCO_2$/kg limestone	
$GWPN_2O$		296(2010) 298(2019)							298 (IPCC 2013with CCF)
$EF1_{urea}$								0.0059	
EF1/EF4		0.01	0.01					0.01	
EF5		0.025(2010) 0.0075(2019)	0.025					0.0075	
FracGASF		0.1	0.1					0.1	
FracLEACH		0.07	0.07					0.07	

[1] Barber and Rothman 2008

Nitrous oxide

Nitrous oxide emissions from all forms of nitrogen added to the soil include synthetic N fertilisers, organic N in the form of compost and crop residues, and are accounted for using the IPCC methodology. The original studies used the IPCC 1996 methodology. While this was a change in the IPCC 2006 methodology up to 2014, New Zealand used the method outlined in the 1996 IPCC Guidelines. To be consistent with the 2006 Guidelines, New Zealand modified the methods/equations for this part of the national inventory in 2015, following a recommendation from the Agriculture Inventory Advisory Panel. In the revised guidelines (2006 IPCC Guidelines for National Greenhouse Gas Inventories), the fertiliser emissions methodology was changed so that fertiliser application is no longer adjusted for volatilisation. A more detailed explanation of this is in footnote 11, page 11.12, chapter 11, volume 4 of the 2006 guidelines. This aligns with the calculations used in He Waka Eke Noa.

Recommendations

Follow the IPCC 2006 methodology, Chapter 11: N_2O Emissions from Managed Soils, and CO_2 Emissions from Lime and Urea Application. Non-urea nitrogen has an emission factor of 5.40 $kgCO_{2eq}/kgN$. Urea coated and not coated with urease inhibitor has emission factors of 3.27 and 3.48 $kgCO_{2eq}/kgN$ respectively.

Limestone and dolomite

Limestone has an emission factor on application to the soil of 0.44 $kgCO_{2eq}/kg$ of $CaCO_3$. This is the IPCC Tier 1 emission factor of 0.12. Wells (2001) assumed limestone was of 90% purity Therefore, the total carbon dioxide emission is assumed to be 0.396 $kgCO_{2eq}/kg$ limestone (0.44 x 90%).

Dolomite has a Tier 1 emission factor of 0.13, or 0.477 $kgCO_2/kg$ dolomite - $CaMg(CO_3)_2$. Like lime Wells (2001) assumed 90% purity to reduce emissions to 0.429 $kgCO_{2eq}/kg$ dolomite.

Recommendations

Continue to follow the IPCC methodology and adjust for purity (IPCC Tier 2) of 90%.

Leaf litter, shoots, and compost

The quantity of nitrogen returned to the soil in organic matter needs to be determined. The equation below is used to determine nitrous oxide emissions during decomposition. Direct soil emissions = $kgN \times EF_1 \times 44/28 \times GWPN_2O$.

2.2 Agrichemicals

Activity or product	Emission factors in earlier studies				Recent Recommendations				
	Dairy	**Kiwifruit**	**Apples**	**Grapes/wine**	**EC PEF dairy**	**EPD Int PCR fruit and nuts**	**ecoinvent/World Food LCA Database**	**MfE GHG Guidance**	**Other**
Unspecified pesticide kgCO_{2eq}/kg							GLO value used for unspecified pesticide		
Fungicide kgCO_{2eq}/kg ai		12.5 (2010) 8.46 (2019) (210 MJ/k gai[1])	5.8						
Inorganic fungicide (S and Cu) kgCO_{2eq}/kg ai		1.42 (2010) 5.9 (2019) (5 MJ/kg ai[1])	0.3						
Fungicide – Sulphur kgCO_2/kg		5		1.42					
Fungicide – Captan WG kgCO_{2eq}/kg		7.32		7.32 (9.77 kgCO_{2eq}/kg ai)					
Fungicide (various brands) kgCO_{2eq}/kg				2.95 - 6.23					
Herbicide kgCO_{2eq}/kg ai		33.1 11.92 (Roundup) (310 MJ/kg ai [1])	12.2						
Herbicide - Glyphosate kgCO_{2eq}/kg ai		(550 MJ/kg ai [1]) (437 MJ/kg ai [1] in kiwifruit report	26.2	16					14.74 (2011 report – kiwifruit specific)
Herbicide kgCO_{2eq}/kg				4.5 (Amitrole) 7.9 (Guardoprim) 4.7 (Brown Out)					
Insecticide		14.8	11.1	18.1 (Karate)					19.17

Activity or product	Emission factors in earlier studies				Recent Recommendations				
	Dairy	**Kiwifruit**	**Apples**	**Grapes/wine**	**EC PEF dairy**	**EPD Int PCR fruit and nuts**	**ecoinvent/World Food LCA Database**	**MfE GHG Guidance**	**Other**
$kgCO_{2eq}/kg$ ai		(not incl oils) (310 MJ/kg ai[1]) 185 MJ/kg ai [1] in kiwifruit report		(4.16 $kgCO_{2eq}/kg$)					
Fungicide $kgCO_{2eq}/kg$ ai									17.37
Growth regulator $kgCO_{2eq}/kg$ ai			5.2						
Biocontrol agent $kgCO_{2eq}/kg$ ai		4.63 (BT)	4.6						
Oil		0.63 (120 MJ/kg ai[1])	0.5						
AgChem general ($kgCO_{2eq}/MJ$)		0.064							
Plant growth regulator (MJ/kg ai) [1]		200							
Biological control agent (MJ/kg ai) [1]		190							
Other (MJ/kg ai) [1]		120							
Agchem transport (MJ/kg agchem)		4.6 (Germany) 0.2 (NZ) 0.4 (Australia) 1.1 (Japan)							

[1] MJ/kg ai figures include manufacturing, packaging, and transport from country of manufacture.

The PCR document specifies that agrichemical production is an upstream process and that the use of agrichemicals is a core process. The emission factors used in the earlier studies (pipfruit and kiwifruit) incorporated both processes and were from a 1987 study (Green, 1987) with chemicals that are, now, mostly outdated. There is a range of agrichemicals used across the various fruit crops with differences in active ingredients and in the location of manufacture.

In the earlier apples study, a discussion of the differences in the emission factors between different sources of information was noted. While these sources were different, the average emission factors were similar within a broad category, say for example 5.8 compared to 6.0 $kgCO_{2eq}$/kg ai for fungicide. Therefore, a consistent use of emission factors from the same study was used, rather than using emission factors for individual chemicals sourced from separate studies.

A more recent report (2011) calculated the GHG emissions following ILCD guidelines (https://lcm.org.nz/data-sets/pesticides-used-new-zealand) for a kilogram of a representative herbicide, fungicide and insecticide used in the New Zealand kiwifruit industry considering cradle-to-NZ port. While these more recent emission factors are similar to those used in the earlier study, the report did highlight differences in emission factors depending on whether they are computed using IPCC characterization factors or using the ecoinvent database.

The total contribution of agrichemicals to on-orchard emissions was relatively small, with 4% (kiwifruit) and 10-14% (apples) of total emissions due to agrichemical use. However, the contribution of agrichemicals to the total greenhouse gas emission of the entire supply chain was even smaller at around 0.7% and 0.6-1.3% for kiwifruit and apples, respectively.

Recommendations

Agrichemical emission factors should be representative of the chemicals currently used by the respective industries. These factors should also be New Zealand-specific, accounting for the location of manufacture and transportation of the chemical to New Zealand. If these updated emission factors are not readily available, then it is recommended to use the same emission factors for the broad categories of agrichemicals of herbicides, fungicides and pesticides, as the earlier studies based on the study of Green (1987). This ensures consistency. Nevertheless, the contribution to supply chain emissions from agrichemical use is less than 1.5%.

2.3 Fuel and electricity

Activity or product	Emission factors in earlier studies				Recent recommendations				
	Dairy	Kiwifruit	Apples	Grapes/wine	Dairy	EPD Int PCR fruit and nuts	ecoinvent/World Food LCA Database	MfE GHG Guidance	Other
Use of diesel in tractors/machinery <18.64kW $kgCO_{2eq}$/hour							GLO value used for low/high load		
Use of diesel in tractors/machinery <74.57kW $kgCO_{2eq}$/hour				13.89 (50hp tractor/loader/ backhoe)			GLO value used for low/high load		
Use of diesel in tractors/machinery >74.57kW $kgCO_{2eq}$/hour				23.56 (120hp tractor) 45.96 (175hp tractor) 156.80 (500hp tractor) 123.66 (500hp off-highway truck)			GLO value used for low/high load		
Diesel burned in agricultural machinery $kgCO_{2eq}$/MJ							GLO value used		
Diesel LCA $kgCO_{2eq}$/l[1]		3.108(2010) 3.147(2019)	3.108						
Diesel-combustion only $kgCO_{2eq}$/l				2.65 (mobile) 2.61(stationary combustion) (2006)				2.69	

Activity or product	Emission factors in earlier studies				Recent recommendations				
	Dairy	**Kiwifruit**	**Apples**	**Grapes/wine**	**Dairy**	**EPD Int PCR fruit and nuts**	**ecoinvent/World Food LCA Database**	**MfE GHG Guidance**	**Other**
Petrol LCA $kgCO_{2eq}/l$[1]		2.735	2.735						
Petrol-combustion only $kgCO_{2eq}/l$								2.45	
LPG-mobile combustion $kgCO_{2eq}$/litre								1.64	
LPG-stationary combustion $kgCO_{2eq}/kg$				2.97 (2006)				1.64	
LPG Lift truck $kgCO_{2eq}/kg$			2.97						
Electricity $kgCO_{2eq}/kWh$		0.2441 (2010) 0.1086 (2019 for 2016)	0.2019	0.209 (Purchased electricity 2006 avg)	0.202 (2020 NZ electricity mix)		0.1203 NZ electricity (2014 mix)		0.166 (2018-2020 average) LCA based[2]

[1] From Barber and Stenning Fuel LCA Emission Factors 2018. Separated into combustion only and LCA (i.e., including upstream and transmission emissions). Revised emission factors are generated using the latest MBIE data, whilst original emission factors were generated from older MBIE datasets.

2 https://lcm.org.nz/data-sets/new-zealand-electricity

Emission factors associated with fuel and electricity use can be separated into two broad categories - those that account for direct emissions only and those that take a full LCA approach. Direct emissions of fuel use will generally be limited to combustion within machinery, and direct electricity emissions are generally limited to combustion at thermal power stations and fugitive emissions associated with geothermal generation. In contrast, an LCA approach will consider the full life cycle emissions incorporating upstream emissions (e.g., mining, gas processing emissions, refinery emissions) as well as transmission and distribution emissions. Values calculated using a full LCA approach will be higher than those which just consider direct emissions. This trend can be seen in the values in the Table above, but the earlier GHG study values are generally well aligned with the recent MfE values, as expected.

Emissions associated with fuel use will also vary depending on the size and type of machinery and its efficiency. For example, for tractors and other similar machinery, the ecoinvent datasets indicate that emissions may be halved or doubled depending upon the size of the engine, and there may be a factor six difference in emissions per hour depending upon the power required for the activity.

The proportion of renewable generation in the New Zealand electricity grid mix varies from year to year but is generally increasing over time and has comprised over 80% of total generation since 2014. This trend is expected to continue due to the lower marginal generation costs of wind and solar compared to fossil fuels (MBIE, 2019). Accordingly, it is important that emission factors for electricity are aligned with the time period covered by an LCA study as older data will tend to overestimate electricity emissions. This trend in the decrease in electricity emission factors can be seen in the table above when comparing the earlier GHG studies with more recent factors.

The PCR for fruit and nuts and wine provide a hierarchy of emission factors to be used for electricity with supplier-specific mixes given preference followed by residual grid mixes and finally national production mixes. However, data on supplier-specific mixes are limited in New Zealand.

Recommendations

Electricity emission factors shall be based on a New Zealand specific full LCA study and aligned to the timeframe of the LCA study as far as practicable. The use of a New Zealand national production mix will be appropriate in most situations unless on-site electricity generation is undertaken, in which case emission factors for the specific generation technology should be used.

For fuel use, emission factors shall be matched to the fuel type, and if data on actual consumption of fuel are not available, then emission factors per hour of activity should be based on the size of machinery and type of activity as far as practicable.

2.4 Machinery production and maintenance

Activity or product	Emission factors in earlier studies				Recent recommendations				
	Dairy	**Kiwifruit**	**Apples**	**Grapes/ wine**	**EC PEF dairy**	**EPD Int PCR fruit and nuts**	**ecoinvent/World Food LCA Database**	**MfE GHG Guidance**	**Other**
Agricultural machinery (unspecified) $kgCO_{2eq}/kg$ machinery							GLO value used		
Vehicle $kgCO_{2eq}/kg$		5.64 lifespan=15yrs	5.64						
Implement $kgCO_{2eq}/kg$		4.91 lifespan=20yrs	4.91						

2.5 Infrastructure on farm

Activity or product	Emission factors in earlier studies				Recent recommendations				
	Dairy	**Kiwifruit**	**Apples**	**Grapes/wine**	**EC PEF dairy**	**EPD Int PCR fruit and nuts**	**ecoinvent/World Food LCA Database**	**MfE GHG Guidance**	**Other**
Building $kgCO_{2eq}/m^2$		12.4 lifespan=20yrs	12.7						
Irrigation							Recommended to use length of irrigation pipes and hoses		
Steel wire $kgCO_{2eq}/m$		3.79 lifespan=30yrs	0.1						
Timber posts $kgCO_{2eq}/post$		0.19 lifespan=30yrs	3.9						
PVC $kgCO_{2eq}/kg$		4.6 lifespan=40yrs	4.6						
LDPE 15 $kgCO_{2eq}/kg$		3.7 lifespan=20yrs (2010) lifespan=30yrs (2019)	3.7						
MDPE 40 $kgCO_{2eq}/kg$		3.7 lifespan=30yrs							
Steel Agbeam $kgCO_{2eq}/m$		3.79 lifespan=50yrs						2.85 average structural steel	
Air flights of seasonal workers $kgCO_{2eq}/person\text{-}km$		0.108-0.154	0.108-0.154						

The PCR specifies that the manufacture of production equipment, buildings and other capital goods shall not be included in the LCA, and the PAS2050 also specifies that it is excluded (BSI, 2011, Section 6.4.4; BSI, 2012, Annex C). However, machinery maintenance such as tractors and packhouse equipment are core processes (EPD International, 2019, Section 4.3.1.2). The PEFCR specifies, “Capital goods (including infrastructures) and their End of life: they shall be included unless they can be excluded based on the 1.0% cut-off rule. The eventual exclusion has to be clearly documented.” (EC PEFCR, 2018, p.234).

The earlier kiwifruit and pipfruit GHG studies included the production of capital equipment used on the orchard. The capital equipment accounted for 6% to 17% of the GHG emissions associated with the orchard operations, and maintenance of this equipment is not mentioned in the reports. Production of capital equipment was not included in the earlier wine GHG study (but trellising wire, replaced posts and irrigation piping were included in the Barry (2011) study).

Seasonal workers contributed from near zero to 7% of the earlier kiwifruit and pipfruit orchard GHG emissions, and 2% of the packhouse/coolstore GHG emissions. However, the PCR excludes “business travel of personnel” and “travel to and from work by personnel” (Section 4.3.1.2).

Recommendations

As orchard infrastructure contributed more than 1% to the carbon footprint of the orchard life cycle stage in the earlier kiwifruit and apple studies, and as its relative contribution is likely to increase in future as orchard management practices change, as a default it should be included in carbon footprint studies. However, for agricultural systems where the infrastructure can be demonstrated to be contributing less than 1% to the carbon footprint of the agricultural stage, it can be omitted.

Maintenance activities associated with capital equipment, such as use of lubricants and cleaning agents should, also be included.

For seasonal workers, although the guidelines recommend exclusion of personnel travel, we recommend that they are included in future carbon footprint studies. Many seasonal workers fly into the country specifically to undertake orchard and packhouse/coolstore work on the Seasonal Work Scheme. These air flight emissions can potentially contribute more than 5% to the orchard and packhouse/coolstore life cycle stages of horticultural crops.

2.6 Packhouse/coolstore

Activity or product	Emission factors in earlier studies				Recent recommendations				
	Dairy	**Kiwifruit**	**Apples**	**Grapes/wine**	**EC PEF dairy**	**EPD Int PCR fruit and nuts**	**ecoinvent/World Food LCA Database**	**MfE GHG Guidance**	**Other**
Electricity use in packhouse/coolstore		0.295–0.590 kWh/tray 72–82% of the electricity use is for refrigeration							
Storage electricity kWh/t per day			10.6						
Refrigerant loss (kg) GWP: R404=3260, R22 = 1500 $kgCO_{2eq}/kg$		Loss (kg) x GWP	Loss (kg) x GWP						
Storage times in coolstore		3 months (green) 6 weeks (gold)							

Recommendations

The emission factors for electricity, refrigerant loss and storage times should be representative of the current practices in packhouses and coolstores that are used to pack and store fresh fruit. The emission factor for electricity should updated for the current mix in the New Zealand grid (see Section 2.3). The GWP for the refrigerants should follow the values from the latest IPCC report, although the loss of refrigerant (kg) will need to reflect modern equipment and management practices, as will any change in the storage times in the coolstore in relation to harvest dates and dispatch to the port.

2.7 Packaging

Activity or product	Emission factors in earlier studies				Recent recommendations				
	Dairy	**Kiwifruit**	**Apples**	**Grapes/wine**	**EC PEF dairy**	**EPD Int PCR fruit and nuts**	**ecoinvent/World Food LCA Database**	**MfE GHG Guidance**	**Other**
Packaging Materials									
Packaging for fertilisers and pesticides							Included in fertiliser and pesticide datasets		
Green glass bottle $kgCO_{2eq}/kg$				0.625 (green glass, at regional store)			GLO value used (no cullet used)		
Screw caps (aluminium) $kgCO_{2eq}/kg$				10.856					
Cartons $kgCO_{2eq}/kg$			0.212	0.824 (Corrugated board)					
Dividers $kgCO_{2eq}/kg$				-0.887 (solid unbleached board)					
Labels $kgCO_{2eq}/kg$				0.218 (coated woodfree paper)					
Plastic $kgCO_{2eq}/kg$			1.97						
Kiwifruit packaging[1] $kgCO_{2eq}/pack$									0.234-0.592
Packaging – End of Life									
Packaging waste to landfill $kgCO_{2eq}/kg$			0.611						

Activity or product	Emission factors in earlier studies				Recent recommendations				
	Dairy	**Kiwifruit**	**Apples**	**Grapes/wine**	**EC PEF dairy**	**EPD Int PCR fruit and nuts**	**ecoinvent/World Food LCA Database**	**MfE GHG Guidance**	**Other**
Paper and textile landfill waste $kgCO_{2eq}/kg$								3.0 paper, 1.80 textile (w/o gas recovery), 0.797 paper, 0.478 textile (w/ has recovery)	
Garden and food landfill waste $kgCO_{2eq}/kg$								1.5 garden, 1.125 food (w/o gas recovery), 0.398 garden, 0.299 food (w/ has recovery)	
Wood landfill waste $kgCO_{2eq}/kg$								3.225 (w/o gas recovery), 0.856 (w/ has recovery)	
Office landfill waste $kgCO_{2eq}/kg$								1.842 (w/o gas recovery), 0.489 (w/ has recovery)	
Polythene to landfill $kgCO_{2eq}/kg$			0.113						
Landfill mixed waste (default)-without landfill gas recovery $kgCO_{2eq}/kg$				0.874				1.170	
Landfill mixed waste (default)--with landfill gas recovery $kgCO_{2eq}/kg$								0.311	

[1] Tray data from 2012 Catalyst report to Zespri.

Previous recommendations in relation to wine packaging materials were based on a combination of European ecoinvent data and an LCA study on container systems for wine based on predominantly USA and Canadian data (Franklin Associated, 2006). Previous studies have shown that the glass bottle is a significant, and often the single largest, contributor to the GHG emissions associated with wine and therefore accurately representing the emissions associated with glass bottles is an important consideration. Some of the more recent ecoinvent data for green and white glass production are significantly higher than previous recommendations for wine studies which were based on ecoinvent v2 (2007).

For the other horticultural products, packaging contributed about 2% and 6% of the total CF of apples and kiwifruit respectively. Emission estimates for specific kiwifruit packaging systems listed in the table above are based on a Zespri report (Catalyst, 2012). Previous values for corrugated board boxes used in wine studies are four times higher than those used in apple studies.

Regarding end-of-life management, MfE have published a range of emission factors for different types of landfill waste, with and without landfill gas recovery. A comparison of the values in the table above shows that the MfE values are generally equivalent to, or higher than, values previously used in kiwifruit and wine studies for landfills without gas recovery but lower than values used in previous studies for landfills with gas recovery.

Recommendations

New Zealand specific EFs for glass used for wine packaging should be specific to the type (colour), weight and recycled content (with or without cullet) of the bottle/glass. For other packaging, emission factors should be specific to the packaging type and composition as far as practicable.

Regarding end-of-life management, the variability between landfill emission factors with and without gas recovery, and between different material types, indicates that both these aspects should be assessed as part of the LCA study.

2.8 Truck transport

Activity or product	Emission factors in earlier studies				Recent recommendations				
	Dairy	**Kiwifruit**	**Apples**	**Grapes/wine**	**EC PEF dairy**	**EPD Int PCR fruit and nuts**	**ecoinvent/World Food LCA Database**	**MfE GHG Guidance**	**Other**
Truck (>32 t) $kgCO_{2eq}/t.km$		0.116	0.116	0.111			EURO5 value used	>30t : 1.534 (diesel pre 2010 vehicles), 1.495 (2010-2015), 1.492 (post 2015)[1] $kgCO_{2eq}/km$	
Truck 16-32 t $kgCO_{2eq}/t.km$							EURO5 value used	1.240 (diesel pre 2010 vehicles), 1.208 (2010-2015), 1.206 (post 2015)[1,2] $kgCO_{2eq}/km$	
Truck >16t, fleet average $kgCO_{2eq}/t.km$		0.125						1.314 (diesel pre 2010 vehicles), 1.28 (2010-2015), 1.278 (post 2015)[1,3] $kgCO_{2eq}/km$	
Truck 3.5-16t				0.331					

Activity or product	Emission factors in earlier studies				Recent recommendations				
	Dairy	**Kiwifruit**	**Apples**	**Grapes/wine**	**EC PEF dairy**	**EPD Int PCR fruit and nuts**	**ecoinvent/World Food LCA Database**	**MfE GHG Guidance**	**Other**
$kgCO_{2eq}/t.km$									
Truck 7.5-16 t $kgCO_{2eq}/t.km$		0.268		0.278				0.770 (diesel pre 2010 vehicles), 0.730 (2010-2015), 0.720 (post 2015)[1,4] $kgCO_{2eq}/km$	
Lorry 7.5-16 t, EURO5, with refrigeration machine $kgCO_{2eq}/t.km$		0.268					GLO value used for CO_{2eq}/R134a refrigerants		
Truck 3.5-7.5t $kgCO_{2eq}/t.km$		0.626	0.626	0.642			EURO5 value used		
Truck 3.5-7.5 t, EURO5, with refrigeration machine $kgCO_{2eq}/t.km$							GLO value used for CO_{2eq}/R134a refrigerants		
Truck 7.5-17.5t $kgCO_{2eq}/km$				0.86					
Truck freight (unspecified) $kgCO_{2eq}/t.km$							GLO value used		
Truck with reefer, cooling $kgCO_{2eq}/t.km$							GLO value used		
Truck with refrigeration machine $kgCO_{2eq}/t.km$							GLO value used		

Activity or product	Emission factors in earlier studies				Recent recommendations				
	Dairy	**Kiwifruit**	**Apples**	**Grapes/wine**	**EC PEF dairy**	**EPD Int PCR fruit and nuts**	**ecoinvent/World Food LCA Database**	**MfE GHG Guidance**	**Other**
Transport from cool store to port $kgCO_{2eq}$/kg of fruit delivered to stores[5]									0.0019
Rail $kgCO_{2eq}$/t.km				0.0393					

[1] MFE Guidance uses different size categories than the ones shown in this table. Averages of multiple size categories have been used for comparison. MFE Guidance also distinguishes between year of vehicle manufacture and contains emission factors for hybrid and Battery Electric Vehicle (BEV) heavy vehicles.

[2] Average of 3 size categories: 15,000 – 20,000 kg, 20,000 – 25,000 kg, 25,000 – 30,000 kg

[3] Average of 4 size categories: 15,000 – 20,000 kg, 20,000 – 25,000 kg, 25,000 – 30,000 kg, >30,000 kg

[4] Average of 3 size categories: 7,500 – 10,000 kg, 10,000 – 12,000 kg, 12,000 – 15,000 kg

[5] Tray data from 2012 Catalyst report to Zespri.

A range of emission factors are available for truck transport, reflecting the range of different sizes and types of vehicles that may be used as well as the presence or absence of refrigeration. The values used in previous studies are generally aligned across the earlier studies, although some specific values were provided for some products (e.g., values for transport between overseas port and retailers for wine). The values in the Table above demonstrate that the emission factor per t.km of product transported decreases markedly as the size of truck increases; the earlier studies used factors that varied by a factor of five between the smallest and largest trucks.

MfE provides a wide range of recommended emission factors for New Zealand road freight differentiated by truck size and age. However, these values are not directly comparable with the earlier studies as MfE expresses the emission factors on a per km basis rather than a per t.km basis as used by other sources.

Recommendations

Data should be collected on the weight and type of vehicle (e.g., with or without refrigeration) relevant to the LCA study as well as the distance travelled.

2.9 Shipping

Activity or product	Emission factors in earlier studies				Recent recommendations				
	Dairy	**Kiwifruit**	**Apples**	**Grapes/wine**	**EC PEF dairy**	**EPD Int PCR fruit and nuts**	**ecoinvent/World Food LCA Database**	**MfE GHG Guidance**	**Other**
Container ship, freight $kgCO_{2eq}/t.km$				0.02 (4500TEU) 0.01 (8000TEU)			GLO value used		IMO (Table 5 below)
Container ship, reefer, with cooling $kgCO_{2eq}/t.km$		0.0525 (Zeebrugge) 0.0509 (Yokohama)	0.016				GLO value used		
Refrigerant leakage $kgCO_{2eq}/day$			0.001923						
Transoceanic freight ship $kgCO_{2eq}/t.km$				0.0107					
Inter-Island Ferry $kgCO_{2eq}/t.km$				0.05					
Fuel LCA ($kgCO_{2eq}/kg$)		3.618							
Fuel combustion ($kgCO_{2eq}/kg$)		3.166							

In horticulture, shipping can be a large component of total GHG emissions. In 2010 shipping was estimated to account for 44% of kiwifruit and up to 57% of pipfruit emissions.

There appear to be large differences between data sources for fuel use and GHG emissions associated with shipping. Recommendations in the previous studies all included that further work was needed on this topic. Moreover, the International Maritime Organisation (2020) has shown a large drop in shipping carbon emission intensity over 10 years between 2008 and 2018, including a 26% reduction for both container and general cargo ships.

Most of the publicly available shipping emissions are combustion, rather than life cycle based. Therefore, they need to be adjusted up when included in carbon footprinting studies. The 2010 kiwifruit study appears to of used a combustion rather than an LCA t.km emission factor (the original models are no longer available so this cannot be confirmed). Replacing the combustion EF with LCA increases the emissions by 16% to 0.0610 kgCO_{2eq}/t.km (Zeebrugge) and 0.0591 kgCO_{2eq}/t.km (Yokohama).

Based on the IMO (2020), a container ship (3000-4999 TEU) has an LCA emission factor of 10.7 gCO_{2eq}/t.km. This has been calculated using the figure of 17.0 gCO_{2eq}/t.nm (Table 61, Option 2 – IMO, 2020). Therefore 17.0 / 1.852 (km/nm) = 9.18 gCO_{2eq}/t.km. As the IMO figures are combustion emissions, they need to be converted to LCA emissions. International Fuel Oil (IFO 380) has a combustion emission factor of 3,030 gCO_{2eq}/L and an LCA emission factor of 3,520 gCO_{2eq}/L (Barber and Stenning, 2021), therefore a conversion factor of 1.162. This was applied to the combustion emissions, 9.18 x 1.162 = 10.7 gCO_{2eq}/t.km. Based on the IMO (2020) values LCA emission factors for a range of ship types and sizes are shown in Table 5.

Table 5 - Shipping emissions (based on IMO, 2020 - t.nm figures)

Ship type	Size category	Units	Combustion Emissions (gCO_{2eq}/t.nm)	Combustion Emissions (gCO_{2eq}/t.km)	LCA Emissions (gCO_{2eq}/t.km)
Container	0-999	TEU	34.2	18.5	21.5
	1000-1999	TEU	26.2	14.1	16.4
	2000-2999	TEU	19.7	10.6	12.4
	3000-4999	TEU	17.0	9.2	10.7
	5000-7999	TEU	16.3	8.8	10.2
	8000-11999	TEU	13.3	7.2	8.3
General cargo	0-4999	dwt	33.1	17.9	20.8
	5000-9999	dwt	29.7	16.0	18.6
	10000-19999	dwt	27.4	14.8	17.2
	20000-+	dwt	13.3	7.2	8.3
Refrigerated bulk	0-1999	dwt	181.7	98.1	114.0
	2000-5999	dwt	102.3	55.2	64.2
	6000-9999	dwt	79.2	42.8	49.7
	10000-+	dwt	58.9	31.8	36.9

Based on Te Manatū Waka Ministry of Transport (2021) data, the median container ship size in Q1 2021 was 3,752 TEU. This has remained consistent since Q4 2018. The largest size category since Q4 2014 is 4,000 – 5,9999 TEU and has been consistently increasing as other size categories have been on a downward trajectory.

Beyond the fuel use, there is uncertainty in the applied shipping methodology. This includes emission factors based on t.km not accurately reflecting that it usually is volume rather than weight that dictates the available space on a ship and its fuel use. Due to ships needing to carry ballast there is not a direct relationship between cargo weight and fuel use. Emission factors based on volume such as TEU (twenty-foot equivalent unit) containers may be a more equitable application of emissions rather than product weight.

The return voyage must also be considered. PAS 2050:2011 states that:

"transport emissions shall include the emissions associated with the entire delivery journey from source to delivery point and back, including those arising from any portion of the journey that products were not being transported. Where return journeys are used to transport other products, the emissions from those journeys shall be allocated to the products transported on the return journey."

Our interpretation of this would be to allocate a portion of the empty return journeys emissions to the outward journey's cargo. Based on an analysis of Ministry of Transport data (2021) in 2018 and 2019 68% of reefer containers that go out full come in empty. It has been very consistent since 2013, ranging between 66% to 70%. While an average of 17% of dry containers were imported empty, 20% of export dry containers went out empty. Therefore, as a proportion of full dry exported containers the average was -4%, i.e., there is a greater number of imports than exports in 2018 and 2019.

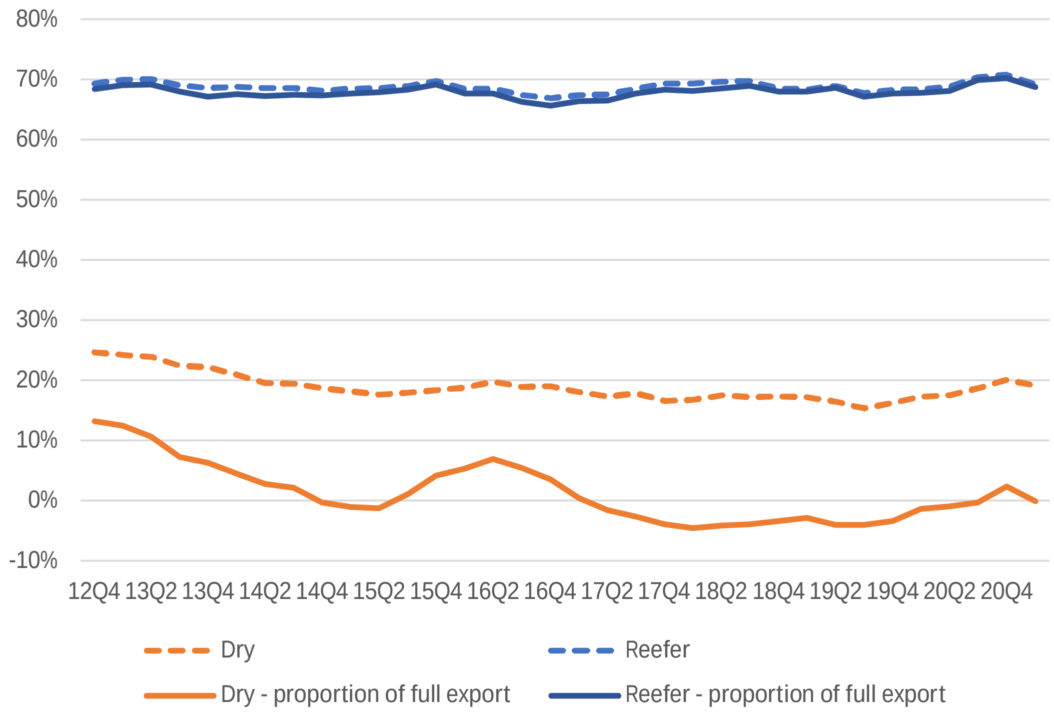

Figure 1.1 - Percentage of empty imported containers (dashed lines) and as a proportion of full exported containers (solid lines) for different years (first two numbers) and quarters of the years (Q1, Q2, Q3 and Q4).

Of all exports, approximately 70% go in dry containers. However, the split between dry and reefer containers is very commodity dependent. In the 12 months to March 2021, 78% of dairy products were sent in dry containers, similar to non-animal food at 86% dry 14% reefer containers. 94% of meat products were sent by reefer containers (Te Manatū Waka Ministry of Transport, 2021).

Recommendations

Where primary fuel use data is available from the shipping company then this should be used. Where this is not available then LCA software or publicly available emission factors should be chosen to match the ship type and size. If unknown, then the median container ship size leaving NZ is currently in the 3000-4999 TEU category. Note the ship size is likely to change on route to distant markets, and where possible this should be modelled.

A portion of the return voyage should be included where it is determined that the ship is returning partially full. Again, this will change on route from distant markets and where possible this should be modelled. As a default, for products sent out in reefer containers, 68% should be added to account for the empty return journey. Further analysis is needed as this approach will overestimate emissions, as empty containers incur less fuel use than full containers even when taking into account the need for additional ballast on a partially full return voyage. For products exported in dry containers, there is no need to account for the return journey component as they are more than balanced by imports.

There is no publicly available information on loading factors for refrigerated bulk shipping. As these are often chartered vessels this information should be collected directly from the shipping company, ideally along with fuel use.

2.10 Air transport

Activity or product	Emission factors in earlier studies				Recent recommendations				
	Dairy	**Kiwifruit**	**Apples**	**Grapes/wine**	**EC PEF dairy**	**EPD Int PCR fruit and nuts**	**ecoinvent/World Food LCA Database**	**MfE GHG Guidance**	**Other**
Helicopter transport $kgCO_{2eq}$/hour		226.9-396.4 (small/large helicopters)					GLO value used		
Air freight $kgCO_{2eq}$/t.km							GLO value used for short/medium/ long haul		

In ecoinvent, the emission factors for air freight (per t.km) vary by about 50% depending upon whether it is a short or long-haul flight. None of the earlier CF studies included air freight.

The earlier kiwifruit study discussed helicopter use for frost protection (Mithraratne et al., 2010, p.15), showing that this activity can make a significant contribution to the overall result

Recommendations

Any activities using helicopters, and air freight, should be assessed as part of a CF study given the high fossil fuel consumption associated with these activities.

3. Updates on methodology guidelines

Since the earlier reports were published, a number of international guidelines and standards have been published that are relevant for GHG accounting of food products. These include:

- ISO 14067 "Greenhouse Gases – Carbon Footprint of Products – Requirements and Guidelines for Quantification" (ISO, 2018).
- UNEP Life Cycle Initiative: Global Guidance on Environmental Life Cycle Impact Assessment Indicators (Volume 1, Frischknecht and Jolliet, 2016) (Volume 2, Frischknecht and Jolliet, 2019).
- International EPD System: "General Programme Instructions for the International EPD System" (Version 4) (EPD International, 2021).
- Australasian EPD Programme: "Instructions of the Australasian EPD Programme. A Regional Annex of the General Programme Instructions of the International EPD System" (Version 3.0) (Australasian EPD programme, 2021).
- International EPD Programme: Product Category Rule for Fruits and Nuts (EPD International, 2019).
- World Food LCA Database: "Methodological Guidelines for the Life Cycle Inventory of Agricultural Products" (Version 3.5, updated 29 February 2020) (Nemecek et al., 2019).

In addition, the European Commission Product Environmental Footprint (PEF) programme has been operating for several years and has produced the following guidance:

- EC PEF programme: "Recommendation of 9 April 2013 on the use of common methods to measure and communicate the life cycle environmental performance of products and organisations (EC, 2013).
- EC PEF programme: "Product Environmental Footprint Category Rules Guidance" (Version 6.3) (EC, 2018).

Also, the original PAS 2050 (BSI, 2008) was updated after the previous GHG studies, and released in 2011 (BSI, 2011). However, it has been superseded by ISO 14067 and so its guidance is not included in this section.

In general, these newer standards and guidelines make recommendations that are in line with the methods used in the earlier studies (and which are based on the ISO 14040 and 14044 standards (ISO, 2006a, 2006b). However, there are a few variations, and they are discussed in this section.

3.1 GHG metrics

Guidelines generally recommend using the GWP100 factors and using the latest values published by the IPCC. In particular, ISO 14046 (Section 6.5.1), the UNEP Life Cycle Initiative (Frischknecht and Jolliet, 2016, Section 3.6.10), and the more recent EC PEFCR (2018, p.69) recommend the use of these factors with climate-carbon feedbacks. The ISO standard states that other time horizons and the GTP may be used in addition to GWP100 but should be reported separately. The UNEP Life Cycle Initiative recommends using two indicators, GWP100 and GTP100, and inclusion of climate-carbon feedbacks.

The International EPD System (and corresponding PCR on fruit and nuts) requires the use of four indicators to represent climate change in EPDs: GWP-fossil, GWP-biogenic, GWP-land use and land-use change (LULUC), and GWP-TOTAL (the sum of the other three GWP indicators) (International EPD Systems, 2021). These categories are also required by the newer EC PEFCR guidelines (EC, 2018, p.66); however, they note that food LCAs model biogenic carbon emissions are treated as having zero emissions (apart from any methane emissions which are modelled).

Recommendation

Assess climate change using the latest IPCC values for GWP100, including climate-carbon feedbacks. It is additionally recommended to use GTP100 factors to represent the longer-term impacts of climate change.

3.2 Allocation Between Co-Products

The International EPD System PCR for fruit and nuts states that, “Where fruits and nuts are destined for human consumption, even though they may be of potentially different grades, they are considered equivalent in terms of the service they deliver, therefore no allocation is appropriate.”

The WFLCD guidelines (Nemecek et al., 2020, Section 2.6.2) state that co-products from crop production systems are allocated on an economic basis.

Recommendation

Fruits and nuts of different grades should be modelled on a mass allocation basis (but note the exclusion of products destined for animal feeds, see Section 3.4).

3.3 Modelling Animal Feed

The International EPD System PCR (2018, Section 4.6.1) for fruit and nuts states that, “Where substandard or waste fruits or nuts are used as animal feed this will be regarded as «near to waste treatment» and it is not considered as a displacement of other feedstock.” Their transport is to be included to the location of the subsequent activity (2018, Section 4.10.2).

Recommendation

Waste fruit and nuts that are used for animal feed should be regarded as "free" inputs to livestock systems (but their transport included).

3.4 Modelling Recycling

At end-of-life of a product, the International EPD System (2021, Section A.3.2) requires the inclusion of transportation of materials for recycling to the scrapyard/collection site to be included in the analysis. Correspondingly, studies of products using recycled material are required to include onward transport and processing from the scrapyard/collection site in the system under analysis. The WFLCD (Nemeek et al., 2019, Section 3.16.1) also follows the approach of allocating recycling processes to the system that is using the recycled material.

However, the EC PEF (2013, Annex V) had a complicated equation (called the End-of-Life formula) which includes factors relating to the quantities of virgin material, input recycled content, output material going to recycling, avoided energy credits and final waste management. The more recent EC PEFCR (2018, p.112) recommends instead the use of the Circular Footprint Formula; this assesses the same aspects but includes additional factors such as material quality factors.

Recommendation

As recycling of materials is largely outside the scope of this report, no specific recommendations are made here.

3.5 GHG Emissions Associated With Land Use and Land Use Change

ISO 14067 (Section E2) species that no GHG emissions are to be associated with ongoing land use where there is no change in soil carbon or above-ground biomass over time. However, where there is a change in land use, there may be an associated change in the average soil carbon content and/or above-ground biomass, and this is to be assessed relative to a reference land use, and allocated over a "specified time period." Section 6.4.9.5 notes that 20 years is frequently used as the time period. Where there is ongoing land use, and it is associated with a change in soil carbon content and/or above-ground biomass, then this is to be assessed over an appropriate time period (e.g., a full crop rotation), provided that "measures are in place to ensure its permanence" (Section 6.4.9.6).

The WFLCD also uses a 20-year time frame for assessing land-use change (Nemecek et al., 2019, Section 3.3), as does the EC PEF (EC, 2013, Section 5.4.9). It is worth noting that the UNEP Life Cycle Initiative (2019, Section 8.6) also recommended recently that changes in soil organic carbon should be used as an indicator of soil quality. And the EC PEF (2013, p.124) states that the impact category "Land Use" indicator is soil organic matter measured as "kg C".

The more recent EC PEFCR (2018, p.69) states that changes in soil carbon cannot be assessed because they cannot be guaranteed over a longer time period.

Recommendation

ISO 14067 should be followed in assessing land use and land-use change.

3.6 Cut-Off Rules

The International EPD System (Section A.3.3., 2021) requires that 99% of the result for any impact category should be included in the analysis. It additionally requires that 99% of both the mass of the product content and of the energy use of the product life cycle shall be accounted for.

The WFLCD guidelines (Nemecek et al., 2019, Section 2.4.1, 2.4.2, 2.4.3) state that processes are excluded when they contribute less than 1% to environmental impacts. The EC PEF (2013, p.94) did not allow cut-offs; however, the more recent EC PEFCR (2018, p.233) specifies that a 1% cut-off is allowed (based on mass, energy and environmental significance) per process but that the total cut-off cannot be higher than 5%.

Recommendation

A study should aim to be as complete as possible in modelling all relevant processes. Any omissions should be fully documented, and an estimate given of their potential contribution to the final results if included.

3.7 Inclusion of Infrastructure

ISO 14067 (2018, Section 6.3.4.1) states (as a Note), "Capital goods can be excluded in accordance with the goal and scope if their exclusion is not expected to significantly alter the conclusions according to specified criteria." A similar approach is followed in the International EPD scheme (2021, Section A3.1.2), which states that infrastructure, manufacturing equipment, etc. are to be included "if they make up a significant share of the overall attributable environmental impact." However, the International EPD Scheme PCR for fruit and nuts (2018, Section 4.3.1.2) states that "manufacturing of production equipment, buildings and other capital goods" shall not be included.

The more recent EC PEFCR (2018, p.234) states, "Capital goods (including infrastructures) and their End of life: they shall be included unless they can be excluded based on the 1.0% cut-off rule. The eventual exclusion has to be clearly documented." (EC PEFCR, 2018, p.234).

Recommendation

As the earlier New Zealand carbon footprint studies indicated that orchard infrastructure contributed more than 1% to the carbon footprint of the orchard life cycle stage, it should be included in carbon footprint studies. However, for agricultural systems where the infrastructure can be demonstrated to be contributing less than 1% to the carbon footprint of the agricultural stage, it can be omitted.

For packhouse/coolstore activities, infrastructure is less likely to make a significant contribution to the final product carbon footprint because of the high volume of product being processed and stored in the facilities. Therefore, it can be omitted from the analysis unless it is anticipated to contribute more than 1% to the carbon footprint of the packhouse/coolstore stage.

3.8 Establishment Stage for Perennial Crops

This life cycle stage is generally not mentioned in the standards and guidelines. However, the International EPD System PCR (2018, Section 4.3.1.1) does state that the initial establishment of a crop should be included "if the crop lifetime is expected to be less than 25 years." It also states that the production of seeds, cuttings or plants required for cultivation should be assessed.

Recommendation

As long as a representative range of orchards of different ages are surveyed, changes in yields associated with both the early and late stages of a perennial crop will be included in the analysis.

Regarding, cultivation of seedlings for subsequent use in orchards, there is currently insufficient research available to determine whether this is relevant or not in the context of GHG analysis of perennial crops.

4. Apples: Changes in Practices and Productivity, and Implications for the CF Results (Brent Clothier and Sam McNally)

The apple industry today in Aotearoa New Zealand is very different from that of 2008/2009, when the original LCAs for calculating the carbon footprint of export apples were carried out (Hume et al., 2009). Before providing an update of the carbon footprint, we first set the context and seek to understand the limitations in comparing the 2009 assessments with the contemporary situation. It is instructive to detail the manifold changes that the New Zealand apple industry has undergone over the last two decades.

Here we look at the changes that have happened to the apple industry since 2009 and assess the contemporary changes in the greenhouse gas emissions from the footprint results from Hume et al. (2009).

Unlike the project of Hume et al. (2009), which received significant resourcing to complete a detailed LCA analysis, this current updating required a smarter approach to obtain a generic assessment of changes. To do this, we carried out a desktop data-mining exercise of changes in industry structure, orchard practices, export patterns, corporate models, and infrastructural changes. This was complemented by discussions with Apple & Pears New Zealand Inc., Bostock, T&G Global, and Mr Apple (Appendices 1, 2, and 3). The latter three corporates comprise about 50-60% of New Zealand's apple exports. They were provided with the 2009 report of Hume et al. (2009) and asked to comment on how contemporary practices have changed. In return, they have commented on a late draft of this report, and their suggestions have been incorporated. Necessarily, the information provided via Appendices 1, 2 and 3 was in a qualitative and generic sense, and this information has enabled our re-assessments could be made. If more detailed LCA analyses are required in the future, these observations would easily lead to a more in-depth project to assess their implications on the current footprint.

The original intention was to make the functional unit one kilogram of apples at the retailer by using the Hume et al. (2009) retailer assumptions. However, given the significant change in the overseas market mix these assumptions do not hold true in 2021. Therefore, the functional unit is 1.0 kg of apples at the weighted average overseas port. The distance to the average port was based on the volume of apples exported. Figure 4.1 shows the system boundary.

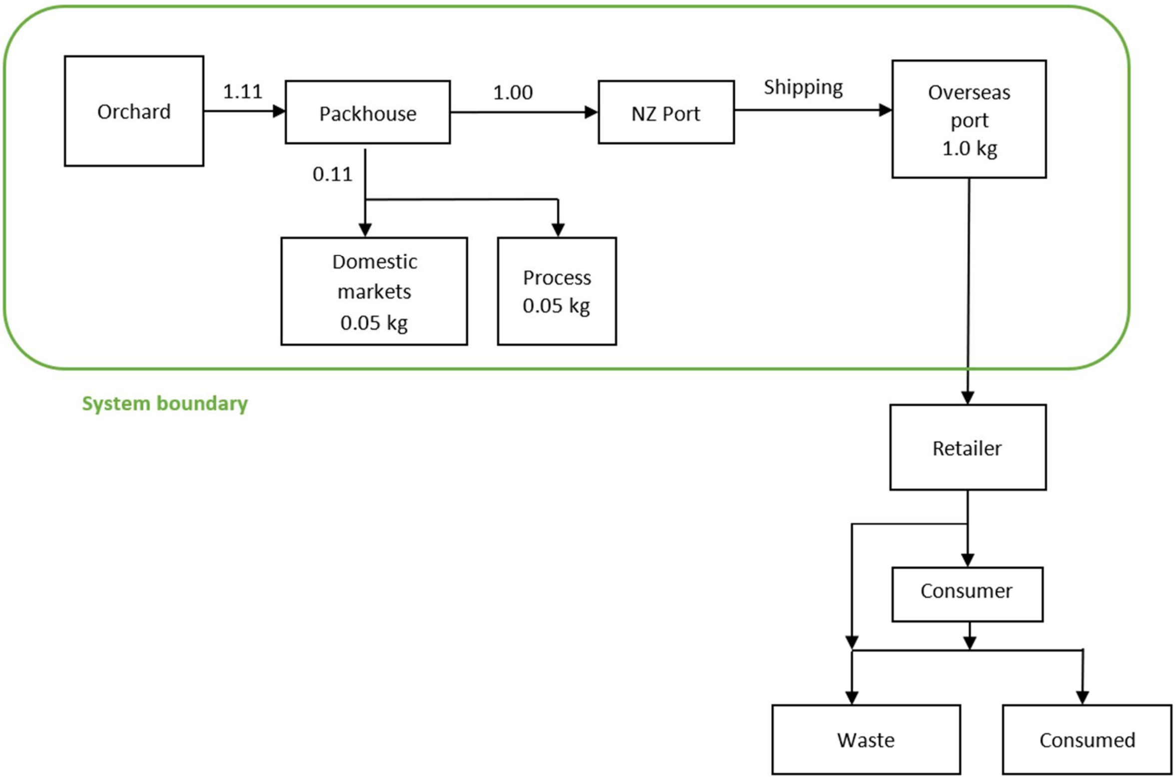

Figure 4.1 - *Apple flow along the supply chain from orchard to 1 kilogram delivered to the average overseas port. The rest of the supply chain is outside of the system boundary.*

4.1 Production and Exports

The World Apple Review carries out an annual survey into international competitiveness across 33 apple-exporting countries. New Zealand was ranked #1 for international competitiveness in both 2017 and 2018, scoring well across the 23 criteria. The Review said, "*New Zealand the Innovator: Because of its relatively small size, heavy export orientation and distance from major markets, the New Zealand apple industry has long relied on innovation to provide it with an edge over major competitors*". In 2009, New Zealand only ranked third, behind Chile and Italy. New Zealand produces about 0.55 million tonnes of apples, which is just 0.7% of the global production of 86.1 million tonnes. China alone produces 39.2 million tonnes or 46% of world production. Yet in 2019, New Zealand exported 42,700 tonnes of apples to China, which is about 11% of our exported apples.

In 1996 the area planted in apples was 15,819 ha, and by 2007 this had dropped to 9,247 ha, a decline of 42%. This drop was primarily precipitated by apple-price volatility between 1992 and 2001, which was compounded by the grower-led industry deregulation in 2001. Not surprisingly, total apple production in New Zealand has been volatile over the last 60 years (Figure 4.2), with a significant drop occurring in 2000. These production data derive from FAOSTAT (FAOSTAT).

Since 2010, the price received for apples has stabilised and grown in recent time. (Table 4.1). Indeed between 2010 and 2019, the price received for export apples has increased 1.7 times. In response, total apple production has increased over this time (Figure 4.2).

Table 4.1. The rise in the export price received for apples between 2010 and 2019 (FreshFacts, https://www.freshfacts.co.nz/)

	2010	2019	Ratio 2019:2010
Apple price ($ FOB per TCE)*	$22.93	$39.58	1.7

* FOB: Free-on-Board TCE: Tray Carton Equivalent = 18 kg

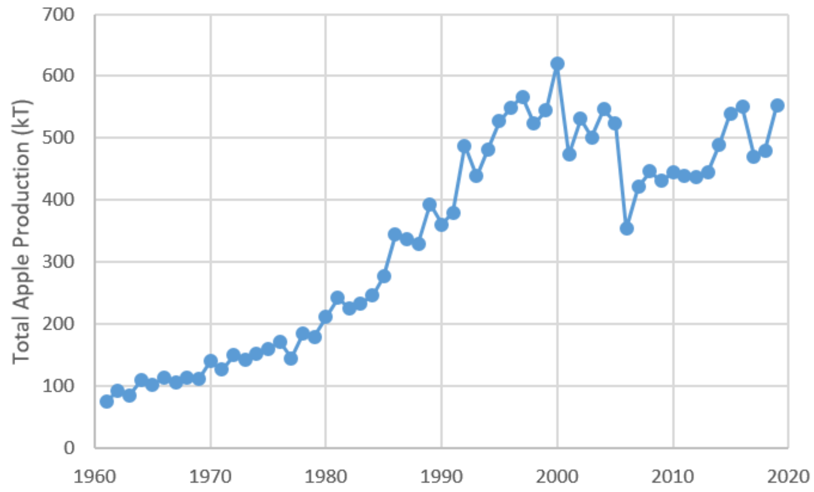

Figure 4.2 - Total apple production in New Zealand since 1960 (FAOSTAT)

Despite this volatility in the planted area of apples and total apple production, apples continue to be grown in three main regions of New Zealand: Hawke's Bay, Tasman-Nelson, and Central Otago (Table 4.2). These areas comprise 88% of New Zealand's apple plantings.

Table 4.2 - The three main apple-growing regions of New Zealand (FreshFacts 2008 and 2019) and the areal percentage split of the regions in 2017

	2007	2017	Percentage 2017 *
Hawke's Bay (ha)	5206	4746	55%
Tasman-Nelson (ha)	2438	2400	28%
Central Otago	472	427	5%

* These 3 regions comprise 88% of apple plantings

To complement Figure 4.2, we present in Table 4.3 more details about total apple production, apples exported and the national pack-out for export. Since the study of Hume et al. (2009), total apple production has increased by 30%, and the export volume has increased by 50%, as the national pack-out for export has risen from 59 to 70% (Table 4.3).

Table 4.3 - Changes in apple production and export volumes between 2008 and 2019 (FreshFacts; FAOSTAT)

	2008	2019	Ratio 2019:2008
Apples produced ('000 tonnes)	446	567 *	1.3
Apples exported ('000 tonnes)	261	395	1.5
National pack-out	59%	70%	

* FAOSTAT

Targeted picking and quality control in the orchard has increased export pack-out rates. From their survey results, Hume et al. (2009) suggested that of total production, 5% of apples went to the domestic market, and 31% for processing and juice. These numbers would be much less today and would merit further investigation. Furthermore, most processed juice is now from China, and export-reject fruit is now more likely to end up as fresh juice, pectins, cider, and to a lesser extent, stock-feed. Quality control now ensures that fruit picked in the orchard has a greater chance of being exported. Fruit picked and then not exported is a costly waste. In our discussions with Mr Apple, they reported pack-out rates as being as high as 85%. For our updating purposes here, we will consider the pack-out rate in 2019 to be 1.2 times (70/59) that in 2008.

The skill of New Zealand's growers is reflected in the long-term trend in the areal productivity of apple growing (Figure 4.3). So, despite volatility in the apple price and changes in production, apple productivity has steadily increased. This is also due in part to higher density plantings, with closer spacings and narrower rows, which are typical on modern production systems.

Hume et al. (2009), through their survey of 51 orchards, found apple production to be 79.8 t/ha for Braeburn and 55.0 t/ha for Royal Gala. In 2008, there were 952 exporting orchards, so the survey of Hume et al. (2009) covered just 5% of New Zealand's orchards. These productivity values are much above the national average figures shown in Figure 4.3, which are 44.6 and 55.3 t/ha respectively for 2008 and 2019. Here we will consider the carbon footprint results of Hume et al. (2009) but apply the national productivity increase of 1.25 (55.3/44.6) times.

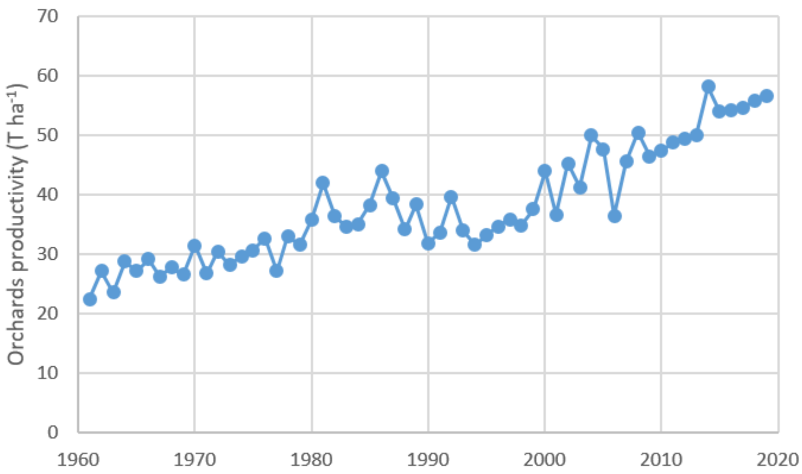

Figure 4.3 - The areal productivity of New Zealand's apple orchards in t/ha (FAOSTAT)

This trend is likely to increase, and possibly even at a faster rate in the near future. Plant and Food Research have developed a FOPS system (Future Orchard Planting System) set to produce a step-change in apple productivity. This FOPS system has been designed to intercept a much higher fraction of the incident sunshine such that apple production is enhanced. Production well above 100 t/ha has been recorded in a number of trials and prototypic FOPS orchards. This will dramatically reduce the LCA carbon footprint of apple production in the future through producing more export apples with the same amount of carbon investment.

We consider that the number of tractors passes to be the same in 2019 as in 2008. This is about 20-24 tractor operations per year. We assume here that similarly sized tractors are being used today. This latter assumption merits further investigation, especially because the spacing between rows has been reduced due to higher density plantings, and so it is likely that smaller tractors are being used. Also, as a result of these higher density plantings, we consider that the time of operation of tractors is some 20-25% more, as the path-length of passages has increased.

The major change within the orchard is we consider due to higher productivity, and we take this as being about 25% greater than in 2009.

4.2 Orchards and Packhouses

Between 2008 and 2019, the planted area in apples has increased by about 20% due to stronger prices of late (Table 4.4). The number of exporting orchards has also increased, but not by as much, reflecting the greater consolidation of orchards and the rise in apple orcharding by corporates such as Mr Apple, T&G Global and Bostock.

Table 3.4 - The planted area in apples between 2008 and 2019, along with the number of export orchards, export packhouses and exporting companies (FreshFacts)

	2008	2019	Ratio 2019:2008
Planted area (ha)	8,832	10,719	1.2
Export orchards (#)	952	996	1.05
Export packhouses (#)	74 *	52	0.7
Exporters (#)	90	73	0.8

* Interpolated

This rise of corporate orcharding has seen the number of exporters decline, and this has also meant that the number of packhouses has decreased by 30%, such that modern packhouses are bigger and more efficient. The gains in the reduction of GHG emissions are likely to be large in the coolstore-packhouse part of the supply chain. There will, however, through there being more orchards and fewer packhouses, a larger distance for trucking of the harvested fruit between the orchard and packhouse. Hume et al. (2009) found this distance to be between 10 and 13 km. Likewise, the distance between the packhouse and port will have increased. Hume et al. (2009) considered this to be between 30 and 90 km. Although the respective distances would be greater nowadays, the emissions of this trucking would most likely be offset by the use of more modern trucks with lower emissions.

So overall, in the orchard and packhouse stages of the apple supply chain, we consider that there have been overall efficiency gains, in terms of emissions per kilogram of apples loaded onto a ship, by a factor of 1.2. The carbon footprint over these two stages we now consider to 83% of those in 2008.

4.3 Export Destinations

The study of Hume et al. (2009) considered the LCA carbon footprint of apples despatched to the United Kingdom and consumed there. They did note that expanding apple exports in Asia and North America would potentially lower the carbon footprint of the apple sector. So it is illustrative to assess the changing destinations of New Zealand's export apples since the study of Hume et al. (2009). Freshfacts in 2008 and 2019 provide a breakdown of the export value from these changing destinations, and these are given in Table 4.5 for five regional groupings.

Table 4.5 - A regional breakdown of the destinations for New Zealand's export apples between 2008 and 2019, and the percentage changes over this period (FreshFacts 2008, 2019)

Export Destination	2008 *		2019 @		Δ %
	Export ($ m)	Percent	Export ($ m)	Percent	2008-2019
Continental Europe	122	35.4%	149	19.2%	-16.2%
SE & E Asia	71	20.5%	460	55.5%	+35.0%
United Kingdom	58	16.8%	74	9.6%	-7.2%
North America	58	16.8%	92	11.1	-5.7%
Middle East	8	2.3%	38	4.6%	+2.3%

* Total exports $345 m @ Total exports $829 m. Δ change =2.5

There has been a dramatic 35 percentage-point shift to South-East and East Asia as the destination for New Zealand’s export apples. Over 55% of New Zealand’s export apples end up nearby in Southeast and East Asia. The leading suppliers of apples to China are New Zealand, Chile, the USA and South Africa. In terms of the shipping footprint of apples imported into China, there will be a similarity between the major exporters.

There has been a 16% decline in the fraction of New Zealand’s apples going to continental Europe, and a drop of 5-7% in the proportion despatched to the United Kingdom and North America.

Nonetheless, there have been increases in the total value across all regional groupings. There has been a 6.5 times rise in the value of exports to Asia, 4.8 times to the Middle East, 1.6 to North America, 1.3 to the United Kingdom, and 1.2 times to Europe.

Our LCA assessment here for the total export sector will now consider these regional weightings of New Zealand’s export apples.

4.4 Cultivars

The original study (Hume et al. 2009) assessed the carbon footprint using both the PAS 2050 LCA methodology and the ISO 14040 series LCA standards. They also considered Braeburn and Royal Gala cultivars, grown under either integrated fruit production (IFP) protocols or organic certification. The ambit of that study reflected the large amount of resourcing provided to carry out detailed investigations by a large team of seven scientists from four organisations. They carried out surveys and interviews in the field on orchards, packhouses and coolstores.

Nowadays, the cultivar make-up of our apple exports has changed (FreshFacts). In 2009, Royal Gala and Braeburn comprised approximately the same fraction of exports, such that in sum, they contributed nearly 70% of the exported cultivars (Figure 4.4). In 2019, Royal Gala still comprised over 30% of export, whereas those of Braeburn had halved.

Figure 4.4 - The cultivar distribution of apple exports in 2009 (left), compared to the latest data for 2019 (right).

For the purposes of this update of the carbon footprint, it was decided to focus only on the average of Royal Gala and Braeburn. Without detailed surveying, it was not possible in this update to distinguish between IFP and Organic. However, the large commercial apple grower Bostock was interviewed as part of this study. They export around 85% of New Zealand Organic apples, so both integrated and organic practices are considered in a generic sense. As there is little difference between the cultivars of Braeburn and Royal Gala in Hume et al. (2009), we will here consider a ‘generic’ apple cultivar. Furthermore, we will focus on integrated fruit production (IFP), although the difference with Organic is quite small.

4.4 Recognised Seasonal Employer (RSE) workers

The average PAS2050 GHG (greenhouse gas) emissions for a kg of apple exported to the United Kingdom is 900 g CO_{2eq} (Hume et al., 2009). Across a spread of countries providing RSE workers, they found that the contribution of international travel to and from New Zealand added about 5 g CO_{2eq} to the footprint, or about 0.6% of the carbon footprint. Covid-19 notwithstanding, the number of RSE workers coming to New Zealand has trebled between 2008 and 2019 (Figure 4.5), such that it is likely that this is now likely to be 15 gCO_{2eq} per kg of apples. This is just 0.015 $kgCO_{2eq}$ per kg of apples or just 1.7% of the total footprint.

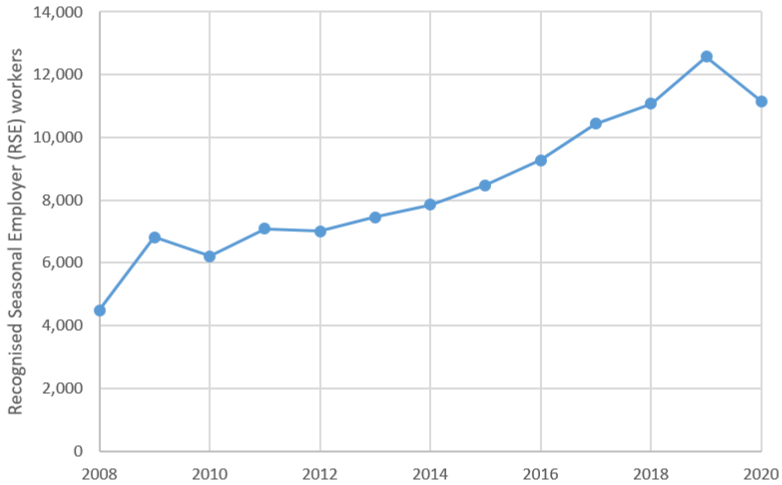

Figure 4.5 - The total number of Recognised Seasonal Employer (RSE) workers coming to New Zealand. This is across all industries and not just related to the apple industry.

It is reported that the New Zealand apple and pear industry employs about one-half of all the RSE workers coming into New Zealand (https://www.nzkgi.org.nz/wp-content/uploads/2020/07/RSE-Doc-June-2020-WEB-FINAL.pdf). Yet, despite this rise in RSE workers, the contribution of their emissions to the total footprint of export apples is still small.

4.5 Revisiting the 2008 Footprint

Hume et al. (2009) computed the LCA GHG emissions for New Zealand apples exported and consumed in the UK. They considered LCA calculations based on the ISO 14040 LCA standards, and the PAS 2050 method, and they applied these to both the cultivars of Braeburn and Royal Gala, grown under integrated fruit production (IFP) (Figure 4.6) and Organic practices (Figure 4.7).

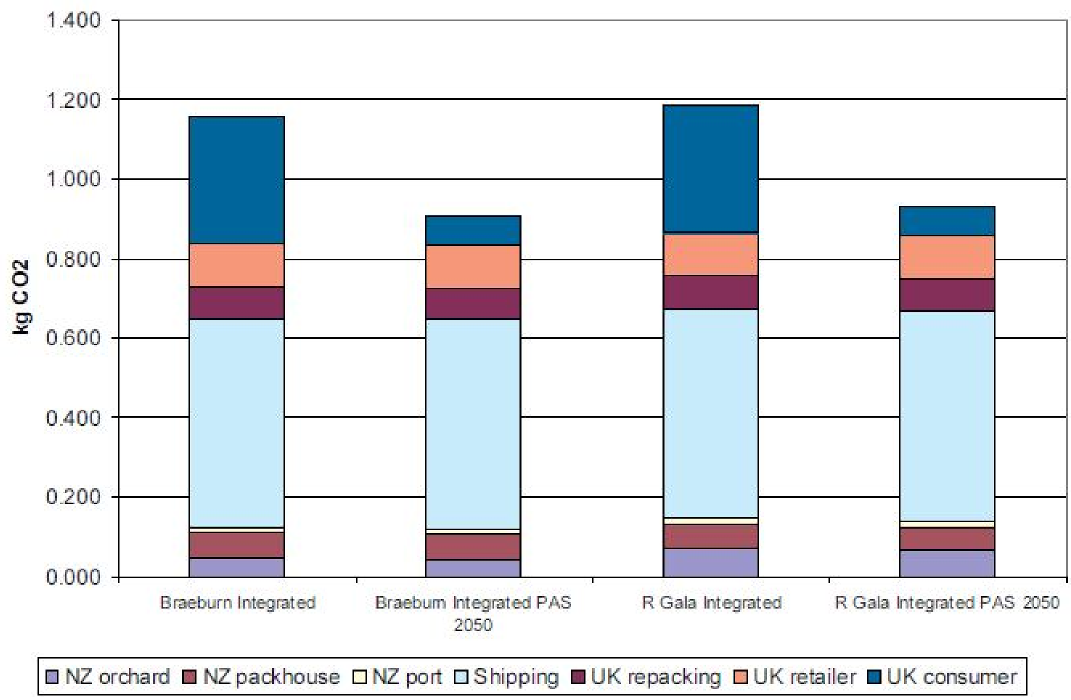

Figure 4.6 - The greenhouse gas (GHG) footprint in $kgCO_{2eq}$/kg apples exported from New Zealand (NZ) and consumed in the United Kingdom (UK). This is Figure 9 from Hume et al. (2009) and shows the footprint calculated using two different protocols: the ISO Series 14040 LCA Standards (left) and the PAS2050 scheme (right) for integrated fruit production (IFP) practices for both Braeburn and Royal Gala cultivars. The various stages of the supply chain are shown in different colours.

Using the PAS 2050 protocol, the footprints for Braeburn and Royal Gala were respectively 0.905 and 0.928 $kgCO_{2eq}$/kg apples exported from New Zealand (Figure 4.6). For Organic orchards, these footprints were respectively 0.980 and 0.970 $kgCO_{2eq}$/kg apples. The differences between cultivars and orchard practices are minimal.

For the updating purposes here, we will consider the PAS 2050 methodology, and we will not distinguish between either cultivars or orchards practices. Further, we will take the carbon footprint to be of the order of 0.95 $kgCO_{2eq}$/kg apples exported and consumed in the UK.

Hume et al. (2009) found that the orchard stage of the supply chain contributed 10% of this footprint, and the packhouse-coolstore stage contributed 7%. Thus, it is considered that in 2008, the orchard practices and the packhouses-coolstores had a carbon footprint of 0.16 $kgCO_{2eq}$/kg apples. We will use this number as the 2008 baseline and consider how orchard and packhouse-coolstore practices have changed.

We will address changes in the footprint of shipping in a later section.

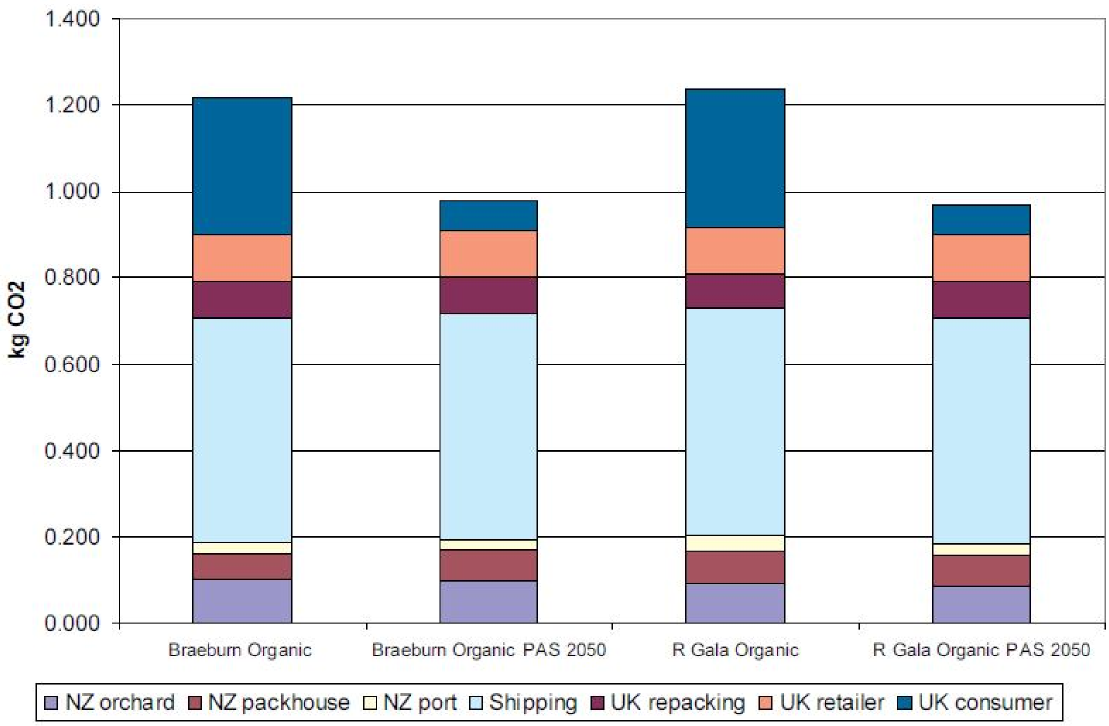

Figure 4.7 - The greenhouse gas (GHG) footprint in kgCO$_{2eq}$/kg apples exported from New Zealand (NZ) and consumed in the United Kingdom (UK). This is Figure 9 from Hume et al. (2009) and shows the footprint calculated using two different protocols: the ISO Series 14040 LCA Standards (left) and the PAS2050 scheme (right) for Organic practices for both Braeburn and Royal Gala cultivars. The various stages of the supply chain are shown in different colours.

4.5.1 Modern Orchard-Packhouse Practices

We now consider how modern orchard processes and practices in the packhouse-coolstore have changed and how these might have affected this baseline footprint of 0.16 kgCO$_{2eq}$/kg apples.

As noted above, apple productivity in 2019, in terms of t/ha, is considered to be 1.25 times that in 2008, and the apple pack-out from the packhouse-coolstore is 1.2 times that of 2008. On the debit side, it is considered that on-orchard trafficking would have increased by 25% due to higher density plantings, whilst the number of RSE workers has trebled, and that there is a longer transport distance between orchards and packhouse-coolstores, and between the packhouse-coolstore and port. In terms of impact on the carbon footprint, we consider that the rise in apple productivity and the increase in the pack-out would dominate. So, we consider that this rise by a factor of 1.2 would result in a reduction in the orchard-packhouse-coolstore footprint by 83%, being (1/1.2).

Our estimate of the orchard-packhouse-coolstore carbon footprint in 2019 is 0.13 kgCO$_{2eq}$/kg apples, a reduction of 0.03 kgCO$_{2eq}$/kg apples, or a drop of 19%.

4.5.2 An Assessment of the Global Footprint of all Apple Exports

Hume et al. (2009) considered the LCA carbon footprint of apples exported and consumed in the UK. The port of destination in the UK was Felixstowe. They considered the GHG emissions of apples exported in refrigerated containers was 0.016 kgCO$_{2eq}$/t/km (Section 2.9). Backhaul emissions in Hume at al. (2009) were set at 0.10 $kgCO_{2eq}$/t/km with there being no refrigeration, and they only allocated 30% of the backhaul to apples (see Table 28 in Hume et al., 2009). We have also used these numbers here in our update. As can be seen above in Table 5 these emission factors currently apply to ships carrying less than 2000 reefer containers in twenty-foot equivalent units (TEU). Currently ships are likely to be much bigger than 2000 TEU. In 2016 just 45% of container ships leaving Port Napier carried more than 4000 TEU. By the end of 2020, some 80% of ships leaving Port Napier had loads of more than 4000 TEU (Deloitte's New Zealand Ports and Freight Yearbook 2021). The emissions factor for ship carrying under 5000 TEU is just 0.107 $kgCO_{2eq}$/t/km, which is 35% less than that of a ship carrying less than 2000 TEU. The recommendation for shipping (Section 2.9) now considers, the default ship size could be considered in this range of 3000-4999 TEU. Should further shipping information be deemed worthwhile to consider, we could carry out a more detailed analysis to find out the actual average ship-size carrying apples in reefer containers, and then choose an appropriate emission factor from Table 5. We could then easily change the arithmetic of our calculations in Table 4.6. It is very likely that the shipping footprint will be much less than that we have calculated here, as the emission factor for reefer sea transport outwards will now be less than the 0.016 $kgCO_{2eq}$/t/km we have used here. We have used this value, so that we could make a direct comparison with the earlier work of Hume et al. (2009) and to highlight the impact of the changed destination for New Zealand's export apples.

With their emission factors for shipping and backhaul, given that the distance from Napier to Felixstowe is 20,683 km, the emissions from shipping apples in refrigerated containers to the UK, Hume et al. (2009) found to be 0.50 $kgCO_{2eq}$/kg (Table 3.5). Shipping was found by Hume et al. (2009) to contribute to 54% of the total carbon footprint (Figures 4. and 4.).

Given that there have been dramatic changes in the destinations for New Zealand's export apples (Table 4.5), a comparison between a single-port destination in 2008 and 2019 would seem inadequate. We have revisited the sole-port destination of Hume et al. (2009) to calculate a globally weighted average footprint for all export apples by considering their respective destinations. The results of these recalculations for both 2008 and 2019 are presented in Table 4.6.

Table 4.6. The regional export destinations (kt) and region-weighted greenhouse gas (GHG) emissions (kt CO_{2eq}) of New Zealand apples exported from Napier in 2008 (Hume et al. 2009) and for 2019. The GHG emissions for both years assume that the emissions from shipping are 0.016 $kgCO_{2eq}$/t.km, and 30% of the backhaul is allocated to apples. The destination ports chosen for each region are given in the column third from the right

Export Destination	2008* ktonnes	Fraction	T-km	Total GHG emissions kT CO_{2-e}	2019# ktonnes	Fraction	T-km	Total GHG emissions k T CO_{2-e}	New Zealand Port - Napier to …	GHG kg CO2-e/kg	(km)
Continental Europe	93.4	0.36	1945	46.3	75.8	0.19	1579	37.6	Antwerp	0.50	20820
SE & E Asia	53.5	0.21	538	11.8	219.2	0.56	2205	48.2	Shanghai	0.22	10058
United Kingdom	43.8	0.17	907	21.9	37.9	0.10	784	19.0	Felixstowe	0.50	20683
North America	43.8	0.17	462	11.0	43.8	0.11	462	11.0	Long Beach	0.25	10527
Middle East	6.0	0.02	63	1.5	18.2	0.05	191	4.5	Jebel Ali	0.25	10527
Total exports kT, Total T-km & Total T CO2-e/kg	**240.6**		**3915**	**92.5**	**395.0**		**5221**	**120.3**			

We have used the emissions factor of 0.016 $kgCO_{2eq}$/t/km for the outward shipping, and 0.010 $kgCO_{2eq}$/t/km for the backhaul with 30% allocated to apples in all calculations for both years (following Box 2 in Hume et al., 2009). This can easily be revisited if need be. Next, we have divided the destination for New Zealand apples into the five regions of Continental Europe, SE and E Asia, the United Kingdom, North America and the Middle East. We consider a modal port in these regions, respectively Antwerp, Shanghai, Felixstowe, and Jebel Ali. The distances from Napier to these ports are given in Table 4.5, along with the emissions from shipping of apples in $kgCO_{2eq}$/kg apples.

Freshfacts (2009) and (2019) were used to determine the export of apples to these various regions. In 2008 there were 240.6 kt of apple exports, and this had risen to 395 kt in 2019, and for the respective distances travelled there were 3915 t.km of apple exports in 2008 and 5221 t.km in 2019. So, whereas the tonnage of exports increased by 1.64 times, the weighted distance in T.km only increased by 1.33 times, as more exports were despatched to ports that were closer and primarily in Asia.

As a result of the increased tonnage of exports and the changing destinations, the total shipping emissions rose from 92.5 kt CO_{2eq} in 2008 to 120.3 kt CO_{2eq} in 2019.

So, the globally weighted total footprint from the shipping of New Zealand's export apples in 2008 was 0.38 $kgCO_{2eq}$/kg apples, being 92.5/240.6 (Table 4.6). This is much less than the 0.5 $kgCO_{2eq}$/kg apples from Hume et al. (2009), which only accounted for the single port destination of Felixstowe in the UK. So, the global footprint for apples in 2008 is 0.12 $kgCO_{2eq}$/kg apples less than the calculation of Hume et al. (2009) for just the UK.

So, rather than the Hume et al. (2009) calculation for just the UK, we find that the globally weighted carbon footprint of New Zealand's apples in 2008 was just 0.83 $kgCO_{2eq}$/kg apple exported. This is 13% less than the 0.95 $kgCO_{2eq}$/kg apples through just considering the UK as a destination.

In 2019, with the greater export of apples to closer ports in SE and E Asia and the Middle East, the globally weighted shipping footprint was only 0.30 $kgCO_{2eq}$/kg apples, being 120.3/395 (Table 4.6).

So, between 2008 and 2019, there has been a 21% drop in the shipping footprint of apples due to the changing mix of destination ports. The shipping footprint has shrunk by 0.08 $kgCO_{2eq}$/kg apples exported.

4.5.3 An Updated Provisional carbon Footprint for Apples

We have carried out a re-analysis of the carbon footprint of New Zealand's export apples from Hume et al. (2009). We have computed a globally weighted analysis of the footprint in terms of all of the destinations for New Zealand's export apples across five regions.

We have calculated that the globally weighted carbon footprint in 2008 would have been 0.83 $kgCO_{2eq}$/kg apples, a footprint some 13% lower than the figure of Hume et al. (2009) for just the UK alone.

Through a quantitative and qualitative assessment of changed orchard practices and protocols in the packhouse-coolstore, we consider there now to have been a reduction in the footprint of these two stages of 0.03 $kgCO_{2eq}$/kg apples. At this stage we have not carried out any further analysis beyond the destination port.

Our quantitative assessment of the relative global weightings of the changes in the destinations of New Zealand's export apples between 2008 and 2019 leads us to conclude that the emissions from the shipping stage of the supply chain dropped from 0.38 to 0.30 kgCO_{2eq}/kg apples, a reduction of 0.08 kgCO_{2eq}/kg apples, or a drop of 21%.

The total reduction between 2008 and 2019 in the supply-chain stages of the orchard, packhouse, coolstore, and shipping amounts to 0.11 kgCO_{2eq}/kg apples.

Thus between 2008 and 2019, we consider that the carbon footprint of New Zealand's export apples along these parts of the supply chain has dropped from 0.83 to 0.72 kgCO_{2eq}/kg apples, a reduction of 13%.

A detailed quantitative survey, akin to that of Hume et al. (2009), would seem to be merited to confirm these numbers.

5. Kiwifruit: Changes in Practices and Productivity, and Implications for the carbon footprint results (Andrew Barber)

5.1 Introduction

Zespri commissioned an updated carbon footprint assessment in 2019 (Barber et al., 2019). The assessment was for Green ('Hayward' variety) kiwifruit produced in NZ (specifically the Bay of Plenty) then distributed to and consumed in Germany. It included surveying 11 Green kiwifruit orchards. Orchard emissions for the Gold variety were also investigated, and while they had higher production their inputs were also higher, with the final orchard emissions being almost the same at just 2% lower.

The 2019 study (Barber, et al., 2019), which covered on-orchard resource use for the 2016/17 growing season, is the basis for the results presented in this report. Differences between the 2019 Zespri study and this 2021 MPI study are due to:

- Methodology and emission factor changes which were identified and subsequently standardised across the sectors included in this MPI study.
- The functional unit was also changed from per kilogram consumed in Germany to per kilogram at the German retailer. The results presented in this report have been updated to use the same functional unit (to the retailer) for comparison purposes.

The change in functional unit means that the results presented in this report are lower than the 2019 values reported elsewhere by Zespri, where the system boundary was the consumer as opposed to the retailer, and so therefore includes accounting for consumer waste.

The Global Warming Potentials (GWP) in this study uses the climate-carbon feedback (CCF) from the IPCC 2014 5th Assessment Report. Methane has a GWP of 34 (28 without CCF), and nitrous oxide is 298 (265 without CCF).

The 2021 values have also been compared to emissions from the 2010 study (Mithraratne et al., 2010) and revised 2010 values. The revision of the 2010 figures has enabled a comparison of system rather than methodology changes. The original 2010 values have also been updated so the functional unit matched this study.

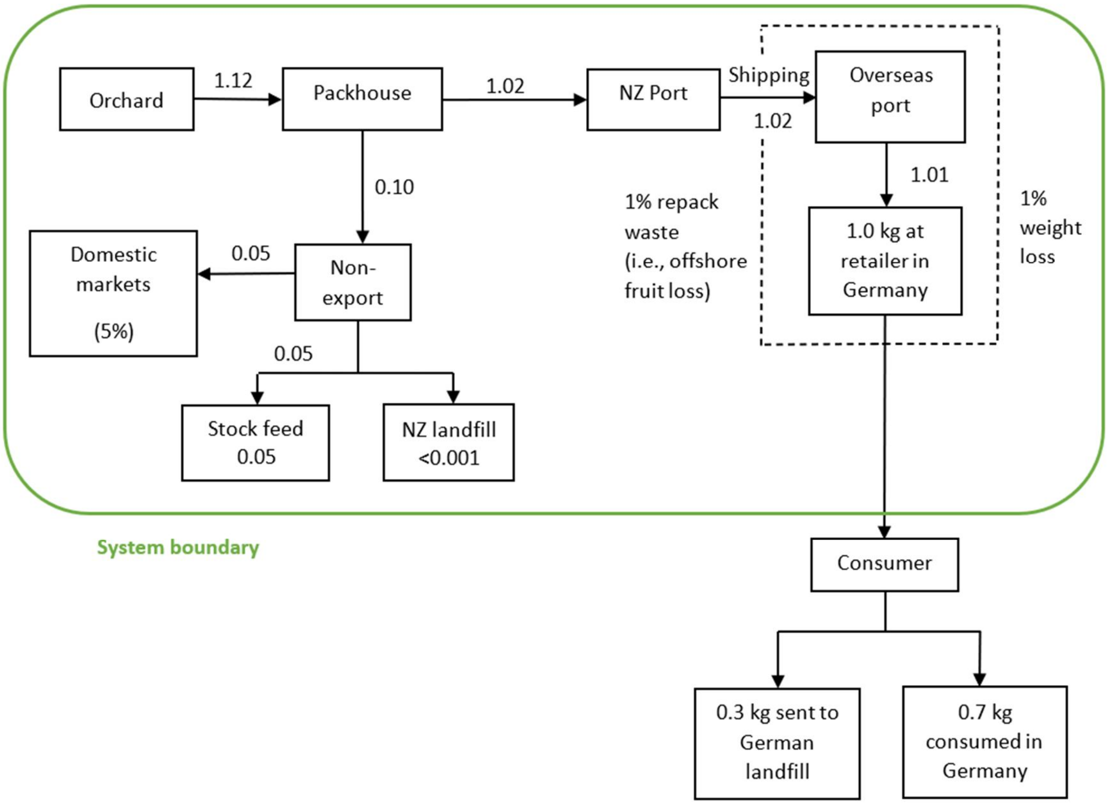

Figure 5.1: Kiwifruit flowing along the supply chain from orchard to 1 kilogram delivered to the retailer in Germany.

5.2 Changes in practices since 2010

5.2.1 Orchard practices

Based on the system boundary being to the German retailer, excluding the consumer (primarily fruit waste), orchard emissions were found to account for 7.6% of total emissions, nearly 50% lower per kilogram of kiwifruit harvested compared to the revised 2010 values.

This change has been primarily driven by a significant 69% increase in production, from 7,400 to 12,500 tray equivalents per hectare. Despite higher production many inputs were lower including fuel use (-38%), electricity (-61%), nitrogen (-21%), and lime (-14%). Agrichemical use changes were a mixture of lower herbicide and synthetic fungicide use, while there were increases in oil, and inorganic fungicides (S & Cu).

Kiwifruit orchards are reasonably capital intensive, primarily machinery and steel support structures. Capital was the second-largest orchard component, just ahead of energy and just below field emissions. As orchard capital makes up 2% of total emission to the retailer, it was included in the study. Even if the functional unit was to the consumer, with their associated 30% fruit waste, orchard capital remained just above the 1% materiality cut-off limit. Capital has been reported separately, so it could be removed if being compared to methodologies that exclude capital.

Compared to the 2010 study, this study has applied updated emission factors to energy, fertiliser, and field emissions, along with methodology and assumption changes. This has resulted in orchard emissions being half of the revised 2010 study.

5.1.2 Packhouse and packaging

The packhouse stage, including transport, accounts for 12% of total emissions (retailer system boundary). The 2019 study included eight packhouses, up from two in the original 2010 study. This is likely to be a significant cause for some of the differences described below. This also underscores the importance of collecting the resource use inputs and production from as many orchards and packhouses as practical in order to make the carbon footprint robust and representative. Ideally the data needs to be collected annually, as Sustainable Winegrowing NZ has just initiated through their WiSE Scorecard, in order to better track change over time.

The 2019 study found packhouse electricity was 55% higher compared to 2010. The 2019 figure is considered more robust due to the larger number of packhouses analysed and does not necessarily reflect a change in energy intensity. Despite this difference, the emissions associated with electricity use were marginally lower due to a reduction in GHG emissions per kilowatt hour of electricity (driven by changes in the electricity grid mix) between the different years. Emissions due to transport were higher, although it is still one of the smallest contributors to total packhouse emissions. Refrigeration emissions were 70% lower. This is due to the large decrease in leakage rates from the coolstore equipment. Again, it is important to use caution around these figures, as the 2010 report's refrigerant emissions were based on a single packhouse. Reporting of accidental leaks can have a significant impact.

Packaging emissions were 73% higher in the 2019 assessment than the original values in the 2010 report. The 2010 results were based on broad assumptions. Subsequently, Catalyst conducted a detailed packaging LCA study (Catalyst, 2012) with the findings used in the 2019 report.

Emissions from RSE workers, which account for approximately 1% of the total packhouse/coolstore emissions, were lower for the 2017 season due to differences in the origins of the workers. Non-RSE emissions again were not included due to the absence of data; their addition is expected to make a negligible difference to total emissions.

Overall, the packhouse emissions values are now considered more robust. The reduction of 28% between the 2021 and revised 2010 values reflects a lower electricity emission factor, offsetting higher electricity intensity - and lower refrigerant losses.

5.2.3 Shipping

Shipping is the single largest component of the kiwifruit carbon footprint. It is also highly influenced by methodology around how to treat back haulage - and the shipping emission factor (reported as $kgCO_{2eq}/t.km$). The 2019 kiwifruit study used primary data - supplied by the Zespri Shipping Team. As described in Section 2.9, where available, primary data should be used. This was the approach used in this study for kiwifruit.

In the absence of new data, the 2017 port emissions were assumed to be the same as 2010. The port stage accounts for only 1% of the total emissions, with most of the emissions (90%) arising from the transport of fruit to the port.

The shipping methodology for kiwifruit involved applying the LCA emission factor for IFO 380 shipping fuel to the total fuel used in a round trip (inward and outward) to the import port at Zeebrugge in Belgium. Total weight of kiwifruit and associated packaging carried by the ship was then used to calculate the emissions per kilogram of kiwifruit transported. This included sending fruit by both reefer container and reefer ships. The split was approximately 70% reefer containers and 30% reefer ships.

The 2019 report, using 2017 primary shipping data, showed emissions were 22% lower than the revised 2010 figure, predominantly due to a higher proportion of larger more efficient container ships being used in 2017. Shipping emissions were found to account for 70% of the total emissions to the German retailer.

Had the kiwifruit supply chain been modelled using the shipping methodology described in Section 2.9 for reefer containers and using a 1635 TEU sized container ship (Zespri supplied size, not the median sized ship leaving NZ) then total emissions to the retailer would have been 9.5% lower (having also adjusted the primary data to be 100% reefer containers) and shipping would account for 63% of total emissions.

5.2.4 Waste

Waste fruit has a significant impact on a product's carbon footprint, particularly if the wastage occurs at the consumer end of the supply chain.

It is extremely important to correctly account for waste throughout the supply chain - and consequently use a multiplier to account for each stage's emissions based on the functional unit.

If the retailer is the system boundary (excluding consumer waste), then 1.12 kilograms must be grown for every kilogramme of fruit delivered to the retailer. This includes fruit sent to domestic consumers, stock food, and waste.

The updated analysis of the supply chain in this study included zero orchard fruit waste (i.e., orchard GHG emissions were divided by the quantity of fruit harvested and sent to the packhouse) and 4.7% fruit waste at the packhouse. There were also other small losses across the supply chain, including 1% during shipping (water loss), and 1% offshore fruit losses during repacking.

To give a sense of how large an impact the consumer has on kiwifruit's carbon footprint the 2019 study used the consumer as the system boundary (1 kg kiwifruit consumed). Consumer waste at 30%, including skin, meant that 1.60 kg of fruit must be grown for every kilogram of fruit consumed. This compared to an estimate of 1.23 kg in the original 2010 study when consumer waste was assumed to be just 10%. The difference is predominantly due to a change in the consumer waste assumption, going from 10% in the 2010 study to 30% waste (edible and inedible) in the 2019 study. On reflection, the 10% figure was a significant underestimate when the inedible component (skin) alone is approximately 18%. The WRAP (2018) study found that 12% of the edible part was also wasted. This is considered low and would be much higher in more commodity-type fruit.

The results presented in this study use a system boundary ending at the German retailer, so consumer wastage of purchased fruit has no bearing on these results.

5.3 Implications for the carbon footprint results

The 2010 original 'cradle to consumer' carbon footprint study was mainly presented in terms of tray equivalents consumed. In 2019 this was changed to 1 kilogram of Green kiwifruit consumed in Germany. This is in line with the Product Category Rules (PCR) for fruit and nuts, which states with regard to the functional unit: “The declared unit shall be 1 kg of product, including its packaging (the weight of the packaging is not included in this 1 kg) and the non-edible parts.” (EPD, 2019).

In this study, the functional unit is 1 kg delivered to the German retailer. To allow for meaningful comparisons across the different studies, the functional unit has been changed on the earlier studies.

The 2010 study, adjusted for the current functional unit, found the carbon footprint of kiwifruit was 1.12 $kgCO_{2eq}/kg$ at the retailer. Based on this study, emissions have increased to 1.24 $kgCO_{2eq}/kg$ at the retailer. However, when compared to the revised 2010 study, which incorporated more accurate packaging and primary data shipping emissions, there has been a 24% decrease in the carbon footprint of kiwifruit delivered to a retailer in Germany, from 1.64 to 1.24 $kgCO_{2eq}/kg$ at the retailer.

The 2010 study (Mithraratne et al., 2010) indicated the significant variability in data for nitrogen fertiliser production and use, coolstore energy use and refrigerant leakage, refrigerated shipping, and changes in soil carbon on orchards. This has been supported by subsequent studies (Gentile et al., 2021; Gentile et al., 2016; Holmes et al., 2014; McLaren et al., 2008; Müller et al., 2015; Page, 2011; Robertson et al., 2014).

Table 5.1: Updated Cradle to Consumer Carbon Footprint for 1 Kilogram of Kiwifruit to the retailer in Germany Compared to Carbon Footprint from the Original and Revised 2010 Values (adjusted for this study's functional unit - to the retailer)

Stage	2021		Revised 2010	Change (2021 – R. 2010)	Original 2010
	$kgCO_{2eq}/kg$	%	$kgCO_{2eq}/kg$	%	$kgCO_{2eq}/kg$
Orchard	0.070	5.7%	0.156	-55%	0.162
Orchard - capital	*0.024*	*1.9%*	*0.031*	-22%	*0.032*
Packhouse	0.151	12.2%	0.210	-28%	0.141
Port	0.011	0.9%	0.012	-3%	0.011
Shipping	0.874	70.4%	1.116	-22%	0.636
Repackaging	0.037	3.0%	0.038	-3%	0.049
Retail	0.073	5.9%	0.073	0%	0.092
Total	**1.241**	**-**	**1.635**	**-24%**	**1.123**

Note: all studies have been updated to use the same functional unit $kgCO_{2eq}/kg$ of fruit at the retailer

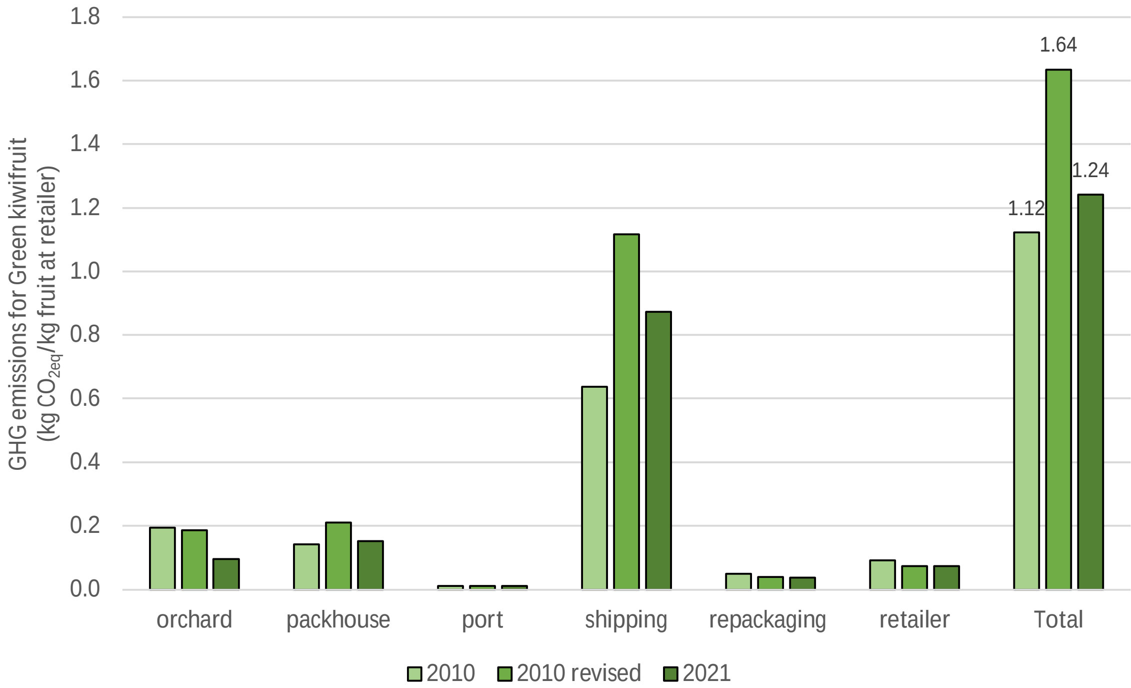

Figure 5.2: Updated Carbon Footprint for 1 Kilogram of Kiwifruit at the Retailer in Germany Compared to the Original and Revised 2010 Values (adjusted for this study's functional unit)

Figure 5.3 shows the proportion of carbon footprint attributable to each life cycle stage. Shipping is the most significant life cycle stage, accounting for 70% of all GHG emissions. Outside of shipping, the next most significant stage is at the packhouse (12%), followed by the orchard (8%). If consumer emissions and their associate wastage were included, then this would be the second-largest emission source.

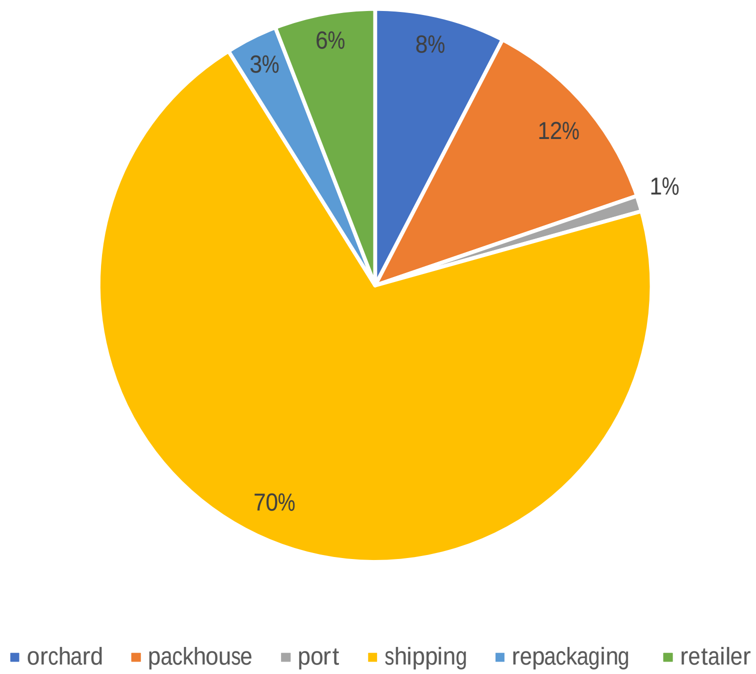

Figure 5.3: Proportion of Carbon Footprint due to Each Life Cycle Stage – kiwifruit to retailer in Germany.

6. Grapes and Wine: Changes in Practices and Productivity, and Implications for the carbon footprint results (Sarah McLaren and Louise Bullen)

6.1 Changes in practices since last studies

6.1.1 Vineyard practices

An updated carbon footprint assessment has been undertaken based on original data from a wine carbon footprint study by Greenhalgh et al. (2008). The original assessment was a case study of an individual winery using data provided by Wairau River Wines for a 750 ml bottle of 2007 Wairau River Sauvignon Blanc. Updated data on vineyard practices for this specific winery were not available to directly assess any changes in practice at this vineyard since the original study. Therefore, and in line with the remit for this report, the industry averaged data were used to estimate likely changes in practice since the original study.

A recent study commissioned by New Zealand Wine (Andrews, 2020) included a summary of chemical use on New Zealand vineyards from winery scorecards in terms of an average application of active ingredient per 100m. This data was applied to the length of the vineyard referenced in the original report to estimate current herbicide, insecticide, and fungicide use. No data were identified on current average fertiliser use, and therefore the original fertiliser quantities were retained.

Updated emissions factors for agrichemical inputs have also been used where available based on the recommendations in Section 2. These updated emission factors are generally higher for fertiliser inputs but lower for herbicides compared to the emission factors used in the original study. Updated emission factors for the use of vineyard equipment and transport of vineyard inputs are generally lower than those used in the original case study.

The combined effect of updated agrichemical use data, revised emission factors and productivity improvements (Section 6.2) is a 20% reduction in the emissions associated with fertiliser use and a 35% reduction in the emissions associated with herbicides, insecticides, and fungicides compared with the original study. This amounts to a 28% reduction in the life cycle GHG emissions for the vineyard stage.

6.1.2 Winery

In the original case study, electricity use in the winery was the source of 17.6% of the total life cycle GHG emissions. A recent report by Barber et al. (2020) analysed electricity use and intensity based on New Zealand winery scorecards and concluded there had been a 22% reduction in electricity use between 2012 and 2020 based on two-year rolling averages. This reduction in electricity use has been applied to the winery stage of the original wine case study. However, it is worth noting that the electricity use in the original case study is relatively high compared to the average electricity intensity figures found by Barber et al. (2020).

Emission factors for electricity have also decreased significantly since the original case study, which was based on the 2007 electricity generation mix. These updated values have been applied in addition to the reduction in electricity use.

Estimated reductions in electricity intensity and use of an updated electricity emission factor result in a 38% reduction in GHG emissions associated with electricity use in the winery.

6.1.3 Packaging

The revised carbon footprint is based on the same bottle type and weight (0.565 kg per bottle) as used in the original study. Due to the significant contribution of the wine bottle (28.4% of life cycle emission) in the original study, a number of potential reduction measures in relation to wine packaging were identified by Deurer et al. (2009). These included a reduction in the weight of the glass bottle, use of recycled glass, use of alternative packaging systems (e.g., PET bottles, Tetra Pak), and bulk shipping of wine.

Recent information on the typical weight of New Zealand wine bottles was not available at the time of this study. However, it is understood that information on the weight and source of wine bottles used by New Zealand wineries has recently been added to the annual information collected by New Zealand Wine and will be available later this year (E.Massey, pers comms.)

There has been a significant increase in the proportion of New Zealand wine shipped in bulk and bottled closer to export markets in recent years. In 2019 approximately 40% of New Zealand wine exports were shipped in bulk and continued growth in bulk shipping is expected (T.Chilala, pers comms). Bulk shipping can significantly reduce distribution GHG emissions due to the reduced weight however these benefits may be partially offset by higher GHG emissions associated with bottling depending on the location of bottling and associated emissions.

The original study utilised an emission factor for packaging glass from the ecoinvent database. The equivalent updated ecoinvent emission factor is used in the revised assessment. This more recent emission factor is 36% higher than the value used in the original study. The impact of this updated emission factor and alternative emission factors for glass are discussed further in Section 6.3.

6.1.4 Shipping

The distribution component of the original study was based on shipping to a range of countries which reflected the export destinations of the specific case study wine. The majority (60%) of wine in the original case study was exported to Australia, and the UK and USA also received significant quantities.

The most important markets for New Zealand wine in 2019 were the USA (31%), the UK (25%), Australia (20%), continental Europe (9%) and Canada (7%), making up a total of 92% of New Zealand wine exports (Plant & Food Research, 2021). In 2007, these same countries also received 92% of New Zealand wine exports, although the proportion of wine sold to the closest market of Australia was higher in 2007 (26%) than in 2019 (20%). It seems likely then that there has been an overall increase in emissions associated with shipping New Zealand wine to overseas markets due to a reduction in the Australian market and no significant increase in nearer markets such as Asia.

On the other hand, more recent emission factors for truck transport and container ships are lower than the emission factors used in the original study. This reflects improved efficiency in transport and shipping emissions since 2007.

For the updated study, two distribution scenarios were modelled. One scenario represents shipping a bottle of wine to the USA and the other to the UK. Distribution emissions also include land-based transport emissions from the winery to a New Zealand port and transport from the receiving port to the retailer.

6.1.5 Types of product

The original study was for a specific product, namely a bottle of 2007 sauvignon blanc, and therefore the updated carbon footprint assessment is also based on a bottle of sauvignon blanc.

6.2 Changes in productivity

There is considerable variability in the productivity of New Zealand vineyards from year to year, as is typical for perennial crops (Section 1.1). During the 10-year period from 2010 to 2019, the average productivity of New Zealand sauvignon blanc per hectare varied from 8.0 t / ha in 2012 to 15.4 t / ha in 2014 (Plant & Food Research, 2021). To estimate the change in productivity in New Zealand sauvignon blanc production since the original case study was undertaken, a comparison was made of the average productivity (tonnes/ha) over the four years 2005-2008 with the average productivity over the four years 2016-2019 based on data from Plant & Food Research (2021). A 27% increase in productivity of sauvignon blanc occurred between these two four-year periods. This average increase in productivity was applied to the original case study.

6.3 Implications for the CF results

The original ‘cradle to retailer’ carbon footprint for a bottle of sauvignon blanc from the case study winery has been updated based on more recent emission factors and estimated improvements in productivity, vineyard agrichemical inputs and winery electricity intensity as described in Sections 6.1 and 6.2. Figure 6.1 shows the ‘cradle to retailer’ carbon footprint for a bottle of sauvignon blanc produced in Marlborough and shipped to a range of destinations (Section 6.1.4) from the original study and the updated carbon footprint based on shipping the bottle of wine to either the USA or the UK. The updated ‘cradle to retailer’ carbon footprint for a bottle of wine shipped to the USA is 1.090 $kgCO_{2eq}$/bottle and for a bottle shipped to the UK is 1.263 $kgCO_{2eq}$/bottle. This compares to 1.216 $kgCO_{2eq}$/bottle in the original study.

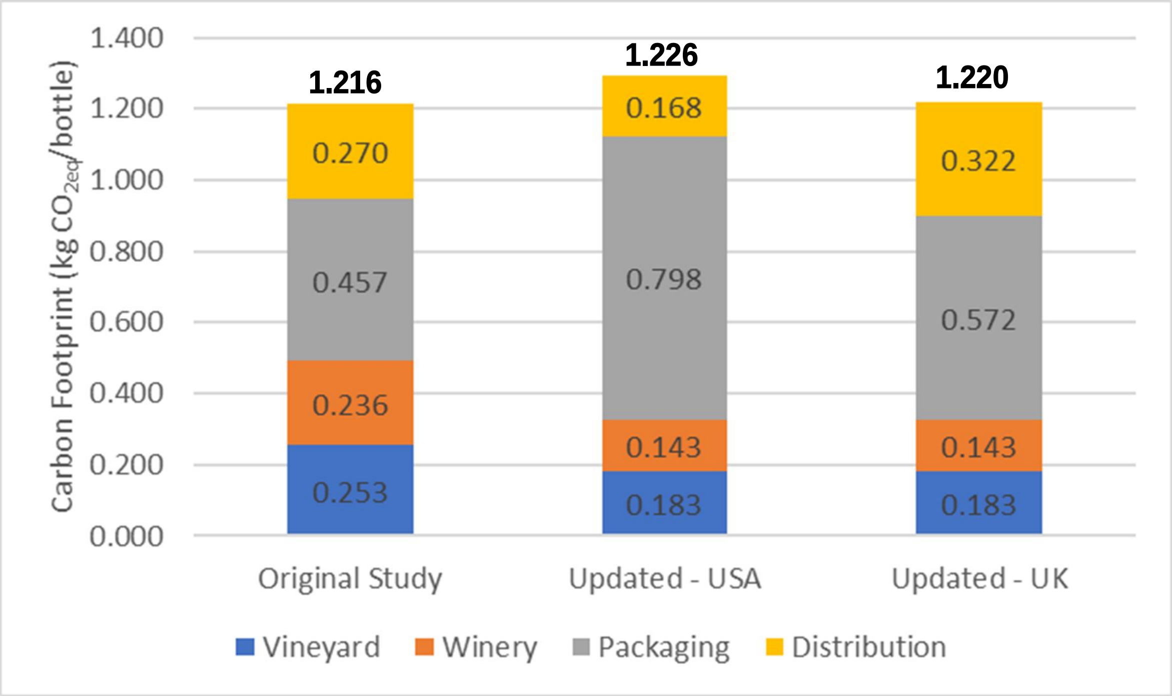

Figure 6.1: Updated Cradle to Retailer Carbon Footprint for a Bottle of Sauvignon Blanc Shipped to the USA or UK Compared to Carbon Footprint from Original Study

Figures 6.2 and 6.3 show the proportion of carbon footprint attributable to each life cycle stage in the updated assessments when shipped to the USA (Figure 6.2) and UK (Figure 6.3). The relative proportion of the carbon footprint due to distribution for the bottle shipped to the UK is 28.8% compared to 17.5% for a bottle shipped to the USA. Consequently, the proportion due to other life cycle stages is lower for the bottle shipped to the UK. The packaging stage is the most significant life cycle stage regardless of the destination.

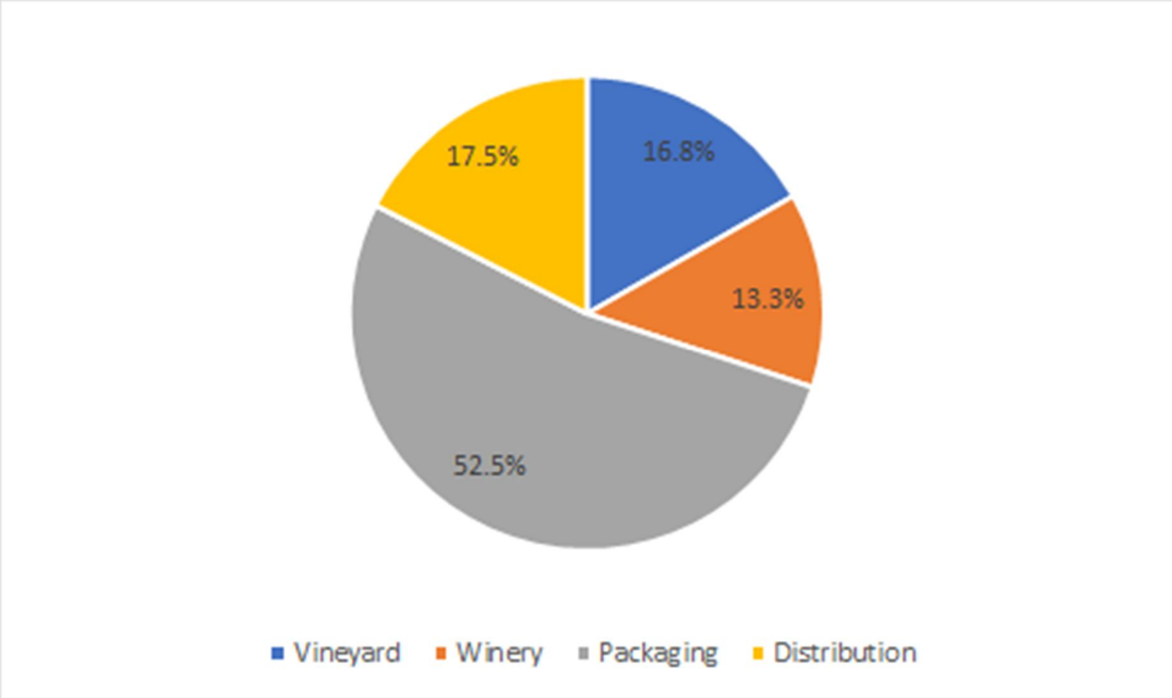

Figure 6.2: Proportion of Carbon Footprint due to Each Life Cycle Stage - Bottle of Wine Shipped to the USA

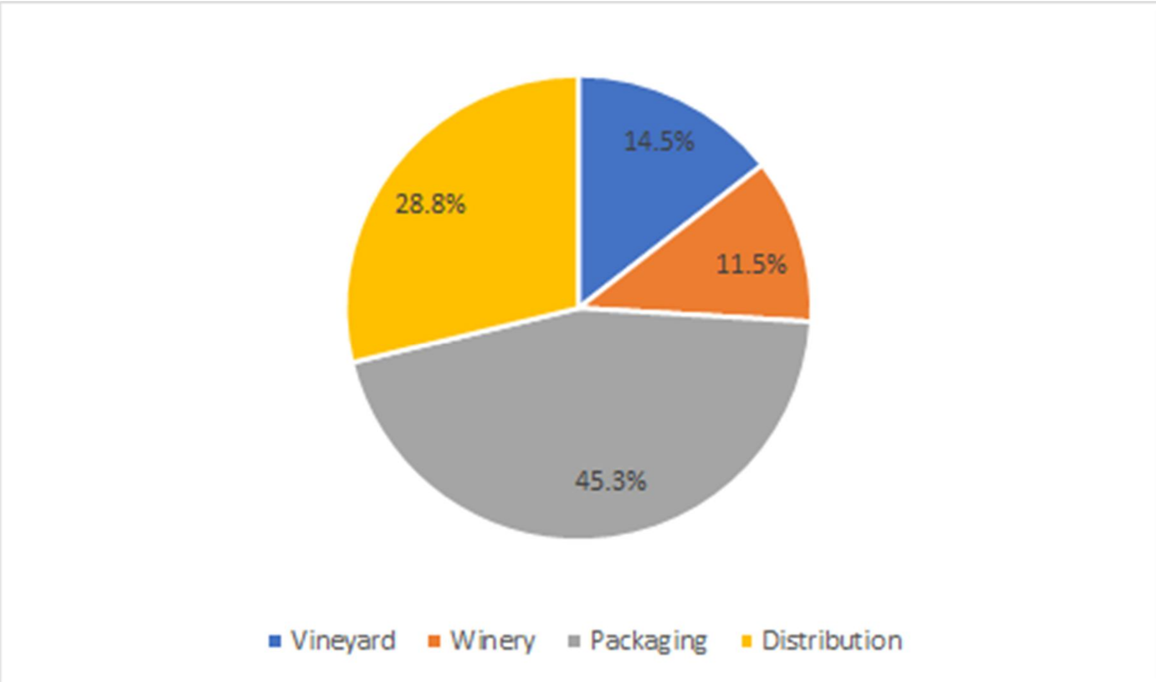

Figure 6.3: Proportion of Carbon Footprint due to Each Life Cycle Stage - Bottle of Wine Shipped to the UK

The carbon footprint associated with the vineyard stage decreased by 28% compared to the original study due primarily to the estimated improvement in productivity since the original study was undertaken as well as the incorporation of more recent agrichemical data and the use of updated emission factors.

The carbon footprint associated with the winery stage decreased by 39% compared to the original study due to a combination of a lower electricity emission factor and estimated improvements in electricity intensity. However, these improvements in the vineyard and winery stages are considered indicative only as these values were estimated from industry average data, which may not be representative of the original case study.

The emissions associated with packaging increased by 25% in the revised assessment compared to the original study. This is predominantly due to the impact of an updated emission factor for the production of packaging glass. Although both the original and revised emission factors for the wine bottle are based on green packaging glass production from the ecoinvent database, the more recent value is 36% higher than the emission factor used in the original study. The updated value is comparable with emission factors used in other carbon footprint studies of New Zealand wine (J. Andrews, pers comm). The original study did not include any allowance for the return shipping journey and this approach has been maintained in the updated study in accordance with recommendations for dry containers (Section 2.9).

The GHG emissions associated with the wine bottle production comprise the single biggest factor influencing the overall cradle-to-retailer life cycle of a bottle of wine. Therefore, the choice of emission factor has a significant impact on the overall life cycle emissions. There is a wide range of emission factors available in the literature and in LCI databases representing the production of packaging glass. Although it is significantly higher than the value used in the original study, the emission factor used in this study is near the lower end of this range. For example, the Glass Packaging Institute (2010) identified a ‘cradle to cradle’ GWP value for North American container glass of 1.25 $kgCO_{2eq}/kg$ of glass. The use of a higher emission factor such as this value would result in a further 40% increase in the packaging stage carbon footprint.

The most significant factors that determine the actual emissions associated with bottle production are the efficiency of the glass production process (US Department of Energy, 2017), the proportion of recycled glass cullet used, and the carbon footprint of the energy supply used in the production process. Specific information on New Zealand wine bottle production and use was not readily available, which makes it challenging to identify an appropriate emission factor for wine bottles used by New Zealand wineries. Data on the mix of locally produced and imported bottles used, the proportion of recycled content, and the emissions associated with specific suppliers would enable a more accurate estimate of the carbon footprint associated with the packaging stage.

As noted above, the life cycle GHG emissions for wine shipped to the UK are 16% higher than wine shipped to the USA. Indeed, and going one step further, Deurer et al. (2009) suggested that one of the long-term emission reduction options was to reorientate New Zealand wine exports to Pacific Rim and Asian markets. It should also be noted that the weight of the bottle is a significant determinant of the distribution of GHG emissions; light-weighting the bottle or shipping wine in bulk would contribute to its reduction.

Overall, the revised ‘cradle to retailer’ carbon footprint for a bottle of sauvignon blanc is comparable to the carbon footprint calculated in the original case study. This is because the more efficient vineyard and winery operations are offset by the higher packaging-related GHG emissions. However, it is important to note that this comparison is based on just one winery, over a single season, and for one variety of wine, assessed over ten years ago.

The collection of up-to-date data from a representative range of New Zealand wineries over a number of seasons would enable a more robust and representative assessment of the vineyard and winery stages of the life cycle. Similarly, as noted above, improved data on wine bottles and other packaging systems used in New Zealand and their production would add robustness to the assessment of the packaging and distribution stages.

6.3.1 Summary

The updated 'cradle to retailer' carbon footprint of New Zealand wine is 1.090 $kgCO_{2eq}$/bottle when shipped to the USA and 1.263 $kgCO_{2eq}$/bottle when shipped to the UK. These values are comparable to the results of the earlier study by Greenhalgh et al. (2008) and are within the range of results for similar international wine carbon footprint studies.

Compared to the earlier study, improvements in the vineyard and winery stages are largely offset by increases in the packaging stage, but this is strongly influenced by assumptions regarding the emissions associated with the production of the wine bottle. The wine bottle remains the single most important input to the carbon footprint of wine; this result is consistent with both previous New Zealand studies and overseas studies.

The revised assessment is largely based on study data that is now over ten years old and represents the emissions for a single winery and vintage. This limits the conclusions that can be drawn on the carbon footprint of the wider wine industry from this study. A comprehensive study of the carbon footprint of New Zealand wine utilising recent data over a range of wineries and seasons as well as New Zealand specific data on packaging systems would enable a more robust and representative assessment of the carbon footprint of New Zealand wine.

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