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Eutrophication and climate change impacts of a case study of New Zealand beef to the European market

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Abstract

Objective: Beef production in the Lake Taupō region of New Zealand (NZ) is regulated for nitrogen (N) leaching. The objectives of this study were to 1) evaluate the implications of nitrogen emission limitations on eutrophication and climate change impacts of NZ beef through its life cycle to a European market and uniquely link it to 2) estimation of the reduction in these impacts that can be funded by the consumer's willingness to pay (WTP) a premium for a low environmental-impact product.

Method: The cradle-to-market Life Cycle Assessment (LCA) of NZ beef on the European market included beef production on farms, meat processing, packaging and transport stages. Various beef production systems in the Lake Taupō region were modelled: farm systems with and without regulated N leaching limits in place (using N fertiliser inputs of 0 and 100 kg N/ha/year respectively) using suckler beef or beef derived from surplus calves from a dairy farm. The FARMAX model was used to model farm productivity and profitability under these various scenarios, whereas the OVERSEER® model was used to model field/farm emissions (N, phosphorus (P)) and the NZ greenhouse gas (GHG) Inventory model was used to estimate total GHG emissions. Eutrophication and climate change impacts of NZ beef to the European market were calculated using recent regionalised LCA indicators. We estimated freshwater and marine eutrophication impacts of European beef using published N emissions to water and air. We estimated the European consumer's WTP

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for beef with positive environmental attributes based on a meta-regression analysis based on 21 published studies and compared farmer’s profit for the farm system scenarios.

Results: When using common P-driven eutrophication indicators, the farms using 100 kg fertiliser-N/ha/year appeared to have a lower freshwater eutrophication impact than farms using no N fertiliser, which is in contradiction with the local freshwater policy for N regulations. When the contribution of both N and P were accounted for, the farms using no N fertiliser had the lowest estimated impact. Comparison with published environmental footprint of beef from Europe showed lower climate change and eutrophication impacts for NZ beef, thus showing potential positive environmental attributes for NZ beef. The European consumer’s WTP (32% price premium) for such a beef product with low environmental impacts could offset the cost to farmers for implementing the reduction of N emissions.

Conclusions: Bridging the gap between local freshwater policy and LCA indicators starts by considering both P and N emissions and impacts. Combining an environmental LCA with an economic analysis revealed that the consumer willingness to pay could compensate for the environmental cost of protecting the lake that currently only the farmers are bearing.

Keywords

LCA, nitrogen, phosphorus, freshwater, environmental attributes, carbon footprint

1. Introduction

Nutrient inputs to waterways can lead to undesirable algal growth. This phenomenon, called eutrophication, is a major issue worldwide (Khan and Mohammad 2014). Research has shown a slow temporal decline in water quality in the largest lake (Lake Taupō) in New Zealand (NZ). Although the lake is almost pristine, measurements have shown a moderate increase over time in nitrogen (N) (Vant 2013). Since Lake Taupō has high environmental, economic and cultural values (Petch et al. 2003), land use and farm management practices are now regulated to protect its water. Governmental policy has set a maximum N leaching value for each individual farm in the catchment (WRC 2019), where sheep and beef farming dominate (Vant and Husser 2000).

The environmental regulations in the Lake Taupō catchment require farmers to restrict some of their farming practices. The implementation of a maximum N leaching value comes with a cost to farmers; previous studies of Taupō farm scenarios have shown that N-regulation results in a lower profit for farmers (e.g. Thorrold et al. 2001). One way of compensating for this restriction (and subsequent loss in profit) is to pass on the costs of compliance to consumers as shown in Ledgard et al. (2016). The Taupō Beef & Lamb company was set up by farmers that are producing beef from low input farm systems, who market it to the local restaurants and national retail outlets and charge a price premium for this "low environmental footprint" beef (Taupō Beef and Lamb 2016). Because NZ is a major exporting nation, it would be interesting to analyse if a similar approach could be used for NZ beef meat sold overseas. The endorsement of NZ products overseas could go beyond the image of pasture-based and free-range products by quantifying their low footprint using internationally recognized indicators and communicating this information transparently to consumers (e.g. through labelling). Previous studies have shown that consumers value the environmental credentials of NZ products (in India, China and the UK), but consumer preferences and their willingness to pay for different food attributes differs across countries. As a result, it may be beneficial for NZ producers to certify NZ products for certain attributes (Saunders et al. 2013).

Promoting environmental attributes requires demonstrating them in a quantitative, transparent and reproducible way. To ensure a consistent measure of environmental performance internationally, the European Commission (EC) proposed the Product Environmental Footprint (PEF) methods based on the life cycle assessment (LCA) of products (EC 2019). LCA is a standardised (ISO 2006a, 2006b) multicriteria decision support methodology for the environmental assessment of products. After a pilot testing phase (including a NZ dairy product), the EC is now exploring the implementation of PEF in policies. PEF indicators for eutrophication are based on a European model that is not appropriate for NZ (Payen and Ledgard 2017), but we can expect these recommendations to change in the future for two reasons. First, Life Cycle Impact Assessment (LCIA – the phase of LCA that concerns the modelling of environmental impacts) indicators are constantly improving as more research becomes available (for a review of eutrophication indicators see

Henderson 2015, for a comparative case study see Payen and Ledgard 2017). Second, the GLAM program (Global Guidance for Life Cycle Impact Assessment Indicators and Methods) from the United Nations Environmental Program/Society of Environmental Toxicology and Chemistry (UNEP/SETAC) Life Cycle Initiative released its latest recommendations for methodology changes based on an international consensus building process (UNEP 2019). GLAM identified the "current best available practice" for a variety of impact indicators, aimed at life cycle assessment practitioners and method developers. The global importance of these impact categories is recognized in the Sustainable Development Goals (UNEP 2019). The indicators recommended for aquatic eutrophication are based on globally-valid models (Helmes et al. 2012, Cosme and Hauschild 2017), thus relevant for application in NZ. However, the indicator for freshwater eutrophication has shortcomings. Although it is spatially-explicit (modelling catchment specificities when quantifying potential impacts), it only accounts for the contribution of phosphorus (P). Even though it is the availability of P that controls eutrophication in many Northern Hemisphere lakes with excess N, this is not the case in many NZ lakes because excess of N is uncommon (Vant and Huser 2000). Consequently, P-driven indicators capture only a part of the problem for freshwater bodies such as Lake Taupō where algal growth is co-limited by N and P (Payen and Ledgard 2017).

Because this co-limitation is occurring in many countries, another recommendation from the GLAM program is the development of N characterisation factors to account for the contribution of N to freshwater eutrophication (UNEP 2019). Freshwater eutrophication characterisation factors have now been developed for both N and P (Payen et al. 2020). These characterisation factors represent the transport and attenuation of dissolved inorganic N and dissolved inorganic P within a river basin and distinguish nutrient emissions from soil and emissions to freshwater. The fate processes modelled include nutrient attenuation from land to stream, in the river network, in reservoirs and lakes, and those associated with water consumption. Characterisation factors were calculated at a river basin resolution with a global coverage.

Payen and Ledgard (2017) showed that the Lake Taupō catchment is a good illustration of a discrepancy between local policy and product-oriented environmental impact indicators (based on LCA). There is an inconsistency between the local environmental policy that regulates N, with the currently accepted indicators of freshwater eutrophication that focus on P. As a result, this work aimed to analyse how local freshwater quality policy and LCA eutrophication indicators can be reconciled using a case study in the N- and P-limited Lake Taupō catchment. Thus, this paper addresses both N and P nutrients when assessing freshwater impacts. Since water and nutrients eventually drain to estuaries, we also considered marine eutrophication impacts. In addition, we assessed climate change to identify potential impact-shifting and to build on previous carbon footprint studies of NZ beef (Lieffering et al. 2010).

Recently published LCA studies of beef meat provide estimates of global warming potential impacts but only a few include calculation of eutrophication potential impacts (Bragaglio et al. 2018, Presumido et al. 2018). There is a need to address more systematically this impact category and to do so by using the latest LCIA methods recommended by the GLAM program.

The main objective of this novel study was to **evaluate the implication of nitrogen emission limitations on eutrophication impacts of NZ beef meat sold on the European market and to estimate the reduction in eutrophication impacts that can be funded by the consumer's willingness to pay (through labelling).** Freshwater eutrophication impacts of NZ beef produced in a range of farm production scenarios were estimated using indicators recommended by the GLAM program and a new method that accounts for the contribution of both N and P. Impacts estimated with different indicators were compared. Impacts from NZ beef were also compared with those from average European beef. The farm costs of reducing emissions were assessed against the consumer's willingness to pay (WTP) for beef products with a low environmental footprint. The main novelty of this work was to combine an environmental LCA with an economic analysis.

2. Materials and Methods

2.1. LCA goal and scope

To calculate potential impacts of NZ beef on the European market, we performed a cradle-to-market LCA (ISO 2006a, 2006b) (i.e., from raw material extraction to the market entrance gate), which included all inputs for beef production (under various farms system scenarios - section 2.2.1) and for post-farm processes (meat processing, packaging and transport to Europe - section 2.2.2). The system boundaries are illustrated in Figure 1, and the main data sources are summarised in Table 1.

The functional unit was 1 kg of live-weight (LW) equivalent on the market for the sake of comparability with other studies (and due to variability in literature of the conversion factors from live-weight to carcass-weight and meat). Where conversions were made, conversion factors from beef meat to carcass weight were 58% for European beef (Weiss and Leip 2012; Lesschen et al. 2011) and 54% for NZ beef (West 1993).

In this study, the environmental focus is on climate change (a global impact) and eutrophication potential (regional impacts).

Table 1. Overview of data sources for the main components of the NZ beef life cycle inventory

	Main data sources
Farm characteristics and inputs	Primary data (Ledgard et al. 2016, Beef+LambNZ 2018)
Animal dry matter intake	Primary farm data and FARMAX model (Webby and Bywater 2007)
Farm emissions (N, P and GHG)	OVERSEER model (Wheeler et al. 2007, 2011), NZ GHG Inventory (MfE 2018)
Farm profit and productivity	FARMAX model (Webby and Bywater 2007)
Processing plant inputs (energy, packaging…)	Primary data (Lieffering et al. 2010) and Ecoinvent 3.4 (Wernet et al. 2016)
Processing plant waste water	Lieffering et al. (2010)
Transport (to and from processing plant)	Primary data on transport (Beef+LambNZ 2018) and Ecoinvent 3.4 (Wernet et al. 2016)

2.2. Beef supply chain description

2.2.1. Farm systems modelling

NZ farms systems are based on year-round grazing of perennial pasture (ryegrass and clover), with reliance on clover fixation of atmospheric N_2 as the main external N input. Beef is derived from farm systems including breeding and/or finishing systems with cattle from traditional beef breeds (e.g. Angus, Hereford) or from the dairy industry. An average beef and sheep farm in the Taupo/Waikato regions is on rolling to steep hill country, with cattle and sheep grazed together and with calving/lambing in early spring so increased feed demand matches the seasonal pattern of pasture growth. Feed intake is from grazed pasture with <5% from pasture silage or hay made during periods of surplus and fed out in winter. In this study the average farm was based on survey data from Beef+LambNZ (2018).

As illustrated in Figure 1, three NZ farm systems from the Lake Taupō catchment were analysed in this study:

- "B" - an average beef and sheep breeding and finishing farm (i.e. animals are finished to slaughter weights on the same farm),
- "B.F" - a beef finishing farm, with one-year-old (yearling) cattle supplied from an average beef and sheep breeding farm,
- "D.B.F" - a beef finishing farm, with weaned calves supplied from a dairy farm and reared to yearlings on a beef breeding farm,

For each NZ farm system, two N fertiliser input scenarios were analysed since there is a strong link between increased N fertiliser use and increased N leaching risk (e.g. Ledgard et al. 1999):

- "0-N" - No fertiliser-N inputs for both the breeding and finishing farms (current N leaching constraints in place (i.e. N regulation)).
- "100-N" - Urea applications to pasture at a total of 100 kg N/ha/year for the breeding and finishing farms (i.e. assuming no N regulation).

Figure 1. Flow diagram for the NZ Beef production, meat processing and transport to the European market showing the main inputs and outputs modelled. Six beef production scenarios were modelled based on farm systems and nitrogen (N) fertiliser input. **B**: Average beef and sheep breeding & finishing farm; **B.F**: Beef finishing farm with cattle supplied from an average beef and sheep breeding farm; **D.B.F**: Beef finishing farm with weaned calves supplied from a dairy farm and reared to yearlings on a beef breeding farm. For each farm system, two N fertiliser input scenarios were analysed: 0-N (0 kg N/ha) and 100-N (100 kg N/ha).

Modelling of the finishing farm system was based on a real cattle finishing farm (Ledgard et al. 2016) in the Lake Taupō catchment (120 ha flat-rolling grassland) that purchases yearling beef cattle and sells them at about 2-years-old. Lake Taupō catchment farms have long-term pastures of perennial grasses and white clover on a coarse-textured pumice soil under relatively high rainfall (1300+ mm/year) and are prone to N leaching. The breeding farm system model was based on an average beef and sheep farm from the wider Waikato/Bay of Plenty region (from Beef+LambNZ 2018). Primary data for inputs on farm were derived from an average of three years (2015-2016, 2016-2017 and 2017-2018) for the finishing farm and from the year 2015-16 for the breeding farm. Farms were modelled using the farm production and economics model FARMAX (Webby and Bywater 2007) to estimate animal pasture dry matter (DM) intake and farm profit. Heifers and steers are sold for meat processing at between about 450 and 650 kg LW (varying with time of year) from all farm systems with cull breeding cows sold at about 500 kg LW. More details on the modelling of each farm system and scenario is provided below, and Table 2 shows key farm inventory data.

The **B - 0-N** scenario is an average beef and sheep breeding and finishing farm from the Taupō catchment. Farm emissions related to cattle were calculated using allocation based on DM intake by cattle and sheep. In this case it resulted in 51% of emissions allocated to cattle. No N fertiliser was used on the farm. The **B - 100-N** scenario is the same farm as described for B – 0-N, but the farm received 100 kg urea-N/ha/year and the increased pasture growth was used to calculate the increase in cattle numbers on the farms. The **B.F - 0-N** scenario is a system with two farms where a breeding farm supplies cattle to the finishing farm. The breeding farm (a Beef+LambNZ Class 4 farm using Taupō area pasture growth and farm survey data for the regions Waikato/Bay of Plenty) was modelled to supply all required cattle to the finishing farm, at the appropriate time of year and weight (varying from 8-24 months age). Note that the breeding farm also sells some other beef (including cull cows) and sheep meat and wool. For analysis, the breeding farm was set up in the OVERSEER® nutrient budgets model (hereafter called OVERSEER (Wheeler et al. 2007, 2011)) and emissions related solely to the cattle sold to the finishing farm were calculated by splitting the farm into the proportion of land needed to produce beef or sheep, based on DM intake requirements. In this case it was 55% allocated to cattle. Then it was further split up based on the relative amount of LW sold to the finishing farm versus LW sold elsewhere. In this case, 42% was allocated to the finishing farm. No N fertiliser was used on the farms. The **B.F - 100-N** scenario is the same two-farm system described as for B.F - 0-N, except that the finishing and breeding farms received 100 kg urea-N/ha/year and the increased pasture growth was used to increase cattle numbers on the farms. This resulted in 48% of total beef LW sold allocated to the finishing farm. The **D.B.F - 0-N** scenario is a system with three farms, where weaned surplus calves from a dairy farm are transferred to a breeding farm that supplies the finishing farm. The breeding farm (a Beef+LambNZ Class 4 farm based on Taupō area pasture growth and farm survey data for the regions Waikato/Bay of Plenty) was modified so that the required cattle for the finishing farm were derived from surplus calves from an average Waikato dairy farm (data from DairyNZ DairyBase survey farm data for 2015-16; e.g. Ledgard et al. 2019). It was assumed that surplus 40 kg dairy calves were sold to the breeding farm and that they were reared from 40 kg to 100 kg using milk powder and cereal grain based on NZ average data (Muir et al. 2000). They were then fed pasture on the breeding farm for a period to reach the same age and weight before sale to the finishing farm as on B.F. – 0-N. No N fertiliser was used on the farm. The **D.B.F - 100-N** scenario is the same as the three-farm system described for D.B.F - 0-N, except that the finishing and breeding farms received 100 kg urea-N/ha/year and the increased pasture growth was used to increase cattle numbers on the farms (and therefore more surplus dairy calves were sourced from the dairy farm to meet the finishing farm's requirements). Dairy farm emissions were allocated between milk and LW sold for meat (which includes surplus calves) using biophysical allocation (IDF 2015).

For all above-mentioned NZ beef and sheep farm scenarios, the only feed source was from pasture (grazed or silage) and there were no feed crops grown. The total pasture Dry Matter Intake (DMI) per ha was approximately 7.0 t DMI /ha across 0-N farms, and approximately 0.3-0.9 t DMI higher for the 100-N farms. Fertiliser P was applied to pasture as superphosphate at 20 kg P/ha/year to the finishing farm and 17 kg P/ha/year to the breeding farms (based on calculated maintenance requirements).

Table 2. Inventory table for each farm included in the different scenarios

Table 2 shows that farms not using N fertiliser (i.e. under N regulation) have a lower productivity (lower net cattle LW sold) due to their lower pasture production.

2.2.2. Post-farm modelling

The post-farm model consists of the processing of the live animals into meat products and the transport and intermediate storage of the chilled beef before it reaches the European market (Rotterdam assumed as entry port).

The processing stage includes all activities from the farm gate to the finished product at the processing plant: transport to the processing plant, processing energy, use of consumables, packaging and waste treatment. These activities were modelled based on surveyed processing plants (Lieffering et al. 2010) and using the Ecoinvent v3.4 database (Wernet et al. 2016). Transport of the live animal to processor was modelled using a 7.5 to 16 tonne truck. Transport of the finished product from the processing plant to the NZ port was modelled using a refrigerated 7.5 to 16 tonne truck. Shipping from NZ (Tauranga) to Europe (Rotterdam) was modelled using a transoceanic freight ship with cooling reefers.

2.3. Environmental Life Cycle Assessment (LCA)

Environmental impact assessment was performed across the whole NZ beef supply chain (from cradle to the European market) and impacts are expressed per kg LW equivalent at the market. We did not expressed results per kg meat for the sake of comparison with other European beef studies that are at the farm gate.

2.3.1. P, N and GHG emissions

Payen and Ledgard (2017) showed the importance of a site-specific inventory for farm nutrient flows. P runoff and N leaching emissions were estimated for all NZ farm scenarios using OVERSEER, which has been validated against field measurements across NZ (McDowell et al. 2005, Wheeler et al. 2007). P runoff is calculated based on soil, climate, hydrologic conditions, application rates and transport factors (McDowell et al. 2005). N leaching is calculated based on the amount and timing of N excreted by animals, applied fertilisers, and is mainly driven by soil properties and drainage (Wheeler et al. 2011). OVERSEER has been used to define maximum N leaching limits for farms in the Lake Taupō catchment and is used as a tool for setting policy on freshwater management (Ledgard et al. 2009). Ammonia and nitrogen oxide emissions were calculated using NZ-specific emission factors from the NZ GHG Inventory (MfE 2018). N and P emissions in wastewater from meat processing were estimated based on data collected from three processing plants (Lieffering et al. 2010).

GHG emissions were estimated for all NZ farm scenarios based on a tier-2 methodology with NZ-specific emission factors from the NZ GHG inventory (MfE 2018). Intake of pasture by animals was calculated from animal productivity data using FARMAX and this was linked with the Inventory factors to calculate methane emissions. Pasture intake data was combined with NZ average pasture N concentrations (MfE 2018) to calculate excreta-N and this was multiplied by the NZ inventory factors to estimate nitrous oxide (N_2O) emissions. All background GHG emissions (including indirect emissions associated with fuel and fertiliser production and use) were accounted for (Ledgard et al. 2019).

GHG emissions from post-farm stages were based a previous carbon footprint study (Lieffering et al. 2010) and updated for truck types and transport distances.

2.3.2. Freshwater and marine eutrophication impacts

Freshwater eutrophication impacts of NZ beef (all farm scenarios) were calculated using three methods. We applied ReCiPe 2016 (Huijbregts et al. 2016, based on the fate factors developed by Helmes et al. 2012 which is recommended by UNEP 2019) and ReCiPe 2008 (Goedkoop et al. 2009). However, these LCA indicators only focus on P. As a result, we also applied the method developed by Payen et al. (2020), accounting for the contribution of both N and P, at the highest spatial resolution as possible (i.e. using characterisation factors (CFs) at the river basin scale). The spatially-explicit CFs represents the transport and attenuation of dissolved inorganic N and dissolved inorganic P within a river basin, distinguishing nutrient emissions from soil and emissions to freshwater. The fate processes modelled include nutrient attenuation from land to stream, in rivers, in reservoirs and lakes, and any associated with water consumption and were built based on Global NEWS2 attenuation factors (Mayorga et al. 2010). This fate model for freshwater eutrophication is consistent with and complements recent advances in marine eutrophication impact assessment (Cosme and Hauschild 2017). See supplementary material for more details (Figure S1).

In the absence of P attenuation methods, we used a conservative approach to estimate attenuation from farm to freshwater, by assuming all P runoff calculated by OVERSEER was contributing to freshwater eutrophication calculated with ReCiPe 2008 and 2016.

Since water (and associated nutrients) eventually drain to estuaries, we calculated marine eutrophication impacts using the indicator recently developed by Cosme and colleagues (Cosme and Hauschild 2017; Cosme et al. 2017), which focuses on N.

Table 3 provides an insight to the characterisation factors (CFs) applied in this study for a few locations only.

Table 3. Freshwater and marine eutrophication characterisation factors used in this study for nitrate and phosphate emissions to river, for the Waikato region (New Zealand), Europe and a Global average.

2.3.3. Climate change impact

The climate change impact of NZ beef (all farm scenarios) was estimated as the sum of direct and indirect GHG emissions, using a global warming potential of 25 kg $CO_{2\ eq}$/kg emissions for methane and 298 kg $CO_{2\ eq}$/kg N_2O for nitrous oxide (IPCC 2007). Land use was assumed as long-term pasture and therefore no effect of land use change was modelled. Soil carbon sequestration was not accounted for.

2.4. Comparison with European beef

Impacts of NZ beef on the European market were compared with impacts from beef produced in Europe using published LCA studies (Buratti et al. 2019, Bragaglio et al. 2018, Presumido et al. 2018, Leip et al. 2015). We used emissions and impact results published in Leip et al. (2015) since it represented "average European beef". This average European beef was derived from a mix of farm systems with contributions from animal housing, use of brought-in crop feeds and dairy-derived cattle. Leip and colleagues (2015) calculated N and GHG emissions from average European beef using the agro-economic Common Agricultural Policy Regionalised Impact (CAPRI) modelling system (Britz and Witzke 2012). They estimated land use change emissions based on Weiss and Leip (2012) and included C sequestration in managed grassland. Climate Change was calculated in Leip et al. (2015) using the same global warming potentials as in this study (IPCC 2007), thus allowing comparison.

We calculated freshwater and marine eutrophication impacts of European beef using published N emissions to water and air (Leip et al. 2015) multiplied by the relevant CFs at the European scale. Aggregation of Payen et al. (2020) CFs from river basin to European scale was based on an emission-weighted scheme (see supplementary material for more details). It is important to note that the resulting eutrophication impacts should be considered with caution due to differences in the spatial resolution of calculations. Although most emissions occur in Europe, some of them may actually occur in other countries (for feed), and most importantly, the variability of eutrophication CFs within Europe is large. As a result, using a European average CF creates a lot of uncertainty. Since the details of P emissions for beef were not available in Leip et al. (2015), we only calculated eutrophication impacts determined by N.

2.5. Economic analysis

To determine if the consumer's willingness to pay (WTP) may offset the cost to farmers for producing beef with low N emissions, we estimated the European consumer's WTP for beef with a "low environmental footprint" and compared farmer's profits for the different farm system scenarios.

We estimated the European consumer's WTP a price premium for beef products with environmental attributes[1] using a meta-regression analysis (Yang and Renwick 2019). Based on a systematic review of relevant studies, a list of 144 WTP estimates was produced from 21 studies focusing on estimating a price premium for credence attributes associated with beef products. Details of the meta-regression analysis is provided in the Supplementary materials.

For the economic profit analysis, our baseline assumption is that beef produced from farms not using N inputs (the 0-N scenarios) could get a price premium from the market in terms of a consumer WTP for environmental premium (USDA, 2018). However, only a proportion of the WTP could be delivered to farms (31%; with the remaining 69% going to post-farm stages) and we assumed that only the finishing farm could get the premium. For the finishing and breeding system (B.F - 0-N with premium), we take both farms as an entity where the profit of the entity takes into account all the finishing farm's profit and a proportion of the breeding farm's profit (only the profit from products sold to the finishing farm). Similarly, for the finishing and dairy-based breeding system (D.B.F - 0-N with premium), the estimation of the entity's profit considers all the finishing farm's profit, and a proportion of the profit from the beef breeding and dairy breeding farm.

[1] Environmental attributes belong to one category of credence attributes associated with food products consumers could not observe or experience. Environmental attributes are also called environmentally friendly and sustainable attributes that are relevant to environmental concerns, such as water quality and carbon emissions.

In addition, to get the products to potential consumers, finishing farms need to get their beef certified as 'environmentally friendly', and thus carry the cost of certification (up to NZD 7500)[2].

Based on these assumptions and the financial data from FARMAX, we estimated the farm profit before tax (profit per kg LW) for all three types of farm systems and produced nine economic scenarios. Note that contrary to emissions and environmental impacts, allocation of costs and profit to beef (versus sheep) was based on economic value instead of DM intake.

3. Results and discussion

3.1. Eutrophying emissions (N and P)

N emissions were lower for NZ beef compared with average European beef (Leip et al. 2015), even when transportation from NZ to Europe was included, except for ammonia (NH_3) emissions that were higher for the NZ scenarios using 100-N (Table 4). The higher NH_3 emissions seem surprising since there is no animal housing in NZ (usually responsible for a large share of NH_3 emissions from livestock). However, NZ NH_3 emissions estimated from excreta are probably overestimated since an emission factor of 10% was used (based on the NZ GHG inventory, MfE 2018) but a review of this EF (Sherlock et al. 2008) showed that lower values are more probable for animal excreta in grazed systems. In a sensitivity analysis, we used an emission factor of 4% for dung and urine N, based on results from Ledgard et al. (1999), which reduced NH_3 emissions from 82 to 55 g N-NH_3/kg LW for the B -100-N scenario. Ammonia emission factors used in the average European beef study were unclear: N emissions were estimated using the CAPRI model (Leip et al. 2015) where they were calculated following a mass-flow approach (Leip et al. 2014). It is mentioned that country-specific emission factors and abatement measures were accounted for (Klimont and Brink 2004), but the detail was not provided. Evidence of a potential methodological discrepancy is that the nitrate leaching for

[2] This is estimated by using BioGro data. The annual base fee ranges from $5,000 for a domestic primary producer to $10,000 if they sell processed products/cosmetics etc. Here, we use the average $7,500 as an estimate of the annual cost of certification.

European beef is almost twice that for the NZ beef, which seems in contradiction with the much higher apparent NH_3 emissions for NZ beef.

When comparing the various NZ beef scenarios, N emissions per kg LW equivalent at the market were lowest for the systems based on dairy-derived cattle and particularly the farm using no N fertiliser (D.B.F - 0-N) (Table 4).

P emissions were not available for the average European beef. When comparing the NZ beef scenarios, P emissions per kg LW were lowest for the systems from dairy-derived cattle, particularly the farm using 100 kg N/ha/year (D.B.F 100-N) (Table 4).

Table 4. Nitrogen, phosphorus and greenhouse gas emissions per kg LW equivalent at the market for all NZ beef production scenarios and per kg LW at the farm gate for average European beef (based on Leip et al. 2015). P emissions were not available for European beef. B: beef and sheep breeding & finishing farm; B.F: beef finishing farm with cattle supplied from a beef and sheep breeding farm; D.B.F: beef finishing farm with cattle supplied from a dairy farm and reared to yearling on a beef breeding farm; 0-N: no fertiliser-N input; 100-N: 100 kg fertiliser-N/ha/year.

Although recent studies on beef produced in Europe often mention that eutrophying emissions depend on soil and climate conditions, it is usually unclear how these emissions (phosphate and nitrate in particular) were calculated and if the pedoclimatic context was actually modelled (e.g. Bragaglio et al. 2017, Presumido et al. 2019).

3.2. Eutrophication impacts

3.2.1. Freshwater eutrophication - What are the benefits of considering N as contributing to freshwater eutrophication in addition to P?

Comparison of freshwater eutrophication impacts calculated with ReCiPe 2008, ReCiPe 2016 and Payen et al. (2020) (expressed in P_{eq}, focusing on the contribution of P) showed the same ranking of farm system scenarios. For example, impacts calculated with ReCiPe 2016 ranged from 0.59 g P_{eq}/kg LW (for D.B.F – 100-N) to 1.46 g P_{eq}/kg LW (for B – 0-N). Systems relying on dairy calves have a lower impact than systems based on a traditional breeding farm, and systems with 100-N inputs have a lower impact than 0-N systems (Figure 2). Impacts are lower for systems using urea fertiliser because their productivity is higher, but they are using the same amount of P fertiliser. These results indicate the constrained scenarios as the most impacting ones (per kg LW), which seems in contradiction with the freshwater policy. Impact indicators expressed per kg of product put the emphasis on system productivity and environmental efficiency. However, this is not the objective of local environmental policy focused on lake water quality, where impacts are determined by emissions per surface area unit.

Conversely, when considering freshwater eutrophication impacts due to N emissions calculated with Payen et al. (2020) (expressed in N_{eq}), the farm systems ranking was different. Systems with 100-N fertiliser inputs have a higher impact than 0-N systems, and systems relying on dairy calves have a higher impact than systems based on traditional breeding farms. Impacts ranged from 19.3 g N_{eq}/kg LW (for D.B.F – 0-N) to 44.6 g N_{eq}/kg LW (for B – 100-N). The main contributor was nitrate emission to water. Impacts were directly correlated with emissions since they occur in the same watershed and have the same characterisation factor (i.e. same attenuation).

This different ranking of farm systems depending on the nutrient considered shows that focussing on P to assess freshwater impacts can be misleading and in contradiction with the local policy in place.

For all scenarios, the contribution of post-farm stages (meat processing, packaging and transport) to freshwater eutrophication impact was very low (less than 2% on average for the four indicators applied).

To determine the impact of sourcing calves from dairy farms instead of from beef and sheep farms, we compared the contribution of various farm stages for the two scenarios having the breeding and finishing farms as separate entities (B.F and D.B.F) using 100 kg N/ha/yr. For the B.F-100-N scenario, the breeding farm was responsible for 72% of the impacts and the finishing farm for 27% on average for the four indicators applied. Conversely, when the calves were derived from a dairy farm (DBF) the relative contribution for D.B.F-100-N from the finishing farm was higher at 50% (for average of the four indicators), since impacts from the breeding farm based on dairy cattle was lower at 48% (including 1-3% contribution from the calves from the dairy farm).

Figure 2. Freshwater eutrophication impacts of 1 kg Taupō beef meat on European market calculated with ReCiPe 2008, ReCiPe 2016 and Payen et al. (2020) for NZ farm systems. B: beef and sheep breeding & finishing farm; B.F: beef finishing farm with cattle supplied from a beef and sheep breeding farm; D.B.F: beef finishing farm with weaned calves supplied from a dairy farm and reared to yearlings on a beef breeding farm; 0-N: no N input; 100-N: 100 kg N/ha/year.

3.2.2. Freshwater eutrophication - How to aggregate the contribution of N and P?

To avoid considering that a single nutrient is limiting algal growth (i.e. determining eutrophication), one possible solution is to express the impact in algae-equivalents, by aggregating N and P using the Redfield ratio (as suggested in Goedkoop 2009). The conversion factors for P and N are 114.5 kg algae/kg P and 15.8 kg algae/kg N. When aggregating freshwater eutrophication impacts from N and P calculated using Payen et al. (2020), impacts ranged from 0.85 to 1.79 kg algae/kg LW. The farm system having the highest impact was B - 100-N, while the farm system having the lowest impact was D.B.F - 0-N, which is in accordance with the freshwater policy. Such an aggregation of N and P also allows clear identification of which system is the most impacting, if the rankings vary when a single nutrient is considered (we had B – 100-N or B – 0-N ranked as the most impacting scenarios for N- and P-driven impacts respectively), and most importantly, to reconcile with local environmental policy.

Aggregation is appropriate for catchments that are predominantly co-limited all year round such as Lake Taupō (Pearson et al. 2016). However, when the co-limitation is seasonal (e.g. N-limited in summer and P-limited in winter), we reach the limit of an approach that is only spatially-explicit. A temporally-explicit impact assessment would need to be used, however, it would have several methodological constraints. First, there is a (usually unknown) time lag between an emission from land and its entry to freshwater. For example, the water in a stream entering Lake Taupō was found to be 38 years old (Vant 2013). Since past N emissions are still being released, there is a risk that N concentration will increase in the future even if mitigation measures are in place. Thus, freshwaters may become more P-limited in the future. Second, we do not always know the exact seasonality of the N or P-limitation status of catchments.

3.2.3. Marine Eutrophication – Focusing on N

Since part of the nutrients emitted in a catchment will eventually reach coastal water, it is important to account for marine eutrophication impacts as well. Impacts calculated with ReCiPe 2008 and Cosme et al. 2017 consistently showed the same ranking of NZ farm systems (Figure 3). It is important to note, that it is also the same ranking obtained for freshwater eutrophication impacts determined by N. The B – 100-N scenario has the highest impact at 0.078 and 0.025 kg N_{eq}/kg LW, calculated with ReCiPe 2008 and Cosme et al. 2017 respectively.

Figure 3. Marine eutrophication impacts of 1 kg Taupō beef meat on the European market calculated with ReCiPe 2008 and Cosme et al. 2017 for the NZ farm system scenarios. B: beef and sheep breeding & finishing farm; B.F: beef finishing farm with cattle supplied from a beef and sheep breeding farm; D.B.F: beef finishing farm with weaned calves supplied from a dairy farm and reared to yearlings on a beef breeding farm; 0-N: no N input; 100-N: 100 kg N/ha/year.

P can also be a limiting factor to marine eutrophication (Henryson et al. 2017), but in the absence of an operational method, the potential contribution of P could not be assessed. There is a need for assessment of the co-limitation status of marine coastal waters and for the development of P fate factors that are applicable globally and are spatially explicit.

3.2.4. Impacts from ammonia emissions

The impact of airborne N emissions are not characterised by Payen et al. (2020) or Cosme et al. (2017), which means that the potential contribution of ammonia to eutrophication (through re-deposition to water) is not accounted for. To address this limitation, we added CFs from ReCiPe 2008 for ammonia to air (0.92) and calculated the difference in impact results. Impacts can be increased by up to 42% for the system having the largest ammonia emissions (B.F 100-N). However, this CF of 0.92 is an average European value derived from a coupling of the CARMEN and EUTREND models (Goedkoop et al. 2009). This shows the urgent need for the development of spatially-explicit and globally-valid CFs for airborne N emissions (which is now ongoing work as part of UNEP 2019).

We also calculated eutrophication impacts with the CML indicator because it accounts for all eutrophication substances (including ammonia), although it corresponds to a worst-case scenario by considering that 100% of emissions contribute to the impact (no attenuation accounted for; and covers both terrestrial and aquatic eutrophication). Impacts calculated with CML ranged from 31.1 g PO_4^{3-}/kg LW (for D.B.F – 0-N) to 83.2 g PO_4^{3-}/kg LW (for B – 100-N), and showed the same ranking as N-driven impact categories (marine eutrophication and freshwater eutrophication-N) calculated with Payen et al. 2020.

3.3. GHG emissions and climate change

The total GHG emissions for NZ beef to the European market (from cradle-to-market-gate) ranged from the equivalent of 7.1 (for D.B.F – 0-N) to 14.1 kg $CO_{2\ eq}$/kg LW (for B – 100-N) (Figure 4; reported on a LW basis to enable subsequent comparison with other studies).

The contribution of post-farm stages (meat processing, packaging and transport to Europe) was low (ranging from 1.3 to 2.6%). The agricultural stage was the major contributor for all systems, with a contribution ranging between 94% (for D.B.F – 0-N) and 97% (for B – 100-N) of the total life cycle emissions. The main contributor was methane from enteric fermentation (ranging from 59% to 77%), followed by N_2O emissions from excreta (ranging from 11% to 16%), and N_2O emissions from N fertiliser (ranging from 0 to 13%).

Interestingly, the farm ranking was similar to the one obtained for N-driven eutrophication impacts. The 100-N farm systems have a higher impact on climate change than the 0-N systems, mainly due to N_2O emissions from fertiliser application (See Table 4 for GHG emissions). The contribution of N fertiliser manufacturing was low (about 0.4% maximum). The average beef and sheep breeding and finishing farm (B) had the highest impact on climate change mainly due to less animals sold per ha. The lower impact of the dairy-based systems (D.B.F) can be explained by the methane from enteric fermentation from the dairy cows being mainly allocated to milk production, while for the beef cattle breeding system all emissions are assigned to beef (methane from enteric fermentation at the finishing farm is the same for all systems). The lower impact of dairy-based beef is in accordance with results from previous studies (De Vries et al. 2015).

Figure 4. Impact on climate change of 1 kg of NZ beef to the European market in kg $CO_{2\ eq}$/kg LW calculated with IPCC GWP 100a. B: beef and sheep breeding & finishing farm; B.F: beef finishing farm with cattle supplied from a beef and sheep breeding farm; D.B.F: beef finishing farm with weaned calves supplied from a dairy farm and reared to yearlings on a beef breeding farm; 0-N: no N input; 100-N: 100 kg N/ha/year.

3.4. Comparison with European beef

Overall, comparison with recently published LCAs of beef produced in Europe showed that NZ beef has a lower global warming potential impact than the Italian, Portuguese and average European beef (Table 5). Regarding eutrophication impacts, results comparison is not possible due to differences in reference unit and impact assessment method. Previous studies used approaches that maximised the impacts by considering that 100% of emissions contributed to the eutrophication impact (Presumido et al. 2019), or they accounted for some attenuation but used a generic fate factor that was not specific to the characteristics of the basin (Bragaglio et al. 2018). Recent methods applied in this study modelled attenuation processes occurring in soil, rivers and lakes, thus acknowledging that only part of the emissions will reach freshwater and marine water (separately). The strength of these approaches is that attenuation is specific to the river basin where the emission occurs.

Table 5. Comparison of global warming potential and eutrophication impacts of beef produced in Europe (recently published studies) and in NZ (this study). Impacts are expressed per kilogram LW.

In comparison with the average European beef (Leip et al. 2015), it is important to notice a difference in the system boundaries (Table 5). We compared NZ beef through to the European market with European beef at the farm gate (in the absence of post-farm transportation and processing data for the European beef). As a result, from a market perspective, European beef impacts were underestimated.

Regarding eutrophication impacts, since emissions of P were not available for average European beef, we can only compare NZ and European systems using indicators focusing on the contribution of N (in N_{eq}). NZ beef had lower impacts than average European beef for marine eutrophication (calculated with Recipe 2008 and Cosme et al. 2017) and freshwater eutrophication (calculated with Payen et al. 2020). Figure 5 shows the comparison of average European beef with the NZ farm system scenario having the greatest impact (B – 100-N).

Regarding climate change, NZ beef had a lower impact than average European beef, even when accounting for the meat processing and transport from NZ to Europe. This can be explained by the minor contribution from transport of NZ beef from NZ to Europe. In addition, the European beef included housing of cattle with feed brought-in and this will have increased the GHG emissions (from manure and greater fuel use) compared to that for year-round grazing of cattle in NZ.

For wider comparison with non-European studies, the climate change impact of NZ beef from cradle-to-market-gate (7.1-14.1 kg $CO_{2\ eq}$/kg LW) was similar to estimates for beef cattle to the farm-gate of 10.6-12.4 kg $CO_{2\ eq}$/kg LW for Australian beef (Wiedemann et al. 2016) and 7-13 kg $CO_{2\ eq}$/kg LW for USA beef (Rotz et al. 2015).

Figure 5. Comparison of climate change impact (calculated with IPCC 2007 GWP 100a, in kg $CO_{2\ eq}$/kg LW), marine and freshwater eutrophication impacts (calculated with ReCiPe 2008, Cosme et al. 2017 and Payen et al. 201920, in kg N_{eq}/kg LW) of 1 kg average European beef (based on Leip et al. 2015) and 1 kg NZ beef from the farm system scenario having the largest impact (B – 100-N).

Future studies addressing eutrophication impacts should (i) make more transparent their estimate of N and P emissions in the inventory, (ii) account for both N and P nutrients (at least at the inventory stage), (iii) model post-farm stages and (iv) include sensitivity analyses. One limitation of this study is that no sensitivity analyses or statistical analyses were performed.

3.5. Can the consumer willingness to pay offset the cost to farmers for the reduction of N emissions?

NZ beef potentially has a positive environmental credence attribute on the European market as demonstrated by the lower environmental impacts.

3.5.1. Consumer willingness to pay (WTP)

Results of a meta-regression analysis indicate that European consumers are willing to pay a 32% price premium on average, ranging from 18% to 103%, for beef products with positive environmental attributes (Figure S1). Notably, compared to consumers from North America, the European consumers are willing to pay 9% more for beef products with low environmental impacts. However, the WTP for environmental attributes is relatively lower than WTP for some other credence attributes, such as animal welfare and organic production (more details in the Supplementary Material).

3.5.2. Impact of WTP on farm level profitability

We first analysed profitability per kg LW for each farm within a scenario, as shown in Figure 6. In the absence of a price-premium, the profitability of the finishing farm (F in the D.B.F and B.F scenarios) showed a similar trend to the breeding and finishing farm (B). Here, when changing from a non-constrained (100-N) to a constrained (0-N) scenario, the profit per kg live weight decreased from \$0.6 to \$0.4 for the B farm and from \$0.31 to \$0.26 for the finishing farm (in NZ dollars; NZ\$1 ≈ 0.58 euros). This indicates that environmental regulation reduced farm profitability for both finishing and B farms. When a price premium was added, the finishing and B farms increased their profit by 73% and 16% respectively compared to the regulated farm. Therefore, when beef is sold at a price premium, the consumer WTP can offset the cost to farmers for mitigating the N emissions, but this would apply only to the farm selling finishing cattle. This can be explained by two reasons. First, we considered that only the finishing farm would get the price premium although in practice there may be some flow-on to the breeding farm as well. Second, in the modelling of the B.F and D.B.F systems in FARMAX, the breeding farm supplied yearling cattle to the finishing farm, but the temporal pattern of feed demand on that farm meant that it was unable to effectively use the extra feed produced from adding 100 kg fertiliser-N/ha/year. Thus, the cattle from the breeding farm for the BF 100-N scenario had a lower profit than those for the 0-N scenario. In practice, this farm would be unlikely to use N fertiliser where the outcome was reduced profitability.

Overall, when we analysed the profitability of the farm systems as a whole (i.e. across all farms involved in producing and rearing cattle), the D.B.F system had the highest profit within each scenario ($0.87 for 100-N, $0.89 for 0-N, and $1.09 for 0-N +premium). For the D.B.F and B.F systems, the overall profit was lower in a non-constrained (100-N) scenario, while the highest profit was achieved in the constrained (0-N) scenario with a price premium. The 100-N scenario had the lower profit for the reasons mentioned above, representing a non-efficient use of the fertilisers in the breeding farms modelled. Conversely, for the B system, the highest profit was achieved in the non-constrained (100-N) scenario and the profit reduced in the constrained (0-N) scenario with no price premium. When a price premium was added, the profit for the B system increased but was not as high as that for the non-constrained scenario.

Figure 6. Comparison of profitability in NZ$ per kg net live weight gain per farm (breeding and finishing farms separated) across nine economic scenarios. B: beef and sheep breeding & finishing farm; B.F: beef finishing farm with cattle supplied from a beef and sheep breeding farm; D.B.F: beef finishing farm with weaned calves supplied from a dairy farm and reared to yearlings on a beef breeding farm; 0-N: no N input; 100-N: 100 kg N/ha/year.

It is interesting to note that the scenario achieving the greatest profit (D.B.F 0-N with premium) is also the one presenting the lowest environmental impacts (for marine eutrophication, freshwater eutrophication N equivalent and in algae-equivalent, and climate change impact).

One limitation is that the meta-analysis did not distinguish which environmental attribute the consumer values the most (their WTP may vary strongly across different environmental issues). In this study, we assumed that water quality and climate change were of concern for European consumers.

4. Conclusions

This paper showed that we could start bridging the gap between local freshwater quality policy and LCA eutrophication indicators by accounting for both N and P nutrients. LCA freshwater eutrophication indicators need to account for the contribution of N in addition to P, and align to freshwater policy needs.

Nitrogen emission limitations in the Lake Taupō catchment in NZ led to lower eutrophication impacts. This conclusion would have been different if only P had been considered in the freshwater eutrophication impact indicator. Calculating impacts with common P-driven freshwater indicators would have pointed the non-constrained scenario as the least impacting, which seems in contradiction with the local freshwater policy. This shows the importance of accounting for both nutrients when assessing freshwater eutrophication impacts, as recommended by the GLAM program of the UNEP/SETAC Life Cycle Initiative.

The freshwater policy across NZ is currently focusing on managing N emissions across most catchments of concern for water quality but it should also actively monitor and manage P emissions. Indeed, because there is a time lag between the application of N on soil (from fertiliser, urine and manure) and its emission to freshwaters, there is a risk that N concentration in freshwater will increase in the future even if mitigation measures are in place (because time-lags mean that some past leached-N is still to enter the lake). As a result, freshwater may become more P-limited in the future.

NZ beef produced in the Lake Taupō catchment and supplied to the European market showed a potential lower impact on climate change, freshwater and marine eutrophication than that for average European beef at the farm gate, although more datasets are required to confirm this. The economic analysis revealed that this lower “environmental footprint” could potentially be used to sell NZ beef with a price premium on the European market. For certain farm systems, this price premium would potentially offset the cost to farmers for farm practices required to achieve the reduction of N emissions. Indeed, although ceasing N fertiliser inputs under an environmental regulation scenario lowers farm profitability, the constrained scenario could actually outperform the non-constrained scenario when a premium is considered. This shows that the consumer willingness to pay could compensate for the environmental cost of protecting the lake that currently only the farmers are bearing.

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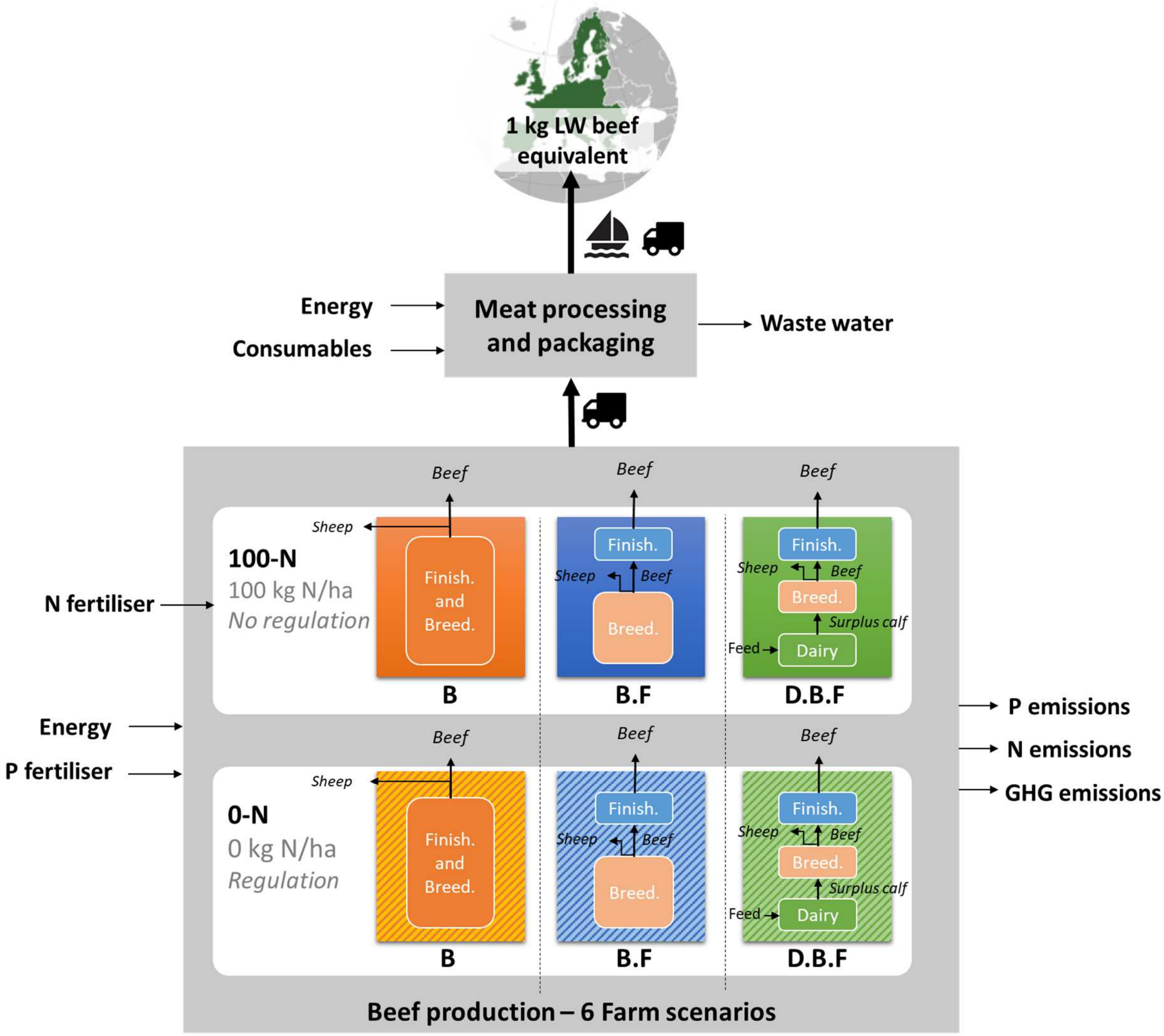


Figure 1. Flow diagram for the NZ Beef production, meat processing and transport to the European market showing the main inputs and outputs modelled. Six beef production scenarios were modelled based on farm systems and nitrogen (N) fertiliser input. **B**: Average beef and sheep breeding & finishing farm; **B.F**: Beef finishing farm with cattle supplied from an average beef and sheep breeding farm; **D.B.F**: Beef finishing farm with weaned calves supplied from a dairy farm and reared to yearlings on a beef breeding farm. For each farm system, two N fertiliser input scenarios were analysed: 0-N (0 kg N/ha) and 100-N (100 kg N/ha).

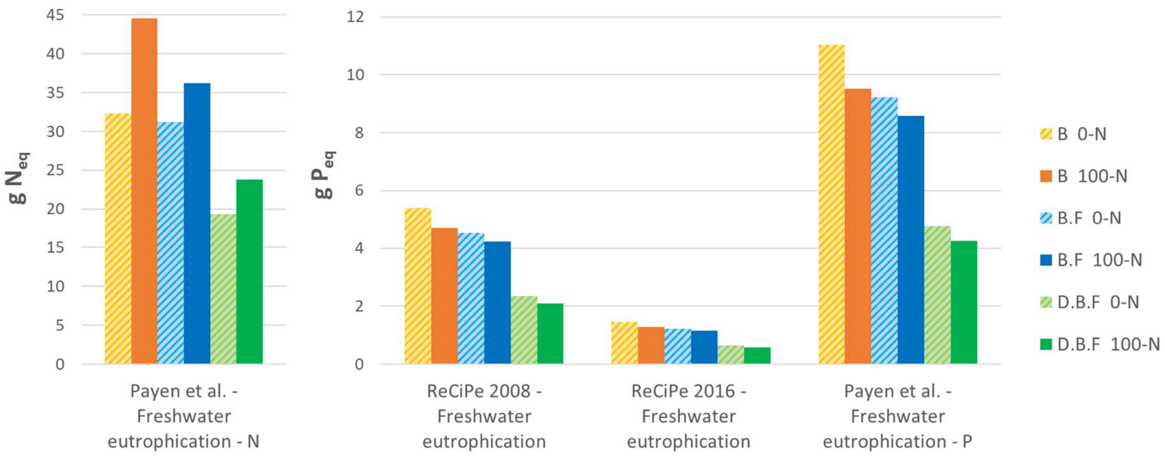


Figure 2. Freshwater eutrophication impacts of 1 kg Taupō beef meat on European market calculated with ReCiPe 2008, ReCiPe 2016 and Payen et al. (2020) for NZ farm systems. B: beef and sheep breeding & finishing farm; B.F: beef finishing farm with cattle supplied from a beef and sheep breeding farm; D.B.F: beef finishing farm with weaned calves supplied from a dairy farm and reared to yearlings on a beef breeding farm; 0-N: no N input; 100-N: 100 kg N/ha/year.

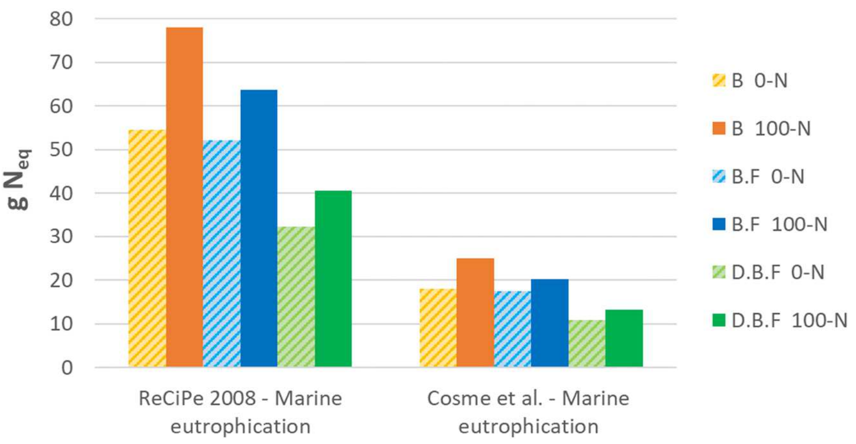


Figure 3. Marine eutrophication impacts of 1 kg Taupō beef meat on the European market calculated with ReCiPe 2008 and Cosme et al. 2017 for the NZ farm system scenarios. B: beef and sheep breeding & finishing farm; B.F: beef finishing farm with cattle supplied from a beef and sheep breeding farm; D.B.F: beef finishing farm with weaned calves supplied from a dairy farm and reared to yearlings on a beef breeding farm; 0-N: no N input; 100-N: 100 kg N/ha/year.

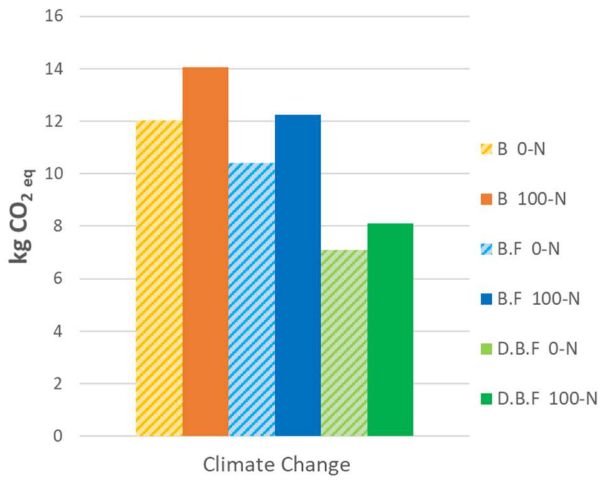


Figure 4. Impact on climate change of 1 kg of NZ beef to the European market in kg $CO_{2\ eq}$/kg LW calculated with IPCC GWP 100a. B: beef and sheep breeding & finishing farm; B.F: beef finishing farm with cattle supplied from a beef and sheep breeding farm; D.B.F: beef finishing farm with weaned calves supplied from a dairy farm and reared to yearlings on a beef breeding farm; 0-N: no N input; 100-N: 100 kg N/ha/year.

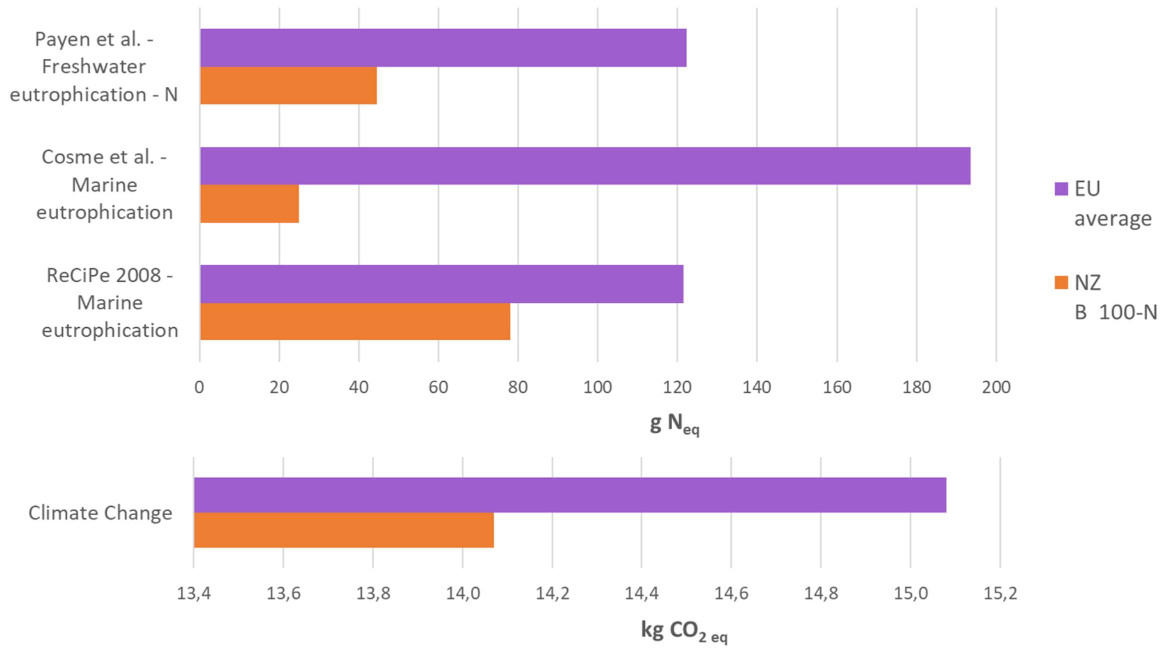


Figure 5. Comparison of climate change impact (calculated with IPCC 2007 GWP 100a, in kg $CO_{2\ eq}$/kg LW), marine and freshwater eutrophication impacts (calculated with ReCiPe 2008, Cosme et al. 2017 and Payen et al. 201920, in kg N_{eq}/kg LW) of 1 kg average European beef (based on Leip et al. 2015) and 1 kg NZ beef from the farm system scenario having the largest impact (B – 100-N).

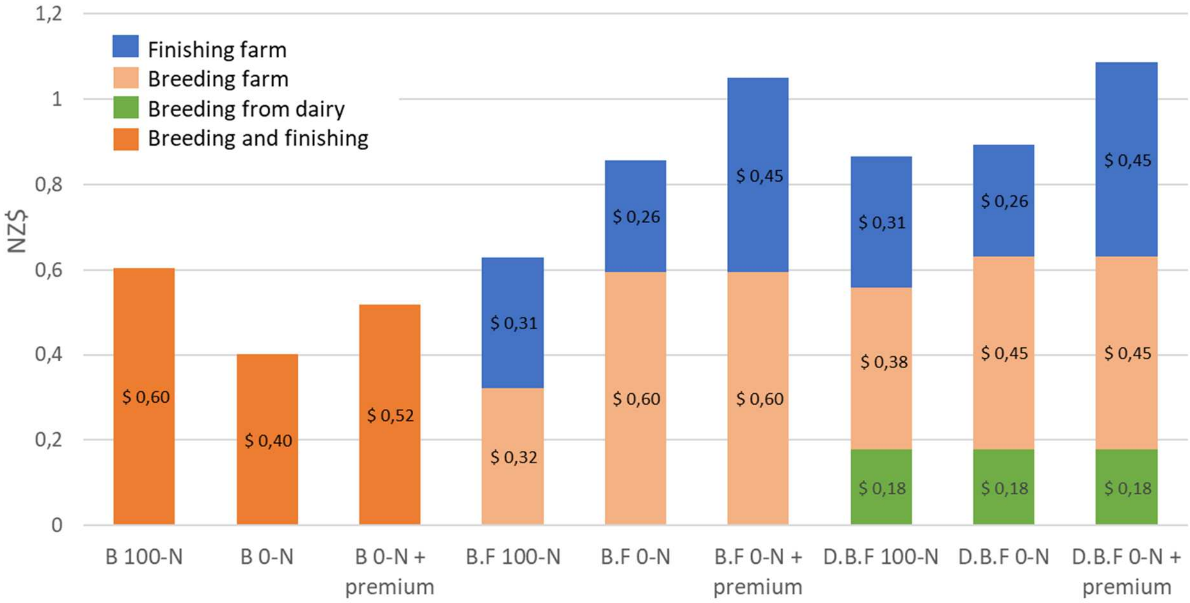


Figure 6. Comparison of profitability in NZ$ per kg net live weight gain per farm (breeding and finishing farms separated) across nine economic scenarios. B: beef and sheep breeding & finishing farm; B.F: beef finishing farm with cattle supplied from a beef and sheep breeding farm; D.B.F: beef finishing farm with weaned calves supplied from a dairy farm and reared to yearlings on a beef breeding farm; 0-N: no N input; 100-N: 100 kg N/ha/year.

Table 2. Inventory table for each farm included in the different scenarios

	Breeding & Finishing farm	**Breeding & Finishing farm**	**Finishing farm**	**Finishing farm**	**Breeding farm (beef cattle)**	**Breeding farm (beef cattle)**	**Breeding farm (dairy cattle)**	**Breeding farm (dairy cattle)**
	B	**B**	**B.F**	**B.F**	**B.F**	**B.F**	**D.B.F**	**D.B.F**
	0-N	100-N	0-N	100-N	0-N	100-N	0-N	100-N
	Finish. and Breed.	Finish. and Breed.	Finish.	Finish.	↑ Breed.	↑ Breed.	↑ Breed. ↑ Dairy	↑ Breed. ↑ Dairy
Area [ha]	324	324	120	120	426#	455#	123##	123##
Total cattle LW purchased [kg/ha/yr]	4.3*	4.3*	709	797	0	0	98 (from dairy farm)	110 (from dairy farm)
Net cattle LW sold [kg/ha/yr]	211	249	597	670	200 (LW sold for purchase by finishing farm)	210 (LW sold for purchase by finishing farm)	692 (LW sold for purchase by finishing farm)	778 (LW sold for purchase by finishing farm)
Urea fertiliser [kg N/ha/yr]	0	100	0	100	0	100	0	100
Allocation factor to cattle vs. cattle +sheep (based on DM intake)	51%	51%	n.a.	n.a.	55%	55%	100%	100%

* breeding bulls only; # Breeding farm area required to supply cattle to finishing farm; ## Breeding farm area based on using an average area solely for rearing weaned dairy calves to sell to the finishing farm.

Table 3. Freshwater and marine eutrophication characterisation factors used in this study for nitrate and phosphate emissions to river, for the Waikato region (New Zealand), Europe and a Global average.

Method	**Emission route**	**Unit**	**Waikato (NZ)**	**Europe**	**Global**
Freshwater eutrophication					
ReCiPe 2008	Phosphate to river	kg P_{eq}/kg	0.330	NA	0.330
ReCiPe 2016 (Helmes et al. 2012)	Phosphate to river	kg P_{eq}/kg	0.087*	NA	0.326
Payen et al. (2020)	Phosphate to river	kg P_{eq}/kg	0.676	NA	0.326
Payen et al. (2020)	Nitrate to river	kg N_{eq}/kg	0.151	0.242	0.226
Marine eutrophication					
ReCiPe 2008	**Nitrate to river**	**kg N_{eq}/kg**	**0.230**	**0.230**	**0.230**
Cosme et al. 2017	Nitrate to river	kg N_{eq}/kg	0.084	0.383	0.226

NA=Not applied; *Using Helmes et al. (2012) fate factors divided by ReCiPe 2016 global average fate factor.

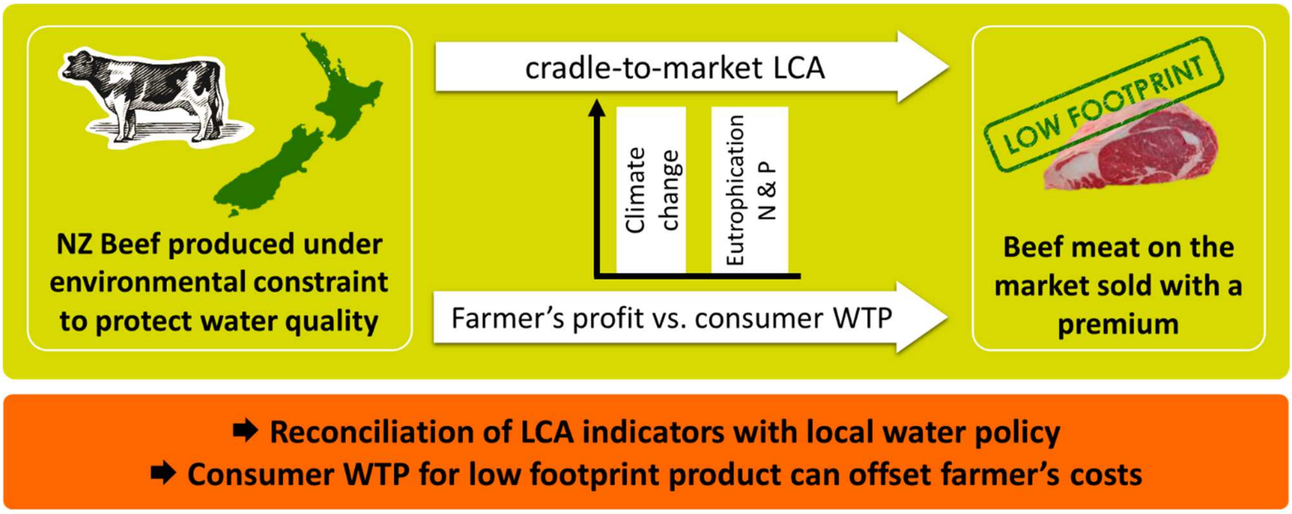
Table 4. Nitrogen, phosphorus and greenhouse gas emissions per kg LW equivalent at the market for all NZ beef production scenarios and per kg LW at the farm gate for average European beef (based on Leip et al. 2015). P emissions were not available for European beef. B: beef and sheep breeding & finishing farm; B.F: beef finishing farm with cattle supplied from a beef and sheep breeding farm; D.B.F: beef finishing farm with cattle supplied from a dairy farm and reared to yearling on a beef breeding farm; 0-N: no fertiliser-N input; 100-N: 100 kg fertiliser-N/ha/year.

Substance & Unit	B 0-N	B 100-N	B.F 0-N	B.F 100-N	D.B.F 0-N	D.B.F 100-N	Average EU Beef
g N-NOx	1.24	1.54	1.23	1.55	1.08	1.24	2.73
g $N\text{-}NH_3$	42.62	81.93	37.31	69.89	23.00	34.48	43.44
g $N\text{-}NO_3$	45.76	63.16	44.21	51.29	27.35	33.64	114.26
g $P\text{-}PO_4^{3-}$	5.05	4.38	4.23	3.95	2.19	1.97	n.a.
kg CO_2	0.85	1.67	0.65	1.59	1.28	1.80	5.57
g N_2O	5.86	9.11	5.41	8.08	3.05	4.81	11.09
kg CH_4	0.38	0.39	0.33	0.33	0.20	0.20	0.23

Table 5. Comparison of global warming potential and eutrophication impacts of beef produced in Europe (recently published studies) and in NZ (this study). Impacts are expressed per kilogram LW.

Reference	Country	System boundary	Global warming potential (kg $CO_{2\ eq}$)	Freshwater eutrophication	Marine eutrophication
Bragaglio et al. 2018	Italy	cradle-to-farm	17.6 - 26.3	779 - 1009 g $NO_{3\ eq}$	
Buratti et al. 2019	Italy	cradle-to-farm	18.2 - 24.6	n.a.	n.a.
Presumido et al. 2018	Portugal	cradle-to-slaughterhouse	16.4 - 22.3	123-154 g $PO_{4\ eq}$	
Leip et al. 2015	Europe (average)	cradle-to-farm	15.1	122 g N_{eq}* (Payen et al.)	193 g N_{eq}* (Cosme et al. 2017)
This study	NZ	cradle-to-market	7.1 - 14.1	19.3 - 44.6 g N_{eq} (Payen et al.) 4.2 - 11.0 g P_{eq} (Payen et al.)	13.6 - 34.8 g N_{eq} (Cosme et al. 2017)

* Estimated in this study (see section 2.4)


NZ Beef produced under environmental constraint to protect water quality
cradle-to-market LCA
Climate change
Eutrophication N & P
Farmer's profit vs. consumer WTP
LOW FOOTPRINT
Beef meat on the market sold with a premium
➨ Reconciliation of LCA indicators with local water policy
➨ Consumer WTP for low footprint product can offset farmer's costs